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Deforestation, Institutions, and Property Rights: Evidence from land titling to indigenous peoples and local communities in Ecuador

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Deforestation is a matter of pressing global concern, contributing to declining ecosystem services, biodiversity loss, and ultimately climate change through growing emissions. We evaluate the effect of assigning property rights to indigenous peoples and local communities (IPLCs) in coastal Ecuador on deforestation and the role polycentric institutions play in policy effectiveness. Informed by a theoretical model, we employ causal methods to 1) evaluate changes in forest coverage for the first 12 years of policy adoption, and 2) evaluate the effect of the presence of non-governmental organizations (NGOs) on policy permanence. We find that assigning property rights to IPLCs significantly decreases mangrove deforestation and that the presence of NGOs funded by foreign aid significantly increases the probability of policy adoption and permanence. We assess the positive development implications of the policy concerning local fisheries provisioning and the role of international aid in achieving environmental outcomes. Our work highlights the importance of IPLCs and civil society as actors for sustainable land stewardship in future climate policy.

KEYWORDS
Deforestation, property rights, polycentric institutions, foreign aid ,instrumental variable, regression discontinuity

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Deforestación, instituciones y derechos de propiedad: evidencia del otorgamiento de títulos de tierras a comunidades indígenas

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La deforestación es un asunto de preocupación mundial, contribuyendo a la disminución de servicios ecosistémicos, la pérdida de biodiversidad y, en última instancia, a el cambio climático a través de emisiones de carbono. Esta investigación evalúa el efecto de otorgar derechos de propiedad a comunidades indígenas y locales (IPLCs, por sus siglas en inglés) en la costa de Ecuador sobre la deforestación y el rol de instituciones de carácter policéntrico en la efectividad de estas políticas. Sobre la base de un modelo teórico, se emplean métodos causales para 1) evaluar los cambios en cobertura forestal para los primeros 12 años de adopción de la política; 2) evaluar el efecto de la presencia de Organizaciones No Gubernamentales (ONGs) en la permanencia de la política. Se encuentra que la asignación de derechos de propiedad a IPCLs disminuye significativamente la deforestación en manglares y que la presencia de ONGs financiadas por asistencia internacional incrementa significativamente la probabilidad de adopción y permanencia de la política. Se evaluan implicanciones positivas de la política en términos del aprovisionamiento en pesquerías y el rol de la asistencia financiera para obtener resultados positivos en términos ambientales. La investigación destaca la importancia de IPLCs y la sociedad civil como actores de la gestión sostenible de la tierra en vista a futuras políticas climáticas.

KEYWORDS

deforestación, derechos de propiedad, instituciones policéntricas, asistencia internacional, variables instrumentales, regresión discontinuada

1 | INTRODUCTION

The problem of the sustainable management of commons is still one of the biggest challenges facing economics. Concerns about water and air quality, pollution and hazardous waste, species extinction, maintenance of stratospheric ozone, and most recently the stability of the global climate have only increased in importance (Stavins, 2011). Amongst these, deforestation is a matter of pressing global concern, contributing to declining ecosystem services, biodiversity loss and growing carbon emissions. In particular the last century has seen an increase of deforestation in previously untouched ecosystems, with the prime example being the amazonian forest. Similarly, mangrove forests, which are inter-tidal forests occurring along tropical, subtropical, and some temperate coasts have been experiencing equal or greater rates of deforestation (Duke et al., 2007; Richards and Friess, 2016). Despite this fact, the issue of mangrove degradation and deforestation has received comparatively little attention (Friess et al., 2019).

Mangrove forests overlap with high and increasing densities of human populations. Therefore, mangroves provide key regulating, provisioning and cultural ecosystem services such as coastal protection, pollution control, food provision, and cultural values for hundreds of millions of people (Barbier et al., 2011). Mangroves' ability to provide relatively larger carbon sequestration when compared to other forests, as well as increased coastal resilience in the face of extreme weather patterns (Del Valle et al., 2020; Hochard et al., 2019), has placed them on the international climate change mitigation and adaptation agenda. Recent global commitments made during COP 26 highlight the critical importance of stopping deforestation. To achieve this, policymakers stressed the central role "[...] and value of knowledge and forest guardianship provided by Indigenous Peoples and local communities, calling for indigenous peoples to be empowered as such" (UNFCCC, 2021).
To curb deforestation numerous policies have been implemented with varying degrees of success. A key contribution of economics to this issue has been the development of market-based approaches to environmental protection of forests (Souza-Rodrigues, 2019). As many forest resources are held as common property or open access, problems pertaining their management have frequently also been addressed by common-property regimes of collective management (Ostrom, 2000). Evidence for the effectiveness of said approaches seems to be available at the local level, but questions remain pertaining their suitability as commons problems have spread beyond communities and even across nations (Stavins, 2011). In that light the promotion of property rights has been proposed as a way to ensure scarcity is well reflected in markets, and large scale land titling interventions have been championed as a policy to reduce deforestation and to achieve development goals, with potential benefits ranging from poverty reduction to food security (Liscow, 2013; Miller et al., 2021). Moreover, such interventions allow for the testing of the effectiveness and suitability of decentralized or polycentric forms of governance to govern and manage public goods (Ostrom, 2010). Specifically, transferring formal property rights to indigenous peoples and local communities addresses environmental justice and human rights issues concerning violence, expropriation and encroachment (BenYishay et al., 2017).

Property rights for indigenous and local communities also play a central, albeit, little recognized, role in the fight against climate change. Amazon indigenous territories alone cover nearly one-third of the region's land area across eight countries, and along with protected areas, protect over 52 percent of existing carbon stocks in the entirety of the Amazon forest (Walker et al., 2020). Theoretically, property rights could have ambiguous effects on deforestation depending on institutional and market settings (Busch and Ferretti-Gallon, 2017). Empirical evidence has found that land titling increased deforestation by small landholders in Brazil (Probst et al., 2020), and Nicaragua (Liscow, 2013), although both studies focus on private land holders. In contrast, recent evidence on communal/indigenous property rights[1] policies, finds mixed evidence on its effect on deforestation (Baragwanath and Bayi, 2020; BenYishay et al., 2017; Blackman et al., 2017; Buntaine et al., 2015). Rigorous analyses of titling campaigns are rare, with most studies not dealing with the non-random assignment of policy, therefore risking biased estimates of policy impact. Moreover, related theoretical and empirical research suggests that tenureship changes could either stem or spur forest damage impact (Miller et al., 2021; Busch and Ferretti-Gallon, 2017).

We identify several gaps in this broader literature, which mostly focuses on the effectiveness of interventions using panel methods, therefore not dealing with potential time-varying omitted variables that might bias estimates of interest (Blackman et al., 2017; Busch and Ferretti-Gallon, 2017; Miller et al., 2021). Furthermore, this body of work seldom presents empirical evidence of why or how these interventions work, nor do they reconcile observed effects with possible theoretical mechanisms behind said successes or failures (Deaton, 2010). We address these gaps in our study, and provide, to our knowledge, the first causal evaluation of a property rights based project targeting a previously understudied ecosystem, i.e. mangrove forests. Our aim is to contribute to the research on deforestation in the tropics and relevant climate and development policy, by first empirically examining the effects of formalizing land rights to "ancestral users" and its effectiveness on reducing mangrove deforestation in Ecuador. Secondly, we propose mechanisms that make said policy work, specifically showcasing the wide diversity of institutional arrangements in place, accounting for the role of international aid and NGO involvement in policy enrollment and outcomes. Specifically, we focus on the effect of local institutions by ancestral users, the presence of common-pool resources such as fisheries, and non-governmental organization involvement as the main mechanisms of policy success. To this end, we establish a simple stylized model to guide the empirical strategy and gain insights into conservation outcomes and policy adoption from communities. Our study is centred on the "Acuerdos de uso sustentable y custodia de manglar" (AUSCM) land titling policy, a pioneering land rights program across coastal Ecuador for mangrove conservation launched in the year 2000, and included in Ecuador's national climate policy structure.

Our research contributes to several strands of literature. First we add to the literature on policy evaluation and causal methods in tropical deforestation (Sims, 2010; Ferraro et al., 2012; Liscow, 2013; Souza-Rodrigues, 2019; Assuncao et al., 2022), specifically the effect of property rights granted to indigenous communities on mangrove deforestation, thereby expanding this literature to an important ecosystem, providing a rich data set and identification strategies. Second, this research aims to add to the body of work on common pool resources and institutions, in the framework of decentralized environmental and climate change policy (Ostrom, 2000; Dietz et al., 2003; Ostrom, 2010), specifically looking to contribute with an empirically rigorous assessment of the existence of a wide variety of institutions for the governance of a large scale commons problem. Finally, we contribute to the literature on the role of transactions costs and property rights in environmental policy making (Libecap and Lueck, 2011; Libecap, 2014; Ayres et al., 2018; Bühler, 2022), and add to the growing body of work studying the impact of the non-profit and non-governmental sector in influencing policy implementation, compliance, and policy relevant outcomes (Usmani et al., 2022; Grant and Grooms, 2017; Fitch-Fleischmann and Kresch, 2021).

