PARKS VERSUS PES: EVALUATING DIRECT AND INCENTIVE-BASED LAND CONSERVATION IN MEXICO

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ABSTRACT

Protected areas (PAs) and payments for ecosystem services (PES) are the top two mechanisms available for countries to achieve international REDD agreements, yet there are few empirical comparisons of their effects. We estimate the impacts of PAs and PES on forest conservation, poverty reduction and population change at the locality level in Mexico in the 2000s. Both policies conserved forest, generating an approximately 20-25% reduction in expected forest cover loss. PES created statistically significant but small poverty alleviation while PAs overall were neutral for livelihoods. Estimates by individual policy type for the same level of deforestation risk indicate that biosphere reserves and PES balanced conservation and livelihood goals better than strict protected areas or mixed use areas. This suggests both direct and incentive-based instruments can be effective but that financial support, flexible zoning and explicit recognition of livelihood goals are key components of conservation that does not harm livelihoods.

INTRODUCTION

Forest loss due to agricultural and pastoral expansion, logging, and urban development remains a persistent global environmental problem despite decades of experimentation with different policy solutions. Protected areas ("PAs") and payments for ecosystem services ("PES") head the list of land conservation policies that countries will rely on as they seek to reduce greenhouse gas emissions from land-use change under international climate agreements (Kerr 2013, Pfaff, Amacher and Sills 2013, REDD Desk 2016). Yet in the majority of countries, people also live on the lands important for reduced emissions from deforestation and forest degradation ("REDD"). This means that policymakers working to reduce emissions confront a choice not just about which mechanism will be most environmentally effective, but how to achieve conservation without compromising other social goals such as poverty alleviation.

In this paper, we provide the first explicit national-scale comparison of direct and incentive-based land conservation instruments across both environmental and social outcomes. Specifically, we estimate the impacts of protected areas and payments for ecosystem services in Mexico on forest conservation, poverty reduction and population change in the most recent decade (the 2000s) at the locality level. We measure changes over time by combining global forest cover change data from 2000-2012 (Hansen et al. 2013) and measures of poverty and population density from 2000-2010 (Mexican census; CONAPO) for each locality. We evaluate the impacts of all protected area types in Mexico, both as a group and broken into categories of strict versus more flexible protection. We compare these estimates to the impacts of Mexico's Federal Payments for Ecosystem Services program, a conditional cash transfer program that provides payments to selected private and community landowners in exchange for maintaining existing land cover. Our identification strategy compares localities with a higher share of land protected by PAs or PES with similar localities that do not benefit from these policies. Our regression model uses differenced outcomes to eliminate level effects and includes controls for pre-trends in each outcome, state fixed effects, and multiple controls for geographic and social criteria that determined selection into PAs or PES and could influence outcomes. We check for parallel trends in the pre-period and demonstrate robustness to a variety of specification checks including matching before estimation and re-calculating treatment effects using bounding methods.

Protected areas are a form of direct, involuntary regulation: they work by prohibiting specific agricultural or extractive activities. In contrast, payments for ecosystem services are incentive-based and generally voluntary: they provide compensation to willing land owners conditional on maintaining a defined land use or fulfilling specific management activities. In the Mexican case, there is considerable overlap in the types of places where these two policies have been applied (15% of localities with some protection have both types), but there are also differences in their geographic distribution due to history and political feasibility. In order to separate the effects of instrument type from location, we estimate marginal impacts across comparable levels of deforestation risk. We also test for possible complementarity of the two policies where they do overlap and consider their relative cost-effectiveness using data on production revenues to proxy for the opportunity cost of land.

Overall, we find that both PAs and PES were environmentally effective across this time period, with full protection resulting in an estimated 20-25% reduction in the expected loss of forest cover. With respect to development goals, the data show that PES led to small but statistically significant poverty alleviation (10-12% increase in locality poverty alleviation index). Parks as a group had no effect on locality poverty trends although strict protected areas did show significantly less poverty alleviation than counterfactual localities. Importantly, none of the park types or PES did harm in an absolute sense: localities with these policies showed decreases in average values of all basic poverty indicators.

Comparing the park types—biosphere reserves, strict PAs and mixed-used PAs—with each other and with PES yields two lessons. First, there are apparent tradeoffs across the individual policies with respect to achieving different social goals. Biosphere reserves generated the most avoided deforestation while PES resulted in the most poverty alleviation. These tradeoffs match theoretical expectations that well-enforced protected areas can offer more complete conservation while PES is more likely than PAs to alleviate poverty. In addition, our analysis of the distribution of avoided deforestation from land of differing agricultural production values indicates lower opportunity costs of avoided deforestation for PES, biosphere reserves and mixed-use PAs than for the strict PAs. However, all policies generated avoided deforestation from a mix of low and high cost land, so there is no clear policy winner with respect to cost-effectiveness.

Second, we find that PES and the biosphere reserves were more successful than other policy types in simultaneously achieving environmental conservation and not harming livelihoods. Both PES and the biosphere reserves differ from other PA designations by allowing a flexible but still well-defined and enforced approach to conservation—PES through voluntary uptake and the biosphere reserves through a combination of strictly protected core areas and buffer zones allowing mixed use. In addition, both PES and the biosphere reserves likely received more financial support during this period than the other park types. Overall, this suggests that both direct and incentive-based mechanisms can succeed, but are more likely to balance environmental and poverty alleviation goals when they combine flexible zoning with sustained funding and efforts to support local livelihoods.

1. PARKS VS PES: BACKGROUND AND EXISTING EVIDENCE

Expected impacts of Parks and PES

Land-use change, including forest loss, is the second largest source of anthropogenic greenhouse gas emissions (IPCC 2013) and reduces biodiversity, water quality and amenity benefits worldwide. Protected areas currently cover 12.7% of global land area (Bertzky et al. 2012), with much of the increase in the past three decades coming from new parks in developing or middle income countries. Backlash against protected areas due to possible conflict with local livelihoods (e.g. Adams et al. 2004, West et al. 2006, Brockington et al. 2006) has led conservationists and governments to explore alternative policies, including more flexibly zoned protected area types and payments for ecosystem services (Ferraro 2001, 2002, Wunder 2007, Jack et al. 2008, Wunder et al. 2008, Pechacek et al. 2013). In addition to being more politically feasible, payments for ecosystem services could potentially be more cost-effective. A central theoretical and empirical finding in the literature on pollution control is that market-based mechanisms are likely to meet abatement goals in a more cost-effective way than command and control regulations by solving the regulator's information problem and allocating more of the control burden to low-cost providers (e.g. Tietenberg 1990, Stavins 2003).

This result does not necessarily generalize to the land conservation context, however. The relative cost-effectiveness of direct versus incentive-based land conservation depends on which types of land are protected under each policy as well as on the costs and success of enforcement. PES programs are likely to be more cost-effective than PAs only if PES can successfully enroll and protect land that has equivalent environmental benefits but costs less to protect. This in turn depends on the relationship between the risk of deforestation and opportunity cost. For example, suppose we consider a flat-payment PES program and define the payment offered to be equal to $\bar{p}$. The true opportunity cost for a landowner considering enrolling a hectare of land is given by c, which is observed privately by the landowner. Then only landowners with $c \leq \bar{p}$, including those with $c = 0$, will voluntarily enroll. If opportunity cost is positively correlated with the risk of deforestation, then PES is likely to attract lands that are low cost but also at low risk of deforestation. This potential adverse selection problem has been described many times in the literature and is formally modeled by Ferraro (2008).[1] To overcome this selection problem, policymakers can target payments based on observable characteristics likely to predict the risk of deforestation (described theoretically by e.g. Alix-Garcia et al. 2008, Ferraro 2008 and practiced by PES programs in Costa Rica and Mexico e.g. Robalino et al. 2008, Sims et al. 2014). Auctions for PES contracts are also theoretically possible but have been used mainly in developed country settings (Ajayi et al. 2012).

[1] Models of differences between voluntary and mandatory programs in a developed country context are published in Stranlund (1995), Segerson and Miceli (1998) and Wu and Babcock (1999). Lewis et al. (2011) evaluates different types of voluntary incentive policies for biodiversity conservation in comparison to a fully efficient solution.

As they do not suffer from the same self-selection challenge as PES and can cover large contiguous areas, protected areas may be sited on lands with either relatively high or low opportunity cost ($c \leq \bar{p}$ or $c \geq \bar{p}$), depending on the location of the PA. Prior literature has shown that, like PES, parks tend to be located on land with lower risk of deforestation due to political economy constraints (Andam et al. 2008, 2010, Joppa and Pfaff 2009). However, some have still been established in high risk areas, and risk frontiers often move with time. In summary, the actual avoided deforestation generated by PES or PAs in a specific context will depend upon whether the programs are able to enroll land at high risk of deforestation and whether they are effectively enforced. For a similar amount of avoided deforestation, the relative cost-effectiveness of PAs vs. PES will depend on the opportunity costs of those lands as well as the administrative costs of each policy.

Direct and incentive-based mechanisms are also likely to have different economic consequences for local populations, impacts that are particularly important in developing countries. PES are voluntary, so barring substantial informational asymmetries or surprise events related to the costs of participation, PES should result in either zero or positive poverty alleviation impacts (Pagiola et al. 2005, Jack et al. 2008, Alix-Garcia et al. 2015). In contrast, the impact of PAs on welfare depends on the extent to which they offset the opportunity costs of use restrictions with benefits such as tourism employment, increased local ecosystem service flows, or more sustainable use of valuable renewable resources (Dixon and Sherman 1990, Robalino 2007, Andam et al. 2010, Sims 2010). Both PAs and PES could also positively or negatively impact population trends, by restricting local development activities or by providing new sources of rural income.

Answering the call for "Conservation Evaluation 2.0"

Given the theoretical ambiguity with respect to avoided deforestation, cost-effectiveness, or social impacts of PAs relative to incentive payments, empirical comparisons are needed. Growing literatures separately evaluate each type of policy (see reviews by Pattanayak et al.

2010, Miteva et al. 2012, Alix-Garcia and Wolff 2014 and further detail below), but it is hard to draw comparative conclusions from this literature because different analyses use different methods or evaluate policies and outcomes across different time periods.

Our paper therefore makes two main contributions to the literature. First, we provide the first simultaneous national-scale evaluation of the environmental and socioeconomic impacts of direct versus incentive-based land conservation. Second, we contribute to the separate evaluation of both PAs and PES in Mexico by using the most recent information on land cover change and by producing new estimates of impacts for locality-level economic outcomes and population. In the spirit of Miteva et al.'s (2012) call for "Conservation Evaluation 2.0" and Vincent's (2015) appeal for more focus on the economics of conservation policies, we seek to use rigorous methods, track both social and environmental impacts, assess heterogeneous effects, and compare cost-effectiveness to the extent possible given data limitations.

The first round of conservation evaluation estimated moderate avoided deforestation due to protected areas in Costa Rica (Pfaff et al. 2009, Andam et al. 2009), Thailand (Sims 2010), Brazil (Nolte et al. 2013, Pfaff et al. 2014a), Guatemala (Blackman et al. 2014), Indonesia (Gaveau et al. 2009, Schwarze and Juhrbandt 2010, Miteva et al. 2015) and Russia (Jones and Lewis 2015). Most studies find that avoided deforestation increases with the risk of deforestation (Pfaff and Robalino 2012). Yet as our results corroborate, avoided deforestation is not necessarily greater for strict protected areas versus those with intermediate levels of restrictions (Ferraro et al. 2013, Pfaff et al. 2014b, Miranda et al. 2014). Previous work in Mexico finds mixed evidence of park effectiveness in preventing land-use change, with results differing across regions and park types. Existing studies analyze impacts on cover change in the 1980's (Deininger and Minten 1999, 2002), between 1993-2000 (Duran-Medina et al 2005, Figueroa and Sánchez-Cordero 2008, Bezaury-Creel 2009, Blackman, Pfaff and Robalino 2015) and 2000-2005 (Pfaff et al. 2014b) and on biodiversity conservation (Ochoa-Ochoa et al. 2009). Our work adds to this literature by considering all park types and new high resolution forest cover data from 2000-2012.

