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LETTER

Evaluating the effects of common-pool resource institutions and market forces on species richness and forest cover in Ecuadorian indigenous Kichwa communities

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Abstract

We compare conservation outcomes between a protected area (PA) and four indigenous common-property regimes (CPRs) under differing degrees of market integration in the Ecuadorian Amazon. We first assess how market forces and common-pool resource institutions governing processes of forest conversion affect biodiversity and forest cover, and whether institutions mitigate the effect of market forces. We then analyze how biodiversity and forest cover differ between a PA, and communities with different market access. Finally, we link biodiversity and forest cover changes within communities to differences in land-use practices. While we show similar levels of forest cover and biodiversity between the PA and large CPRs with little access to local markets, institutions appear not to attenuate market effects on conservation outcomes in our case studies. We discuss results within a common-property theory context and highlight the importance of disentangling how market integration, common-pool resource institutions, and resource health interact over time.

Introduction

Communities managing forest commons (CMFCs) can reduce forest degradation without affecting livelihoods by designing and enforcing rules (institutions) to regulate forest-based common-pool resources (Persha *et al.* 2011). Few studies, however, have compared the conservation value of CMFCs relative to protected areas (PAs) (Berkes 2009) and no comparisons focus on the relationship between market-oriented livelihoods and biodiversity (Oldekop *et al.* 2010).

Land reforms, agricultural credits, and infrastructure (e.g., roads) encourage the adoption of market-oriented agricultural practices (Geist & Lambin 2002). Market forces provide incentives for short-term exploitation and long-term management by influencing perceptions of gains and losses associated with resource overexploitation. While communities contribute to forest degradation by integrating market-oriented practices into traditional land uses (Killen *et al.* 2008), they have also been shown to implement institutions in response to market forces, threats to (or changes in) common-pool resources and related socioeconomic inequalities among members (Agrawal & Yadama 1997; Ostrom 2009; Andersson & Agrawal 2011). Although institutions can moderate the effect of market forces (Agrawal & Yadama 1997) with no significant economic tradeoffs (Persha *et al.* 2011), few studies have evaluated relationships between local institutions and commercial livelihoods in CMFCs and how they relate to forest degradation and biodiversity in Latin America (Chhatre & Agrawal 2008; Oldekop *et al.* 2010; Persha *et al.* 2011).

Forest cover change in Latin America has been influenced by agrarian reforms and colonization programs since the 1960s and 70s (Rudel *et al.* 2009). Faced with competition from settlers, many indigenous communities in Amazonia secured land titles by adopting state recognized forms of organization (Perreault 2003) and market-oriented agriculture (Behrens *et al.* 1994). Many indigenous groups continue managing common-pool resources (Bremner & Lu 2006) but have also contributed to regional deforestation (Killeen *et al.* 2008) by adopting market-oriented agriculture (Gray et al 2008**)**. Natural resource management decisions under many common-property regimes (CPR) are, therefore, likely influenced by past and current incentives for commercial agricultural production. Disentangling how institutions and economic incentives interact to influence CMFC outcomes, will help identify tradeoffs affecting the relationships between livelihoods, forest cover change, and biodiversity, and elucidate the role that local communities can play in conservation efforts (Berkes 2009).

We compare species richness of two commonly used biodiversity indicators (ferns and leaf litter frogs), measures of forest cover (gap fraction, normalized difference vegetation index [NDVI] and near infrared reflectance [NIR]) inside a PA and in forests in four indigenous Kichwa communities with CPRs in the Ecuadorian Amazon. We use these biodiversity indicators because they have been shown to decline with environmental degradation in the neotropics and have stricter habitat requirements than other commonly used indicators (Barlow *et al.* 2007; Pardini *et al.* 2009). Similarly, gap fraction, NDVI, and NIR are negatively related to forest age and biomass (McMorrow 2001; Vieira *et al.* 2003; Asner *et al.* 2004).

