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Complex Tenure and Deforestation: Implications for Conservation Incentives in the Ecuadorian Amazon

MARGARET B. HOLLAND
University of Maryland, Baltimore, MD, USA

FREE DE KONING
Conservación Internacional—Ecuador, Quito, Ecuador

MANUEL MORALES
Corporación ECOLEX, Quito, Ecuador

LISA NAUGHTON-TREVES
University of Wisconsin-Madison, WI, USA

BRIAN E. ROBINSON
University of Minnesota, St. Paul, MN, USA

and

LUIS SUÁREZ *
Conservación Internacional—Ecuador, Quito, Ecuador

Summary. — This paper analyzes deforestation in areas of overlapping land tenure in the northern Ecuadorian Amazon. We use a random coefficients model to test for differences in forest cover across tenure forms over time. Tenure categories are significantly associated with changes in deforestation, even after controlling for multiple factors. Deforestation slows dramatically in the latter time period; and model results link parks with reduced deforestation. The same is true for lands where indigenous territories overlap with forest protection. Our results suggest that Ecuador's conservation incentive program could refine its targeting by focusing on indigenous areas and communal lands outside of parks.

Key words — land tenure, deforestation, protected areas, indigenous territory, payment for ecosystem services, REDD, Ecuadorian Amazon

1. INTRODUCTION

Land tenure is a key factor affecting landholders' investment decisions on property and forest use. The impact of land tenure conditions on forest outcomes is difficult to predict given that in any area land tenure can be intertwined with multiple other factors shaping land use in unique ways. Thus much of the recent research on land tenure and tropical forest conservation is largely limited to comparing deforestation rates inside and outside protected areas (Andam, Ferraro, Pfaff, Sanchez-Azofeifa, & Robalino, 2008; Joppa & Pfaff, 2010, 2011). Other studies focus on the variable land use outcomes related to tenure security vs. insecurity and show that securing property rights can slow deforestation, or hasten agricultural expansion depending on local context (Barbier & Burgess, 2001; Robinson, Holland, & Naughton-Treves, 2011). Many studies of land tenure and deforestation assume that local land can be clearly classified into a single category (e.g., private, communal, or public) when in fact a given area may be subject to overlapping or even contradictory designations. This paper recognizes the complexity of land tenure in tropical forests and asks how various and overlapping *forms* of tenure are associated with deforestation rates. With a detailed dataset on land tenure designations, infrastructure, and population data over nearly two decades in an ecologically important development frontier of the Ecuadorian Amazon, we provide a novel assessment of forest change, including in areas where tenure is clear and where there is ambiguity and overlap in tenure regime.

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Land tenure has emerged as a critical yet poorly understood component of incentive-based conservation mechanisms, such as Payments for Ecosystem Services (PES). The highest profile of these incentive-oriented policies, REDD (Reducing Emissions from Deforestation and Degradation), is part of the ongoing negotiations related to an international climate change mitigation strategy for post-Kyoto Protocol. Tenure is a key equity concern given that it determines the flow of benefits and assignation of responsibility for forest protection (Bruce, Wendland, & Naughton-Treves, 2010). Tenure regimes are often complex and overlapping in areas targeted for REDD, and this condition deserves more careful attention. Further, there is a lack of empirical analysis regarding the relationship between certain forms of land tenure and forest change.

Here, we look to the two northernmost provinces within the Ecuadorian Amazon (Sucumbíos and Orellana) and pose three questions:

1. Is there a significant variation in forest change across different forms of land tenure?
2. Are forest outcomes markedly different for areas where tenure forms overlap?
3. How might the interplay between land tenure form and deforestation help inform the implementation of a forest conservation incentive mechanism, such as Socio Bosque, which is included as a component of Ecuador's REDD strategy?

For this analysis, we conceptualize land tenure as it is defined by the US Agency for International Development (USAID): the institutional framework that determines how land (and its related resources) is accessed by individuals or groups, who is allowed to possess and use the land resources, under what conditions, and for how long (USAID, 2008). Tenure *form* refers to the individual, group, or entity to which the rights to the land are attributed and then administered. The most common forms tend to fall into four categories: individual, communal, state, and open-access (USAID, 2008). In this paper, we focus on state, individual, and communal land tenure forms in the northern Ecuadorian Amazon, paying special attention to areas where communal tenure overlaps with state forms.

Overlapping land tenure is of particular interest since it could be associated with contested, unclear or uncertain access rules, and thus imply potential tenure insecurity. Conversely, overlapping tenure could bolster, or reinforce, access rules in cases where such rules do not conflict, even if there are multiple forms. Specifically relevant to the Ecuadorian Amazon, where indigenous communities hold communal title, we aim to test whether indigenous areas that overlap with protected and restricted use areas have significantly different deforestation rates relative to other types of landholdings. In theory, this could have either a positive or negative effect on deforestation. Given that indigenous lands have management goals beyond forest preservation, one would expect higher deforestation rates than areas designated "protected" by the state. However, empirical analyses reveal highly variable deforestation rates, and in some cases indigenous lands appear more effective in maintaining forest (Nepstad *et al.*, 2006; Porter-Bolland *et al.*, 2011).

Proximity to roads, local institutional strength, and national government recognition are among the factors complicating this simple comparison (Nepstad *et al.*, 2006; Porter-Bolland *et al.*, 2011). Previous studies on deforestation and tenure ambiguity offered mixed signals: uncertain tenure status can discourage or hasten deforestation depending on an array of socioeconomic conditions (Robinson *et al.*, 2011). Few studies address overlapping tenure despite the fact that many indigenous areas are inscribed within or overlap with protected areas (Naughton-Treves *et al.*, 2006). Here we examine overlapping tenure as a distinct condition in itself. Because our definition of tenure form implies that each individual, group, or organizational identity can determine the access rules and management of resources, we assume that not all indigenous lands behave similarly with respect to deforestation. Where indigenous lands overlap with state-managed protected areas, we expect deforestation rates to be higher than in nonoverlapping, or "pure" protected area land. Outside of overlaps with tenure forms emphasizing forest protection and management, we expect forest change on indigenous lands to reflect similar rates to privately-owned lands.

(a) *Study region*

The dense forests of Ecuador's northern Amazon region (and the provinces of Sucumbíos and Orellana, specifically), contain globally significant biological and cultural diversity, and store significant carbon in the form of biomass. They also grow above substantial oil reserves (Figure 1). Land in this area is subject to multiple designations with different rules concerning deforestation and access to resources (Naughton-Treves *et al.*, 2006). We use a random effects model to test forest change across different tenure categories, overlapping forms, and across two time periods (1990–2000 and 2000–08). Recognizing that local drivers of Amazonian deforestation can change abruptly with national-level political change (Alvarez & Naughton-Treves, 2003), we note that these two time periods are marked by distinct political and economic conditions, as well as land tenure and land use developments. Previous research in the region reveals that deforestation is associated with road construction triggered by oil exploration, and is also affected by sociopolitical factors, especially indigenous vs. colonist land use practices (Mena, Bilsborrow, & McClain, 2006; Mena, Barbieri, *et al.*, 2006; Pan, Carr, Barbieri, Bilsborrow, & Suchindran, 2007).

This region is also targeted for Ecuador's national forest conservation incentive program, *Socio Bosque* or *Forest Partners*, a program aiming (1) to conserve 36,000 km^2 of forest and other native ecosystems, and (2) safeguard livelihoods and increase income for between 0.5 and 1.5 million people nationwide (de Koning *et al.*, 2011). One-quarter of the total national area and 16% of the agreements currently enrolled in Socio Bosque lie in this study region.

(b) *Background*

Increasing encroachment into the Amazon is a result of the dynamic frontier interplay of road expansion, resource extraction activities, and agricultural settlement. But the pattern and rate of deforestation are affected by more than the proximity of roads and rivers or human numbers—land tenure and land policy shape outcomes (Geist & Lambin, 2002). Amazonian frontier forest is typically cleared more rapidly in areas where land ownership is uncertain (Araujo, Bonjean, Combes, Combes Motel, & Reis, 2009). Here individuals clear forest to claim land and insure against the risk of expropriation or invasion (Fearnside, 2001). At a broader scale, official land use designations also affect forest clearing and fires, despite pervasive problems of "paper parks" and weak enforcement of indigenous reserves (Nelson & Chomitz, 2011; Nepstad *et al.*, 2006). If rural colonization policies, commercial resource extraction and road construction propel the initial sweep of forest loss in these frontier regions, land tenure shapes finer patterns of forest conversion.