With respect to the impact of PAs on local livelihoods, earlier work shows mixed results, with improvements due to PAs found in Costa Rica, Thailand, Uganda, and Indonesia (Andam et al. 2010, Sims 2010, Naughton-Treves et al. 2011, Gurney et al. 2014), no negative impacts in Cambodia (Clements et al. 2014) and mixed results in Zambia (Bandopadhyay and Tembo 2010, Richardson et al. 2012), a different park in Uganda (Tumusiime and Sjaastad 2014), and Tanzania (Baird and Leslie 2013). Work on the mechanisms by which PAs could improve livelihoods is still at early stages but existing efforts suggest that tourism is the main driver of positive material impacts (Sims 2010, Ferraro and Hanauer 2014, Robalino and Villalobos-Fiatt 2015). In Mexico, preliminary work has estimated negative municipal-level economic impacts of protected areas prior to 2005 (Blackman et al. 2011). In contrast, our paper tests for locality-level impacts on poverty and population using changes from the most recent decade (2000's). We include population density growth as an outcome because of heated debates about the effects of parks on population trends (e.g. see Wittemyer et al. 2008 and response letter from Igoe and Brockington) and to test whether poverty alleviation impacts might be explained by migration.

Compared to the literature on protected areas, there are substantially fewer rigorous evaluations of payments for ecosystem services, particularly large-scale national programs (Pattanayak et al. 2010, Alix-Garcia and Wolff 2014). Recent evaluations in Costa Rica (Arriagada et al. 2012, Robalino and Pfaff 2013), Mexico (Alix-Garcia et al. 2012, Alix-Garcia et al. 2015) and Ecuador (Jones and Lewis 2015) estimate statistically significant but modest impacts on environmental outcomes. They find small but positive impacts on poverty reduction in Mexico (Alix-Garcia et al. 2015) and small or neutral impacts on livelihoods in Costa Rica (Robalino et al. 2014, Arriagada et al. 2015). Both sets of studies generally show that avoided deforestation from PES also increases with the risk of deforestation. With respect to PES's economic impacts, prior work on China's payments for reforestation programs traces possible increases in welfare through additional assets and reallocation of labor to off-farm opportunities (Uchida et al. 2007, 2009). A recent review of cases from Guatemala, Cambodia and Tanzania

(Ingram et al. 2014) also provides positive evidence for PES as a way to conserve biodiversity and support local livelihoods.

Despite these growing literatures on the separate effects of PA and PES, direct retrospective comparison of the two conservation types is extremely limited. Work in progress by Baylis et al. (2012) compares protected areas and PES for areas in and near the Monarca Reserve in Mexico. Clements et al. (2015) evaluates the impacts of PES on forests and outcomes for four villages within two PAs in Cambodia. Papers by Siikamaki and Layton (2007) and Busch and Grantham (2013) discuss the differential targeting of direct versus incentive-based systems in Finland and Indonesia and simulate potential policy improvements but do not evaluate ex-post impacts. In Costa Rica, a recently published analysis by Robalino et al. (2015) does examine interactions between national parks and PES with respect to forest cover outcomes. We are not aware of any national-level empirical analyses that retrospectively compare the two policies on both conservation and development dimensions, or that consider cost-effectiveness.

2. EMPIRICAL STRATEGY AND MEXICO'S POLICIES

Our empirical strategy relies on comparisons of changes in outcomes in the 2000's between localities with different shares of land protected. In this section we outline our estimating equation and then explain in detail the two policies of interest, our unit of analysis, outcomes data, treatment variables and pre-trend data.

We face the standard empirical challenge of conservation evaluation, which is that parks and PES were not randomly sited. Given the absence of suitable instruments to separately predict park presence and PES[2], we model changes over time in the outcomes in the past decade as a function of the share of land treated during this period, controlling for state fixed effects, pre-

[2] We created potential instruments for park location based on biodiversity indicators including the ranges of charismatic mammal species, endemic birds and proximity to bird survey sites but these instruments did not have sufficient power in the first stage. We also experimented with historically-based instruments intended to exploit the relationship between the Agrarian Reform and the goals of the Cardenas administration for parks, again without sufficient power.

trends and geographic characteristics that determined selection criteria. This identifies impacts based on a comparison of changes in outcomes between localities with similar baseline characteristics and pre-trends but with greater versus less share of area with protected status during the past decade. We check for potential differences in pre-trends and run multiple robustness checks, including estimating bounds under the assumption of remaining omitted variables.

Our main estimating equation is:

$$(1)\ IHS(\Delta Y_{imj,2000s}) = \beta_0 + \beta_1 SharePES_{imj,2000s} + \beta_2 SharePA_{imj,2000s} + \theta' \boldsymbol{IHS}(\boldsymbol{\Delta Y}_{imj,1990s}) + \delta' \boldsymbol{X}_{imj,2000} + \alpha_j + \varepsilon_{imj}$$

where $IHS(\Delta Y_{imj,2000s})$ is the inverse hyperbolic sine of the change in outcome for locality *i* in municipality *m*, state *j* during the past decade (the 2000s). $SharePES_{imj,2000s}$ and $SharePA_{imj,2000s}$ are the share of each locality receiving PES or PAs by 2010 and $\boldsymbol{IHS}(\boldsymbol{\Delta Y}_{imj,1990s})$ is a vector of transformed locality-level controls values of the pre-trend for each outcome (details and data sources described below). $\boldsymbol{X}_{imj,2000}$ is a vector of other locality-level covariates related to selection into PAs or PES and likely to influence outcomes (described below). Finally, α_j are state fixed effects, so impacts are identified from changes over time within the states. To ensure comparability, we use the same specification for all three outcomes. The outcome and pre-trend variables are transformed using the inverse hyperbolic sine function to mitigate the influence of outliers (Burbidge et al. 1988). This transformation is defined at zero and for negative numbers and can otherwise be interpreted similarly to a log-linear specification: the coefficients are approximately the expected percent changes in the untransformed dependent variables for a change in share protected from zero to 1 (from no to full protection).[3] Standard errors are clustered at the municipality level to account for spatial correlation.

[3] Note that this is an approximation; marginal effects for the untransformed variables can be calculated by: $\frac{\partial \Delta Y}{\partial x} = \beta * \sqrt{1 + \Delta Y^2}$. For PES, the actual mean share conditional on treatment is approximately 0.3 while for PAs it is approximately 0.7.

Protected Areas

Mexico's protected areas system includes federal, state, and municipally designated protected areas, as well as locally designated and managed "Certified Areas." We analyze all available mapped protected areas created before 2010 (Table 1), which together cover 12% of terrestrial territory. As shown in Figure 1 and Table 1, protected areas vary substantially in terms of their stringency, locations and coverage area. Approximately 80% of protected areas are federally managed (WDI 2014, Bezaury-Creel et al. 2009) and in contrast to many other countries, only a small percentage is strictly protected. Rather, most protected land is in biosphere reserves or mixed-use areas including flora and fauna protection areas, natural resource protection areas, and certified areas (Table 1). For the purposes of our analysis, we group protected area types into four major categories based on a review of the specific rules for each type and an assessment of the correspondence between these rules and the IUCN categories. (Ley General del Equilibrio Ecológico y Protección al Ambiente, 1988, de la Maza Elvira 1999, Bezaury-Creel 2009, CONABIO 2012, Chavez 2012, CONANP 2012, 2014)

Many of Mexico's protected areas have existed in some form for decades (the earliest in 1876), although coordinated management and funding increased dramatically starting in the late 1990's (CONANP 2014). Early parks tended to be close to population centers and focused on watershed conservation and recreational or educational opportunities for urban residents (Wakild 2011). Approximately three-quarters of the areas now categorized as national parks were established during the Cardenas administration between 1934-1940 (Simonian 1995). A second major push for protection in the 1970's and 80's tripled the area under protection, mainly through the establishment of biosphere reserves (calculations based on data in Bezaury-Creel 2009). Unlike the earlier national parks, biosphere reserves were targeted to protect relatively pristine landscapes representing unique and biodiverse ecosystems (Simonian 1995). Biosphere reserves were also intended to be more participatory and more focused on community-level sustainable development (UNESCO 2015), although their management has still been relatively top-down in the Mexican context (Ruiz-Mallén et al. 2015). Biosphere reserves are flexibly zoned, as they combine core areas with strict protection and strong enforcement with buffer zones that allow sustainable use.

Despite the increase in area, Mexican PAs in the 1980's and early 1990's suffered from lack of resources and a divided management regime: at the beginning of the 1980s, five different ministries were simultaneously responsible for the federal protected areas. This situation improved in the 1990's with the establishment of the National Commission for the Knowledge and Use of Biodiversity (CONABIO), and in 2000 with the establishment of the National Commission of Protected Areas (CONANP). Between 1995 and 2000, federal investment in the protected areas system increased substantially from 10.9 million pesos to 142.7 million pesos per year (Bezaury-Creel 2009, p 402-409) while the area protected rose to approximately 7% of land by 2000 (World Development Indicators 2014). Spending on protected areas continued to increase in the most recent decade, indicating that enforcement and support may have been different in the 2000's than in the past (Bezaury-Creel 2009). Total spending in 2008 was 1,100 million pesos per year for federal protected areas and recent figures give a budget for CONANP in 2015 of 1,185 million pesos.[4]

Factors influencing the siting for the protected areas depend on the type of park and the timing of establishment. As mentioned above, biosphere reserves tend to be targeted towards places with more intact ecosystems, while national parks were designed to preserve areas with watershed or recreational value. Systematic criteria for park establishment across time do not appear to exist, so our empirical strategy relies on controls for standard factors used in evaluations of protected areas globally or in Mexico, including ecosystem type, distance from city, slope, elevation, distance from roads, poverty level and baseline population density.

As in many countries around the world, the stated goals and official rules of protected areas in Mexico often conflict with local use on the ground. Biosphere reserves, for instance,

[4] Figures from: "Presupuesto de egresos de la Federación 2015; Ramo 16 Medio Ambiente y Recursos Naturales," January 2015.

have the legal goal of preserving "ecosystems undisturbed by human activity" (Ley General del Equilibrio Ecológico y Protección al Ambiente, 1988) but in reality these areas overlap with land that is used and legally owned by communal and private property holders. Of the localities with more than 90% of land in a protected area, on average 41% of land is under common property ownership. By constitutional right, Mexico's government can limit the use of resources on any private property, provided it is "in the public interest." In practice, however, it may be politically difficult to enforce restrictions, particularly if they limit subsistence use.

Payments for Ecosystem Services

In part due to the conflicts around protected areas, federal conservation efforts in Mexico shifted in the 2000's towards more incentive-based programs, including major financial support for payments for ecosystem services and smaller programs to support commercial plantations, reforestation, or community forestry.[5] The federal payments for ecosystem services program began in 2003 with dual goals of preventing land cover change and maintaining rural livelihoods. It is managed by the Mexican National Forestry Commission (CONAFOR), which was established in 2001. The 2015 federal budget numbers show that the total budget for CONAFOR (7,743 million pesos) is larger than that for CONANP, but much of this budget supports the cash transfer payments.

The federal PES program offers annual payments under five-year contracts to landowners in exchange for maintaining existing forest or other vegetative land cover on enrolled parcels (Muñoz-Piña et al. 2008, Alix-Garcia, Shapiro and Sims 2012, Alix-Garcia, Sims and Yanez-Pagans 2015). We include payments from the three major PES "modalities": hydrological services, biodiversity conservation, and carbon capture and storage. In the period from 2000-2010, PES payments reached a total of more than three million hectares of land.

