STRENGTHENING AREA-BASED CONSERVATION TO SUPPORT BIODIVERSITY AND PEOPLE'S WELLBEING: A PERSPECTIVE FROM TROPICAL REGIONS

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Table of Contents

List of Tables

List of Figures

List of Boxes

Resumen

Las naciones de las regiones tropicales tienen la responsabilidad de preservar los lugares más biodiversos de la Tierra y de gestionarlos para garantizar la provisión de las “Contribuciones de la Naturaleza a la calidad de vida de las Personas” (NCP). Las medidas de conservación basadas en áreas, como las áreas protegidas (AP), son instrumentos que estas naciones han implementado para cumplir con dicha responsabilidad. Sin embargo, fallas persistentes en la gestión y ubicación de las AP han disminuido su contribución a la biodiversidad y el bienestar humano. En esta tesis, exploro los retos y las alternativas que tienen las regiones tropicales para fortalecer tres aspectos clave de la conservación basada en áreas: la sostenibilidad financiera, la representatividad ecológica y la gestión equitativa.

En el **Capítulo 1**, evalúo si una financiación adecuada para el manejo de AP conduce a una protección más efectiva de los bosques. Los resultados indican que déficits en la financiación han reducido el impacto de las AP de Ecuador en evitar la deforestación, mientras que niveles más bajos de desarrollo humano en los países de Latinoamérica se asocian a sistemas AP con impactos más bajos. En el **Capítulo 2**, investigo cómo expandir las AP en la Amazonia occidental para aumentar la representación de la biodiversidad al menor costo posible de manejo y de oportunidad por actividades agrícolas. Para alcanzar este objetivo, encuentro que las priorizaciones deben incorporar información espacial sobre la variación de costes de conservación de la tierra, considerar a los territorios indígenas e implementar colaboración internacional. En el **Capítulo 3**, identifico cuáles son las brechas más críticas en la representación ecológica y de NCP en las áreas de conservación de los Andes. Con base en una revisión de artículos y ejercicios de priorización, determino que la planificación de la región necesita tres acciones transformadoras para cerras tales brechas:

(1) integrar a los NCP en análisis de priorizaciones para la conservación de la biodiversidad, (2) diversificar la gobernanza y objetivos de manejo de las AP, y (3) fortalecer la colaboración entre los países andinos y el financiamiento privado e internacional. En el **Capítulo 4**, examino los riesgos a los que se enfrenta la diversidad cultural humana debido a la deforestación. Mediante la revisión de estudios de caso, detecto tres vías por las que la pérdida de bosques transforma los sistemas culturales de poblaciones locales. Simultáneamente, análisis espaciales muestran que la deforestación de este siglo se ha expandido rápidamente hacia los territorios de ~1.400 grupos etnolingüísticos, lo que puede suponer una amenaza para al menos el 20% de la diversidad mundial. Aunque investigaciones demuestran que las AP son un instrumento eficaz para frenar deforestación, la falta de equidad en su gestión también es una amenaza a las culturas locales.

Esta tesis demuestra que una protección efectiva de la biodiversidad tropical requiere una expansión sustancial de áreas de conservación en los lugares adecuados y un incremento significativo de recursos para su gestión. Además, los resultados sugieren que el establecimiento y manejo de áreas de conservación exitosas y el bienestar de las comunidades locales están fuertemente conectados, lo que exige una mejor comprensión de estos vínculos al planificar la expansión y gestión de áreas de conservación. Con base en estos hallazgos, propongo cinco áreas de acción para mejorar la sostenibilidad financiera, la representación ecológica y la equidad social de la conservación basada en áreas, y que, en conjunto, buscan armonizar la protección de la biodiversidad con las necesidades de los actores locales. Estas y otras perspectivas de las regiones tropicales sobre cómo fortalecer la conservación basada en áreas también son fundamentales para informar los acuerdos globales sobre protección de la biodiversidad.

Abstract

Nations in tropical regions are responsible for preserving the most biodiverse places on Earth, managing them to ensure the delivery of Nature's Contribution to People's quality of life (NCPs). Area-based conservation measures, such as protected areas (PAs), are among the main instruments these nations have implemented to fulfil this responsibility. However, persisting shortcomings in the management and location diminish the contribution of area-based conservation to sustaining biodiversity and people's wellbeing. This thesis explores the challenges and alternatives for tropical regions to strengthen three key aspects of area-based conservation: financial sustainability, ecological representativeness, and equitable management.

In **Chapter 1**, I assess the hypothesis that adequate funding for management contributes to more effective forest protection in PAs of Ecuador and PA systems of Latin American countries. Results show that funding deficits reduce the PA' impact in avoiding deforestation, while countries' human development dimensions are the most relevant drivers of impact at the PA system level. **Chapter 2** explores ways by which an expansion of PAs for the western Amazon can increase biodiversity coverage at the least possible management and agriculture opportunity costs. I find that prioritizations that incorporate spatial data on conservation costs, involve indigenous lands, and assume international collaboration allow maximizing species representation at more affordable budgets. In **Chapter 3**, I identify critical gaps in ecological representation and NCPs in the current PA system of the Andes. Based on a literature review and prioritization exercises, I find that conservation planning in the region needs three transformative actions to close such gaps: (1) optimizing the coverage of both biodiversity features and NCPs when planning for the

expansion of PAs, (2) diversifying administration regimes and management objectives of PAs, and (3) increasing collaboration among Andean countries, engaging with private and international financial support. Finally, **Chapter 4** examines the risks that human cultural diversity faces due to deforestation in tropical regions. By reviewing case studies, I detect three pathways by which forest loss can transform forest people cultures. Also, spatial analyses show that this century's deforestation has rapidly expanded into the territories of ~ 1,400 ethnolinguistic groups, posing a threat to at least 20% of the world's linguistic diversity. Although research suggests that PAs are an effective instrument to curb deforestation, the lack of social equity in their management also poses a threat to local cultures.

This thesis shows that effective protection of tropical biodiversity requires a substantial expansion of areas-based conservation targeted at the right places, and a better allocation of funds for its management. Results also suggest that the establishment and management of successful area-based conservation and the wellbeing of local communities are strongly connected, which calls for a better understanding and consideration of these links when planning the expansion of conservation areas. Considering this requirement, I propose five areas of action to enhance the financial sustainability, ecological representation, and social equity of area-based conservation efforts, which together also seek to harmonize biodiversity protection with the concerns and aspirations of local actors. These and other perspectives from tropical regions on how to strengthen area-based conservation are also critical to inform global agreements on biodiversity protection.

Introduction

Global agendas for biodiversity conservation and sustainability will be moving forward in this decade. The United Nations declared the 2020s “a decade of action” to meet the Sustainable Development Goals (UN 2020). The Convention on Biological Diversity (CBD) has also called for “transformative” actions to achieve the vision of world nations for 2050 of ‘Living in Harmony with Nature’ (CBD 2018). As a part of this vision, countries are negotiating an ambitious expansion of area-based conservation coverage to protect at least 30% of the planet’s land and oceans before 2030 (~15.3% and 7.5% of the terrestrial and marine realm are currently protected, respectively; CBD 2020). Area-based conservation includes protected areas (PAs), geographically defined areas designated and managed to achieve the long-term conservation of nature (CBD 2011). Objectives of PAs also incorporate the promotion of sustainable use of natural resources by local communities, an adequate provision of Nature’s Contributions to People’s good quality of life (NCPs), climate change mitigation and adaptation, and the protection of cultural values associated with biodiversity (Dudley 2008; Stolton & Dudley 2010; Hannah et al. 2020). Thus, PAs serve as a fundamental tool to maintain diverse values of nature and the benefits it provides people, contributing to the CBD’s aim of building a sustainable relationship between humanity and nature.

Countries from tropical regions are crucial partners in achieving the aspirations of the CBD’s agreement, as global priority sites for biodiversity protection concentrate in the tropics (e.g., Wilson 2016; Hannah et al. 2020; Dinerstein et al. 2020). Many of these priority areas are also shared with thousands of indigenous people and local communities (IPLCs), who directly rely on nature for their livelihoods, commercial activities, and identities (Devenish & Gianella 2012; Garnett et al. 2018). Thus, perspectives from tropical regions on how to set goals for area-based conservation are especially relevant for building an actionable global agreement with high biological and social impact. This agreement will also require actions to address persisting problems that undermine the long-term success of tropical PAs in curbing environmental degradation, preserving biodiversity, and contributing to people's wellbeing (Laurance et al. 2012; Pringle 2017). Among several deficiencies, PAs in tropical regions urgently need to enhance their financial sustainability, balance their ecological representativeness, and strengthen social equity in management, which are often hindered by ecological characteristics and socioeconomic constraints shared by tropical regions.

A sound area-based conservation system should be **financially sustainable**, which means that countries and relevant institutions are able to cover all costs associated with effective management that ensure the protection of nature (Bovarnick et al. 2010). In tropical regions, most PAs are dependent on public funding (Bovarnick et al. 2010; Aseres & Sira 2020). However, many tropical countries are low income, which combined with competing social needs, often prevents the allocation of adequate resources for PAs (Bradshaw et al. 2009; Büscher et al. 2017). Consequently, PAs in the tropics are usually underfunded, compromising their ability to manage wildlife, restore degraded landscapes, enforce conservation and control of threats within their borders, among other actions (Bovarnick et al. 2010; Laurance et al. 2012; Coad et al. 2019). Lack of funding also hampers compensation to local people for the opportunity costs resulting from foregone incomes when land is declared protected (Aseres & Sira 2020). Underfunding is also the result of the misconception that just declaring new PAs is enough to produce positive impacts on conservation, regardless of funds for enforcing management. To fight this assumption, empirical evidence on the links among funding, human pressures, and conservation outcomes in PAs are urgently needed for tropical regions (Flores & Bovarnick 2016).

Ecological representativeness is the degree to which an area-based network covers the full variety of biodiversity facets (e.g., target taxa, habitat types) in a way that contributes to their long-term persistence (Kukkala & Moilanen 2013). Achieving an ecologically representative conservation network is particularly challenging in biologically rich regions such as the tropics, as it has been shown this would demand a large extent of conservation areas and resources compared to higher latitudes (Rodrigues & Gaston 2001; Jenkins et al. 2013). This goal has been further complicated because of flawed historical decisions on where to locate PAs, which created a bias towards protecting places with low economic value that contribute little to represent important groups, such as threatened vertebrate species (Rodrigues et al. 2004; Venter et al. 2018). Consequently, a large part of the tropical biodiversity remains unprotected and exposed to anthropogenic threats (Butchart et al. 2015; Maxwell et al. 2020; Hannah et al. 2020). Also, there is limited knowledge of the extent to which tropical PAs cover important areas for retaining and providing NCPs, such as freshwater services, non-timber forest products, or sacred places (Neugarten et al. 2020). Thus, science needs to provide clear guidance to tropical regions on where to allocate the limited resources for expanding conservation areas in order to maximize the representation of biodiversity and demanded NCPs by local and global human populations.

Equity is closely related to social justice aspects concerning all involved stakeholders when conservation areas are established (Franks et al. 2018). Achieving equitable management in PAs is critical in tropical regions since they are home to a large rural population and most of the indigenous peoples of the world, whose livelihoods are sensitive to the land-use regulations that PAs imposes (Chazdon et al. 2009; Oldekop et al. 2016; Garnett et al. 2018). Although numerous tropical PAs are known to deliver benefits to neighboring IPLCs and rural populations, such as poverty reduction (Andam et al. 2010; Ferraro & Hanauer 2014; Naidoo et al. 2019), there are also cases of PAs with deficiencies in equitable management. Specifically, reports have shown deficiencies in terms of effective participation in decision-making, access to justice in conflicting situations, respect of identity and cultural differences, and recognition of customary and ancestral rights to land and natural resources (Martin et al. 2016; Zafra-Calvo et al. 2019). These social inequities diminish the quality of life of local communities and often undermine the PA impact on biodiversity conservation (Oldekop et al. 2016). Thus, a successful expansion of equitable PAs in the tropics requires a sound understanding and sensitive considerations of the long-term interactions between local people and nature and how the establishment of PAs can alter these interactions and affect people’s wellbeing, cultural continuity, and natural resource use (Agnoletti & Rotherham 2015; Linnell et al. 2015).

New agreements in the post-2020 global biodiversity framework are expected to create a new momentum in the expansion of area-based conservation. Therefore, it is timely for conservation science to inform decision makers about actions that might help enhance PA financing, ecological representativeness, and social equity. Many recent global and regional studies have provided helpful guidance on where and how much land should be set aside for protection across the tropics to enhance biodiversity representation (e.g., Wilson 2016; Pimm et al. 2018; Hannah et al. 2020). However, these studies seldom explore the consequences that might arise from the proposed conservation-area expansions, such as its financial feasibility, impacts on local and national economies, as well as on rural and indigenous populations inhabiting areas of high biological value (Mehrabi et al. 2018; Ellis & Mehrabi 2019; Schleicher et al. 2019). Thus, these studies might offer limited guidance to decision-makers on how to navigate the complex decisions and actions needed to strengthen area-based conservation for people and nature (Knight et al. 2006; Kuempel et al. 2020). Contrastingly, a more effective, fair, and actionable global strategy for biodiversity conservation needs to be informed by research that addresses how to overcome the persisting financial, ecological, and social shortcomings of area-based conservation in a more integrated and comprehensive way.

This thesis explores the challenges and alternatives of tropical regions for building more financially sustainable, ecologically representative, and socially equitable area-based conservation systems. The thesis consists of four independent research chapters focusing on specific study areas and dealing with the knowledge gaps in area-based conservation presented in this introduction. The objective of each Chapter 1s:

- **Chapter 1.** To assess whether funding for management contributed towards effective forest protection in Ecuadorian PAs and Latin American PA systems. Here, I use counterfactual analysis to isolate PAs' impact in curbing deforestation. Then, I explore the socioeconomic circumstances in which proper funding is more critical for PAs to deliver positive outcomes. Based on these results, I discuss actions to enhance the financial sustainability of PAs and maximize their conservation impact.

- **Chapter 2**: To design a potential expansion for the PA system in the Western Amazon that increases species representation at minimum management and opportunity costs. As a first step, I explore the main drivers of the costs associated with PAs in the region. This information is used in spatial prioritization exercises to identify additional sites that would protect species at more affordable management costs and minimize conflicts with agricultural production.

- **Chapter 3**: To identify spatial conservation needs in the Andes for biodiversity protection and people's wellbeing. In this research, I provide a comprehensive review of critical gaps on ecological representation and NCPs in the current PA system of the Andes. I also examine location and management alternatives for PAs to close these gaps and make area-based conservation more equitable and financially viable.

- **Chapter 4:** To examine the risks faced by human cultural diversity due to this century's unprecedented deforestation in the global tropics. The chapter builds on a literature review and spatial analysis to delve into the relationships between nature and human cultural evolution in forest landscapes. This analysis also serves as a basis for understanding the positive and negative impacts that PA governance and management might have on the cultural continuity and wellbeing of forest-dependent people.

Chapter 1 - Does money matter? The role of funding in the performance of Latin American protected areas

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Abstract

Conservationists have long argued that inadequate funding for managing protected areas (PAs) jeopardizes their ability to achieve conservation goals. However, this claim has rarely been substantiated by quantitative evaluations. Here, we examined whether funding contributed towards more effective forest protection in 27 individual PAs in Ecuador and 17 PA systems of Latin American countries. We found that although most of the PAs reduced deforestation between 2000 and 2010, these conservation impacts were highly variable. Within the PA system of Ecuador, lower PA impacts were associated with larger funding deficits, especially for PAs facing major human pressures on forests. At the system level, human development scores of countries partially explained the variation on impact. We, therefore, emphasize that maximizing the conservation impact of Latin American PAs

needs a multi-level approach that includes better resource allocation for PAs, combined with strategies for strengthening institutions and governance of PA systems.

Keywords: management, budget, deforestation, effectiveness.

1. Introduction

Protected areas (PAs) are one of the chief instruments for conserving biodiversity (CBD 2010) and have been instrumental in slowing the loss of forests in the face of increasing human pressures (e.g., Joppa Lucas & Pfaff 2011; Geldmann et al. 2013; Pfaff et al. 2015). However, while PAs are losing less forests than non-protected forests, PAs are not immune to forest loss, with an estimated 21.9 million hectares of forests cleared between 2000 and 2012 globally within PAs (Heino et al. 2015) - an area similar to the size of Guyana or Great Britain. Moreover, examining the average effect across national or regional PA networks hide considerable variation in effectiveness of individual PAs within the same study area (Nolte et al. 2013; Eklund et al. 2016; Schleicher et al. 2017). Thus, understanding what factors contribute to delivering successful PA outcomes is critical for realizing the full potential of PAs (Geldmann et al. 2018, 2019).

Most PAs around the world only receive a fraction of the required resources for their management. The annual cost of managing the global networks of terrestrial and marine PA is estimated to be US$ 68 billion, but spending is closer to US$ 24 million per year (Waldron et al. 2020). It is thus argued that underfunding PAs jeopardizes their impact in achieving conservation goals (Watson et al. 2014; Coad et al. 2019). However, the quantity of empirical evidence supporting this statement is limited and inconclusive (IPBES 2018). In fact, few studies have investigated the connections between funding and PA impact (Gill et al. 2017; Geldmann et al. 2018, 2019; Schleicher et al. 2019b), probably in part, because reliable and high-quality data on financial spending and needs in PAs is incredibly rare (Lindsey et al. 2018). Moreover, some studies have suggested that governance types, law enforcement, and corruption are more important than management and financial factors in influencing conservation outcomes (Nolte et al. 2013; Amano et al. 2018; Schleicher et al. 2019b). Thus, understanding what levels of funding are needed to achieve conservation outcomes under different contexts remains one of the most important knowledge-gap related to PA effectiveness.

Latin American countries are at the forefront of global forest conservation. Over the past two decades, the region has built an impressive PA system that covers the largest extent of protected forests in the world (Heino et al. 2015; FAO 2016). The forests of Latin America are home to an unparalleled biological and cultural diversity, provide essential ecosystem services that contribute to the economies of the countries and are essential for global carbon and water cycling (FAO 2016; Potapov et al. 2017; Lovejoy & Nobre 2018). Thus, ensuring effective management of Latin American protected forests is a task of global importance. In response to this need, in 2010, a joint effort of national governments and international NGOs estimated the financial requirements for managing the national PA systems (Bovarnick et al. 2010). The report found that Latin American PA systems had only 55% of their basic budget covered (Bovarnick et al. 2010). In parallel, these PAs suffered significant forest degradation of about 1,097,618 hectares between 2004 and 2009 — an area the size of Jamaica (Leisher et al. 2013). Such chronic underfunding could negatively affect the impact of Latin American PAs in preserving forests. However, the region lacks indicators on the positive impact of funding, which hinders the possibilities to persuade government offices to increase PAs' budget and attract additional international support for PAs (Flores & Bovarnick 2016).

Here, we use two unique datasets on PA funding to assess the degree to which financial resources contributed to the impact of Latin American PAs in avoiding deforestation for the period of 2000 to 2010. We examine the role of funding at two levels at which precise data on spending for PA management was available for this period (**Fig. 1.1**): (1) for individual PAs within a country; analyzing 27 PAs of Ecuador as a case study (Galindo et al. 2005), and (2) for national PA systems of 17 out of 24 Latin American countries (Bovarnick et al. 2010). To estimate the PA impact, we use a counterfactual approach (i.e., statistical matching) that compares deforestation rates of PAs with the unprotected landscape that has similar contextual attributes as the PAs (Schleicher et al. 2019a). We then test the influence of different potential drivers of impact, including funding deficits. We also explore drivers behind the variation in PA funding deficits to provide additional insights on how to improve the financing and performance of the PAs in the region. Our analysis is the largest to date to test the relationship between actual conservation spending in terrestrial PAs and conservation outcomes. It provides evidence on the importance of strengthening the countries' governance structures and improving the budget of PAs, as these factors appear to be critical for reducing forest loss within PAs.

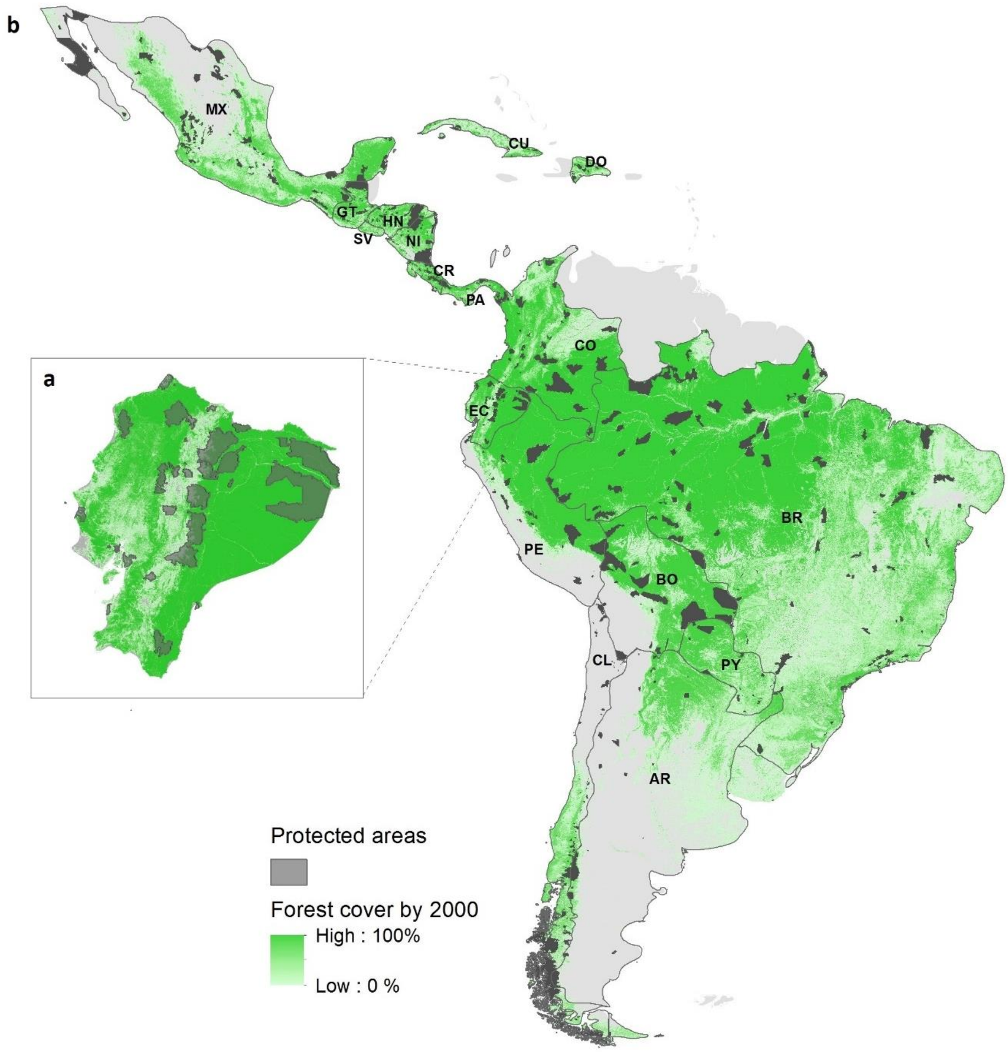

Figure 1.1. Study area**.** We evaluated the impact in avoiding deforestation of (a) 27 PAs within Ecuador and (b) national PA systems of 17 Latin American countries.

2. Methods

2.1. Protected area financial data

We examined the role of funding deficits on PA impact in avoiding deforestation at two administration levels (**Fig. 1.1**). First, at the PA level, focusing on all native-forest PAs of Ecuador declared by 2003 (27 PAs). This analysis level is critical to assess the role of funding since most activities that deliver forest protection take place in individual PAs (e.g., control and monitoring). Ecuador is also an excellent country for such an assessment, being one of the few nations of the region that built specific financial data for each PA over the 2000's decade. Second, we focused on the PA-system level, defined as the aggregation of individual PAs within a country and central operations that affect all PAs (e.g., budget management, setting PA fees; (Bovarnick et al. 2010). Thus, available funding for managing the systems might help explain the overall impact of the PA systems across countries. For this assessment, we included the national PA systems of 17 Latin American countries for which financial data was available. See Supporting Information (Section 1) for more detail on the PA data set.

Funding deficits for PA management was defined as the percentage of the funds required to meet the basic management needs of PA systems or individual PAs not covered by the allocated budget. Basic management refers to the minimum funding needed to operate key conservation programs, including sustaining ecosystem functions in PAs (Bovarnick et al. 2010). This management scenario typically includes administration, participatory planning, and control and surveillance activities in PAs (Galindo et al. 2005). For Ecuador, financial data was from a survey led by the Minister of Environment (Galindo et al. 2005), with the participation of park managers and stakeholders, and it covered the period 2003-2010. Funding deficit data for the PA systems of each country were based on reports verified by individual governments and used for evaluating trends among countries (Bovarnick et al. 2010). This data was primarily constructed with information ranging from 2003 to 2008, and we used them as a broad indicator of the financial situation of the PA systems over the entire decade (2000-2010).

2.2. Forest cover change data

Data on forest cover change was obtained from the Global Forest Change (GFC) time-series analysis (Hansen et al. 2013). The GFC data is a remote sensing product of the percentage of canopy cover per grid cell (30 x 30 m) for all vegetation taller than 5 meters. Forest loss is defined as a stand-replacement disturbance or a change from a forest to a non-forest state. Following Heino et al. (2015), we aggregated data on forest cover and forest loss extent to ~0.5 km^2 and ~1 km^2 cell resolution (at the equator) for the analyses at the PA level and PA-system level, respectively. For the PA systems in Latin America, we estimated the forest cover of each cell in 2000 and the forest loss between 2000 and 2010, which approximates the period covered in the financial report. For the Ecuador analysis, accumulated forest loss was calculated from 2003 to 2010. We discuss the caveats of this forest cover dataset in Supporting Information, Section 2.

2.3. Assessing the protected-area impact

We used matching to account for the potential bias in the locations of PAs when estimating the impact. Statistical matching allows us to compare deforestation rates between treatment sites (i.e., PA sites) and unprotected sites (i.e., control sites) that are similar in respect to a set of covariates hypothesized to affect both the location and impact of protection. Matching was performed in R using the MatchIt package (Ho et al. 2011) and the propensity score matching (PSM) method, after also testing Coarsened Exact Matching (CEM), which did not perform as well as PSM for our dataset (Supporting Information, Section 3). Matching was done without replacement using the nearest neighbor method and a caliper of 0.25 standard deviations of the propensity score as a cutoff for included matches (Stuart 2010). Based on the literature (e.g., Nolte et al. 2013; Carranza et al. 2014; Schleicher et al. 2017; Cuenca et al. 2018) and available spatial information for each level of analysis, selected covariates were related to (1) accessibility, (2) agricultural suitability, (3) initial tree cover, (4) topography and (5) human pressure on the environment. Additionally, sites were matched by exact ecoregion in the case of the PAs within Ecuador and by the exact biome for the analysis across PA systems. See Supporting Information (Section 3) for detailed methods on the matching analysis.

Following matching, we estimated the PA impact as the baseline deforestation avoided by PAs. Specifically, this metric shows how far baseline deforestation rates (those in matched control sites) have been changed by protection, thereby allowing for comparing results of countries or regions with very different baselines (Carranza et al. 2014). For each PA in Ecuador, we calculated the difference between the deforestation rate in the pool of matched control sites and the deforestation rate in the pool of matched PA sites, divided by the deforestation rate found in the pool of matched controls. We used the same metric for estimating the impact at the system level. In this case, we pooled sites across the entire PA system (and their matched controls) without considering which sites belong to which individual PA (Carranza et al. 2014). In this way, each matched PA site had equal weight in the estimation of the system impact. Positive values indicate deforestation rates inside PAs are lower than in control sites. We also use this metric as an indicator of deforestation pressure on protected areas (see below).

2.4. Statistical analyses

As explanatory variables of PA impact, we included factors that have been shown to influence deforestation rates and PA impact (Geldmann et al. 2018), such as (1) socio‒economic and governance attributes of countries, (2) direct and indirect human pressures on forests, and (3) PA design and management characteristics (**Table 1.1**). We also assess whether money is flowing to the places most in need and why some PAs are funded better than others. To investigate this, we produced additional models that explore the circumstances influencing the variations in the funding deficit itself for PAs within Ecuador and among PA systems of Latin America (**Table 1.1**). As explanatory variables for the variation of funding deficit within Ecuador, we included: years since PA establishment (newer PAs might need more time for financial consolidation), elevation and slope (PAs in the Andean mountain range usually receive more revenues from tourism), size (smaller PAs are usually more expensive to manage per unit area), whether the PA had a management plan (which may facilitate the allocation of resources), and deforestation rates in its control sites, as an indicator of the overall pressure on protected forests (PAs under intense human pressures are usually more expensive to manage). For the analysis of funding deficits among PA systems of Latin America, we tested the relevance of the socio-economics variables (countries with healthier economies and better governance may cover the PA needs), PA average size, and deforestation rate in control sites.

Table 1.1. Explanatory variables tested in the models of PA impact and PA funding deficits. We also analyze the response variables at two administration levels: PA level for Ecuador and PA-system level for Latin American countries. Justification for the selection of explanatory variables and their description is given in Supporting Information.

Explanatory variable	**Response variable**	
Socio-economic and governance	*Impact*	*Funding deficits*
Gross Domestic Product (GDP) at purchasing power parity (USD) of countries	System level	System level
GDP growth (%) of countries	System level	System level
Human Development Index (0–1) of countries	System level	System level
Percentage of the rural population (%) of countries	System level	System level
Corruption index perception (0–10) of countries	System level	System level
Polity index (-10–10) of countries	System level	System level
Rule of law (-2.5–2.5) of countries	System level	System level
Poverty index of the municipality (0–1) where PAs are located	PA level	
Direct and indirect human pressure on forests		
Human population density (per km^2) of countries	System level	
Average human population density (per km^2) in the buffer zone of PAs (10 km)	System level	
Human population density growth (%) of countries	System level	
Population growth (%) of countries	System level	
Percentage of agricultural land (%) of countries	System level	
Average travel time to cities (h) from the PA system	System level	
Average travel time to cities (h) from PAs in the system	PA level	
Average opportunity cost for agriculture of (USD per Ha, year)	System level	
Average opportunity cost for agriculture of the PA (USD per Ha, year)	PA level	
Average PA slope (º)	PA level	
Average PA elevation (m)	PA level	
PA perimeter (%) under pressure (top quintile of Human Footprint scores)	PA level	
Deforestation rate (%) in the pool of matched control sites		System and PA levels
PA design and management		
Average size (km^2) of PAs	System level	System level
PA size (km^2)	PA level	PA level
PA perimeter-area ratio	PA level	
Funding deficits (%) of the PA system	System level	
Funding deficits (%) of the individual PA	PA level	
Type of PA management (0: Strict (I-II), and 1: that allow use (III-VI), according to IUCN categories)	PA level	
Years since the PA establishment	PA level	PA level
Management plan (0: no, 1: yes)	PA level	PA level
Overlap of the PA with indigenous lands (0: no, 1: yes)	PA level	

We used Generalized Linear Models in R (R Core Team 2014) to fit models for the PA impact in avoiding deforestation. We transformed the variable of impact (which is continuous, negatively skewed and with negative values) into “permitted deforestation” (i.e., one minus the impact) and fitted models using a Gamma distribution (log link function). In the case of funding deficits, we fitted Beta Regression models since data is proportional and ranging from zero to 1 (Cribari-Neto & Zeileis 2010). We tested models with all possible combinations of explanatory variables for each level of analysis (site and system), included testing polynomials (orthogonal squares, Barnes et al. 2016). We selected the best models based on Akaike Information Criteria (corrected for small sample sizes, AICc), excluding models containing collinear explanatory variables (those with a correlation coefficient higher than 0.5, Burnham & Anderson 2002). Trailing models with higher AICc are reported in Supporting Information (Section 5). Results from model selection were consistent when tested for sensitivity to outliers and different matching parameters (Supporting Information, Section 5).

3. Results

3.1 Impact of protected areas within Ecuador

Twenty-three out of 27 PAs within Ecuador had lower deforestation rates than their matched unprotected landscapes between 2003 and 2010 (**Fig. 1.2a**). The PA with the best performance avoided 99.5% of deforestation, while the PA with the lowest impact had a deforestation rate 157% higher than a similar unprotected area (i.e., for every hectare lost outside, the PA lost ca. 2.7 hectares). Simultaneously, funding deficits for management ranged from 33% to 100%. According to the most parsimonious models, based on AICc

(**Table 1.2**, **Fig. 1.3a**), a lower impact in avoiding deforestation was associated with larger funding deficits ($\beta = 0.028$, $p = 0.021$). Moreover, avoided deforestation was smaller for PAs that overlap with indigenous lands ($\beta = 1.042$, $p = 0.031$) and PAs with most of their perimeter under human pressure ($\beta = 0.018$, $p = 0.025$). Together, these variables explained 36% of the deviance of the dataset.

3.2 Impact of protected-area systems of Latin American countries

Between 2000 and 2010, 16 out of 17 Latin American countries had lower deforestation rates in their PA systems than in matched unprotected sites (**Fig. 1.2b**). Brazil had the PA systems with the highest conservation impact, avoiding 82% of the baseline deforestation. In contrast, Honduras was the only country with higher rates of forest loss inside PAs than matched forests outside (8% higher). According to the financial report, deficits for covering the basic management needs of the evaluated PA system of Latin America ranged from 5% (Bolivia) to 87% (Paraguay).

Based on the AICc, our most parsimonious model explained 62% of the deviance in the data and showed that more positive impacts on avoiding deforestation were associated with higher scores on national Human Development Index ($\beta = -5.23$, $p = 0.001$; **Table 1.2**, **Fig. 1.3b**). Thus, this model suggests that countries with healthier societies, access to a good education, and a higher standard of living are better at protecting forests. The average size of PAs in the system was also retained in the most parsimonious model ($\beta = -0.00001$, $p = 0.03$), but only if including the influential data from Bolivia (Supporting Information, Section 5). The funding deficit was not retained in the most parsimonious model when looking across countries at the level of their entire PA system.

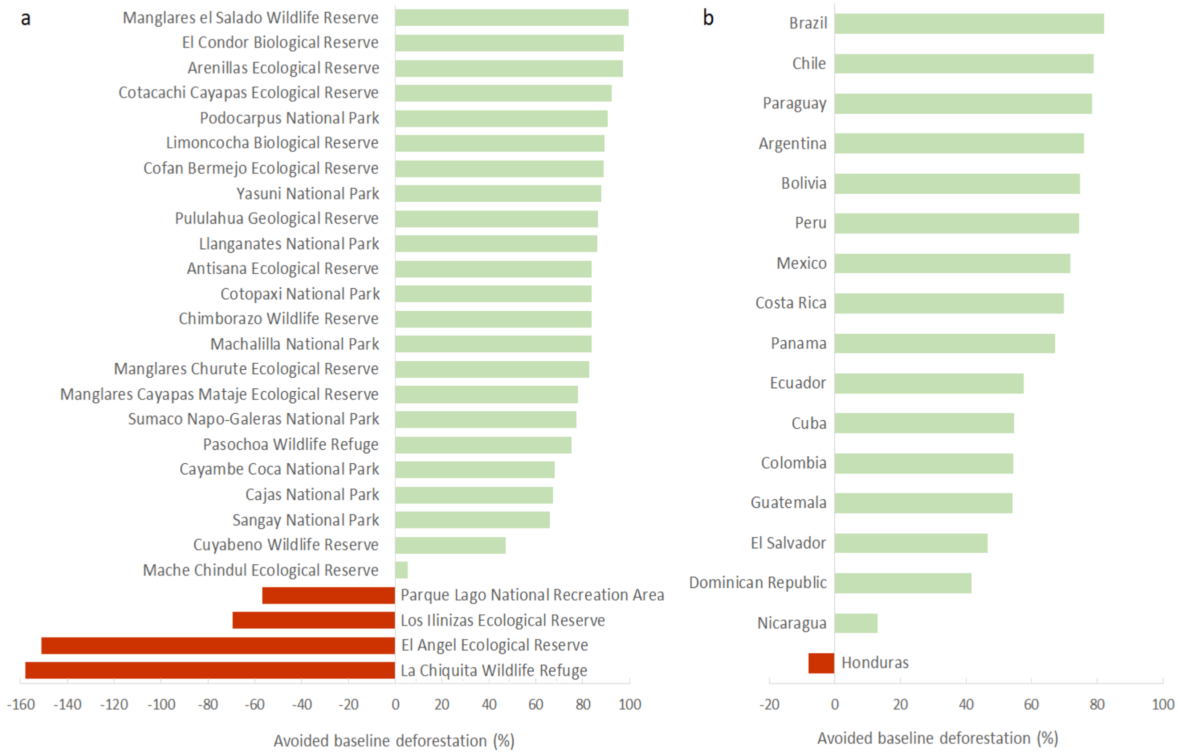

Figure 1.2. Impact of PAs in avoiding deforestation. This impact metric shows how far PAs have changed baseline deforestation rates (i.e., in matched control sites), expressed as percentage. Results are shown for (a) individual PAs of Ecuador (2003-2010) and (b) national PA systems in Latin American countries (2000-2010). Negative impact values (red) indicate PAs with higher deforestation rates than in matched unprotected sites, while positive values (green) correspond to PAs that managed to reduce baseline deforestation.