Figure 1. *The study region spans the two northeastern Ecuadorian provinces of Sucumbíos and Orellana.*

The discovery of oil in 1967 by the US Texaco-Gulf consortium effectively opened access to the northern Ecuadorian Amazon (Uquillas, 1984). The sequence of settlement and change in the forest frontier in this region reflects initial waves of spontaneous colonization and state-driven deforestation through the land settlement incentives and other related policies starting in the 1970s, to more enterprise-driven deforestation by the 1990s (Rudel, 2007). Colonization resulted in deforestation (−2.3%/year estimated for the entire Ecuadorian Amazon, or *Oriente*, during 1977–85) and rapid population growth (Rudel & Horowitz, 1993; Bilsborrow, Barbieri, & Pan, 2004; Sierra, 2000).

Two agrarian reform laws (1964 and 1973) brought a rapid influx of Andean colonists to the area, and have reshaped land use. The 1964 law classified large portions of Ecuador's Amazon region as unsettled (*tierras baldías*), thus ignoring the ancestral territories of multiple indigenous groups (Bremner & Lu, 2006). As elsewhere in Latin America, land under "productive use" was eligible for title and credit (Morales, Naughton-Treves, & Suarez, 2010). Although the laws were changed in 1994, many settlers continue to associate forest clearing with improved tenure security (Morales *et al.*, 2010).

The 1990s brought political and economic turmoil to Ecuador, marked by the entry and exit of five different presidents and inflation rates that frequently surpassed 50% (Beckerman, 2001). During this period, both oil development and land use policy were subject to abrupt shifts. In 1992 the Durán administration pulled Ecuador out of OPEC, and immediately increased the country's oil production targets by more than 50% (Alsalem, Sharma, & Troutt, 1997). Then in the mid-1990s, the Government of Ecuador signed contracts with foreign companies releasing the rights to indigenously-controlled and environmentally sensitive lands that were previously restricted. The government also shifted forest reserve boundaries and designated special zones in parks to accommodate oil drilling (Naughton-Treves *et al.*, 2006). With the oil boom came accelerated population growth rates: over 5% in our study region, compared with 2.2% across the rest of Ecuador (INEC, 2011). By 2001, approximately 300,000 people inhabited the region encompassed by our study area and the province of Napo (the northern Ecuadorian Amazon), representing an overall population increase of 63% from 1990 counts (INEC, 2011).

For the 30 years after its discovery in 1967, oil from the Ecuadorian Amazon represented more than 50% of the country's exports and government revenue. By the late 1990s, earnings began to fall as national GDP stagnated, and the country formally declared 1998–99 a financial crisis (Valdivia, 2005). The new millennium saw significant structural reforms in Ecuador. During 2002–06, the economy grew by 5.2% on average, bolstered by high oil prices, remittances, and an increase in nontraditional exports (Beckerman, 2001). Despite a trend toward a stabilizing economy after 2000, political turmoil continued and three more presidents took office during 2000–07.

Rafael Correa took office as president in 2007, and by 2008, the Government of Ecuador passed a new Constitution. A key component of the new Constitution was to define indigenous tenure over ancestral lands as communal and indivisible, effectively assigning indigenous peoples' legal rights to more than half of Ecuador's remaining forest land. In the same year, Ecuador launched the Socio Bosque program.

Meanwhile, in our study area, although the rate of population growth has slowed in recent years (3.5% for Sucumbíos province, 5.1% for Orellana), the population of the northern Ecuadorian reached nearly 417,000 inhabitants by 2010 (INEC, 2011). Oil production has continued apace, facilitated by the fact that the Government of Ecuador maintains all subsurface rights, regardless of individual or indigenous communal title to surface lands. In fact, oil extraction occurs in several sites within national parks in the study region (Finer, Jenkins, Pimm, Keane, & Ross, 2008; Finer, Moncel, & Jenkins, 2010).

The pattern and pace of forest loss in the region have also been influenced by shifts in national forest conservation policy (Pan *et al.*, 2007; Sierra, 2000). Previous research points to the influence of rapid migration within the forest frontier, increasing urbanization within the region, declining fertility, and household lifecycles on forest change and overall land use in this study region (Barbieri, Bilsborrow, & Pan, 2006; Carr, Pan, & Bilsborrow, 2006). Most agricultural production by colonists to the region has remained small-scale, with an emphasis on a mix of subsistence food crops, livestock for domestic consumption, and some cash cropping (primarily coffee) (Marquette, 1998). During the early 1990s, 10% of the country's coffee production came from this region, and by the end of the decade the high demand for on-farm labor to cultivate coffee influenced a slowing in forest conversion (Hicks, 1990; Marquette, 1998). This study presents an opportunity to explore some of these macro-level influences of population and extractive industries (including oil and mining) on forest dynamics in the region using updated forest change data. Furthermore, our analysis explores these factors alongside tenure forms. Previous studies refer to the importance of local land tenure for forest conservation (e.g., Messina, Walsh, Mena, & Delamater, 2006), but this has yet to be tested specifically for overlapping forms of tenure or with updated forest change.

(c) *Land tenure dynamics in the Ecuadorian Amazon*

The development and exploitation of natural resources, along with rapid political and social change in the Ecuadorian Amazon have shaped land tenure and resource access rules, taking complex and sometimes contradictory form. This complexity is evident in the formal administration of land. Rural lands in Ecuador presently fall primarily under the jurisdiction of two agencies: the sub-Secretariat of Lands within the Ministry of Agriculture (MAGAP) and the Ministry of the Environment (MAE). Lands administered by MAGAP were previously under the jurisdiction of the Agrarian Development Institute (INDA, from 1994 to 2010), and before that, under the Ecuadorian Institute for Agrarian Reform and Colonization (IERAC, from 1973 to 1994). When MAE was created in 1996, all lands within the forest patrimony and protected forests, which had previously been administered by the Institute for Forestry, Natural Areas and Wildlife (INEFAN), were handed over to MAE.

The turnover in land management agencies has intensified titling challenges and tenure insecurity. Acquiring a title can be slow and costly: roughly $1500 USD per individual title, or $30/hectare for average landholdings in our study region (Freire, J. L., personal communication, May 28, 2012). Of those lands now administered by MAGAP and MAE, close to 81,000 km^2 had yet to be titled by 2000 (27% MAGAP lands and 10% of MAE lands) (Morales *et al.*, 2010). Untitled lands are still found in remote forested areas, including within our study region (Morales *et al.*, 2010). Furthermore, contested claims are common, even to titled land (as much as 30% of Ecuador's land area) (Morales *et al.*, 2010). In some conflict areas, occupants convert forest in order to establish their claim.

(d) *Land tenure categories*

Our study area spans ~40,000 km^2 across two provinces, Sucumbíos and Orellana. We define five distinct forms of tenure across this study region: (1) Protected Areas (PAs), (2) forest patrimony areas (PF), (3) protected forests (BP), (4) indigenous lands, and (5) lands held privately or as colonization areas adjudicated by the newly created sub-Secretariat of Lands within the Ministry of Agriculture, Livestock, and Fisheries (MAGAP) (Table 1 and Figure 2). More than half of the region is designated as indigenous lands (60% of study region), followed by private/MAGAP lands (21%), and protected areas (15%), according to the most recent figures.

According to Ecuador's Forestry Law (Ley Forestal, 1981), forest extraction for commercial purposes is not permitted in the national system of protected areas. This extends to indigenous lands located within protected areas, although cutting of forest for subsistence use by indigenous groups in parks is permitted (Morales *et al.*, 2010). More flexible than park law, protected forests (*bosques protectores, or BP)*, and the forest patrimony areas of the State (*patrimonio forestal del Estado, or PF)*, are restricted-use categories originally created as part of the same Forestry Law (1981). Protected forests can be privately-owned or publicly-held and fall under the jurisdiction of MAE. Forest patrimony areas are recognized as the property of the government; however they may be converted to private or communal ownership through a petition and adjudication process administered by MAE. Only once the lands have been adjudicated and transferred can forest extraction occur. As such, it must be submitted as a plan and approved by MAE (Ley Forestal, 1981). For indigenous territories (*tierras ancestrales*), land title is recognized as communally-held and cannot be sold. Forest extraction for commercial gain in indigenous communal areas is permitted only if the community has a forest management plan approved by MAE, and only on lands outside of protected areas.

In our study region, the three forms of forest protection and management (protected areas, protected forests, and forest patrimony areas) do not overlap, but indigenous territories are circumscribed in all three (Figure 2). Thus we separately analyze three categories of spatial overlap: indigenous areas with (1) protected areas, (2) protected forests, and (3) forest patrimony areas. This categorization does not represent the full set of conflicting claims in this region (e.g., indigenous territories also overlap with lands held by the MAGAP (Morales *et al.*, 2010), but it covers the majority of forested land.

