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Community managed forests and forest protected areas: An assessment of their conservation effectiveness across the tropics

Luciana Porter-Bolland [a,*], Edward A. Ellis [b], Manuel R. Guariguata [c], Isabel Ruiz-Mallén [d], Simoneta Negrete-Yankelevich [a], Victoria Reyes-García [e]

[a] Instituto de Ecología, A.C., Veracruz, Mexico
[b] Centro de Investigaciones Tropicales, Universidad Veracruzana, Veracruz, Mexico
[c] Center for International Forestry Research (CIFOR), Bogor, Indonesia
[d] Institut de Ciència i Tecnologia Ambientals, Universitat Autònoma de Barcelona, Spain
[e] ICREA and Institut de Ciència i Tecnologia Ambientals, Universitat Autònoma de Barcelona, Spain

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ABSTRACT

This paper assesses the role of protected and community managed forests for the long term maintenance of forest cover in the tropics. Through a meta-analysis of published case-studies, we compare land use/cover change data for these two broad types of forest management and assess their performance in maintaining forest cover. Case studies included 40 protected areas and 33 community managed forests from the peer reviewed literature. A statistical comparison of annual deforestation rates and a Qualitative Comparative Analysis were conducted. We found that as a whole, community managed forests presented lower and less variable annual deforestation rates than protected forests. We consider that a more resilient and robust forest conservation strategy should encompass a regional vision with different land use types in which social and economic needs of local inhabitants, as well as tenure rights and local capacities, are recognized. Further research for understanding institutional arrangements that derive from local governance in favor of tropical forest conservation is recommended.

1. Introduction

Slowing tropical deforestation and forest degradation remains an enormous challenge at both national and global scales with concomitant social, environmental, and economic implications (Geist and Lambin, 2002; Grainger, 1993; Uriarte et al., 2010). While tropical forest cover continues to decrease globally (FAO, 2010), strategies for reverting this trend are often contentious, since causal explanations and drivers of deforestation are varied and context specific (Angelsen and Kaimowitz, 1999; Geist and Lambin, 2002). Nevertheless, a general agreement exists that a mix of different forest conservation strategies are needed across the tropics that integrate public-, private-, and community-managed areas (Bray et al., 2008; Naughton-Treves et al., 2005; Nepstad et al., 2006). There is still disagreement, however, as to what are the "best practices" for forest conservation (Shahabuddin and Roa, 2010; Wilshusen et al., 2002) with some advocating strict protection and others arguing for alternative schemes such as community-based, locally-implemented conservation.

The debate partly originates from the fact that most forests have traditionally been, and still are, inhabited and managed by local people (Heckenberger et al., 2007; Noble and Dirzo, 1997), and that even forested areas considered under strict protection regimes are either inhabited or within the limits of expanding human populations (Nagendra et al., 2009). Furthermore, evidence that most areas considered important for biodiversity conservation, particularly in the tropics, coincide with long term human habitation and use is changing conservation paradigms (Lele et al., 2010). There are at least three research findings that argue for the need to develop alternatives to strict forest protection. First, empirical accounts indicate significant social and economic costs for local populations derived from the establishment of strictly protected forests (Ferraro, 2002; West et al., 2006; but see Andam et al., 2010). Second, recent research suggests that after controlling for (statistically) confounding variables, the effectiveness of strict forest protection in reducing deforestation rates may not be as high as previously estimated (i.e., a 10% reduction *vs.* earlier estimates of up to 65% reduction; Andam et al., 2008). Third, there is evidence that within the same region, forests managed by local or indigenous communities for the production of goods and services can be equally (if not more) effective in maintaining forest cover than those managed under solely protection objectives (Bray et al., 2008; Ellis and Porter-Bolland, 2008; Nepstad et al., 2006), or with

* Corresponding author. Present address: Carretera Antigua a Coatepec #351, El Haya, Coatepec, Veracruz 91070, Mexico. Tel.: +52 228 8421800x4317.
E-mail address: luciana.porter@inecol.edu.mx (L. Porter-Bolland).

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respect to the wider forest landscape (Duchelle, 2009; Stocks et al., 2007).

Other research attempts to seek if there are broad patterns that support the findings mentioned above in the quest for alternatives to strict forest protection. It is now widely recognized that plans for the management of protected areas should take into account the needs of those living within these areas. According to Naughton-Treves et al. (2005), after decades of expanding protected forest areas, the necessity of integrating human-rights concerns and equity into management objectives is now unquestionable. Furthermore, several international agreements fully recognize that biodiversity conservation must (ideally) encompass economic benefits at multiple scales, alleviate poverty, protect threatened cultures, and promote peace (Naughton-Treves et al., 2005). Also, it is widely argued that efforts to expand forest protected areas should explicitly consider the landscapes in which both protected and co-managed forest areas are embedded (Bray et al., 2008; DeFries et al., 2007; Hayes, 2006; Nagendra et al., 2009; Ostrom and Nagendra, 2006). Finally, recent assessments of change in land use/cover indicate that while protected areas can help to reduce tropical deforestation (Bruner et al., 2001), they are nevertheless becoming increasingly isolated (DeFries et al., 2005; Nagendra, 2008; Naughton-Treves et al., 2005) thus disregarding ecological, cultural, and social processes that are known to influence the permanence of forest ecosystems at landscape scales (DeFries et al., 2007; Hansen and DeFries, 2007; Sayer, 2009).

Previous land use/cover change assessments across the tropics have primarily focused on the underlying causes of deforestation (e.g., Geist and Lambin, 2002; Gibbs et al., 2010; Rudel et al., 2009) or examined the performance of forest reserves in conserving forest cover (e.g., Bruner et al., 2001; DeFries et al., 2005; Wright et al., 2007). Others have discussed the potential role of tropical forest management for the production of goods and services as a conservation tool (Dickinson et al., 1996) and also attempted to determine the environmental impacts of implementing good practices in tropical forests managed primarily for timber (e.g., Auld et al., 2008; Marx and Cuypers 2010; van Kujik et al., 2009). Much less attention has been paid, however, to discerning how these different tropical forest management strategies may contribute to reducing deforestation. Thus, the objectives of this paper are: (1) to compare the effectiveness of two broad types of tropical forest management, protected areas and community managed forests, in maintaining forest cover over time; and (2) to assess the underlying causes that may explain differences between these two management types. Specifically, we refer here to protected areas as those areas identified within the IUCN categories of: (I) strict nature reserve or (Ib) wilderness area, (II) national parks, (III) natural monument or feature, and (IV) habitat/species management area (IUCN/WCMC 1994). By community managed forests we refer to those where multiple use takes place under a variety of tenure, benefit sharing and governance schemes and that include local, rural, and/or indigenous groups (Pagdee et al., 2006). These are sometimes categorized as IUCN protected areas categories V and VI: protected landscape/seascape and protected area with sustainable use of natural resources, respectively. Our rationale to apply this comparison of management types is underlined by the assumption that local management for multiple goods and services is thought to enhance the ecological, economic, and social functions of tropical forests (Panayotou and Ashton, 1992) by promoting greater resource access (Charnley and Poe, 2007) and by including the voices of different stakeholders (Kant, 2004). Advocates of multiple-use forest management further argue that both a social and financial edge can be gained over timber dominated models (Ashton et al., 2001; Wang and Wilson, 2007). We are aware that some of these assumptions have been recently challenged to apply to specific socio-economic and development contexts (García-Fernández et al., 2008). Yet our hypothesis is that, on a pantropical scale, rates of deforestation within or around community managed forests are either equal to or less than forests under strict protection.