3.3 Drivers of funding deficits

Among several explanatory variables tested, we found that PAs in Ecuador (**Table 1.2**, **Fig. 1.3c**) with smaller deficits in funding were those with a management plan (β = -1.291, $p < 0.001$), and located at steeper slopes (β = -0.044, $p < 0.001$, 64% of the variance explained). Funding deficits in PA systems (**Table 1.2**, **Fig. 1.3d**) were higher in countries with higher deforestation pressure (β = 2.453, $p = 0.001$) as described by the rate of forest loss in matched unprotected sites (41% of the variance explained). Contrary to what we expected, countries with higher economic growth (measured as changes in GDP) did not spend more resources to close funding deficits for PA management (β = 0.008, p = 0.02).

Table 1.2. Most parsimonious model (according to AICc) for explaining (a) the impact of PAs in Ecuador, (b) the impact of national PA systems in Latin America, (c) funding deficits of PAs in Ecuador, and (d) funding deficits of national PA systems of Latin America. This table presents the modeling results for the variable of impact transformed as "permitted deforestation" (i.e., one minus impact). Thus, explanatory variables with a positive estimate indicate an increase in PA ineffectiveness. 2nd order polynomials are indicated by superscript. Significance of regression coefficients: ***p < 0.001, **p < 0.01, *p < 0.05

Model	**Explanatory variables**	**Estimate**	**SE**	***t***	***p***	**Explained deviance**
a. (1 – impact) of PAs of Ecuador	Intercept	-4.458	1.014	-4.396	< 0.001***	0.36
	PA perimeter (%) under pressure	0.018	0.007	2.401	0.025*	
	Funding deficits (%)	0.028	0.011	2.486	0.021*	
	Overlap with indigenous lands (yes: 1; no: 0)	1.042	0.452	2.304	0.031*	
b. (1 – impact) of PA systems	Intercept	2.869	0.849	3.382	0.004*	0.62
	HDI	-5.295	1.229	-4.306	0.001***	
	Average PA size (km^2)	-0.00001	0.000	-2.422	0.03*	
Model	**Explanatory variables**	**Estimate**	**SE**	***z***	***p***	**Pseudo R^2**
c. Funding deficits of PAs of Ecuador	Intercept	2.425	0.332	7.313	< 0.001***	0.64
	Management plan (yes: 1; no: 0)	-1.291	0.305	-4.228	< 0.001***	
	Slope (º)	-0.044	0.015	-2.933	0.003**	
d. Funding deficits of PA systems	Intercept	-1.511	0.487	-3.103	0.002**	0.41
	GDP growth (%)	0.008	0.003	2.326	0.02*	
	Deforestation (%) in control sites	2.453	0.764	3.212	0.001**	
	Deforestation (%) in control sites ^{2}	-1.539	0.767	-2.008	0.045*	

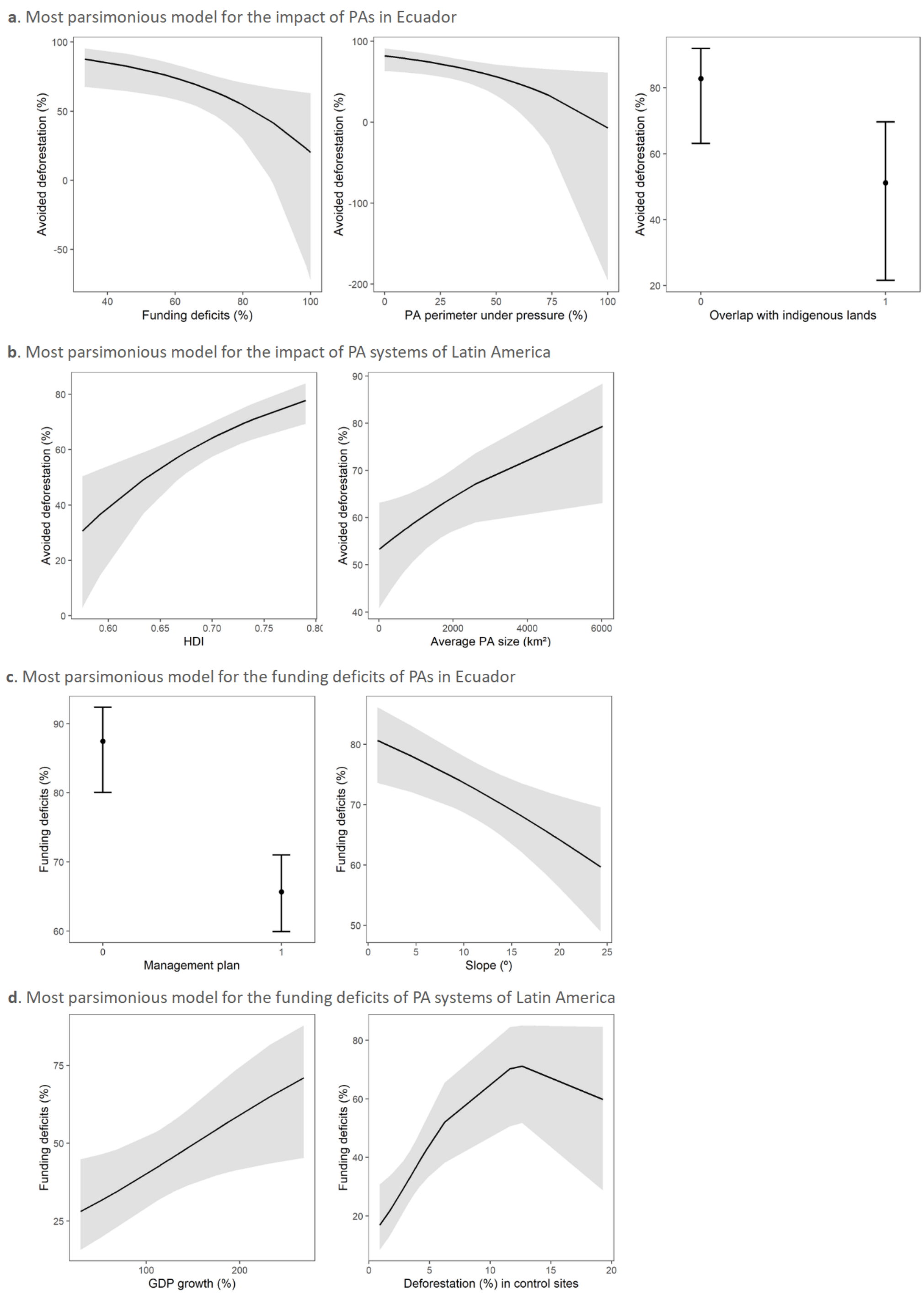

Figure 1.3. Marginal effects plots for the most parsimonious models (according to AICc). Models explain the variation in the impact in avoiding deforestation for (a) individual PAs of Ecuador and (b) national PA systems of Latin American countries. Additional models were built to explore the factors behind the levels of funding deficits for (c) individual PAs of Ecuador and (d) national PA systems of Latin America. Each plot shows the relationship between the response variable and the indicated explanatory variable while adjusting for interference from other explanatory variables. Grey shadows are 95% confidence intervals.

4. Discussion

Over the decade between 2000 and 2010, most individual Ecuadorian PAs had significantly less forest loss than matched control sites. Similarly, most national PA systems of Latin American countries experienced lower deforestation rates than analogous areas in the unprotected landscape. These findings are encouraging and suggest that PAs have helped to avoid deforestation. However, our results also show that none of the PA systems was able to completely prevent losses of forest cover, with some PAs being ineffective. Thus, our results concur with other studies in the region (e.g., Cuenca et al. 2016; Herrera et al. 2019; Schleicher et al. 2019b) in making clear that merely declaring PAs does not guarantee long-term preservation of forests in Latin America.

Our analysis is the first to explicitly explore the relationships between funding allocation and conservation outcomes at a PA level within a country, and at the system level for multiple countries. Results show that funding deficits are a significant driver of PA impact within Ecuador. Moreover, PAs were less effective when insufficient funding for management was coupled with intense human pressures. Other studies have found similar links between human pressure and PA impact but without being able to show the effects of funding (Geldmann et al. 2019). By contrast, PAs in remote areas of the Amazon, with low human pressures, were effective in abating deforestation almost entirely, even when severely underfunded. While these results show that ensuring adequate funding and active management in PAs is critical, particularly in areas under higher pressure, we caution that this may lead to neglecting more remote areas that often harbor the last of Earth’s wild places and will likely come under increased pressure (Watson et al. 2016).

We also found that avoided deforestation was lower in PAs of Ecuador that overlap with indigenous lands, as similar analyses for the Ecuadorian Amazon have also detected (Holland et al. 2014). This finding does not imply that PAs located in indigenous lands are ineffective. Indeed, several studies have shown that indigenous lands can be as effective or more effective than PAs (e.g., Schleicher et al. 2017; Herrera et al. 2019). Instead, this result likely reflects that PAs in indigenous lands per definition are inhabited and might have multiple objectives besides strict preservation of forests, such as subsistence crop production (Vasco et al. 2018). We also stress that effective forest protection in PAs requires the genuine participation of local communities in the decisions of rules that regulate the use of forest resources (Ostrom 2015). However, in Ecuador, indigenous and local communities often have little input in PA management decisions (Negru et al. 2020). Thus, future research could explore whether factors related to equitable management helps explain the conservation impact of PAs inhabited by indigenous people.

When looking across countries, national human development scores (as an indicator of governance) better described the PA-system impact. Lower human development has been associated with weak law enforcement, high corruption, and intense pressure on forest resources by populations in poverty, which together can undermine ecological outcomes of PAs (Barnes et al. 2016; Amano et al. 2018; Geldmann et al. 2019) and increase forest loss rates in the countries (Jha & Bawa 2006; Kauppi et al. 2018). Thus, it is perhaps not surprising that national socioeconomic factors were the most influential variable in determining deforestation rates at the country level, which in turn sets the “baseline” for deforestation rates within the PA systems. Results from Ecuador suggest that once country-level effects kept constant across all PAs, funding plays a key role in explaining the PA impact. We also emphasize that funding deficits tended to be higher in national PA systems exposed to larger deforestation pressures, precisely where resources would have the most significant conservation impact. As a consequence of this non-random resource allocation, analyses may not detect a relationship between funding and avoided deforestation across PA systems, even if the actual underlying relationship is positive (Schleicher et al. 2019b). By contrast, within Ecuador, funding deficits were not biased towards PAs experiencing higher deforestation pressure, which partially explains why this analysis could capture the influence of funding in the impact of individual PAs.

Our use of a counterfactual approach allowed for isolating the impact of PAs as compared to more appropriate controls than comparing to all non-protected land (Schleicher et al. 2019a). Moreover, since our indicator of PA impact was based on remote sensed forest cover, it was not influenced by the resource allocation in PAs. This independence is crucial for ensuring an unbiased evaluation of the effect of funding on PA impact. We also analyze unique data on funding deficits at a PA-system level, which offered an uncommon opportunity to explore trends on the role of funding at a regional scale. Still, methods to estimate these financial needs were not 100% uniform among countries (Bovarnick et al. 2010). Therefore, to improve the precision of future analysis, countries must develop, update, and standardize the documentation on financial needs and resource allocation for individual PAs, including information on how much money these PAs lose because of inefficient use of resources (Flores 2010).

According to our results, the impact of PAs on preserving forests can be significantly improved through a multi-level approach that includes better resource allocation for PAs, combined with considerations of the national socioeconomic conditions that support PA management. Our exploration of the forested PAs in Ecuador shows that closing funding gaps is likely to have a greater impact on curbing deforestation if funds are directed first to PAs experiencing stronger anthropogenic pressures. Moreover, investing in developing proper management plans for each PA seems to facilitate allocating the required budget and reducing the gap (Flores & Bovarnick 2016). These findings are informative for managing individual PAs in other Latin American countries, where many forest PAs lack proper planning, are experiencing large funding deficits and high pressure from economic activities, as occurred in Ecuador (Flores & Bovarnick 2016; Coad et al. 2019). Our results also show a need of strengthening institutions and governance of PA systems to improve their impact. This approach will also require a holistic strategy focuses on poverty alleviation and improving the livelihood of rural populations (Jha & Bawa 2006; Barnes et al. 2016). The fact that national PA systems under higher pressures tended to have larger funding deficits merits serious attention. Thus, to increase the impact of PAs on halting biodiversity loss, we encourage future global conservation agreements to generate mechanisms to assist biodiverse countries financially and to establish more ambitious goals in terms of the quality of national PA governance.

5. Appendices

Supplementary Information. Extended methods.

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Chapter 2 - Cost-effective protection of biodiversity in the western Amazon

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Abstract

The western Amazon needs to expand its protected-area system to ensure the conservation of its immense and threatened biodiversity. However, potential expansions often meet with resistance because of scarce government resources and competing social priorities. Here, we proposed an expansion of the protected-area system for the western Amazon that increases biodiversity conservation at minimum costs. We started by evaluating biological data to establish conservation targets for enhancing the protection of 2419 species of plants and vertebrates. We then built a map that shows the variation in costs of effectively managing lands as protected areas. We also adapted an opportunity cost layer for agriculture and livestock to approximate realistic foregone incomes when a particular extent of land is protected. These cost estimates were used in a decision-support tool to find the most inexpensive places to achieve the conservation targets. We found that this cost-optimized expansion would reduce annual costs by 22% in comparison to an expansion planned without cost data. Moreover, without collaboration with indigenous peoples and without cooperation among the western Amazon countries costs would be 39% and 49% higher, respectively. The cost of the proposed expansion, estimated at US$ 100 million annually, is only a fraction of the regional Gross Domestic Product (0.018%). Thus, this study may help governments and conservation agencies to improve the financial planning of the region's reserve network by maximizing species protection at more affordable costs.

Keywords: management cost, opportunity cost, systematic conservation planning, protected-area budget.

1. Introduction

Establishment of protected areas is a fundamental strategy to preserve the large biodiversity of the western Amazon. This region, which includes the Amazon of Colombia, Ecuador, and Peru, stands out as one of the most biodiverse places in the world for amphibians, birds, mammals, fish, and vascular plants (Bass et al. 2010; Jenkins et al. 2013). Moreover, the western Amazon still retains large tracts of intact tropical forests (Potapov et al., 2017) and is home to at least 140 indigenous peoples (RAISG 2012). However, the biological and cultural diversity of the region faces numerous threats resulting from the expansion of oil extraction, mining, hydroelectric projects, illegal logging, agriculture, and large-scale road infrastructure (Finer et al. 2015; Finer and Jenkins 2012; RAISG 2012; Venter et al. 2016). To address these threats and maintain the integrity of ecosystems, Colombia, Ecuador, and Peru have established large protected area systems that harbor the most preserved ecosystems of the region and, in several cases, have been proved to be successful in curbing anthropogenic threats and conserving biodiversity (Rodriguez et al. 2013; Schleicher 2018; Schulman et al. 2007).

Unfortunately, the protected areas of the western Amazon have serious deficiencies in funding and biological representation. According to economic studies, several Amazonian protected areas face large funding shortfalls that hinder their effective management and jeopardize their ability to accomplish the conservation targets they pursue (Bovarnick et al. 2010; Ministerio del Ambiente 2015). At the same time, Amazonian protected areas have numerous gaps in terms of biodiversity representation, which means that many species and ecosystems are insufficiently covered or are absent from protected-area systems (Fajardo et al. 2014; Lessmann et al. 2016; Lessmann et al. 2014; Schulman et al. 2007). In this context, the long-term persistence of Amazonian biodiversity requires both meeting the funding needs of protected areas and expanding protected-area systems for improving biodiversity representation.

Given the limited budget destined to protected areas and the competing social priorities in the western Amazon, the resources for expanding its protected-area system should be allocated efficiently (Brown et al. 2015; Margules and Pressey 2000). As the cost of protecting sites varies widely (Armsworth 2014), a cost-efficient expansion requires a thoughtful design that identifies places which protection optimizes increases in biodiversity representation at the least possible costs, such as management and opportunity costs.

Management costs are those associated with enforcing and maintaining protected areas, including personnel and operating costs (Naidoo et al. 2006). Prioritizing the protection of areas that would have low management needs implies lower budgets and better chances for these areas to be adequately funded (Armsworth et al. 2018; Green et al. 2012). Moreover, protecting sites of minimal opportunity costs, which reflects foregone incomes when land is protected, may reduce the conflicts between resource users and conservationists, the annual payments to compensate for lost incomes, and the land acquisition costs (Ban et al. 2011; Naidoo et al. 2006).

Originally formulated in the conservation biology field, the prioritization of conservation areas has more focused on fulfilling biological needs, while economic factors have been rarely integrated into the planning framework (Albers et al. 2016). In addition, detailed spatial data on land-conservation costs for the western Amazon remains poorly explored. Therefore, several prioritization studies in the region have not been able to evaluate the direct costs of land protection (e.g. Cuesta et al. 2017; Lessmann et al. 2016; Rodriguez and Young 2000). Instead, they have focused on minimizing surrogates of costs, such as the amount of land to be protected or the ecological impact of human activities in these areas. Such lack on explicit cost data may have limited our ability to identify priority areas of high cost-effectiveness.

This study addresses two fundamental questions for conservation planning in the western Amazon (**Fig. 2.1**): 1) what is the most cost-effective way to expand the region's protected-area system? and, 2) how much resources would that expansion save? As a first step, we drew on published studies to build regional models of the cost for effectively managing current protected areas. Based on these models, we built a map for the western

Amazon that reflects the different costs of managing lands if they are protected. In addition, we used existing information on current and potential crop revenues as the starting point for creating a map of opportunity costs for agriculture and livestock. These cost layers, together with species distribution maps, were used to identify priority areas that increase the representation of the region's species diversity at the least possible costs. We also estimated the savings resulting from incorporating conservation costs into the prioritization and discussed the challenges and the recommendations for expanding the protected-area system of the western Amazon in a cost-effective way.

2. Methods

2.1 Study area

The western Amazon (**Fig. 2.1**) covers 1 414 641 km^2 and represents 43% of the surface of Colombia, 38% of Ecuador, and 63% of Peru (RAISG 2012; Sierra 1999). This region is classified into three sub-regions according to elevation (Hoekstra et al. 2010): Amazonian lowlands (<250 m), Amazonian foothills (250–800 m), and Andean Amazon (>800 m). The region has an extensive reserve network composed of 67 state-protected areas partially or totally located within the western Amazon: 18 in Colombia, 13 in Ecuador, and 36 in Peru. These protected areas occupy 20% (287 412 km^2) of the region and approximately 20% of the Amazon of each country and sub-region. Thirty-three percent of the western Amazon and 16% of its protected areas overlap with indigenous lands, which include titled lands, ancestral lands, and areas inhabited by people in voluntary isolation (RAISG 2012).

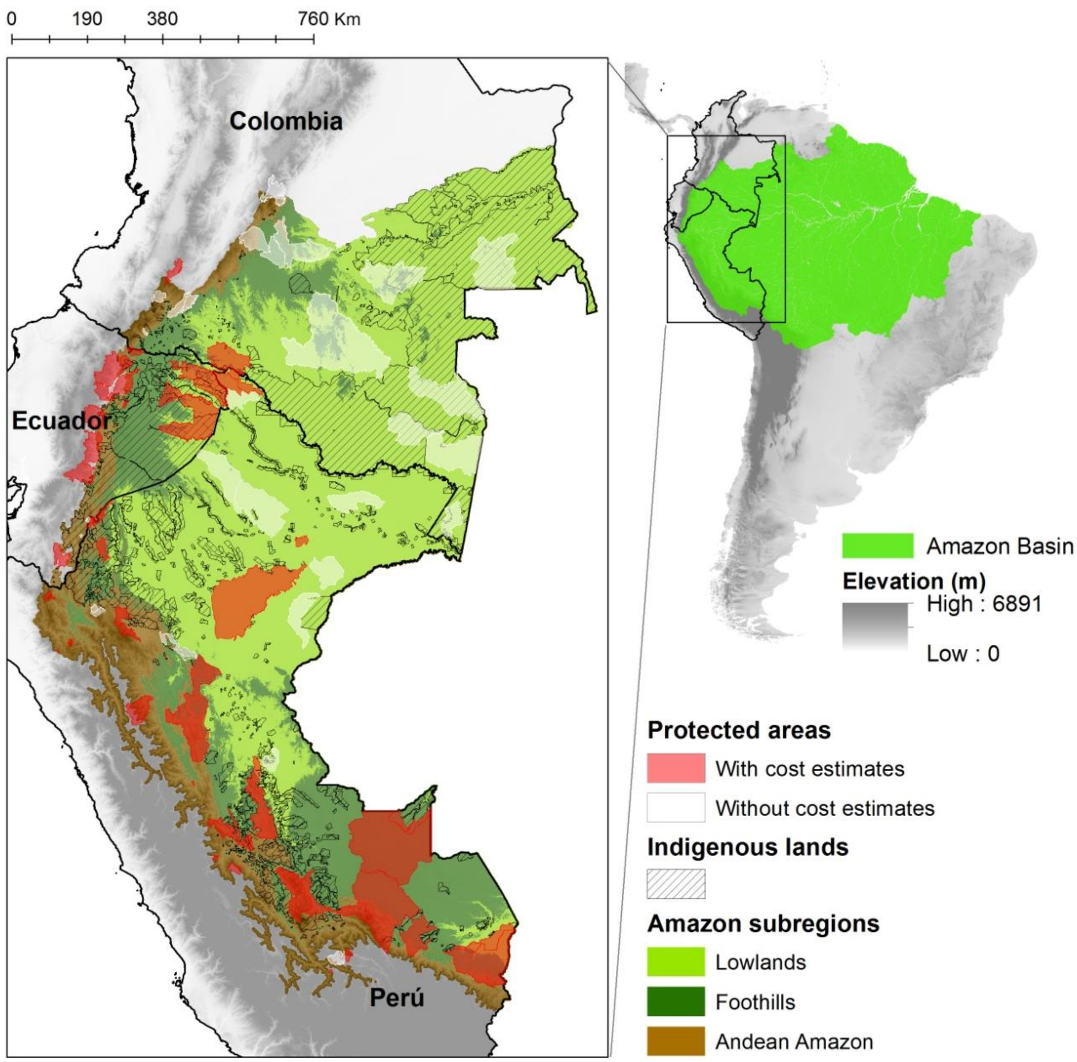

Figure 2.1. Western Amazon (Colombia, Ecuador and Peru). Amazonian protected areas with estimates of costs for effective management are highlighted in red, whereas those protected areas without available economic estimates are cleared.

2.2 *Estimating land conservation costs for the western Amazon*

We evaluated and mapped two types of costs associated with land conservation: (1) costs of effectively managing land as a protected area, and (2) opportunity costs from foregone agriculture and livestock when setting a piece of land for conservation. Both maps were used to guide the prioritization towards areas with the greatest species value per cost.

2.2.1 Costs for an effective management

Previous studies at global and continental scale have modeled the costs of effectively managing protected areas (e.g. Balmford et al. 2003; Bruner et al. 2004; Moore et al. 2004). According to these studies, the size, location, and design of protected areas explain a significant percentage of their differences in management costs. Therefore, these models could be spatially projected to build a map that reflects the management costs of protecting a specific site. Also, these models may guide the design of new protected areas to minimize their management costs. Here, we built models at a regional scale that consider drivers that may influence the management costs of Amazonian protected areas. As the independent variable for this modeling, we used available estimates of the costs of basic and effective management of current protected areas in Ecuador (Galindo et al. 2005), Peru (León 2005), and Colombia (Londoño 2013). While current spending is insufficient, estimated costs of effective management are the funds required to ensure basic operations within protected areas (Balmford et al. 2003). A total of 44 out of the 67 Amazonian protected areas had specific estimates of their annual basic management cost: 11 from Ecuador, 31 from Peru, and two from Colombia (**Fig. 2.1**; Supplementary Information). Although few Colombian protected areas had management cost data, the Amazon regions of the three countries share many natural and social characteristics, justifying the adequacy of building a regional model from the available data.

We constructed linear regression models to explain the variation in the annual management cost per-area (US$/year per km^2). As a preliminary step, we inflated these management costs to 2016 values (COIN NEWS 2016) and transformed the values using a base-10 logarithm scale (Balmford et al. 2003; Bruner et al. 2004). As predictor variables, we evaluated the following attributes of Amazonian protected areas that may influence management costs: size, management objectives, years since their establishment, inaccessibility, human population density inside them, average human footprint, distance to the nearest other protected area, distance to villages, average slope, presence of indigenous lands, proportion of land under operative oil blocks, and national Gross Domestic Product per capita of the country where the protected area is located. When needed, we transformed predictor variables to improve the normality of data. The potential importance of each variable on management costs is specified in **Table 2.1**.

We constructed two regression models. For the first one (hereafter, General Model) we tested all predictor variables, and therefore, it represents a best estimate of the drivers of variation in management costs. However, this model may depend on attributes of potential new protected areas that are difficult to pre-establish in a prioritization exercise, such as their size (which is a result of the prioritization algorithm) or management objectives. Thus, we built a second model (the Context Model) which only includes variables that describe the location of protected areas (**Table 2.1**) and that we projected spatially to build a map of the costs of managing sites (using square cells of 1 km^2) as protected areas.

Models were obtained by using both forward and backward stepwise procedures, with the alpha-to-enter and alpha-to-exit value set at 0.05. Best models were selected according to their Akaike Information Criterion (AIC), Bayesian Information Criterion (BIC), and explanatory power (R^2) values. Models containing collinear predictors (those with correlation Pearson coefficient higher than 0.7) were discarded. We also tested if the interaction between the selected variables improved the explanatory power of models. We performed all the analyses in R (R Core Team 2014).

Table 2.1. Predictor variables for management cost models of protected areas in the western Amazon. Location variables reflect the geography characteristics and anthropogenic pressures of the protected areas. Information on these variables were obtained from the sources specified in the Supplementary Information.

Variable	Transformation	Importance
Non-location related		
Size (km^2)	Log_{10} x	Larger protected areas have been associated with lower management cost per-area.
Management objectives. Strict protection (0: I-IV categories) or sustainable use (1: V and VI categories).	Categorical variable	Different management objectives may require different activities and expenditures.
Years since establishment	√x	Recent protected areas may need more budget for their consolidation.
Location		
Average inaccessibility (travel time to major cities)	√x	Inaccessible protected areas may be less vulnerable to human activities, requiring lower costs for protection.
Average population density (inhabitants / km^2)	Log_{10} (x+1)	Protected areas in highly populated zones may require more funding to resist pressures.
Average human intervention (1-100)	Not needed	Protected areas in highly intervened zones may require more funding to control and restore ecosystems.
Average distance to villages (km)	Log_{10} x	Protected areas close to populated areas may suffer greater human pressure, requiring more funding.
Average distance to the other protected areas (km)	√x	Nearby protected areas may reduce the overall human pressure, decreasing management costs of neighbor areas.
Average slope (0-90)	-	Protected areas with greater average slope may be more complex to access, decreasing the threats and the surveillance costs.
Presence of indigenous lands (0: no, 1: yes)	Categorical variable	Indigenous territories may be a support for protected area management, reducing overall costs.
Proportion in operative oil blocks (0-100)	√x	An overlap with oil blocks may increases the management costs in order to address the environmental impacts of this industry.
Gross Domestic Product per capita, PPP.	-	Countries with higher GDP have been related to expensive reserves.

2.2.2 Opportunity cost map

We created a map layer of opportunity costs of clearing land based on the global map of gross economic rents from agricultural lands and cattle rising produced by (Naidoo and Iwamura 2007). Since this map reflects gross earnings (i.e., costs of production are not included), we made an adjustment to net profits, assuming a profit margin of 15% for all types of crops (Busch et al. 2009; Strassburg et al. 2008). In Naidoo and Iwamura's original map, profits from clearing for agriculture are mainly based on the characteristics of climate and soil, rather than actual production or potential productivity as dependent on accessibility (Naidoo and Adamowicz 2006). To adjust for this, we multiplied the net profits by the probability of land conversion (ranging between 0 and 1), which is based on the concurrence of current and projected roads in the area from (Soares-Filho et al. 2006). All profits were adjusted to 2016 values (COIN NEWS 2016). Opportunity costs related to other economic activities in the western Amazon, such as oil extraction and mining, were not considered since the potential and current profits have not been systematized and are difficult to access in the region. Moreover, since oil blocks cover a large portion of the western Amazonia (RAISG 2012), conservation planning at a regional scale has been forced to coexist with these blocks, despite of the negative impact of this economic activity on local biodiversity.

2.3 Target species

We based the identification of priority areas on birds, amphibians, mammals, reptiles, and vascular plants. For these species, there are more complete inventories across the western Amazon, as well as available references about their taxonomy and distribution

(Supplementary Information). Occurrence records for these species were gathered from specimen databases of natural history collections (Supplementary Information). Then, we approximated the geographic distributions of these species by building species distribution models with Maxent (Phillips et al. 2006), using the bioclimatic variables from Worldclim 1.4 at ~1 km^2 spatial resolution as ecological predictors. We also included the modeled distribution maps of 62 species (42 amphibians, 15 birds, and 5 mammals) endemic to the east slope of the Andes in Peru, generated by (Young et al. 2007). As a result, the species set was composed of 1,445 birds, 132 mammals, 327 amphibians, 219 reptiles, and 296 endemic vascular plants (a total of 2,419 species). See the Supplementary Information for more information on species distribution modelling and its caveats for conservation planning.

To guide the selection of priority areas maximizing species representation, it is necessary to establish conservation targets, here defined as the minimum proportion of each species' distribution to be included in a protected-area system. When these targets are met, species are considered as represented. Ideally, conservation targets should be established for each species according to their sensitivity to forest loss and habitat disturbance (Ardron et al. 2008). However, assessing species sensitivity requires specific information on their particular ecological requirements, which are not available for all species included in our study. Instead, we used species geographic range size to inform the establishment of conservation targets. The geographic range size is strongly related to the probability of species persisting in degraded habitats, with populations of narrow-ranged species being more vulnerable to local extinction in human-disturbed land (Newbold et al. 2018). Thus, we scaled the targets between 50% of the distribution for species with a range of 2,000 km^2

and smaller, and 5% for those with ranges larger than 200 000 km^2 (Rodrigues et al. 2004). Moreover, for species classified as vulnerable, endangered, and critically endangered (IUCN 2014) the maximum possible target was set to 75% of their distribution. Thus, species of high priority for conservation (those with small ranges and threatened) received the highest conservation targets. See the Supplementary Information for more information on the definition of species conservation targets.

2.4. Identification of priority areas of high cost-effectiveness

We used the decision support tool Marxan (Ball et al. 2009) to identify areas of high cost-effectiveness. Marxan's simulated annealing algorithm permits the selection of a set of planning units (PUs, squares of ~ 13 km^2) that meet predefined species conservation targets, while minimizing the total cost of protecting the selected PUs. For this prioritization, we followed a complementary-based approach, in which the proportion of species distributions already protected were considered for the target achievement.

Information on costs generated as described in the previous sections was included in Marxan in two ways. First, the cost of each PU was calculated as the sum of its management and opportunity costs, taken from both cost maps. Hence, the selection of PUs in relatively more expensive sites was minimized. Second, since small protected areas have higher management cost per unit area (see Results), we calibrated Marxan's Boundary Length Modifier (BLM) parameter to prioritize larger, better connected areas, rather than smaller and scattered areas. To accomplish this task, Marxan may add sites that connect smaller areas to make a single larger protected area, but such land addition may increase opportunity costs. Thus, to generate optimal results, we calibrated the algorithm by progressively increasing land connectivity until we detected the start of an exponential rise in the total extent and opportunity costs (Ardron et al. 2008). Finally, we excluded highly intervened areas from the selection (PUs with average human footprint index >10, (Venter et al. 2016) since conservation actions may be unfeasible there and ecosystems no longer have habitat to harbor species.

Additionally, we assessed the efficiency of the selected priority areas. Each priority area was composed of adjacent PUs selected by the algorithm. We calculated the efficiency as the contribution to increase species representation offered by the protection of a priority area, divided by its total cost (management costs from the General Model plus opportunity costs). Specifically, the contribution consisted on estimating how much of the target (in terms of percentage) of an unrepresented species is reached by a priority area. Then, for each priority area, we calculated the average contribution to all unrepresented species. Thus, priority areas that offer large conservation gains per dollar were considered good candidates for being protected first.

We also reran the prioritization under different scenarios to understand the scope of the savings resulting from including data on conservation costs and the constraints and needs for achieving these savings. In the first rerun, we used uniform costs by fixing the PU costs at US$500 (which approximates to the average total cost of all PUs). Thus, the optimization was based on minimizing the total extent of the expansion. The second re-run excluded the indigenous lands from the selection of priority areas, because it has been suggested that establishing protected areas in these lands could have additional challenges and complex implications for indigenous peoples (Schuster et al. 2018). To understand the advantages of this prioritization at the regional scale, a third re-run assumed no collaboration between the three western Amazon countries (Colombia, Ecuador, and Peru). Here, priority areas were specified to meet the targets independently within each country, overlooking the potential contribution of the other western Amazonian countries to achieve overall species representation.

3. Results

3.1. Management and opportunity cost estimates

Observed annual cost per unit-area for an effective management varied widely among current protected areas of the western Amazon, from US$ 7 per km^2 to US$ 9,156 per km^2 (average US$ 871 per km^2). The size of protected areas alone explained 76% of this cost variation. Therefore, the General Model, which includes only the protected-area size and whether they overlap with indigenous lands, was highly explicative ($R^2 = 0.81$). This model indicates that management cost per unit-area is higher in smaller protected areas and in protected areas outside indigenous lands (**Table 2.2**).

The Context Model, which excluded protected area size, was less explicative than the General Model, but was still able to explain much of the variation in management costs ($R^2 = 0.55$) through the incorporation of accessibility. Specifically, management costs were higher for protected areas located in accessible zones, as well as (again) outside indigenous lands (**Table 2.2**). An interaction between indigenous lands and inaccessible areas was also significant and positive, which means that managing protected areas in indigenous lands that are accessible is particularly cost-efficient.

Table 2.2. Best regression models for protected-area management cost per year and unit area (in log scale) for the western Amazon. The General Model was built with all predictor variables, whereas the Context Model used only evaluated variables related to the location of the protected areas.

Dependent variable ($\log_{10}$ US$/ year, km^2)	**Selected predictor variables**	**Coefficient**	**P-value**
General Model $R^2 = 0.81$ $P < 0.001$	Intercept	4.799	< 0.001
	Size ($\log_{10}$)	-0.696	< 0.001
	Indigenous lands	-0.336	0.0028
Context Model $R^2 = 0.55$ $P < 0.001$	Intercept	4.361	< 0.001
	Inaccessibility ($\sqrt{x}$)	-0.047	< 0.001
	Indigenous lands	-1.783	0.0011
	Inaccessibility ($\sqrt{x}$) * Indigenous lands	0.036	0.0112

The map of management cost built from the Context Model (**Fig. 2.2A**) showed that the Andean Amazon concentrates areas of high management cost for protection, whereas Amazonian lowlands are, in general, less expensive. This pattern is generally mirrored by opportunity costs, as the highest opportunity costs also occur in the Andean Amazon (**Fig. 2.2B**). Although several remote lowland areas in the western Amazon have high agricultural potential, their access to markets is limited, resulting in low opportunity costs.