2. METHODS

(a) *Data sources*

(i) *Deforestation*

We worked with the Ministry of the Environment (MAE) to acquire a portion of the Historical Deforestation Map of Ecuador for 1990, 2000, and 2008 (MAE, 2011a). MAE pro-

Table 1. *Descriptive statistics for tenure categories within study region, separated according to those present in the first time period (T1), 1990–2000, and the second time period (T2), 2000–08*

Tenure form (singular and overlapping)	# Areas	Total area in study region (km^2)	Average size (Mean, km^2)	% Study region	Permitted forest use
Private/MAGAP lands					Forest extraction for commercial gain permitted where privately owned
T1	Undefined	8564.46	n/a	21.5	
T2	Undefined	8286.23	n/a	20.8	
Protected areas (PA)					Strict conservation
T1	4	5717	1429.2	14.4	
T2	6	5995.2	999.2	15.1	
Forest patrimony areas (PF)					Restricted use forest extraction permitted with plan only if privately owned
T1	7	1588.8	227	4	
T2	7	1588.8	227	4	
Protected forests (BP)					Restricted use forest extraction permitted with plan only if privately owned
T1	7	272.8	39	0.7	
T2	7	272.8	39	0.7	
Indigenous community lands (nonoverlapping)					Forest extraction for commercial gain allowed with plan (communal ownership)
T1	11	10215.8	928.7	25.7	
T2	10	9519.7	952	23.9	
PA overlap with Indigenous community lands					Subsistence forest extraction permitted (communal title and government-owned)
T1	13	8375.6	644.3	21.1	
T2	15	9020.8	601.4	22.7	
PF overlap with Indigenous community lands					Restricted use forest extraction permitted with plan (communal ownership)
T1	14	4261.4	304.4	10.7	
T2	14	4261.4	304.4	10.7	
BP overlap with indigenous community lands					Restricted use forest extraction permitted with plan (communal ownership)
T1	3	804.9	268.3	2	
T2	5	858.8	171.8	2.2	

duced this Map to determine the national forest baseline and estimates for CO_2 emissions from deforestation. Using Landsat and ASTER as the source imagery, the final land use product maintains a pixel resolution of approximately 30 m^2. Land use was classified separately for each reference year, using the six Level 1 land use categories as defined by the Intergovernmental Panel on Climate Change (IPCC): forest, agriculture, grasslands, wetlands, settled areas, and other (MAE, 2011a). After image pre-processing, analysts used an unsupervised classification algorithm to perform the initial classification, and then corrected any thematically or spectrally-mixed pixel issues using visual and manual editing. The accuracy assessment for the final land use product resulted in an overall Kappa coefficient of 0.7 for the entire country (MAE, 2011a).

Using the land use maps as our base, we masked out any cloud cover from all three reference years and combined and calculated forest cover and change following Tabor, Burgess, Mbilinyi, Kashaigili, and Steininger (2010). We then calculated the forest baselines for 1990 and 2000, along with observed deforestation for our two time periods of analysis: 1990–2000 (T1) and 2000–08 (T2). We defined observed deforestation as that which was known forest in the first date and visibly deforested by the second date. With our change calculations, we retained information on the resulting land use post-deforestation.

We generated a grid of 1-km^2 cells across the study region from which to derive observations of deforestation rates. We chose a grid cell size of 1-km^2 given that it captures variability from the land use change classification. It also represents one to two times the size of the average land settlement parcel for this region from the agrarian reform period (on average 50 hectares). For each grid cell, we calculated the percent area in forest base as of T1 for each time period, as well as the percent of the forest base deforested by T2, as defined in Tabor *et al.* (2010).

(ii) *Factors associated with land use change*

For each grid cell, we calculated a cell's proximity (km) to five factors associated with deforestation, as informed by past research (Barbieri, Carr, & Bilsborrow, 2009; Carr *et al.*, 2006). For each grid cell we calculate the proximity (km) to the nearest road, navigable rivers, markets (towns with more than 1000 people), and oil production infrastructure, all of which have been linked with higher deforestation since the 1990s (Mena, Bilsborrow, *et al.*, 2006; Mena, Barbieri, *et al.*, 2006). We also include the distance to mining concessions, mean elevation, soil fertility and, with census data at the parish level, population density (people per km^2) in each time period. All datasets related to transportation networks, population centers, oil and mining concessions were provided by MAE and published by various government entities.

Our dataset allowed us to assess the deforestation impacts associated with two prime macro-level economic forces, the mining and oil industries, alongside impacts related to land tenure characteristics. Although the literature has addressed general links between forest cover and macro-forces (Burton & Berck, 1996; Deacon, 1994; Wunder, 2005), the relative effect of macro forces and institutional factors like land tenure deserves greater attention. Here we were able to examine the relative effects of these two important factors.

(iii) *Tenure form*

The data for Protected Areas (PAs), protected forests (BP), and forest patrimony areas (PF) were provided by MAE, Ecolex, and CI-Ecuador. We defined indigenous lands through two sources: Ecolex (2009) for area within the

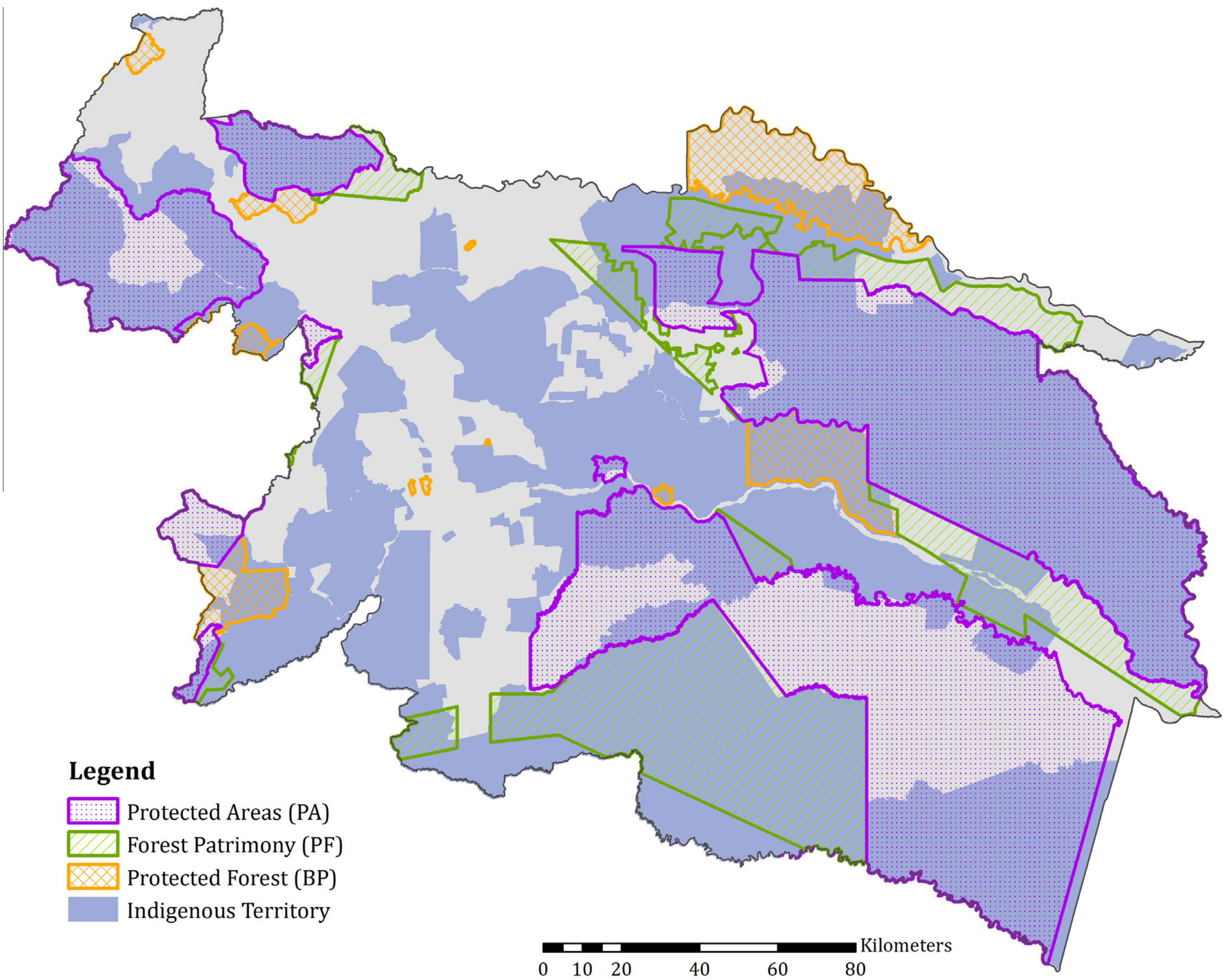

Figure 2. *Map of study region with spatial distribution of main tenure categories. (Note: All land in gray is considered private or held by MAGAP without title.) Source: MAE (2010) and Sierra and Maldonado (2009).*

Cuyabeño Reserve; Sierra, Maldonaldo, & Zamora (2011) for elsewhere in the study region. While not all of these indigenous areas are formally delimited yet according to law, they reflect the most comprehensive and up-to-date map for the study region.