The AUSCM policy was formulated in 1999 and implemented in 2000 in response to the rampant deforestation the country experienced during the 20th century which lead to losses of over 40% of all mangrove coverage in Ecuador. This historical process was

[1] Communal property rights are usually held in common ownership and may not be transferred or used as collateral (Probst et al., 2020).

characterized by systematic encroachment and episodes of violence, leading to not only the loss of the ecosystems, but also to the loss of the traditional means of subsistence of ancestral communities, with adverse effects on development and food security (Beitl, 2014a; Veuthey and Gerber, 2012). By 2020, 60 communities had property rights assigned, and 94 in total had been historically part of the policy. Communities voluntarily join the program, and over 30 percent of all remaining mangrove coverage in the country is covered by the policy.

Given that both policy and NGO presence are not assigned randomly, to achieve our stated goals presented above we develop an identification strategy that deals with the endogeneity of a) policy adoption by communities and b) NGO involvement. First, we evaluate the causal impact of the AUSCM policy on mangrove deforestation by employing an instrumental variable strategy, using the presence of aquatic organisms and relevant soil types as exogenous predictors of policy adoption. Second, by exploiting the variation in NGO involvement across time and communities, we investigate the causal impact of non-profit and non-governmental involvement on policy adoption and permanence. We employ a regression discontinuity design exploiting partisan voting behavior in the United States Congress as exogenous predictor of foreign aid disbursements and thus NGOs presence in policy uptake. We use this approach, since most of the NGOs who were working with ancestral communities in the periods we study were at least partly funded by the United States Agency for International Development (USAID).

Our results confirm that the adoption of communal property rights by ancestral communities reduces mangrove deforestation. This result is robust across different specifications, with the chosen instrumental variable being a strong predictor of policy adoption. This has positive implications for climate policy seeking to reduce emissions from deforestation, and north-south payments compensation mechanisms as part of the global climate mitigation strategies. We estimate that the policy prevented a total of 1.5 million tCO2 emissions between 2010 and 2012. Valued at the social cost of carbon[2], this corresponds to almost 60 million US$ of avoided damages.

Additionally, we find that devolution of property rights to ancestral communities provides more protection to mangrove forests against deforestation when compared to state-led protected areas, and that the presence of commercially important fisheries in mangrove forests is a strong predictor of property rights adoption by communities. These results therefore have positive implications for both development and food security benefits of the policy, notwithstanding the environmental justice component of devolution of rights to ancestral communities.

With regards to our second aim of assessing the effect of external actors' involvement on policy uptake, our results show that involvement of the non-profit and non-governmental (NGO) sector has a positive effect on the adoption and permanence of policy by ancestral communities. Our regression discontinuity design (RDD) relies on the partisan vote share margin in the US Congress as an exogenous predictor of foreign aid disbursements and hence the degree of NGO support. Our results suggest that NGO involvement affects policy adoption positively. We deduce that the mechanism for this is the reduction of transaction costs communities would otherwise have to bear to full fill the bureaucratic requirements of the policy. Likewise, our results highlight both the important role NGO involvement plays in environmental policymaking as well as that desired policy outcomes are at least partly dependent on the availability of international aid.

This paper is organized as follows: in Section 2 , we detail the study and policy context. Section 3, describes our theoretical model and the mechanisms affecting a community's

[2]We used the estimate of 112.86 US$=tC by Wang et al. (2019).

decision to adopt property rights. In section 4 we present the data used in this study. In Section 5, we describe our identification strategy. Section 6.1 includes findings from the grid-level analysis of the effect of property rights on deforestation, whilst section 6.2 includes our results on the presence of NGOs on policy adoption. Section 7 continues with a brief discussion of our results.

2 | POLICY DESCRIPTION AND STUDY CONTEXT

The AUSCM policy was formulated in 1999 and implemented in 2000 in response to the failure of command-and-control approaches and the deforestation the country experienced during the 20th century. It is estimated that approximately 30% to 40% of all mangrove coverage in Ecuador was lost since 1970 (Friess et al., 2019).

The degradation and deforestation of mangroves has been mainly driven by aquaculture and specifically, shrimp-farming. Onshore aquaculture was the leading cause of mangrove deforestation during the second half of the twentieth century, with its expansion entailing the conversion of standing mangrove forests into aquaculture farms (Friess et al., 2019). Historically, shrimp-farming started in Ecuador in the late 1960's, promoted both by the state and international development agencies. It boomed during the 20th century benefiting local economic elites, with its expansion encroaching on mangrove ecosystems and the traditional lands of indigenous peoples and local communities (IPLCs)[3] in coastal Ecuador. Ecuador experienced a loss of approximately 30% to 40% of all mangrove coverage, despite the deforestation ban and additional protection status of mangroves provided by protected areas created between 1979 and 1995 (Rodríguez, 2018; Beitl, 2011; Veuthey and Gerber, 2012). Apart from the overall biodiversity loss caused by deforestation, there were direct impacts on local communities as traditional users of mangroves' provisioning services, who depend on the respective marine resources as their main source of income. Specifically, two fisheries stand out as the main sources of income of mangrove dependent communities, the red crab and the fishery for mangrove cockles. Both are of artisanal nature, where local fishers collect crabs and cockles in mangrove forests. As communities were not themselves involved in shrimp farming and associated economic benefits, the destruction of mangrove ecosystems resulted in a loss of subsistence for these communities (Rodríguez, 2018; Veuthey and Gerber, 2012; Beitl, 2014b)[4]. Hence, this historical process was characterized by systematic encroachment and episodes of conflict, leading to not only the loss of ecosystems, but also the traditional means of subsistence, with implications on development and food security (Veuthey and Gerber, 2012; Beitl, 2014a).

In light of these impacts, the AUSCM institutionalized a "process of devolution of rights to communes, communities, peoples and ancestral nationalities, who may request an AUSCM for their subsistence, use and sustainable exploitation of mangrove based resources" (Bravo, 2013). The explicit aim of the project was to preserve mangrove forests and support the rights and well-being of ancestral communities. The policy is based on the rationale that communities have a self-interest in conserving mangrove ecosystems, which they had managed successfully historically. From the IPLCs perspective, mangrove preservation is a rational behavior, as a functioning mangrove ecosystem provides an ideal breeding and nursery habitat for marine species to prosper (Barbier, 2017). These marine species are highly valued in local markets, and thus increase the resulting benefits for communities

[3]Please note that in this paper we use the terms IPLCs and ancestral communities interchangeably.

[4]The process of mangrove degradation differs slightly from that of other land forest ecosystems. This is the case, as alternative land-uses such as conventional farming which local communities might otherwise have reverted to are not possible in the mangrove areas as they are regularly flooded by seawater.

(Beitl, 2014b)[5]. AUSCM were and are granted upon request, are non-transferable and are collectively held for 10-year periods subject to renewal. Importantly, communities needed to have a legally established collective entity to apply to the program, be it in the form of an association, cooperative, or commune. Furthermore, a detailed management plan had to be submitted when applying to the policy. These requirements related to the policy were challenging at least for some communities who already had monetary and time constraints and suddenly had to cope with additional and largely unfamiliar work. Many communities thus received technical assistance by external organizations such as NGOs and universities, to successfully handle bureaucratic tasks and formal complaints to officials when they noticed infringement of the mangrove deforestation ban (Beitl et al., 2019).

If accepted, communities gained exclusive rights over resources within the mangrove forests, also adopting duties of monitoring, and reporting on compliance per semester according to the submitted management plan. We understand both the administrative as well as the monitoring tasks as transaction costs associated with the policy. The communities were and are facing a trade-off between policy-related transaction costs and the benefits resulting from harvesting natural resources from the mangrove ecosystem.

The final granting of property rights involved the previously described application process, which was then followed by a demarcation period, entailing the required approval by executive decree by the Environmental Ministry of Ecuador. By 2020, 60 communities located in all coastal provinces of Ecuador where within policy, with over 90 having been historically part of the policy. Figure 1 showcases the distribution of all currently remaining mangroves across the Pacific coast of Ecuador. Please refer to Figure A.3, for a depiction of mangroves with and without property rights in the gulf of Guayaquil.

[5]According to Beitl (2014b), the people in the communities strongly identify as a fisher men and women and hope to protect the mangroves also to be able to pass traditions on to their children.

FIGURE 1 Mangrove coverage (marked in black) in Ecuador are our study area. Source: Own depiction.

3 | THEORETICAL FRAMEWORK AND HYPOTHESES

In the following we present our theoretical model which builds upon work by Usmani et al. (2022) and Souza-Rodrigues (2019). With this model we aim at investigating the theoretical effect of communities' property rights on illegal conversion of mangrove area[6], as well as the effect of NGO involvement on policy uptake and renewal, before addressing the questions empirically in section 5.

Each community is treated as a representative agent, assuming that the members of a community have a joint utility function $U(c, E)$. Utility is a function of consumption c and cultural and regulating ecosystem services $E(I, a)$. Ecosystem services $E(I, a)$ are affected by illegal land conversion activities $I(p)$. These are carried out by external actors and are a function of exogenous global shrimp prices p.[7] We assume that cultural and regulating

[6]Please note that mangrove deforestation was codified as illegal in Ecuador since 1994, thus total mangrove deforestation is equivalent to illegal deforestation. In the following we use the term 'illegal deforestation'.