To test the above hypothesis, we rely on quantitative information derived from studies reporting land use/cover change. In the last few decades, the number of these type of studies has increased dramatically as remote sensing techniques have become widely available and cost-effective (Uriarte et al., 2010). That said, accounting for forest cover change (i.e., deforestation or forest recovery) is not sufficient to represent complex tropical land use dynamics or to understand the legacy of past management and abandonment cycles (Lawrence et al., 2010), nor to provide enough information to understand forest degradation processes (Putz and Redford, 2010). Through review and meta-analysis of case studies that account for change in forest cover, however, it is possible to provide a robust indication of the effectiveness of land use types (in our case, community managed forests *vs*. strict forest protection) in conserving tropical forest cover (Rudel, 2008). We further recognize that forest cover is only one metric for assessing conservation effectiveness. In the case of community management, no change in forest cover may imply a reduction in socioeconomic benefits (e.g., Vadjunec et al. 2009). In the case of protected areas, no change in forest cover does not necessarily imply that the mammalian fauna, for example, is fully present (e.g., Redford, 1992). Yet the maintenance of forest cover is being widely agreed as a robust indicator of environmental integrity and biodiversity status at local and global levels (Bruner et al., 2001; Butchart et al., 2010). Based on a literature review and meta-analysis of case studies, we assess differences in forest cover change over time and the causal scenarios behind forest cover trends for both protected and community managed forests and compare the effectiveness of these two management types in the maintenance and conservation of tropical forest cover.

2. Methods

We conducted a meta-analysis of published empirical case studies that assessed forest cover change in tropical environments either under Protected Area status (including national parks, biosphere reserves, and wildlife reserves; hereafter PA) or community managed forests (including indigenous reserves, extractive reserves, community forest management or areas with communal forest resource use; hereafter CMF). Our meta-analysis included a statistical comparison of deforestation, specifically forest cover change rates in PA and CMF along with a Qualitative Comparative Analysis (QCA) to identify the main drivers or causal conditions that correlate with either deforestation or forest conservation (Rudel, 2008).

2.1. Identification of case studies

During 2010, we searched three web-based engines: (i) SCOPUS (www.scopus.com); (ii) EBSCO (http://search.ebscohost.com); and (iii) CABI (http://www.cabi.org), to locate peer-reviewed journal articles cited by the index of the Institute for Scientific Information (ISI; http://science.thomsonreuters.com) that describe land use/cover change in either PAs or CMFs across the tropics. Keywords entered in database searches included: *deforestation*, *land use and cover change*, *protected area*, *community conservation*, *community forest management*, *extractive reserves, indigenous reserve, working forest, tropical forest, and multiple-use forest*. Since many articles report examples and data of land use/cover change for individual or groups of PA and/or CMF in one or several tropical regions, we conducted our analysis at the case study level. By "case study" we mean each individual, or in some cases, group of individual PA or CMF located in a specified geographical region for which temporal change in forest cover is both evaluated and reported. Because the analysis was conducted at the case study level, we ended up analyzing more case studies than the total number of articles selected. From a pool of 109 articles identified we selected 27 articles that included a total of 73 case studies that met the following criteria: (i) evaluated land use/cover changes in PA and/or CMF of specific locations (i.e., articles reporting broad or regional changes in forest cover were excluded); (ii) reported a net annual rate of forest cover change or else provide the data to calculate such a rate; (iii) the location of PA and/or CMF occurred within the Tropic of Cancer and Tropic of Capricorn; (iv) provided information on drivers of change of forest cover in the study area; and (v) for PAs, reported changes in forest cover that occurred after the establishment of the protected status. Buffer zones (from 2 to 5 km) were included in the meta-analysis when these were part of the PA. We purposefully excluded articles dealing with colonization in agricultural frontier regions (since they deviate from our characterization of "community managed forests"; e.g., Aldrich et al., 2006; Caldas et al., 2007; Walker et al., 2000). Articles that only reported percent deforested area but not deforestation rates *per se* or that did not report forest areas needed to calculate annual rates in forest cover change were also excluded (as in the case of a CMF in Brazil [Vadjunec et al., 2009] and in Nicaragua [Stocks et al., 2007]).

Case studies were classified into 40 PAs and 33 CMFs. Table 1 shows information on the case studies used in the meta-analysis. Information in the table includes identification code, name of the study area, country, whether it refers to a PA or a CMF, reported (or calculated) deforestation rate, and the bibliographic reference. The selection of case studies comprised a total of 16 countries, most of them (11) located in Latin America and the Caribbean, followed by three countries in Southeast Asia, and two countries in Africa (Table 1).

2.2. Analysis of forest cover change in PA and CMF

To compare the effectiveness of PAs and CMFs in deterring deforestation, we compared annual deforestation rates in both forest types using Mann–Whitney test for two samples, since the data did not fulfill distributional assumptions of parametric statistics. Unless indicated otherwise, we set an *a priori* significance level of 10% following other land use/cover change studies that compare deforestation rates and justify it either because of the small number of samples analyzed (Armenteras et al., 2009), or in metadata analysis, given the disparity of research objectives, approaches and methods employed (Nagendra, 2008).

We acknowledge that particularly in tropical countries, rates of forest cover change vary according to geographical location, forest area, as well as the spatial and temporal scale of analyses (Ludeke et al., 1990; Mertens et al., 2004; Ewers, 2006; Joppa and Pfaff, 2010; Ellis et al., 2010). As described in the selection criteria above, we addressed this issue as much as possible by limiting our selected PA and CMF case studies and forest cover change rate data to specific locations by excluding regional or national scales of analysis. Our data set of selected case studies had a mean of 496,244 ha and a median of 67,750 ha; 82% of the sample had areas of analysis between 6,000 and 350,000 ha. Likewise, possible confounding effects of time in determining the effectiveness of PA and CFM in conserving forest cover are minimal. Among the 73 case studies, forest cover change was reported across periods that ranged from 3 to 38 years (mean = 12.5 years; median = 12 years). Eighty percent of case studies had periods of analysis from 1985 to 2005, split in periods from 4 to 12 years and from 12 to 20 years. Furthermore, we found that the comparison of rates of annual forest cover change between PAs and CMFs was not confounded by either forest area ($n = 60$, $R^2 = 0.013$; $p = 0.379$) or number of years within the periods of analysis ($n = 73$, $R^2 = 0.003$; $p = 0.626$).

2.3. Qualitative Comparative Analysis (QCA)

We applied a Qualitative Comparative Analysis (QCA) as a meta-analysis tool, which reduces the complexity of a multivariate binary data set (presence or absence of certain variables from a specific site) by detecting scenarios in which a response variable (deforestation rate in our case) most often takes the *present* or *absent* value. In other words, QCA provides the analytical tools for comparing explanatory models across case studies through an output of a minimized set of factors (scenario) that in combination produce a particular outcome in the response variable (Ragin, 2008; Rudel, 2008). In our study, the input for the QCA was a matrix that contained all 73 case studies and included information regarding presence or absence of deforestation (the response variable) and presence or absence of explanatory variables that are usually associated with forest cover change. The set of explanatory variables included all site characteristics that were reported across case studies as possible deforestation or forest recovery causal factors, such as population growth, market integration (urbanization and transport infrastructure extension), economic policies and programs leading to commercial wood extraction, livestock development, and agricultural expansion, and changes in land tenure (see 'QCA Variables'). These are also some of the main underlying factors of deforestation in the Millennium Ecosystem Assessment (MEA, 2005). The cells in the matrix were filled with a binary code representing whether each variable was present or absent in each case study. The last column of the matrix represented the response variable, deforestation (DEF).

QCA uses Boolean algebra (or Boolean minimization) to construct an output matrix named a "truth table" with a list of the minimum set of causal conditions or "prime implicants" that explain the outcomes of the response variable. A "truth table" contains cases grouped by similar configurations of causal conditions after considering all logically possible combinations of factors and cases. It chooses the groups that minimize the number of factors that produce a particular outcome in the response variable. Moreover, the "truth table" identifies those combinations of causal conditions and case studies that are contradictory in the dataset, that is, the case studies that result in both outcomes of deforestation or forest maintenance/recovery. Results can then be evaluated to determine what set or combinations of causal conditions or "prime implicants" are responsible for the outcome of deforestation or forest conservation, and which case studies exemplify these conditions and outcomes. To carry out the QCA we used the software TOSMANA version 1.3.1 (Cronqvist, 2009), which applies the Quine algorithm to perform the Boolean minimization.