First, we determine how market forces and institutions governing agriculturally led processes of forest conversion affect species richness and forest cover, and whether institutions influence the effect of market forces. Next, we analyze how species richness and forest cover differ between a PA, and forested areas managed by communities with different market access. Finally, we link changes in species richness within communities to land-use practices. Examining these relationships helps us understand how livelihood decisions and common-pool resource institutions interact to affect environmental outcomes.

Methods

Study area

We conducted our study in the Sumaco Biosphere Reserve (SBR), which contains the Sumaco Napo-Galeras national park (SNGNP), an IUCN category II PA. The SBR is located within the Ecuadorian Amazon and within the Tropical Andes biodiversity hotspot (Myers *et al.* 2000). Approximately 70% of the SBR's inhabitants are indigenous Kichwa (Valarezo *et al.* 2001) and many Kichwa communities secured collective land titles and credits by forming state-recognized agricultural cooperatives after the discovery of oil and agrarian reforms of 1964 and 1973 (Perreault 2003). While families often hold usufruct rights to land parcels used for agriculture, many communities implement institutions to regulate the use and distribution of land and natural resources.

Sampling

We examined socioecological relationships using a series of statistical models linking quantitative environmental data with household surveys assessing agricultural livelihoods and rankings of common-pool resource institutions derived from qualitative interviews.

We conducted household surveys in four Kichwa communities with similar populations (290–300 inhabitants divided into 40–80 households) living within the SBR's buffer zone and sampled species richness in those four communities and the SNGNP (see Figure S1 for details). All sites are tropical moist forest (Navarrete 2001) and lie between 200 and 400 m AMSL (sites within the SNGNP are the only ones accessible between 200 and 400 m).

We visited the communities of San José de Payamino (Payamino) and Chontacocha during August to December 2008, and the SNGNP and the communities of Verde Sumaco and Cascabel 2 (Cascabel) during August to December 2009. Additional interviews were held in each community in August 2010. Verde Sumaco and Payamino own over 16,000 ha each and are situated approximately 20 km from the nearest market town (Loreto). Prior to 2007, the communities were inaccessible by road. Chontacocha and Cascabel own approximately 2,000 ha each and have been connected to Loreto by a 10 km road with regular public transport for over 10 years. All communities sell agricultural