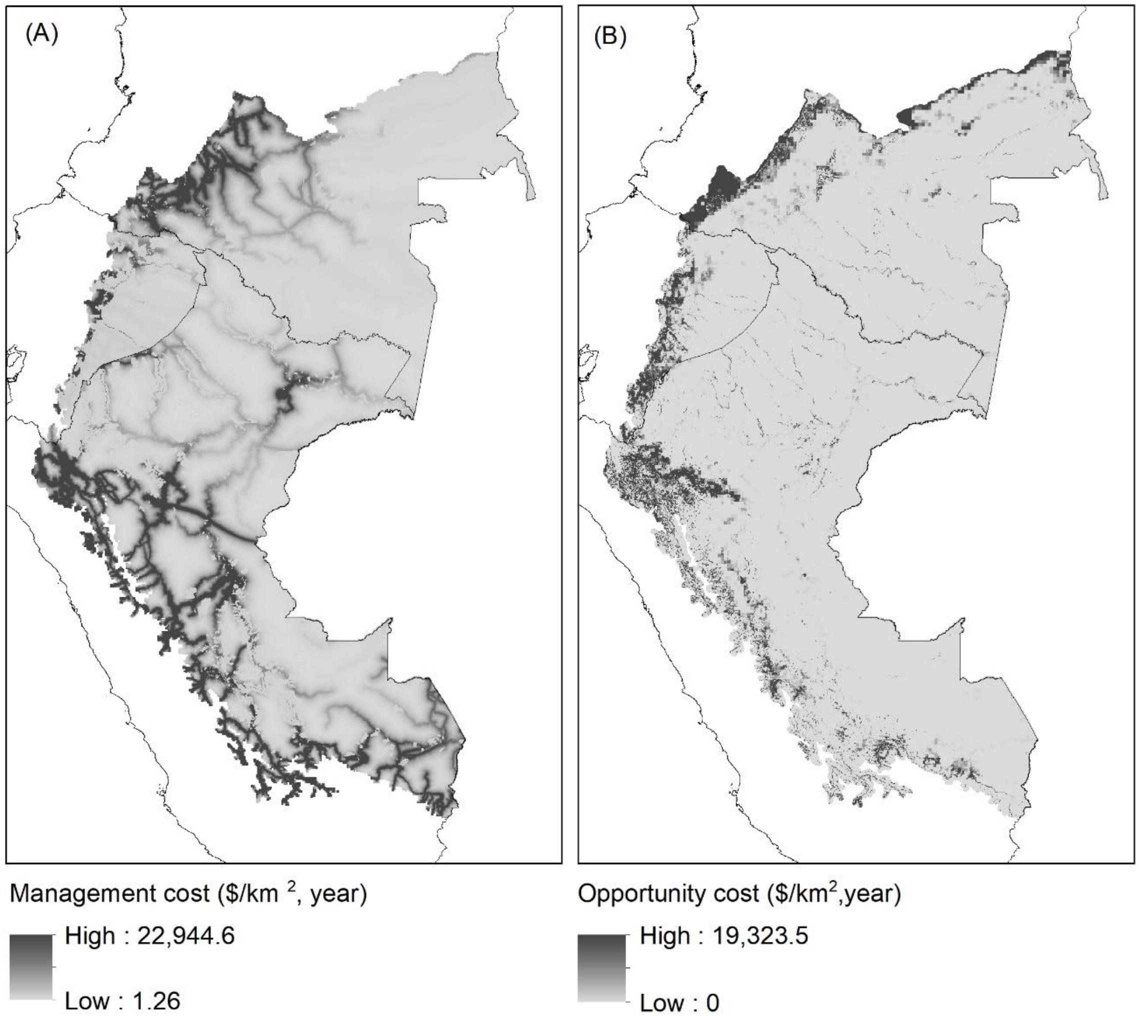

Figure 2.2. Maps of potential conservation costs of protected areas in the western Amazon. (A) Annual costs per-area for an effective management based on the location attributes of protected areas. (B) Opportunity cost is the result of combining information on the net profits of crops and livestock, with the likelihood of land conversion.

3.2 Cost-effective priority areas

A total of 1,777 out 2,419 analyzed species were adequately covered by the current protected area systems according to the established targets. Thus, an expansion of the protected areas is still needed to cover 27% of the species (n = 642). Importantly, a percentage as large as 70% of these insufficiently protected species are partially or totally restricted to the Andean Amazon sub-region (Fig. 2 in Supplementary Information). As a result, the Andean Amazon is the sub-region with the highest proportion of its extent in priority areas for expansion (38%), followed by Amazonian foothills (14%), and Amazonian lowlands (9%). Overall, we identified 297 priority areas (**Fig. 2.3A**), most of them of small size (< 100 km^2) and with different levels of accessibility, from very remote (5 days of travel to major cities) to highly accessible lands (6 min of travel; average of all priority areas = 1,079 min). Together, priority areas occupy ~16% of the western Amazon (223 622 km^2) and, with the current protected areas, achieved the targets for 2,411 species (99.7% of the assessed species).

Using the General Model of management cost, we calculated that an expansion of the protected-area system based on the selected priority areas would cost ~ US$ 71.4 million per year to cover for effective management. According to the 95% confidence interval associated to our model, this estimate on management cost could range between US$ ~48 million and US$ ~108 million per year. An additional ~ US$ 28.5 million correspond to opportunity costs (**Table 2.3**). Thus, the annual total cost of the priority areas ranged from to US$ 68 per km^2 to US$ 6,399 per km^2 (average US$ 1,925 per km^2). Most priority areas are located in Peru, thus US$ 68 million per year of the total cost would corresponds to this country, US$ 23 million to Colombia, and US$ 8 million to Ecuador (Table 2 in Supplementary Information). Moreover, our analysis of efficiency (**Fig. 2.4**) suggests that there are priorities even within these priorities. The 20 most efficient priority areas would only cost US$ 20 million per year (US$ 9.3 million for management, and US$ 10.7 million for opportunity costs), cover as little as 7% of the western Amazon, and contribute to achieve the conservation target of 437 species (67% of those insufficiently protected).

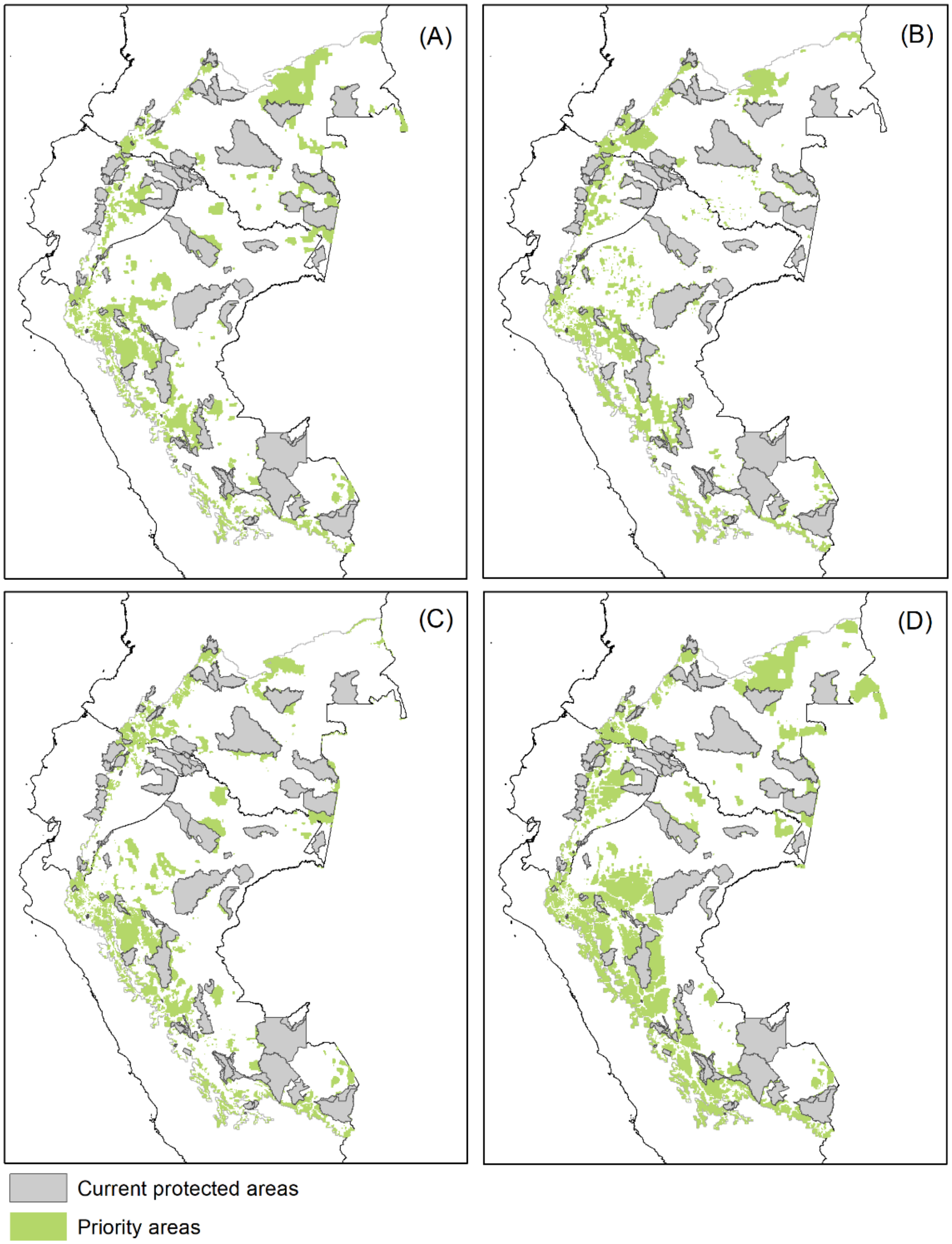

Figure 2.3. Priority-area networks identified under different scenarios. (A) Cost-optimized priority areas, (B) priority areas when considering uniform costs, (C) priority areas when excluding indigenous lands from the selection, and (D) priority areas when assuming non-international collaboration among the western Amazon countries.

Table 2.3. Conservation costs of priority-area networks in the western Amazon identified under different scenarios. Annual costs for an effective management were estimated from the General Model, whereas opportunity costs correspond to adapted profits from agriculture. * LCL: Lower Control Limit; UPL: Upper Control Limit (UCL).

Prioritization scenarios	**Cost (US$ million / year) of the priority-area network**					**Area (km²)**	**Cost per represented species (US$/ # species, year)**
	Management			Opportunity	Total		
	Estimate	95% LCL	95% UCL				
Cost-optimized	71.3	47.9	108.2	28.6	99.9	223,622	41,476
Uniformed costs	94.9	59.1	156.8	32.3	127.3	182,687	52,791
Excluding indigenous lands	108.2	68.3	175.9	31.2	139.3	198,805	58,103
Without international collaboration	109.1	70.3	173.6	40.4	149.5	325,104	62,005

Regarding alternative prioritization scenarios (**Fig. 2.3**, **Table 2.3**), we found that priority areas resulting from the rerun that assumed uniform costs (i.e., simulating expansion planned without using cost information) implies an annual investment of ~US$ 127 million (~US$ 95 and ~US$ 32 million per year for management and opportunity costs, respectively). Therefore, this scenario, although it encompasses 13% of the western Amazon, increases costs by 27% in comparison to the planning that considered cost variation. When indigenous lands were excluded from the selection, the total extent of the priority areas encompasses 14% of the western Amazon and requires an investment 39% higher (total costs ~ US$ 139 million) than the original solution. Under the assumption of no international collaboration, the area needed to achieve targets increases to ~23% of the western Amazon, and it is 49% more expensive (total costs ~US$ 149 million) than a conservation plan developed with international collaboration. Finally, the uniform-cost scenario and the non-international collaboration scenario had the same target achievement as the cost-optimized solution (99.7% of species represented). By contrast, when the indigenous lands were excluded, the target achievement dropped to 99% (20 species unrepresented). See the Supplementary Information for more details on these scenarios.

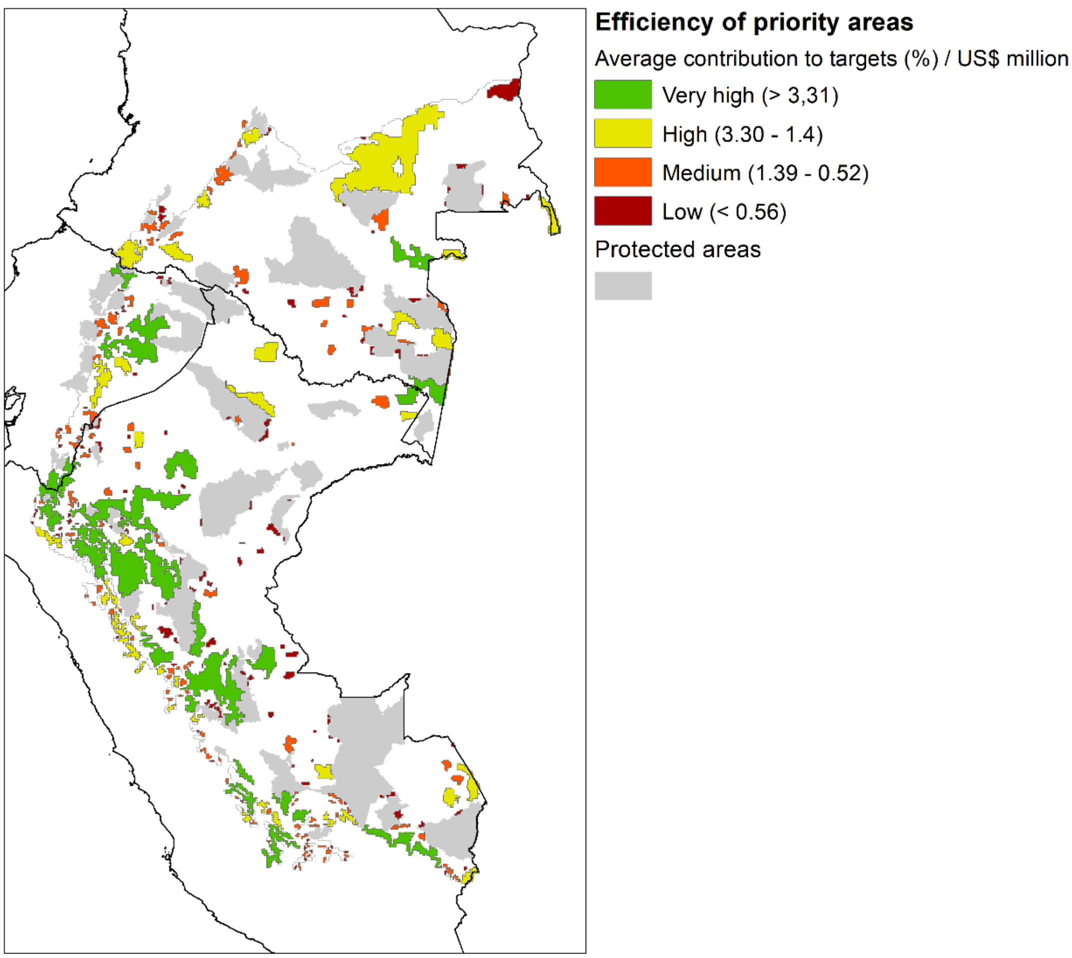

Figure 2.4. Efficiency on investment of the priority areas identified in the western Amazon. For the cost-optimized prioritization, we calculated the efficiency of each priority area as its contribution to achieve species conservation targets divided by its total cost. The 20 most efficient areas (green) are those that contribute more to achieve the targets at lower costs.

4. Discussion

4.1 What is the most cost-effective approach to expand the protected-area system in the western Amazon?

In the western Amazon, protected areas that are large, have low accessibility, and lie within indigenous lands, are associated with lower costs per unit area for an effective management. According to our modeling, annual management cost per-area and size of protected area were related in a log scale, which implies that larger protected areas achieve great savings in management costs, compared to the smallest ones. These savings are the result of the smaller perimeter/surface relationship that large reserves have, which limits the incursion of threats towards their core and hence, reduces the overall costs for surveillance (Cantú-Salazar and Gaston 2010). More importantly, large protected areas in the western Amazon are usually situated in remote zones, where there are fewer socio-economical impediments to create big protected areas. Thus, the lower human pressure around and within these remote protected areas allows for less intensive surveillance, reducing the total management costs, even though inaccessible protected areas demand higher transportation costs for operations. Given this relationship between the size of reserves and their location, size can be understood to explain costs both due to reduced perimeter per area, and also as a good indicator of remoteness and less complex management needs. Moreover, since the lower costs of large protected areas could be associated with the low pressure of the places where they are usually created, the Context Model (and the management cost map) could explain much of the variation in management costs, mainly through the inaccessibility level.

The livelihoods and cultural practices of many indigenous people from the western Amazon depend on natural resources. This relationship has long encouraged indigenous communities to take special measures to prevent forest fires and to control hunters, loggers, miners (Kaimowitz 2015; Zimmerman 2013). In this manner, the traditional protection, collaboration, and governance of local indigenous peoples appears to lower the management costs that need to be assumed by protected-area agencies. Surprisingly, management costs of protected areas in indigenous lands that are accessible were even lower. Indeed, indigenous peoples in the Peruvian Amazon that live close to large population centers and whose lands are titled have had a pronounced effect in maintaining their forests probably because of their closeness to regulatory-agency offices (Blackman et al., 2017).

These non-uniform distributions of land-conservation costs in the western Amazon influenced the selection of cost-effective priority areas. Particularly, the Andean Amazon sub-region concentrates areas that are expensive to manage given their high accessibility and limitations to create large protected areas. Also, soils and climate in this region make it highly suitable for crop expansion, which combined with better accessibility to markets, increases opportunity costs for agriculture. Since the decision-support algorithm we used is directed by the minimization of costs while increasing biodiversity representation, the selection of priority areas towards the lowland Amazon was favored in comparison to an expansion planned with uniform costs (Fig. 2 in the Supplementary Information). Moreover, within the Andean Amazon, the selection of several areas expensive areas was replaced by less accessible lands when cost data were considered. Together, these lowlands and less accessible sites selected would be more affordable and just as important for achieving the species conservation targets as some expensive zones at higher elevations. Nevertheless, the set of priority areas in the cost-optimized solution also included several accessible, productive, and small areas in the Andean Amazon and Foothills that, although expensive, are highly irreplaceable given their large concentration of unprotected and restricted-range species. In fact, 47% of the areas selected in the cost-optimized solution (mostly in the Andean Amazon) were also present in the expansion planned without cost because of their unique species composition that make them crucial to achieve the targets, independently of their costs. Thus, to achieve a cost-effective solution, a combination of sites with different sizes, human pressure levels, and costs needs to be included in the protected-area system expansion of the western Amazon.

4.2 How much resources would the cost-optimized expansion save?

Our cost-optimized expansion of the protected-area system would cost 78% of an expansion only based on minimizing the total extent of land to be protected (i.e., 67sing uniform costs). Thus, as other studies have found (Armsworth 2014), considering cost variation when prioritizing generated more cost-effective alternatives. Moreover, we found that the collaboration between countries generates is vital to cost-effective solutions. When collaborating, less expensive opportunities to protect species arise from the larger territory under planning, and redundancies can be avoided through accounting for the contribution of the regional protected-area system (Kark et al. 2009). As a result, without international collaboration, much more land and costs are needed to achieve the conservation targets.

Our study also highlights that, in economic terms, it is more efficient to expand the protected areas in collaboration with indigenous groups, given comparatively lower management costs and their contribution to close current gaps in biodiversity coverage. However, such an expansion of protected areas in indigenous lands has implications that need to be accounted for. First, partnerships between indigenous peoples and federal or state governments should be encouraged, including informed consent and true participation in design (Schuster et al. 2018). Second, the design and management of any new reserve must consider the particular aspirations for agricultural use, forest resources and other basic needs that each indigenous people may have (Beltrán 2000). On some occasions, conservation strategies in indigenous lands may require efforts towards reconciling and coordinating biodiversity and indigenous needs within the same territory. For example, rates of wildlife harvest have been found to often exceed maximum sustainable levels for several indigenous communities in the Amazon (e.g. Zapata-Ríos et al. 2009). However, any access restrictions should be agreed on with the communities concerned, and appropriate compensation should be given in cases where such restrictions are considered necessary by all parties (Beltrán 2000). These considerations also apply for any priority area that overlap with other traditional forest communities in the western Amazon, which often produce positive results at avoiding deforestation (Porter-Bolland et al. 2012).

The proposed expansion of the protected-area system at US$ 99.9 million annually (for both management and opportunity costs) far exceeds current spending but seems feasible and affordable for governments and the international community. This expansion represents only a small fraction of the annual GDP of the western Amazon countries (0.018%) and of other government spending priorities, such as the military expenditure (2.2% of the GDP; World Bank 2016). Moreover, according to the management cost model that incorporate the size of the protected areas, the management costs could be lowered if the priority areas adjacent to existing protected areas are managed as an extension of the last ones, rather than as new reserves. Nevertheless, protected-area agencies in the western Amazon will still need to reduce funding gaps in current protected areas and increase the annual budget to manage new reserves. In this context of high investment, the efficiency analysis of the priority areas is an additional piece of information to help deciding which areas may be protected first. Specifically, the 20 most efficient areas would cost 20% of the total priority area system, while they contribute to reach the conservation targets for 67% of the insufficiently protected species.

4.3 Final considerations

Besides the economic efficiency of priority areas, other criteria are also relevant to inform conservation planning. For example, several identified priority areas of high cost-efficiency have a relatively high risk of tree cover loss for the next years (Soares-Filho et al. 2006, Fig. 3 in the Supplementary Information). Thus, a strategy with high impact on conservation could be to focus on protecting those efficient and more vulnerable areas. Further studies may also evaluate the trade-offs of prioritizing areas with high vulnerability and immediate threats instead of areas with a high return-on investment in terms of species representation. In any case, ensuring the proper funding for managing these vulnerable areas will be crucial, since the exposition of these areas to large pressures make them more dependent of funding to reduce the threats and ensure their effectiveness (Galindo et al. 2005). Finally, the urgency of protecting any of the identified priority areas also depends on the degree at which the forest landscapes are effectively preserved by other management strategies implemented in the western Amazon, such as private and municipal reserves, or local programs of economic incentives to protect private and communitarian lands (e.g., Socio Bosque or Bosques Protectores in Ecuador). While incorporation of relevant laws and strategies is beyond the scope of this analysis, the cost-effective conservation areas identified may be useful for other approaches to conservation as well.

Protected areas in the western Amazon are experiencing a complex dynamic. Several events of downgrading, downsizing and degazetting of protected areas have been reported**,** mainly as a response of increasing extractive pressure on nature resources (Mascia and Pailler 2011). But at the same time, at least 18 new state reserves have been created in the last 10 years (UNEP-WCMC and IUCN 2019), which reflects the region's commitment to enhancing biodiversity protection. In this context, the long-term success of the protected-area expansion in the western Amazon may depend on careful decisions informed with a combination of biological, social and economic information. To accomplish that goal, our assessment on cost-effective protection offers valuable and new information that, together with other prioritization criteria, different forest management strategies, in situ evaluations and social aspirations, could enhance the feasibility and effectiveness of future expansions of protected areas.

5. Conclusions

Our study demonstrates the challenge in seeking to adequately protect the remarkable biodiversity of the western Amazon since many unprotected species are concentrated in expensive areas. In this context, we provided a map of priority areas for expansion that would reduce annual costs by 25%. Moreover, we highlight the importance of international collaboration when defining conservation targets, as well as of involving indigenous lands in planning to propose cost-effective solutions. Therefore, this contribution may help governments and institutions to improve the financial planning of the region and to achieve scientifically based biodiversity conservation objectives at more affordable costs.

6. Appendices

Supplementary Information. Extended methods and results

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Chapter 3 - Protected areas for positive futures for nature and people in the Andes

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Abstract

We present the most comprehensive review to date of the spatial conservation needs for both biodiversity and Nature's Contributions to People (NCP) in the Andes, a mountain range that hosts immense biodiversity and supports the livelihoods of more than 100 million people, but that is also under rapid environmental change. Although protected areas cover ~21% of the Andes, we found that several critical areas for species and ecosystems conservation remain unprotected. Closing these gaps would require protecting large land areas with high economic costs associated. Also, there is little information on which areas should be protected to ensure the provision of NCP. To overcome these limitations and achieve positive futures for biodiversity and people in the Andes, the upcoming protected area agendas in the region need to: (1) mind about their cost-effectiveness when designing reserves while optimizing the protection of both biodiversity and benefits from nature to people, (2) diversify governance regimes and management objectives of protected areas, with indigenous and local communities at the forefront of the decision making process, and

(3) call for collaboration among Andean countries, accompanied by global financial support. These transformative actions can make protected areas more supportive of nature and people's well-being in mountain landscapes.

Keywords: conservation planning, nature's contribution to people, biodiversity, mountain systems, indigenous peoples, human well-being.

1. Introduction

The Andes are not only the longest continental mountain range of the world [1] (**Fig. 3.1**). This region displays unparalleled terrestrial and freshwater biodiversity levels that are the result of strong environmental gradients and complex topography [2,3]. The long history of interactions between people and nature in the Andes has also produced a distinct and rich agrobiological diversity that has been shaped by an astonishing cultural diversity, currently made up by more than 200 indigenous groups (~ 28 million people) [4–6]. The livelihoods and well-being of these indigenous groups depend on Nature's Contributions to People (NCPs) [5,7] provided by Andean ecosystems, such as food, materials, water, energy, health security, supporting identities, or learning and inspiration [8,9]. In addition to indigenous groups, more than 80 million upland and lowland inhabitants benefit from the NCPs provided by this mountain range. The Andes is clearly a region of exceptional biological and cultural value that, however, is under a large pressure from human actions (**Box 1**).

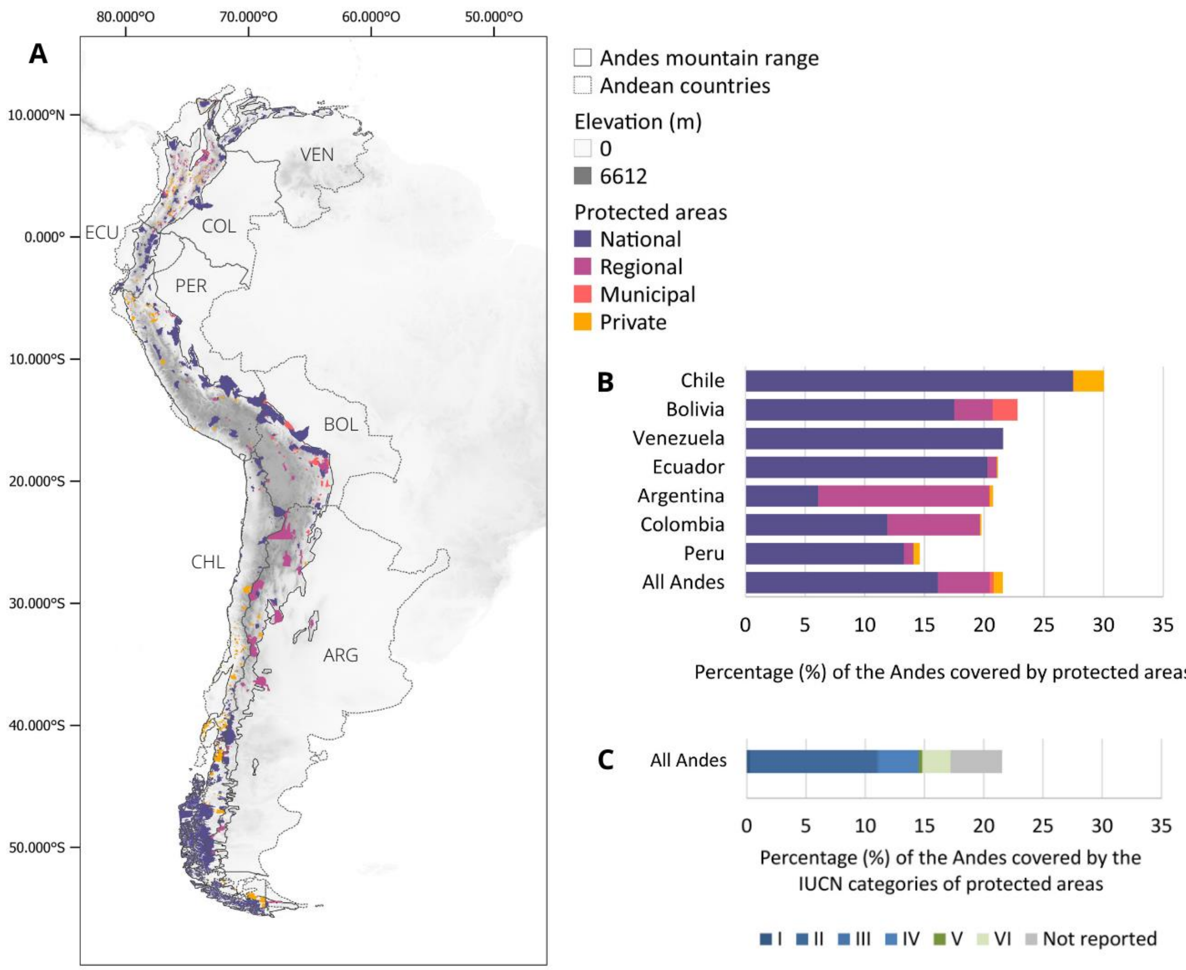


Figure 3.1. Protected areas in the Andes. (A) The Andes limits, according to Körner et al. [18]. Andean countries have built an extensive network of protected areas with diverse governance (National, Regional, Municipal or Private). (B) Percentage of the Andes covered by protected areas according to the governance type and country, and (C) according to IUCN categories of management objectives (blue: categories that imply strict protection of resources, green: categories that allow multiple uses of resources).

Human occupation in Andean ecosystems has shaped the landscapes since pre-hispanic times [10,11]. During the last century, productive activities such as agriculture, mining, and power generation through hydroelectric plants have expanded and gained relevance for the Andean nations' economies [5,6]. These activities, together with growing urbanization and migration, unsustainable exploitation of natural resources, and climate change significantly transform the ecosystems [12], threatening Andean biodiversity [13], and reducing nature's capacity to provide water, food, or support the identities of indigenous and local communities [14–17]. Today the Andes are home to critical biodiversity hotspots (**Box 1**) and some of the poorest and most marginalized local communities on the planet [6]. This crisis highlights the need for sustainable development strategies in the Andes, including greater support for the conservation of biodiversity and provision of NCPs, while respecting the rights of human communities settled in this mountain range.

Worldwide, protected areas have been crucial for sustaining mountain biodiversity and ensuring its sustainable use [42,43]. In the case of the Andes, the region is repeatedly identified as a critical region for expanding the current protected area network and thus, safeguard much of the world's biodiversity (**Box 1**). This expansion is often suggested as a vehicle to increase the current representation of biodiversity under protection, which means to ensure that a sample of each required biodiversity feature is included within protected areas and separated from human pressures [44,45]. Aside from preserving biological richness, there is growing interest in securing mountain NCPs that underpin the quality of life of millions, and protected areas have been recognized as a means by which to help satisfy this need [46–48]. Decision-makers from the Andes thus need guidance on where and how the expansion of protected areas can maximize the protection of biodiversity and multiple NCPs.

Box 1. Evidence of the exceptional value of the Andean mountain range to the conservation of global nature diversity.

Biodiversity

The tropical Andes (which spans from western Venezuela to northern Chile and Argentina) stand out by having a high cross-taxon congruence of total species richness, threatened species, and endemic species [19,20]. Specifically, this region is of global botanical importance based on its high plant endemism [21,22]. It also stands out for its high concentration of restricted-range bird species [23] and endemic amphibians [24].

The Andes also concentrate areas in need of urgent protection since their high irreplaceability regarding plants and vertebrate species (and their ranges of environmental conditions) are poorly covered by protected areas [25–29]. Eleven Global 200 Ecoregions [30] are found in the Andes, those with high priority of protection since they represent the world's most unique, irreplaceable, and biologically diverse regions. The tropical Andes are also a global priority region for protecting places that capture high species richness, evolutionary potential, and ecosystem functions [31].

Agrobiodiversity

The Andes are a center of origin for domesticated plants and traditional agriculture systems [32,33]. The region also harbors a high richness of globally relevant Crop Wild Relatives. Therefore, several priority areas for wild relatives conservation are located in the Andes (e.g., potato, common bean, quinoa, squash) [34,35].

Biocultural diversity

A great diversity of species and native languages (>100) co-occur in the Andes [36]. The region also overlaps with critical Global Language Hotspots, places with extremely high diversity, very little documentation, and immediate threats of endangerment [37].

Biodiversity and habitat loss

The Andes harbor two global Biodiversity Hotspots (Tropical Andes and Chilean Winter Rainfall-Valdivian Forests). These are regions with an exceptional concentration of endemic species undergoing accelerated loss of habitat and where in-situ conservation actions are required [24,38]. Tropical Andes are also considered a Hyper Hotspot, a priority hotspot for conservation investment in light of its exceptional totals of endemic species [13].

Biodiversity and climate change

The Andes are an especially vulnerable biodiversity hotspot given that >2,000 endemic plant species could become extinct because of climate change [39]. The ecosystems of the region are also vulnerable to vegetation shifts due to climate change [40]. The Andes also concentrate a large coverage of high priority areas that would minimize the extinction risk of tropical species in future climates [41].

Drawing concrete recommendations for protected-area location and management actions in the Andes is, however, extremely challenging because of the intricate patterns of Andean biodiversity [49], resource conflicts among different actors, accelerated human population growth [17,50] and political borders [51]. With the aim to find solutions to this multidimensional choice problem [52], several studies on spatial conservation prioritization have used transparent objectives and methods to identify important sites in the Andes that should be protected to promote the persistence of biodiversity and other natural values [45,53]. These studies have also evaluated different conservation targets, planning approaches, spatial scales, and socioeconomic constraints, delivering diverse proposals for conservation. Crucially, these results must be systematized and integrated to identify consensus areas for protection, the most adequate governance and management actions, as well as persisting knowledge gaps, similarly to what has been done in other regions (e.g., Amazonian savannahs [54] or Mediterranean Basin [55]). Such analysis will set a sound basis for a coordinated and effective protected area strategy in the Andes.

Here, we provide the first comprehensive review of the spatial conservation needs for biodiversity and nature's contributions to people in the Andes. We started by examining studies published in the last decade on spatial conservation planning for Andean systems in order to identify the current and most critical representation gaps in the region's protected areas in terms of biodiversity and NCPs. We also highlight the main challenges and opportunities for efficiently expanding and managing the protected area network, thus narrowing the conservation gaps of the Andes, and ensuring benefits to local communities. With this contribution, we aim to help guide future agendas on protected areas to achieve better outcomes for nature and the quality of life of Andean people. As major providers of global goods and reservoirs of unique biodiversity, mountain ecosystems of the world need to be the target of specific assessments, as the one presented here, to address with sensitivity and specificity their needs when planning, designing, and managing protected areas [42].

2. Conservation gaps in the Andean protected areas

Currently, the Andes display an impressive network of protected areas. This network covers 21.6% of the region (**Fig. 3.1,** See Methods), exceeding the proportion of global terrestrial surface area in protected areas (15.1%) [56] and the Aichi Biodiversity Target of protecting 17% of important terrestrial ecosystems by 2020 [57]. Most of the land protected is under strict protection of resources (IUCN categories from I to IV; 14.5% of the Andes) and administered by national governments (16.1% of the Andes) (**Fig. 3.1**). This large network of protected areas harbors key sites for Andean biodiversity conservation. For instance, some regions with high species richness have a large concentration of protected areas, such as the humid Yungas systems in Peru, Bolivia and Argentina or the pluvial mountain forests in the tropical Andes [58,59]. In the eastern Andean slopes of Peru and Bolivia, protected areas partially cover four-fifth of the endemic bird species [60]. Overall, the largest richness of plant species captured throughout the protected sites in South America is located in the Andes, [61]. Similarly, the representation of species diversity of vertebrates in protected areas is higher for the Andes than for other regions of tropical countries, such as coastal forests in Ecuador [62,63], or the Llanos region in Venezuela [64].

Despite exceeding the target of protecting 17% of its total area, severe deficiencies in the Andean protected-area system remain. Historically, many protected areas in the Andes have been located towards upper elevations and low productive lands that usually have low conservation costs and limited biological value [65,66]. These biases have contributed to creating recurring biological representation gaps in the protected area system [67]. Here, we reviewed 43 studies published between 2009 and 2019 that report gaps and priorities areas for conservation in the Andes (see Methods). All Andean countries are represented in this set of studies, but the vast majority of research was conducted in the tropical Andes (Colombia, Ecuador, Peru, and Bolivia; **Fig. 3.2**). Most of the reviewed studies also focused on assessing the representation of the diversity of vertebrate (23) and plant species (20) in protected areas (**Fig. 3.2,** Supplementary Information), concluding that more land conservation efforts should be directed towards:

- Subandean forests (500 m – 2000 m) in Colombia, Ecuador, and Peru, which have a striking taxonomic diversity of plants and vertebrates per unit area that is also under a high fragmentation and threats from migratory agriculture, illegal crops, and human settlements. These forests have a lower species representation in reserves than the more widely protected highlands ecosystems or lowland Amazonian forests [68–72].