2.3.1. QCA Variables

A total of 18 binary variables were used in the QCA matrix. Seventeen of these refer to explanatory variables or causal conditions derived from the selected articles and reported by authors as drivers of forest cover change. The eighteenth variable refers to the response variable of deforestation or forest conservation derived from forest cover change rates obtained for each case study. To assign values to each explanatory variable for each case study, a code of (1) was assigned if authors mentioned it as a driver of forest cover change or maintenance/recovery in the studied area and a (0) if it was not mentioned or not regarded as important. The variables used in our QCA are the following:

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Table 1
Reported deforestation rates in protected forest areas (PA) and community managed forest (CMF) case studies across the tropics used for the meta-analysis.

Case	Case ID	Case study area	Country	PA	CMF	Y = rate[a]	References
Belize BS	PABE1	Community Baboon Sanctuary	Belize	Yes	No	−2	Wyman and Stein (2010)
Bolivia MNP	PABO1	Madidi NP	Bolivia	Yes	No	0.4	Forrest et al. (2008)
Brazil NP	PABZ1	National Parks	Brazil	Yes	No	−0.05	Nepstad et al. (2006)
Colombia MNP	PACO1	Macarena National Park (inside)	Colombia	Yes	No	−0.1695	Armenteras et al. (2009)
Colombia TNP	PACO2	Tuparro National Park (inside)	Colombia	Yes	No	−0.062	Armenteras et al. (2009)
Colombia NNP	PACO3	Nukak National Park (inside)	Colombia	Yes	No	−0.059	Armenteras et al. (2009)
Colombia PNP	PACO4	Puinaway National Park (inside)	Colombia	Yes	No	−0.0518	Armenteras et al. (2009)
Colombia CNP	PACO5	Chirinbiquete National Park (inside)	Colombia	Yes	No	−0.0162	Armenteras et al. (2009)
Costa Rica BCNP1	PACR1	Braulio Carrillo NP (North Region and Surrounding Areas)	Costa Rica	Yes	No	−19.4	Schelhas and Sanchez–Azofeifa (2006)
Costa Rica BCNP2	PACR2	Braulio Carrillo NP (Northern Region and Surrounding Areas)	Costa Rica	Yes	No	−6.7	Schelhas and Sanchez–Azofeifa (2006)
Costa Rica CNP	PACR3	Corcavado Natl. Park (no buffer)	Costa Rica	Yes	No	0	Sanchez-Azofeifa et al. (2002)
Guatemala SLNP	PAGU1	Sierra Lacandon Natl Park	Guatemala	Yes	No	−1.07	Sader et al. (2001)
Guatemala DLR	PAGU10	Dos Lagunas Reserve	Guatemala	Yes	No	0	Sader et al. (2001)
Guatemala TNP	PAGU11	Tikal Natl. Park	Guatemala	Yes	No	0	Sader et al. (2001)
Guatemala MBR (5I)	PAGU2	Maya Biosphere Reserve (5 Mgmt. Units-Inhabited)	Guatemala	Yes	No	−0.694	Bray et al. (2008)
Guatemala LDTNP	PAGU3	Laguna del Tigre Natl Park	Guatemala	Yes	No	−0.33	Sader et al. (2001)
Guatemala CCR	PAGU4	Cerro Cahui Reserve	Guatemala	Yes	No	−0.11	Sader et al. (2001)
Guatemala EZR	PAGU5	El Zotz Reserve	Guatemala	Yes	No	−0.09	Sader et al. (2001)
Guatemala LDTR	PAGU6	Laguna del Tigre Reserve	Guatemala	Yes	No	−0.03	Sader et al. (2001)
Guatemala MBR(5U)	PAGU7	Maya Biosphere Reserve (5 Mg mt. Units-Uninhabited)	Guatemala	Yes	No	−0.018	Bray et al. (2008)
Guatemala EMNP	PAGU8	El Mirador Natl Park	Guatemala	Yes	No	0	Sader et al. (2001)
Guatemala RANP	PAGU9	Rio Azul Natl Park	Guatemala	Yes	No	0	Sader et al. (2001)
Honduras CNP1	PAHO1	Celaque Natl. Park (inside boundaries)	Honduras	Yes	No	−1.04	Southworth et al. (2004)
Honduras CNP2	PAHO2	Celaque Natl. Park (inside boundaries)	Honduras	Yes	No	−0.47	Southworth et al. (2004)
India TART (I)	PAIA1	TATR Interior	India	Yes	No	0	Nagendra et al. (2006)
Indonesia GRWS	PAIO1	Gunung Raya Wildlife Sanctuary (GRWS)	Indonesia	Yes	No	−2.74	Gaveau et al. (2007)
Indonesia GPNP	PAIO2	Gunung Palung National Park (and a 10 km buffer)	Indonesia	Yes	No	−2.2	Curran et al. (2004)
Indonesia HR	PAIO3	Hydrological Reserve (HR)	Indonesia	Yes	No	−2.13	Gaveau et al. (2007)
Indonesia BBSNP	PAIO4	Bukit Barisan Selatan National Park (BBSNP)	Indonesia	Yes	No	−0.64	Gaveau et al. (2007)
Jamaica BMRPE	PAJM1	Blue Mountain range after Park Establishment	Jamaica	Yes	No	−0.26	Chai et al. (2009)
Malaysia SBFR	PAMA1	Sungai Buloh Forest Reserve	Malaysia	Yes	No	−9.07	Jusoff and Manaf (1995)
Malawi LMNP	PAMI1	Lake Malawi Natl. Park	Malawi	Yes	No	−0.83	Abbot and Homewood (1999)
Mexico LTBR	PAMX1	Los Tuxtlas BR	Mexico	Yes	No	−4.3	Dirzo and Garcia (1992)
Mexico MBR	PAMX2	Monarch Butterfly Reserve	Mexico	Yes	No	−2.4	Brower et al. (2002)
Mexico LM1	PAMX3	La Montaña (8 Ejidos-Buffer and Surrounding Calakmul BR)	Mexico	Yes	No	−0.7	Ellis and Porter-Bolland (2008)
Mexico MABR	PAMX4	Montes Azules BR within boundaries	Mexico	Yes	No	−0.33	Mendoza and Dirzo (1999)
Mexico LM2	PAMX5	La Montaña (8 Ejidos-Buffer and surrounding Calakmul BR)	Mexico	Yes	No	−0.3	Ellis and Porter-Bolland (2008)
Mexico CBR	PAMX6	Calakmul Biosphere Reserve (Buffer Zone and Core)	Mexico	Yes	No	−0.16	Vester et al. (2007)
Peru PA	PAPE1	Protected Areas	Peru	Yes	No	−0.008	Oliveira et al. (2007)
Zimbabwe SWR	PAZE1	Sengwa Wildlife Reserve	Zimbabwe	Yes	No	−0.7	Mapaure and Campbell (2002)
Bolivia TIR	WFBO1	Tacana Indigenous Reserve	Bolivia	No	Yes	−0.05	Forrest et al. (2008)
Brazil IR	WFBZ1	Indigenous Reserves	Brazil	No	Yes	−0.2	Nepstad et al. (2006)
Brazil ER	WFBZ2	Extractive Reserves	Brazil	No	Yes	−0.17	Nepstad et al. (2006)
Brazil AJER	WFBZ3	Alto Jurua Extractive Reserve	Brazil	No	Yes	0	Ruiz-Perez et al. (2005)
Colombia EI	WFCO9	El Itilla (inside)	Colombia	No	Yes	−0.1265	Armenteras et al. (2009)
Colombia BC	WFCO1	Barranco Colorado (inside)	Colombia	No	Yes	−1.99	Armenteras et al. (2009)
Colombia IR1	WFCO10	Inside IR	Colombia	No	Yes	−0.0933	Armenteras et al. (2009)
Colombia TDCGLP	WFCO11	Inside Tucan de Caño Giriza La Palma	Colombia	No	Yes	−0.0422	Armenteras et al. (2009)
Colombia LF	WFCO12	Inside La Fuga	Colombia	No	Yes	−0.0254	Armenteras et al. (2009)
Colombia PNP	WFCO13	Inside Puerto Nare	Colombia	No	Yes	−0.0041	Armenteras et al. (2009)
Colombia PV y PE	WFCO14	Inside Puerto Viejo y Puerto Esperanza	Colombia	No	Yes	0.0349	Armenteras et al. (2009)
Colombia PC	WFCO2	Inside Piaroa de Cachicamo	Colombia	No	Yes	−0.8227	Armenteras et al. (2009)
Colombia LY	WFCO3	Inside Llanos de Yari	Colombia	No	Yes	−0.8127	Armenteras et al. (2009)
Colombia IR2	WFCO4	Inside IR	Colombia	No	Yes	−0.7644	Armenteras et al. (2009)
Colombia LS	WFCO5	Inside La Sal	Colombia	No	Yes	−0.5839	Armenteras et al. (2009)
Colombia CMD y M	WFCO6	Cano Mesetas-Dagua y Murcielago (inside)	Colombia	No	Yes	−0.2247	Armenteras et al. (2009)
Colombia BC	WFCO7	Barranquillita (inside)	Colombia	No	Yes	−0.2184	Armenteras et al. (2009)
Colombia IR3	WFCO8	Inside IR	Colombia	No	Yes	−0.2072	Armenteras et al. (2009)
Guatemala P4	WFGU1	Peten (4 Concessions recently inhabited)	Guatemala	No	Yes	−0.716	Bray et al. (2008)
Guatemala P2	WFGU2	Peten (2 Concessions-Long inhabited)	Guatemala	No	Yes	−0.022	Bray et al. (2008)
Guatemala P6	WFGU3	Peten (6 Concessions-Uninhabited)	Guatemala	No	Yes	−0.003	Bray et al. (2008)
India TART (P)	WFIA1	TATR Periphery	India	No	Yes	−0.25	Nagendra et al. (2006)
Mexico EXM	WFMX1	X-Maben Ejido, QROO	Mexico	No	Yes	−0.6	Dalle et al. (2006)