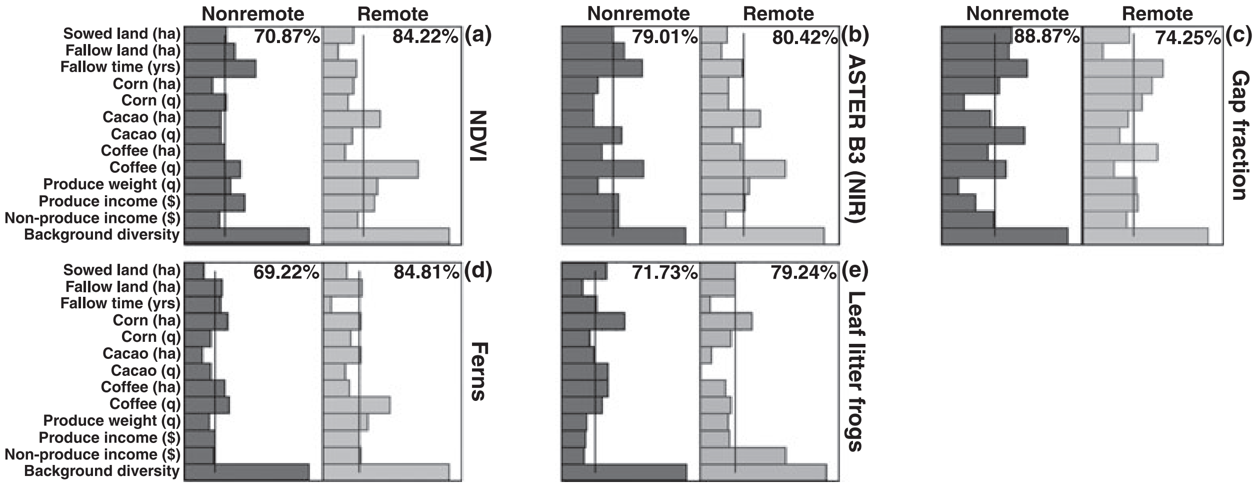

Figure 1 Partial least squares regression results ($n = 32$) for NDVI, NIR, gap fraction, and biodiversity indicators. Percentage values represent the total variation explained by the regression model. Horizontal bars represent individual variable contributions. Background diversity is the contribution of forest cover, infrared reflection, gap fraction, or species richness from paired uninhabited (UH) sample points; e.g., NDVI at uninhabited sites in (a), NIR in (b), gap fraction in (c), ferns in (d), and leaf litter frogs in (e). Vertical black bars represent significance levels (variable importance to projection).

produce in Loreto but market intermediaries also regularly visit Chontacocha and Cascabel. We therefore classified Verde Sumaco and Payamino as remote and Cascabel and Chontacocha as nonremote. Remote communities in the SBR have substantially larger territories than nonremote communities making it difficult to control for both territory size and market distance. We selected communities based on (1) similar population sizes, (2) distance to markets: prior to 2007 Verde Sumaco and Payamino were only accessible via a 3–6 hour canoe ride, while Chontacocha and Cascabel are 30 minutes by road, and (3) similar territory sizes in remote and nonremote communities, respectively.

Household surveys

We surveyed land-use practices in 32 households using questionnaires (nine households each in Payamino and Chontacocha, and seven households each in Verde Sumaco and Cascabel, ∼15–20% of households in each community). To ensure community participation, we selected households with approval of community leaders and accounted for small sample sizes by selecting households involved in subsistence and commercial agriculture, and spread (rather than clustered) over the inhabited areas in each community. Variables included the amount of land currently dedicated to agriculture, the amount of fallow land and fallow time, as well as the land dedicated to, and the most recent yields of, the most important regional cash crops (corn, coffee, and cacao) (Figure 1). Total yields, agricultural and nonagricultural income generated during the last year (e.g., temporary employment) were also included in the survey.

Institutional analysis

We adapted Ostrom's (2009) analytical framework for socioecological systems and identified the main management practices regulating Kichwa CPRs in the SBR. Sampled communities adopted the same state-recognized organizational system and secured collective land titles during the 1980s. Management decisions, rules, and sanctions are approved by general consensus during public meetings whose proceedings serve as official records of resolutions.

We built categorical indices of community common-pool resource institutions by assessing the presence or absence of rules and sanctions regulating the agricultural use of land and timber, which is commercially sold, and whether communities had established forest reserves. Forest reserves, land-use change, and timber extraction can significantly affect our chosen forest cover measures and biodiversity indicators (Barlow *et al.* 2007; Pardini *et al.* 2009).

We assessed the presence or absence of rules (de-jure) and whether these rules were upheld in practice (de-facto) through monitoring and sanctions, by conducting semistructured interviews with subsets of sampled households and additional key informants who had held community management committee roles (3–6 interviews in each community). All interviews were conducted in Spanish. We corroborated responses between interviewees and cross-referenced answers with community meeting records. While these methods cannot yield enough data to trace the internal processes leading to the implementation of institutions, they are similar to qualitative techniques used in comparative socioecological studies (e.g., Persha & Blomely 2009) and provide sufficient information for rankings of institutional strength based on both presence and enforcement of rules and sanctions. We ranked institutional arrangements by scoring the presence (or absence) of individual de-jure and de-facto rules and generating a total institutional score for each community.

Biodiversity sampling

We sampled fern and leaf litter frog species richness along 49 500 m long transects: five in the SNGNP, three per community in uninhabited forested areas deemed to be in "pristine" condition by community leaders (12), and on the individually managed lands of the 32 interviewed households (inhabited). In remote communities, uninhabited areas were several hours by canoe and foot from the communities' center. In nonremote communities, uninhabited areas were either a 20 ha forest reserve within short walking distance from the communities' center with no agricultural plots (Chontacocha) or an area within 40 ha of communal forest that had not yet been allocated to community members and showed no signs of recent agricultural use (Cascabel).