[7]Illegal land conversion activities are also affected by other exogenous factors that have an impact on shrimp farm profitability as the sole alternative land use. Among others these factors are monetary sanctions for illegal mangrove deforestation and distance to centers of commerce. Please note at this point that the exogenous environmental variables e affecting the availability of fishing resources $F(a, I, e)$ are not the same as the factors driving shrimp farm profitability. We will come back to this point in section 5.1 which illustrates our identification strategy. Please also note that other land-uses such as other agricultural activities are not possible here as the land is regularly flooded by seawater.

ecosystem services decrease in illegal land conversion activities l:

$$\frac{\partial E}{\partial l} < 0 \tag{1}$$

as illegal conversion of mangrove area to shrimp farms entails deforestation of mangroves and thereby reduces mangrove cover and the related ecosystem services such as flood protection and carbon sequestration. Each community decides whether to acquire property rights a by adopting the policy described in section 2 and acts as a representative rational utility maximizing agent with a common income and time constraint T. Property rights a are defined as share of land parcels with communal property rights assigned to them.

We assume that cultural and regulating ecosystem services linearly increase in a, as communities are interested in keeping the ecosystem they acquired property rights for intact to increase extractable fishery resources :

$$\frac{\partial E}{\partial a} > 0; \frac{\partial^2 E}{\partial a^2} = 0, \tag{2}$$

Each community's utility function is assumed to be twice differentiable, continuous, and concave and increases in consumption, as well as in regulating and cultural ecosystem services:

$$\frac{\partial U}{\partial E} > 0; \tag{3}$$

$$\frac{\partial U}{\partial c} > 0 \tag{4}$$

A community's available budget consists of extracted environmental goods $F(a, l, e)$, i.e. edible water organisms, such as crustaceans and molluscs with the price p_f.

The level of fisheries resources $F(a, l, e)$ depends on habitat state and size (Barbier, 2017). As property rights are intended to prevent alternative land-uses and the concurrent destruction of the organisms' hydrological habitat, the level of fisheries is affected by the amount of land for which a community holds property rights a. We assume that F increases linearly in a:

$$\frac{\partial F}{\partial a} > 0; \frac{\partial^2 F}{\partial a^2} = 0, \tag{5}$$

Apart from a, the available fisheries resources are impacted by exogenous environmental parameters, such as salinity, temperature and soil type. This is the case as the mentioned organisms can only exist and survive under specific environmental conditions. We define e as the share of soil type with the adequate mineral composition for marine organisms to flourish available within a land parcel. Heterogeneity in e thus represents an exogenous source of variation in fisheries resources $F(a, l, e)$ across land parcels and communities, we will exploit in our identification strategy. We assume

$$\frac{\partial F}{\partial e} > 0; \frac{\partial^2 F}{\partial e^2} < 0 \tag{6}$$

Since there are other restrictions to the growth of marine resources such as limited food availability, the resources grow in e at a decreasing rate. Just as with the regulating and cultural services we furthermore assume that provisioning services, i.e. marine resources decrease in illegal land conversion activities:

$$\frac{\partial F}{\partial l} < 0; \tag{7}$$

If a community decides to acquire property rights it is entitled to extract the available provisioning services as a source of income. It is however also obliged to comply with legal and administrative requirements associated with the policy as well as an obligation to monitor the mangroves. For these activities the community incurs transactions costs $TAC = w \cdot t(N) \cdot a$ by allocating time $t(N)$ measured in time per land parcel with property rights to administrative tasks and monitoring activities at wage rate $w > 0$.
Involvement of external institutions, such as non-governmental organizations (NGOs) $N(a, f)$ can however support communities with regards to administration time effort. We assume

$$\frac{\partial t(N)}{\partial N} < 0, \frac{\partial^2 t(N)}{\partial N^2} < 0 \tag{8}$$

Implying that administration time connected to policy roll-out can be reduced by NGO involvement at a decreasing rate. Apart from a NGO involvement is also affected by the availability of exogenous (international) funding f. We assume that NGO involvement linearly increases with amount of property rights

$$\frac{\partial N}{\partial a} < 0, \frac{\partial^2 N}{\partial a^2} = 0 \tag{9}$$

and increases with available exogenous funding f at a decreasing rate:

$$\frac{\partial N}{\partial f} > 0, \frac{\partial^2 N}{\partial f^2} < 0 \tag{10}$$

The former relies on the intuition that the administrative effort for communities increases with the amount of land with property rights (e.g. a larger area needs more mapping and surveying effort). Income is allocated to consumption c and transaction costs of the policy [8] resulting in equation 12. In the following we use subscripts to represent partial derivatives.

Thus, the following optimization problem represents the decision each community faces in a specific year when deciding whether to enter or stay in the policy. For ease of reading, community and time-specific subscripts are left out here.

$$\max_{a,c} U = U[c, E(l(p), a)] \tag{11}$$

s.t. income constraint

$$F(a, l, e) \cdot p_f = c + w \cdot t(N(a, f)) \cdot a \tag{12}$$

[8]The time spent fishing is not included in the model, as it is not necessary to derive the following hypotheses.

and s.t. time constraint

$$T > t(N(a,f)) \quad a \tag{13}$$

The Lagrangian for this optimization problem is thus:

$$\begin{aligned} L = {} & U[c, E(I(p), a)] + \\ & (F(a, I, e) \quad p_f - c - w \quad t(N(a, f) \quad a) + \\ & \qquad (T - t(N(a, f)) \quad a) \end{aligned} \tag{14}$$

We use the first order conditions to apply the implicit function theorem using Cramer's rule (please refer to B) and get:

$$\frac{\partial I}{\partial e} = \frac{F_e}{F_I} < 0 \tag{15}$$

We assume that fishery resources increase with the share of soil type with an adequate mineral composition for marine species to prosper within a land parcel (assumption made in equation 6), which renders the enumerator of the previous expression positive. We furthermore assumed that fishery resources decrease in illegal mangrove deforestation activities (see equation 7), due to the concurrent destruction of the marine organisms' habitat, rendering the denominator and with it the whole expression negative. The corresponding hypothesis we test with the specifications further described in section 5.1 is thus:

***H* 1:** *Soil type is an exogenous predictor of fishery resources, which in turn results in an increased utility of acquisition of property rights by local communities leading to reduced deforestation of mangroves.*

In order to derive our second hypothesis we apply the implicit function theorem using Cramer's rule again (please refer to B) and get:

$$\begin{aligned} \frac{\partial a}{\partial f} = {} & \frac{U_c \quad w \quad t_N \quad N_f}{U_c \quad w \quad 2t_N \quad N_a} = \\ & \frac{N_f}{2N_a} > 0 \end{aligned} \tag{16}$$

With assumption 10 we get a positive enumerator and with assumption 9 a positive denominator, resulting in an overall positive expression.
This leads to the second hypothesis, which we test using the identification strategy described in section 5.2:

***H* 2:** *Exogenous international aid disbursements as a predictor of NGO involvement increase the probability of property rights acquisition and continuation by local communities*

4 | DATA

In the following we describe how we compiled and processed our data set from various sources. The variable of interest, mangrove forest coverage in a defined spatial unit is available as a $1\ \mathrm{km}^2$ resolution grid for the years 2000 – 2012 (Hamilton, 2015). The raster value indicates the mangrove coverage between zero and 955 square meters. We included raster cells that had a mangrove forest coverage larger than zero square meters in one of the mentioned years as observations in our final data set and used the centroid of each cell to extract data from other spatially overlapping data sources [9]:
Please refer to Hamilton and Casey (2016) for detailed information on data pre-processing of the mangrove cover grid. Next, we created a deforestation variable by subtracting the forest cover in of a year t with the mangrove cover in $t - 1$ and multiplied the variable with minus one. The larger the value of the new variable is, the stronger the illegal land conversion in a cell.
In order to define whether a cell is treated, i.e. was part of the policy and had property rights assigned to it, we acquired the geographical demarcation of communities with property rights in the AUSCM as vector data and information on the duration of treatment for each community.

Additionally, we extracted covariates describing the communities such as number of members, number of reports issued to the government as well as their size from the management reports communities submitted. We merged the data on communities with the spatial information using the names of communities as unique identifiers. Moreover, we acquired data on the presence and timing of involvement of external actors, as well as on the type of organisation and source of funding supporting each community from the same source.
Furthermore, we included covariates which drive the profitability of shrimp-farming as the sole alternative land-use. We included mean annual temperature, calculated based on Harris et al. (2014) down scaled with the procedure by Fick and Hijmans (2017), population density extracted from Center for International Earth Science Information Network - CIESIN - Columbia University (2018), as well as euclidean distance to the closest major city, calculated based on a vector data set by World Bank (2017).

Since several protected areas were created previous to the policy (Rodríguez, 2018), we include the legal protection status of each raster cell as a control variable. We extracted it from a spatial data set providing information on the start date, type and geographical position of legal protection status acquired from the Ministry of the Environment of Ecuador. For our identification strategy described in section 5.1 we compiled spatial data on the presence of shellfish, as well as red and blue crabs with maps provided by the Ministry of Environment and from the National Institute of Aquaculture and Fisheries Research in Ecuador and a map of soil types by Dijkshoorn et al. (2005).[10] Furthermore we acquired returns for elections to the U.S. House from MIT Election Data and Science Lab (2017) and data on annual foreign aid distributed by USAID (USAID, 2021) for the identification strategy described in section 5.2. Please refer to Table C for a list of variables with respective sources.