[5] Ley General de Desarrollo Forestal Sustentable 2003, Reglamento de la Ley General Desarrollo Forestal Sustentable 2005.

Applications to the program are voluntary and landowners enrolled over the period we analyze can freely choose how to spend the funds received after meeting land management goals. The structure of payments has changed somewhat over time (see Alix-Garcia, Sims and Yanez-Pagans 2015) with payments ranging from approximately 20-40 USD/ ha depending on land use type. Household surveys of a representative sample of the 2008 cohort of participants in the hydrological services program showed that annual per capita payments for households in common properties were approximately $130 USD, which was greater than 1 month of minimum wage work (Alix-Garcia, Sims and Yanez-Pagans 2015).

The federal PES program enrolls both private and communal landowners, including those living in protected areas, and the rules of selection have evolved over time to prioritize land of ecological and social priority. These rules and the distribution of lands over time in the largest modality (hydrological services) are described in detail in Sims et al. (2014). In brief, in the early years of the program, eligible land was required to be upstream from urban centers or inside priority mountain areas, to be above overexploited aquifers, and to have > 80% forest cover. Within eligible applicants, priority was given by greater baseline forest cover. In 2006, the eligible zones were expanded and eligible parcels were required to have only 50% forest cover. Priority was given on the basis of a points system which combined predicted deforestation risk, water availability, location in protected areas or priority mountain areas and location in a high poverty or majority indigenous municipality. Our regression covariates, which are listed below and in Table 2, were collected in order to proxy for the major factors influencing this point system.

Unit of analysis, outcomes, and pre-trends

Our unit of analysis is the locality, the smallest administrative unit in Mexico. As the detailed boundaries of localities are not mapped, we use the point locations (from INEGI) to create Thiessen polygons around each locality in order to assign locality areas. As illustrated in Figure 2, Thiessen polygons assign land to each locality based on the closest point and avoid the problem of double-counting (methodology follows Alix-Garcia et al. 2013). We use the 1995 locality polygons as the unit of analysis to maintain a constant area over time and calculate area-weighted means from similarly constructed Thiessen polygons based on point data from other years. This results in N= 105,647 localities for the entire country; for this analysis, we restrict the sample to the N= 59,535 localities which had some forest cover in 2000.[6] An advantage of measuring outcomes at the locality level is that any local spillovers (positive or negative) from PES or PAs will already be incorporated into locality changes. This mitigates concerns that estimates could be biased by localized spillover effects.[7]

We study three possible outcomes from the most recent decade: the net change in forest cover from 2000-2012, changes in the locality level poverty alleviation index from 2000-2010, and changes in population density from 2000-2010. The outcomes are measured in slightly different years due to data availability. Data on forest cover in 2000 and forest cover change from 2000-2012 comes from Hansen et al. (2013), which is the only available data source providing wall-to-wall analysis of forest change during this period.[8] The dataset is based on Landsat satellite images (30 m resolution). We sum Hansen et al.'s gain and loss areas in order to create net forest cover change for the period; thus our forest outcome variable is positive if a locality gained forest, negative if it lost forest. We provide several alternate specifications of this outcome measure as robustness checks in the Appendix. We also note that there are inherent

[6] Our main estimation sample keeps localities with greater than 5% and at least one 1 ha in area of forest cover at baseline. Fewer than 10 observations are dropped for missing data on slope or elevation and one is dropped because of missing data from the 93-00 forest cover change. We also exclude 11 localities which had more than 5% of land area in municipal parks, as these are very small areas of land in highly urbanized environments and we do not have a sensible counterfactual for municipal parks.

[7] Prior work (Alix-Garcia, Shapiro and Sims 2012) indicates both substitution and output price slippage in deforestation due to an early cohort of PES, but subsequent analysis of household impacts did not show substantial substitution slippage (Alix-Garcia et al. 2015). Other analyses of protected area impacts, including in the prior decade in Mexico, have generally found deforestation spillovers to be small and in some cases positive (Andam et al. 2010, Sims et al. 2010, Baylis et al. 2013, Blackman et al. 2015). Detailed comparison of slippage for Parks vs. PES goes beyond the scope of this paper but is an important avenue for future research.

[8] Hansen et al.'s data also provides estimated tree cover in 2000 for each pixel. Given the different types of forest in different regions of Mexico, we use the following cutoffs in order to calculate percentages of area in tree cover in 2000: region 1(North) > 40%; region 2 (Central) > 60%; regions 3 and 4 (South) > 70%. We also check robustness of results to a simple > 50% or > 30% cutoff; results available on request.

limitations with the Hansen et al. data. The data may understate true loss of natural forest, because it may classify plantations and agroforestry crops as forested areas, even though these areas provide fewer ecosystem services (Tropek et al. 2014). It may also understate selective logging, an important source of forest degradation (Burivalova et al. 2015) or very small areas of deforestation. Yet, because Hansen's data counts forest loss due to timber harvest in addition to forest loss due to conversion to agriculture, it could overstate apparent deforestation in sustainably managed areas. As we are measuring outcomes at the locality level, truly sustainable forest rotations in our data should on average net out to zero forest cover change: harvests in some areas should be matched by regrowth in nearby areas, with the possible exception of timber harvests near the end of the decade. Given these different factors, it is most likely that the Hansen et al. data understates true forest loss, so our estimates of impacts are likely conservative.

Poverty data comes from CONAPO and is based on a weighted average of indicators including rates of literacy, primary schooling, availability of potable water, sanitation and electricity and housing characteristics (weights determined by principal components analysis done by CONAPO[9]). We re-normalize each year's index values to have mean zero and standard deviation one. We then multiply by negative one so that higher values of this index, which we refer to as the "poverty alleviation" index, represent <u>less</u> poverty. We also analyze impacts on each of the individual indicators that are common across years. To do so, we used data on all individual components of the index that are common between 2000 and 2010 and calculated the area-weighted means to account for changes in locality boundaries. Each indicator is normalized to have mean zero and standard deviation one and we calculate the change in normalized values from 2000 to 2010. Population data is also from CONAPO and is converted into density measures (hundreds of people per square km). Finally, our analysis also includes measures of the pre-trends in each variable ($\boldsymbol{\Delta Y}_{imj,1990s}$). The data on forest cover comes originally from UNAM (Velázquez et al. 2002, Mas et al. 2004) and measures the change in forest cover from 1993-

[9] This methodology is documented in "Anexo C: Metodología de estimación del índice de marginación por localidad", http://www.conapo.gob.mx/work/models/CONAPO/indices_margina/2010/anexoc/AnexoC.pdf

2000. The change in the poverty alleviation index is based on area weighted means of the locality marginality index from 1990 and 2000, and the change in population density is based on area weighted means from 1995 and 2000.[10]

Treatment variables and selection covariates

Our measure of PES or PA "treatment" is the share of each locality (ranging from zero to one) that was enrolled in PES between 2003-2010 or protected in a park established by 2010. As some parks were still being established in the 2000's and PES are for five year contracts, that means in both cases our policy variables sometimes reflect partial treatment. We use the share protected by 2010 in both cases as it is the most conservative measure of treatment.

Using the share protected at any point before 2010 conceptualizes PA status as an ongoing treatment, not a one-time event. We think this matches the reality of continuing pressure on natural resources and the need to continuously monitor and enforce protected area regulations. The large increase in investment in parks in the past decade (described above) is a second reason to use the total cumulative share of parks in each locality as our key measurement of this policy. Prior to the budget increase in the late 1990s, parks likely did not have sufficient resources to effectively operate, while the recent budget increases have hopefully increased their impact.

Thus, we do not seek to capture the total lifetime impacts of protected areas, but rather to estimate the relative impacts of PAs during the 2000s to PES in the same time frame.[11] We

[10] Population data at the locality level was only available from 1995, not 1990.

[11] We do not use a panel specification across the two periods of data. While a panel analysis would be helpful to control for time-invariant unobservable characteristics, it requires substantial variation across time within localities (which is not the case for the PAs) and truly comparable outcomes data. The forest data is measured in a scale of 1:60,000 for the most recent decade and 1:250,000 for the pre-trend and was constructed using different methodologies. The index of marginality was in its first years of being developed in 1990 and includes some different indicators. In a locality fixed effects regression, these differences may create measurement error that can both introduce bias and affect precision (Hyslop and Imbens 2001). Furthermore, a fixed effects analysis would only identify impacts from locality-level changes in parks, but we are interested in the effects of all parks during the most recent decade. The changes in policy and funding in the late 1990s suggest that the effect of parks is likely to be substantially different in more recent years, even for existing parks. We therefore focus this analysis only on the impact of all PAs (and PES) in the most recent decade; but note that longer term panel analysis is an important priority for future work as data sources and policy become more directly comparable.

recognize that protected areas established before 2000 may have had prior effects on forest cover or livelihoods that we do not measure (although work by Blackman, Pfaff and Robalino 2015 suggests that federal parks did not actually have substantial impacts in the 1990's). In this analysis, we explicitly remove those earlier impacts by differencing the outcomes from 2010-2000 and by including controls for pre-trends and 2000 levels of the outcomes. This produces estimates of relative program impacts for PAs and PES only in the decade of the 2000's. However, given potential concerns about this way of measuring the treatment variable, we include robustness checks breaking parks into those established before and after 2000 and scaling the PES treatment by number of years treated (online Appendix). We also note that a prior analysis using annual variation in PES contracts (Alix-Garcia et al. 2015) finds similar treatment effects from PES when using year to year variation.

Finally, all specifications include a set of covariates ($\boldsymbol{X}_{imj,2000}$) measuring geographic characteristics and baseline characteristics. These covariates were chosen on the basis of the histories of PAs and PES described above and are: average slope and elevation (5 categories of each to allow for non-linear effects), the log of distance to nearest locality with population greater than 5000, the locality anti-poverty index in 2000, municipality anti-poverty index in 2000, log population density in 2000, log distance to nearest road, log distance to nearest urban area, log percent forest cover in 2000, average availability of water, overexploited watershed status, log locality area, share in major ecoregions, whether municipality is majority indigenous and percent of the locality under common property ownership. We include robustness checks removing controls that might be endogenous to the protected area treatment before 2000 (see Appendix).

Risk of deforestation index

PAs and PES are often sited in similar areas, but given their different histories, may also cover different types of land. In order to assess average differences and to attempt to isolate the impacts of conservation mechanism type, we create an index measuring risk of deforestation. To do so, we regress changes in forest cover within all non-treated localities (no PES or PAs) on all covariates except state fixed effects and use the coefficients to predict expected cover change for all localities. This predicted cover change is then normalized to have mean zero and standard deviation one and multiplied by negative one. This is referred to throughout as our "risk of deforestation" index; the values are interpreted as standard deviations away from the mean; higher values indicate more risk of deforestation.[12]

3. RESULTS

Summary statistics

Avoided deforestation impacts of conservation policies will depend on management and enforcement but also on whether protected lands are at high risk of deforestation. Table 2 shows the means of treatment variables, outcomes, and covariates for all localities, localities with a greater than 5% share in PES, and localities with greater than 5% share in protected areas. We also show summary statistics for localities with strict protection (categories 1-4 in Table 1), biosphere reserves (category 6), and mixed use (categories 7-9). Figure 3 shows the density distributions of our aggregate predicted risk of deforestation index for these categories.

As shown in Table 2, the aggregate predicted risk of deforestation is higher for PES than all types of PAs together, but there is considerable overlap in the distribution of risk. Averages are also mixed for individual characteristics that usually predict risk of deforestation. PAs on average contain land that has less steep slope, is closer to densely populated areas, and is slightly closer to roads—characteristics which are usually associated with higher risk of deforestation. Areas with PES had more forest cover to start, a larger fraction of land in high poverty municipalities and localities and a larger fraction of land in common properties. These

[12] The Mexican Institute of Ecology and Climate Change (INECC) has also created a risk of deforestation measure using similar characteristics that was used in targeting the PES program. "Index of Economic Pressure to Deforest / Risk of Deforestation" version 1. Methodology at http://www.inecc.gob.mx/irdef-eng.) Unfortunately, this measure relies on older cover change data and is missing data in multiple places, which is why we create and use our own index.

characteristics have a less clear theoretical relationship with deforestation but were associated with higher risk of deforestation in this period. The regional distribution of PAs and PES is also different, with more PAs in the central region (lower risk) and more PES in the southwest (higher risk).