We used households as points of origin for each transect and avoided roads and agricultural plots. We visited transects on two consecutive days and sampled ground dwelling and epiphytic ferns (below 2 m) during the first sampling day within 5 m × 5 m quadrats set out every 50 m along each transect. We deposited reference collections of identified species (Navarrete 2001) at the Ecuadorian Museum of Natural History (MECN).

We sampled leaf litter frogs (Family: Strabomantidae) on the second sampling day by walking transects at a rate of 50 m per person per hour for a total of 10 person-hours per transect, and searching within 1 m distance from either side of the transect midline. We sampled during both day and night, and deposited reference photographs of identified species (Rodriguez & Duellman 1994; Valencia *et al.* 2008) at the MECN.

Forest cover

We calculated NDVI values in ArcGIS 9.3 (ESRI) for a 2007, orthorectified and radiometrically corrected, 15 m resolution advanced spaceborne thermal emission radiometer (ASTER) image (RMSE = 9.52 pixels, where RMSE is root mean square error), spanning the areas between 200 and 400 m within the SBR. We calculated average NDVI and NIR values across quadrats within each transect.

We estimated gap fraction by taking digital photographs of the canopy with a 16 mm fish-eye lens in each quadrat along each transects, and analyzing images in Can-Eye v6.2 (INRA, 2007).

Statistical analysis

We calculated the effect of institutions and remoteness on forest cover measures and species richness using simple and multistep linear models. We used institutions and remoteness as predictor variables in the simple models. In our multistage models, we used remoteness as predictor for institutions in the first-stage model, and used predicted values of the institutions outcome variable to predict forest cover measures and species richness in second-stage models.

We also used linear models to calculate differences in agricultural practices between remote and nonremote communities and differences in forest cover measures and species richness among sampled sites. We used generalized linear models with gamma distributions when data were not normally distributed. We used contrast tests to compare forest cover measures and species richness between sites and corrected for multiple tests using Sequential Goodness of Fit tests (de Uña-Alvarez & Carvajal-Rodriguez 2010). We conducted all linear and generalized linear models in *R* (R Development Core Team 2011).

We assessed relationships between agricultural variables, and forest cover measures and species richness using partial least squares regressions in JMP 10 (SAS Institute Inc.). We included forest cover measures and species richness values in uninhabited sites to control for background levels of forest cover and diversity, and set significance to 0.8 (Wold 1994).

Results

Socioecological analysis

Nonremote communities dedicated more land to agriculture ($t = 2.31$, $P = 0.028$) and earned more income from agricultural produce sales ($t = 2.20$, $P = 0.036$). These differences are due to greater corn production (hectares: $t = 2.29$, $P = 0.029$; yield: $t = 2.28$, $P = 0.031$) and cacao-dedicated land ($F = 8.80$, $P = 0.006$). Although fallow land did not differ between communities, nonremote communities reported shorter fallow times

($t = -2.41$, $P = 0.023$) suggesting agricultural intensification. While nonagricultural income, coffee-dedicated land, and cacao yields did not differ between remote and nonremote communities, remote communities reported higher coffee yields ($t = 3.57$, $P = 0.003$) suggesting that they also respond to markets through cash crop sales and labor.

Agricultural variables and background diversity (i.e., contribution of forest cover measures or species richness from uninhabited areas) explained 69–85% of variation in forest cover measures and species richness (Figure 1). Many agricultural variables were significantly associated with forest cover measures and species richness, suggesting a clear link between land-use, forest cover, and species richness in both remote and nonremote communities.

Interview responses and community proceeding records' examinations suggest institutional arrangements of different strengths among communities (Table 1). Verde Sumaco and Chontacocha established strictly managed reserves and implemented de-jure and de-facto institutions regulating timber extraction. Cascabel also established a reserve but the area was weakly managed and several community members owned agricultural plots within the reserve. Only Verde Sumaco regulated forest conversion to agricultural land, although Chontacocha was considering similar restrictions.