Each grid cell was assigned a binary value corresponding to its unique tenure category or combination of tenure categories. For the statistical analysis, we removed any observations for which tenure changed between the two time periods. SI Table 1 provides descriptive statistics for these data across tenure categories.

(b) *Estimation strategy*

Since the form of land tenure remains static for our overall time period of analysis, a random coefficients (or hierarchical multilevel) model can appropriately take advantage of the panel nature of our dataset to control for time-varying unobservables and also identify the effect of the tenure parameters of interest. We estimated a reduced form equation of the net benefits of deforestation in grid cell i in municipality (*canton*) j in time period t as

$$y_{ijt} = \alpha + \beta x_{ij} + \gamma z_{ijt} + \eta d_t + \zeta_{0j} + \zeta_{1j} x_{ij} + \varepsilon_{ijt} \quad (1a)$$

$$= (\alpha + \zeta_{0j}) + (\beta + \zeta_{1j}) x_{ij} + \gamma z_{ijt} + \eta d_t + \varepsilon_{ijt} \quad (1b)$$

where y_{ijt} is the natural log of the annualized deforestation rate. The variable x_{ij} represents time-invariant characteristics that affect the net benefit of forest clearing, like tenure form, distance variables, and location characteristics (elevation, soil fertility index, and forest area in 1990). Variable z_{ijt} denotes time-varying characteristics that include population density in time t and a measure of the dynamic spatial nature of deforestation. Here we used the percent deforestation in the first time period (y_{ij1}) as a predictor of deforestation in the second time period. Variable d_t is a time dummy that equals one for the second time period and captures unobserved time-varying characteristics common over the study region that might influence land use such as urbanization, migration, population growth, and broader macro-level policy changes, such as shifts in oil prices, that are not captured in the distance to mining and oil concession lands. To aid interpretability, all continuous data were centered at their cluster means (Enders & Tofighi, 2007) so all results are relative to the average values of the data in our dataset.

Rearranging terms, Eqn. (1b) highlights how the random intercept and slope parameters ζ_{0j} and ζ_{1j} modify the standard estimation parameters α and β, respectively. In our model the random coefficient ζ_{0j} is canton j's estimated deviation from the mean intercept α, and ζ_{1j} is a tenure category in canton j's deviation from the mean effect β on the relationship (slope) between covariates and deforestation. Terms γ and η are additional parameters to be estimated and the term ε_{ijt} is the time-varying residual error.

The random coefficients model decomposed the error structure of the model into its nesting components so that strict independence across spatial units is not assumed, helping control for spatial autocorrelation (Anselin, 2002; Wendland *et al.*, 2011). With random intercepts at the canton level, this model controls for correlation across cantons. Model specifications that also include nesting at grid cell level additionally control for spatial correlation among grid cells. Including this additional level did not affect the results of our models, so spatial dependence seems to have little influence on our estimates. As an additional check, we calculated Moran's I for spatial dependence among model residuals.

We are primarily interested in the *average* association between tenure on deforestation rates, thus we focus our discussion on the parameter coefficients β rather than the canton-specific deviations from the average, ζ_{1j}. Nevertheless, the random slope structure is important in our estimation since it allows for canton-specific heterogeneity in tenure's effect on deforestation. If this heterogeneity is important, not accounting for it downward biases the standard errors of β, making parameters more precisely estimated relative to a random coefficient specification, implying potentially unwarranted significant associations with the dependent variable (Rabe-Hesketh, Skrondal, & Pickles, 2005; Vance & Iovanna, 2006). If this heterogeneity is not important, parameter estimates will not be significantly affected. In the results that follow we present a random-intercept model for comparison to show the influence of canton-specific tenure relationships on the *average* effects of tenure on deforestation. Importantly, for unbiased estimates the model still assumes the random terms are uncorrelated with the covariate measures and ε_{ijt} is uncorrelated with ζ_{0j} or ζ_{1j} (Rabe-Hesketh & Skrondal, 2012). Random intercepts and slopes are also assumed independent across cantons, and the overall error term is assumed independent across cantons and grid cells.

(c) *Econometric issues*

There are two main econometric issues that affected our choice of estimation strategy. First, our dataset contains a nontrivial amount of observations for which we observe zero deforestation in either time period (~45% grid cells). Datasets exhibiting a "corner solution" in the dependent variable are often estimated via a Tobit model, but existing adaptive quadrature techniques that allow for multilevel specifications with limited dependent variables (e.g., Rabe-Hesketh *et al.*, 2005) are not compatible with our dataset. However, the linear random coefficients model used here still provides accurate information on the average relationship between our independent variables and deforestation, particularly at the mean of the data (Wooldridge, 2002, p. 525).

Second, there may be endogeneity between the assignment of tenure and deforestation activity, meaning that tenure may be sought or assigned *due to* deforestation rates as opposed to tenure having a causal influence on deforestation rates. A growing literature uses methods that attempt to create credible counterfactual outcomes for the observed land use choices, allowing the researcher to estimate the difference in the observed outcome from what might have happened in an alternate universe, thus effectively controlling for the potential endogeneity of the observed outcomes (e.g., Andam *et al.*, 2008; Joppa and Pfaff, 2010). Such methods, however, are largely limited to studies in which one can define a "treated" vs. "untreated" group (for instance, areas that are protected vs. not protected). Here we use a regression approach since we are interested in effects over multiple tenure categories. Further, adequately controlling for covariate measures like distance and location features have produced similar results to matching approaches in other studies (Butsic, Lewis, & Ludwig, 2011; Pfaff *et al.*, 2009; Pfaff *et al.*, 2014).

If endogeneity *is* significant in the tenure–deforestation relationship, our parameter estimates on tenure categories could be biased. For example, if groups seek tenure to strengthen land rights to prevent deforestation, then parameter estimates on associated tenure categories would likely overstate the protective effect of tenure. On the other hand, if high rates of deforestation help landholders obtain private land rights with, say, the intent to engage in agriculture, then estimates associated with these categories of tenure are likely to overstate their propensity for deforestation. However, even in these cases we cannot make clear predictions about the direction of the endogenous effect on our parameter estimates since potential correlations between tenure variables and other regressors could change the predicted sign of bias.

A final issue concerns ensuring our dataset is appropriately comparable across tenure categories. We drew from the matching literature's focus on confirming common support among the treated and untreated groups (Caliendo & Kopeinig, 2008) and removed outlying values from our dataset that were not statistically comparable with values in other tenure categories. Here we define "common support" for values of each continuous independent variable as overlap in the statistical distribution (observations that fall between the 5th and 95th percentile) of at least two tenure categories. This definition reduces our original dataset from 56,565 observations to 34,092. Using more strict definitions of common support that require observations to have support in three (25,864 observations) and four (18,190 observations) tenure categories does not affect the sign, significance, or relative magnitude of our parameter estimates.

3. RESULTS

(a) *Absolute forest change*

During the first time period (1990–2000), 4076 km^2 (12.1% of forest area) was deforested (Figure 3, Table 2). This represents an annual deforestation rate of −1.3%, nearly double Ecuador's national rate for the same time period (MAE, 2011a). Both the extent and percent of deforestation decrease dramatically for the second time period (2000–08), to 1125 km^2 and 3.8% of forest area respectively, reducing the annual deforestation rate to −0.5% for the study region, slightly less than the national rate of −0.6% (MAE, 2011a).

Looking across the tenure categories, protected areas experienced the lowest percent and rate of deforestation across both time periods. The fractional loss of forest, (representing the annual rate of deforestation), remained level throughout the study for forests in PAs, whereas it dropped significantly for all other categories during second period (2000–08,