Table 2 and Figure 3 also reveal heterogeneity in characteristics according to park type. Strict protected areas tend to be in higher population density areas, to have less baseline poverty, and to be more frequently located in the central region and in lower risk areas. The biosphere reserves and mixed-use protected areas are more similar to the PES areas and follow more of the classic "high and far" pattern (Joppa and Pfaff 2009) than the strict protected areas. This matches the mandate of biosphere reserves to preserve intact ecosystems, which are more likely to be located in relatively remote areas. Yet despite being in more remote areas, the biosphere reserves and mixed-use areas were more likely to be at high risk of deforestation in the 2000's than the strict protected areas (Table 2 and Figure 3). This is due to the fact that during this period, substantial deforestation occurred in these more distant locations, possibly because of the mobility constraints resulting from non-transferable common property rights in Mexico. In general, the summary statistics confirm that while there is substantial similarity in the types of land where PES and PAs are sited, there are enough differences to also motivate comparisons across the range of deforestation risk.

Avoided deforestation

Table 3 presents the results of our main specifications estimating the impacts of PES and PAs during the 2000's. In Table 3 we include both naïve comparisons and our preferred specifications. The preferred specifications correspond to equation 1 and are shown in bold text and graphed in Figure 4. The positive and significant coefficients for share PES and share PA in Table 3, Panel A, column 3 indicate that both policy types prevented forest loss in the 2000's (i.e. had a positive effect on net forest cover change). However, consistent with previous literature showing the importance of controlling for confounding differences when estimating impacts, there is a substantial difference between naïve estimates with no controls (Table 3, Panel A, column 1) and estimates that include controls (Table 3, columns 2 and 3). Without any controls, full PA protection is associated with a greater than 50% increase in net forest cover change between 2000-2012. Yet when controlling for other characteristics of deforestation risk and state fixed effects, the estimated impact of PAs drops to approximately 24% (column 3).

The naïve estimate for localities with PES (column 1) suggests that PES were associated with small and not statistically different from zero avoided deforestation relative to all other localities. This reflects the fact that PES were targeted to higher deforestation risk areas, as discussed above. However, when controls for characteristics affecting selection and deforestation risk are added, the estimates of PES impact are statistically significant and similar to those of PAs on average – the coefficients indicate an approximately 24-25% increase in expected net forest cover change. The coefficients on PES and PAs are not statistically different from each other (test for equality of coefficients gives p = .85).

To put these results in context, the mean net change in forest cover across all localities with no protection or PES during this period was -2.16 percent of land area. Therefore a positive 20-25% impact of PAs and PES translates to about 0.43-0.54 percentage points less loss of forest cover. While as a percentage of area, these impacts appear small, the large size of Mexico means that both protected areas and PES contributed to meaningful increases in forest cover compared to the counterfactual trend.[13] The total land area in the localities analyzed with at least 5%

[13] Alix-Garcia, Sims and Yanez-Pagans (2015) use annual variation in enrollments and NDVI to assess forest cover and find that the payments for hydrological services program reduced the downward trend in forest cover by 40-51%. The differences between that estimate and this come from three sources: the unit of analysis, the outcome variable, and the time frame. Here the unit of analysis is a locality and the treatment is the share enrolled; in previous work we used points as a unit of analysis, and classified them as enrolled or unrolled in each year. Previous estimates therefore did not include localized leakage. The current paper includes unenrolled areas of forest within the locality and thus gives a net impact of the program within the locality area. Our previous outcome measure was NDVI, or "greenness", which does not translate directly into forest area. However, estimation of the program's impact using the outcome from this paper and a similar methodology to the other paper shows similar results (appendix to Alix-Garcia, Sims and Yanez-Pagans 2015). Finally, here we assess the impact of PES over the entire 2000-2010 period, even though the program was not put into place until 2003. This may result in a smaller estimate compared to the annual impacts. The annual variation strategy works well for PES but does not make sense for the PAs because there are not enough new parks established during this period and because the timing of actual enforcement of and resources devoted to the PAs does not necessarily match well with the official dates of establishment.

protection or payments is 23,748,880 hectares, so this translates to approximately 102,100-128,200 hectares of avoided deforestation. Given that there are similar effectiveness estimates for PES and PAs, the much larger share of land area covered by the protected areas (13.4% vs. 3.14% within the localities we analyze) means they are responsible for the majority of this avoided deforestation to date. Yet the popularity of the PES program suggests that it could probably be scaled up to cover more land area.

Panel B of Table 3 breaks protected areas into strict protected areas, biosphere reserves, and mixed-use areas. We find that the biosphere reserves are most effective on average in preventing forest loss (coefficient = 0.34 or a greater than 30% change in expected trend) while mixed-use areas also show significant avoided deforestation (coefficient = 0.16), and strict protected areas do not yield statistically significant results, although the point estimate is positive. These results are likely to be partly explained by the higher risk profiles of mixed use areas and biosphere reserves compared to strict protected areas (Figure 3), but may also be due to differences in park management type, an issue which we will explore below.

Poverty and population: do no harm?

Table 3 also shows impact estimates of PAs and PES on poverty alleviation and population growth. We find that PES generated significant but small increases in locality poverty alleviation while PAs have on average not significantly affected poverty trends. As shown in Panel A of Table 3 (column 6), full protection under a PES scheme would lead to a greater than 10% (coefficient of 0.1169) increase in the change in the poverty alleviation index while PAs were overall neutral (coefficient of -0.027, not significantly different from zero). When we examine impacts by park type (Panel B), there is a significant decrease in poverty alleviation for localities with a greater share in strict protected areas (-0.106), a positive but not significant increase in poverty alleviation for localities with greater share in biosphere reserves (0.043), and a negative but not statistically significant change in poverty alleviation for mixed-use areas (-0.039). Average estimates of both types of protection on population trends suggest that PES and use protected areas have led to decreases in population (-0.042 and -0.049) while strict protected areas and biosphere reserves have not. However, the significance of the population result on PES is not robust to several of the specification checks, and so should be interpreted with caution.

To better understand the poverty alleviation results, we also examine the changes in individual components of the poverty index. Panel A of Table 4 repeats our main regression analysis using the change in the poverty index (the opposite of the poverty alleviation index shown in Table 3) and changes in individual indicators as the dependent variables. For these individual components of poverty, all PES coefficients have negative signs, indicating possible reductions in poverty measures compared to counterfactual localities. PES significantly reduced illiteracy and the percent of people without access to electricity. The decreases in percent without primary school, dirt floor and refrigerator are also marginally significantly different from zero. In contrast, strict protected areas had positive coefficients on all poverty measures and significantly increased the percent of people without refrigerators, compared to the counterfactual. The biosphere reserves showed mixed results, possibly reducing the percent with dirt floor, but generally did not show significant differences. In the mixed-used PAs, there is a significant increase in the percent of the population that is illiterate and several other coefficients have positive signs, indicating increased poverty due to the mixed-use PAs.

While the above analysis gives impacts relative to the counterfactual, it is also important to consider how populations in and near protected areas fared in an absolute sense. Panel B of Table 4 shows summary statistics for the absolute changes in poverty indicators for each category of protection. The columns give the change in the mean values for the normalized poverty index and for each indicator from 2000 to 2010. The consistent negative signs on average changes indicate that communities with PES and all types of PAs were better off in an absolute sense in 2010 than in 2000. For example, localities with all categories of conservation saw the percentage of the population without primary schooling decrease by 14-16 percentage points (Table 4, Panel B, column 3). So although communities near strict protected areas and possibly in mixed use protected areas fared worse in a relative sense than the counterfactual (similar communities that were not protected), they were not worse off in an absolute sense according to any of these poverty indicators. In fact, all types of localities saw substantial progress in average absolute poverty measures during this decade.[14]

Management type or location?

In this section, we return to the question of whether impacts of different PA types and PES are due to location or to management type. To isolate the impact of management type, we would theoretically like to compare the marginal impacts of different management types for the same types of land (a point made by several others, e.g. Ferraro et al. 2013, Pfaff et al. 2014). To explore this empirically, we regress outcomes on the original covariates from Table 3 as well as the risk of deforestation and 3rd order polynomial interactions between each policy and risk. We then map out the estimated marginal effects of each policy across the predicted risk of deforestation, with the results shown in Figure 5. Heterogeneous marginal effects are shown for the 10th to 90th percentile range of risk for each policy with confidence intervals given by the dotted lines.

As expected, Figure 5a shows that all policies tend to have greater avoided deforestation as risk increases. This partly explains the greater magnitude of avoided deforestation impacts found in Table 3 for the biosphere reserves, as they were located in areas that experienced higher deforestation risk during this period. Yet the marginal effects by risk of deforestation in Figure 5a also indicate that the biosphere reserves were more effective than other types conditional on having the same deforestation risk (although the confidence intervals for the marginal effects overlap). With respect to heterogeneity in poverty alleviation impacts by deforestation risk, we find (Figure 5b) that both PES and the biosphere reserves appear more likely to alleviate poverty than the strict protected areas or mixed-use areas.

[14] Note that the unit of analysis is the locality, so it is possible that part of this result could be explained by selective migration away from these localities.

Putting together the results in Figure 5a and Figure 5b suggests tradeoffs between direct and incentive-based policies in high risk areas: biosphere reserves appear to be better at protecting forest cover while PES may alleviate more poverty. These apparent tradeoffs are consistent with the theoretical expectations outlined in Section 1: PAs give more complete coverage within localities by fully enrolling land, leading to greater avoided deforestation, but PES directly compensates landowners for opportunity cost, leading to greater poverty alleviation.[15]

Figure 5 also implies that within PAs, park type matters. It is difficult to sort out potential channels, but better funding and flexible zoning provide possible explanations. Biosphere reserves may have been able to attract more funding than other protected areas, due to their high profile international status. For instance, Bezaury-Creel (2009) shows budget increases for several biosphere reserves in the 2000's and notes that biosphere reserves were fourteen of the nineteen protected areas that received financing through a special Mexican fund to support conservation. Unfortunately, comprehensive data on the budgets for individual parks or park types is not available.

A second possible channel is that communities can benefit more directly from forest conservation within the biosphere reserves, for example by attracting tourists or selling sustainable forest products. To explore the tourism hypothesis, we obtained data on the revenue from entrance fees to each protected area from 2002-2010 to serve as a proxy for visitation rates to the parks.[16] Revenues are assigned to localities on the basis of area within the park. Table 5 shows means for revenues in two ways: in pesos per square kilometer of park area within that locality and as pesos per person within that locality. While far from a perfect measure of gains from tourism, this does give a sense for the potential visitation values across different park types. These figures suggest that the strict protected areas actually had the highest revenue, so tourism

[15] This is also illustrated in Appendix Figure 1: the biosphere reserves appear to dominate environmental impact across predicted opportunity costs while PES dominates poverty alleviation impact.

[16] We gratefully thank Paulo Quadri for this data. Park revenues may also contain some small amounts from construction authorizations or vehicle use, but these are relatively rare.

does not offer a clear explanation for the relative success of biosphere reserves. However, as previously noted in the literature (e.g. Sims 2010, Robalino and Villalobos 2015), the net benefits of tourism must also take into account the opportunity cost of protection. For example, tourist revenue is likely to be high close to cities but opportunity costs are also high in those areas. Biosphere reserves may have benefitted from intermediate levels of tourism combined with being located in lower opportunity cost locations (see additional discussion in the online appendix). Strict PAs clearly do protect land with higher predicted opportunity costs, as shown in Table 5 and discussed in the next section. It is also possible that due to the flexible zoning within the biosphere reserves, tourists could stay longer or spend more money on related activities inside the biosphere reserves but outside of the core zones, which would not be captured in the revenues data.