Differences in species richness and forest cover

NDVI ($F = 8.38$, $P < 0.0001$), NIR ($F = 11.75$ $P < 0.0001$), gap fraction ($F = 6.08$, $P = 0.0006$), fern ($F = 9.5$, $P < 0.0001$), and leaf litter frog ($F = 5.25$, $P < 0.0015$) species richness were highest in the SNGNP and decreased across uninhabited and inhabited areas in remote and nonremote communities (Figure 2).

Remoteness was a better predictor of forest cover measures and species richness than institutional arrangements and remained significant in all cases after controlling for the effect of institutions (Table 2).

Discussion

Results suggest that economic incentives for agricultural production in our sampled communities compromise the potential role of institutions in forest and biodiversity conservation. In our study, economic incentives are likely to either offset resource management effects of these institutions, or create conditions that undermine the emergence of such institutions. Only Verde Sumaco regulated how much land could be dedicated to agriculture and the absence of institutions governing agricultural forest conversion in other communities could be explained by the existence of tradeoffs between the establishment of institutions and livelihood decisions. These tradeoffs could be driven by short-term individual cost-benefit analyses, rather than long-term communal evaluations.

Urbanization and market demand are affecting agricultural pressures on tropical forests (DeFries *et al.* 2010). While small-scale farmers supplying urban markets continue to be significant drivers of tropical land-cover change (Rudel *et al.* 2009**)**, remote areas often remain "protected" by distance and inaccessibility (Joppa *et al.* 2008). Our results show similar levels of forest cover and species richness between the SNGNP and remote communities. Remote communities in our study, however, had 8–10 times more land than nonremote communities, confounding potential effects of territory size (and population density) with remoteness. Despite these disparities, even in the most densely populated community (Cascabel), individual families had access to more than 20 ha of land each, suggesting that potential effects of territory size and population density may be small. Nonetheless, without replication and larger sample sizes with more variation in territory size and market access it is difficult to differentiate conclusively between the effects of territory size and remoteness. Critically, remote communities also integrated readily into markets. Such integration is likely to increase, as planned infrastructure and roads connecting remote areas to urban centers in Ecuador and elsewhere in the Amazon basin provide access to regions hitherto considered "protected" (Laurance *et al.* 2001; Swing 2011).

While market access (and related agricultural production) was a better predictor of forest cover measures and species richness than institutions, institutional arrangements are dynamic entities influenced by a suite of internal and external factors (Agrawal 2001; Ostrom 2009). Market forces and other social processes such as population growth are linked to agricultural production and natural resource exploitation (Geist & Lambin 2002) but there is evidence that communities can implement and adapt institutions to mediate the effect of external and internal factors threatening common-pool resources (Agrawal & Yadama 1997; Ostrom 2009; Andersson & Agrawal 2011). The evolution of such institutions, however, relies on social processes linking natural resource degradation to overexploitation and underlying driving factors (e.g., market production) (Lu 2005; Ostrom 2009).

Our study, like many others linking institutions to environmental outcomes (e.g., Agrawal & Yadama 1997), is limited by a lack of temporal data. While likely that indigenous market-oriented resource management practices continue to be influenced by past agrarian reforms and current market demands, our data do not allow us