Table 1 (*continued*)

Case	Case ID	Case study area	Country	PA	CMF	*Y* = rate[a]	References
Mexico UEFHG	WFMX2	10 Ejidos Union de Ejidos Forestales Hermenegildo Galeana	Mexico	No	Yes	−0.4	Durán-Medina et al. (2005)
Mexico CFM	WFMX3	Quintana Roo CFM (7 Ejidos-Long inhabited)	Mexico	No	Yes	−0.024	Bray et al. (2008)
Mexico ZM1	WFMX4	Zona Maya (12 Ejidos-Community Forest Management)	Mexico	No	Yes	−0.0004	Ellis and Porter-Bolland (2008)
Mexico ZM2	WFMX5	Zona Maya (12 Ejidos-Community Forest Management)	Mexico	No	Yes	0.002	Ellis and Porter-Bolland (2008)
Mexico OEPFZM	WFMX6	12 Ejidos de la OEPFZM FCP QROO	Mexico	No	Yes	0.63	Durán-Medina et al. (2005)
Peru IT	WFPE1	Indigenous Territories	Peru	No	Yes	−0.096	Oliveira et al. (2007)

[a] *Y* = rate refers to forest cover change rate with negative values being deforestation.

1. Protected Area (PA) was coded as (1) if case study area is under protected area status.
2. Community managed forest (CMF) was coded as (1) if the study site was considered a working forest.
3. Presence of infrastructure and roads (INF), coded as (1) if the source mentioned road development and access as well as industrial or urban expansion in the area as a deforestation driver.
4. Population pressure (POP), coded as (1) when demographic pressure reported as population growth or migration was claimed as a problem associated with land use/cover change.
5. Agricultural expansion (AGX), coded as (1) if the source reported increase in area of agricultural production systems, ranging from slash and burn subsistence agriculture to commercialized cash crops and plantations.
6. Cattle (CAT), coded as (1) if the source reported agricultural expansion generating deforestation pressures from pasture establishment for livestock production.
7. Development and agricultural policy (DEV), coded as (1) when government and international development policies and institutions (such as agricultural colonization projects, cash crop development programs and incentives or other non-agricultural development initiatives including residential development or industrial forestry) were present in the study area.
8. Markets and prices (MKT), coded as (1) when the source reported presence of markets for agricultural, timber, and non-timber forest products in the area.
9. Natural disasters (NDI): coded as (1) when the source reported occurrence of fires and hurricanes in the study area.
10. Private land use and tenure (PRI), coded as (1) when the source reported the presence of private land tenure systems of agricultural production parcels or lands or other types of land uses in the study area.
11. Communal land use (CLU), coded as (1) when the communities were reported to use and manage their territory under communal land use and tenure arrangements.
12. State owned property (STA), coded as (1) when the case study area was reported to be territory owned by the regional or national governments, as in the case of most PAs.
13. Conservation policy and institutions (CON), coded as (1) when there is the presence of conservation policies and institutions for natural protected areas aiming at biodiversity conservation or for the sustainable use and management of natural resources.
14. Natural resource management (NRM), coded as (1) when inhabitants in the study area actively use and manage the forest environment for subsistence and economic purposes.
15. Forestry (FOR), coded as (1) when the study area contains forest extraction and management activities.
16. Indigenous population (IND), coded as (1) when the case study area was reported to be mostly inhabited by indigenous populations as in the case of indigenous reserves.
17. Remote frontier regions (REM), coded as (1), when the study area was considered to have the condition of remoteness or inaccessibility, as indicated by the source.
18. Deforestation (DEF), as the response variable, was coded as (1) when the annual rate of forest cover loss reported in the article was greater than −0.2%, and (0) otherwise (forest cover maintenance/recovery). This threshold was selected based on the distribution of our deforestation rate for all case studies and considering general land cover change literature where often rates of forest cover loss greater than −0.25% are associated with, or claimed as, having a deforestation trend.

3. Results

3.1. Deforestation rates in PAs and CMFs

We compared rates of forest cover change between PA ($n = 40$) and CMF ($n = 33$) case studies. The mean annual rate of forest cover change in PAs was −1.47, indicating a net loss of forest cover. There was, however, a wide variation in the data (SD = 3.46) with a maximum annual rate of deforestation of −19.40 and a maximum rate of forest recovery of 0.40. In contrast, for the CMFs case studies, the mean rate of forest cover change was higher than for PAs (−0.24).

Table 2
Protected area (PA) and community managed forest (CMF) case studies undergoing deforestation (annual percent forest cover change rates equal to or below −0.2). Total number of case studies analyzed were 40 (PA) and 33 (CMF).

Protected area	Annual deforestation rate	Community managed forest	Annual deforestation rate
Belize BS	−2	Brazil IR	−0.2
Costa Rica BCNP1	−19.4	Colombia BC	−1.99
Costa Rica BCNP2	−6.7	Colombia BC	−0.2184
Guatemala LDTNP	−0.33	Colombia CMD y M	−0.2247
Guatemala MBR (5I)	−0.694	Colombia IR2	−0.7644
Guatemala SLNP	−1.07	Colombia IR3	−0.2072
Honduras CNP1	−1.04	Colombia LS	−0.5839
Honduras CNP2	−0.47	Colombia LY	−0.8127
Indonesia BBSNP	−0.64	Colombia PC	−0.8227
Indonesia GPNP	−2.2	Guatemala P4	−0.716
Indonesia GRWS	−2.74	India TART (P)	−0.25
Indonesia HR	−2.13	Mexico EXM	−0.6
Jamaica BMRPE	−0.26	Mexico UEFHG	−0.4
Malawi LMNP	−0.83		
Malaysia SBFR	−9.07		
Mexico LM1	−0.7		
Mexico LM2	−0.3		
Mexico LTBR	−4.3		
Mexico MABR	−0.33		
Mexico MBR	−2.4		
Zimbabwe SWR	−0.7		
Total number of cases	21	Total number of cases	13
Proportion of PAs with positive rates	52.5%	Proportion of CMFs with positive rates	39.4%
Average rate	−2.77	Average rate	−0.59

In other words, CMFs had a lower average rate of deforestation than PAs. There was also less variation between CMF case studies (SD = 0.439) with a maximum deforestation rate of −1.99 and a maximum rate of forest recovery of 0.63.