Cost effectiveness

Vincent's 2016 paper on the impact evaluation of forest conservation programs and Miteva et al.'s 2012 "Conservation 2.0" paper both call for more comparison of the economic benefits and costs of conservation instruments. Vincent writes, "A good first step toward this new version of impact evaluation would be to develop outcome measures that more directly relate to conservation benefits or costs." In this section, we take a first step towards comparing the relative cost-effectiveness of PAs vs. PES.

From a social perspective, a full calculation of cost-effectiveness should include both the direct costs of administering the program and the opportunity costs of forgone land use. This is not possible given the data available; instead we are able to give the budgetary expenditures for each program and how the opportunity cost of land in each type relates to avoided deforestation. First, using the budget numbers for CONANP (Section 2 of the paper) we calculate that the annual federal spending for PAs per hectare is approximately 50 pesos or 4-5 USD per hectare.[17]

[17] The exchange rate across this period was approximately 11-12 pesos / USD on average, although it has increased in more recent years.

This is an underestimate, however, since it does not include spending by national or international conservation NGOs. In comparison, the PES program offered annual payments of ~250-400 pesos per hectare (~20-40 USD) depending on land type (with additional administrative costs of up to 4% (10-16 pesos or ~1-2 USD). Thus PES was likely significantly more expensive to implement per hectare when considered from a pure budgetary perspective, but the majority of funds went to transfers, which are not true resource costs.

Most of the true resource costs of PES and PAs are likely to be due to the opportunity cost of foregone land use, which depend on the forgone profits from the highest value non-forest use. Data on production profits across the country is not available, so we create a proxy based on production values, which is the best available data. To create the locality predicted production revenues measure, we start with data on production revenues for principal crops and livestock at the municipal level from 2003, normalized by the non-forested area of the municipality (to give an approximate measure of revenues per hectare in agricultural or pastoral use).[18] We then regress these production revenues on municipal averages of the same covariates used to create the risk of deforestation variable and predict the locality-level production values using the locality values of those covariates. By using locality-level characteristics, we mitigate the problem that municipality production revenues are likely to overstate true opportunity cost because they give the value of land already in production rather than the potential production values for the extensive margin. We use data from 2003 because it is the earliest year in our period with fairly complete municipal data. Although agricultural prices fluctuate over time, we found that in a panel of production revenues from 2003, 2007, and 2010, the elasticity of present to past prices is 0.82. This suggests that price fluctuations, while large over time, do not differ substantially across space, so our measure is likely to capture relative differences in production

[18] The production revenues data is from: INEGI Sistema Estatal y Municipal de Base de Datos, http://sc.inegi.org.mx/sistemas/cobdem/.

potential across space. Finally, we emphasize that our proxy is a measure of predicted revenues, not true profits, because it does not take into account production costs.

The mean predicted locality production revenues for each policy are given in Table 5. They are highest overall for the strict PAs. PES and mixed-use PAs have slightly higher average values than the biosphere reserves, but all are of a similar order of magnitude. To understand how avoided deforestation impacts compare in terms of foregone revenues, we show the full distribution of predicted production revenues by impacts for each policy (Figure 6). This figure was generated by using the coefficients from the regressions with polynomial interactions with deforestation risk (Figure 5) to estimate the amount of avoided deforestation due to each policy in each locality. We then calculated the cumulative level of expected avoided deforestation for each level of predicted revenues, and normalized the x-axis by the total area of avoided deforestation generated by each policy to show the proportional distribution of foregone production revenues for each policy.[19] Figure 6 can thus be interpreted in the same way as a supply curve, but with the amounts as proportions of the total land enrolled in each policy and with actual avoided deforestation supplied based on land enrolled by 2010.

Figure 6 shows that all policies have generated avoided deforestation from a mix of low and high value lands, implying that the most cost-effective allocation would come from a combination of all policy types. The distribution of foregone production revenues is everywhere higher for strict protected areas, which makes sense given their locations closer to population centers and more frequently in the agriculturally productive center of the country. For PES, use PAs, and biosphere reserves, the distributions of forgone revenues are fairly similar. For all three policies, about 60% of the avoided deforestation comes from land of relatively low value: less than 800 pesos of revenue per hectare. Over this range, a slightly greater share of PES avoided deforestation comes from lower value land than either of the other two policies. Most of the remaining avoided deforestation for these three policies (about 35%) comes from land in

[19] We limit the y-axis at 4000 hectares because using the full range of production values distorts the figure significantly, and there is very little land enrolled with these high values.

localities with estimated production values under than 2000 pesos. For the biosphere reserves, a somewhat greater share comes from land in localities with higher predicted production revenues. This may be because PES payments were not high enough to generate complete coverage within high value localities, while mixed-use PAs did not receive enough funding for effective enforcement in high value areas.

Overall, Figure 6 shows that avoided deforestation from strictly protected areas in this case is relatively more costly, but there is no obvious cost winner between PES, the biosphere reserves and the mixed-used PAs. Although we should interpret the results with caution because production revenues are only a proxy for opportunity cost, Figure 6 represents an important finding: PES is not necessarily more cost-effective simply because it is an incentive-based rather than command and control conservation mechanism.

Complementarity between PES and PAs?

An additional policy-relevant question is whether there is possible complementarity between direct and incentive-based policy in this case. There is certainly significant overlap: 15% of localities with some form of protection have both parks and PES. Table 6 introduces an interaction between the Share PES and the Share PA to our preferred specification (columns 1-3). Complementary effects would result in positive coefficients on the interaction terms. However, the interaction terms are actually negative for forest cover and poverty alleviation and not significantly positive for population. For forest cover, the fact that the coefficient is close in magnitude to the main effect for PES indicates that PES would provide no additional avoided deforestation where the areas receiving PES are already fully inside PAs.[20] This makes sense if existing protection is already effective. Interestingly, the lack of environmental complementarity between PAs and PES echoes the results of Robalino et al. (2015) for Costa Rica.

[20] Consider for instance a locality that that is fully inside a protected area. The effect of a change in PES (from 0-1, or no share to full share) is the coefficient on Share PES plus the coefficient on Share PES x Share PA, which is 0.0216 and is not significantly different from zero.

However, we do find possible complementarity when we look more specifically at the borders of protected areas. Complementarity might be greater for border localities because they are only partly protected and so at higher risk of deforestation. The last three columns in Table 6 include a dummy variable for whether a locality is at the border of a protected area (less than 50% share in a PA) and an interaction term between the border dummy, share PA and share PES. This allows the potential complementarity effect of PAs and PES to vary depending on whether localities are fully inside protected areas or on the border. The combined marginal effects show possible positive complementarity near borders for forest cover (coefficient is 0.47 but not significant) and significant positive complementarity with respect to poverty alleviation (coefficient is 0.556). For example, for a locality with a 0.25 share in a protected area, the estimated marginal effect of PES on forest cover change is approximately 34% and is significantly different from zero ($p<.05$). Similarly, the expected change in the poverty alleviation index is approximately 25% ($p<.01$). However, for a locality with a 0.75 share in a protected area, the estimated marginal effect of PES on forest cover change is only approximately 9% and only approximately 8% on poverty alleviation, and neither is statistically significant. Taken together, these results suggest that targeting PES to the borders of PAs could have more environmental and social impact than targeting PES to the core areas.

4. ROBUSTNESS CHECKS

Although we control for the factors most likely to affect siting and outcomes, it is possible that unobservable sources of bias or simultaneity issues remain. We conduct a variety of robustness checks in order to ensure that we are using the best comparison group and that our results are not threatened by potential omitted variables or differences in pre-trend.

First, our identification strategy relies on the parallel trends assumption: trends in the outcomes for localities with similar observable characteristics as those that received PES or different types of PAs would have been similar in the absence of these regulations. This is a more plausible assumption if trends in the pre-period were parallel. To test this, we regress the change in forest cover, poverty and population in the 1990's on the share in PES and different park types by 2010, with controls for all time-invariant covariates (Table 7). We find no significant differences ($p < .05$) in pre-trends for forest change, poverty, or population for any of the treatments. Strict PAs show marginal significance indicating slower growth in poverty alleviation in the pre-period; and biosphere reserves show possibly meaningful forest protection (~7%) and slower population growth in population (~1.5%, marginally significant) in the pre-period. These results motivate our inclusion of controls for pre-trends and for the 2000 levels of forest cover, poverty and population, in order to ensure that our estimates pick up only differences in the 2000's and not from the pre-period.

Next, we conducted a variety of robustness checks to see whether our results are sensitive to limiting comparisons to counterfactual localities that are more similar in terms of baseline levels and trends of key variables, to including only exogenous geographic characteristics, and to different measures of the forest cover variable. These are shown in the online Appendix.

Finally, while we have attempted to include a full set of relevant controls, it is possible that there is a key omitted variable correlated with both the policies and the outcomes over the period of interest that might overturn the result. In order to assess this possibility, we conduct an exercise based on the premise that changes in the coefficients of interest with the introduction of covariates can be informative about potential changes in treatment effects with the inclusion of omitted variables (Murphy and Topel 1990, Altonji, Elder and Taber 2008, Oster 2013). The key unknown factor in making this assessment is the covariance between the omitted variable and the treatment variable. If the covariance between the omitted and treatment variable equals the covariance between the observables and the treatment variable, the coefficient of proportionality is one. Following the recommendation of Oster (2013), we calculate both the treatment effect that is implied by an assumed coefficient of proportionality equal to one (β) and the coefficient of proportionality that would overturn our results (δ).[21] Both are shown in Table 8, which also includes the preferred specification results from Table 3 for comparison. We find that for the policy effects which are statistically significant, the coefficients of proportionality required to overturn the results are all greater than 1. Thus, an omitted variable would have to be more correlated with the treatment variable than the current set of observables to produce a true treatment effect equal to zero.

CONCLUSION

Comparisons of conservation policies are important for informing future choices, yet there has been little empirical study of the impacts of direct versus incentive-based mechanisms for land conservation. Evaluating PAs and PES in Mexico, we found that both conserved forest, with similar estimates of avoided deforestation by share of land protected. Relative changes in the poverty alleviation index were negative for strict protected areas but not significantly different from zero for biosphere reserves or mixed-use protected areas. Absolute changes in poverty indicators indicate that on average communities with PES and all types of PAs showed gains in factors such as access to primary school, improved housing conditions, and access to refrigeration. We believe our results are good news overall for policymakers facing a choice between PAs and PES to achieve REDD: both conservation instruments achieved avoided deforestation throughout this period while affected communities saw improvements in economic development indicators. This demonstrates an important point, which is that all forms of environmental protection during this period achieved the goal of "do no harm" in an absolute sense. Yet it raises the ethical question of whether the "do no harm" goal of international conservation should be assessed in an absolute sense or relative to the best available counterfactual.

[21] We assume an Rmax equal to 1.3 times the R-squared achieved by the full regression specification for each outcome. The Rmax is an estimate of the R-squared that would be achieved in the case where we were able to include all the key unobservables. We apply the rule of thumb suggested by Oster (2013).