- Species and ecosystems in the Western Andean slopes in Colombia, Ecuador and Peru, which have lower levels of representation in protected areas compared to the Eastern slopes and the transition to the Amazon forests [12,62,69,73–76]. This insufficient protection is especially alarming for the western slopes of Colombia and Ecuador given their high diversity of endemic plants and birds and rates of environmental degradation.

- Important endemism centres in the eastern slopes from Peru to Bolivia, which are only partially covered by protected areas [12,60,77,78].

- Tropical dry and xeric forests and shrub remnants (e.g., in the inter-Andean valleys and foothills of the mountain range), often less represented than tropical moist forests [58,73,77,79]. Many tropical dry ecosystems have been preferred zones for agriculture and human settlement, reducing the original vegetation and threatening a large richness of endemic plants and birds.
- Montane forests and inter valleys from the South of Ecuador to the North of Peru, which concentrate significant gaps in plants and vertebrate protection [62,63,76,80–82].
- Ecosystems and species located in the biologically rich and highly populated Central Chile, which are extremely under-represented compared to the high Andes and the Southern Andes in the country [65,83,84]. Also, Chilean arid ecosystems are poorly covered by protected areas [65,85].
- Endemic or nearly endemic and threatened species in the Andes, which are among the worst covered groups [59,63,78,86–90]. Species groups with more restricted ranges, such as amphibians and reptiles, often have lower representation than birds [78,87,91].
- Important sites required to cover shifting species distributions and support species migration under climate change, which often lack protection and concentrate in the high Andes (>2000 m) and eastern foothills of tropical countries [69,81,86,88,92,93].

Preserving the habitats of these critical species and ecosystems can also underpin NCPs in the Andes [7]. Still, priority areas for protecting biodiversity and sustaining diverse NCPs do not always overlap [94,95]. Thus, it is fundamental to assess the spatial conservation needs for NCPs, especially in the Andes, where millions of people directly depend on natural resources. However, only seven of the 43 studies for conservation planning in the Andes explicitly assessed elements of nature that contribute to people's quality of life **(Fig. 3.2)**. For example, according to three of these studies, most of the assessed crop wild relatives (CWR) of potato in Bolivia, pseudocereals in Argentina, and chile peppers in the Andes lack protection [96–98]. Expanding such evaluations for diverse CWR across the Andes is crucial because CWR are associated with various categories of NCPs, such as Medicinal, biochemical and genetic resources, Food and Feed, Supporting Identities, and Learning and Inspiration of local communities. Another example of NCP assessment needs comes from Chile, where protected areas and priority sites for biodiversity poorly represent carbon storage, agricultural production and plant productivity (related NCPs: Regulation of Climate, Food and Feed) [66]. Also, in Central Chile, protected areas play an essential role in preserving forests at high altitudes, but their biased locations have limited the access of people with lower incomes to the cultural services that these protected areas provide (related NCPs: Learning and Inspiration, Physical and Physiological Experiences) [99]. In the Central Andes of Colombia, protected areas have a low overlap with priority vertebrate species and natural areas with multiple benefits to people (related NCPs: Learning and Inspiration, Physical and Physiological Experiences, Regulation of Water Quantity) [100]. Finally, in the Ecuadorian Andes, agriculture is a crucial resource for local economies. A study shows that priority areas for plant diversity conservation and agriculture production overlap (NCP: Food and Feed) in the Northern and Southern dry forests, suggesting that land-sharing approaches to sustain and manage both features are needed [82].

We did not detect studies on conservation planning directly assessing other components of NCPs relevant in the Andes, such as traditional landraces of crops, culturally important species, or pollination provided by wild bees [101]. Consequently, critical areas for sustaining and providing NCPs in the Andes may be overlooked. The lack of studies is probably due to the fact that spatial conservation planning has been traditionally developed from a more naturalistic ecological perspective, focusing heavily on ecosystems component and species rather than in social and biocultural diversity, but also because the spatial modelling of NCPs and ecosystems services and their integration in prioritizations have remained a major challenge [95].

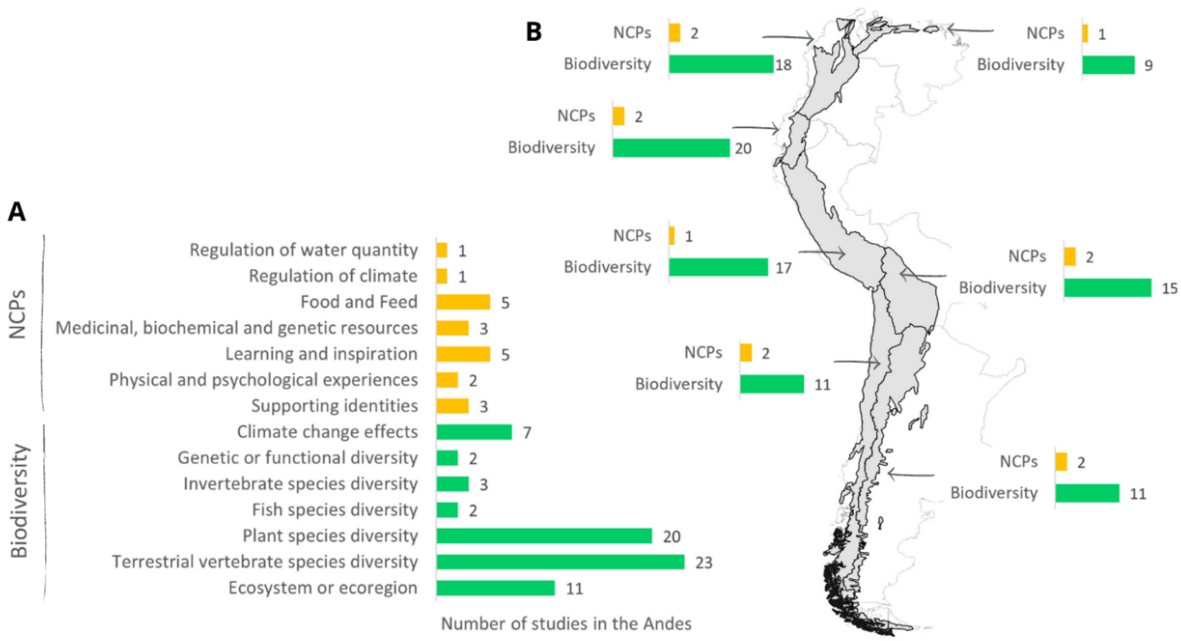


Figure 3.2. Studies on spatial conservation planning in the Andes that focus on biodiversity and/or NCPs features. (A) Number of studies (from 2009 to 2019) that directly analyze different features of biodiversity (n = 43) or one of the 18 general categories of NCPs (n = 7). (B) Number of the studies focusing on biodiversity or NCPs by country. One study can assess more than one feature and country.

3. Challenges of expanding protected areas in the Andes

Large land area extent needed

At the global level, the exact percentage of land needed to safeguard habitats and species is unknown and a matter of discussion [44], as available estimates also depend on target groups and target levels of protection. Still, most studies indicate that the Andes needs a considerably large expansion of protected areas, such as doubling the current coverage [69,71]. This requirement is partially due to its high number of endemic and narrow-ranged species demanding coverage of their particular geographic extent [41,71,102](**Fig. 3.3**). By contrast, the broad and congruent geographic ranges of the species in Amazonian lowlands allow a smaller number of well-chosen conservation areas to cover many species [71,103]. Because a large extent of the Andes is already protected, such an expansion via traditional protected areas (centralized governance with strict use of resources) would have considerable impacts on different actors such as indigenous communities, who are important stewards of the environment and have ancestral rights to land management [107]. However, the reviewed studies on conservation planning in the Andes poorly addressed or mentioned social values, equity and environmental justice concerns, acceptability, and feasibility of the proposed large expansions of conservation areas [108].

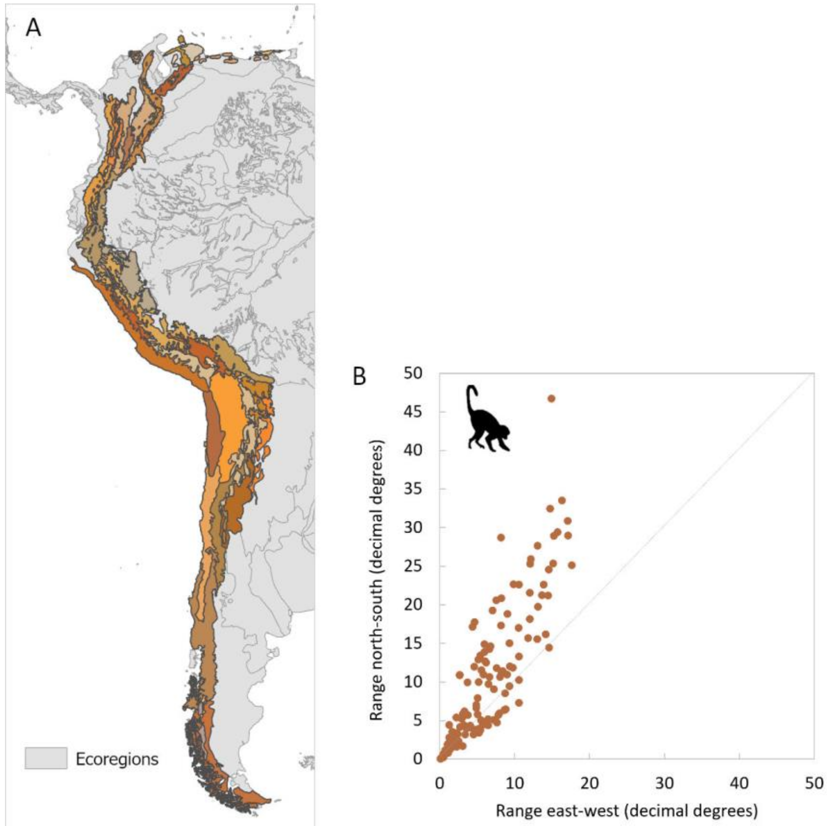


Figure 3.3. Configuration of ecoregions and species geographic range in the Andes. (A) Ecoregions in the Andes [104] (shown in an orange scale of colors), which are oriented predominantly north-south. (B) Maximum north-south and east-west dimensions of the geographic ranges of 131 mammal endemic species in the Andes [105]. Geographic ranges of equal dimensions would fall on the diagonal line [106]. Andean species tend to exhibit a great altitudinal replacement and narrow ranges oriented north-south (above the diagonal), since major climatic belts and biome types that influence species distribution also run north-south. Given this species configuration, improving biodiversity protection in the Andes requires establishing multiple conservation areas of small sizes distributed along the altitudinal gradient.

Costly protected area expansion

Developing a much more representative network of protected areas is expected to be expensive in the Andes, compared to the conservation costs of Amazonian reserves. In general, a better representation of Andean biodiversity requires protecting many small areas (e.g., less than 10,000 ha) with ecosystem fragments remnants that contain important populations of endemic species [78,109], as well as protecting and restoring degraded habitats where generalist species persist [2]. Many of these areas, especially in the northern Andes, are highly accessible, close to urban centres, or in the transition toward ecosystems where people graze their cattle [59,87]. This means that priority areas are exposed to considerable human pressures, and they would require large budgets for restoring, managing and preserving them [61,71]. Moreover, due to economies of scale, protecting these small patches leads to higher management costs than the same total extent arranged in large-sized areas [61,71]. These reasons explain why estimated average annual management costs of priority areas in the Amazon regions of Colombia, Ecuador and Peru, with lower human intervention level and larger sizes, are four-time lower than those in the Andean slopes [71,110]. The Andes is also characterized by a good climate and appropriate soils for cropping, and better accessibility to markets than many regions in the Amazon, which leads to high opportunity costs for agriculture and land acquisition costs for establishing new reserves [60,71,75].

National budgets available to manage protected areas are usually very deficient in the Andean countries [111], making the scenario of a large expansion of protected areas with strict protection and governmental administration even more difficult. Adequate funding is crucial because it fosters the ability of protected areas to safeguard biodiversity and prevents protected-area downsizing and degazettement events [112,113]. Thus, although it is regrettable that countries have not been able to protect all that is needed in the Andes, it is also alarming that current or future protected areas lack proper funding to secure their long-term functioning and persistence.

The urgency for a rapid response

Most of the priority areas in the Andes urgently need direct conservation efforts. These places represent the last opportunities to protect natural ecosystems where much of their original extent has been converted. Moreover, these priority areas still face massive conversion and little protection. For example, priority areas for dry valleys of the Tropical Andes [69] or invertebrates in the Southern Andean Yungas in Argentina [114] include the last remnants of these regions and face severe pressure from intensive agriculture. Overall, opportunities for establishing site-based conservation in wildland areas are much less abundant in the Andes (60% of the non-protected surface) than in the Amazon forests (86% of the non-protected surface) [115]. Climate change effects on species range also demand rapid response to enhance the in-situ protection of thousands of species that will undergo reductions of their climatic niche or are expected to migrate towards unprotected sites, particularly in the Andes [86,88]. Therefore, there is a pressing need for planning and implementing spatial conservation strategies. Still, limitations in resources and competing land uses pose a serious challenge to practitioners and governments for timely implementation.

4. Opportunities for expanding the protected areas in the Andes

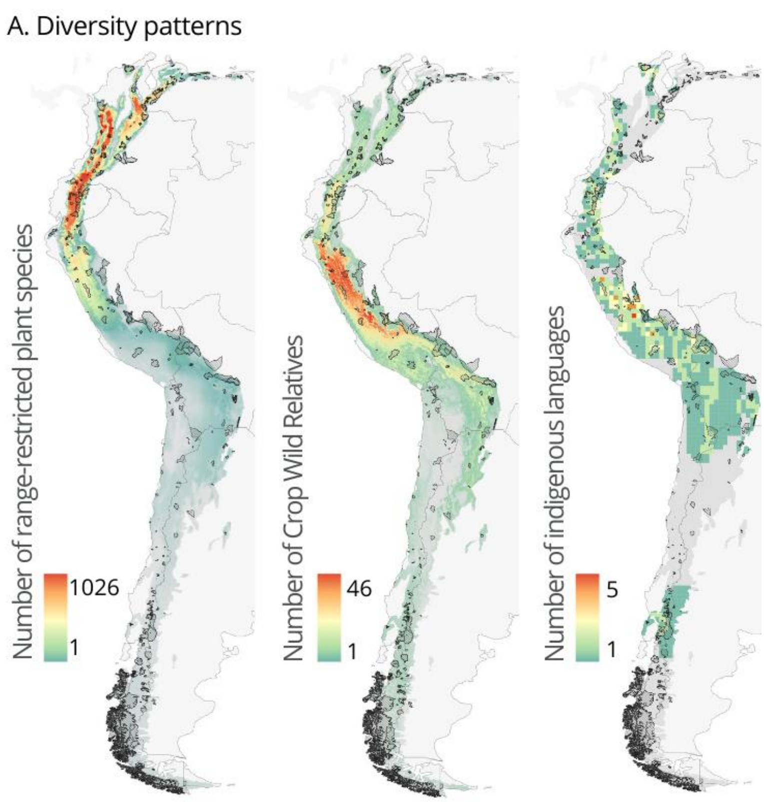
A conservation plan for expanding Andean protected areas that is ecologically sound, feasible and socially equitable [43] needs to address the socioeconomic challenges mentioned above. We argue that opportunities still exist to expand conservation areas and help change current negative trends for biodiversity and NCPs in the Andes. To achieve this goal, we identified three main tasks for the region’s conservation planning: (1) mind about the cost-effectiveness of protected areas securing biodiversity and NCP preservation, (2) diversifying governance regimes and management objectives of protected areas, and (3) calling for regional coordination for supporting the protection of Andean biodiversity accompanied by global financial support. We desperately need a “biodiversity diplomacy” that would help coordinate efforts across political boundaries in the region, improving the cost-effectiveness of conservation.

Cost-effective expansion of protected areas for people and nature

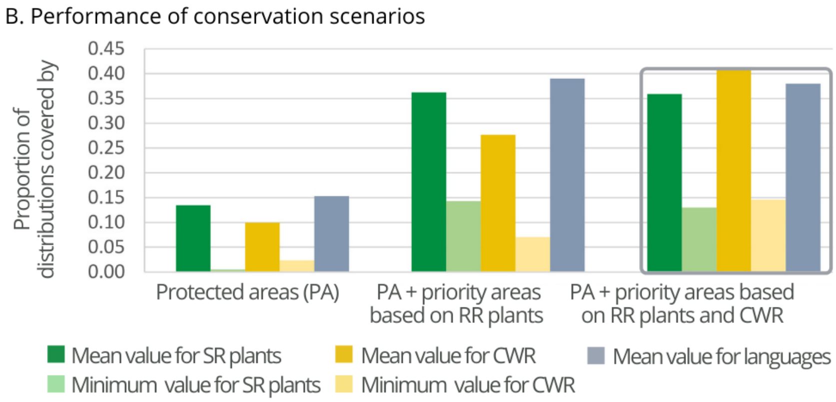
In the Andes, competing land uses, the conservation of critical species and ecosystems gaps, and multiple nature values can converge in the same territory. Thus, future expansion of conservation areas in the Andes should aim at reconciling views and needs in order to gain feasibility and long-term stability. For this, it could focus on places that satisfy diverse conservation needs at the least possible costs and bring direct benefits to Andean people [116]. Thus, when possible, reserve planning could take the supply, demand and flow of relevant NCPs into account [48,94]. In this way, local people’s social, economic, and cultural needs are attended together with the protection of biodiversity. For example, the protection of known range-restricted plant species (which is a traditional conservation object in the region) might confer a limited benefit to the protection of CWR of importance in the Andes (such as wild potato, beans, or cereals) (**Box 2**). Instead, integrating both groups into conservation planning would increase CWR representation in reserves with a minimal decrease of benefits for range-restricted species. Thus, we ensure that preserving these areas from environmental degradation would simultaneously contribute to mitigating biodiversity loss, sustain people's nutrition and the traditional agricultural practices and identities of local communities.

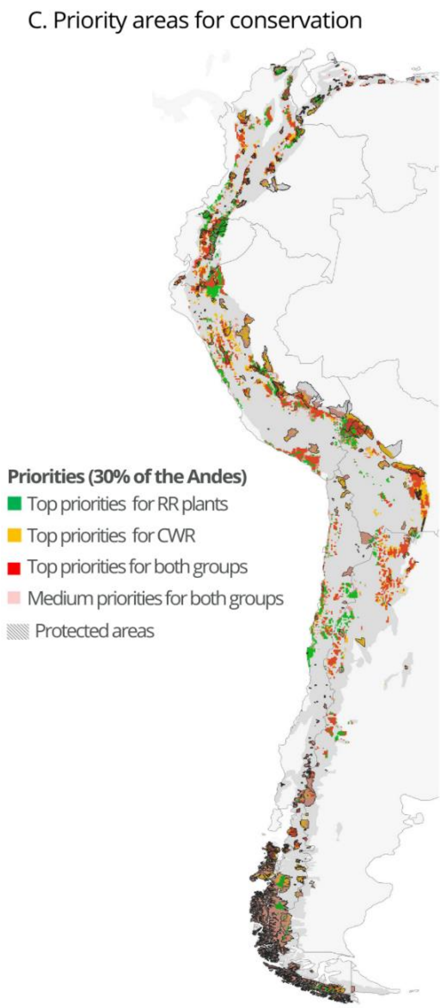
Box 2. Balancing conservation of biodiversity and NCPs in the Andes

Conservation planning in the Andes overwhelmingly focuses on prioritizing areas that sustain intrinsic values of biodiversity. For example, many studies highlight important areas for protecting range-restricted plant species (RR) [77,78], which are highly diverse in the northern Tropical Andes **(A).** However, reserve planning could also seek to maximize the protection and access of other Nature's Contribution to People (NCPs). For example, Crop Wild Relatives (CWR), which are the wild plant cousins of cultivated crops, harbour a vast resource of genetic diversity that is critical for underpinning the long-term resilience of the crops [119]. CWR reach high levels of richness in the Central Andes **(A),** where indigenous and local communities have traditionally planted CWR alongside crops to promote the natural crossing of beneficial traits [120,121]. Currently, climate change and land-use change are threatening with extinction of landraces and CWR of importance for the Andes and the world [122–124]. However, CWR have rarely been a target of biological inventories or explicitly included in spatial prioritizations [35,122].



Here, we balanced protection for both RR plants and CWR when expanding the protected coverage from 16% to 30% [41,125] of the Andes **(B)**. This exercise used the Zonation algorithm to find priority areas for conservation with low levels of human intervention and that complement current protected areas. A total of 1726 species of SR plants [126] and 118 CWR [127] were simultaneously included as target groups, allowing us to identify areas that retain a higher probability of species occurrence from both groups. Identified priority areas could increase the average protection of species distributions from the current 10% coverage to 41% for CWR, and from 14% to 36% for RR plants **(B).** Although a conservation-planning scenario based only on RR plants achieves similar benefits for this group, this scenario confers a much lower coverage for CWR (average protection of species distributions: 26%) **(B)**. This exercise shows how directly incorporating NCP indicators can help enrich the value and benefit that zoned protected areas provide whilst optimizing the efficient investment of resources.





The identified priority areas and current protected areas could have different management priorities based on the diversity levels of RR plants and CWR they harbour. For example, several priority areas in Bolivia retain the most significant areas of many CWR (yellow areas). Actions to manage and monitor the genetic diversity in natural populations of CWR are essential in these areas [128]. Efforts to assist the protection of RR plants would be especially relevant in the selected areas in the North of Ecuador and Central Chile (green areas), where the diversity of RR plants is high. However, most priority and protected areas have equally high conservation values for both RR plants and CWR (red areas), which demands simultaneous actions for managing these groups [128].

Protected and priority areas in the Andes also embrace landscapes that sustain a large part of the region's cultural diversity **(A)**. This network overlaps with the geographic ranges [129] of 91 of the 100 indigenous linguistic groups in the Andes, with an average of 38% of their ranges retained **(B)**. Thus, to guarantee these cultures' right to thrive, a land conservation strategy for the Andes must include indigenous and local communities' aspirations and perspectives and ensure their access and management of biodiversity and NCPs, including CWR of interest.

(See Supplementary Information for the methodological details of this prioritization).

In addition, future studies on conservation planning in the Andes should integrate indigenous and local communities' conceptions and values of nature and biodiversity at the core of plans and decisions [116,117]. It is also important to keep in mind that integrative conservation planning combining NCPs, social values, and biodiversity must assess if protection goals and management strategies for the different targets are compatible [118], or if integrating multiple NCPs could mask key biodiversity areas for protection [48]. Other significant research gaps in cost-effective planning for the Andes that should be addressed are (**Fig. 3.2**): (1) assessment of other taxa with poor data available, such as invertebrates, fungi, fish species, and the effects of climate change in these groups, and (2) complementary metrics of biodiversity, such as functional and genetic diversity.

Diversifying models of protected areas

Most of the current protected areas in the Andes are centralized and conceived for restricting the use of natural resources (**Fig. 3.1**). To overcome the socioeconomic challenges mentioned above, the Andes need to encourage protected areas under more diverse governance regimes and management objectives that respond to specific conservation needs and socioeconomic contexts. For example, given the fine-scale of protection required to preserve endemisms and their closeness to human centres and agricultural lands, many of the small-sized priority areas for conservation could be taken over by local communities, private owners and foundations [78,109]. These areas could use innovative ways to combine revenues for their managers and positive outcomes for biodiversity [60,77,109]. In other cases, highly vulnerable biodiversity might need strict protection, such as paramos in the Northern Andes [130]. New protected areas could also integrate the management and use of diverse NCPs while contributing to biodiversity protection. In Peru, the Potato Park is an example of the potential that new protected-area models have in contributing to the maintenance of local livelihoods and biocultural diversity in traditional agricultural landscapes [50]. Local communities lead this conservation model, and it focuses on the conservation and sustainable use of plant genetic resources through traditional Andean approaches to agrobiodiversity and landscape protection [50]. Overall, we emphasize the importance of supporting diverse conservation-based area models where people are acknowledged as an integral element of protected areas, particularly indigenous people, and traditional economic practices.

The expansion of in-situ protection in the Andes could also rely on other effective area-based conservation measures (OECM), that deliver effective protection despite not listing biodiversity conservation as their primary objective [131]. For example, in the Mediterranean region of Chile Central, where less than 2% of extent is formally protected, there are sustainability programs at vineyards that seek to promote practices compatible with biodiversity protection and enhance the cultural benefits of nature [132]. Integrating OECM into a regional strategy has the potential to make area-based conservation more socially equitable, as they are managed to deliver positive outcomes for biodiversity conservation and landowners, especially where natural resources are major components to sustain livelihoods [133].

Regional collaboration and international support

The Andes often lack a coordinated and regional vision for the conservation of its biodiversity and NCPs. For instance, most reviewed studies on systematic conservation planning in the Andes (26 out of 43) were carried within national boundaries. However, the geographical range of many species and ecosystems, watersheds, etc., are not restricted by national borders (**Fig. 3.4**). Therefore, national analyses can miss opportunities to identify less expensive opportunities to protect species arise from the larger territory under planning [71,87]. Moreover, within the tropical Andes, high number of mammal and bird species are projected to move across political borders under climate change [51]. These species movements demand regional planning and proactive cooperation among countries to identify macro corridors for the dispersion and flow of species with high mobility [134]. Thus, such national-scale analyses, although relevant, provides little basis for a regional protected-area strategy. Similarly, spaces for regional collaboration and cooperation are scarce in the Andes (such as Alianza para las Montañas). By contrast, the protection of Amazon biome counts on several initiatives where scientists, local communities, NGOs, and governments across countries collaborate (e.g., Science Panel for the Amazon, Red Amazónica de Información Socioambiental Georreferenciada RAISG, the Amazon Conservation Vision, Amazon Sustainable Landscapes Program, among others). Creating these opportunities for the Andes is crucial to enhance its biodiversity and NCP conservation. In these spaces, local and regional actors can share capacities and experiences of successful outcomes in protected areas and mobilize different perspectives to change the current direction towards a positive future for nature and people in the Andes.

The economic investment required for the rapid response to enlarge conservation needed by the Andes is considerably high. At the same time, public financing for the management of Andean protected areas has historically been insufficient [111]. It is clear that Andean nations could increase their investment in protecting nature [135]. However, international cooperation for financing a more extensive protected-area network is not only well justified but urgently needed. The vulnerability and importance of the contribution of Andean ecosystems to other ecosystems in the region and the Amazon in particular, makes it an important component of world's well-being, a worth of supporting their conservation as 'global common good' [42,136] under the mechanism proposed by the new Global Agreement for Biodiversity [137]. In this agreement, besides establishing a global target for land protection, wealthier countries and private industries could commit to assist conservation in developing and biodiverse regions, such as the Andes, by setting specific financial support.

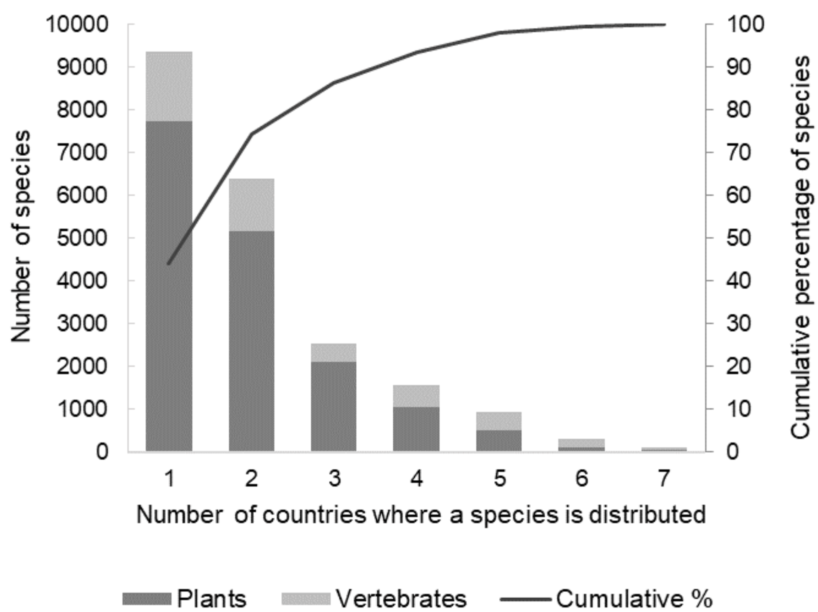


Figure 3.4. Number of species shared among Andean countries. A large proportion (~ 55%) of Andean plant and vertebrate species is distributed in more than one country of the region, which calls for better cross-border collaboration when planning for species conservation. These estimates were based on the geographic ranges or occurrence records available for 16,708 plant species [126] and for 4,516 species of birds, mammals, reptiles, and amphibians [105].

5. Conclusions

This study presents the most comprehensive review to date of the spatial conservation needs of the Andes, a region where nature and quality of life of people are strongly interconnected. Our review makes it evident that critical areas for species and ecosystems remain unprotected, but more importantly, that nature contributions to people are not usually taken into account in these assessments. We consider that, when possible, an expansion of area-based conservation in the Andes should integrate the protection and access of critical NCPs associated with human welfare while maintaining and restoring habitats that support biodiversity. Moreover, indigenous peoples and local communities need to be at the centre of the discussions, decisions, and governance to respect their land rights, recognizing the value of traditional knowledge to manage biodiversity and NCPs. These suggestions involve coordinated cross-boundary efforts, moving towards multifunctional protected areas, and redefining their current purpose in the Andes, which is mainly the protection of biodiversity via restricted resource use. Implementing these and other transformative changes also require tracking their positive and negative impact on biodiversity, indigenous and local communities, and land managers. Although protected areas are only some of the possible interventions in nature conservation, they have an unequalled potential to make human-dominated landscapes more supportive of biodiversity and people's well-being from mountain systems.

6. Methods

Coverage of protected areas in the Andes

We used the definition and spatial information of Körner et al. 2016 [18] to set the limits of the Andes mountain range (**Fig. 3.1**). Based on these limits, the Andes cover 2.9 million km^2 distributed across Venezuela (4% of the Andes), Colombia (12%), Ecuador (4%), Peru (24%), Bolivia (15%), Chile (22%) and Argentina (19%). The Andes occupy a large extent of these countries, ranging from 11% of Venezuela to 84% of Chile's terrestrial extent. To calculate the surface of the Andes that is covered by protected areas, we gathered the most updated information (by 2020) of the protected-area boundaries from national agencies or the Minister of Environment offices of each country. We included reserves that are part of national, regional, municipal, and private conservation networks when available and applicable. We could not access comprehensive spatial data for private reserves in Venezuela and Bolivia, which consequently were not included in the analyses.

Literature review

We conducted a state-of-the-art review [138] of articles on spatial prioritizations and protected area evaluations that have assessed Andean regions. This includes regional, national, or subnational-scale studies across the seven countries that comprise the Andes: Venezuela, Colombia, Ecuador, Peru, Bolivia, Chile, and Argentina. The first selection of articles, written in English or Spanish, used the search engine Web of Science and the following combinations of words in the topic (TS): (Andes OR Andean) AND ("protected areas" OR "priority areas" OR prioritization OR "gap analysis" OR "conservation planning” OR “systematic conservation planning” OR “reserve selection”). We also conducted the search in Google using the same combination of keywords to identify theses, governmental or NGOs reports not published in scientific databases (grey literature). The search was applied to published works between 2009 and 2019 to obtain information relevant to the decision-makers, as these studies are based on more updated protected area coverage. Since the Andes cover a high proportion of the continental extent of Chile (~84%), we included all studies on conservation planning carried out national level.

We explored abstracts and full texts to filter articles including insights of spatial conservation priorities for the Andes explicitly, with a focus on (1) reviewing existing protected areas based on their coverage of different conservation objects and/or (2) identifying additional areas to protect according to different objectives and approaches, such as closing representation gaps, retaining nature’s benefits to people, preserving wilderness, endemism centers, etc. Filtering by these criteria resulted in 43 articles, reports, or documents for reviewing (Supplementary Information). We extracted the following information from each item: (1) year of publication and type of study (published article in a peer-reviewed journal or grey literature), (2) study location and country, (3) whether the study is intended at protecting biodiversity or/and nature features and services directly related to people’s quality life (i.e. NCPs), (4) specific target feature assessed (biodiversity: species, ecosystems/ecoregions, functional diversity, genetic diversity, morphological diversity; or NCP categories), (5) whether the study considered the effect of climate change, (6) representation of features in protected areas, (7) characteristics of the identified priority areas (if including a prioritization), and (9) highlighted challenges and opportunities to protect these areas. To classify the NCPs, we used the 18 categories of the

IPBES generalizing perspective [7]. Finally, we stress that evaluating protection effectiveness of biodiversity features and NCPs (which can affect their representation level) is beyond the scope of this paper.

7. Supplementary Information

1. List of reviewed studies.
2. Detailed methods for "Box 2 - Balancing conservation of biodiversity and NCPs in the Andes".
3. List of species used in "Box 2 - Balancing conservation of biodiversity and NCPs in the Andes".

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Chapter 4 - Large-scale deforestation poses a threat to the world's cultural diversity

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Abstract

Effects of deforestation on biodiversity and climate change are widely recognized. Here we show that this century's deforestation also poses a significant threat to the world's cultural diversity. We gathered emerging evidence that forest loss often precipitates processes of cultural erosion in forest dwellers through three pathways: (1) restricting people's interactions with their biocultural landscape, (2) increasing their exposure to dominant cultural groups, and (3) reducing their cultural group size. We also examined changes in forest cover between 2000 and 2016 in the distributions of all ethnolinguistic groups linked to tropical forests. Our results raise concern about the prospects for cultural survival of ~1,400 ethnolinguistic groups (20% of the world's diversity), whose territories have been exposed to significant rates of forest loss. To halt cultural loss from forests, world governments must substantially advance in recognizing indigenous peoples' land tenure

rights, which are critical for strengthening indigenous' organizations, reducing deforestation, and mitigating climate change. Furthermore, national and international environmental agendas should promote actions for forest protection that respect cultural identities and customary use of biodiversity.

Keywords: indigenous and local communities, cultural evolution, cultural extinction, land rights, protected areas, forest loss and degradation.

1. Introduction

Renowned for their biological diversity (Giam 2017), forest landscapes are also regions of high cultural diversity. Thousands of indigenous peoples and local communities (IPLCs) live in forests (Garnett et al. 2018; Watson et al. 2018; Fa et al. 2020). Although IPLCs constitute a numerical minority, they are the vessel of unique languages, rituals, spirituality, and traditional knowledge that are inextricably linked to forests and that together embody a large part of the world's cultural diversity (**Fig. 4.1a**) (Sutherland 2003; Loh & Harmon 2014; IPBES 2018a; Begotti & Peres 2020). However, this diversity is currently vanishing at an alarming rate, as reflected in the extinction and endangerment of thousands of indigenous languages (Austin & Sallabank 2011; Anderson 2011; Amano et al. 2014; Loh & Harmon 2014) and the marked erosion of local ecological knowledge from forests (Gómez-Baggethun et al. 2013; Aswani et al. 2018). This cultural erosion from forests is part of the global crisis of cultural extinction, in which it is estimated that around 90% of the world's 6,000 languages will disappear by the end of the 21st century (UNESCO Ad Hoc Expert Group on Endangered Languages 2003).

With this cultural loss, the world loses valuable ecological knowledge that is instrumental in promoting the sustainable management of biodiversity, as well as notions and philosophies that are crucial to make humanity more adaptive and resilient in the face of changes (Dunn 2008; Robbins 2015; Lyver et al. 2019). For many forest IPLCs, cultural erosion is also associated with losing an individual sense of belonging, purpose, social support, and spirituality, among other experiences, that are positively related to good mental health and well-being (Shepherd et al. 2017; Díaz et al. 2018; Ferguson & Weaselboy 2020). In this context, the relevance of having a culturally diverse humanity calls for a better understanding of the factors that endanger the cultures that inhabit forests.

Large-scale deforestation for commodity production, infrastructure projects and energy generation is burgeoning and intensifying in tropical regions where countless of IPLCs survive as distinct people (**Fig. 4.1b**) (Geist & Lambin 2002; Lewis et al. 2015; Allan et al. 2020; Fa et al. 2020; FAO & FILAC 2021). Even though IPLCs have been shown to be effective guardians of tropical forests, many of them are reporting a rapid forest loss and depletion of biodiversity in their territories that is resulting in an undesired and pronounced erosion of their cultural systems and well-being (IPBES 2018b, **Fig. 4.2**). Despite these claims, there is little research on the cumulative impact of forest loss on global, regional, or local cultural diversity. By contrast, most of the attention has focused on studying the role of socioeconomic factors, such as global economic integration, modernization, or economic growth, as underlying drivers of recent losses in cultural diversity (Nettle & Romaine 2000; Mufwene 2004; Austin & Sallabank 2011; Amano et al. 2014).