The normality (Shapiro–Wilk) test indicates that forest cover change data from PAs and CMFs did not have a normal distribution ($W = 0.472$, $p < 0.0001$ and $W = 0.780$, $p < 0.0001$). The two-sample Kolmogorov–Smirnov test further demonstrated that the distributions of the two samples (PAs and CMFs) were statistically different at the 0.1 significance level ($D = 0.283$, $p = 0.085$). Likewise, the Mann–Whitney test showed that forest cover change rates were statistically different ($U = 817.5$, $p = 0.082$).

Twenty one out of the 40 PA case studies analyzed (52.5%) presented annual deforestation rates higher than −0.2. In the case of CMFs, only 13 case studies out of a total of 33 (39.4%) showed deforestation rates greater than −0.2%. In sum, the statistical analysis from our meta-analysis suggests that CMFs seem to perform better than PAs in having (i) lower annual deforestation rates and (ii) less variation in rates of forest cover change as compared to PAs. Indeed, the top 10 cases with the highest annual deforestation rates, ranging from −1.99 to −19.4, consisted of nine PAs located in Costa Rica, Mexico, Belize, Malaysia, and Indonesia and only one CMF (from Colombia) with the lowest annual deforestation rate (−1.99).

3.2. Underlying factors of land use change

3.2.1. Deforestation in PAs and CMFs

As already mentioned, high annual deforestation rates were found for both PA and CMF case studies, although high deforestation rates were most prominent for PAs (Table 2). The QCA analysis helped in understanding the relative role of underlying drivers of deforestation for 20 of the PA cases (Table 3). The 20 PAs that experienced deforestation were located in a wide range of countries (Indonesia, Guatemala, Honduras, Mexico, Belize, Costa Rica, Jamaica, India, Brazil, Zimbabwe, Malawi, and Malaysia). Most of these PAs were managed by the government (90% of cases examined), although half were under private land tenure. Although framed by conservation policy institutions, development policies were also present in most of the PAs that experienced deforestation (in up to 80% of the cases). Those development policies resulted in agricultural expansion (including cattle ranching for 50% of these cases; particularly in Latin America). For 70% of the PAs with high rates of deforestation, human population growth was considered

Table 3
Portion of the QCA truth table from the meta-analysis dataset with information for the outcome of deforestation (DEF = 1) showing the 20 "protected area" cases (out of a total of 40) that had high deforestation rates.

Case ID	Case identifier	PA	CMF	INF	POP	AGX	CTL	DEV	MKT	PRI	NDI	CLU	STA	CON	NRM	FOR	IND	REM	DEF
PAIO1	Indonesia GRWS	1	0	0	1	1	0	1	1	1	1	0	1	1	0	0	0	0	1
PAIO3	Indonesia HR	1	0	0	1	1	0	1	1	1	1	0	1	1	0	0	0	0	1
PAIO4	Indonesia BBSNP	1	0	0	1	1	0	1	1	1	1	0	1	1	1	1	0	0	1
PAGU2	Guatemala MBR (5I)	1	0	1	1	1	1	1	0	0	1	0	1	1	1	0	1	0	1
PAHO2	Honduras CNP2	1	0	1	1	1	0	0	0	0	1	0	1	1	0	0	1	1	1
PAHO1	Honduras CNP1	1	0	1	1	1	0	0	0	0	1	0	1	1	0	0	1	1	1
PAMX5	Mexico LM2	1	0	1	1	1	1	1	0	0	1	1	0	1	1	0	1	0	1
PAMX3	Mexico LM1	1	0	1	1	1	1	1	0	0	1	1	0	1	1	0	1	0	1
PAMX4	Mexico MABR	1	0	1	0	1	1	1	0	0	1	0	1	1	0	0	1	0	1
PAMX1	Mexico LTBR	1	0	1	1	1	1	1	1	1	0	0	1	1	0	0	1	0	1
PABE1	Belize BS	1	0	1	0	1	1	1	1	1	1	0	1	1	1	0	0	0	1
PACR1	Costa Rica BCNP1	1	0	1	1	1	1	1	1	1	0	0	1	0	0	0	0	0	1
PACR2	Costa Rica BCNP2	1	0	1	1	1	1	1	1	1	0	0	1	1	1	0	0	0	1
PAIO2	Indonesia GPNP	1	0	0	0	0	0	1	1	0	1	0	1	1	0	1	0	0	1
PAJM1	Jamaica BMRPE	1	0	0	1	1	0	1	1	1	1	0	1	1	0	1	0	0	1
PAGU1	Guatemala SLNP	1	0	0	0	1	1	1	0	0	1	0	1	1	0	0	0	0	1
PAMX2	Mexico MBR	1	0	0	0	1	1	1	1	1	0	0	1	1	0	1	0	0	1
PAZE1	Zimbabwe SWR	1	0	0	0	0	0	0	0	0	1	0	1	1	1	0	0	0	1
PAMI1	Malawi LMNP	1	0	0	1	0	0	0	0	0	0	0	1	1	1	0	0	0	1
PAMA1	Malaysia SBFR	1	0	1	1	0	0	1	1	1	0	0	1	1	1	0	0	0	1
Totals		20	0	11	14	16	10	16	11	10	14	2	18	19	9	4	7	2	20

PA = Protected Area; CMF = community managed forests; INF = Infrastructure; POP = population; AGX = agricultural expansion; CTL = cattle; DEV = development and agricultural policy; MKT = market and prices; PRI = private land use tenure; NDI = natural disasters; CLU = communal land use; STA = state owned property ; CON = conservation policy institutions; NRM = natural resource management; FOR = forestry; IND = indigenous populations; REM = remote and inaccessible areas; DEF = annual deforestation rate greater than −0.2%.

Table 4
Portion of the QCA truth table from the meta-analysis dataset with information for the outcome of deforestation (DEF = 1) showing the 4 "community managed forest" cases (out of a total of 33) that were found to have high deforestation rates.

Case ID	Case identifier	PA	CMF	INF	POP	AGX	CTL	DEV	MKT	PRI	NDI	CLU	STA	CON	NRM	FOR	IND	REM	DEF
WFMX2	Mexico UEFHG	0	1	0	0	1	0	1	0	0	1	1	0	1	1	1	0	0	1
WFMX1	Mexico EXM	0	1	0	1	1	0	1	0	0	1	1	0	1	1	1	1	0	1
WFIA1	India TART (P)	0	1	1	1	1	1	1	1	0	1	1	1	1	1	1	1	0	1
WFBZ1	Brazil IR	0	1	1	0	1	1	1	1	0	1	1	1	1	1	1	1	0	1
Totals		0	4	2	2	4	2	4	2	0	4	4	2	4	4	4	3	0	4

PA = Protected Area; CMF = Community managed forests; INF = Infrastructure; POP = population; AGX = agricultural expansion; CTL = cattle; DEV = development and agricultural policy; MKT = market and prices; PRI = private land use tenure; NDI = natural disasters; CLU = communal land use; STA = state owned property ; CON = conservation policy institutions; NRM = natural resource management; FOR = forestry; IND = indigenous populations; REM = remote and inaccessible areas; DEF = annual deforestation rate greater than −0.2%

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to be exerting pressure on protected forests (of which only 35% were of indigenous origin and referred to cases located in Honduras, Guatemala, and Mexico). Fifty five percent of the PAs that experienced deforestation were either influenced by infrastructure development (particularly roads, mostly in Latin America) or by economic activities outside the forest (e.g., coffee production in Costa Rica and Jamaica). A few of the PA cases were subject to natural resource use or timber exploitation. In two of the PAs, one in Indonesia and one in Mexico, the presence of commercial forestry activities and timber markets was associated with loss of forest cover. Two of the PA cases examined (10%), both in Honduras, referred to remote areas that can be considered to be located in forests frontier regions and that hence suffered from infrastructure development, population pressure, and agricultural expansion.