When we consider the park types separately and account for both location and management type, we find that along with PES, biosphere reserves came the closest to a "win-win" policy for forests and livelihoods. Looking for commonalities, we note that both PES and biosphere reserves recognize the need to promote livelihoods. PES explicitly compensates landowners for their conservation efforts while biosphere reserves are designed to "promote solutions reconciling the conservation of biodiversity with its sustainable use" (UNESCO 2015). Both also include provisions to actively monitor and enforce conservation restrictions—PES through conditionality of payments and biosphere reserves through strictly protected core areas and possibly greater funding for enforcement. Biosphere reserves have received national and international attention in the past decade, so their significant avoided deforestation impacts along with neutral livelihood impacts are encouraging. Still, the results indicate that there are tradeoffs in social goals between mechanism types, with biosphere reserves achieving more environmental impact while PES produced more poverty alleviation. We take a first step towards evaluating the cost-effectiveness of PAs vs. PES by using budget data and production revenues. We find that PES is more expensive in terms of budget outlays and that the strict protected areas likely had the highest opportunity costs of avoided deforestation, but that there is no clear winner between PES and biosphere reserves with respect to cost-effectiveness. These findings suggest that PES is not necessarily cheaper than PAs for reducing carbon emissions for climate change and that policymakers should also expand and fortify existing protected area systems in seeking to achieve REDD goals.

In addition, the relative success of biosphere reserves and PES compared to strict and mixed-use PAs results suggest that it is less important whether conservation instruments are direct or incentive-based than whether they are well-funded and combine enforceable protection with zoning that allows some continued local resource use. Future economic research should focus on more detailed estimates of the social costs of both types of protection while research in multiple disciplines should continue to investigate the institutional and social mechanisms through which direct and incentive-based conservation measures can produce change.

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Figure 1: Parks and PES in Mexico

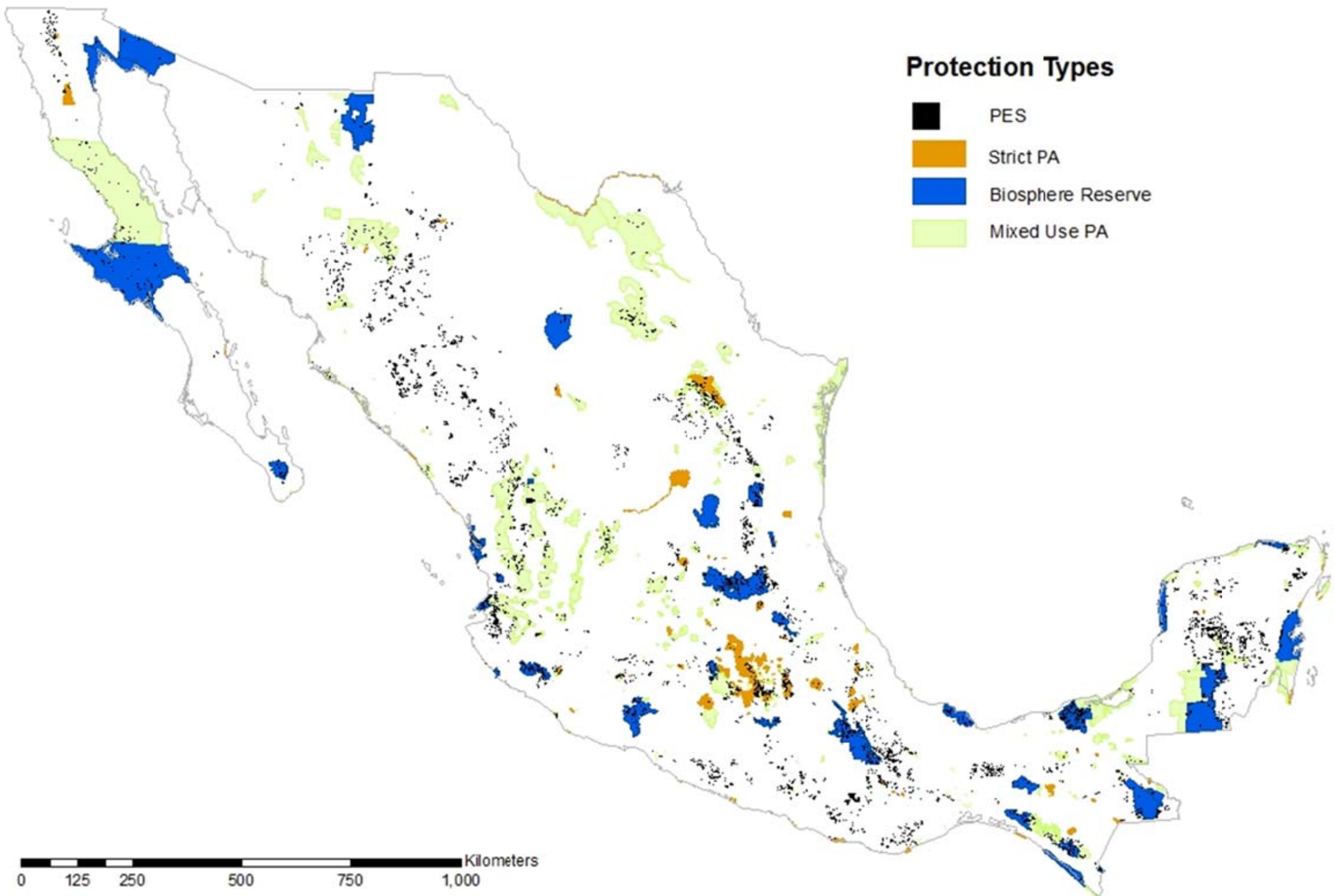
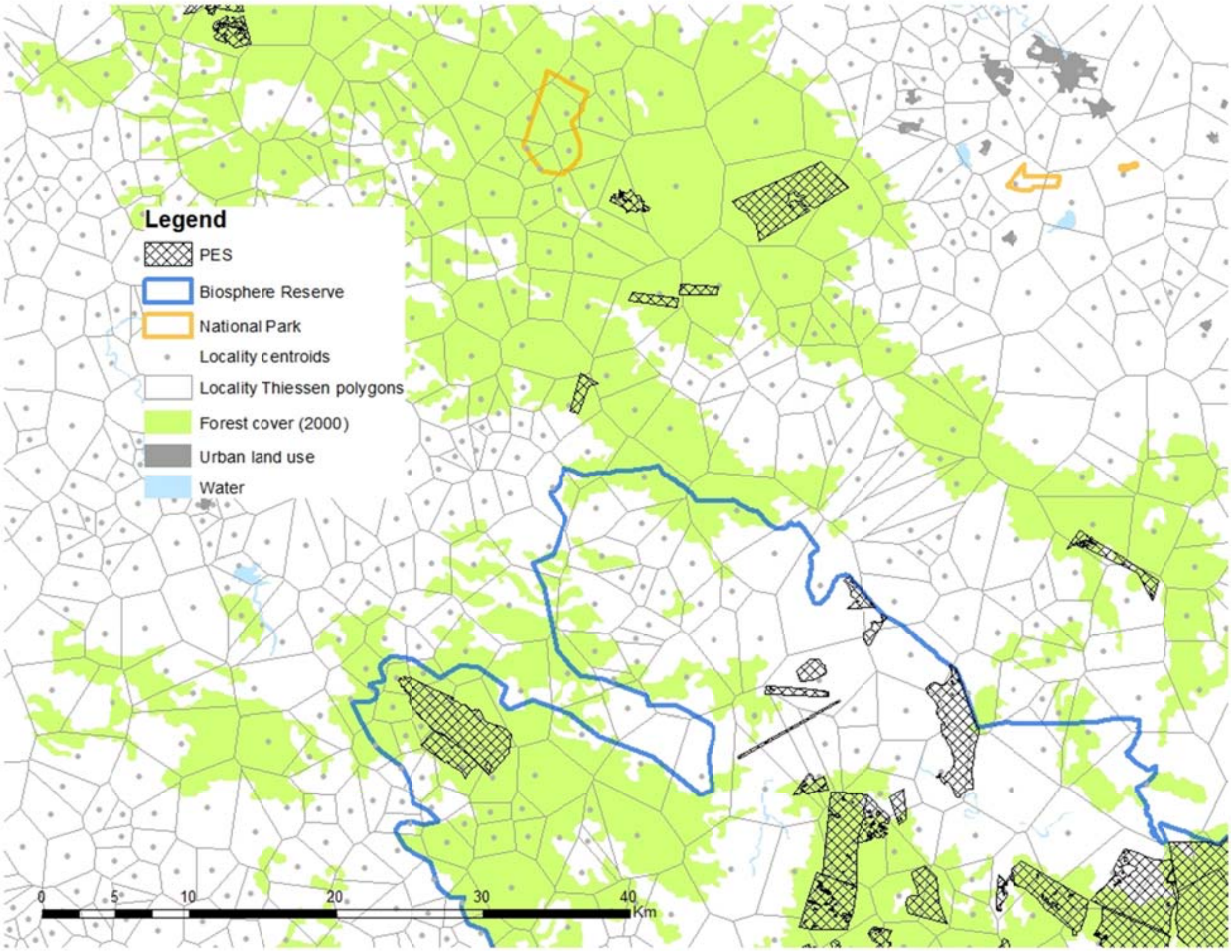
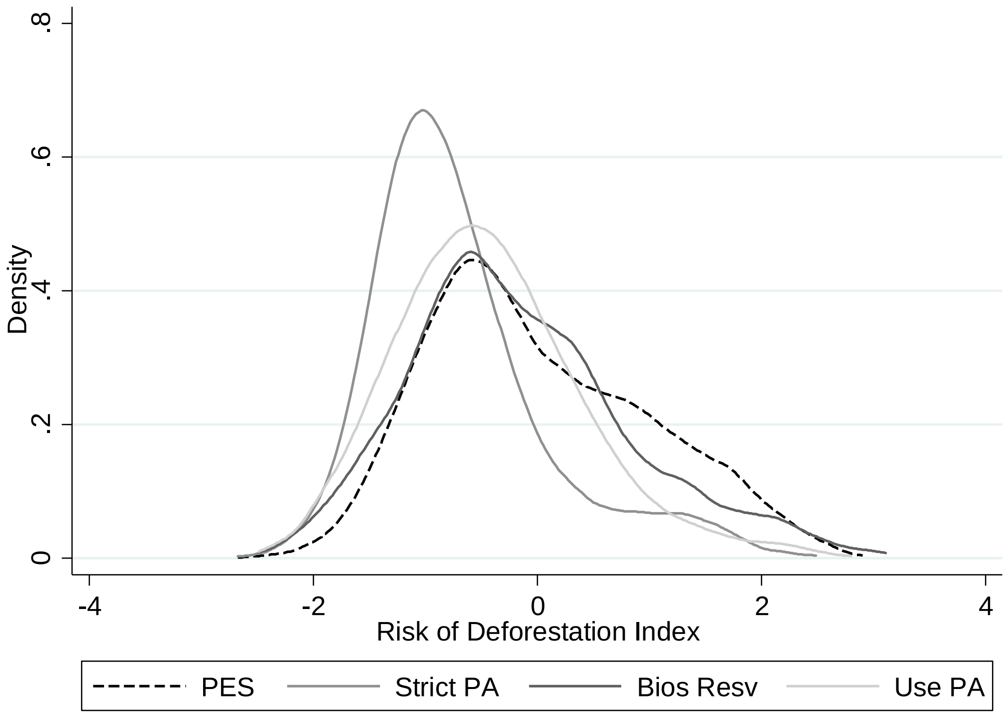