[9] All available data sets were reprojected to the WSG 84 / UTM zone 17S projection and preprocessed in QGIS and R

[10] Please refer to Figure A.4 for a screenshot of the data showing the spatial distribution of fishery resources in an example community.

5 | ECONOMETRIC FRAMEWORK AND IDENTIFICATION STRATEGY

In order to test the hypotheses presented in section 3, we need to account for the endogeneity of policy adoption by communities and NGO involvement.
To test hypothesis H1 we exploit the dependence of aquatic organisms on environmental parameters, i.e. selected soil types as exogenous predictors of policy adoption, by carrying out an instrumental variable approach, which is illustrated in Figure 2 and further described in subsection 5.1. Second, we investigate the causal impact of NGO involvement on policy adoption by testing hypothesis H2 with a regression discontinuity design illustrated in Figure 2 and described in more detail in subsection 5.2.

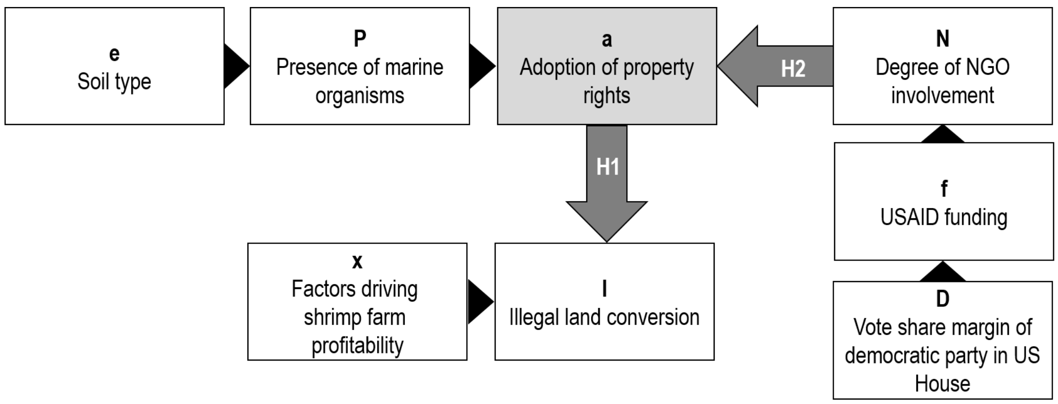

FIGURE 2 Illustration of the identification strategy carried out in sections 5.1 and 5.2, to test hypotheses H1 and H2 derived in section 3 respectively. We use selected soil classes as exogenous environmental parameter e with $\mathsf{cov}(\mathsf{e},\mathsf{P}) >> 0$ and $\mathsf{cov}(\mathsf{a},\mathsf{P}) >> 0$. Source: Own depiction.

5.1 | Effect of Property rights on mangrove deforestation

As stated in section 3 local communities rely on harvesting marine organisms to secure their subsistence. The presence of the aquatic species *Ucides Occidentalis* (red mangrove crab), *Cardisoma crassum* (blue mangrove crab) and *Anadara tuberculosa* (black cockle) is thus an important predictor of treatment, as their presence makes it more attractive for communities to protect the mangrove ecosystem. As depicted in Figure 2, their presence P is dependent on exogenous environmental parameters, one important environmental parameter being the soil type e. As communities self-select into the program, adoption of property rights is endogenous. In order to identify the effect property rights a have on illegal land conversion activities l, we thus exploit the exogenous variation in soil types as instrumental variable. *Ucides Occidentalis* and *Cardisoma crassum's* habitat is characterized by silty-clayey substrates (Alemán and Ordinola, 2017) while *Anadara tuberculosa* requires a "muddy" substrate with a high water content (Diringer et al., 2019). To verify that the soil types we theoretically expect to be exogenous predictors of the respective species are in fact suitable instruments, we regressed soil type classes extracted from Dijkshoorn et al. (2005) on the presence of each species. In line with the literature, cambisols (CM) and planosoils (PLe) seem to be habitats of shellfish and crabs[11]. Hence, we created a dummy variable indicating the presence of those soil classes as our instrumental variable, hereinafter called soil IV[12]. In the context of forest conservation policy evaluation, instruments are expected not to affect land-cover change except through the probability of treatment. For the instrument to work, it needs to be truly exogenous and sufficiently correlated with the treatment variable (Sovey and Green, 2011; Ferraro and Hanauer, 2014). We argue that the exogeneity assumption is met, as the soil type is exogenous to the decision of shrimp farmers' illegal land conversion activities l. The diagnostics in Table 1 show that the soil type used as instrument is sufficiently strong, since a first-stage partial-F test by Staiger and Stock (1997) rejects the Null that the soil IV is weak, i.e. not sufficiently correlated with the treatment at a 0.1% significance level. A Wu-Hausman test reveals that a two- stage least squares estimation is more consistent than ordinary least squares (OLS) estimation. Please refer to Figure A.5, which shows the overlap of the two identified soil types used as combined soil IV with policy adoption.

TABLE 1 Diagnostics of soil type used as instrument in Models 1 to 3 of Table 2 show that the instrument is sufficiently strong. A Wu-Hausman test indicates that the IV approach is more consistent than a simple OLS regression.

	df1	df2	statistic	p-value
Weak instruments (F-Test)	1	13, 991, 767	115, 565	$< 2e-16$
Wu-Hausman	1	13, 991, 766	508.5	$< 2e-16$

With the necessary model assumptions met, we hence carried out an instrumental-variable regression by two-stage least square estimation:

[11] Cambisols are characterized by sandy or loamy surface horizons with at least 8% clay content, while Planosols have a coarser top horizon which shows signs of water stagnation due to a clayey, slowly permeable sub-horizon (WRB, 2014). Please refer to Table D.5 for the results of the auxiliary regression.

[12] Our work here is most similar to the approach of Sims (2010), who uses provision of hydrological services and watershed status as an instrument for land conservation policies on development outcomes. We combined all CM and PLe into a single dummy variable and used it as an instrument.

$$I = \quad + \quad a + \quad L + \quad x + u \tag{17}$$

$$a = \quad + \quad s + \quad L + \quad x + k \tag{18}$$

With I as illegal deforestation variable, calculated as described in section 4, treatment a, legal protection status L, a matrix of variables driving the profitability of the alternative land-use x, the soil IV s and error terms u and k. Please note that our chosen IV design identifies a local average treatment effect (LATE) (Ferraro and Hanauer, 2014).

5.2 | Effect of Non-governmental organization involvement on adoption of property rights

For testing hypothesis H2 we applied a regression discontinuity design within treated units. Regression discontinuity relies on treatment status being fully or partly dependent on a "running" or "forcing" variable crossing a known threshold (Lee and Lemieux, 2010). For the identification of the causal effect of NGO involvement N on policy adoption a, we thus rely on partisan differences over foreign aid allocation in the US Congress. We argue that the NGOs which are fully or partly funded by USAID will adjust their level of support to communities depending on the available funds allocated to Ecuador. The legislative bodies in US government play a defining role in determining the amount of US foreign aid, as they authorize policy and appropriate funds (Lee and Lemieux, 2010). According to Ahmed (2016), the composition of Congress influences foreign aid disbursements[13], with a liberal Congress supporting foreign aid more than a conservative one. We thus defined the vote margin of the Democrats to Republicans in the US House as running variable D[14]. A positive running variable indicates a relative majority of democrats in the US House. At the cutoff point $c = 0.004$[15], the democrats have a slight relative majority of votes. The running variable is exogenous to the factors that determine whether a community adopts policy or an NGO supports a specific community, as voters who elect the House of Representatives cast their vote based on, or in response to national or regional political and economic conditions in the US (Ahmed, 2016). As the level of NGO involvement is likely not fully determined by the described mechanism, we use a fuzzy regression discontinuity design. The treatment effect $_1$ can thus be estimated as follows:

$$_1 = \frac{\lim_{\#0.004} E[a|D = c + \ ] - \lim_{"0.004} E[a|D = c + \ ]}{\lim_{\#0.004} E[a|N = c + \ ] - \lim_{"0.004} E[a|N = c + \ ]} \tag{19}$$

[13] Evidence in the literature shows that the composition of congress also influences environmental policy, with a democratic leaning congress being more supportive of stringent environmental policy (Sussman, 2004; Kim and Urpelainen, 2017; Pacca et al., 2021).

[14] We used MIT Election Data and Science Lab (2017) to calculate the share of votes each of the party received nationwide in each election. In a second step we took the difference of the democrats and the republican vote share. We use the resulting vote margin in the House as opposed to the Senate as running variable, since all members of the house are subject for re-election in every bi-annual election, introducing more variability, which is not the case in the Senate (Ahmed, 2016).

[15] The value 0.004, i.e. 0.4% is the minimum positive value measured in the data. At 0 no party has the relative majority of votes.

With involvement of NGOs fully or partly funded by USAID N, running variable D, adoption of property rights a and cutoff c. Please refer to Lee and Lemieux (2010) for detailed information on the regression discontinuity design and to F.2 for the results of falsification tests to inspect empirical regularities that are expected to hold in most cases where the identifying assumptions of the regression discontinuity design are met (Cattaneo et al., 2020b).