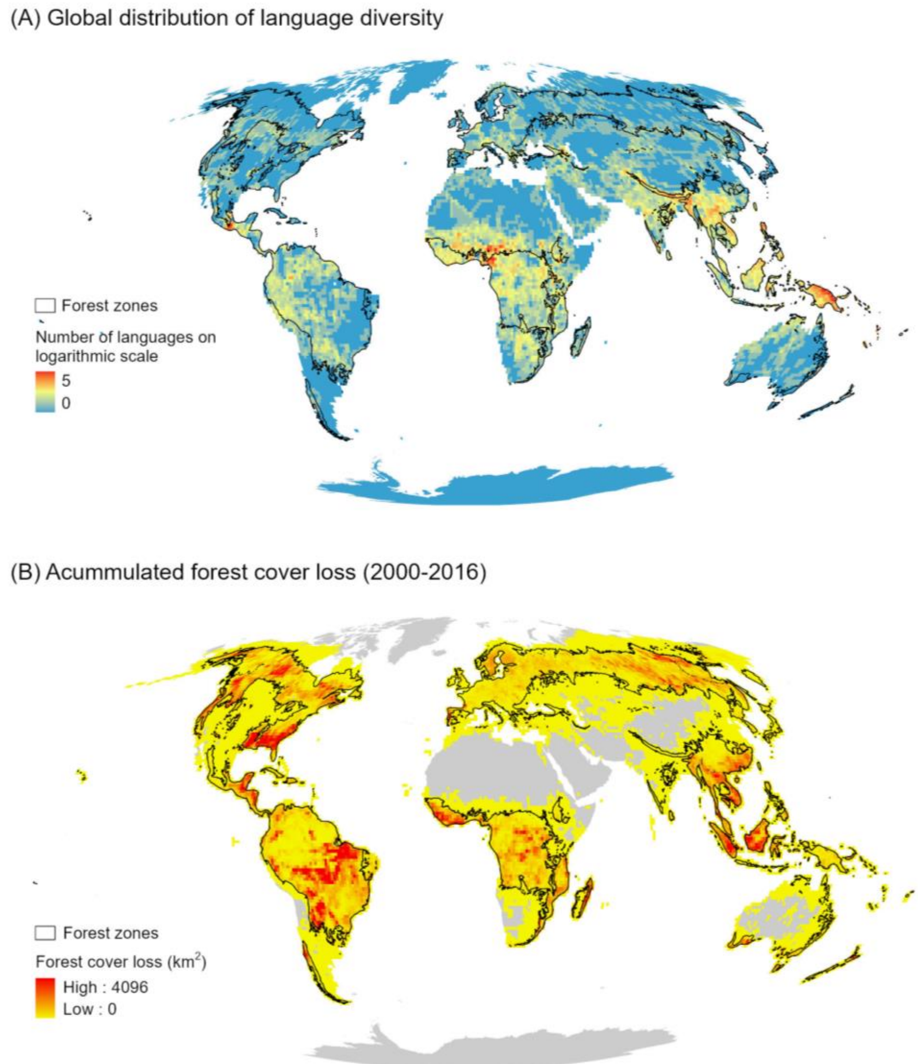


Figure 4.1. Global distribution of language diversity and forest loss. For 1-degree cells, we show (a) the number of languages (Lewis 2017) in a logarithmic scale, and (b) the accumulated forest loss between 2000 and 2016 (Hansen et al. 2013). The figure also shows the extent of the forest zones of the world (Potapov et al. 2017).

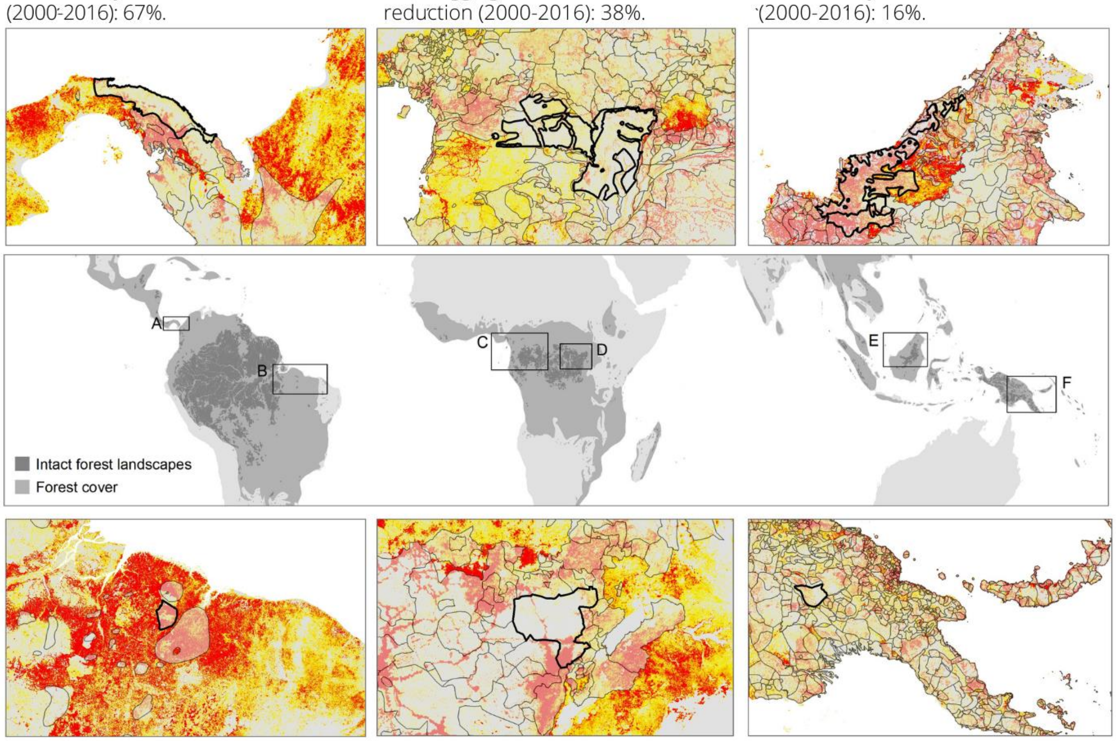
Today, it is widely recognized that cultural diversity is an important part of biodiversity and nature, and its emergence is driven by a process similar to the one driving the rest of biodiversity and natural processes. An example of this is the high global spatial correlation between species and linguistic richness (Gorenflo et al. 2012; Loh & Harmon 2014; Hamilton et al. 2020). However, the interconnections between ecological changes and cultural transformations have been largely overlooked in the literature, in part because these are complex and multidimensional (Austin & Sallabank 2011; Pérez-Llorente et al. 2013; Dunn 2018), and thus it is unclear to what extent and under which conditions forest loss and degradation can undermine human cultural diversity (Pérez-Llorente et al. 2013; Cámara-Leret et al. 2019a). Closing this knowledge gap is instrumental to effectively approximate the conservation status of forest cultural diversity and project their future trajectory, as well as to investigate and accommodate better policies that support IPLCs to adapt or prevent the impacts of forest loss.

In this perspective, we discuss whether this century's deforestation poses a significant threat to the world's human cultural diversity. We start by examining recent case studies looking at connections between forest loss and degradation and changes in cultural systems of IPLCs. We then take a spatial approach to estimate how many and which cultural groups from tropical regions could be at risk due to the loss and degradation of forests on which people depend. Finally, we summarize actions that might reduce deforestation impacts on cultural diversity and discuss the success and challenges for their implementation. This piece brings together sources primarily from anthropology, geography, and ecology to delve into the intertwinement of cultures and their natural environments and demonstrate that the maintenance of the world cultural diversity necessarily includes forest protection and access of IPLCs to them. We acknowledge that we are non-indigenous researchers addressing a global issue that affects indigenous people and other traditional communities and that our knowledge systems influence our position concerning the impact of deforestation on cultural survival. Our point of view embraces the notion that we, as just another species, become what we are through cumulative cultural evolution, a process that takes place in an ecological context and that affects our technological complexity, settlement pattern, and ideological repertories and which can change and eventually disappear as ecosystems and their contributions to people change (Boyd et al. 2011; Marquet et al. 2012; Santoro et al. 2017; Weinberger et al. 2017).

A. Kuna (*cuk*). For the past two hundred years, the Guna people traditionally managed one of the best-preserved forests in Central America. However, climate change and uncontrolled increase in deforestation by colonos are threatening Guna's food security, shelter, and identity. Intact forest reduction (2000-2016): 67%.

C. Baka (*bkc*). The Baka pygmies from the rainforests of Cameroon and Gabon are semi-nomadic, hunter-gatherers. They are struggling to maintain their traditional way of life since their territory is shrinking for alleged conservation projects, mining, and logging activities. Intact forest reduction (2000-2016): 38%.

E. Iban (*iba*). The Iban people, Borneo, have a more sedentary way of life. They farm, build wood houses, hunt, and harvest trees for local use. Expansion of oil palm plantations and invaders have reduced Iban's access to forests and land, eroding their core identity. Forest cover reduction (2000-2016): 16%.



B. Guajá (*gvj*). The Awá people, one of the last groups of hunter-gatherers of Brazil, is teetering on the edge of extinction. Logging companies and settlers keep exploiting Awá's rainforest, hunting the animals on which the Awá depend on and exposing them to diseases and violence. Intact forest reduction (2000-2016): 47%.

D. Lese (*les*). The Lese people live in relative isolation in the Ituri rainforest, Democratic Republic of Congo. The Lese are agriculturalists, live in small villages and get forest products from nomadic tribes. The Lese have been resistant to cultural change but deforestation is threatening their survival. Intact forest reduction (2000-2016): 6%.

F. Huli (hui). The Huli people reside in the remote highlands of Papua New Guinea. They live by hunting, gathering plants, and growing crops. Their cultural rituals, clothing, and shelter rely on forests. Deforestation and natural gas extraction is threatening the Huli's habitat and culturally valuable species. Intact forest reduction (2000-2016): 70%.

Figure 4.2. Examples of ethnolinguistic groups in tropical forests that are experiencing negative effects of forest loss and degradation on their livelihoods and cultures. These ethnolinguistic groups are communities that have traditionally inhabited tropical forests. For each group, the language name and 3-characters ISO identification is shown (Lewis

2017). The geographic range of each group is highlighted in dark black, while borders for neighbor groups are shown in a lighter grey. Gridded data from yellow to red indicate forest loss (km^2) between 2000 and 2016, with red representing the highest loss by 1 km^2 cell (see Fig. 4.2b). Descriptions of the impacts from forest loss and degradation on each group were obtained from: (A) Guidi (2014); Mateo-Vega et al. (2018); Minority Rights Group International (2020a), (B) Wallace (2016); Paiva et al. (2020), (C) Church & Page (2017); Minority Rights Group International (2020b), (D) Morelli & Wilkie (2000); Gay (2001), (E) Human Rights Watch (2019); Pahlevi & Butler (2019), (F) Kolinjivadi (2011).

2. Impacts of large-scale forest loss and degradation on cultural systems

Culture can be defined as information that affects behavior and is passed among individuals and across generations through social learning (Boyd & Richerson 1996; Whiten et al. 2017). This information constantly evolves through generating changes in the environment that can become selective pressures on the population that generated the change, giving rise to evolution through niche construction, and thus, to persistence or extinction, adaptation, or disappearance in response to changing conditions (Laland et al. 2001). This process is also expected to be species-dependent and idiosyncratic. Despite the dynamic nature of cultural systems, multiple assessments, mainly based on language diversity, suggest that we, humans, are currently facing a global cultural loss crisis (Sutherland 2003; Maffi & Woodley 2010; Austin & Sallabank 2011; Loh & Harmon 2014).

Here, we gathered recent research and IPLCs views detailing how forest loss and degradation can result in extensive direct and cascading impacts on multiple traits of IPLCs' cultural systems. This body of research is largely based on individual case studies for specific ethnic groups and forest regions. We analyzed and synthesized this information into three non-excluding pathways (and feedback mechanisms) by which forest loss and degradation transform the socio-environmental context of forest people (**Fig. 4.3**). Often, these transformations result in the erosion and a cultural tipping point (section 2.1), or instead, it can lead to cultural adaptation and renovation (section 2.2).

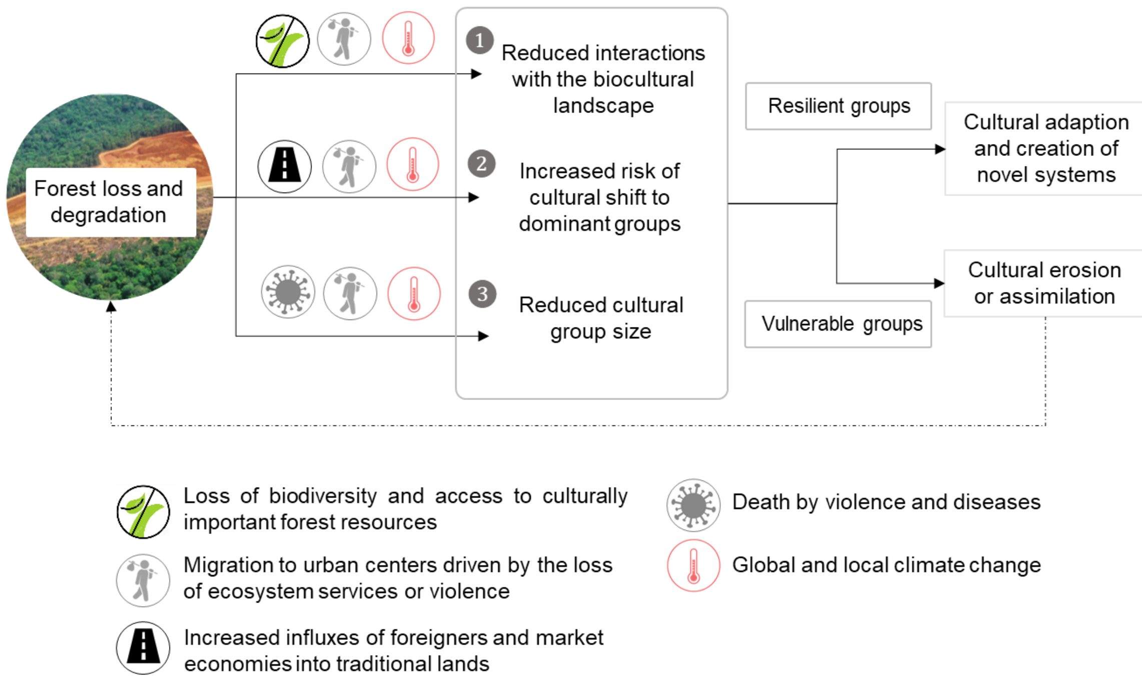


Figure 4.3. Connections between forest loss and degradation and cultural change in forest IPLCs. Circular symbols represent specific socio-environmental transformations driven by forest loss and degradation that forest IPLCs could face. Numbers indicate the pathways in which these transformations might affect the generation, maintenance, and transmission of cultural information. Traits that can confer groups resiliency and lead to adaptation of their cultural systems include having a large population size, strong sovereignty, social cohesion, and communication, among others. Instead, having smaller population sizes, undermined autonomy, exposure to unequal power relationships are traits associated with increased vulnerability to cultural erosion and assimilation in the face of deforestation. Furthermore, positive feedback (dashed line) increasing forest loss and degradation might result when cultural erosion involves disconnection between people and nature via the loss of conservation attitudes and the decline of traditional ecological knowledge.

2.1 Cultural erosion and assimilation

Pathway 1- Deforestation reduces access and interactions with the biocultural landscape.

Traditional ecological knowledge, management systems, identities, or languages of many IPLCs have been shaped by time and land (Robbins 2015; Ferguson & Weaselboy 2020; Ford et al. 2020). Thus, the continuity and evolution of these cultural systems rely on the interactions of people with land and local environment as a spiritual and material source (Lyver et al. 2019; Ferguson & Weaselboy 2020; Ford et al. 2020). Simultaneously, traditional communities can influence the landscapes that they inhabit. For example, in the Amazon forests, IPLCs have contributed to creating a cultural landscape that is the product of their traditional land use, including the introduction of novel combinations of wild and domesticated species (Lombardo et al. 2020). Traditional land-use might also include forest clearing but, because this use often involves sustainable practices or because many IPLCs live at low densities, most IPLCs have been able to maintain forest cover extent historically and some of the most biodiverse areas remaining on Earth (Freitas et al. 2004; Frainer et al. 2020; FAO & FILAC 2021; Ellis et al. 2021).

In contrast to traditional land uses, large-scale deforestation can profoundly change the biophysical environment and trigger a cycle of disconnection between IPLCs and the natural landscapes they inhabit. For instance, there is mounting evidence that plantations and deforestation erode landscapes that have served as cultural inspiration, alter sacred places, deplete culturally significant and useful species, and make traditional lifestyles no longer viable (Garibaldi & Turner 2004; Tang & Gavin 2016; Cámara-Leret et al. 2019a). Also, the loss and degradation of rainforests can undermine critical ecosystem services and the basis for the economic self-sufficiency of many IPLCs, often forcing them to migrate to urban settlements where their lifestyles may finally become alienated from forests (Austin & Sallabank 2011; Tang & Gavin 2016). As a result of long-term declines in access to forests and the disruption of interactions with the biocultural context, IPLCs lose opportunities for acquiring, maintaining, and transmitting traditional knowledge, rituals, vocabulary, and ways of life related to the environment (Pyle 2003; Reyes-García et al. 2013a; Kai et al. 2014). This is supported by comparative studies which have found that traditional communities or members exposed to larger declines on access, extent, and quality of native forests, or its local biodiversity, tend to exhibit major losses on cultural traits associated with forests (e.g., Kai et al. 2014; Atreya et al. 2018; Paneque-Gálvez et al. 2018; Cámara-Leret et al. 2019a; Parra et al. 2019; Turvey et al. 2021). Similarly, studies based on interviews and case reviews describe how several IPLCs have abandoned cultural practices and lost traditional knowledge partially due to the loss of native forests and the biodiversity that sustain such practices (e.g., Reyes-García et al. 2013a; Kodirekkala 2015; Kim et al. 2017; Ukam 2018). Noteworthy, loss of knowledge might occur at a slower pace than land-use change, which results in a time-delayed erosion of knowledge, a process similar to biological “extinction-debt” (Parra et al. 2019). Thus, the impacts of more recent deforestation events on cultural traits could emerge or be more evident years ahead.

<u>Pathway 2 - Deforestation increases the risks of cultural shift and assimilation towards dominant cultures.</u>

The cultural extinction crisis is often the result of processes that force or foster minority groups to disrupt the transmission of their cultural systems and adopt or assimilate that of more dominant groups (i.e., demographically or due to power or affluence)

(Weinreich 1953; Austin & Sallabank 2011). Thus, although interaction and exchange among cultures can be a key driver of cultures evolving (Axelsen & Manrubia 2014), under some circumstances it can also trigger cultural shifts and assimilation tipping points, in which the recovery of the former cultural expression state becomes unlikely (Mufwene 2004; Lyver et al. 2019).

Large-scale deforestation is likely to boost the contact between traditional people from forests and dominant outsider groups, increasing the risk of cultural shifts. In tropical regions, forest loss often comes in hand with an increase in accessibility and a large influx of foreigners, extractive industries and market economies into previously remote and sparsely populated forest areas (Curtis et al. 2018; Vilela et al. 2020). Under this socioeconomic change, IPLCs might voluntarily leave behind their traditional ways of life, cultural values and languages to adjust to the new conditions and access more competitive salaries, health system and consumer goods (Nettle & Romaine 2000; Mufwene 2004; Reyes-García et al. 2005; Austin & Sallabank 2011; Reyes-García et al. 2013a; Bozigar et al. 2016). Studies from the Amazon confirms such processes finding that communities that are closer to main roads and with higher access to urban settlements, market economies, and extractive centers usually experience greater cultural changes and erosion than those communities living in more pristine and less accessible forest areas (Peralta & Kainer 2008; Suárez et al. 2009; Pérez-Llorente et al. 2013; Reyes-García et al. 2013a; Vasco et al. 2018; Castro et al. 2020). In other cases, the arrival of extractive industries and intensive agriculture might lead to land conflicts, serious human rights abuses, and forced displacements of IPLCs to urban or rural centers where they are assimilated into dominant cultures (Nettle & Romaine 2000; Austin & Sallabank 2011). A case example is the palm oil industry in Malaysia and Indonesia, which has acquired large expanses of land for production, leading to the evictions and impoverishment of local indigenous communities with customary rights of use upon them (Colchester 2011).

Pathway 3 - Deforestation is linked to processes that can reduce the size of cultural groups.

Deforestation also associates with processes that increase violence and deaths in IPLCs. Large scale illegal activities, such as mining, logging and colonization, frequently occur inside the territories of many indigenous people in tropical regions (e.g., Escobar 2020; Baragwanath & Bayi 2020). These illegal activities increase the contact of indigenous people with outsiders (miners, loggers) who may bring infectious diseases to which indigenous people have no immunity or cure if they do not have access to health systems (Walker et al. 2015). For example, a recent study for the Brazilian Amazon estimates that illegal miners are the main vector of coronavirus transmission in the territory of Yanomami indigenous people, leading 40% of the population at risk of contracting COVID-19 (ISA 2020). Worldwide, illegal economic activities are also known to be associated with cases of brutal violence that takes the lives of many members of indigenous groups or coerces many others to migrate (Fraser 2017; Andreoni & Casado 2019). In this context, the last group members can also be forced to assimilate into more dominant or prosperous cultures to survive as individuals (Nettle & Romaine 2000). Since the generation, learning, and maintenance of cultural information benefits from the presence of conspecifics and their interactions, rapid reductions of group size, via displacements or deaths, ultimately leads to a pronounced deterioration of their knowledge systems, language, practices, triggering a process of cultural simplification (Carneiro 1967; Henrich 2004; Kline & Boyd 2010; Prochazka & Vogl 2017).

Feedback mechanisms among forest loss, climate change, and cultural erosion

Tropical deforestation is one of the main drivers of current climate change, accounting for about 12% of anthropogenic carbon emissions (Baccini et al. 2012; IPCC 2020). Indigenous territories are stewards of huge forest carbon pools, containing 34% of all above-ground carbon stored in tropical forests (Walker et al. 2014; Woods Hole Research Center & Environmental Defense Fund 2015). If lost, these would lead to the release of 168.3 Gt CO_2, or three times the global emissions in 2014 (Woods Hole Research Center & Environmental Defense Fund 2015). Impacts of climate change can also lead to significant alterations on native forests, such as increased frequency and intensity of wildfires, higher temperatures, changes in species compositions, and land conversion and degradation (Voggesser et al. 2014). IPLCs are among the first to witness and experience these impacts on forests, as well as on their livelihoods and cultures (Parrotta & Agnoletti 2012; Voggesser et al. 2014; Dunn 2018). Overall, climate change impacts on forests can boost socioenvironmental changes that lead to the three cultural transformation pathways (see **Fig. 4.3**). For example, in New Guinea, more than 700 endemic plant species used by forest indigenous people are projected to suffer contractions of their geographic ranges due to climate change, which pose a threat to the practice and transmission of knowledge and traditions associated with the use of these plants (Cámara-Leret et al. 2019b). In the Amazon forest, the number of wildfires within indigenous lands dramatically increased in the last few years, driven by a combination of illegal fires and changes in climate, such as more prolonged drought (Brando et al. 2020; IPCC 2020). These wildfires have increased respiratory diseases among native peoples and affected food sources of uncontacted groups, pushing them from their homes and increasing their exposure to other cultures and new diseases (Hanbury 2020). Importantly, climate change impacts are more intense in deforested areas but also reach out to IPLCs in more remote landscapes (Vargas Zeppetello et al. 2020).

The disconnection between nature and IPLCs triggered by deforestation, climate change, and other socioeconomic drivers can precipitate further forest clearing and degradation (Zent 2009; Pérez-Llorente et al. 2013; Paneque-Gálvez et al. 2018; Lyver et al. 2019). For instance, modifications on conservation behavior in many indigenous communities in the Amazon (e.g., adherence to traditional hunting norms, environmental awareness) and losses of their traditional knowledge related to sustainable resource use have been associated with an increase in deforestation (Fernández-Llamazares et al. 2015; Pérez & Smith 2019). Similarly, many groups in the Amazon have replaced sustainable forest-based activities with market-oriented activities requiring forest clearing, such as commercial agriculture and logging (Vadez et al. 2004; Castro et al. 2020). This deterioration of forests, due to the reduced engagement of IPLC with their environment, is then a feedback that reinforces cultural erosion (Zent 2009; Lyver et al. 2019).

2.2 Cultural resilience and adaptation in the face of forest loss and degradation

Cultural erosion and homogenization are not the only possible outcome of deforestation. Many IPLCs have a wealth of traditional knowledge that help them adapt to climatic, environmental, and social changes and avoid cultural assimilation (Reyes-García et al. 2013b; Davis et al. 2017). For example, some forest local communities in Honduras and Bolivia have responded to exposure to market economies by using and enriching their ecological knowledge to trade timber, non-timber forest, and fisheries goods (Godoy et al. 1998; Reyes-García et al. 2007; Aswani et al. 2018). Input from other cultures is, of course, not only a pressure on existing practices. Traditional knowledge, practices, and beliefs can evolve and adapt through hybridization with other cultures to create new forms and systems (Gómez-Baggethun et al. 2013; Aswani et al. 2018). For instance, despite being exposed to external economies and formal education, some Andean Quechua communities have enriched their knowledge on plant medicinal uses by incorporating species from other ecological belts and exchanging knowledge with Amazonian communities at local markets (Mathez-Stiefel et al. 2012).

These examples evidence the resilient and adaptive nature of IPLCs' cultural systems in the face of socio-environmental changes (Reyes-García et al. 2013b). Although IPLCs may lose specific pieces of cultural traits, they can also retain the ability to generate, transform, and transmit their cultural information, including their language, worldview, and values through social learning (Gómez-Baggethun et al. 2013; Reyes-García et al. 2013b; Lyver et al. 2019). This cultural adaptation process is likely to occur when key factors such as collective actions, sovereignty and communication are secured and reinforced in IPLCs (Gómez-Baggethun et al. 2013; Ford et al. 2020). However, the speed and magnitude of forest loss and degradation are usually too large for many cultural systems to regenerate and adapt (Fernández-Llamazares et al. 2015; Robbins 2015; Lyver et al. 2019; Ford et al. 2020). Moreover, deforestation usually comes in hand with unequal power relationships, with companies and governments sometimes directly seeking to weaken social cohesion within IPLCs through limiting their autonomy and ability to interact with the land on their own terms (Gómez-Baggethun et al. 2013; Bozigar et al. 2016; Lyver et al. 2019). This ultimately affects their possibilities to escape from cultural assimilation. Thus, rapid forest loss and degradation pose a significant challenge for many IPLCs, even for those with traits that potentially confer them resilience and the ability to cope with environmental changes (see **Fig. 4.3**).

Our examination of case studies shows that deforestation can have complex interactions with intrinsic and extrinsic factors related to IPLCs, such as group size, their historical contacts, the degree of diversification of their livelihoods, their social cohesion, sovereignty and recognition of their land rights, the political context, or additional socioeconomic drivers associated to cultural loss. As a result of these interactions, IPLCs' cultures can have different long-run prospects for survival that are difficult to predict with accuracy. Nevertheless, the evidence shows that large-scale deforestation has the potential to increase the risk of cultural erosion and assimilation, and that such risks must be addressed and mitigated.

3. Cultural diversity exposed to 21st-century deforestation in the tropics

Despite all the emerging evidence of the negative impact of deforestation on local cultures, there have not been efforts to quantify how many and which forest-dependent cultures across the tropics might be facing this threat. To explore this, we assessed the overlap between forest cover reduction from 2000 to 2016 (Hansen et al. 2013) and the geographic range of all living ethnolinguistic groups from tropical regions whose livelihoods and cultures are likely to rely on forests (Lewis 2017), as we exemplified in **Figure 4.2**. To get a close identification of forest-dependent ethnolinguistic groups, we used a spatial perspective, in which a group living within forests is a useful proxy for forest dependence (Newton et al. 2016, 2020). We selected only minority ethnolinguistic groups with at least 50% of their ranges overlapping with forest cover by 2000, as these are more likely to have a larger part of their cultural system related to forests (see Methods for more details in this analysis). This spatial criterion comprises many traditional and indigenous communities inhabiting forests for many generations (Newton et al. 2016).

According to our analysis, forest ethnolinguistic groups in the tropics (n = 3478) represent almost half of all known living languages (49%) concentrated in only 7% of the land surface of the world (~ 9.9 million km^2). Most of the forest linguistic groups inhabit Melanesia (28% of the forest diversity), South-Eastern Asia (28%), Middle Africa (12%), and South America (9%) (**Fig 4.4a**). Within this diversity, 475 ethnolinguistic groups overlapped, at least in half of their range, with the last remaining "intact" forests in the world by 2000. The number of groups living in intact forests is higher in South America (n = 155), followed by South-Eastern Asia (n = 123) (**Fig. 4.4b**). Intact forests are unbroken expanses of forests without remotely detected signs of human intervention (Potapov et al. 2017). These forests are also home to forest people with limited contact with the outside world, including traditional hunter-gatherer, semi-nomadic and horticultural societies (Watson et al. 2018; Begotti & Peres 2020). Therefore, these groups in intact forests are considered highly vulnerable to the rush of development and environmental transformations (Fraser 2017; Watson et al. 2018; Ford et al. 2020).

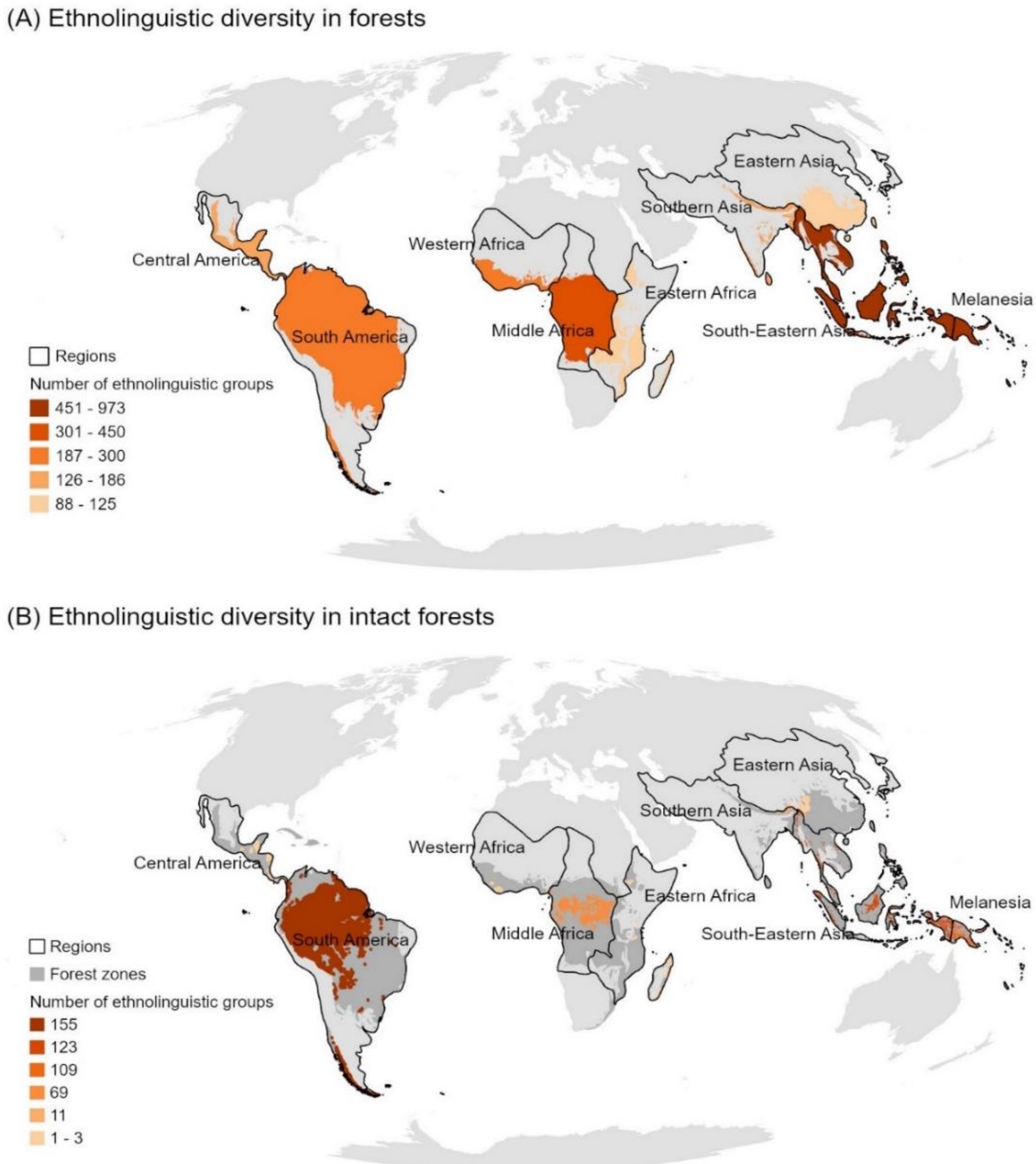


Figure 4.4. Forest ethnolinguistic diversity in the tropics of the world. Number of minority ethnolinguistic groups with (A) high overlap (≥ 50 %) with forest cover by 2000 (n = 3478), and (B) with high overlap (≥ 50 %) with intact forests by 2000 (n = 475). Data is summarized by the United Nations' regions. The analysis was limited to the forests in the Neotropics, Afrotropics, and Indo-Malayan biogeographic realms, with the Papua New Guinea portion of the Australasian realm included.

Between 2000 and 2016, about 41% of forest ethnolinguistic groups (n = 1436) experienced reductions in forest cover or intact forest cover ranging between 5% (0.33% annual) to 100% (6.7% annual) (**Fig. 4.5**). Such rates on forest loss might exceed those expected from the traditional land use by forest dwellers. For example, in the territories of indigenous agricultural communities in Indonesia, and indigenous communal lands in the Amazon basin, Costa Rica, and Zambia, annual forest loss rates from indigenous management are less than 0.3% (Blackman & Veit 2018; Nugroho et al. 2018; RRI 2020). Therefore, our finding suggests that almost half of forest ethnolinguistic diversity has recently faced high environmental degradation levels, which raises concern about their prospects for survival. Moreover, 40% of ethnolinguistic groups from intact forests (n = 191) had high intact forest reductions (**Fig. 4.5b**). The livelihoods and cultural systems of many of these groups are particularly susceptible to sharp socioenvironmental changes. For example, uncontacted, or in voluntary isolation, indigenous people are part of the ethnolinguistic groups facing intact forest reductions, such as: Awá in Brazil (47% of intact forest reduction), Yanomami in Brazil (5% of intact forest reduction), Cacataibo in Peru (37% of intact forest reduction), and Arará in Brazil (16% of intact forest reduction. It is also worrying that 31% (438) of the forest groups facing forest reductions already have low intergenerational language transmission (EGIDS: 6b-9), as registered by the language vitality indicator in the global database (Lewis 2017). This erosion of language transmission can weaken the practice and learning of their traditional ecological knowledge that is crucial to face climate change and deforestation impacts (Ford et al. 2020).