Figure 2: Illustration of locality Thiessen polygons



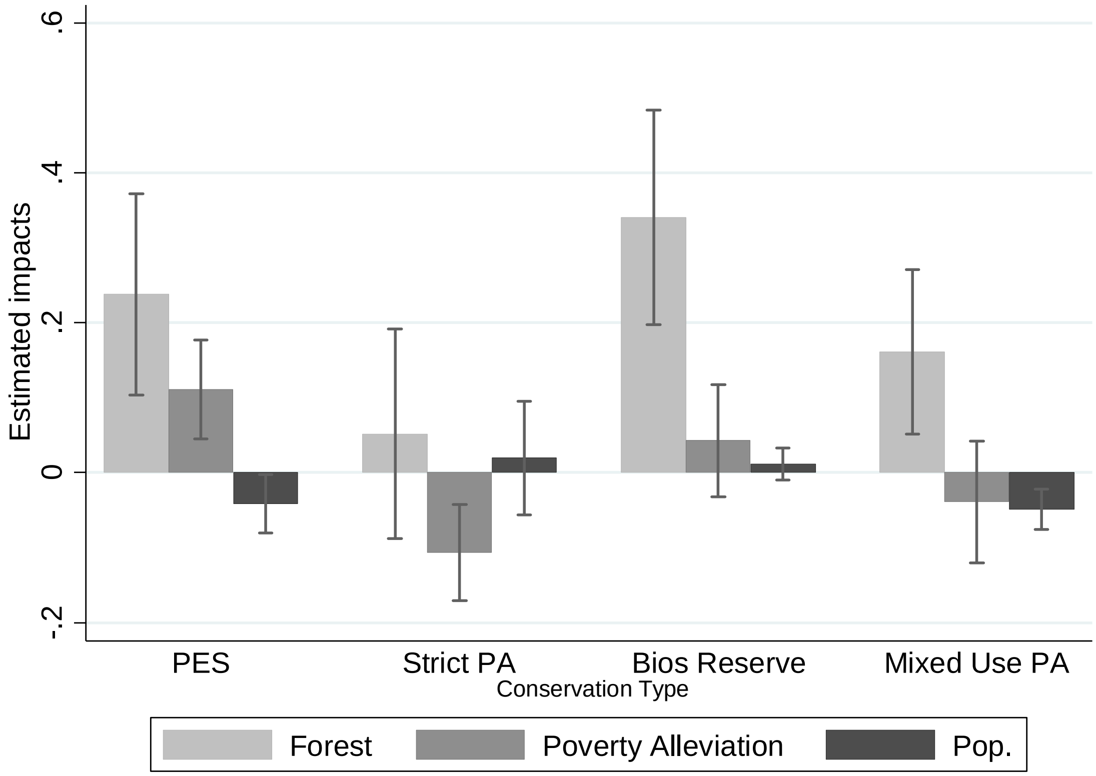
Dots indicate locality centroids; grey lines show Thiessen polygons for each locality (unit of analysis).

Figure 3: Risk profiles of PES, Strict PAs, Biosphere Reserves and Mixed-use PAs



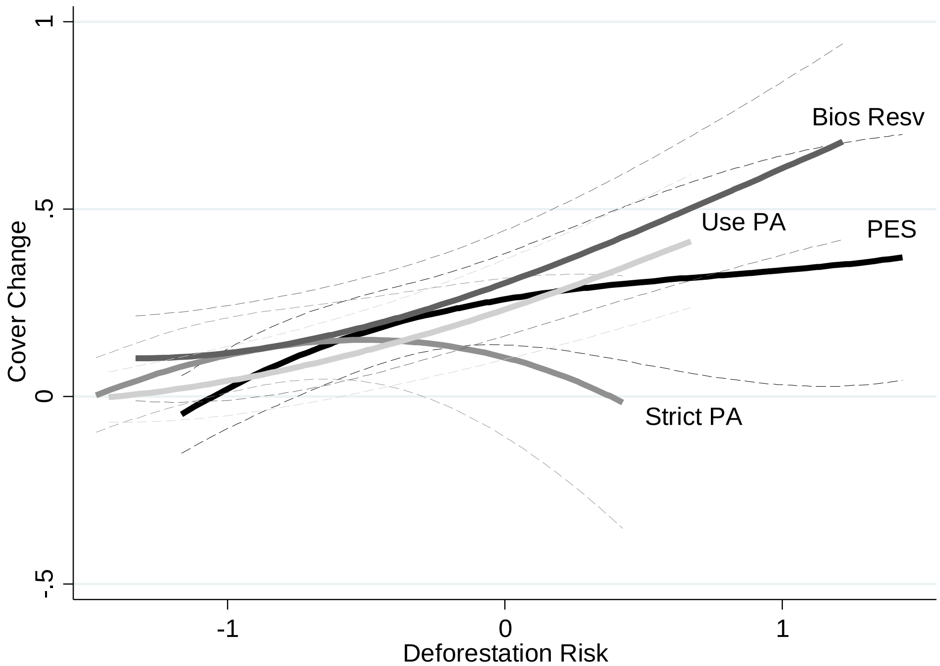
Kernel density distributions of risk of deforestation index by protection type (bandwidth = .2) Predicted risk of deforestation is based on net forest cover change for all non-treated localities as described in the text; index values are normalized to mean zero and SD = 1.

Figure 4: Impacts of PAs and PES across conservation and development dimensions

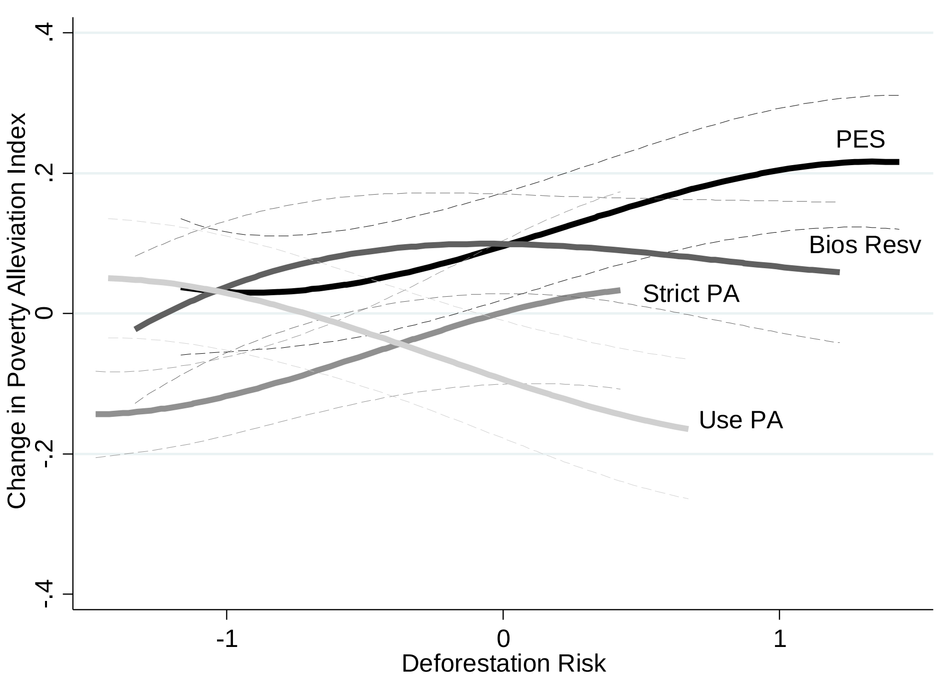


Coefficients and 95% confidence intervals from main specification (equation 1 and Table 3 columns 3, 6, 9). A "win-win-win" situation for forests, poverty alleviation, and population would have positive values for all estimated coefficients.

Figure 5: Marginal impacts of all policy types by predicted risk of deforestation
a: Forest cover change

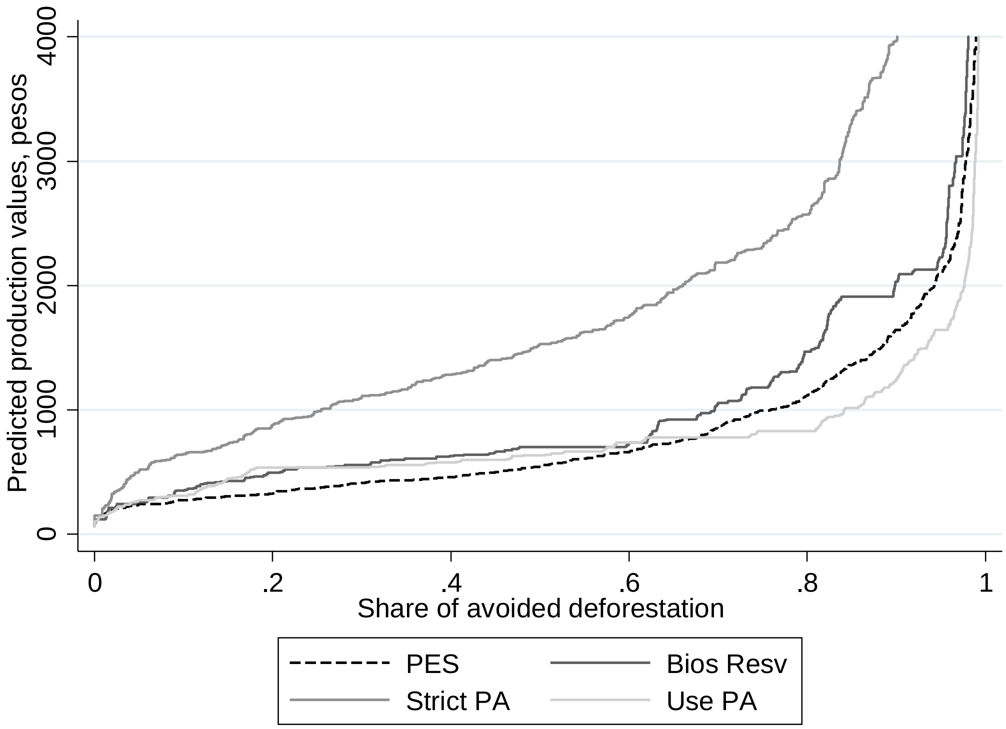


b: Change in poverty alleviation index



Coefficients and 95% confidence intervals from main specification with interactions with deforestation risk, risk squared and risk cubed. Marginal effects for the 10th to 90th percentile of deforestation risk.

Figure 6: Locality production revenues vs. cumulative proportion of avoided deforestation



For each policy, graph shows the relationship between predicted locality production revenues and estimated cumulative avoided deforestation as a proportion of the total estimated avoided deforestation for each policy.

Table 1: Protection types and area protected within localities analyzed

Category	Analysis grouping	Stringency	Management Level	Area protected (sq km)	Mean % protected
Protected areas					
1. Natural Monument	Strict PAs	Strict protection (IUCN III)	Federal/State/ Municipal	287	0.017
2. Sanctuary	Strict PAs	Strict protection (IUCN 1a)	Federal/State	454	0.040
3. National Park	Strict PAs	Strict protection (IUCN II)	Federal	5,538	0.657
4. State Park	Strict PAs	Strict protection (IUCN II)	State	4,447	1.105
5. Muni Park		Strict protection (IUCN II)	Municipal	omitted	0.000
6. Biosphere Reserve	Biosphere Reserve	Mixed: Core areas strict protection (ICUN 1); Buffer zones (IUCN VI)	Federal/State	42,051	2.709
7. Flora and Fauna Protection	Mixed Use	Sustainable use (IUCN IV)	Federal/State/ Municipal	38,195	1.996
8. Natural Resource Protection	Mixed Use	Sustainable use (IUCN VI)	Federal/State/ Municipal	30,157	2.122
9. Certified Area	Mixed Use	Sustainable use (IUCN VI)	Local	1,124	0.077
Payments for Ecosystem Services					
Hydrological Services, Biodiversity Conservation, Carbon Sequestration	PES	Maintain existing vegetative cover, implement management plan	Federal	26,844	2.532

Column 1 gives the categories as defined by the Mexican government; column 2 gives our grouping according to stringency of legal restrictions. Column 3 lists the corresponding IUCN category and column 4 gives the level of government responsible for management. The IUCN categories are based on correspondence between definitions under Mexican law and IUCN definitions. Finally, the last two columns list the total area protected under that category and the mean percent of land protected across the localities within the sample of localities analyzed.