The QCA analysis explains deforestation in only four of the CMF cases (two in Mexico, one in India, and one in Brazil; Table 4). According to our analysis, the underlying factors behind deforestation in those cases were agricultural expansion, development policies, and population pressures. Two of these, an indigenous reserve in Brazil and a managed forest area in the periphery of a protected reserve in India, also had cattle production and the influence of regional markets that influenced forest cover loss, despite having the presence of conservation policy and institutions, being state owned property, and having natural resource management and forestry activities. Compared to deforestation case studies of PAs, these CMFs had comparatively lower deforestation rates (between −0.2 and −0.6).

3.2.2. Forest conservation and/or recovery in PAs and CMFs

Maintenance of forest cover or its recovery occurred in 19 out of the 40 PA case studies (47.5%) and in 20 out of the 33 CMF cases (60.6%; Table 5). The QCA explained this process for 18 PA case studies (45%; Table 6) and for 13 CMF case studies (39.4%; Table 7). Of the 18 PAs where there was a process of forest cover maintenance or recovery that could be explained by the QCA, most (78%; 14 of the cases) were characterized by the absence of infrastructure development, population pressure, agricultural expansion, cattle ranching, development policies, and markets. In addition, most case studies (78%) were also characterized by being remote sites and having the presence of conservation policies and institutions. These PAs were located in Guatemala, Brazil, Costa Rica, Bolivia, Peru, Colombia, and India. Four PAs with outcomes of forest conservation, located in Colombia, Guatemala, and Mexico, reported the presence of infrastructure development, population pressure, agricultural expansion, development policies and/or market pressures. Two of these cases, however, were either remote or presented particular conditions of inaccessibility. Based on these results, the PA with highest forest recovery, i.e., with a forest cover change rate of 0.4, and the presence of underlying drivers of deforestation, was a particularly remote site in Guatemala.

Table 5
Protected Area (PA) and Community Managed Forests (CMF) case studies undergoing forest maintenance/recovery (annual percent forest cover change rates greater than −0.2).

PA (n = 40)	Annual deforestation rate reported	CMF (n = 33)	Annual deforestation rate reported
Bolivia MNP	0.4	Brazil ER	−0.17
Brazil NP	−0.05	Colombia EI	−0.1265
Colombia CNP	−0.0162	Peru IT	−0.096
Colombia MNP	−0.1695	Colombia IR1	−0.0933
Colombia NNP	−0.059	India TART (P)	−0.08
Colombia PNP	−0.0518	Nicaragua BNRR	−0.07
Colombia TNP	−0.062	Bolivia TIR	−0.05
Costa Rica CNP	0	Colombia TDCGLP	−0.0422
Guatemala CCR	−0.11	Colombia LF	−0.0254
Guatemala DLR	0	Mexico CFM	−0.024
Guatemala EMNP	0	Guatemala P2	−0.022
Guatemala EZR	−0.09	Colombia PNP	−0.0041
Guatemala LDTR	−0.03	Guatemala P6	−0.003
Guatemala MBR(5U)	−0.018	Mexico ZM1	−0.0004
Guatemala RANP	0	Brazil AJER	0
Guatemala TNP	0	Mexico ZM2	0.002
India TART (I)	0	Colombia PV y PE	0.0349
Mexico CBR	−0.16	Bolivia MIMA	0.2
Peru PA	−0.008	Mexico OEPFZM	0.63
Total number of PAs with negative deforestation rates (proportion regarding n)	19 (47.5%)	Total number of CMF with negative deforestation rates (proportion regarding n)	19 (57.6%)
Average rate	−0.022342105	Average rate	−0.001

Table 6
Portion of the QCA truth table from the meta-analysis dataset including information for the outcome of forest conservation/recovery (DEF = 0) showing the 18 “protected area” cases (out of a total of 40) that showed negative deforestation rates.

Case ID	Case identifier	PA	CMF	INF	POP	AGX	CTL	DEV	MKT	PRI	NDI	CLU	STA	CON	NRM	FOR	IND	REM	DEF
PAMX6	Mexico CBR	1	0	0	1	1	1	1	1	1	1	1	1	1	0	0	1	1	0
PAGU8	Guatemala EMNP	1	0	0	0	0	0	0	0	0	1	0	1	1	0	0	0	1	0
PABZ1	Brazil NP	1	0	0	0	0	0	0	0	0	1	0	1	1	0	0	0	1	0
PAGU9	Guatemala RANP	1	0	0	0	0	0	0	0	0	1	0	1	1	0	0	0	1	0
PAGU7	Guatemala MBR(5U)	1	0	0	0	0	0	0	0	0	1	0	1	1	0	0	0	1	0
PAGU10	Guatemala DLR	1	0	0	0	0	0	0	0	0	1	0	1	1	0	0	0	1	0
PACR3	Costa Rica CNP	1	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0
PABO1,	Bolivia MNP	1	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	1	0
PAPE1,	Bolivia MNP	1	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	1	0
PACO5,	Colombia CNP	1	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	1	0
PACO3,	Colombia NNP	1	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	1	0
PACO4,	Colombia PNP	1	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	1	0
PACO2	Colombia TNP	1	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	1	0
PAIA1	India TART (I)	1	0	0	0	0	0	0	0	0	1	1	1	1	1	0	0	1	0
PACO1	Colombia MNP	1	0	1	1	1	1	1	1	1	0	0	1	1	0	0	0	0	0
PAGU6	Guatemala LDTR	1	0	1	1	1	1	1	0	0	1	0	1	1	0	0	0	1	0
PAGU4	Guatemala CCR	1	0	0	1	1	1	1	0	0	1	0	1	1	0	0	0	0	0
PAGU11	Guatemala TNP	1	0	0	0	0	0	0	0	0	1	0	1	1	0	0	0	0	0
Totals		18	0	2	4	4	4	4	2	2	10	2	18	18	1	0	1	14	18

PA = Protected Area; CMF = Community managed forests ; INF = Infrastructure; POP = population; AGX = agricultural expansion; CTL = cattle; DEV = development and agricultural policy; MKT = market and prices; PRI = private land use tenure; NDI = natural disasters; CLU = communal land use; STA = state owned property ; CON = conservation policy institutions; NRM = natural resource management; FOR = forestry; IND = indigenous populations; REM = remote and inaccessible areas; DEF = deforestation rate greater than −0.2%.

Table 7
Portion of the QCA truth table from the meta-analysis dataset including information for the outcome of forest conservation/recovery (DEF = 0) showing the 13 "working forest" cases (out of a total of 33) that showed negative deforestation rates.

Case ID	Case identifier	PA	CMF	INF	POP	AGX	CTL	DEV	MKT	PRI	NDI	CLU	STA	CON	NRM	FOR	IND	REM	DEF
WFMX3	Mexico CFM	0	1	1	1	1	0	1	0	0	1	1	0	1	1	1	1	0	0
WFGU3	Guatemala P6	0	1	0	0	0	0	0	0	0	1	1	1	1	0	1	0	1	0
WFMX4,	Mexico ZM1	0	1	0	0	0	0	1	0	0	1	1	0	1	1	1	1	0	0
WFMX5,	Mexico ZM2	0	1	0	0	0	0	1	0	0	1	1	0	1	1	1	1	0	0
WFMX6	Mexico OEPFZM	0	1	0	0	0	0	1	0	0	1	1	0	1	1	1	1	0	0
WFBO2	Bolivia MIMA	0	1	1	1	1	1	1	1	0	0	0	1	1	1	0	1	0	0
WFPE1	Peru IT	0	1	1	1	1	1	1	0	0	0	1	1	1	1	1	1	0	0
WFBZ2	Brazil ER	0	1	1	0	1	1	1	1	0	1	1	1	1	1	1	0	0	0
WFBZ3	Brazil AJER	0	1	0	0	1	1	0	0	0	0	1	1	1	1	1	0	1	0
WFBO1	Bolivia TIR	1	1	0	0	0	0	0	0	0	0	0	1	1	1	0	0	0	0
WFIA2	India TART (I)	1	1	0	0	0	0	0	0	0	1	1	1	1	1	0	1	1	0
WFIA3	India TART (P)	1	1	1	1	1	1	1	1	0	1	1	1	1	1	0	1	0	0
WFNA1	Nicaragua BNRR	1	1	0	1	1	1	0	0	0	0	1	1	1	1	1	1	1	0
Totals		4	13	5	5	7	6	8	3	0	8	11	9	13	12	9	9	4	13

PA = Protected Area; CMF = Community managed forests ; INF = Infrastructure; POP = population; AGX = agricultural expansion; CTL = cattle; DEV = development and agricultural policy; MKT = market and prices; PRI = private land use tenure; NDI = natural disasters; CLU = communal land use; STA = state owned property ; CON = conservation policy institutions; NRM = natural resource management; FOR = forestry; IND = indigenous populations; REM = remote and inaccessible areas; DEF = deforestation rate greater than −0.2%.