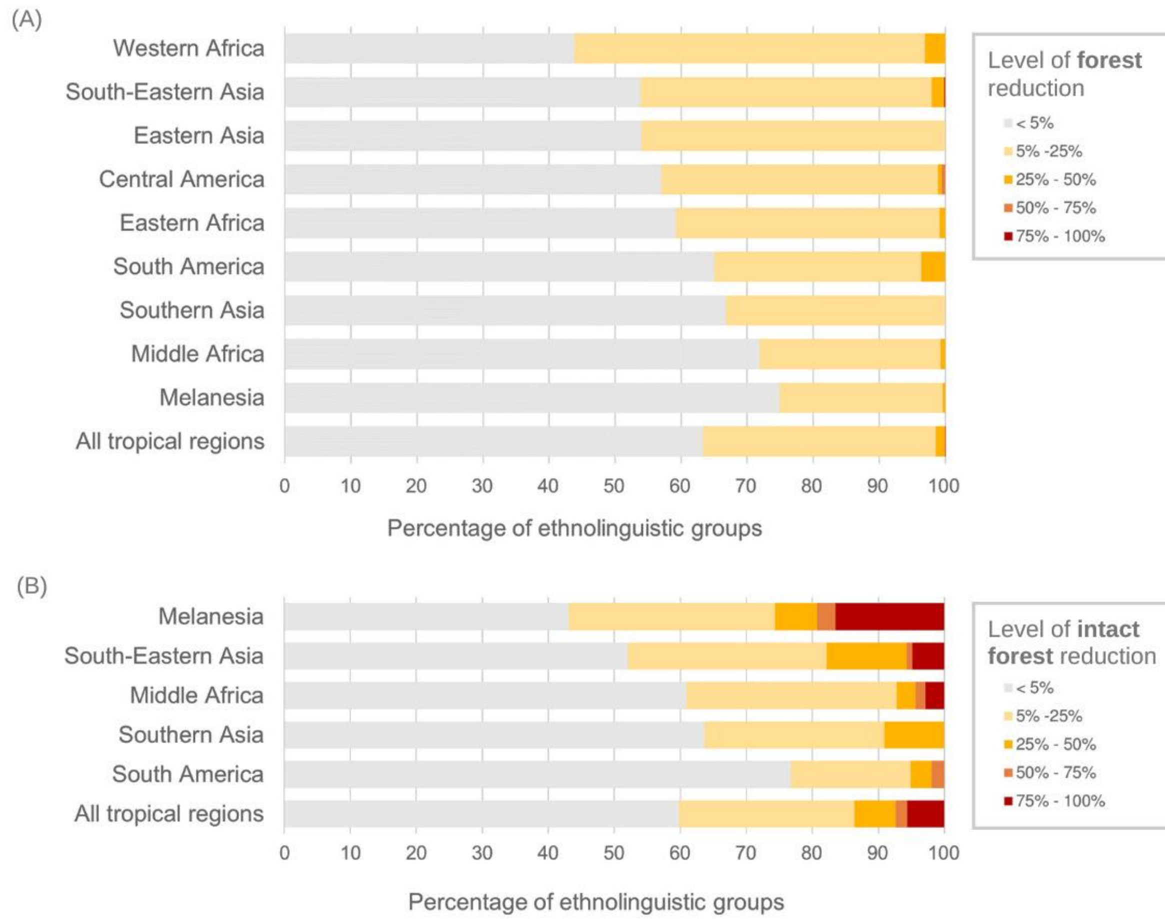


Figure 4.5. Forest reduction (2000 - 2016) across forest ethnolinguistic groups from tropical regions. Bars show the percentage of ethnolinguistic groups under each level of forest reduction rate. Results are shown separately for reductions in (A) forest cover and (B) intact forest cover. Reductions in intact forests combine forest cover loss and the increase of human intervention and fragmentation of forests (Potapov et al. 2017).

Forest reduction rates within ethnolinguistic groups varied regionally (**Fig. 4.5**). The regions with the largest percentage of groups overlapping with high rates of forest cover loss were Western Africa, South-Eastern Asia, and Eastern Asia have (56%, 46% and 46% of their diversity, respectively; **Fig. 4.5a**). Considering only the groups inhabiting intact forests, Melanesia and South-East Asia are the regions that include the highest proportion of groups facing intact forest reductions (57% and 48%, respectively; **Fig. 4.5b**). Overall, these results highlight South-Eastern Asia and Melanesia as the regions with the highest richness of forest ethnolinguistic groups exposed to deforestation. Although our study does not aim at addressing the factors behind this pattern, we detected some potential explanations. Ethnolinguistic groups from Melanesia and South-Eastern Asia have very small geographic areas (median: 183 km^2 and 715 km^2, respectively), in contrast to those in South America or Middle Africa (median: 3034 km^2 and 1948 km^2, respectively). Having small ranges increases the likelihood that deforestation and other stochastic events affect large proportions of their geographic ranges and speakers (Amano et al. 2014). In the last decades, these regions also exhibited the largest reductions of their forest and intact forest cover, mainly driven by commodity production and industrial logging (Rosa et al. 2016; Potapov et al. 2017; Curtis et al. 2018). Other factors, such as lack of recognition of many IPLCs' land rights and weak governmental environmental regulation (see below), might also contribute to the advance of forest reduction into the territories of ethnolinguistic groups from these regions.

4. Halting the impact of deforestation on cultural diversity

Our study gathers IPLCs' views and research evidence that exposes the connection between forest loss and degradation and the erosion of many forest cultures (**Fig. 4.3**). Threats to IPLCs from forest loss and degradation seem clear and have the potential to affect a large part of cultural diversity across tropical regions (**Fig. 4.5**). In this context, we argue that actions for revitalizing cultures and curbing their erosion must consider enhancing the protection of the forests and biodiversity that sustain cultures. We underscore that supporting forest cultural diversity needs a more integrative approach involving: (1)

greater efforts at tracking the impact of forest change on cultural systems of IPLCs, (2) ensuring IPLCs' land rights, and (3) incorporating IPLCs voices and needs into forest governance and conservation strategies.

4.1. Tracking the impact of forest change on cultural systems

Globally, there is a lack of assessments and databases explicitly designed for tracking temporal changes in different cultural traits of a group (Reyes-García et al. 2013a). For example, information on speaker numbers is a useful indicator for evaluating language endangerment (Lewis 2017). However, the available information for all ethnolinguistic groups is not always updated, limiting our ability to assess the impact of recent environmental change on cultural groups. This problem is exemplified by the case of Doso and Ukuriguma ethnolinguistic groups in Papua New Guinea, whose intact forests completely degraded between 2000 and 2016, but our current knowledge in their speaker numbers dates from 1973 and 2003 respectively (Lewis 2017).

As deforestation and land degradation advance, it is urgent to collect long-continued datasets based on large samples on how different cultural traits of IPLCs change over time and how these changes relate to forest loss and other social drivers of change (Pérez-Llorente et al. 2013). Combined with data about intrinsic and extrinsic factors related to cultural groups, this information would help (1) drawing robust, global, and regional scale conclusions about their short- and long-term impact of environmental degradation on cultures, (2) understanding the factors that help IPLCs retain the ability to generate, transform, and transmit their culture in the face of a rapid environmental change, as well as (3) designing strategies to support IPLCs coping with change.

4.2 Ensuring IPLCs' collective land rights

In many countries, progress has been made in recognizing and granting land titles as common property to indigenous people who have traditionally occupied and claimed these territories (Alden Wily 2018; Tubbeh & Zimmerer 2019; Tauli-Corpuz et al. 2020). This measure provides IPLCs with legal rights to manage forests crucial for their livelihood, cultural development and revitalization, and to take legal actions against those who would illegally or unethically exploit resources (Newton et al. 2016; Ferguson & Weaselboy 2020; Baragwanath & Bayi 2020). As a result, areas with secured community land rights are more prone to the successful implementation of conservation initiatives (Ostrom & Nagendra 2006; Santika et al. 2017; Griffiths 2018) and tend to report lower annual deforestation rates than other forms of governance (Araujo et al. 2009; Porter-Bolland et al. 2012; Nolte et al. 2013; Robinson et al. 2014; Blackman et al. 2017), although the latter is not always the case (Krishna et al. 2017; Busch & Ferretti-Gallon 2017).

Unfortunately, many other IPLCs throughout tropical regions lack proper conditions to exercise their land and autonomy rights. Therefore, their communal lands have become spaces of vulnerability and depletion of natural resources (Tubbeh & Zimmerer 2019). In many cases, governments have opposed conferring final decisions on indigenous peoples over export-oriented agriculture and industrial projects in their lands (Colchester 2011; Tubbeh & Zimmerer 2019; Tomlinson 2019; He et al. 2019). In countries with high forest cultural diversity, such as Brazil, Indonesia, and Papua New Guinea, there has been no significant effort to protect legal indigenous lands from deforestation and violence from illegal mining, logging, and expansion of forestry corporations (Laurance et al. 2011; Wallace 2016; Harris 2019). Also, in Brazil, recent presidential decrees are undermining

Indigenous Peoples' rights to their traditional lands (Begotti & Peres 2020). Furthermore, there seems to be an increase in the forced displacement of IPLCs globally driven by a sharp acceleration in the acquisition of traditional lands by foreign investors (Gilbert 2017). Thus, significant challenges remain in restoring, implementing, and ensuring land rights as part of strategies to support forest people's cultures and secure their livelihoods and autonomy.

4.2. Incorporating IPLCs' voices into forest governance and conservation

Safeguarding cultural diversity in tropical forests requires effective governance that protects forest integrity and functioning and equitable governance that employs inclusive processes and produces fair outcomes for IPLCs (Bennett & Satterfield 2018; FAO & FILAC 2021). For instance, in Indonesia, a weak regulation and insufficient local governance capacity undermined certification systems for sustainable palm oil that aimed to reduce the negative impacts of these industries on forests and local communities (McCarthy & Zen 2010). In Brazil, central forest governance created complex registration and permitting requirements for legal timber certification that puts this market out of reach of many Amazonian local communities, whose practices are precisely more sustainable and beneficial for local economies than those implemented by large logging industries in the area (McDermott et al. 2015).

Effective and equitable governance is also critical for protected areas. Globally, about 21% of indigenous peoples' lands are within protected areas, encompassing at least 40% of the global terrestrial area under protection (Garnett et al. 2018). In the tropics, protected areas have an important role in maintaining natural landscapes that are culturally and economically vital for many IPLCs (Lockwood et al. 2006). For instance, Communal Reserves in Peru are co-managed protected areas declared at the local communities' request to safeguard biodiversity for the benefit and sustainable use by these communities (Amend et al. 2017). However, the establishment of protected areas has not always have resulted in positive experiences for forest IPLCs. Worldwide, many IPLCs have perceived a general loss of access and their rights over land and natural resources due to inequitable management and governance of protected areas (RRI 2020; Tauli-Corpuz et al. 2020). In the Congo Basin, the expansion of protected areas has displaced several indigenous pygmy communities from their lands and prohibited traditional hunting, devastating the livelihoods of these communities, and eroding their cultural identity (Pemunta 2019) (**Fig. 4.2**). This critical situation reflects the need of securing IPLCs rights in and around protected areas, grating participatory management, and managing the potential conflicts with the conservation aims of administrative authorities of protected areas (Worboys et al. 2015; Pemunta 2019). Equitable management in protected areas is also critical for effective forest conservation. When local communities are genuinely engaged in decisions regarding rules affecting natural resource use, the likelihood of following the rules and monitoring others is greater than when an external authority imposes the rules (Ostrom & Nagendra 2006). Nevertheless, IPLCs are highly diverse in their land use and views on nature protection (Gray & Bilsborrow 2020; Pascual et al. 2021), which includes the fact that not all IPLCs could be interested in managing their forest for nature-conservation objectives (Garnett et al. 2018).

Recognizing IPLC's voices and rights also has important implications for designing future actions and goals for biodiversity conservation. Some conservationists are proposing expanding conservation areas to cover half the Earth by 2030 (Wilson 2016). Based on this goal, “intact” areas have been identified as good candidates to focus on proactive nature conservation efforts, including protected areas (Riggio et al. 2020). Our study underscores that many of these low impact lands are also reservoirs of high cultural diversity linked to forests. Therefore, the design of such ambitious conservation goals should explicitly address local acceptability and guarantee the right of local cultures to thrive. It is also critical to ensure that IPLCs voices are effectively articulated in international forums and policy-related processes for biodiversity conservation, such as the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services and the Post-2020 Biodiversity Targets framework in preparation by the Convention on Biological Diversity (Tauli-Corpuz et al. 2020; McElwee et al. 2020). These articulations have the potential to encourage governments to adopt policies that secure the rights of IPLCs to cultural distinction when designing actions for the protection of forests and other ecosystems.

5. Conclusions

Worldwide, research on deforestation has mostly focused on its drivers and impacts on biodiversity loss (Gibson et al. 2011; Busch & Engelmann 2017; Alroy 2017), erosion of ecosystem services (IPBES 2018a), or climate change (IPCC 2020). In this perspective, we underscore that deforestation can also erode natural diversity, from microbes to human cultures. Evidence from case studies illustrates the vital importance of the forest-culture connection, showing that forest loss and degradation might precipitate cultural erosion and homogenization processes in many IPLCs. Considering this, a large part of the world's cultural diversity could be at risk, with at least one-fifth of all living ethnolinguistic groups witnessing substantial losses of forest cover in their territories since 2000. Unfortunately, there is a lack of global efforts to monitor and project how these cultural groups could respond and adapt to this change. Nonetheless, we do know that many of them have already low intergenerational transmission of language and that they often lack recognition of land rights, limiting their ability to cope and adapt without eroding their cultural systems. This cultural extinction crisis advocates for the creation of the subdiscipline of anthropological conservation, to advance our understanding of the global changes in cultural traits and drivers. We also underscore the immediate need for global and national environmental policies and agreements that recognize the vital values of forests for sustaining human diversity, as well as the importance of ensuring land rights and autonomy of IPLCs to guarantee cultural adaptation and continuity. Overall, the outstanding cultural diversity contained in forest dwellers provides a global service that involves assets of innovations, knowledge, and wellbeing that merits increased political protection and financial support, particularly in the face of climate change.

6. Methods

This section details the spatial analysis for assessing the overlap between forest reduction (from 2000 to 2016) and cultures from tropical regions that are potentially linked to forests. Our analysis is a first attempt to approximate the extent that the impact of deforestation can have in terms of the number of cultural groups affected.

6.1 Forest cover dataset

Our analysis used two types of indicators of forest cover. First, we gathered data on forest cover and loss from the Global Forest Watch (Hansen et al. 2013). This dataset provides annual tree cover loss (since 2000) as a binary presence/absence value, defined as complete stand replacement or a change from a forest to a non-forest state within a pixel with a resolution of 30x30 m. We acknowledge that this product addresses tree cover and is unable to separate the natural forest from forest plantations in regions such as Southeast Asia. To reduce this source of error, we considered as non-forests all tree cover that overlapped with plantations (of native or introduced species) according to the Tree Plantation data, also available in Global Forest Watch. Second, we evaluated the changes in the intact forest landscapes of the world (Potapov et al. 2017). While global tree cover includes forests of different levels of intervention, intact forest landscapes refer to forests without remotely detected signs of human activity or habitat fragmentation. The estimation of the reduction of intact forests considers the tree-cover loss and the increase of fragmentation and anthropogenic disturbances that may negatively affect the provision of forest resources and expose IPLCs to different threats. Finally, for each linguistic group (see below), we estimated the percentage of its geographic range overlapping with forest cover and intact forest cover in 2000 and the rate of forest reduction and intact forest reduction until 2016.

6.2. Cultural diversity dataset

We used the geographic range of living languages from the 21st Edition of Ethnologue Database (Lewis 2017) as an indicator of cultural diversity (Cámara-Leret et al.

2019b). Language is the primary medium of cultural transmission, and thus, the ability of many societies to name, identify, use and share knowledge about their surrounding resources is linked to each language group (Cámara-Leret et al. 2019b; Ferguson & Weaselboy 2020). Thus, in many cases, a group of people that share a common ethnicity, territorial identity, and cultural heritage also share the same language (Barrett 2003). Hereafter, we refer to these languages as ethnolinguistic groups. Ethnologue maps use polygons to show the approximate boundaries of the traditional homelands of each ethnolinguistic group. When possible, this dataset also shows the number of speakers, level of endangerment, and other traits for each group. Nevertheless, no precise information on the year of the data used to build the map is indicated, and no claim is made for precision in the placement of these boundaries. These limitations might reduce the precision of our analysis. Still, Ethnologue is the most comprehensive and trusted geographic data set of the locations of the world's ethnolinguistic peoples. There have been important efforts to map the geographic ranges of indigenous groups (see Garnett et al. 2018), which is also a useful indicator of cultural diversity. However, this spatial information is still absent in some culturally diverse tropical countries, such as Papua New Guinea.

6.3. Identification of cultures potentially linked to forests

Currently, there is not available systematized information on which cultures of the world rely on forests for their survival. Moreover, the criteria for defining "forest-dependent people" (i.e., people that get benefits to some degree from forests) vary considerably in the literature (Newton et al. 2016, 2020). In this context, we used two criteria to identify groups whose survival and culture are probably linked to forests in the three major tropical biogeographic regions – the Neotropics, Afrotropics, and Southeast Asian tropics (Dinerstein et al. 2017). First, we followed a spatial perspective, in which people living in forests might be more likely to interact directly with forests and depend on them for their livelihoods, traditional practices, and well-being, compared to people living farther from forests (Newton et al. 2016, 2020). From the language database, we filtered ethnolinguistic groups, whose geographic range overlapped in more than a half (≥ 50%) with forest cover by 2000 (Hansen et al. 2013). Using this large percentage as a threshold, we expected to increase the probabilities of identifying groups with stronger connections and dependence on forests. From this set, we also distinguished those linguistic groups that also overlap with intact forest landscapes in at least half (≥ 50%) of their range. As a second criterium, we filtered linguistic groups with small size (those with less than one million speakers; Canvin & Tucker 2018) because people that heavily rely on forests usually are minority groups or live in low population densities (e.g., less than 100,000 people in the case of the indigenous lands of Brazil; (Begotti & Peres 2020).

The use of these criteria resulted in a set of 3478 ethnolinguistic groups from tropical forests. Altogether, this set of forest linguistic groups has at least 158 million speakers worldwide. Reports suggest there are 200 million indigenous peoples that depend on forests and 1.6 billion rural people depending upon forests to some extent (Chao 2012; Newton et al. 2020). Thus, our estimates of forest linguistic groups are probably closer to reflect the diversity of forest indigenous peoples and other local or tribal communities. Furthermore, when possible, we consulted available literature, reports and news to verify the nature of the relationship between these linguistic groups and forests (e.g. world information http://www.forestpeoples.org/, https://www.survival.es/, https://www.iwgia.org/, https://minorityrights.org/, or national database such as Brazil: https://pib.socioambiental.org/, Peru: https://bdpi.cultura.gob.pe/). We found that this set is composed by an array of groups with different ways of life and associations with forests, such as isolated indigenous peoples in the Amazon basin, traditional agriculture and hunter gatherers tribes in Papua New Guinea, or semi-nomadic communities in the Congo Basin (e.g., **Fig. 4.2**). Nevertheless, for most ethnolinguistic groups in the world, especially for Melanesia, there is no available and detailed information on their specific ways of life, economy or demography.

Our analysis used conservative thresholds in forest cover and speaker population size to reduce commission errors when identifying ethnolinguistic groups that are potentially linked to forests. However, this might have erroneously excluded other groups linked to forests, for which a long history of deforestation may have reduced their forest cover below 50% by 2000 (such as the Xavantes people in Brazil or Mapuches in Chile). Thus, we carried out a sensitivity analysis where we assessed the results of (1) increasing the threshold of speaker number from 1 million to 10 million, and (2) allowing the inclusion of linguistic groups overlapping with forest cover in at least 25% of their geographic range (Supplementary Information). Compared to the principal analysis, combined both criteria increased the number of identified forest ethnolinguistic groups from 3478 to 4167, representing an increase from 49% to 59% of all living languages of the world, respectively. Moreover, the number of ethnolinguistic groups exposed to significant forest loss rates (5% - 100%) also increased from 20% to 25% of the living languages. By expanding the analysis towards ethnolinguistic groups with smaller forest coverage, we also consider territories with vulnerable forests to degradation events (due to their smaller sizes) or that have been under deforestation pressures for a long time. This explains why this additional analysis offers a more critical picture of the deforestation rates and advance in cultural groups.

6.4. Additional caveats

The present spatial analysis has some caveats to be aware of. First, in some cases, the same linguistic group could include very distinct IPLCs with unique traditional knowledge, rituals, or organizations that could be exposed to different rates of forest loss. Second, although spatial proximity is a useful proxy of forest dependence, some IPLCs might rely on neighboring forests, which means that their livelihoods and culture could be affected by deforestation even if they do not overlap with a large extent of forests or if the deforestation event occurs outside their territory (Newton et al. 2020). Third, our analysis assesses deforestation rates between 2000 and 2016 in the territories of cultural groups, overlooking previous deforestation events. For example, in the 1970s and 1990s, deforestation first accelerated in the Amazon and South-East Asia, respectively (Rosa et al. 2016). Overall, these caveats might lead us to underestimate the accumulated impact of deforestation on the world's cultural diversity. Thus, there is room for future studies to improve the precision of this analysis. However, we require better data on the current and historical location of different cultural groups and the nature of their relationships with forests.

7. Appendices

1. Summary of the ethnolinguistic diversity associated with forests and deforestation rates in these territories according to the world's regions.

2. Results of the sensitivity test for the criteria used to identify ethnolinguistic groups associated with forests.

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General Discussion

It is well established that future expansions of area-based conservation should place a particular focus on protecting and restoring tropical ecosystems, given their exceptionally high biodiversity value. The research presented in this thesis goes a step further and delves into the economic, ecological, and social challenges that tropical regions will face to expand and build a system of area-based conservation that is able to contribute meaningfully to biodiversity conservation and people's wellbeing.

In Chapter 1, I assessed the relationship between funding and the performance of PAs in Latin America using unique and precise financial data and state-of-the-art statistical matching to explore how funding shortfalls affected the ecological outcomes of PAs. Our results offer novel insights on the drivers of PAs effectiveness and how they operate and interrelate at regional and national scales. When looking across Latin America, I found that the quality of national governance is the strongest predictor of impact, suggesting that protected area performance is first and foremost a result of the overall country-level policies that affect forest loss inside and outside PAs. But when looking within a specific country, I detected that funding plays a crucial role in improving the impact of individual PAs. Thus, I show that to improve PA effectiveness, we need to address both country-level governance structures as well as to ensure sufficient funding for site-level management, especially for PAs facing major human pressures on forests.

Many tropical regions lack high-quality data related to the cost of expanding their PA system. The absence of reliable cost data precludes the ability to assess cost-effectiveness when identifying conservation priority areas. In Chapter 2, I generated spatial information on PA costs (management and opportunity) and combined it with species distribution maps to identify cost-effective priority areas for conservation in the western Amazon. This contribution may thus help governments and conservation institutions to achieve biodiversity conservation objectives while maximizing scarce conservation resources. It also helps to understand the conditions that enable less expensive conservation, such as collaboration with indigenous peoples and among countries. Moreover, this analysis offers the first geospatial model that explains the variation in the annual management costs of Amazonian PAs, which conservation planners and scientists could use to estimate the budget needed for different prioritization scenarios.

Chapter 3 enables a view of the Andes as a single entity, whose high biological and cultural diversities are an important heritage that needs to be preserved and managed in an integrated manner. Based on this view, I provide the most comprehensive review of the current spatial protection needs for biodiversity and Nature's Contribution to People wellbeing along the Andes. I also delve into the economic challenges associated with enhancing such protection, and recognize the diversity of actors living in the region and how their different interests need to be made compatible. Thus, this study seeks to enable transboundary, effective, and fair conservation planning that maintains the critical socio-environmental services that the region provides to the local and global human population.

Worldwide, hundreds of indigenous and local communities claim that undesired and rapid deforestation in their territories negatively affects their livelihoods and erodes their cultural systems. As a response to this claim, in Chapter 4, I reviewed individual case studies to build the first conceptual model that explains the connections between forest loss and cultural change in local communities. Moreover, through spatial analysis, I detect that, since 2000, rapid deforestation has been taking place in the territories of ~1,400 cultural groups across the tropics, which might threaten a large part of the world's cultural diversity. These results point out the need for a more integrative approach to support cultural continuity; preserving and documenting languages, beliefs, and practices will achieve little at curbing cultural erosion if not accompanied by the protection of the landscapes that support those cultures. We urgently need to support stronger forest protection policies and the rights of local communities to the land and resources that allow their cultures to thrive.

Lessons learned and action areas to strengthen area-based conservation

In this section, I integrate the main findings of the four chapters of this thesis into five areas of actions, which together could enhance the financial sustainability, ecological representativeness, and equitable management of conservation areas (**Fig. 1**). Below, I provide arguments for these actions and discuss the conditions and challenges of their implementation.

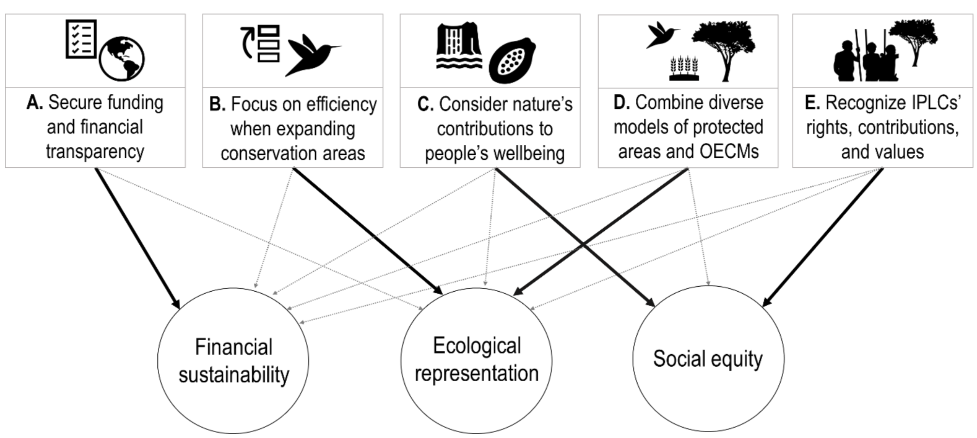

Figure 1. Areas of actions (A-E) to build more financially sustainable, ecological representative, and socially equitable networks of area-based conservation in tropical regions. Each arrow connects an action to the aspects of conservation areas that are

expected to be improved. Dark arrows indicate direct positive impacts, while dashed arrows refer to additional positive impacts.

Action A. Secure adequate funding and financial transparency.

This thesis provides evidence that current and future PAs should be accompanied by effective governance and adequate funding to avoid the proliferation of "paper parks" with low conservation success (**Chapter 1**). Ensuring a proper budget is critical for PAs exposed to high human pressures since their effectiveness is more vulnerable to underfunding (**Chapter 1**). The budget needed for covering basic management in current and new PAs represents a small percentage of the gross domestic product of, at least, some Latin American countries, suggesting that moderate increases in government spending could reduce financial and ecological gaps (**Chapter 2**). However, some countries have critical economic constraints, PAs under chronic underfunding and large deforestation pressure (**Chapter 1**). Thus, increased and stable international and private financing is particularly relevant for supporting these countries in covering PA management needs in the short term (**Chapter 1**, **Chapter 2**).

A financially sustainable and effective PA system also requires nations to produce transparent reports on PA budget, investment allocation, and resource use efficiency (**Chapter 1**). In this context, conservation cost modelling can help governments to project and plan more adequate budgets for PAs (**Chapter 2**). This thesis has discussed other actions that can also contribute to achieving a financially sustainable system, such as protecting places that deliver high return of investment for biodiversity (see Action C) or that provide NCPs that are valued by the population and are potential sources of revenues

(Action D) or governing conserved areas in partnership with diverse landowners to fairly share the costs and benefits of protection (Actions E and D).

Action B. Focus on efficiency when expanding conservation areas.

Findings of this thesis support the need to expand area-based conservation efforts but targeted at the right places. Analyses for the western Amazon (**Chapter 2**) and the Andes mountain range (**Chapter 4**) show that significant additions of conservation areas are needed to achieve a higher and more balanced representation of species, ecosystems, or NCPs of interest. Importantly, future investments in expanding areas under protection should be efficient, which means that PA additions must be strategically targeted at places with high biodiversity conservation value. Efficiency is crucial because it minimizes the risk of constructing a PA system that is unnecessarily too large and expensive to manage (**Chapter 2**). As this thesis exemplifies, areas of high value for biodiversity are not evenly distributed in space. In the western Amazon (**Chapter 2**) and the Andes (**Chapter 3**), systematic conservation planning exercises detected specific areas that represent cost-effective solutions to the goal of covering species and ecosystems currently lacking protection. Overall, to guide efficient decisions, national or regional policies should encourage the documentation of the value of all sites of significance for biodiversity (e.g., wilderness retention, restoration, connectivity, prevention of species extinctions under climate change) and establish representation targets for them (Watson et al. 2016; Di Marco et al. 2016; Visconti et al. 2019). These different values could lead to overlapping spatial plans but also to distinct suggestions on where to locate PAs that would need to be further discussed by decision-makers and stakeholders.

Action C. Consider nature's contribution to people when expanding and managing conservation areas.

Natural and semi natural areas represent the main source of subsistence and cultural identity for countless IPLCs in the tropics (**Chapters 3**, **Chapter 4**). Therefore, current and future PAs must ensure the integrity of people-nature interactions that sustain significant ecological and cultural values (**Chapters 3**, **Chapter 4**). In addition to targeting biodiversity representation, Systematic Conservation Planning may include among its objectives safeguarding and providing demanded NCPs and other values of nature. This would contribute towards promoting conservation schemes that enhance the equitable distribution of the benefits delivered by protection, while also helping to justify funding and investment opportunities (**Chapter 3**). In this context, spatial prioritization can be used to boost the efficiency of PA systems by identifying areas whose protection would close ecological representation gaps while also capturing demanded NCPs (**Chapter 3**). Overall, it is clear that the design of future PAs is sensitive to people's needs and how they value nature. Negotiations with various interest groups are therefore critical to achieving a more satisfactory conservation planning for people and nature.

Action D. Promote area-based conservation with diversified management and governance.

Many of the sites identified in this thesis as priority for improving biodiversity representation overlap with lands that sustain the livelihoods and cultures of thousands of people and with territories managed by indigenous people and other smallholders (**Chapter 2**, **Chapter 3**). Therefore, protecting these priority sites requires strategies that address the aspirations, needs, and rights of local actors. This could be achieved by promoting alternative governance and management schemes to the widely implemented state-owned strict protection, which usually restrict human activity and lead to social conflict. For example, an alternative is to document and recognize landscapes that are well-managed by private owners and local communities and support their efforts as "Other Effective area-based Conservation Measures" (OECMs) (**Chapter 3**). Moreover, since processes and actors that influence the impact of PAs can occur at multiple levels (e.g., indigenous communities at site-PA level, and the governance quality at a national level, **Chapter 1**), a polycentric institutional scheme can provide a useful framework of governance (i.e., with multiple centers of semiautonomous decision making, Nagendra & Ostrom 2012).

By combining diverse governance schemes and management objectives of reserves and OECMs it is possible to complement biodiversity representation, improve protection effectiveness and connectivity while providing a range of benefits for local actors. Moreover, by engaging with local actors managing other areas (e.g., community lands, private owners), the conservation costs of closing ecological representation gaps can be shared (**Chapter 2**). Still, there are major challenges related to private rights and conservation success that policies need to overcome, such as determining and negotiating the responsibilities that accompany private conservation and what conservation outcomes are expected from it (Moon et al. 2021).

Action E. Ensure IPLCs' land rights and recognize their contribution to biodiversity conservation and diverse ways of valuing nature.

In tropical regions, the establishment and management of PAs can have a profound impact on the wellbeing of IPLCs, who in turn are central actors to achieve successful PAs. For instance, in the western Amazon and Andes, an efficient PA expansion for biodiversity conservation needs IPLCs as allies because sites for closing representation gaps often overlap with territories inhabited by IPLCs (**Chapter 2**, **Chapter 3**). Moreover, establishing new PAs in Amazonian indigenous lands is associated with more affordable management costs (**Chapter 2**). IPLCs' traditional knowledge can also help achieve sustainable management of biodiversity and NCPs within PAs (**Chapter 3**). IPLCs might also influence the impact of PAs in avoiding deforestation (**Chapter 1**). Likewise, PAs can positively affect the prospects of cultural survival of IPLCs by protecting the forests that sustain these communities. However, PAs can also affect them negatively by evicting IPLCs from their territories against their will (**Chapter 4**). All of this suggests that, to deliver positive outcomes for IPLCs and nature, conservation planning in tropical regions must investigate and address these interconnections. For instance, PAs need to guarantee IPLCs access to their biocultural environment and space for reproducing their biocultural practices because this is a key factor to sustain their traditional ecological knowledge and conservation attitudes that also benefits PAs (**Chapter 4**). Also, since traditional indigenous management seems to lead to more effective forest protection than other governance regimes (**Chapter 4**), many suggest that enforcing collective IPLCs' tenure rights and advocating indigenous-led conservation is a more cost-effective policy for improving ecological representation and social equity than establishing state and strict forest PAs (RRI 2020; Tauli-Corpuz et al. 2020). Nevertheless, the success of community rights-based conservation needs a sound understanding of the conditions that have traditionally enabled the long-term protection of biodiversity. For example, there is concern that the rising global demand for palm oil and rubber could lead to upward shifts in land prices in Indonesia, which could provide new incentives for local communities to sell their lands or cultivate plantation crops compromising biodiversity (Krishna et al. 2017). IPLCs are also highly diverse, and in some cases, their aspirations, economic interests, or perspectives on how to manage biodiversity might not be compatible with the views of conservation science (Pascual et al. 2021, **Chapter 4**). In such situations, efforts towards recognizing, negotiating, and coordinating actions for biodiversity and IPLC's perspectives are required to come up with fairer conservation interventions within the same territory (**Chapter 2**, **Chapter 4**). Overall, scientific findings support the conservation role of IPLCs, but it is important to avoid uniform policies that homogenize indigenous groups and their socioeconomic contexts (Gray & Bilsborrow 2020).

The five areas of actions recognize the tropics as highly biodiverse and as human-inhabited landscapes, where there are close and long-term interdependencies between people's wellbeing and nature (Fa et al. 2020; Ellis et al. 2021). These actions also align with the "inclusive conservation perspective" (Tallis et al. 2014) or "People and Nature" approach (Mace 2014), in which different visions of biodiversity values and conservation area management, ranging from biodiversity-centered to socio-economically driven motivations, are balanced to achieve ecologically and socially relevant outcomes. Nevertheless, the implementation of these actions may involve trade-offs and unintended consequences. For example, it has been suggested that sharing conservation areas for sustaining biodiversity and NCPs may deliver poor outcomes for both goals compared to lands managed exclusively for biodiversity or NCP utilization (Karp et al. 2015; Ellis & Mehrabi 2019). Overall, measuring the success of area-based conservation and optimizing trade-offs is difficult when nature and people are considered together (Mace 2014). Thus, it is critical for science to produce coherent, inclusive, and measurable metrics on the social and biological impact of area-based conservation (Faith et al. 2010; Mace 2014).

Informing global agendas for area-based conservation

The world's nations are currently defining the post-2020 Global Biodiversity Framework to set the planet on a path to a sustainable future for biodiversity. This framework will formulate a new and ambitious target for area-based conservation ("Target 2") that involves increasing its effectiveness and coverage by 2030 (CBD 2020). Here, I based on the main findings of this thesis on how to strengthen area-based conservation in the tropics, I discuss the proposed Target 2, as presented in the Update of the Zero Draft of the Post 2020 Global Biodiversity Framework (CBD 2020):

> *"Target 2. By 2030, protect and conserve through well connected and effective system of protected areas and other effective area-based conservation measures at least 30 per cent of the planet with the focus on areas particularly important for biodiversity."*

According to the results of this thesis, the following elements of Target 2 fit the needs of area-based conservation in the tropics: (1) scaling up the proportion of land that is covered by conservation areas focusing on areas of importance for biodiversity, (2) through PAs and OECMs, and (3) highlighting the need for building a well-connected network and with effective management (which demands proper funding). However, there are two aspects relevant to tropical regions that are not directly addressed in the currently proposed target and suggested indicators to track its progress.

First, the proposed Target 2 focuses solely on preserving important areas for biodiversity. Thus, its implementation might fail to recognize the need for building a system of conservation areas that maximize the provision of a broader range of nature's benefits and services required by people (Action C). Likewise, this target overlooks the role of PAs in meeting different United Nations' Sustainable Development Goals (e.g., poverty alleviation, food and water security, disaster risk reduction) and supporting climate change mitigation deals (Bhola et al. 2020; Arneth et al. 2020).

Second, the current draft for Target 2 lacks explicit recognition of IPLCs' land sovereignty and contribution to biodiversity conservation (Action E). Moreover, it does not address the importance of ensuring equitable management, which was explicit in the former area-based target in the 2011-2020 Biodiversity Framework). Consequently, the current wording has raised concern in the Global South about the negative social impact that a massive scaling up of conservation area coverage might have on IPLCs, small landowners, and rural population who inhabit priority places for biodiversity protection (Büscher et al. 2017; Ellis & Mehrabi 2019; Minority Rights Group et al. 2020; RRI 2020; Agrawal et al. 2021). This thesis has treated some of these risks, including the loss of access to land and resources, the erosion of cultural systems, or the reduction of life opportunities (RRI 2020; Agrawal et al. 2021). Moreover, although the CBD and the OECM framework identify indigenous-conserved lands (e.g., ICCAs) as a critical governance regime, their recognition remains in practice marginal compared to state PAs and other OECMs (Tauli-Corpuz et al. 2020).