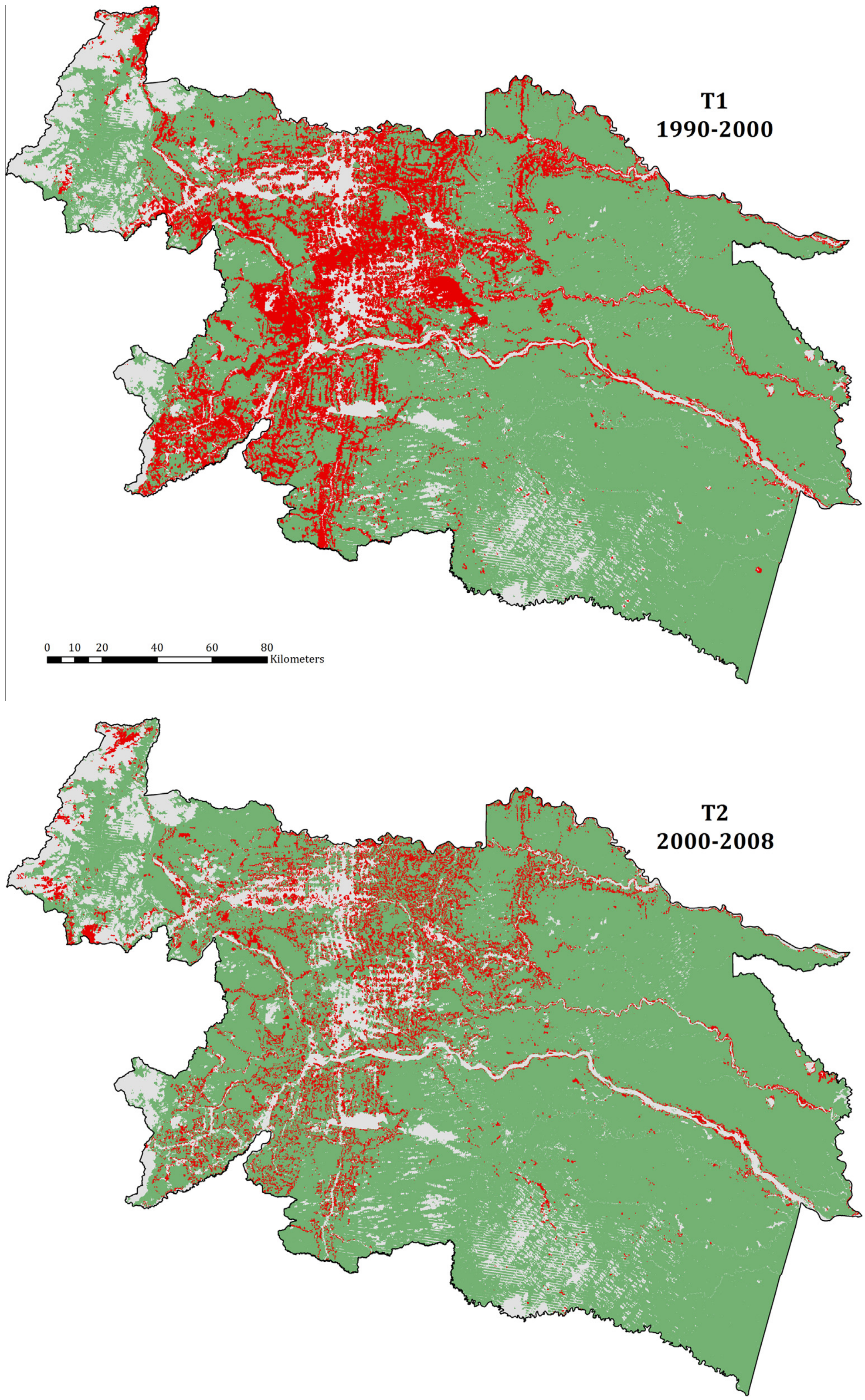

Figure 3. *Observed deforestation in the study region (provinces of Sucumbíos and Orellana) during the two time periods of analysis: 1990–2000 (T1) and 2000–08 (T2).*

Table 2). As expected, the rate of deforestation remained highest for the mixed category of private-MAGAP lands, but was reduced by more than half by 2008. Indigenous lands occupy the greatest proportion of the study region (64% of total area) and, overall, exhibited the highest volume of forest loss by total area for the first time period.

Table 2. *Deforestation calculations for 1990–2000 and 2000–08, for the study region and eight categories of land tenure*

	Total area (km^2)	Forest base (km^2)	De-forested (km^2)	% De-forested	Defor./year	% Deforested to agriculture	Fractional loss of defor.
1990–2000							
Study region	39762.7	33606.8	4079.6	12.1	408.0	78.4	−1.3
PA (no overlap)	5717.0	5381.5	49.4	0.9	4.9	50.2	−0.1
PA-INDIG (overlap)	8375.6	7477.5	134.1	1.8	13.4	20.2	−0.2
PF (no overlap)	1588.8	1472.8	88.4	6.0	8.8	69.1	−0.6
PF-INDIG (overlap)	4261.4	3564.9	136.6	3.8	13.7	47.1	−0.4
BP (no overlap)	272.8	164.8	31.4	19.0	3.1	62.6	−2.1
BP-INDIG (overlap)	804.9	738.5	89.1	12.1	8.9	63.4	−1.3
INDIG (no overlap)	10215.8	8549.8	1858.4	21.7	185.8	81.1	−2.4
Private-MAGAP	8564.5	6278.4	1693.7	27.0	169.4	85.0	−3.1
2000–08							
Study region	39762.7	29966.5	1135.0	3.8	141.9	55.6	−0.5
PA (no overlap)	5995.2	5088.6	25.0	0.5	3.1	54.1	−0.1
PA-INDIG (overlap)	9020.8	7883.0	120.1	1.5	15.0	9.4	−0.2
PF (no overlap)	1588.8	1392.2	38.5	2.8	4.8	49.2	−0.4
PF-INDIG (overlap)	4261.4	3476.3	52.4	1.5	6.5	46.7	−0.2
BP (no overlap)	272.8	140.1	9.9	7.1	1.2	22.2	−0.9
BP-INDIG (overlap)	858.8	695.0	18.2	2.6	2.3	51.5	−0.3
INDIG (no overlap)	9519.7	6307.4	411.5	6.5	51.4	65.2	−0.8
Private-MAGAP	8286.2	4616.4	460.2	10.0	57.5	61.5	−1.3

In the case of overlapping categories of tenure, where indigenous community areas overlapped with protected forests (BP) or forest patrimony areas (PF), the rate of deforestation was *less* than that which occurred in any of those tenure categories separately. This was true across both time periods. While there was no change in the rate of deforestation for indigenous lands overlapping protected areas, there is a noticeable reduction in the amount of deforested land being converted to agriculture in protected areas that overlap indigenous areas by 2000–08 (T2). However, these descriptive statistics of forest cover change do not take into account the relative pressure acting on any particular tract of forest. Protected areas located in remote areas tend to avoid deforestation by nature of their remoteness, not necessarily their protected status (Joppa & Pfaff, 2011). Thus we controlled for remoteness and other factors that affect forest clearing, to see if these broad associations still hold.

(b) *Econometric results*

Table 3 presents several specifications of the random coefficient model described above. Models I–III present effects for all the tenure categories in our dataset. Models IV and V collapse tenure into four categories that reflect our initial hypothesis that tenure's effect on deforestation rates should be ordered such that: (1) protected areas avoid the most deforestation, followed by (2) restricted use forests (BP and PF), then (3) overlapping tenure, and (4) indigenous areas avoiding the least. We assume unmitigated clearing occurs on mixed private/MAGAP land, therefore private/MAGAP land serves as the reference group for our analysis, and all tenure effects are interpreted as relative to this group. The first model estimates random intercepts at the canton level for reference. Model II allows for random intercepts at the canton level and random slopes on tenure categories, so that the relationship between tenure and deforestation can vary within each canton. Model III has the same structure as Model II, but estimates separate effects for each time period. Models IV and V are analogous to Models II and III but use aggregate tenure categories in the estimation.

We see that Model I shows inflated significance relative to the other models presented, a result we expect if the variation of the tenure–deforestation relationship within each canton is important as noted above. Model II shows that over the 18-year period protected areas and all three categories of protected land that overlaps with indigenous communities saw significantly less deforestation than private-MAGAP lands after controlling for the location and proximity characteristics described above (for covariate parameter estimates see SI Table 2). For example, the β coefficient (taking into account the log-transformation) for protected areas in Model II represents a 145% decrease in the annual deforestation rate relative to private-MAGAP lands. All other continuous independent variables were also log-transformed so, for instance, the model predicts that a 1% change in the distance to a town was associated with a 46% decrease in annual deforestation rate (SI Table 2).

The statistical significance of the tenure variables in the model only implies that these tenure categories had a distinguishably different average effect relative to private-MAGAP land. Comparing other tenure categories within Model II, we reject Wald tests that the tenure coefficients are not distinguishable from the estimates for indigenous lands for categories PA, BP, PA + Indigenous and PF + Indigenous, that is, most of the protected categories (all $\chi^2 \geqslant 9.52$ and $p \leqslant 0.01$). Thus the results from Model II broadly imply a significant difference in deforestation rates between protected and unprotected areas, irrespective of whether communities live within the protected areas. The results from Model III tell a similar story, with the notable exception that restricted use protected forests (BP) and forest patrimony (PF) areas were associated with avoided deforestation in the latter time period, but not in the first one, demonstrating a dynamic effect over time.