6 | RESULTS

6.1 | Effect of property rights on illegal mangrove deforestation

Table 2 shows the results of the estimation strategy described in section 5.1. Column (1) shows the significant negative effect of treatment a on illegal mangrove deforestation I. It indicates that adoption of property rights a significantly reduces illegal mangrove deforestation. Hence we do not reject hypothesis H1, which means that the assignment of communal property rights to ancestral communities has a causal link to reduced mangrove deforestation in Ecuador.

The coefficient of control L, indicating the legal protection status of each cell is negative and significantly different from zero. This suggests that legal protection measures such as the designation of protected areas (PA) seem to work in preventing mangrove deforestation as legal protection status reduces illegal deforestation in our model. The latter results are exploratory, since legal protection status is likely endogenous. Compared to the effect of property rights assignment PA are around 80% less effective in preventing illegal land conversion. The results are robust to the inclusion of spatial fixed effects (Column 2) and further covariates (Column 3).

As expected the coefficients of the covariates population density (pop_T) and temperature (tmp_T) are positive and significant, which we propose capture the effects of the profitability of alternative land uses on deforestation. The coefficient of the variable indicating the euclidean distance to relevant business centers has an unexpected negative coefficient. We argue that the chosen distance measure is the reason for this unexpected result, as it cannot account for topography and possibly bad road quality in remote areas, thus not representing driving time or distance to the closest relevant business center accurately. Please refer to Table F.8 for robustness checks with clustered standard errors to account for spatial autocorrelation.

With our results it is possible to estimate the total amount of CO2 emissions avoided by the policy, which we then value at the social cost of carbon (SCC). We make use of the estimates presented in Table 2 to calculate the area of prevented mangrove conversion in each year between 2000 and 2012 [16]. Next we multiply the conserved area in hectare with an average blue carbon emission factor by Alongi (2020) quantifying the annual emissions resulting from conversion of a mangrove ecosystem to aquaculture: 614.4 $\mathrm{tC\ ha^{-1}\ a^{-1}}$ [17]. Based on these calculations, we find that the policy prevented the release of 529,380 tC to the atmosphere between 2000 and 2012 which is equivalent to more than 1.5 million tCO2. Valued at the SCC, 112.86 US\$=tC (Wang et al., 2019), this corresponds to 59.7 million US \$ in avoided damages due to future climate change impacts.

Aside from benefits of avoided emissions, which are a global public good provided by the

[16] We do this by multiplying the average policy effect with the total mangrove area with property rights in each year.

[17] 1802.2 $\mathrm{tCO2\ ha^{-1}\ a^{-1}}$ 0.34 tC=tCO2, with the conversion factor from EPA (2021).

TABLE 2 Results of effect of treatment a on illegal mangrove deforestation I identified by two-stage least squares estimation with a soil IV which is further described in section 5.1.

	Dependent variable:		
	I		
	(1)	(2)	(3)
Constant	0.523	0.652	- 7.499
	(0.020)	(0.027)	(0.316)
a	- 1.767	- 2.016	- 1.765
	(0.076)	(0.088)	(0.083)
L	- 0.168	- 0.385	- 0.335
	(0.012)	(0.020)	(0.018)
distance			0.00001
			(0.00000)
pop_T			0.00003
			(0.00001)
tmp_T			0.309
			(0.013)
Spatial fixed effects	False	True	False
Observations	13,991,770	13,991,770	13,991,770
Residual Std. Error	9.960 (df = 13991767)	9.964 (df = 13991762)	9.957 (df = 13991764)

Note: p<0.1; p<0.05; p<0.01

policy, we also explore the fisheries benefits of the intervention, which are received by the local communities.
We calculate the additional estimated fisheries income provided by the mangroves protected by the policy [18]. We do this for the two main fisheries, which comprise 90% of all existing concessions in our sample, the *Ucides Occidentalis* (red crab) fishery and *Anadara tuberculosa* (black cockle) fishery. In the final year of our period of analysis a total of 4,878 families held communal property rights over mangroves and earned their income from the revenues stemming from both fisheries.

For the red-crab fishery we employ the available catch per unit effort (CPUE) (Alemán-Dyer et al., 2019) associated with this fishery (14±2 units per man-hour), which was calculated based on communities within policy. Data on seasonal closings and effective fishery days per year (240 days) were obtained from Alemán-Dyer et al. (2019). Average prices for a 14 crab bundle are 15 US$, with ex-vessel prices that fisheries receive being 50-60% of the final price (Bravo, 2013). We take into account the average costs per effective fishing day (Bravo, 2013) correcting for annual inflation. [19] Making use of the total number of red-crab fishers within policy, we estimate an average undiscounted net income of more than 134 million $ earned within the first 12 years of policy. Expressed in average benefits of the policy, understood as additional red-crab fishery resources provided by mangrove conservation, we estimate a total added benefit of 281 k $ for the years 2000-2012.

We employ a similar approach for the *Anadara tuberculosa* (black cockle) fishery. The average CPUE of 180 units per fisher per effective day of fishery is calculated for communities within policy. We employ data from the National fisheries institute on both market prices and costs for the fisheries arriving at a daily net benefit of 14 $ per fisherman and day (Cáceres and Gaibor, 2019). There are 23 effective days of fishing a month in the black cockle fishery. Making use of net fishery income per day per fisherman, and the number of fishermen per year, we arrive at an average undiscounted net income of 75.4 million $ earned within the first 12 years of policy. Expressed in average benefits of the policy, understood as additional red-crab fishery resources provided by mangrove conservation, we estimate a total added benefit of 158 k $ for the years 2000-2012.

6.2 | NGO presence and policy adoption

Table 3 shows the effect of NGO involvement on policy uptake and continuation, estimated within treatment. We find a positive and significant effect of NGO involvement[20] and do not reject hypothesis H2. The second and third column show the results estimated with a first and third order parametric polynomial estimator of a fuzzy regression discontinuity design (RDD) with clustered and heteroskedasticity corrected standard errors, using the Democrat's vote share margin compared to the Republicans in the US House as running variable D. The involvement of NGOs increases the probability of property rights adoption.

We believe that apart from external institutions such as NGOs, endogenous institutions and leadership on community level play a role in policy uptake. As we do not have a suitable instrument for strength of endogenous institutions, we cannot formally test this intuition. Arguing that the number of reports issued to the authorities by communities who

[18]Just as in section 3 we assume a linear relationship between mangrove area and population sizes and resulting fishing effort.
[19]We use cost data to avoid over-estimations of fisheries benefits that stem from gross income measures.
[20]Please note that we only considered data on the NGOs with full or partial funding from USAID here.

decided to adopt property rights might be a useful proxy for the strength of endogenous institutions, we compare deforestation rates between communities that issued a number of reports above and below the median number of reports issued by all communities. We find that the communities who issued more than the median number of reports have significantly lower deforestation rates than the comparison group (please refer to figure E.6). We consider these findings to be exploratory results as we do not have a suitable instrument for strength of endogenous institutions.

TABLE 3 Effect of NGO presence on policy uptake and continuation. The results were estimated with a first and third order parametric polynomial estimator of a fuzzy regression discontinuity (RDD) both with clustered and unclustered standard errors using the Democrat's vote share margin compared to the Republicans in the US house as running variable D

	Dependent variable:		
	a		
	(1)	(2)	(3)
Constant	0.035	0.035	- 0.070
	(0.001)	(0.091)	(0.041)
N	1.491	1.491	1.745
	(0.004)	(0.144)	(0.265)
D	- 0.001	- 0.001	- 0.001
	(0.00001)	(0.0004)	(0.001)
D^2			0.0001
			(0.0001)
D^3			0.00000
			(0.00000)
D_right	0.001	0.001	0.015
	(0.00002)	(0.001)	(0.009)
'D^2_right'			- 0.0004
			(0.0003)
Observations	6,737,471	6,737,471	6,737,471
Order of the polynomial	1	1	3
Clustering on District level	FALSE	TRUE	TRUE
Residual Std. Error	0.534 (df = 6737467)		

Note:

p<0.1; p<0.05; p<0.01

7 | DISCUSSION

Our work presents a causal analysis of the effect of assigning communal property rights to local and indigenous communities on deforestation, providing the, to our knowledge, first assessment for mangroves ecosystems. We evaluate 12 years of the AUSCM property rights based policy in Ecuador, finding that property rights have a positive impact on reducing deforestation, which had destroyed 30 to 40 percent of all mangrove coverage in the country (Friess et al., 2019). Our results are significant and robust across specifications and when accounting for alternative land-use, using fixed effects and when clustering standard errors to control for spatial auto-correlation.

The existing literature provides mixed evidence, both empirical and theoretical, on the role of property rights on deforestation. On one hand, more secure land tenureship can reduce mangrove deforestation by increasing the present value of standing forests. This in turn discourages land conversion to productive use as a way to reduce expropriation risk, as has been found in Brazil, Haiti, and Malawi (Araujo et al., 2009; Place and Otsuka, 2001; Busch and Ferretti-Gallon, 2017). On the other hand, more secure land tenure might spur an increase in deforestation by encouraging greater investment in productive activities (Busch and Ferretti-Gallon, 2017). Our findings add to this debate, in line with the recent body of literature which has found positive conservation effects of communal property rights in the Peruvian and Brazilian amazonian forest (Blackman et al., 2017; Baragwanath and Bayi, 2020), expanding this literature by providing a novel identification strategy and unique dataset.