Table 1 Common-pool resource institutions in the sampled communities

	Verde Sumaco	Chontacocha	Payamino	Cascabel 2
Distance to the nearest market town of Loreto	Six-hour canoe ride from nearest trafficable road <Remote>	Ten kilometer road with regular public transport <Nonremote>	A 22 km road fording several streams was constructed in 2007; rains regularly cut off the community and residents hire vehicles to access Loreto; prior to 2007, access was only possible via a 3-hour canoe ride <Remote>	Ten kilometer road with regular public transport <Nonremote>
Size (ha)	24,734	2,000	16,800	1,900
Settlement history	Residents settled the area during the 1970s after Ecuador's agricultural reforms	Residents settled the area during the 1960s after Ecuador's agricultural reforms	Residents claim ancestral rights to the area	Residents relocated to the area after the eruption of the Reventador volcano in 1987
Institutional arrangements				
Forest reserve	Established a large reserve that encompasses 90% of the community's territory (18,546 ha); it was established as part of a resource management plan with help of the Ministry of Environment in 2007; the reserve cannot be used for agriculture and aims to conserve natural resources <Score: 2>	Established a 20 ha reserve during the 1980s; a small area (∼ 4 ha) has been used for communal agricultural projects; the reserve's principal aim is environmental protection and consists mainly of primary forest <Score: 2>	No established reserve <Score: 0>	Established a 20 ha reserve after relocating; reserve can be used for personal agricultural projects and several members currently hold plots within the reserve; the reserve consists largely of secondary forest <Score: 1>
Forest conversion rules and sanctions	de-jure: the management plan stipulates areas that can or cannot be used for agriculture. The community has also set land use limits to 50 ha for each family de-facto: transgressions are monitored informally and would be verbally reprimanded before being fined; no severe transgressions have been reported <Score: 1>	de-jure: families have been assigned individual plots and may convert as much forest to agricultural land as they want. de-facto: residents are in the process of implementing rules limiting the amount land that can be farmed at any one time <Score: 0>	de-jure: no clear rules or regulations and residents may convert as much forest to agriculture as they want <Score: 0>	de jure: families have been assigned individual plots and may convert as much forest to agricultural land as they want <Score: 0>
Timber extraction rules and sanctions	de-jure: residents must seek prior community consent de-facto: transgression are monitored and fined; members have only extracted timber with prior permission <Score: 2>	de-jure: residents must seek prior community consent de-facto: transgressions are monitored and fined. Transgressors have been fined in the past <Score: 2>	de-jure: no clear rules or sanction; on the only occasion that a sale occurred, the community sought a pay settlement between the community and the buyer <Score: 0>	de-jure: no clear rules or sanctions; residents regularly extract and sell timber <Score: 0>

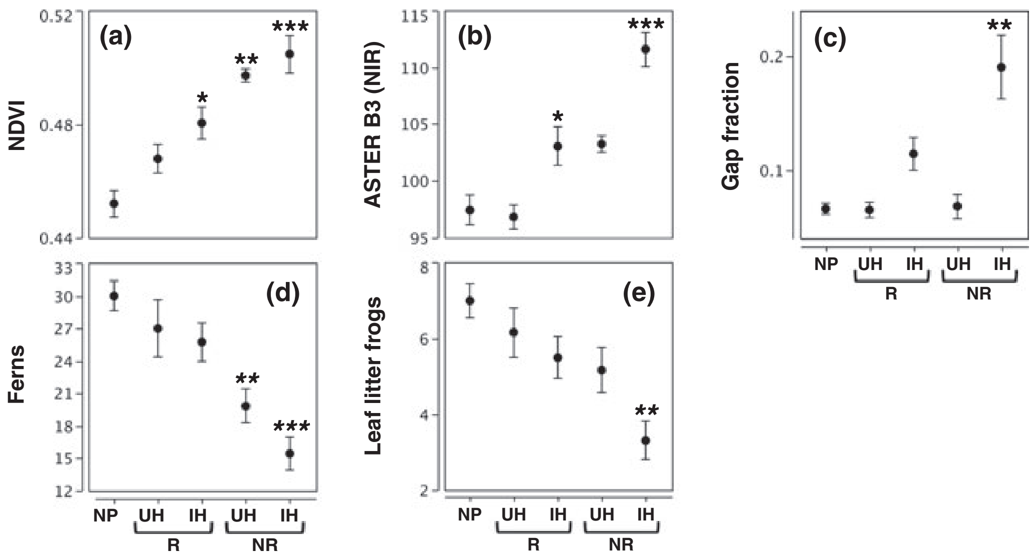

Figure 2 Differences in (a) NDVI ($F = 8.28$, $P < 0.0001$), (b) NIR ($F = 11.75$, $P < 0.0001$), (c) canopy gap fraction ($F = 6.08$, $P = 0.0006$), (d) ferns ($F = 9.5$, $P < 0.0001$), and (e) leaf litter frogs ($F = 5.25$, $P < 0.0015$) in the national park (NP) and in uninhabited (UH) and inhabited (IH) areas in remote (R) and nonremote (NR) communities. Data ($n = 49$) are presented as means ± 1 SE. Asterisks represent significant differences from the national park (***$P < 0.0005$, **$P < 0.005$, *$P < 0.05$).