Models IV and V collapse all instances of tenure in which indigenous claims overlap with some level of protection into one category (*overlapping*), and put forest patrimony (PF) and protected forests (BP) into one category (*restricted use*). Model IV estimates average relationships over the whole time period and Model V estimates each period separately. Not sur-

Table 3. *Forest change by tenure category, random coefficient model results*

Dependent variable: ln(annualized % deforested)	I: Random intercept	II: Random Slope (RS)	III: RS × period	IV: RS, aggregate tenure	V: RS, Agg. tenure × period	Time period
				Protected areas		
Protected area (PA)	-0.58^{***} (0.06)	-1.47^{***} (0.36)	-1.45^{***} (0.32)	-1.10^{***} (0.32)	-1.81^{***} (0.38)	1
			-1.15^{***} (0.32)		-1.51^{***} (0.38)	2
Protected forest (BP)	−0.54 (0.33)	-1.19^{*} (0.55)	−0.43 (0.59)			1
				Restricted use (BP + PF)		
			-1.81^{**} (0.59)	-0.51^{*} (0.23)	0.33 (0.18)	2
Forest patrimony area (PF)	-0.64^{***} (0.08)	-0.62^{*} (0.29)	−0.35 (0.22)		-0.62^{***} (0.18)	1
			-0.58^{**} (0.22)			2
PA + Indigenous overlap	-0.91^{***} (0.05)	-1.00^{***} (0.19)	-0.70^{***} (0.17)			1
			-1.19^{***} (0.17)			2
				All overlapping tenure		
BP + Indigenous overlap	-0.82^{***} (0.12)	−0.60 (0.39)	0.07 (0.33)	-0.85^{***} (0.23)	-0.51^{*} (0.24)	1
			-1.40^{***} (0.33)		-1.13^{***} (0.24)	2
PF + Indigenous overlap	-0.92^{***} (0.06)	-1.20^{***} (0.21)	-0.89^{***} (0.23)			1
			-1.68^{***} (0.23)			2
				All indigenous lands		
Indigenous only	-0.15^{***} (0.04)	−0.07 (0.22)	0.18 (0.19)	−0.06 (0.22)	0.17 (0.20)	1
			-0.39^{*} (0.19)		-0.41^{*} (0.20)	2
ln(dist to oilfield)	-0.23^{***} (0.02)	-0.25^{***} (0.02)	−0.03 (0.02)	-0.25^{***} (0.02)	−0.05 (0.02)	1
			-0.37^{***} (0.02)		-0.37^{***} (0.02)	2
ln(dist to mine)	-0.26^{***} (0.03)	-0.21^{***} (0.03)	0.11^{**} (0.04)	-0.22^{***} (0.03)	0.10^{**} (0.04)	1
			-0.48^{***} (0.04)		-0.49^{***} (0.04)	2
ln(%defor.)|time period 1	0.85^{***} (0.00)	0.84^{***} (0.00)	1.00^{***} (0.00)	0.84^{***} (0.00)	1.00^{***} (0.00)	
Time period 2 (dummy)	-0.92^{***} (0.03)	-0.90^{***} (0.03)	-0.26^{***} (0.05)	-0.90^{***} (0.03)	-0.26^{***} (0.05)	
Constant	0.56^{***} (0.12)	0.52^{**} (0.16)	−0.18 (0.20)	0.52^{***} (0.16)	−0.19 (0.21)	
N	34,092	34,092	34,092	34,092	34,092	
# Random intercepts (cantons)	11	11	11	11	11	
# Random slopes (tenure)		7	7	4	4	
Moran's I	0.03^{*}	0.04^{*}	0.03^{*}	0.03^{*}	0.03^{*}	
ln(likelihood)	−81371.7	−81201.5	−78197.5	−81217.6	−78237.5	

Other covariates included in all models: distance variables (km to: road, town > 1000, river), elevation, % fertile soil, % forest in 1990, and population density. See Supplementary Information Table 2 for the full set of results.
Model I includes random intercepts at the canton level. Models II–V include random slopes on tenure dummy variables with random intercepts at the canton level.
Due to computational limitations in calculating Moran's I, the estimates (and *p*-values) presented here are averages from 50 randomly chosen 400 km^2 patches from the study area.
For all specifications the reference category for tenure is private (MAGAP) land.
$^{*}p < 0.05$.
$^{**}p < 0.01$.
$^{***}p < 0.001$.

prisingly, this re-categorization of tenure essentially averages the effects of the more specific-form tenure from Models II and III. Wald tests in Model IV show that the coefficient on protected areas is significantly different from indigenous land ($\chi^2 = 6.39$; $p = 0.03$) as is overlapping tenure ($\chi^2 = 9.64$ and $p < 0.00$). The effects of tenure in Model V show a similar relationship to Model III with time—for all areas except for PAs, tenure had a stronger impact on avoiding deforestation in the second time period. The strictest protection (PA) and protection in areas with indigenous inhabitants avoided similar amounts of deforestation. Our models suggest these areas avoided roughly 100% more deforestation than private areas, on average. Restricted use areas, indigenous areas without additional restrictions, and private lands had rates of deforestation that are, for the most part, indistinguishable.

Both macro-level variables, the distance to mining and oil concessions, were associated with deforestation rates. The effects are relatively consistent across models specifications over the whole time period: a 1% increase in the distance to an oilfield or a mining concession was associated with more than 0.2% decrease in annual deforestation rate. When looking at the time periods separately, we found no relationship between oil activity and deforestation in the first period, but a greater protective effect in the second period. Mining concessions in the first period were associated with significantly more deforestation (a 1% increase in distance to a mine is associated with more than 0.1% increase in annual deforestation rates), but in the second period were associated with nearly 0.5% decrease in deforestation rates. Others have also shown macro-forces like oil may at least indirectly provide a protective effect over forests since such livelihood and revenue-generating opportunities help limit incentives to clear more forest for agriculture (Wunder & Sunderlin, 2004).

Our measure of *dynamic deforestation*, the rate in the first period as a predictor for the second, suggests that a 1% increase in deforestation in the first period was associated with nearly a 1% increase in deforestation in the second period. Other covariates showed expected signs and significance (SI Table 2), with distance to a town having the strongest negative correlation with deforestation rates. Moran's I statistics were all nearly zero, indicating little concern for spatial autocorrelation (Table 3).

(c) *Implications for incentive-based conservation program (Socio Bosque)*

Our analysis presents both opportunities and challenges for implementing incentive based forest conservation programs. Identifying tenure categories associated with rapid deforestation (e.g., indigenous lands that do not overlap with parks or protected forests) could help administrators through improved targeting for investment. Yet our analysis also revealed the variability of deforestation rates over time and that the relationship between deforestation and tenure itself also varies. Such variability makes the calculation of baseline and additionality more challenging for incentive-based forest conservation programs.

Ecuador's conservation incentive program, Socio Bosque, defines priority areas based on deforestation threat, ecosystem services, and poverty (Figure 4), which can help determine the order of enrollment of landowners in situations where demand outpaces the supply of available funds for incentives, a situation Ecuador experienced for the first time in 2012 (Celi, G., personal communication, May 30, 2012; MAE, 2008, 2011b).

As of March 2012, in our study region Socio Bosque had brokered 223 individual agreements and 20 community agreements, representing a total forest area of 2150 km^2 (94% of this falls within the community agreements). Three communal contracts were also located within the protected areas of Cuyabeño Reserve and Yasuní National Park, which account for nearly half of the total area enrolled under communal contracts in the study region. Lands within protected areas were not eligible for Socio Bosque until 2010, and MAE has set the overall national limit for the amount of area Socio Bosque may enroll within protected areas to 15% (Celi, G., personal communication, May 30, 2012).

Table 4 shows forest outcomes over the study period by Socio Bosque priority categories. Note this table shows *past* changes in forest cover and the land tenure categories within *current* Socio Bosque priority areas since all our data are prior to implementation of the program.

Areas now under individual agreements with Socio Bosque experienced 8.8% deforestation in 1990–2000, and 4% deforestation for 2000–08. The community agreement areas registered 1.8% forest loss during 1990–2000, and 1.3% in 2000–08. Thus deforestation had already slowed considerably, even before the active implementation of Socio Bosque (Figure 5), consistent with our model results for the study region. Interestingly, the lowest ranked priority areas (Priority 3) had the highest rates of deforestation in both time periods. However, Priority 3 areas have the lowest percent forest cover (50% compared to 68% and 64% in Priority 1 and 2), thus Socio Bosque prioritization may implicitly include the potential amount of forest to be saved and/or favor communal land with potentially lower transaction costs (SI, Section III).

4. DISCUSSION

Our study makes two main contributions to the literature on deforestation and land governance. First, using a unique and detailed dataset for two provinces in the northern Ecuadorian Amazon, we presented explicit rates of deforestation over a complex set of tenure categories, controlling for a broad suite of confounding variables including macro-economic drivers like mining and oil activity. Second, we presented changes in deforestation rates within each of these tenure categories over two time periods: 1990–2000 and 2000–08.