This highlights the role local and indigenous communities play for improved land stewardship, which is vital to achieve the climate change goals set forth in the Paris Agreement (Griscom et al., 2017). As mangrove deforestation is not only associated with increased carbon emissions, but also reduced future carbon uptake, our evaluation of property rights to local and indigenous communities supports its application as an effective climate change mitigation strategy. Our estimates show that the policy prevented additional emission of more than 1.55 million tCO_2 between 2000 and 2012 which is equivalent to almost 60 million US$ of avoided damages resulting from climate change. This has positive implications for climate change related north-south transfers, specifically as this provides strong evidence of additionally avoided emissions. Recent experimental evidence suggest that collective ownership might be well-suited for payments of ecosystem services (Kaczan et al., 2017), which seems to align with our empirical results.

Thus our work provides evidence that nature based solutions to climate change provide much promise, but their implementation must be based on a track record of success, which we believe we asses and provide evidence for through our empirical approach. Moreover, our results are in line with recent evidence on the role of local communities and indigenous peoples in safeguarding forests and their associated carbon stocks. In the Amazon indigenous land tenure and management programs between 2003 to 2016 were more than twice as effective in safeguarding carbon sinks than other approaches (Walker et al., 2020). As the role of mangroves in coastal protection has been well documented, the positive implications of successful conservation interventions can be extended to the nascent policy debate around climate change adaptation (Del Valle et al., 2020; Hochard et al., 2019).

Nevertheless, our result contrasts with similar studies finding little effect of property rights on deforestation (Probst et al., 2020; BenYishay et al., 2017). Our work differs from this existing literature in that we propose a theoretical model of the mechanisms behind the drivers of policy adoption at the community level, and derive identification strategies from its underlying intuition. This ensures that we not only asses the effectiveness of granting property rights to local and indigenous communities on deforestation, but also identify the possible causal drivers behind policy failure or success (Deaton, 2010; Ferraro et al., 2012; Busch and Ferretti-Gallon, 2017). A potential weakness of our study is that it fails to capture the dynamic nature of the policy, methods such as staggered diff- in diff not possible due to missing pre-policy data necessary to verify parallel trend assumption.

The presence of common-pool resources such as fisheries in mangrove forests, provide clear incentives for ancestral communities to enroll in time consuming and costly property rights based policy. The effects of exclusivity rights over common pool resources, and its potential positive effect on their management is backed by the commons literature (Schlager and Ostrom, 1992; Ostrom, 2000; Ostrom, 2010). We exploit the presence of these, and in particular the exogeneity of soil type as predictor of policy adoption/fisheries presence, to derive our results through an instrumental variable approach. Our findings imply that property rights are associated not only with the conservation of mangroves, but possibly the fisheries associated with them. A significant percentage of mangroves under AUSCM policy have a multitude of fisheries which are of high local market value. This might be suggestive of positive food security, development and poverty impacts of property rights, which would be in line with described causal pathways linking property rights interventions and poverty outcomes (Miller et al., 2021; Ferraro and Hanauer, 2014). Moreover, this provides a potential explanation for our result's divergence from the empirical property rights literature on land forest ecosystems, as mangrove forests offer different provisioning services than other forest ecosystems which are not flooded regularly.

Furthermore, through our theoretical model we propose that high transaction costs of the policy have a negative effect on policy uptake, but that the presence of exogenous institutions such as NGOs plays a role in absorbing said costs. This is in line with recent evidence from both experiments and quasi-experiments looking into the role of NGOs in reducing transaction costs (Usmani et al., 2022) and fostering environmental compliance (Grant and Langpap, 2019; Grant and Grooms, 2017). Additionally our results providence evidence on the role of transaction costs in addressing global environmental externalities (Libecap, 2014; Libecap and Lueck, 2011; Ayres et al., 2018). We test the effect of the involvement of NGOs across communities on policy enrollment and renewal, by exploiting data on USAID funding and, specifically partisan preferences for environmental aid from the U.S Congress as source of exogenous variation in NGO involvement to address endogeneity of NGO presence. We find that the presence of NGOs increases the probability of policy enrollment and continuation. This result might also be indicative of preliminary evidence of the effectiveness of international aid, which has been subject to questioning (Bourguignon and Sundberg, 2007; Deaton, 2010), whilst also adding evidence of partisan preferences in the US for both aid disbursements and environmental policymaking (Pacca et al., 2021; List and Sturm, 2006). To our knowledge our work presents the first evidence on the effect of partisan preferences from a large donor country in environmental policy outcomes in a recipient country.

We propose two mechanisms through which we believe property rights policies can successfully decrease mangrove deforestation 1) increased benefits for indigenous and local communities through exclusive rights over resources and 2) decreased transaction costs by the presence of external institutions like NGOs. Furthermore, we are convinced that endogenous institutions play a central role in the success of management of a valuable resource (Sutter et al., 2009), albeit, we only present exploratory evidence of that role.Moreover, a quick browse over our data of all legal entities that enrolled in the property rights policy shows that 95 percent are associations of fishermen, with existing internal rules and regulations. This is both supportive of our proposed mechanisms of fisheries benefits being a driver of policy success, but can also be interpreted in that endogenous institutions play an important role in the observed conservation outcomes. Finally we believe that our work showcases the wide diversity of institutional arrangements in place for the management of a common-pool resource, albeit at large scale, effectively presenting a polycentric governance system where international donors, NGOs and civil society, indigenous peoples, local and national governments effectively tackle one of the oldest issues concerning economics, the management of the commons. Given the value of the vast swathes of land under indigenous tenureship as some of the last reservoirs of untouched natural capital on Earth, we hope our work contributes to the discussion regarding the two ultimate commons problems of the twenty-first century, global climate change and the species extinction crisis.

DECLARATION OF INTEREST

No potential conflicts of interest were reported by the authors.

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FIGURE A.3 Treated (orange) versus non-treated (green) mangroves in the gulf of Guayaquil. Own depiction based on data acquired from the Ministry of Environment of Ecuador and Ecuador Coastal Resources Management Project.

A | SPATIAL DISTRIBUTION OF TREATED AND NON-TREATED MANGROVES IN THE LARGEST ESTUARY IN ECUADOR

FIGURE A.4 Spatial distribution of shery resources in an example community. Blue represents the presence of shell sh, while the red polygons indicate the presence of crabs. Source: Ministry of Environment and the National Institute of Aquaculture and Fisheries Research in Ecuador.

FIGURE A.5 Overlap of treatment with soil types CM and PLe used as combined instrumental variable in the Gulf of Guayaquil. This gure is supposed to give the reader an intuition of the spatial distribution of relevant soil types and policy adoption in a example estuary.

B | A COMMUNITY'S OPTIMIZATION PROBLEM

$$\max_{a,c} U = U[c, E(I(p), a)] \tag{B.20}$$

s.t. income constraint

$$F(a, I, e) \quad p_f = c + w \quad t(N(a, f)) \quad a \tag{B.21}$$

and s.t. time constraint

$$T > t(N(a, f)) \quad a \tag{B.22}$$

The Lagrangian for this optimization problem is thus:

$$\begin{aligned} L = {} & U[c, E(I(p), a)] + \\ & (F(a, I, e) \quad p_f - c - w \quad t(N(a, f)) \quad a) + \\ & (T - t(N(a, f)) \quad a) \end{aligned} \tag{B.23}$$

The corresponding rst order conditions for the Lagrangian shown in equation B.23 are as follows:

$$L_a = U_E \quad E_a + \quad (F_a \quad p_f - w \quad (t(N(a, f)) + t_N \quad N_a \quad a)) - \quad (t_N \quad N_a \quad a + t(N(a, f))) = 0 \tag{B.24}$$

$$L_c = U_c - \quad = 0 \tag{B.25}$$

$$L \quad = F(a, I, e) \quad p_f - c - w \quad t(N(a, f)) \quad a = 0 \tag{B.26}$$

$$L \quad = T - t(N(a, f)) \quad a > 0; \quad (T - t(N(a, f) \quad a) = 0 \tag{B.27}$$

Assuming that the members of a community spend some non-zero hours T with other activities such as shing and leisure, has to be equal to zero to full ll the constraint.