Among the 13 CMFs where the outcome of forest conservation was explained by the QCA analysis, 12 had very low annual forest cover change rates (greater than −0.09). The exception was a case study of an extractive reserve in Brazil with a rate of annual forest cover change of −0.17. Six of the CMF case studies with forest conservation outcomes had forest cover change rates that were negligible (greater than −0.003), showing forest maintenance and even forest recovery. These CMF case studies were located in Bolivia, Guatemala, and Mexico. The most common underlying factors among CMFs with forest conservation outcomes were the presence of conservation policies and institutions, communal land use, government ownership of land, and natural resource management. Community forest management was present in nine of the CMFs (70%) and indigenous populations were also present in 50% of the CMF case studies that presented the outcome of forest conservation, which shows their relevance to conservation within CMFs in the tropics.

3.2.3. Case studies showing no trends

The QCA analysis conducted for the 73 case studies did not explain the outcomes of deforestation or forest conservation as a function of the selected variables for a total of 18 cases (approximately 25% of the sample) (Table 8), which can be considered "contradictory" cases. That is, these 18 cases presented similar causal factors of forest cover change, but did not follow the trends of the remaining 55 cases for which a given combination of outcomes explained either deforestation or forest conservation/recovery. The majority of these 18 case studies pertained to CMFs (14 in Colombia and 2 in Guatemala), while two others referred to PAs (Guatemala). These 18 cases presented deforestation drivers such as infrastructure development, population migration, agricultural expansion, cattle development, and markets. The 14 cases that refer to CMFs within Colombia consisted of indigenous reserves in the Guyana Shield region that were affected by guerrilla and illicit coca cultivation, so included additional factors that might affect

Table 8
Portion of the QCA Truth Table cases that failed to explain either deforestation or forest conservation/recovery using meta-analysis dataset.

Case ID	Case identifier	PA	CMF	INF	POP	AGX	CTL	DEV	MKT	PRI	NDI	CLU	STA	CON	NRM	FOR	IND	REM	DEF
PAGU3	Guatemala LDTNP	1	0	1	1	1	1	1	0	0	1	0	1	1	0	0	0	0	C
PAGU5	Guatemala EZR	1	0	1	1	1	1	1	0	0	1	0	1	1	0	0	0	0	C
WFGU2	Guatemala P2	0	1	1	1	1	1	1	0	0	1	1	1	1	1	1	1	0	C
WFGU1	Guatemala P4	0	1	1	1	1	1	1	0	0	1	1	1	1	1	1	1	0	C
WFCO1	Colombia BC	0	1	1	1	1	1	1	1	0	0	1	1	1	1	0	1	0	C
WFCO7	Colombia BC	0	1	1	1	1	1	1	1	0	0	1	1	1	1	0	1	0	C
WFCO6	Colombia CMD y M	0	1	1	1	1	1	1	1	0	0	1	1	1	1	0	1	0	C
WFCO9	Colombia EI	0	1	1	1	1	1	1	1	0	0	1	1	1	1	0	1	0	C
WFCO8	Colombia IR3	0	1	1	1	1	1	1	1	0	0	1	1	1	1	0	1	0	C
WFCO4	Colombia IR2	0	1	1	1	1	1	1	1	0	0	1	1	1	1	0	1	0	C
WFCO10	Colombia IR1	0	1	1	1	1	1	1	1	0	0	1	1	1	1	0	1	0	C
WFCO12	Colombia LF	0	1	1	1	1	1	1	1	0	0	1	1	1	1	0	1	0	C
WFCO5	Colombia LS	0	1	1	1	1	1	1	1	0	0	1	1	1	1	0	1	0	C
WFCO3	Colombia LY	0	1	1	1	1	1	1	1	0	0	1	1	1	1	0	1	0	C
WFCO2	Colombia PC	0	1	1	1	1	1	1	1	0	0	1	1	1	1	0	1	0	C
WFCO13	Colombia PNP	0	1	1	1	1	1	1	1	0	0	1	1	1	1	0	1	0	C
WFCO14	Colombia PV y PE	0	1	1	1	1	1	1	1	0	0	1	1	1	1	0	1	0	C
WFCO11	Colombia TDCGLP	0	1	1	1	1	1	1	1	0	0	1	1	1	1	0	1	0	C
Totals		2	16	18	18	18	18	18	14	0	4	16	18	18	16	2	16	0	18

PA = Protected Area; CMF = Community managed forests ; INF = Infrastructure; POP = population; AGX = agricultural expansion; CTL = cattle; DEV = development and agricultural policy; MKT = market and prices; PRI = private land use tenure; NDI = natural disasters; CLU = communal land use; STA = state owned property ; CON = conservation policy institutions; NRM = natural resource management; FOR = forestry; IND = indigenous populations; REM = remote and inaccessible areas; DEF = deforestation rate greater than −0.2%.

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forest conservation/recovery outcomes, but were not considered in our analysis. These were CMF cases with high deforestation rates, although included the conditions that explained forest conservation/recovery in other cases.

4. Discussion

The findings presented here support our hypothesis that community managed forests may be at least as, if not more, effective in reducing deforestation as PAs at the pantropical scale. In fact, we found that CMFs included in this analysis had lower annual deforestation rates, which were less variable than those reported for PAs. Our finding that PAs are not always successful at avoiding deforestation concurs with previous research on the topic. For example, Nagendra (2008) conducted a meta-analysis of studies referring to deforestation in PAs and showed that net rates of forest clearing within PA boundaries were reported in 50% of the cases examined. Naughton-Traves et al. (2005) compared the annual deforestation rates reported in the literature for 36 PAs and found that while most had higher pressures outside than inside their boundaries, these outside pressures may signal conservation failures in the long run. Using high resolution data to obtain tropical deforestation rates, DeFries et al. (2005) compared 198 PAs and found that 25% of them had forest losses within their boundaries and that 66% of them had clearings in the surrounding landscapes. Although Bruner et al. (2001) carried out an earlier study using survey data of 93 PAs across 22 tropical countries and found that strict protection seemed as an effective mean for preventing land clearing, their study has been criticized by those who indicate that asking performance to park officials may introduce significant bias (Hayes, 2006; Nepstad et al., 2006; Ostrom and Nagendra 2006; Joppa et al., 2008; Joppa and Pfaff, 2010).

Our findings also mesh with other studies that compare the performance of PAs and CMFs in deterring deforestation. Regional studies on the topic have concluded that both CMFs and PAs can be effective ways of protecting forests, although performance of specific cases is context specific and depends on different factors (Armenteras et al., 2009; Oliveira et al., 2007). Despite the need to acknowledge the specific context of each case study, as in the results presented here, previous researchers have found that both PAs and CMFs can be effective in deterring deforestation. For example, using fire activity as a proxy for tropical deforestation, Nelson and Chomitz (2009) concluded that PAs are successful at lowering the incidence of forests fires, but multiple-use protected areas (where some degree of productive use is allowed) can be as, or more, effective as areas designated for strict protection. In a more localized study, Bray et al. (2008) report that, for the Maya Forests of Mexico and Guatemala, CMFs were more effective in reducing deforestation than PAs, although the difference was not statistically significant. They concluded, however, that although the two management schemes might be effective in reducing deforestation, CMFs were apparently better at delivering local benefits.