Given the ambitious expansion of conservation-based areas proposed by Target 2 (covering at least 30% of the planet), I argue that this target should be adapted to better accommodate the needs of both nature and local people and avoid unintended negative social impacts. For this, Target 2 should reflect on the types of areas-based conservation that are promoted and how they are going to be sustained. Although there is no single strategy to solve these challenges, I suggest that a more ecologically sound, cost-effective, and socially just proposal for Target 2 would be:

> *By 2030, protect and conserve through a well-connected, effectively, and* ***equitably managed*** *system of protected areas and other effective area-based conservation measures, in partnerships with* ***lands owned or governed by indigenous peoples and local communities****, at least 30% of the planet with the focus on areas particularly important for biodiversity and other* ***nature's contributions to people's wellbeing****.*

This alternative formulation of Target 2 seeks to reconcile an ambitious expansion of conservation areas with the needs and aspirations of diverse sectors and local communities.

In this manner, the target for area-based conservation aligns better with the 2050 Vision for Biodiversity of “living in harmony with nature” that was adopted as part of the Strategic Plan for Biodiversity (CBD 2018).

Conclusions

Chapter 1 – Impact of funding deficit on conservation effectiveness

- Most assessed PAs within Ecuador and PA systems in Latin America curbed deforestation compared to unprotected lands, which evidences their contribution to biodiversity conservation.
- Within a PA system (Ecuadorian case), lower PA impact was associated with larger funding deficit, especially in PAs facing major human pressure over their forests. Low human development reduces the overall impact of a country's PA system in curbing deforestation.
- PA impact can be maximized with better resource allocation for individual PAs, combined with strategies for strengthening institutions and governance of PA systems.
- PA systems under higher deforestation pressure tended to have larger funding deficits, which calls for better international funding to support nations in closing these shortfalls in the short term.

Chapter 2 – Cost-effective protection of biodiversity

- Costs associated with PAs are not uniform in the western Amazon. Lands with lower management costs (per unit area) have lower accessibility, offer opportunities to create large reserves, and overlap with indigenous lands. High opportunity costs from agriculture occur in mountain landscapes, where land is highly suitable for crop expansion combined with better accessibility to markets.

- Prioritizations that incorporate spatial data on conservation costs, involve indigenous lands, and assume international collaboration allow maximizing species representation at lower management and opportunity costs.
- Based on our modeling, the aggregated management cost of current and complementary proposed PAs far exceed the current spending but seems feasible and affordable for governments and the international community.

Chapter 3 – Andean protected areas for nature and people

- Closing species and ecosystem representation gaps in the Andes is particularly challenging; this requires many small conservation areas in places with relatively high economic and social costs.
- To address these challenges while meeting local people's needs, three actions for area-based conservation are required: integrating NCPs when planning for cost-effective expansion of biodiversity protection, diversifying administration regimes and management, and enhancing regional collaboration and international financial support.

Chapter 4 – Deforestation threats to cultural diversity

- Forest loss and degradation can precipitate processes of cultural erosion and assimilation in forest-dwellers by three pathways: (1) reducing people's interactions with their biocultural landscape, (2) increasing their exposure to dominant cultural groups, and (3) reducing cultural group sizes.
- The prospect for the survival of ~1,400 ethnolinguistic groups from tropical forests (20% of the world's diversity) is likely jeopardized by the significant expansion of forest loss towards their territories during this century.
- Guaranteeing land tenure rights of IPLCs is often a cost-effective strategy to preserve forests, mitigate climate change, and avoid cultural erosion.
- Curbing deforestation within PAs may not be sufficient to safeguard forest-dependent cultures. It is also needed that PA management ensures IPLCs' land rights, guarantees their access to natural resources and their participatory role in forest management.

Integrated conclusions

- The success of area-based conservation and the wellbeing of IPLCs are strongly connected, which calls for a better understanding of these links and their sensible consideration when planning the expansion and management of conservation areas in tropical regions.
- Five areas of actions can help to enhance the financial sustainability, ecological representation and social equity of area-based conservation in the tropics: (1)

enhancing international and private funding support and transparent financial reporting, (2) continuing the expansion of conservation areas focusing on preserving sites that efficiently improve biodiversity protection and (3) NCP retention , (4) through a combination of diverse models of PAs and OECMs, and (5) in partnerships with IPLCs.

- To deliver higher benefits for nature and people, post-2020 global agreements for area-based conservation should incorporate actions that harmonize biodiversity protection needs with the concerns and aspirations of the local human population.

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Appendices – Chapter 1

1. Protected area dataset

We gathered spatial information on the limits of the protected areas (PAs) of Latin American systems and Ecuador that were evaluated by Bovarnick et al. (2010) and Galindo et al. (2005), respectively (Tables S1 and S2). For the analysis across Latin America, we excluded Venezuela, the Guyanas, Haiti, and Belize due to lack of financial data, and Uruguay due to its reduced native forest cover within PAs already at the beginning of the study period. When possible, PA limits for each country were obtained from national PA national agencies. Otherwise, we used the Protected Planet Database (IUCN & UNEP-WCMC 2015). For some countries, we were able to analyze a higher representation of the PA systems that were assessed in the financial study (e.g., 100% of the PAs of Bolivia were included) than for others (e.g., 42% of the PAs of Peru). This is due to the different availability of spatial data on PAs in each country, and because some systems encompass many PAs without forest cover (e.g., savannas, arid ecosystems, etc.), which are beyond our objective. In general, financial estimates applied to those systems managed directly by the central government and depending on data availability, some countries could include PAs with other governance types (e.g., private, or federal PAs).

Table 1. Dataset on the PA systems of Latin American countries analyzed in this study.

Country	Spatial data source *	PAs included in the analysis (i.e., those evaluated in the financial studies, with forest cover, and available spatial data)		Funding deficit of the PA system (%)	Avoided deforestation (%)
		Number	Percentage of the entire PA system		
Argentina	National agency	30	81	21	0.7585
Bolivia	National agency	22	100	5	0.7483
Brazil	National agency	175	62	56	0.8210
Chile	National agency	92	55	49	0.7888
Colombia	National agency	44	83	20	0.5448
Costa Rica	WDPA	119	73	7	0.6989
Cuba	WDPA	47	48	33	0.5462
Dominican Republic	WDPA	39	49	54	0.4168
Ecuador	National agency	25	64	41	0.5761
El Salvador	WDPA	59	53	14	0.4665
Guatemala	WDPA	74	100	48	0.5423
Honduras	WDPA	66	71	38	-0.0796
Mexico	National agency	124	80	33	0.7177
Nicaragua	WPDA	69	57	73	0.1305
Panama	WDPA	39	46	52	0.6722
Paraguay	WDPA	27	52	87	0.7836
Peru	National agency	29	42	48	0.7437

* World Database on Protected Areas https://www.protectedplanet.net/

Table 2. Dataset on the PAs of Ecuador analyzed in detail in this study.

Protected area	Forest cover (%)	Funding deficits (%)	Avoided deforestation (%)
Antisana Ecological Reserve	42	77	0.804
Arenillas Ecological Reserve	27	89	0.964
Cajas National Park	11	33	0.681
Cayambe Coca National Park	52	59	0.669
Chimborazo Wildlife Reserve	15	54	0.801
Cofan Bermejo Ecological Reserve	81	80	0.880
Cotacachi Cayapas Ecological Reserve	59	67	0.929
Cotopaxi National Park	7	59	0.878
Cuyabeno Wildlife Reserve	75	79	0.503
El Angel Ecological Reserve	10	77	-1.755
El Condor Biological Reserve	100	100	0.971
La Chiquita Wildlife Refuge	104	99	-1.621
Limoncocha Biological Reserve	73	64	0.891
Llanganates National Park	37	66	0.871
Los Ilinizas Ecological Reserve	72	88	-0.820
Machalilla National Park	57	63	0.859
Mache Chindul Ecological Reserve	98	77	-0.143
Manglares Cayapas Mataje Ecological Reserve	27	72	0.809
Manglares Churute Ecological Reserve	44	57	0.817
Manglares el Salado Wildlife Reserve	33	99	0.995
Parque Lago National Recreation Area	42	100	-0.598
Pasochoa Wildlife Refuge	82	45	0.864
Podocarpus National Park	60	64	0.904
Pululahua Geological Reserve	83	67	0.855
Sangay National Park	40	61	0.683
Sumaco Napo-Galeras National Park	75	59	0.759
Yasuni National Park	79	75	0.879

2. Limitations of forest loss data from Global Watch Forest

Tree cover data set from Global Watch Forest (GWF) represents the most comprehensive globally and regionally available spatial information on forest loss. Thus, GWF data have been used by other studies to assess the effectiveness of PAs at preserving forests (Heino et al. 2015; Bowker et al. 2017; Potapov et al. 2017; Leberger et al. 2020). However, GWF data present important limitations to be discussed. Tree cover loss estimates from GWF have been criticized for inaccuracies in distinguishing vegetation types at the local scale (Tropek et al. 2014), such as plantations. This limitation could lead to considering losses in plantations as deforestation of natural forests, when in fact reflects the harvest of products grown explicitly for human extraction. To reduce this source of error for the analysis of PA impact within Ecuador, we excluded from the analysis all plantations by 2000 (Socio Bosque 2012). In the case of other Latin American countries, analyses have shown that the majority of tree cover loss (~90%) detected in Brazil, Peru, and Colombia for 2013-2014 was likely loss of natural forests (Petersen et al. 2016).

Global error rates of GWF at classifying tree cover data in forest loss are relatively low compared to similar data sets (Hansen et al. 2013). Thus, there is high confidence in using the data to examine trends and patterns at large scales (e.g., global, regional, national) (Weisse & Petersen 2015). Moreover, some studies have found that GWF performed almost as well as the more resource-demanding, locally calibrated data (Burivalova et al. 2015). For example, in Brazil, GWF more accurately detects forest loss than the coarser-resolution FORMA or Brazil's national-level PRODES product (Milodowski et al. 2017). However, GWF usually underestimates the rate of loss for losses driven by small-scale disturbances (Milodowski et al. 2017). Similarly, in the case of Ecuador, we found that GWF reports lower deforestation rates (39,000 Ha per year) than the national data (77,000 Ha per year) (Socio Bosque 2012), during 2000-2008. Still, we detected a strong and positive correlation (Pearson coefficient r: 0.71, $p < 0.0001$) between the extent of forest cover loss in PAs of both data set for this period. It is important to emphasize that national estimates for Ecuador also used GWF information to eliminate areas without information due to clouds.

3. Matching analysis

Comparison of matching methods

We assessed our data using Coarsened Exact Matching (CEM) and Propensity Score Matching (PSM). Comparing the two matching methods showed that PSM performed better than CEM in terms of successfully matching more treatment sites within Ecuador (73% vs 10% out of 81 733 treatment sites) and in maintaining the properties of the original treatment sites (Table 3). In other words, for all covariates, post-matching treatments were closer to pre-matched with PSM than CEM. For instance, in the case of CEM, most of the treatment sites from Andean PAs (those at high elevations) were not matched. We obtained similar results when analyzing other Latin American countries, such as El Salvador (Table 4).

Table 3. Performance comparison between PSM and CEM for the analysis of PAs within Ecuador. Green colors indicate which of PSM and CEM best matched the original characteristics of PAs.

	Treatment			Control			Absolut difference (matched and pre-matching treatment sites)	
Variables	Pre matching	CEM	PSM	Pre matching	CEM	PSM	CEM	PSM
Distance	0.52	0.28	0.43	0.13	0.27	0.38	0.245	0.094
Elevation	1442.61	587.29	1260.59	914.78	607.10	1279.19	855.320	182.024
Travel time	1415.96	1217.58	1272.06	520.51	1205.84	1161.67	198.384	143.898
Human footprint	2.12	1.49	2.37	6.48	1.70	2.62	0.625	0.251
Initial tree cover	0.42	0.44	0.42	0.37	0.44	0.41	0.016	0.004
Rain	189.69	236.00	197.66	172.16	236.97	196.04	46.307	7.962
Distance to towns	11.21	7.95	9.92	4.11	7.86	8.83	3.258	1.286
Distance to urban areas	55.19	90.31	57.76	38.35	90.33	58.66	35.118	2.567
Slope	3.25	0.83	2.95	2.62	0.85	2.94	2.427	0.303
Ecoregions	31.53	39.32	33.35	37.28	39.32	33.35	7.790	1.826

Table 4. Performance comparison between PSM and CEM for the analysis of PA system of El Salvador. Green colors indicate which of PSM and CEM best matched the original characteristics of PAs.

	Treatment			Control			Absolut difference (matched and pre-matching treatment sites)	
Variables	Pre matching	CEM	PSM	Pre matching	CEM	PSM	CEM	PSM
Distance	0.131	0.050	0.067	0.029	0.042	0.066	0.081	0.064
Initial tree cover	0.579	0.556	0.567	0.399	0.546	0.555	0.023	0.012
Travel time	111.831	84.333	104.983	88.309	85.957	111.961	27.498	6.848
Elevation	374.907	487.889	576.139	455.220	511.790	516.788	112.982	201.233
Human footprint	13.285	13.043	12.929	14.884	13.018	13.634	0.241	0.356
Rain	156.254	99.158	156.169	152.725	103.077	155.662	57.096	0.085
Human population density	44.153	4.548	47.245	417.723	4.592	99.345	39.605	3.092
Opportunity costs for agriculture	149.483	158.049	117.913	142.592	158.406	139.831	8.567	31.569
Slope	3.715	48.444	5.560	3.750	62.957	4.568	44.730	1.845

Selection of control sites

To account for potential effects of local leakage, we excluded a 5 km buffer around PAs to be selected as control sites (Schleicher et al. 2017). For the analysis of PA impact within Ecuador, we also excluded as possible control sites all PAs established after 2003 and other types of forest governance regimes, such as Protected Forests or lands under the payment for ecosystem services (*Socio Bosque* program) (data from Sistema Nacional de Información 2015). Although indigenous lands may have a strong influence on forest protection, we did not exclude them as potential control sites because they occupy a large proportion of the national territory.

For the analysis across Latin American countries, we excluded as control sites any state, private, regional, and municipal PAs established after 2000. Nevertheless, the region has a great diversity of other governance regimes not consistent across all countries (e.g., military bases in Brazil, Reserved Zones in Peru). Moreover, spatial information on indigenous lands is not homogenous or available for all countries. Thus, control sites throughout the region were unavoidably composed of different forms of governance and varying legal restrictions on natural resource extraction, which can influence deforestation rates (Schleicher et al. 2017). Therefore, future impact assessments would benefit from standardized regional data on such governance types and the level of protection they offer.

Covariates

Table 5. Set of covariates used for the matching analysis. The level of analysis refers to the two different matching analysis carried out in this study: for the PAs across Latin American countries and for PAs within Ecuador. For each scale of analysis, we used a different set based on data availability. Layers of these covariates were resampled at 1 km^2 and 0.5625 km^2 of resolution for each analysis, respectively. Covariates were also selected following the literature (Nolte et al. 2013; Carranza et al. 2014; Schleicher et al. 2017; Potapov et al. 2017; Cuenca et al. 2018).

Type of variable	Variables	Level of analysis	Description
Accessibility	Travel time to major cities (h)	PA system PA	The time needed to travel from the nearest city to a given point, through the network of roads and rivers. Data from Nelson (2008).
	Distance to the nearest town (km)	PA	Euclidian distance to the nearest town, sourced from Instituto Geográfico Militar del Ecuador (2014)
	Distance to the nearest urban area (km)	PA	Euclidian distance to the nearest urban area, sourced from Instituto Geográfico Militar del Ecuador (2014)
Agricultural suitability	Opportunity cost for agriculture (USD/ha, year)	PA system	Potential economic benefits from agricultural lands. Data from Naidoo & Iwamura (2007).
	Rain (mm)	PA system PA	Mean annual precipitation data were obtained from the WorldClim Global Climate data (~1950-2000) provided at 30 arc-seconds. Data from Hijmans et al. (2005).
Human pressure	Human population density (per km^2)	PA system	Global population data from Bright et al. (2011).
	Human footprint (index)	PA system PA	Describe levels of human impact and resource use on Earth by 1993. Data from Venter et al. (2016).
Topography	Elevation (m)	PA system PA	Elevation was based on the Shuttle Radar Topographic Mission (SRTM) 90m digital elevation. Data from Jarvis et al. (2006).
	Slope (º)	PA system PA	Derived from an elevation model at 90m resolution.
Biophysics	Initial forest cover (km^2)	PA system PA	Extent of forest cover in each pixel by 2000 in the case of Latin America, and by 2003 in the case of Ecuador. Data from Hansen et al. (2013).

Checking balance for the analysis in Ecuador

After matching, we assessed the resulting balance between the control and treatment sites across all covariates. For the analysis in Ecuador, all covariates had a standardized difference in mean below 0.25 (Schleicher et al. 2017), which indicates that covariate distributions are relatively close in the two groups (Table 6).

Table 6. Covariate balance after matching data for PAs within Ecuador, using an exact match by **ecoregions**.

Covariates	Means Treatment	Means Control	SD Control	Std. Mean Difference	eCDF Med	eCDF Mean	eCDF Max
PSM distance	0.4268	0.383	0.24	0.1598	0.0427	0.0513	0.1014
Elevation	1260.588	1279.193	1296.27	-0.0133	0.0072	0.0124	0.1895
Travel time	1272.061	1161.667	692.86	0.1424	0.0317	0.0316	0.0708
Human footprint	2.3679	2.6158	3.23	-0.0927	0.022	0.0228	0.0477
Initial tree cover	0.4244	0.4148	0.138	0.07	0.0372	0.0354	0.3241
Rain	197.6562	196.0375	102.172	0.0203	0.0662	0.0762	0.1781
Distance to towns	9.9212	8.8307	7.46	0.1517	0.0051	0.0152	0.1418
Distance to urban areas	57.7581	58.6618	46.362	-0.0229	0.0224	0.0255	0.0569
Slope	2.9502	2.9383	4.139	0.0028	0.0082	0.0102	0.1347
Ecoregions	33.3541	33.3541	16.53	0	0	0	0

The presence of indigenous lands can influence deforestation rates. However, a matching analysis using an exact match by ecoregions and indigenous lands, simultaneously, resulted in a low percentage of successfully matched treatment sites. Therefore, in an additional analysis for Ecuador, we used an exact match by the presence or absence of indigenous lands, instead of an exact match by ecoregion. This analysis performed well in terms of the balance of covariates (Table 7), but it was slightly inferior in maintaining the properties of the original treatment sites (Table 8). Thus, we based our main results on the analysis using an exact match by ecoregions. Then, we compared these results with those from an exact match by indigenous lands (see 5.3. Testing the sensitivity of model selection).

Table 7. Covariate balance after matching data for PAs within Ecuador, using an exact match by the presence of **indigenous lands**.

Variables	Means Treatment	Means Control	SD Control	Std. Mean Difference	eCDF Med	eCDF Mean	eCDF Max
PSM Distance	0.42	0.38	0.25	0.13	0.04	0.04	0.07
Elevation	1278.34	1371.87	1248.25	-0.07	0.03	0.04	0.24
Travel time	1252.25	1127.65	719.94	0.16	0.03	0.04	0.08
Human footprint	2.47	2.84	3.09	-0.14	0.04	0.05	0.10
Initial tree cover	0.42	0.42	0.13	0.00	0.02	0.04	0.31
Rain	195.28	186.47	99.90	0.11	0.09	0.08	0.15
Distance to towns	9.79	8.37	7.62	0.20	0.01	0.02	0.17
Distance to urban areas	58.64	52.94	43.11	0.14	0.04	0.04	0.09
Slope	2.98	3.45	4.35	-0.11	0.04	0.04	0.14
Indigenous lands	0.50	0.50	0.50	0.00	0.00	0.00	0.00

Table 8. Performance comparison between the matching analysis using an exact match by the presence of indigenous lands (IL) and by ecoregions of Ecuador. Green colors indicate which analysis best matched the original characteristics of PAs.

	Treatment			Control			Absolut difference (matched treat and pre-matching treat)	
	Pre matching	Match by IL	Match by ecoregions	Pre matching	Match by IL	Match by ecoregions	Match by IL	Match by ecoregions
Distance	0.52	0.42	0.43	0.13	0.38	0.38	0.10	0.09
Elevation	1442.61	1278.34	1260.59	914.78	1371.87	1279.19	164.27	182.02
Travel time	1415.96	1252.25	1272.06	520.51	1127.65	1161.67	163.71	143.90
Human footprint	2.12	2.47	2.37	6.48	2.84	2.62	0.35	0.25
Tree cover	0.42	0.42	0.42	0.37	0.42	0.41	0.001	0.004
Rain	189.69	195.28	197.66	172.16	186.47	196.04	5.58	7.96
Distance to towns	11.21	9.79	9.92	4.11	8.37	8.83	1.42	1.29
Distance to urban areas	55.19	58.64	57.76	38.35	52.94	58.66	3.45	2.57
Slope	3.25	2.98	2.95	2.62	3.45	2.94	0.28	0.30

Checking balance for the analysis in Latin American countries

Matching analysis was carried out separately by each country, using the same nine covariates (Table 3). This approach allows us to compare the impact of the PA systems across all countries while controlling for the same bias of PA location and effect of the treatment. However, four countries showed covariates with a standardized mean difference above the recommended 0.25 (Table 9), which means that in these cases, treatment sites were matched to control sites with important differences in the covariates. Thus, we ran a second type of matching analysis in which we chose the covariates that maximized the number of covariates with standardized differences in mean below 0.25 (Schleicher et al. 2017) (Table 9). As a result, three out of four countries showed an increase in the impact of their PA systems. Thus, we compared model selection results from both analyses (see 5.3. Testing the sensitivity of model selection).

Table 9. Comparison of the balance after matching data under two different approaches of covariate selection across Latin American PA systems.

Country	Matching using the same covariates for all countries (nine covariates)		Matching using different sets of covariates, with standardized difference lower than 0.25	
	Covariates with a standardized difference higher than 0.25	Avoided deforestation %	Covariates excluded	Avoided deforestation %
Argentina	Initial tree cover, rain	75.8	Initial tree cover, rain, elevation	82.4
Bolivia	0	74.8		74.8
Brazil	0	82.1		82.1
Chile	0	78.8		78.8
Colombia	0	54.5		54.5
Costa Rica	0	69.9		69.9
Cuba	Population density	54.6	Population density	56.7
Dominican Republic	Population density	41.7	Population density	37.7
Ecuador	0	57.6		57.6
El Salvador	0	46.6		46.6
Guatemala	Initial tree cover	54.2	Rain, slope	61.6
Honduras	0	-7.9		-7.9
Mexico	0	71.8		71.8
Nicaragua	0	13		13
Panama	0	67.2		67.2
Paraguay	0	78.4		78.4
Peru	0	74.4		74.4

Matching success of treatment sites

The percentage of treatment sites that were successfully matched to control sites varied among countries (Fig. 1) and among PAs within Ecuador (Fig. 2). This result means that countries such as Paraguay and Peru (100% of treatment sites with a match) had a better representation of their PAs in the assessment of impact than Honduras, Guatemala, or Costa Rica (with ~ 50% of treatment sites with a match). For these last countries, we explored the characteristics of not matched treatment sites. We found that, overall, they belong to PAs with low human pressures. For example, these sites exhibited higher inaccessibility (measured as travel time to cities) than matched sites (Fig. 3). The low availability of proper controls for these PA sites in these countries might be due to the absence of remote areas in the unprotected landscape, or because these PAs protect remnants of forests that already disappeared outside PAs. Similarly, PAs in the Andean mountain range of Ecuador had the lowest percentages of matched treatment sites (e.g., Cotopaxi and Llanganates National Parks). Probably, these PAs harbor remnants of Andean forests, which in Ecuador have been exploited by humans for centuries.

The bias of successfully matching towards treatments with high accessibility might have underestimated the impact of the PA systems, especially for Guatemala or Costa Rica, whose treatment sites without a match show lower deforestation rates than the matched ones (Fig. 4). To test the influence of this bias in our results, we included the percentage of successfully matched sites as a potential explanatory variable of the impact in additional models (see 5.3. Testing the sensitivity of model selection).

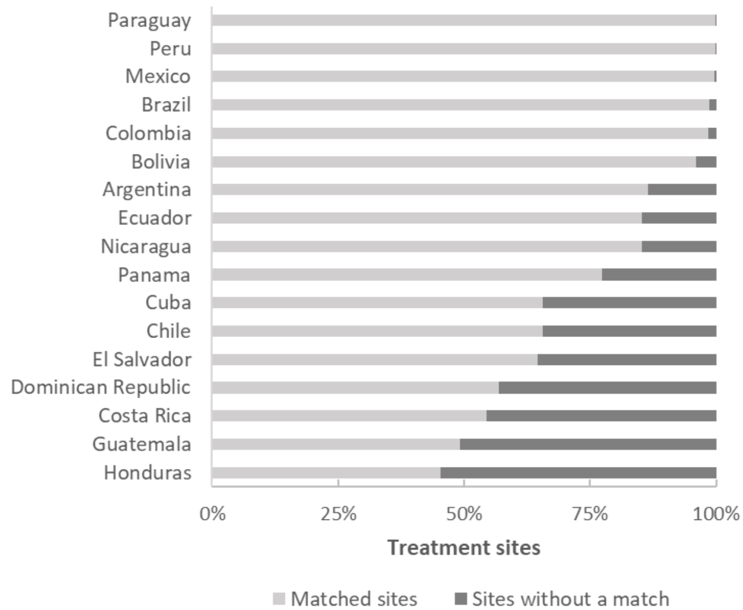

Figure 1. Matching success of treatment sites of the PA system of each country in Latin America.

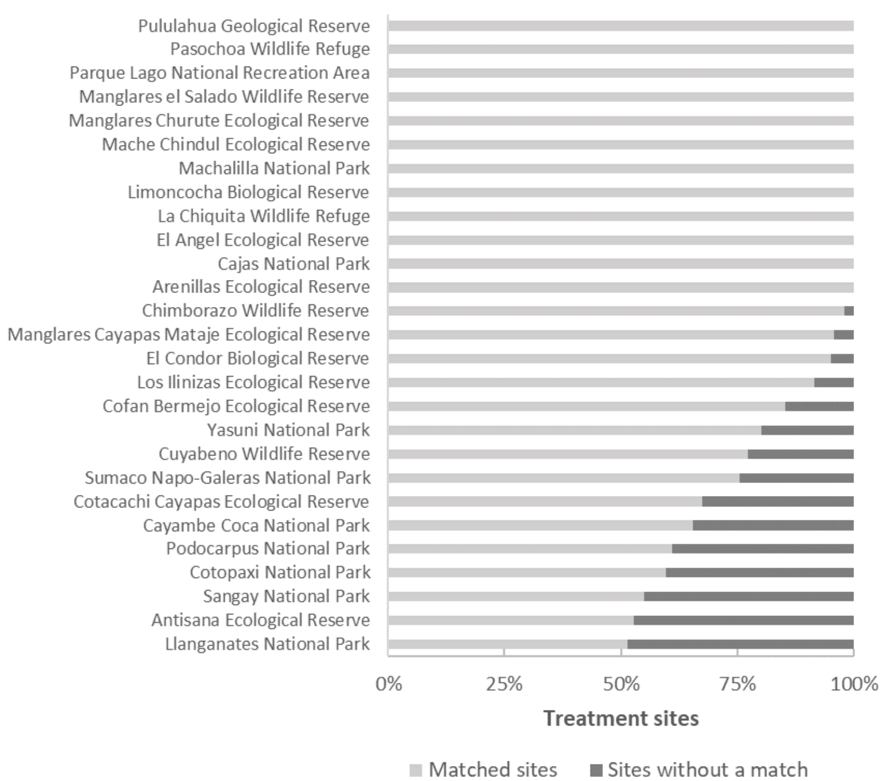

Figure 2. Matching success of treatment sites of PAs within Ecuador.

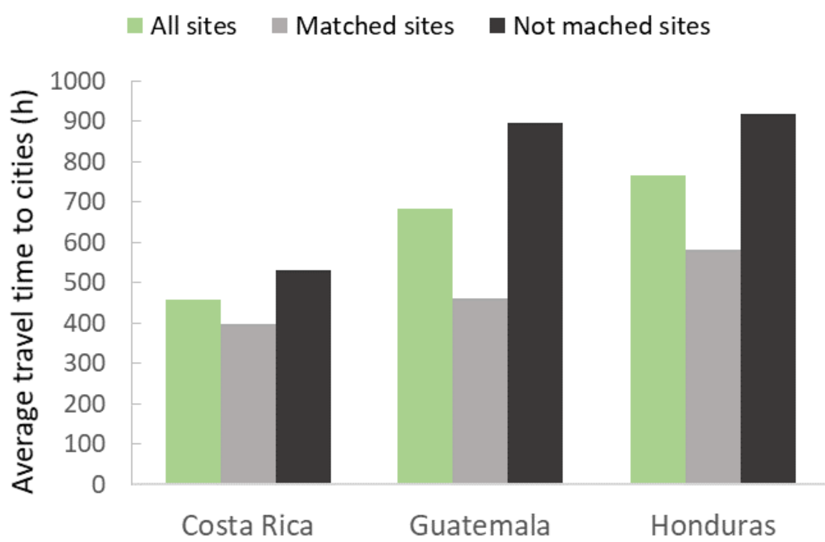

Figure 3. Comparison of the average travel to time to cities among (1) all treatment sites, (2) treatment sites without a match and (3) treatment sites with a match, for Costa Rica, Guatemala, and Honduras.

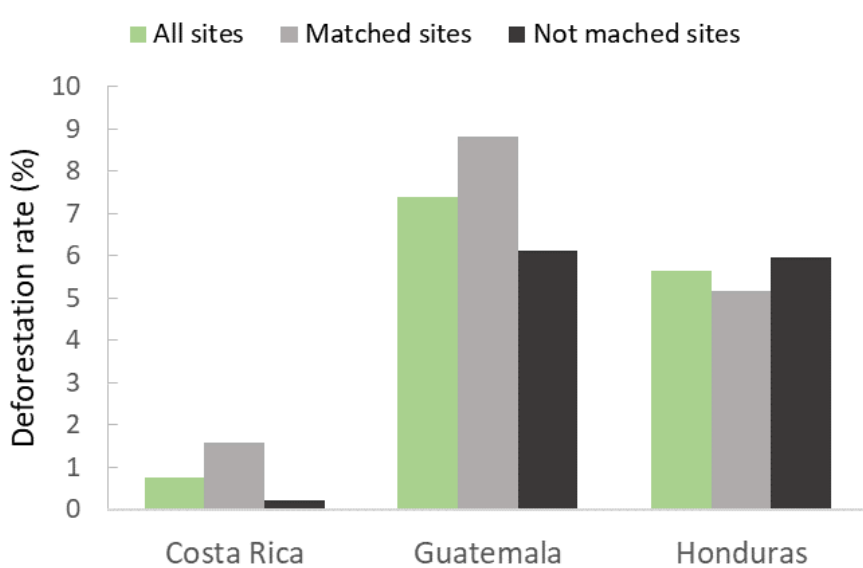

Figure 4. Comparison of total deforestation rates (%) (2000-2010) cities among (1) all treatment sites, (2) treatment sites without a match and (3) treatment sites with a match, for Costa Rica, Guatemala, and Honduras.

4. Explanatory variables

Table 10. Description of the explanatory variables tested in the models of PA impact. The level of analysis refers to the different modeling carried out in this study: across Latin American PA systems, and for PAs within Ecuador. The expected direction of the relationship between the variable and avoided deforestation is indicated as positive (+), negative (-), or both.