With = 0 and by substituting = U_c from B.25 in B.24, we get the following system of equations:

$$f_1 = U_E \quad E_a + U_c \quad (F_a \quad p_f - w \quad (t(N(a, f)) + t_N \quad N_a \quad a) = 0 \tag{B.28}$$

$$f_2 = F(a, I, e) \quad p_f - c - w \quad t(N(a, f)) \quad a = 0 \tag{B.29}$$

The Jacobinian of the two equations w.r.t. l and a is thus defined as

$$J_A = \begin{bmatrix} \frac{@f1}{@l} & \frac{@f1}{@a} \\ \frac{@f2}{@l} & \frac{@f2}{@a} \end{bmatrix} = \begin{bmatrix} 0 & U_E \;\; E_{aa} + U_c \;\; F_{aa} \;\; p_f - U_c \;\; w \;\; (t_N N_{aa} \;\; a + 2t_N \;\; N_a) \\ F_l \;\; p_f & F_a \;\; p_f - w \;\; (t_N \;\; N_a \;\; a + t(N(a, f))) \end{bmatrix} \tag{B.30}$$

with the determinant

$$\left|J_A\right| = -(U_E \;\; E_{aa} + U_c \;\; F_{aa} \;\; p_f - U_c \;\; w \;\; (t_N N_{aa} \;\; a + 2t_N \;\; N_a)) \;\; F_l \;\; p_f \neq 0 \tag{B.31}$$

The Jacobinian w.r.t. e and f is defined as

$$J_B = \begin{bmatrix} \frac{@f1}{@e} & \frac{@f1}{@f} \\ \frac{@f2}{@e} & \frac{@f2}{@f} \end{bmatrix} = \begin{bmatrix} 0 & -U_c \;\; w \;\; t_N \;\; N_f \\ F_e \;\; p_f & -w \;\; t_N \;\; N_f \;\; a \end{bmatrix} \tag{B.32}$$

we apply the implicit function theorem using Cramer's rule:

$$\begin{aligned} \frac{@l}{@e} &= \frac{\begin{vmatrix} 0 & U_E \;\; E_{aa} + U_c \;\; F_{aa} \;\; p_f - U_c \;\; w \;\; (t_N N_{aa} \;\; a + 2t_N \;\; N_a) \\ F_e \;\; p_f & F_a \;\; p_f - w \;\; (t_N \;\; N_a \;\; a + t(N(a, f))) \end{vmatrix}}{\left|J_A\right|} \\ &= \frac{-(U_E \;\; E_{aa} + U_c \;\; F_{aa} \;\; p_f - U_c \;\; w \;\; (t_N N_{aa} \;\; a + 2t_N \;\; N_a)) \;\; F_e \;\; p_f}{-(U_E \;\; E_{aa} + U_c \;\; F_{aa} \;\; p_f - U_c \;\; w \;\; (t_N N_{aa} \;\; a + 2t_N \;\; N_a)) \;\; F_l \;\; p_f} \\ &= \frac{F_e}{F_l} \end{aligned} \tag{B.33}$$

In order to derive our second hypothesis we apply the implicit function theorem using Cramer's rule again:

$$\begin{aligned} \frac{@a}{@f} &= \frac{\begin{vmatrix} 0 & -U_c \;\; w \;\; t_N \;\; N_f \\ F_l \;\; p_f & -w \;\; t_N \;\; N_f \;\; a \end{vmatrix}}{\left|J_A\right|} \\ &= \frac{F_l \;\; p_f \;\; U_c \;\; w \;\; t_N \;\; N_f}{-(U_E \;\; E_{aa} + U_c \;\; F_{aa} \;\; p_f - w \;\; (t_N N_{aa} \;\; a + 2t_N \;\; N_a)) \;\; F_l \;\; p_f} \\ &= \frac{U_c \;\; w \;\; t_N \;\; N_f}{-(U_E \;\; E_{aa} + U_c \;\; F_{aa} \;\; p_f - U_c \;\; w \;\; (t_N N_{aa} \;\; a + 2t_N \;\; N_a))} \end{aligned} \tag{B.34}$$

With assumptions 2, 5 and 9 this expression is reduced to:

$$\frac{@a}{@f} = \frac{U_c \;\; w \;\; t_N \;\; N_f}{U_c \;\; w \;\; 2t_N \;\; N_a} = \frac{N_f}{2N_a} \tag{B.35}$$

C | DATA SOURCES

TABLE C.4 Variable description and sources. Notation, description and sources of variables. A: Ministry of Environment of Ecuador; Ecuador Coastal Resources Management Project (PMRC), B: Management reports submitted to Ministry of Environment of Ecuador, C: Ministry of Environment of Ecuador; National Institute of Aquaculture and Fisheries Research in Ecuador.

Notation	Variable	Unit	Temporal resolution	Source
M	Mangrove coverage	m^2	2000-2012	Hamilton (2015)
L	Legal protection status	-	2000-2012	A
a	Treatment, i.e. demarcation of communities with property rights	-	cross-section	A
	Duration of treatment for each community	-	2000-2012	A
-	Size of communities	m^2	cross-section	B
-	Number of members of a community		cross-section	B
r	Total number of reports issued by each community	-	cross-section	B
N_all	Presence of external institution supporting communities		2000-2012	B
N	Presence of NGO funded by USAID supporting communities	-	2000-2012	B
mussle_rep	Presence of Mussles	-	cross-section	C
redcrab_re	Presence of red crabs	-	cross-section	C
-	Presence of blue crabs	-	cross-section	C
soiliv	Soil type	-	-	Dijkshoorn et al. (2005)
D	Vote shares of Democrats' in the U.S. House	%	2000-2012	Vote shares calculated based on MIT Election Data and Science Lab (2017)
N_aid	Annual foreign aid distributed to Ecuador by USAID	Mio US $	2000-2012	Calculated with USAID (2021)
pop_T	Population density in 2005	-	cross section	Center for International Earth Science Information Network - CIESIN - Columbia University (2018)
tmp_T	Mean annual temperature	°C	2000-2012	Calculated based on Harris et al. (2014) downscaled with Fick and Hijmans (2017)
distance	Distance to business centers	km	-	Calculated based on World Bank (2017)

D | AUXILIARY REGRESSION TESTING THEORETICAL ASSUMPTIONS ABOUT SOIL TYPE USED AS INSTRUMENTAL VARIABLE

TABLE D.5 Auxiliary regression with the variables indicating presence of mussels and red mangrove crabs ad dependent variables respectively and soil classes as predictors of their presence. As expected the cambisoils (CM) and planosoils (PLe) are positive predictors of the presence of mussels and red mangrove crabs, confirming soil class as suitable instrumental variable.

	Dependent variable:	
	mussle_rep	redcrab_re
	(1)	(2)
factor(DOMSOIL_UN)ARh	- 0.155***	- 0.127***
	(0.020)	(0.018)
factor(DOMSOIL_UN)CMd	- 0.094***	- 0.127***
	(0.007)	(0.006)
factor(DOMSOIL_UN)CMe	0.311***	0.250***
	(0.002)	(0.001)
factor(DOMSOIL_UN)CMo	0.532***	- 0.127*
	(0.081)	(0.075)
factor(DOMSOIL_UN)FLe	- 0.155*	- 0.127
	(0.093)	(0.086)
factor(DOMSOIL_UN)FLt	- 0.052***	- 0.041***
	(0.001)	(0.001)
factor(DOMSOIL_UN)LVf	- 0.136***	- 0.126***
	(0.003)	(0.003)
factor(DOMSOIL_UN)LVh	- 0.145***	- 0.127***
	(0.003)	(0.003)
factor(DOMSOIL_UN)PHl	- 0.155	- 0.127
	(0.228)	(0.211)
factor(DOMSOIL_UN)PLe	0.067***	0.076***
	(0.005)	(0.004)
factor(DOMSOIL_UN)VRe	- 0.112***	- 0.084***
	(0.009)	(0.008)
Constant	0.155***	0.127***
	(0.001)	(0.001)
Observations	1,090,700	1,090,700
Residual Std. Error (df = 1090688)	0.323	0.299
F Statistic (df = 11; 1090688)	6,438.786***	4,952.813***
Note:	*p < 0.1; **p < 0.05; ***p < 0.01	

E | DESCRIPTIVE STATISTICS

TABLE E.6 Descriptive statistics of sample for estimation strategy described in section 5.1. With mangrove cover within a cell M and illegal mangrove deforestation I in square meters, treatment status a, legal protection status L, euclidian distance to the closest business center [m], (water) temperature tmp_T [Celsius] and population density pop_T, as well as the instrumental variable indicating a suitable soil type (cambisols and planosoils) and thus habitat for marine species relevant for communities' subsistence.

Statistic	N	Mean	St. Dev.	Min	Pctl(25)	Pctl(75)	Max
M	13,991,770	758.29	235.82	0	754	859	955
I	13,991,770	0.11	9.94	0	0	0	955
a	13,991,770	0.19	0.39	0	0	0	1
L	13,991,770	0.48	0.50	0	0	1	1
distance	13,991,770	45,237.00	23,159.60	87.46	30,681.83	53,015.39	110,417.70
tmp_T	13,991,770	25.04	0.40	22.76	24.76	25.27	26.18
pop_T	13,991,770	498.85	495.12	0.00	11.83	1,024.20	2,384.71
redcrab	13,991,770	0.10	0.30	0	0	0	1
mussle	13,991,770	0.13	0.33	0	0	0	1
soiliv	13,991,770	0.05	0.23	0	0	0	1

TABLE E.7 Descriptive statistics of sample for estimation strategy within treatment described in section ??.With treatment a, presence of external organizations N_all, presence of non-governmental organizations with access to USAID funding N, voteshare margin in the US House $\mathsf{diff}_\mathsf{h}\mathsf{ouse}$ and variable indicating NGO access to overall funding distributed to Ecuador N_aid [million US$]. The mean of a is smaller than 1, as some communities dropped out of the policy or joined the policy later than other communities.