Our results also indicate that our hypothesis does not hold at the level of individual case studies since some CMFs in our sample show deforestation rates that are similar to those in PAs. This is because deforestation results from multiple interacting factors that combine to provide site-specific outcomes. In particular, we observed that deforestation pressures do not necessarily result in forest clearing as institutional arrangements may overcome those pressures. Based on QCA results, 10 out of a total 13 CMFs where forest cover conservation or recovery was explained included some combination of deforestation pressures (e.g., infrastructure development, population growth, agricultural expansion, cattle production, and development initiatives). For instance, in community management *ejidos* in Quintana Roo (Mexico), deforestation drivers such as infrastructure development, population growth, agricultural expansion, and development programs do not necessarily result in increased annual deforestation rates mostly because communities have working rules for managing forested areas (Dalle et al., 2006; Ellis and Porter-Bolland, 2008). The *ejidos* in Mexico exemplify how the maintenance of forest cover in CMFs can occur even with the presence of deforestation pressures. The above example also suggests that institutional arrangements for natural resource management play a crucial role that is visible at the landscape scale. Moreover, while 78% of cases that showed conservation/recovery processes explained by the QCA were characterized as spatially remote PAs or located in remote frontier regions, only 30% of CMF case studies under this category included this attribute, and two of them (Brazil AJER and Nicaragua BNRR; Table 1) also displayed deforestation pressures. Again, this underscores the issue that many context specific factors play a role in explaining changes in forest cover over time.

As far as maintenance of forest cover is concerned, we found ineffective PAs present in tropical regions of all continents reviewed (America, Southeast Asia, and Africa). Community managed forests were underrepresented as they included areas mostly in the American tropics (and two case studies in India). Yet, they seem to provide good examples of forest maintenance at the landscape level. Our case studies and the QCA results in general, and in particular the "contradictory" cases (see 'Case studies showing no trends'), show the complexity involved in assessing changes in land use/cover in both PAs and CMFs in the tropics. This finding calls for caution when making generalizations. For instance, the Colombian Guyana Shield study (Armenteras et al., 2009) provides examples of CMFs cases with both high and minimal rates of deforestation under similar conditions. The authors attribute these disparities to the presence of guerrilla and illegal coca growing activities, which is further influenced by differential colonization of non-indigenous populations across localities, which may be breaking down institutional arrangements where the guerilla may be exerting their influence. For their part, the two CMF case studies in Guatemala that could not be explained by the QCA (Guatemala P4 and P2) were both community forest management concessions also subject to deforestation drivers such as roads, colonization, agricultural expansion, cattle production, and development programs. In this case, Bray et al. (2008) indicate that one of the cases was a recently inhabited community forest concession that had a high annual rate of forest cover loss (−0.72), while the other was a long-inhabited community forest concession with a much lower rate of forest loss (−0.02). There were two PAs in Guatemala that the QCA failed to explain that included the same deforestation pressures and conservation policies and institutions present in other cases. Yet, the smaller biological reserve (Guatemala SLNP) showed a low annual rate of forest cover loss (−0.09), while the larger national park (Guatemala LDTNP), which experienced more colonization, had a much higher deforestation rate (−0.33; Sader et al., 2001).

We believe that the most important finding of our analysis is that, although the QCA results do demonstrate that deforestation occurs both in PAs and CMFs, there were higher deforestation rates in PAs than in CMF on a pantropical scale. In this regard, Chhatre and Agrawal (2009) argue that local ownership and autonomy in rule making positively influence forest outcomes regarding forest dynamics. This argument is also supported by Hayes' (2006) findings in that conservation outcomes were largely influenced by the rules made and acknowledged by local forest users; the author further argues that PAs may not be the optimal governance structure for promoting forest conservation. However, broader social and political processes, and the existing legal frameworks at different levels may also interact in determining how local rules affect

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conservation outcomes. Colding and Folke (2001) indicate the importance of social taboos as informal institutions that have positive roles for biological conservation. It may also be important to consider that broader social and political processes, and the existing legal frameworks that determine local rules, are also important (Chhatre and Saberwal, 2005). These social and political dimensions are some of the fundamental aspects that make PAs vulnerable and that must be considered crucial in conservation debates (Brechin et al., 2002).

We recognize that our findings are circumscribed to both the number and type of case studies selected particularly for CMFs which may be underrepresented from a potentially larger sample. The peer-reviewed literature seems to provide few examples on the environmental performance of this forest management type both in the number of cases and their geographical spread (our sample is biased towards the neotropics). Naughton-Treves et al. (2005) also found that deforestation studies in Africa were scarce, particularly in comparison to studies in Latin America. Selection bias may further generate misleading results if, for example, researchers working on CMFs have mostly selected ''successful'' CMFs to conduct their analysis, or if there is a tendency to publish only significant results (the ''file drawer problem''; Fernandez-Duque and Valeggia, 1994). Under such circumstances, the published case studies selected here will likely reflect a biased sample. Further, we also acknowledge the possibility that CMFs in our sample show lower deforestation rates than PAs as a function of historical patterns in forest cover change across space and time (i.e., a country's 'forest transition curve'; Rudel et al., 2005). For example, PAs may show higher annual rates of deforestation because they could have been established where threat of forest conversion to other uses were high. On the contrary, CMFs may show lower annual deforestation rates because they could have been allocated under specific circumstances where either the threat or the perceived consequences of deforestation were deemed not as serious.

Despite these potential biases, our results nevertheless suggest that tropical forest PAs may not always represent the best way to conserve forests *vis à vis* tropical forests locally managed for production of goods and services. A complement of different management strategies may be needed in order to integrate a more resilient and robust conservation strategy in tropical landscapes (Mascia and Pailler, 2011). This vision requires the integration of development variables (i.e., rights, capacity, governance, and revenue) into conservation objectives (Balint, 2006). Although some authors have shown skepticism with respect to the broad adoption of multiple-use forestry models (Bowles et al., 1998; García-Fernández et al., 2008) and although there are clear challenges in balancing tropical forest conservation with local livelihood development (Kusters et al., 2006), there is evidence that applying specific institutional arrangements may work towards this needed balance (Bray et al., 2008; Hayes and Persha, 2010). This recognition is important when considering both existing and potential roles of diverse alliances for conservation (Nepstad et al., 2006; Schwartzman and Zimmerman, 2005).

Our results do not provide information regarding potential loss of the provision of goods and services under no change in forest cover (i.e., ''forest degradation''; Sasaki and Putz, 2009) as we have no way to determine how (un)sustainable those practices may be within the set of case studies that included community management. For example, Shahabuddin and Roa (2010) claim that while forest management in community conservation areas may show an improvement over open access areas in terms of reducing deforestation pressures, forest quality was not necessarily maintained over time. That said, and to fully realize the positive role that locally-based management may have in the conservation of tropical forest cover, further research and development efforts may be needed to distill existing obstacles impeding the implementation of sustainable management practices in a community context. The large amount of research on the subject of community-based forest management in developing countries has led some authors to synthesize the main factors underlying its success (see e.g., Agrawal and Angelsen, 2009 and references therein). This has implications for site selection, design and implementation of incentive schemes aimed at reducing emissions from deforestation and forest degradation and enhancing forest carbon stocks in developing countries (REDD+; Angelsen et al., 2009) particularly given the millions of ha of tropical forest currently under control of local and indigenous communities (Sunderlin et al., 2008). Targeting community managed forests for the purposes of maximizing the success of implementation of REDD+ schemes may be a sensible approach to follow by further discerning under what biophysical, institutional, market and policy settings, community managed forests are more likely to persist in time and space in relation to other types of forest conservation strategies (e.g., Hayes and Persha, 2010; Phelps et al., 2010). At a minimum, tropical forest managers and practitioners will have to work towards the application of environmentally friendly norms that go beyond timber, that incorporate ecosystem goods and services of both local and global significance, and that are adaptive, inclusive, efficient and flexible (Guariguata et al., 2010; Lawrence, 2007; Michon et al., 2007; Nasi and Frost, 2009) under fair and equitable tenure and resource access regimes (Larson et al., 2010).

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