Similar to previous studies, we found a fast pace of deforestation for this region during the 1990s (Bilsborrow *et al.*, 2004; Mena, Bilsborrow, *et al.*, 2006; Mena, Barbieri, *et al.*, 2006) that slowed considerably after 2000. Local citizens and managers anecdotally attributed the slowdown in deforestation to the financial crisis of the late 1990s as well as the dramatic drop in coffee production (triggered by both the El Niño event of 1997/98 and a global coffee crisis), shocks that elsewhere in the tropics caused farm abandonment and increased rural-urban migration (Rudel, Perez-Luego, & Zichal, 2000). In the late 1990s coffee dominated as the main small-scale cash crop (responsible for ten percent of national production), with a potentially mitigating influence on deforestation (Marquette, 1998). Once coffee production was no longer economically viable due to the crisis, this continued slowing in the rate of deforestation could be attributed either to plot abandonment, or households simply shifting and diversifying strategies, instead of extensification and further forest conversion.

Another factor associated with slowed deforestation was the decrease in oil production, which cost jobs and spurred land abandonment (Delgado, Larco, García, & Al., 2002; Piaz, personal communication, May 28, 2012). Indeed our results showed strong and significant decreases in deforestation near oilfields from 2000 to 2008, even after controlling for tenure

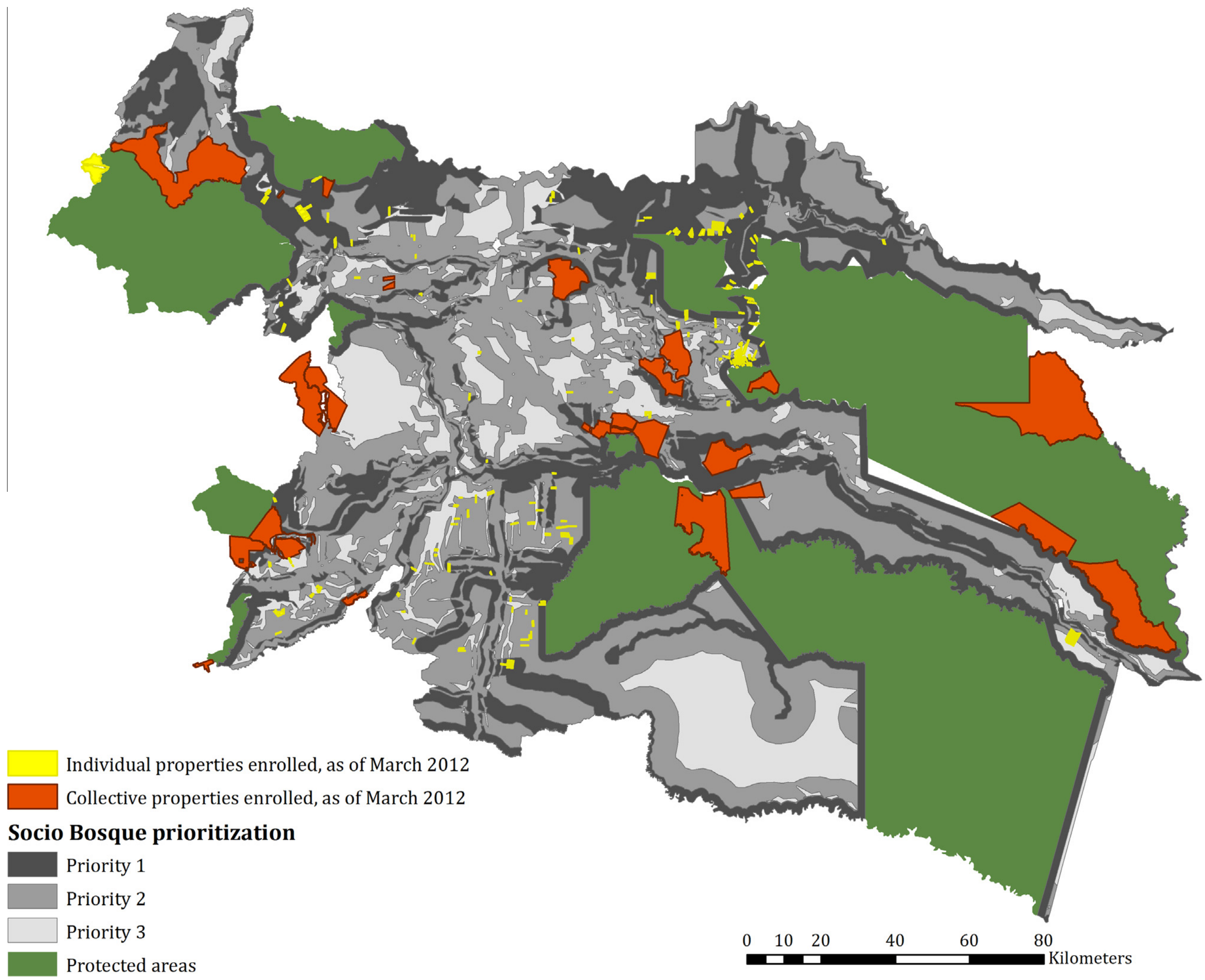

Figure 4. *Prioritization of Socio Bosque in study region, superimposed with current incentive agreements and protected areas. Source: MAE (2012).*

Table 4. *Forest outcomes and tenure categories across the Socio Bosque priority areas*

	Priority 1	Priority 2	Priority 3
Total area (km^2)	8651.7	11701.3	4223.2
Forest base, 1990 (km^2)	7237.7	9607.3	3208.6
Forest base, 2000 (km^2)	6269.0	7957.7	2274.8
% Deforested, 1990–2000	14.5	18.9	31.5
% Deforested, 2000–08	5.9	5.4	7.4
% SB priority area still in forest, 2008	68.1	64.2	49.6
% SB priority area currently under contract with SB (both individual and communal agreements)	5.3	3.6	1.6
% Area in tenure categories			
BP	2.3	0.5	0.4
BP-INDIG	3.4	3.8	2.8
PF	9	4.3	5.6
PF-INDIG	13.6	16.3	26.9
INDIG (no overlap)	38.3	41.9	29.8
Private-MAGAP	32.8	33.3	34.6

and other factors. Increased and organized indigenous community resistance to oil exploration could also help explain an additional factor that, along with the financial crisis, resulted in decreased oil production (Finer *et al.*, 2008). Overall, both the slowed deforestation due to coffee and diversifying on-farm strategies, as well as the decrease in forest loss close to oil production sites, echo similar patterns observed in other parts of the Ecuadorian Amazon, suggesting the possibility of