Statistic	N	Mean	St. Dev.	Min	Pctl(25)	Pctl(75)	Max
a	6,737,471	0.39	0.49	0	0	1	1
N_all	6,737,471	0.37	0.48	0	0	1	1
N	6,737,471	0.22	0.41	0	0	0	1
r	6,737,471	5.72	4.70	0	1	10	13
diff_house	6,737,471	- 0.002	0.06	- 0.07	- 0.06	0.07	0.09
N_aid	6,737,471	13.10	26.12	0	0	0	89

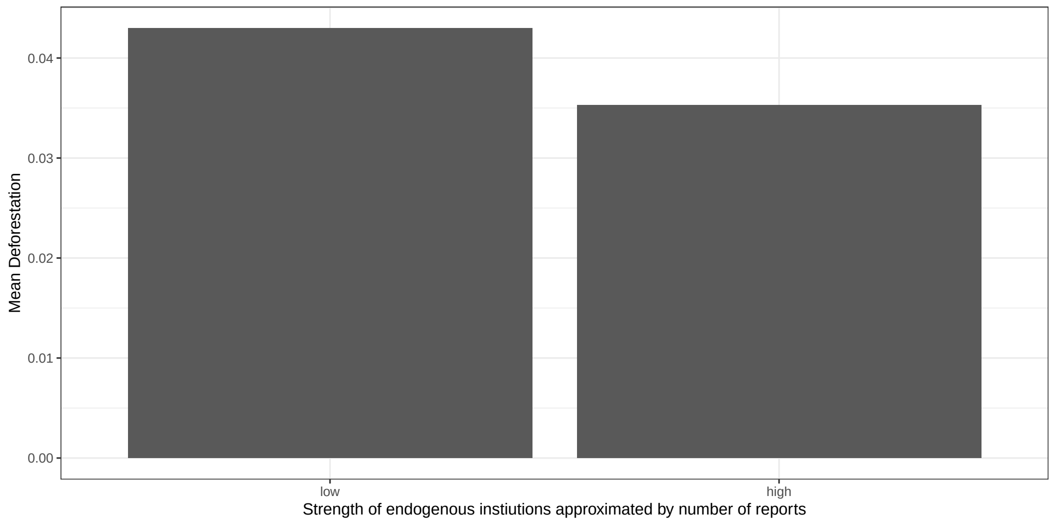

FIGURE E.6 The mean levels of deforestation for communities issuing more (high) or less (low) than the median number of reports, i.e. 6, issued by all communities to the authorities. A t-test reveals that the difference between the mean levels of deforestation is significant at a 0.01% significance level.

F | ROBUSTNESS CHECKS AND FALSIFICATION TESTS

F.1 | Accounting for spatial auto-correlation for Instrumental variable estimation strategy

As a robustness check we include specifications accounting for spatial auto-correlation by clustering standard errors in Table F.8 Unfortunately, choosing the correct level of clustering is not trivial, as it requires in-depth knowledge of the underlying spatial interaction processes. According to Cameron and D. L. Miller (2015) fewer and larger clusters have more variability than smaller clusters but result in less bias. We argue that clustering at the smallest, the community level is not appropriate in case of the specification testing for the effect of treatment on illegal deforestation (Table F.8) for two reasons: First of all, there might be (spatial) correlation between communities, for example, when two communities are geographically close and support each other in their monitoring efforts. Second, if we were to cluster at the community level, each community would represent a separate cluster and all non treated observations would belong to one big remaining cluster. Clustering at the community level would thus not represent the possible spatial interaction of non-treated cells belonging to one large cluster with proximate treated mangrove cells within a treated community. Due to these considerations, and the general consensus to avoid bias by using bigger more aggregate clusters further described in Cameron and D. L. Miller (2015), we cluster at the province level. Additionally we ran a k-means clustering algorithm to determine clusters from the data with unsupervised learning methods. The algorithm creates a random split of the data into a predefined number of k groups to then incrementally adjust clusters based on an Euclidean dissimilarity criterion. The algorithm converges when within-cluster variation is minimized (Please refer to Hastie et al. (2009, p. 460) for a detailed summary on the clustering method). The results remain significant regardless of the chosen level of clustering.

TABLE F.8 Robustness checks with clustered standard errors at province level and at cluster level determined by a k-means clustering algorithm which identifies clusters from the data with an unsupervised learning method.

	Dependent variable:						
	I						
	(1)	(2)	(3)	(4)	(5)	(6)	(7)
Constant	0.523	0.523	0.523	0.523	0.523	0.523	0.523
	(0.020)	(0.133)	(0.037)	(0.037)	(0.035)	(0.036)	(0.209)
a	- 1.767	- 1.767	- 1.767	- 1.767	- 1.767	- 1.767	- 1.767
	(0.076)	(0.491)	(0.091)	(0.134)	(0.122)	(0.138)	(0.719)
L	- 0.168	- 0.168	- 0.168	- 0.168	- 0.168	- 0.168	- 0.168
	(0.012)	(0.151)	(0.022)	(0.021)	(0.020)	(0.020)	(0.114)
Cluster	None	Province	k-means, k=5	k-means, k7	k-means, k=10	k-means, k=12	Year
Observations	13,991,770						
R^2	- 0.004						
Adjusted R^2	- 0.004						
Residual Std. Error	9.960 (df = 13991767)						

Note: $p<0.1$; $p<0.05$; $p<0.01$

F.2 | Falsification tests and robustness checks of Regression Discontinuity Design

In the following, we present the results of three falsification test showing that empirical regularities that should hold if the assumptions necessary to identify a causal effect are not breached. Table F.9 shows falsification tests on the available predetermined covariates i.e. relevant variables that might be correlated with USAID funded NGO involvement N, here the treatment, before treatment is actually assigned. Since covariates are measured before treatment occurs, the effect of treatment on the predetermined covariates should be zero (Cattaneo et al., 2020b). Table F.9 shows that this is the case for both available predetermined covariates. We conclude that relevant covariates are balanced around the cutoff and continue with analyzing the density of the running variable around the cutoff.

TABLE F.9 Falsification tests of predetermined covariates. The tests show no significant effects of treatment N on predetermined covariates. There were some missing values in the variable measuring the population of 2000.

	Dependent variable:	
	Cov	
	(1)	(2)
Constant	537.783	750.391
	(0.831)	(0.435)
N	0.000	0.000
	(3.441)	(1.802)
D	- 0.000	- 0.000
	(9.186)	(4.823)
D_right	0.000	0.000
	(14.329)	(7.523)
Observations	6,712,303	6,737,471
Cov	Population of 2000	Mangrove coverage in 2000
Residual Std. Error	455.641 (df = 6712299)	239.683 (df = 6737467)

Note:
$p<0.1$; $p<0.05$; $p<0.01$

Even though we deem it very unlikely that USAID funded NGOs based in Ecuador would be able to manipulate or influence US House election outcomes, we employ the robust density estimator by Cattaneo et al. (2020a), to test the null hypothesis that the density of the vote share margin used as running variable is continuous at the chosen cutoff. The test results presented in Table F.10 show that the null hypothesis continuous running variable is not rejected.

TABLE F.10 Results of the robust density estimator by Cattaneo et al. (2020a), showing that the null hypothesis of a continuous running variable is not rejected.

Number of obs =	6737471	
Model =	unrestricted	
Kernel =	triangular	
BW method =	estimated	
VCE method =	jackknife	
c = 0.004295	Left of c	Right of c
Number of obs	4146136	2591335
Eff. Number of obs	4146136	2591335
Order est. (p)	2	2
Order bias (q)	3	3
BW est. (h)	0.088047	0.088047
Method	T	P > jTj
Robust	-0.006	0.9952

Next, we carry out placebo tests using alternative cutoff values on all treated observations. By construction no treatment effect should be detectable at the placebo cutoff values, since there is no variation in treatment status (Cattaneo et al., 2020b). Table **??** shows the results of the placebo tests. No significant treatment effects were found using artificial cutoff values.

Even though we do not control for endogeneity of NGO presence in Table F.12, the results confirm that in presence of both policy and NGOs there are significant effects on reductions of deforestation. This, is in line with both our results, where NGO presence affects the adoption of the policy, and the policy itself significantly reduces deforestation. Moreover, we believe this is tentative evidence in support for presence of polycentric governance system, given the wide diversity of institutional arrangements that humans craft to govern, provide, and manage public goods and common-pool resources **Ostrom2014**

TABLE F.11 Placebo falsifications tests revealing that no treatment effect can be found at artificial cutoff values.

	Dependent variable:			
	a			
	(1)	(2)	(3)	(4)
Constant	1.000	1.000	1.000	1.000
	(0.000)	(0.000)	(0.000)	(0.000)
N	0.000	0.000	0.000	0.000
	(0.000)	(0.000)	(0.000)	(0.000)
D	0.000	0.000	0.000	0.000
	(0.000)	(0.000)	(0.000)	(0.000)
D_right	0.000	0.000	0.000	
	(0.000)	(0.000)	(0.000)	
Observations	2,655,757	2,655,757	2,655,757	2,655,757
Alternative cutoff	0.05	0.06	0.07	0.08
Residual Std. Error	0.000 (df = 2655753)	0.000 (df = 2655753)	0.000 (df = 2655753)	0.000 (df = 2655754)
Note:				$p<0.1$; $p<0.05$; $p<0.01$

TABLE F.12 Additional robustness check to test hypothesis using a two-stage least squares regression with the soil IV.

	Dependent variable:
	I
Constant	0.077
	(0.010)
a	- 0.388
	(0.074)
N	1.083
	(0.096)
a*N	- 0.843
	(0.128)
Observations	5,836,935
Residual Std. Error	5.272 (df = 5836931)
Note:	p<0.1; p<0.05; p<0.01