Variable	Scale of analysis	Description and source	Importance for the PA impact at avoiding deforestation	Expected relationship
Socio-economic variables and governance				
Gross Domestic Product (GDP) at purchasing power parity (USD) of countries	PA system	Average from 2000-2010. Data from The World Bank.	GDP improves conservation capacity to respond to increased pressure, but it could also be associated with higher pressures on the environment (Balmford et al. 2003; Barnes et al. 2016)	+ or -
GDP growth (%) of countries	PA system	From 2000 to 2010. Data from The World Bank.		
Human Development index of countries HDI (0-1)	PA system	Average from 2000 to 2010. HDI measures income, health, and education. Data from the United Nations Development Program (http://hdr.undp.org).	More developed nations might have more money and capacity available for PAs (Barnes et al. 2016; Geldmann et al. 2019).	+
Percentage of rural population (%) of countries	PA system	Average from 2000 to 2010, for each country. Data from The World Bank.	Some studies have found that rural population is not associated with forest loss (DeFries et al. 2010), while others suggest that depopulating rural landscapes will reduce pressures on forests (Wright & Muller-Landau 2006).	+ or -
Corruption index perception (0–10) of countries	PA system	For 2010. Perceived levels of public sector corruption. High values associated with low corruption perceptions. Data from Transparency International (http://www.transparency.org/).	High levels of corruptions, week democracies and absence of the rule of law are associated with deficiencies in management, funding, transparent administration, and enforcement of PAs (Heino et al. 2015; Abman 2018).	-
Polity index (-10 – 10) of countries	PA system	For 2010. Low polity index corresponds with low democracy. Data from Marshall & Gurr (2014).		+
Rule of law (-2.5 – 2.5) of countries	PA system	For 2010. Perceptions of the confidence in the rules of society, contract enforcement, property rights, the police, the courts. Data from The Worldwide Governance Indicators project (https://info.worldbank.org).		+
Poverty index of the municipality (0-1)	PA	Data from SENPLADES (2007).	While in some cases, land areas with high poverty appear to be cleared more rapidly (Kerr et al. 2004), there is yet no consensus regarding the link between both aspects.	+ or -

Direct and indirect human pressure on forest				
Human population density (per km²)	PA system	Average from 2000 to 2010, for each country. Data from The World Bank.	Large human population and growth are likely to increase pressure on forests (Jha & Bawa 2006; Wright & Muller-Landau 2006).	-
	PA	Average human population density in buffer zones (10 km) of PAs. Population data from SENPLADES (2007).		
Human population density growth (%)	PA system	From 2000 to 2010, for each country. Data from The World Bank.		-
Population growth (%)	PA system	Average from 2000 to 2010, for each country. Data from The World Bank.		-
Percentage of agricultural land (%)	PA system	Average from 2000 to 2010, for each country. Data from The World Bank.	Globally assessment showed forest loss having a strong connection with agricultural land extent (Heino et al. 2015).	-
Travel time to cities (h)	PA system	Average estimates of the PAs. The time needed to travel from the nearest city to a given point, through the network of roads and rivers. Data from Nelson (2008).	Remote areas are less likely to suffer a stronger human pressure on forests (Bruner et al. 2004).	+
	PA			
Opportunity cost for agriculture of the PAs (USD per Ha, year)	PA system	Average estimates of the PAs. Economic benefits from agricultural lands. Data from Naidoo & Iwamura (2007).	High opportunity costs indicate major incentives to agricultural expansion towards these PAs.	-
	PA			
Slope (º)	PA	Average estimates of the PAs. Derived from an elevation model at 90m resolution. Data from (Jarvis et al. 2006).	Protected areas with greater average slope may be more complex to access, decreasing the threats and the costs of surveillance (Joppa & Pfaff 2009).	+
Elevation (m)	PA	Average elevation of each PA.	PAs at higher elevations tend to be further from areas of high human population densities and are often less agriculturally suitable (Joppa & Pfaff 2009).	+
Protected area perimeter under pressure (%)	PA	Percentage of the PA perimeter under zones with high human footprint index (top quintile of the country). Data from Venter et al. (2016).	This indicator is considered by the Ecuadorian government to evaluate the human pressure on PAs (Galindo et al. 2005).	-
PA design and management				
PA size (km²)	PA system	Average size of forested PAs that were included in the analysis.	Large PAs are associated with lower pressures towards their core (Cantú-Salazar & Gaston 2010). The management costs per unit area of larger PAs are usually lower than for small PAs.	+
	PA	Total size of each PA.		
Perimeter-area ratio	PA	PA perimeter in km divided by PA area in km².	PAs with boundaries are more prone to edge effects (Barnes et al. 2016)	-
Funding deficits for management (%)	PA system	Data from Bovarnick et al. (2010)	Without adequate funds, PAs cannot conserve biodiversity or provide healthy functioning ecosystems (Bovarnick et	-
	PA	Data from Galindo et al. (2005)		

			al. 2010)	
Type of management (0/1)	PA	0: Strict PAs (I-II), and 1: PAs that allow use (III-VI), according to IUCN categories.	PAs that allow use may experience more pressure on their forests. However, studies have shown mixed results for the impact of PAs when comparing strict vs multi-use reserves (Nelson & Chomitz 2011; Nolte et al. 2013).	0: + 1: -
Years since establishment	PA	Since the PA official declaration.	Newer PAs may require more efforts (e.g., funding, monitoring) to consolidate their effectiveness (Barnes et al. 2016).	+
Overlap of PAs with indigenous lands (0: no; 1: yes)	PA	Data from Minister of Environment of Ecuador (http://suia.ambiente.gob.ec/) and Kingman (2007)	Some lands managed by indigenous people have proved to be effective in preserving forests (e.g., at avoiding forest fires).	0: - 1: +
Management plan by 2007 (0: no, 1: yes)	PA	According to Kingman (2007)	PA management plans act as the foundation for all cost estimates and key activities required to achieve management objectives (Bovarnick et al. 2010).	0: - 1: +

5. Model selection

Sub-optimal models

Table 11. Summary output for the three most parsimonious set of models for explaining the impact (a) of PAs within Ecuador, (b) of PA systems among Latin American countries, funding deficits of (c) PAs within Ecuador, and (d) of PA systems of Latin America. 2nd order polynomials are indicated by superscript. These tables present the modeling results for the variable of impact transformed as "permitted deforestation" (i.e., one minus impact). Thus, a variable with positive estimate indicates an increase in PA ineffectiveness. Significance of regression coefficients: *** p < 0.001, ** p < 0.01, * p < 0.05

(a) "Permitted" deforestation (1-impact) among the PAs within Ecuador

Model rank	Standardized coefficients of selected variables				AICc	ΔAICc	Weight
	Funding deficits	Overlap with indigenous lands	Perimeter under pressure	Opportunity costs for agriculture			
1	0.028 *	1.042 *	0.018 *		11.74	0	0.366
2	0.028 *			0.014 *	11.98	0.236	0.325
3	0.025 *	0.781		0.015 *	12.08	0.337	0.309

(b) "Permitted" deforestation (1- impact) among the national PA systems in Latin America

Model rank	Standardized coefficients of selected variables			AICc	ΔAICc	Weight
	HDI	Average PA size	Opportunity costs for agriculture			
1	-5.295 ***	-0.00001*		-13.41	0	0.485
2	-5.444 **			-12.31	1.108	0.279
3	-5.220 **		0.003	-11.97	1.445	0.236

(c) Funding deficits (proportion) among the PAs within Ecuador

Model rank	Standardized coefficients of selected variables					AICc	ΔAICc	Weight
	Management plan	Slope	Elevation	Years	phi			
1	-1.291 ***	-0.044 **			13.26 ***	-36.818	0	0.457
2	-1.230 ***		-0.0002 **		13.06 ***	-36.243	0.575	0.343
3	-0.953 **		-0.0002 **	-0.018	14.07 ***	-35.161	1.656	0.2

(d) Funding deficits (proportion) among the national PA systems in Latin America

Model rank	Standardized coefficients of selected variables					AICc	ΔAICc	Weight
	Deforestation rate in control sites	Deforestation rate in control sites ^{2}	GDP growth	Corruption	phi			
1	2.453 **	-1.539 *	0.008 *		7.37 **	-0.216	0	0.448
2	1.888 *	-0.945			5.58 **	0.284	0.5	0.349
3				-0.193	4.2 **	1.377	1.592	0.202

Scatter plots between response and explanatory variables

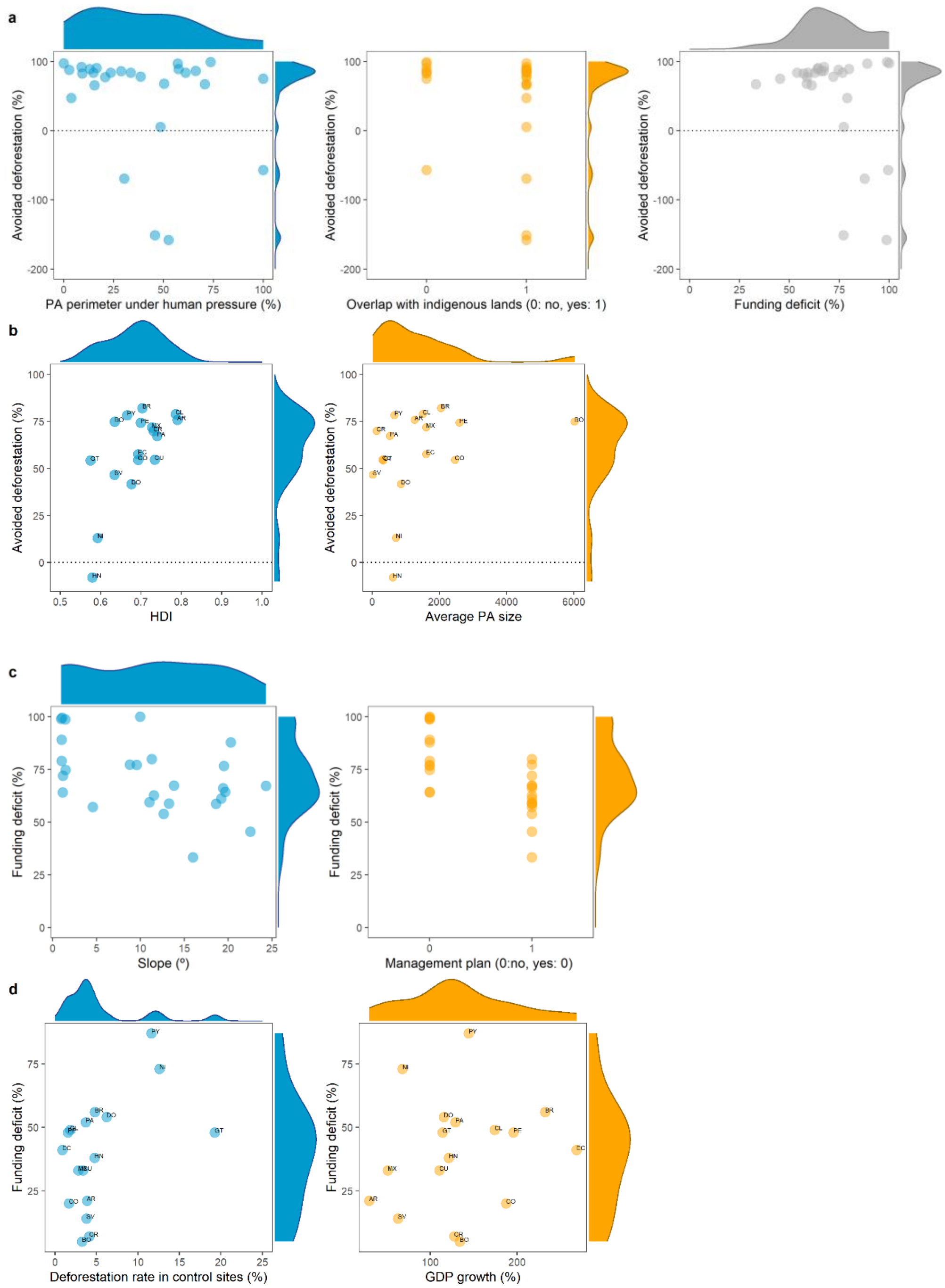

Figure 5. Plots of the selected explanatory variables in the most parsimonious model for (a) the impact of PAs within Ecuador, (b) the impact of national PA systems of Latin American countries, (c) funding deficits of the PAs within Ecuador and (d) funding deficits of national PA systems of Latin America.

Testing the sensitivity of model selection

We tested the sensitivity of our model selection by running additional models under different scenarios. Overall, results from additional model selection were consistent with those reported in our study (Table 12 and Table 13). HDI remains as an important explanatory variable of avoided deforestation among the PA systems, with similar parameter estimates.

Table 12. Summary output for the most parsimonious models (AICc) when testing additional scenarios of modeling for the impact of PAs within Ecuador.

Scenario	Response variable	Description	Selected explanatory variables	Standardize coefficients	Explained variance
(a)	(1- impact) of the PAs within Ecuador	Impact calculation from a matching analysis that used an exact match by the presence of indigenous lands (see Table 7).	Overlap with indigenous lands (%) Funding deficits (%) Opportunity costs for agriculture ($/ha, year)	-0.564 *** -0.305 * -0.401 **	0.59
(b)	(1- impact) of the PAs within Ecuador	Inclusion of matched treatment sites (%) of each PA as a possible explanatory variable (see Fig. 2).	Overlap with indigenous lands (%) Funding deficits (%) Opportunity costs for agriculture ($/Ha, year)	1.520 ** 0.055 * 0.028 ***	0.55

Table 13. Summary output for the most parsimonious models (AICc) when testing additional scenarios for modeling the impact and funding deficits of PA systems in Latin America.

Scenario	Response variable	Description of the model	Selected explanatory variables	Estimates	Explained variance
(a)	(1- impact) of the PA systems	Inclusion of matched treatment sites (%) of each PA system as a possible explanatory variable (see Fig. 1).	HDI Matched treatment sites (%)	-4.68 ** -0.0097 *	0.62
(b)	(1- impact) of the PA systems	Impact estimate based on the matching analysis that used a different set of covariates for each country (see Table 9).	HDI Average PA size	-5.295 ** -0.00001*	0.62
(c)	(1- impact) of the PA systems	Removing Bolivia from input data since it was identified as a possible outlier according to Cook's distance leverage plots of the model residuals.	HDI	-5.823 ***	0.58
(d)	Funding deficit of the PA systems	Removing Guatemala from input data since it was identified as a possible outlier according to Cook's distance leverage plots of the model residuals.	Deforestation rates in control sites (%) Deforestation rates in control sites (%) ^{2} GDP growth (%)	2.94 *** 0.593 0.0061	0.5

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Appendices – Chapter 2

1. Extended Methods

Data used for the management-cost models

Table 1. Protected areas of the western Amazon with information on their costs for an effective management (basic scenario). PN: Parque Nacional, RB: Reserva Biológica, RE: Reserva Ecológica, RPF: Reserva de Produccion Faunística, BP: Bosque Protector, RC: Reserva Comunal, RN: Reserva Natural, SH: Santuario Histórico, SN: Santuario Nacional.

Country	Protected Area	Management cost (US$/year, km^2)	Area (km^2)
Colombia	PN LA PAYA	7,02	5128,29
Colombia	PN PURACÉ	40,56	947,83
Ecuador	PN Cayambe Coca	117,00	4.082,85
Ecuador	PN Llanganates	75,83	2.211,45
Ecuador	PN Podocarpus	259,40	1.384,93
Ecuador	PN Sangay	85,48	4.867,29
Ecuador	PN Sumaco Napo-Galeras	138,16	2.061,62
Ecuador	PN Yasuní	42,49	10.152,13
Ecuador	RB El Cóndor	248,38	79,04
Ecuador	RB Limoncocha	2.302,31	28,09
Ecuador	RE Antisana	364,10	1.205,81
Ecuador	RE Cofán Bermejo	377,21	550,26
Ecuador	RPF Cuyabeno	46,52	5.852,36
Peru	BP Alto Mayo	115,92	1.777,50
Peru	BP de Pagaibamba	9.156,47	20,31
Peru	BP de San Matias San Carlos	198,40	1.493,24
Peru	BP Pui Pui	885,28	545,05
Peru	PN Alto Purús	30,43	25.147,75
Peru	PN Bahuaja Sonene	23,12	11.020,66
Peru	PN Cordillera Azul	71,79	13.531,98
Peru	PN Cutervo	1.871,69	82,38
Peru	PN del Manu	62,18	16.985,55
Peru	PN Güeppí-Sekime	191,90	2.036,29
Peru	PN Ichigkat Muja-Cordillera del Cóndor	350,86	885,22
Peru	PN Otishi	94,45	3.059,73
Peru	PN Río Abiseo	324,17	2.724,08

Peru	PN Tingo María	3.783,39	47,78
Peru	PN Yanachaga-Chemillén	518,14	1.136,14
Peru	RC Amarakaeri	202,42	4.038,14
Peru	RC Ashaninka	203,17	1.844,67
Peru	RC El Sira	63,33	6.164,17
Peru	RC Huimeki	206,03	1.412,34
Peru	RC Machiguenga	203,17	2.189,06
Peru	RC Purus	202,56	2.026,43
Peru	RC Tuntanain	804,20	949,87
Peru	RC Yanesha	836,84	333,89
Peru	RN Allpahuayo Mishana	804,20	580,69
Peru	RN Pacaya Samiria	48,68	21.702,47
Peru	RN Tambopata	158,58	2.802,35
Peru	SH Machupicchu	2.534,39	373,03
Peru	SN de Ampay	4.794,42	38,53
Peru	SN Megantoni	203,17	2.158,69
Peru	SN Pampa Hermosa	4.453,11	115,44
Peru	SN Tabaconas Namballe	823,07	322,74

Table 2. Data source of the predictor variables (attributes of protected areas) used in the modeling of management costs.

<table>
<tr><th>Predictor variables</th><th>Source</th></tr>
<tr><td>Size</td><td rowspan="4">Shapefiles and database of protected areas were obtained from national agencies:
Colombia (Parques Nacionales de Colombia 2015), Ecuador (Instituto Geográfico Militar del Ecuador 2014) and Peru (SERNANP 2015).</td></tr>
<tr><td>Management objective (IUCN categories)</td></tr>
<tr><td>Years since establishment</td></tr>
<tr><td>Distance to the nearest protected area</td></tr>
<tr><td>Inaccessibility</td><td>(Nelson 2008)</td></tr>
<tr><td>Human population density</td><td>(Bright et al. 2012)</td></tr>
<tr><td>Human intervention (Human Footprint)</td><td>(Sanderson et al. 2002)</td></tr>
<tr><td>Distance to villages</td><td>Map of villages:
Ecuador (Instituto Geográfico Militar del Ecuador 2014), Peru (Secretaria de Gobierno Digital 2015), Colombia (DANE 2015)</td></tr>
<tr><td>Slope</td><td>Derived from an elevation model (Jarvis et al. 2006)</td></tr>
<tr><td>Presence of indigenous lands</td><td>(RAISG 2012) and Instituto del Bien Común, Peru (http://www.ibcperu.org/)</td></tr>
<tr><td>Overlap degree with operative oil blocks</td><td>Colombia: (Agencia Nacional de Hidrocarburos 2014)
Ecuador: (Secretaría de Hidrocarburos del Ecuador 2013)
Perú: (Perupetro 2014)</td></tr>
</table>

Species dataset

We used a set of species as biodiversity surrogates. To achieve maximum representation of biodiversity, we tried to include the largest possible number of species from several taxonomic groups (terrestrial vertebrates and plants).

Species occurrences were obtained from specimen databases (GBIF 2014; Vertnet 2014) of the following museums: Missouri Botanical Garden Tropicos Database, American Museum of Natural History; Academy of Natural Sciences of Drexel University; The University of Colorado Museum of Natural History; Delaware Museum of Natural History; Denver Museum of Nature & Science; Museum of Natural History University of Kansas; Natural History Museum of Los Angeles County; Louisiana State University Museum of Natural Science; Museum of Comparative Zoology-Harvard University; Michigan State University Museum; The Museum of Vertebrate Zoology at Berkeley; Oklahoma Museum of Natural History; Royal Ontario Museum; Santa Barbara Museum of Natural History; San Diego Natural History Museum, Alabama Museum of Natural History; The University of Arizona Museum of Natural History; Museum of Zoology, University of Michigan; Museum of the University of Nebraska State Museum; Smithsonian Institution National Museum of Natural History; University of Washington Burke Museum of Natural History and Culture; Yale Peabody Museum of Natural History.

See the list of target species on: https://ars.els-cdn.com/content/image/1-s2.0-S0006320718316264-mmc2.xlsx

Species distribution models

To construct species distribution models (SDMs), we used Maxent, a machine-learning algorithm based on the principle of maximum entropy. Maxent is an adequate technique for our goal because it performs adequately when modeling presence-only occurrence data with low sample sizes, and with moderate errors in their georeferencing (Elith et al. 2006; Graham et al. 2008). Thus, based on studies about Maxent performance with low sample sizes (Pearson et al. 2007), only species with five or more data records were modeled, excluding species with high uncertainty and obvious errors in their locality records. Moreover, for a better sampling of the species environmental niche, these occurrences were obtained for the bounding-box delimited by -79.67/ -66.79 decimal degrees of longitude and 4.99/ -14.52 degrees of latitude, thus incorporating records from a large extent of the Andean range.

We used 11 of the 19 Worldclim1.4 bioclimatic variables as ecological predictors (bio 01, bio 02, bio 04, bio 05, bio 06, bio 07, bio 10, bio 11, bio 12, bio 13, bio 16), at a 1 km^2 spatial resolution. The other bioclimatic variables we excluded from the modeling process because they show unlikely climatic patterns over the western Amazon.

Species occurrence records in the western Amazon are biased towards main roads, large river and urban centers. To reduce the impact of this bias in the modeling, SDMs were constructed using their presence localities as background data. Thus, these backgrounds reflect the same sample selection bias as the occurrence data, improving the performance of the SDMs (Phillips et al. 2009).

SDMs were developed with Maxent 3.3.3e setting the convergence threshold to 10−5, maximum iterations to 500, and the regularization parameter to ‘auto’. We discarded species’ SDMs that had AUC values below 0.7 (Elith and Leathwick 2007). To reclassify the 0.0–1.0 suitability map into presence/absence areas, we used the Maximum Training Sensitivity Plus Specificity threshold, which minimizes the false-positive errors that may identify reserve areas that do not actually contain the target species (Jiménez-Valverde and Lobo 2007).

Other variables, such as the soil properties, are important predictor of plant species range in Amazonia. However, and although digital soil data have recently become available, they have limitations when they are used to infer species edaphic niches, such as (1) insufficient resolution and thematic accuracy, (2) georeferencing problems, and (3) absence of relevant variables (Moulatlet et al. 2017).

SDM algorithms generally indicate the geographic region with the appropriate set of abiotic factors for each species, but there are other factors influencing their distributions, such as biotic interactions, dispersal abilities or geographical barrier, and biographic history that are not addressed by the models (Soberón and Peterson 2005). Therefore, to obtain more accurate approximations of species distributions, we evaluated the species models by comparing them with species distributions in the literature Thus, when necessary, we removed areas of over-prediction and discarded models whose distributions were very different to those reported by other sources (e.g. (IUCN 2014; Ridgely and Greenfield 2007; Tirira 2007).

Using SDMs for systematic conservation planning as important caveats. For example, SDMs may generate false species absences due to the bias sampling, missing the opportunity to protect other valuable areas for the biodiversity (Lessmann et al. 2016). Still, SDMs usually provide more realistic outcomes than species geographic ranges (Carvalho et al. 2010; Rondinini et al. 2006), especially at fine geographic scales (Pineda and Lobo 2012). For example, in contrast to point maps and geographic ranges, SDMs improve reliability of species distribution estimates by minimizing both commission (false species presences) and omission errors (false species absences) in the estimated distributions

(Bombi et al. 2011). However, we recommend that any initiative for protecting the identified priority areas should first conduct proper field validations and rapid biological inventories.

Species conservation targets

Targets were setted in order to favor the increase of protection for the most vulnerable species, and to generate achievable solutions in terms of the extent of the priority areas and resources needed. Specifically, targets were selected through a sensitivity test analysis. We first tested targets of 100% for restricted-range species and 10% for species with wide ranges, following Rodrigues et al. (2004). For species with ranges of intermediate distribution size, the target was interpolated linearly between the target extremes. However, given the high number of endemic species across the western Amazon, protecting the 100% of these species involved unachievable goals for many species and require unfeasible large conservation areas across the region (~ 60 % of the Andean Amazon, for example). Thus, in order to produce more feasible proposals, small-ranged species received maximal targets of 50% (for non-threatened species) and 75% (for threatened species). At the same time, we changed the minimal conservation target from 10% to 5% for species that have wide ranges.

2. Extended Results

Species representation in current protected areas of the western Amazon

Species richness maps for the western Amazon, which show the number of species by ~1 km^2, were generated for all species and insufficiently protected species, by summing all individual SDMs (Fig 1).

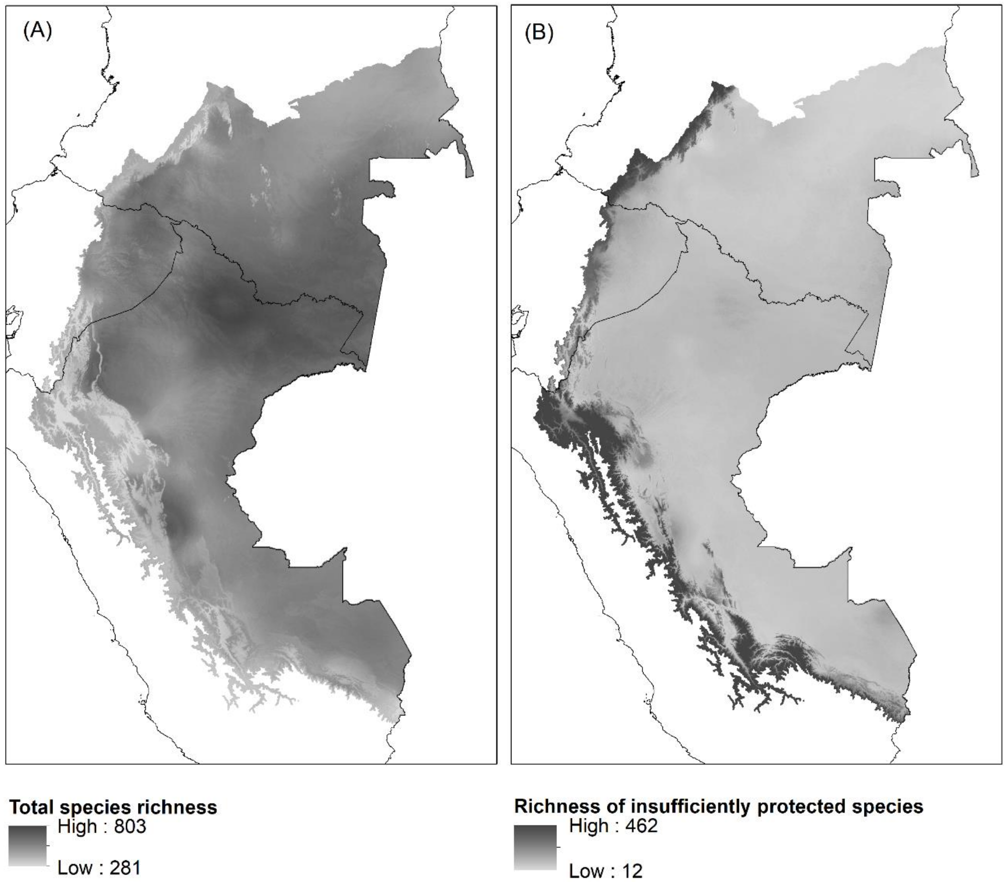

Figure 1. Richness of all species evaluated by our study (A), and richness of insufficiently protected species according to established conservation targets (B).

Identification of cost-effective areas

Table 1. Land conservation costs for priority areas in the western Amazon. Annual costs for an effective management were estimated from the General model, whereas opportunity costs correspond to adapted profits from agriculture.

Amazon countries	**Cost for priority conservation areas (US$ millions per year)**				
	Management			**Opportunity**	**Total**
	Estimate	95% LCL	95% UCL		
Colombia	13.5	9.7	20.3	9.8	23.3
Ecuador	4.5	3.1	6.6	3.7	8.2
Peru	53.4	35.1	81.2	15	68.4
Western Amazon	71.4	47.9	108.1	28.5	99.9

* LCL: Lower Control Limit; UPL: Upper Control Limit (UCL)

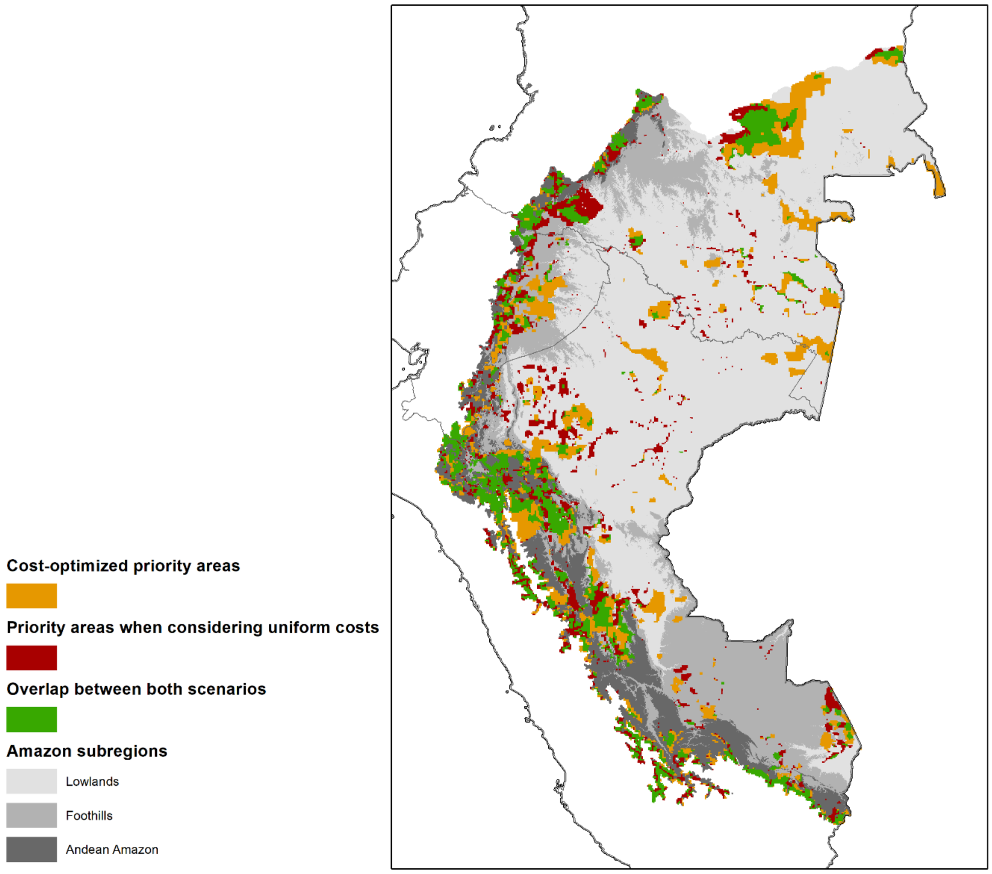

Figure 2. Spatial overlap between the priority areas selected in the cost-optimized scenario and in the cost-uniform scenario.

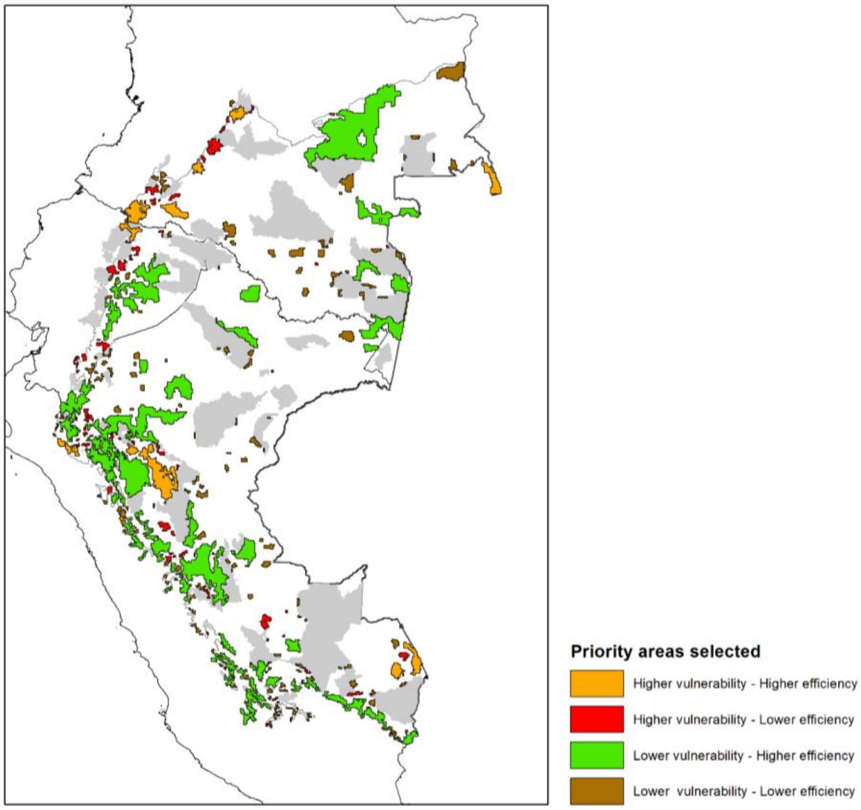

Figure. 3. Vulnerability and efficiency of the cost-optimized priority areas. Vulnerability was measured as the risk of forest loss according to the agriculture expansion and road projects in the western Amazon (Soares-Filho et al. 2006). All priority areas of high vulnerability had an average probability of deforestation higher than 0.3 (values ranged between 0-1). The 60 most efficient priority areas were considered of higher efficiency.

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Appendices – Chapter 3

1. List of reviewed studies on conservation planning that include the Andes.

Reference	Study area
Cuesta et al. 2009	Montane forests of tropical Andean countries
Galindo et al. 2009	Andes and Amazon foothills of Colombia
Josse et al. 2009	Tropical Andes
Young et al. 2009	Eastern Andean slope of Peru and Bolivia
Londono-Murcia et al. 2010	Andes of Colombia and Ecuador
Londono-Murcia & Sanchez-Cordero 2011	Tropical Andes of Colombia and Ecuador
Pliscoff & Fuentes-Castillo 2011	Chile
Thomassen et al. 2011	Ecuador
Delgado-Jaramillo 2013	Venezuela
Ramirez-Villegas et al. 2012	South America
Swenson et al. 2012	Eastern Andean slopes in Peru and Bolivia
Velasquez-Tibata et al. 2012	Andes and Tumbes-Choco-Magdalena hostspot of Colombia
Durán et al. 2013	Chile
Mateo et al. 2013	Ecuador
Cadima et al. 2014	Bolivia
Ferretti et al. 2014	Argentina
Godoy-Buerki et al. 2014	Southern Central Andes of Argentina
Lessmann et al. 2014	Ecuador
Ocampo-Penuela & Pimm 2014	Western Andes of Colombia
Avalos & Hernández 2015	Andes of Boliva and Peru
Cuesta et al. 2015	Ecuador
Fajardo et al. 2014	Peru
Lessmann et al. 2019	Eastern Andean slopes and Amazon forests in Colombia, Ecuador, and Peru
Schutz 2015	Chile
Young et al. 2015	Tropical Andes biodiversity hotspot
Banda-R et al. 2016	Neotropical dry forests
García Márquez et al. 2016	Central Colombia
Asner et al. 2017	Andean Amazon of Peru
Cuesta et al. 2017	Ecuador
Curti et al. 2017	Argentina
Espinel et al. 2017	Eastern Cordillera of Colombia
Martinez-Tilleria et al. 2017	Chile
Reyes-Puig et al. 2017	Ecuador
Martinez-Harms et al. 2018	Chilean biodiversity hotspot
Bax & Francesconi 2019	Tropical Andes
Bennett et al. 2019	Distribution of Andean cats Leopardus jacobita (High Andes)
Fajardo et al. 2019	Tropical Andean countries
Khoury et al. 2019	Distribution of chile pepper genus (includes the Andes, except Chile)

Marquet et al. 2019	Chile
Ovando et al. 2019	Argentina
Quintana et al. 2019	Ecuadorian Dry Inter-Andean Valleys
SPARC 2019	Countries in the Neotropics and Tropical Andes
Tognelli et al. 2019	Tropical Andes

See details on the reviewed studies in: https://doi.org/10.5281/zenodo.4890967

2. Detailed methods for "Box 2 - Balancing conservation of biodiversity and NCPs in the Andes".

Objective

To understand the extent to which Crop Wild Relatives (CWR) in the Andes can benefit from range-restricted (RR) plant conservation actions and whether CWR protection can be increased through integrated planning for both groups. This reserve planning was aimed to increase the surface area under protection from 16% (national protected areas) to 30% of the Andes. The analysis covered the limits of the Andean mountain range [1] (~ 2.9 million km^2).

Species distribution dataset

We collected available information for the species distribution of RR plants and CWR in the Andes. For RR plants, we consulted the integrated Botanical Information and Ecology Network (BIEN) v4.1[2]. BIEN has produced estimated range maps for over 88 000 species of New World plants, mostly using maximum entropy (MaxEnt) algorithm and climatic variables. We included species with range sizes below ~500 000 m^2, the median value [3] for all species in BIEN database with at least half of their potential distribution in the Andes. These criteria resulted in 1726 plant species selected for the prioritization analysis.

In the case of Andean CWR, we used the database of Useful Wild Plants [4], a recent effort for gathering spatial information on useful plants worldwide. This database provides geographic ranges of the potential native distribution of CWR (species with recorded uses in crop breeding or close relatives to cultivated crops). Potential ranges were built with

MaxEnt from occurrence data, using climatic and other eco-geographic covariates. We included all CWR species with potential distribution estimates and at least 10% of their range within the Andes limits, resulting in 118 species of CWR. This set comprises CWR of peanut, chile peppers, quinoa, tobacco, beans, potato, golden berry, nuts, among others. Only nine of these 118 species are also mapped in BIEN database, which highlights how CWR are poorly covered by general plant inventories and databases, and therefore, probably excluded from most spatial prioritizations.

Priority conservation areas

Decision support tool

We used Zonation 4.0.0rc [5] to identify priority areas for the conservation of RR plants and CWR in the Andes. Zonation establishes a hierarchical prioritization (ranking) of areas of the study region, allowing the identification of top priority areas for the conservation of species based on their distributions. These priority areas are identified by a complementarity-based and balanced ranking of conservation value over the entire landscape, maximizing species occurrence.

Zonation setup

The intems below present the Zonation setup used in the final analyses.

Target features: We ran two different planning scenarios where the set of target features varied. In the first scenario, we used all RR plant species for running the identification of the priority areas; while in a second analysis, we used species from both RR plants and CWR. For both scenarios, we searched for the top 30% fraction of the Andes that produces the highest benefits for the target features.

Cell removal rule: We used Core Area Zonation (CAZ) rule because this configuration helps to identify high-priority areas that have a high occurrence level for each species separately. In this way, we were able to enhance the representation level for most of the species, especially for species occurring in areas with low species richness. Instead, ABF cell removal rule, despite leading to results that can reach higher average proportion of feature distributions retained, would have resulted in very small protection for species occurring over areas with low overall richness, as is the case of many CWR.

Weights: all features had the same weight in the analysis.

Condition: We used the Global Terrestrial Human Footprint's map [6] as a condition layer. This last step prevented the software from selecting highly modified areas and assigning them with high conservation value.

Connectivity: We used a boundary length penalty (0.1) to produce a more compact reserve network solution. Prioritizations were run with the "edge removal" option to remove cells from the edges of remaining landscape, increasing connectivity of priority and protected areas in the landscape.

Costs: the use cost layer is not recommended with the CAZ cell removal rule.

Hierarchical mask: national protected areas were included using a hierarchical mask, an approach developed to select areas of the landscape for optimal and balanced expansion of existing PAs (which are preferably selected as the first option in the analysis).

Spatial resolution: All variables for the final priority analyses were used at a spatial resolution of ~10 km^2, which is the finer resolution at which RR plant species data was available. Cells with more than 49% of their surface covered by PAs were considered as protected in the hierarchical mask.

Comparison of planning scenarios

We compared the results of the two scenarios in terms of the average and minimum proportion of the species ranges of RR plants and CWR that would be under protection in the top 30% fraction of the landscape included in solutions. Based on this comparison, the second scenario was the most cost-effective solution, since it increases the protection of CWR for a small reduction in RR plants benefits.

Potential management zoning of selected areas

The species richness for CWR and RR plants have different spatial patterns in the Andes. Therefore, the selected priority areas might have differentiated contributions to the protection of each group that are important to distinguish. To do so, we compared the distribution of the ranking of cells for RR and CWR, separately, within the resulting network of priority and protected areas (which cover 30% of the Andes that is most valuable for conservation). Based on this assessment, areas that comprises the network can be classified in four types: (1) areas with the highest priority for CWR management, as those within the top 15% of the Andes for CWR but that for RR plants have lower priority (within the top 30%-15%), (2) areas with the highest priority for managing RR plants, as those within the top 15% for RR plants but lower priority for CWR, (3) areas that are the most priority places for simultaneously managing RR plants and CWR (those that were the top 15% for both groups), and (4) the remaining areas (30% top priority for CWR and RR plants), which are also important for the conservation of both groups but have lower priorities than the previous.

3. List of species used in "Box 2 - Balancing conservation of biodiversity and NCPs in the Andes"

https://doi.org/10.5281/zenodo.4890967

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Appendices – Chapter 3

1. Summary of the ethnolinguistic diversity associated with forests and deforestation rates in these territories according to the world's regions.

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2. Results of the sensitivity test for the criteria used to identify ethnolinguistic groups associated with forests.

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