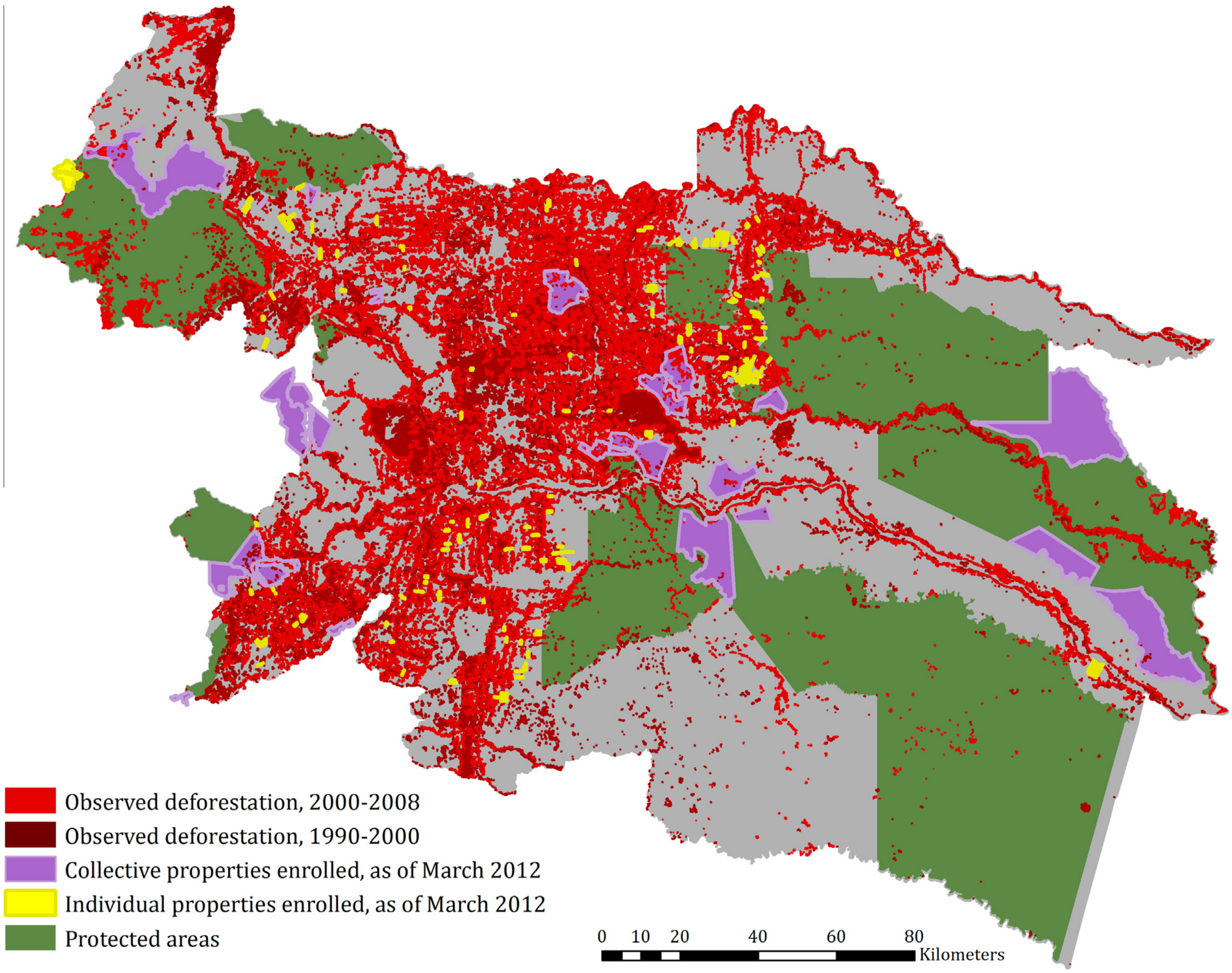

Figure 5. *Observed deforestation for each time period (T1 and T2), compared with protected areas and current contracted lands in the Socio Bosque program.*

a forest transition underway (Rudel, Bates, & Machinguiashi, 2002).

Our results offer new insight into the variable relationship between forest change, tenure overlaps, and tenure dynamics, and allow for an examination of these significant relationships, even after detangling them from macro-level influences (e.g., oil and mining). But the analysis is limited in its ability to distinguish between cases where deforestation is impacted by a location's form of tenure and a case where assignment of tenure category is a function of its ongoing forest (or other associated land use) changes. As such, here we ultimately provide a description of factors (including both macro-level influences and tenure) that are correlated with, but may or may not ultimately cause deforestation. Further, given the scale of our analysis, these results represent the average situation over the landscape. Investigation into plot-level dynamics is a fruitful next step that would offer a better understanding of the relative incentives and pressures faced by the ultimate land managers living within each of these tenure categories.

Overall, our results suggest we can reject our initial hypothesis that restricted use areas avoid more deforestation relative to overlapping "complex" forms of tenure. Instead we found equally low deforestation rates in "pure" protected areas and in areas where indigenous territory and protected area overlap. Outside protected areas, deforestation rates are statistically indistinguishable on indigenous lands and private-MAGAP lands (or communal vs. private tenure) (Table 3). Protected forests and forest patrimony areas without indigenous inhabitants show little significant difference from unprotected land. However, these two restricted use categories are associated with much less deforestation in the second time period, illustrating the importance of exploring tenure-land use relationships over various time scales.

By separating the categories of tenure overlap, we also observe a strong association between reduced deforestation and indigenous lands that overlap with either protected forests or forest patrimony areas. This appears most consistent for indigenous overlap with forest patrimony areas, particularly when compared to "pure" forest patrimony's inconsistent relationship over time. Essentially, rather than be an indicator of contested claim or insecure tenure, here areas of overlapping tenure point to a bolstered effect of forest protection.

Bremner and Lu (2006) also suggest overlapping designation with protected areas in this region may improve tenure security of indigenous territory. In fact, indigenous groups across the *Oriente* have actively formed federations and attempted to delineate their lands and include them within protected areas to defend against incursions from extractive industries. The Shuar people living in southern Ecuadorian Amazon have sought protected area designation and enforcement on top of ancestral lands in order to safeguard their territory against resource exploitation, especially mining and oil exploration (Rudel & Horowitz, 1993). Conversely and more recently, some indigenous communities have promoted extractive industries and plantation agriculture in territories adjacent to protected forests, such as clearing forest for oil palm by a Secoya community in Sucumbíos (Gómez, 2011)

This suggests that some indigenous groups might welcome the added layer of forest protection as a way of strengthening their *de jure* land claim and defend against external pressures. Bremner and Lu (2006) also conclude that the *de facto* rights exhibit a high degree of variability across indigenous groups in this region, particularly in the efficacy of internal enforcement of withdrawal rights for forest and other natural resources among community members. In this example, external community-level factors (extractive industries) affect local land use change and drive the desire for more restrictive land tenure classifications as opposed to land tenure affecting land use decisions.

The majority of contracts already negotiated and enrolled in Socio Bosque for this region are with communities, including indigenous groups holding communal title. These contracts involve lower per hectare incentive payments than the agreements Socio Bosque negotiates with individual landowners, while also securing more forest area for conservation with each contract (SI, Section III). Although this discrepancy in incentives highlights issues of equity in the approach (de Koning *et al.*, 2011), based on these low transaction and investment costs on the part of the government, we anticipate that community agreements will continue to dominate in this region, and potentially throughout the Ecuadorian Amazon. With this in mind, our results suggest that Socio Bosque may have higher potential for additionality by targeting indigenous lands and community organizations holding communal title outside of protected areas, which is particularly relevant for REDD.

5. CONCLUSION

From these overall results, we derive three key messages for practitioners, policymakers, and future research. The first is simply that the form of tenure matters for understanding the future pattern and extent of land use change in this region. Even when accounting for documented drivers of deforestation in this region (proximity to oil exploration, roads, and markets) and macro-level influences, tenure plays a role in slowing forest loss. These observed trends over 18 years, a time of initially rapid, and later slowed but continued deforestation, along with the significant influence of tenure form, can additionally help refine the spatial priorities for Socio Bosque, which currently consider primarily distance-related factors in determining the risk of deforestation. Taking the tenure landscape into account can improve Socio Bosque's ability to target areas under high deforestation threat, as well as pinpoint key opportunities for establishing connectivity within the landscape.

The second message is that we cannot assume all indigenous areas are homogeneous in the factors that influence forest change. Our results show that while indigenous lands on their own trend more closely toward the high deforestation in privately-held or MAGAP-administered lands, the existence of an overlap points to a protective effect for forests. This variation we observe between nonoverlapping indigenous areas and those which overlap with protected areas, protected forests, and forest patrimony areas also matches well with other observations in this area of a high degree of variability between indigenous communities with respect to agricultural activity and land use (Bremner & Lu, 2006; Gray, Bilsborrow, Bremner, & Lu, 2007; Lu *et al.*, 2010). Notably, ignoring overlapping tenure misses policy-relevant nuance that has implications for plot-level analysis of tenure impacts on land use change.

Our third and final message relates to a broader challenge facing Ecuador and other countries in similar situations as they continue to plan for the potential entry of REDD incentives. For Ecuador's Socio Bosque program, the results from the forest change analysis point to dramatically slowed deforestation rates in this region, which historically experienced among the highest rates nationally. This is a trend that has occurred even prior to the implementation of a single Socio Bosque incentive. To ensure additionality in REDD, our results suggest Socio Bosque could further refine its set of priority regions, as well as advocate for titling and the resolution of land conflict in specific tenure forms, such as forest patrimony areas and indigenous territory outside of existing forest management areas. In fact, such initiatives are currently underway in the forest patrimony region surrounding the headwaters region of the Cuyabeño Reserve, indicating a clear opportunity to evaluate the impact of a shift in tenure security related to forest change. As Socio Bosque moves forward, it will be important to closely monitor the impact of the incentives on forest outcomes over time, as well as on household and community livelihoods, in order to track the effectiveness of the mechanism, particularly if any of the enrolled areas become eligible for future accounting related to REDD.

In effect, even in the absence of REDD, Socio Bosque offers a complementary approach to incentivizing continued forest conservation outside of protected areas in the northern Ecuadorian Amazon, particularly as we might expect outside pressures on forest resources, fossil fuel extraction, and even emerging pressures from intensive agriculture such as palm oil and in-migration from Colombia to continue. If Socio Bosque agreements can effectively act as an additional "layer" of protective tenure, our analysis suggests this may, in and of itself, hold power to help communally-held or privately-owned forests retain forest cover.

More broadly, this analysis signals that land tenure helps shape forest outcomes. But the direction of impact functions in concert with macroeconomic conditions that frame land use decisions, as well as opportunity costs and pressures acting on individuals and communities. This complexity should not keep us from purposefully incorporating such factors into an analysis of land use change. Indeed, for forest carbon strategies and conservation incentive programs to effectively target limited resources, while also having a positive impact on mitigating climate change and improving local livelihoods, we must continue to disentangle the myriad drivers that affect land use change, of which land tenure is a crucial factor.

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APPENDIX A. SUPPLEMENTARY DATA

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.worlddev.2013.01.012.