to assess whether institutions in our sampled communities have emerged in response to market-related overexploitation (e.g., Chontacocha) or designed with foresight to prevent future risks to local livelihoods (e.g., Verde Sumaco). Furthermore, our small sample size does not allow us to look for potential interaction effects between size, remoteness, and institutions, which have been previously shown to influence conservation outcomes (Chhatre & Agrawal 2008).

There have been substantial theoretical (Ostrom *et al.* 2009) and empirical (Chhatre & Agrawal 2006; Persha & Blomley 2009; Persha *et al.* 2011) advances in the study of social-ecological systems, yet the interactions between CPRs and exogenous factors such as markets or economic incentives from government conservation policies are still poorly understood. We contribute to the community-based conservation debate by assessing the importance of market forces and CPR institutions in mediating CMFC ecological outcomes, and comparing those to a PA. While our results provide further evidence of the importance of markets in influencing CPR ecological outcomes, they also highlight that local institutions might not necessarily offset the effect of market forces influencing agricultural production.

While our study shows no overall impact of institutions in our sampled communities, we cannot provide information on how institutions influence overall rates of change. Communities implementing institutions in response to degradation, might for example, be able to

Table 2 Multiple regression results for institutional arrangements and remoteness as predictors of forest cover and species richness

Indicator	Model	*t*-value	*P*
NDVI	Remoteness	−3.03	0.0003
	Institutional score	−0.64	0.53
		F-value	
	Remoteness (institutional score)	15.33	0.0003
		t-value	*P*
ASTER B3 (NIR)	Remoteness	−4.06	0.0002
	Institutional score	−0.86	0.39
		F-value	
	Remoteness (institutional score)	16.13	0.0002
		t-value	*P*
Gap fraction	Remoteness	2.30	0.027
	Institutional score	−1.61	0.12
		t-value	
	Remoteness (institutional score)	4.50	0.031
		t-ratio	*P*
Ferns	Remoteness	5.30	<0.0001
	Institutional score	2.06	0.046
		F-value	
	Remoteness (institutional score)	24.51	<0.0001
		t-ratio	*P*
Leaf litter frogs	Remoteness	2.97	0.005
	Institutional score	−0.04	0.97
		F-value	
	Remoteness (institutional score)	9.11	0.0043

significantly dampen rates of environmental degradation or even reverse them (Nagendra *et al.* 2008). To date, there exist no large-scale, multitemporal socioecological studies focusing on the interaction between collective management decisions and environmental impacts. While we need to better understand the internal (e.g., livelihood decisions) and external forces (e.g., market forces) driving institutional change, future conservation policies will greatly benefit from understanding the temporal dynamics of socioecological systems; how institutions emerge, how they respond to socioeconomic, political, and environmental changes over time, and finally what their long-term ecological outcomes are.

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Supporting Information

Additional Supporting Information may be found in the online version of this article at the publisher's web site:

Detailed maps of the location of the communities and the sampled areas within the communities and the national park are available as part of the on-line article. The authors are responsible for the content and functionality of these materials. Queries should be directed to the corresponding author.

Figure S1: Map of the Sumaco Biosphere Reserve and sampled areas within the national park (grey dashed line), Verde Sumaco (A), Payamino (B), Cascabel 2 (C), and Chontacocha (D). Triangles represent sampled location within the national park, squares represent samples in uninhabited areas and circles represent inhabited areas. The solid black circle shows the location of the nearest market town, Loreto.

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