Table 2: Summary statistics (means)

	All Localities	PES > 5%	PA > 5%	Strict PA > 5% (Cat 1-4)	Biosphere Reserves > 5% (Cat 6)	Mixed Use > 5% (Cat 7-9)
Treatment	(1)	(2)	(3)	(4)	(5)	(6)
Share locality in PES by 2010	0.025	0.294	0.068	0.064	0.094	0.063
Share locality in PA by 2010	0.080	0.231	0.716	0.728	0.794	0.693
Outcomes						
IHS (% forest cover change, 2000-2012)	-0.879	-0.733	-0.483	-0.421	-0.541	-0.472
IHS (change poverty alleviation index, 2000-2010)	0.033	0.026	-0.026	-0.096	0.097	-0.064
IHS (change population density, 2000-2010)	0.062	0.042	0.078	0.238	0.025	0.035
Covariates						
Ln (km to loc. w/ pop > 5000)	2.830	3.001	2.805	2.147	2.991	2.979
Average elevation (m)	1029.8	1452.2	1456.5	2110.6	1121.2	1348.8
Average slope (deg)	8.865	12.31	10.38	9.965	10.89	10.21
Ln (locality area in km^2)	2.148	2.496	2.489	2.048	2.536	2.664
Locality poverty alleviation index, 2000	-0.368	-0.444	-0.190	0.196	-0.396	-0.260
IHS (change poverty alleviation index, 1990-2000)	-0.001	-0.007	-0.024	-0.057	0.042	-0.043
Population density, 2000 (100 people per sq km)	0.354	0.278	0.342	0.689	0.222	0.246
Ln (km to any road)	1.218	1.332	1.280	0.836	1.427	1.385
Ln (km to urban area)	3.813	3.821	3.649	2.844	4.023	3.804
IHS (% forest loss, 1993-2000)	-0.940	-0.967	-0.958	-0.204	-1.162	-1.132
IHS (change in pop density 1995-2000)	0.037	0.040	0.067	0.190	0.006	0.040
Ln (% locality w/ tree cover, 2000)	3.524	3.961	3.508	3.441	3.630	3.502
Municipal poverty alleviation index, 2000	-0.436	-0.497	-0.108	0.280	-0.402	-0.125
Water availability	7.284	6.979	6.451	4.951	6.975	6.777
Overexploited watershed (0/1)	0.075	0.117	0.132	0.352	0.034	0.080
Majority indigenous muni (0/1)	0.264	0.329	0.123	0.071	0.118	0.150
Coniferous forests (share)	0.375	0.548	0.557	0.747	0.388	0.561
Dry broadleaf forests (share)	0.202	0.115	0.142	0.083	0.132	0.181
Moist broadleaf forests (share)	0.376	0.325	0.222	0.116	0.368	0.181
North (region 1)	0.283	0.243	0.284	0.109	0.276	0.397
Center (region 2)	0.343	0.370	0.531	0.695	0.507	0.425
Southwest (region 3)	0.148	0.204	0.054	0.037	0.048	0.082
Southeast (region 4)	0.226	0.183	0.132	0.159	0.169	0.097
% common property	42.43	53.93	42.01	41.42	42.44	42.54
Predicted deforestation risk						
Deforestation risk index	-0.000	0.009	-0.422	-0.700	-0.160	-0.433
N localities	59535	4984	6630	1567	2107	3662

Table 3: Impacts of PES and Protected Areas: simple differences, controls and state fixed effects: two and four categories

Dependent variable:	Forest change (2000-2012)			Poverty alleviation (2000-2010)			Population growth (2000-2010)		
PANEL A	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)
Share PES	0.062	0.2865***	**0.2516*** **	0.027	0.1116***	**0.1169*** **	-0.0642***	-0.0692***	**-0.0398** **
	(0.1470)	(0.0800)	**(0.0685)**	(0.0400)	(0.0368)	**(0.0334)**	(0.0211)	(0.0209)	**(0.0202)**
Share PA	0.5611***	0.1918***	**0.2360*** **	-0.0690**	0.0293	**-0.027**	0.0121	0.0022	**-0.0183**
	(0.0583)	(0.0472)	**(0.0448)**	(0.0337)	(0.0378)	**(0.0285)**	(0.0145)	(0.0123)	**(0.0125)**
R^2	0.016	0.304	**0.338**	0.001	0.151	**0.192**	0.000	0.050	**0.059**
PANEL B	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)
Share PES	0.0811	0.2801***	**0.2384*** **	0.0125	0.1007***	**0.1110*** **	-0.0608***	-0.0671***	**-0.0416** **
	(0.1474)	(0.0797)	**(0.0685)**	(0.0407)	(0.0370)	**(0.0336)**	(0.0202)	(0.0208)	**(0.0200)**
Share Strict PA	0.5864***	0.0357	**0.0519**	-0.1576***	-0.0501	**-0.1061*** **	0.2030***	0.0771**	**0.0197**
	(0.1013)	(0.0699)	**(0.0714)**	(0.0351)	(0.0344)	**(0.0327)**	(0.0425)	(0.0385)	**(0.0387)**
Share Biosphere Res	0.3454***	0.2344***	**0.3406*** **	0.1195**	0.1524***	**0.0429**	-0.0329***	-0.0057	**0.0118**
	(0.0927)	(0.0747)	**(0.0730)**	(0.0465)	(0.0529)	**(0.0380)**	(0.0087)	(0.0101)	**(0.0109)**
Share Mixed Use PA	0.4804***	0.1511**	**0.1617*** **	-0.1344***	-0.043	**-0.0388**	-0.0431***	-0.0249*	**-0.0489*** **
	(0.0854)	(0.0658)	**(0.0560)**	(0.0438)	(0.0483)	**(0.0415)**	(0.0122)	(0.0136)	**(0.0136)**
R^2	0.014	0.304	**0.338**	0.004	0.153	**0.193**	0.004	0.050	**0.059**
Covariates		Y	Y		Y	Y		Y	Y
State FE			Y			Y			Y
N	59535	59535	59535	59535	59535	59535	59535	59535	59535

* p< .10 ** p < .05 *** p < .01 Robust standard errors, clustered by municipality in parentheses. Dependent variables are inverse hyperbolic sine transformations of the percent change in forest cover from 2000-2012, the change in the standardized marginality index * -1, and the change in population density.

Columns 1, 4, 7: Differences in mean outcomes regressed on share of locality in each category of protection; Columns 2, 5, 8 add covariate controls; Columns 3, 6, 9: add state fixed effects and match the full model as given in Equation 1. Covariate controls are: average slope and average elevation (spline function, 5 categories), log distance to nearest locality with pop > 5000, locality anti-poverty index in 2000, change in forest cover 1993-2000 (hyperbolic sine transformed), change in locality anti-poverty index 1990-2000 (hyperbolic sine transformation), municipal anti-poverty index in 2000, log population density in 2000, change in population density 1995-2000 (hyperbolic sine transformed), log distance to nearest road, log distance to nearest urban area, log percent forest cover in 2000, average availability of water, overexploited watershed status, log locality area, share in major ecoregions, whether municipality is majority indigenous and percent in common property.

Table 4: Effects on individual components of poverty index and mean changes in poverty indicators from 2000-2010

Dependent variable:	**Full index (higher values are more poor)**	**% of population that is illiterate**	**% without primary school**	**% with dirt floor**	**% without refrig**	**% without piped water**	**% without electricity**
PANEL A: Impact effects of Parks and PES (regression coefficients)							
	(1)	(2)	(3)	(4)	(5)	(6)	(7)
Share PES	-0.111***	-0.114***	-0.089*	-0.175*	-0.108*	-0.017	-0.207***
	(0.034)	(0.043)	(0.046)	(0.094)	(0.064)	(0.063)	(0.073)
Share Strict PA	0.106***	0.055*	0.044	0.041	0.226**	0.021	0.083
	(0.033)	(0.029)	(0.035)	(0.057)	(0.093)	(0.064)	(0.085)
Share Biosphere Reserve	-0.043	0.017	-0.045	-0.111*	0.038	0.01	-0.051
	(0.038)	(0.045)	(0.037)	(0.065)	(0.072)	(0.054)	(0.064)
Share Use PA	0.039	0.091**	0.059	0.085	0.043	-0.019	0.031
	(0.041)	(0.041)	(0.038)	(0.086)	(0.070)	(0.056)	(0.068)
N	59535	59535	59535	59535	59535	59535	59535
R^2	0.193	0.077	0.091	0.113	0.324	0.07	0.234
PANEL B: Summary statistics for the changes in individual components (percentage points)							
	(1)	(2)	(3)	(4)	(5)	(6)	(7)
PES > 5%	-0.023	-6.89	-15.48	-33.23	-18.70	-6.59	-27.58
Strict PA > 5%	0.108	-6.11	-14.17	-19.41	-20.30	-7.95	-13.91
Biosphere Reserves > 5%	-0.102	-5.94	-15.87	-33.08	-27.21	-7.36	-20.84
Use PA > 5%	0.080	-5.46	-14.44	-24.94	-22.96	-6.28	-20.64

Panel A: Regressions where dependent variables are the changes from 2000 to 2010 in the poverty index and the standardized values of each component of this index that was available in both years. Area weighted means are calculated to account for changes in locality boundaries across time. Specifications include the same covariates as Table 3, columns 3, 6, 9 (state fixed effects and full controls). Stars indicate: * p< .10 ** p < .05 *** p < .01 Robust standard errors, clustered by municipality are in parentheses. Panel B: Absolute changes in mean values of the index for sub-samples with PES and different park types. The number of observations for each category matches the numbers given in the summary statistics (Table 2).

Table 5: Park revenues collected and predicted locality production revenues by policy

	All Localities	PES > 5%	PA > 5%	Strict PA > 5% (Cat 1-4)	Biosphere Reserves > 5% (Cat 6)	Mixed Use > 5% (Cat 7-9)
Park revenues collected (proxy for tourism):						
PA revenues (1000 pesos/sq km of PA)	476	9.68	2874	11,891	17.5	111
PA revenues (1000 pesos/person)	47.7	26.6	416	1501	149	64.2
Predicted locality production revenues (proxy for opportunity cost):						
Municipality average production revenues (pesos/ha)	2416	2070	2772	4266	2195	2725
Predicted locality-level production revenues (pesos/ha)	1866	1444	1717	2747	1315	1427

Municipality average production revenues are from 2003 (the date closest to the start of the analysis period that did not have substantial missing data). Locality level production revenues are predicted based on locality covariates as described in the text. During this time period, the exchange rate fluctuated around 11-12 pesos per US dollar.

Table 6: Complementary Effects of Parks and PAs?

Dependent variable:	Forest change (2000-2012)	Poverty alleviation (2000-2010)	Population growth (2000-2010)	Forest change (2000-2012)	Poverty alleviation (2000-2010)	Population growth (2000-2010)
	(1)	(2)	(3)	(4)	(5)	(6)
Share PES	0.3101***	0.1261***	-0.0421*	0.2944***	0.1220***	-0.0455**
	(0.0841)	(0.0387)	(0.0227)	(0.0843)	(0.0391)	(0.0228)
Share PA	0.2543***	-0.0242	-0.019	0.2518***	-0.0231	-0.0194
	(0.0462)	(0.0296)	(0.0136)	(0.0459)	(0.0294)	(0.0136)
Share PES x Share PA	-0.2885**	-0.0454	0.0114	-0.2775**	-0.0576	0.0123
	(0.1238)	(0.0765)	(0.0445)	(0.1224)	(0.0757)	(0.0448)
PA border				0.0663***	-0.0275*	0.0101
				(0.0231)	(0.0146)	(0.0125)
Share PES x Share PA x PA border				0.471	0.5556**	0.1425
				(0.4886)	(0.2620)	(0.1799)
N	59535	59535	59535	59535	59535	59535
R^2	0.338	0.192	0.059	0.338	0.192	0.059

* p< .10 ** p < .05 *** p < .01 Robust standard errors, clustered by municipality. Dependent variables are inverse hyperbolic sine transformations of the percent change in forest cover from 2000-2012, the change in the marginality index * -1, and the change in population density. Same specifications as Table 3 columns 3, 6, 9 (state fixed effects and full controls). PA border is a dummy variable equal to 1 if a locality has more than zero and less than 0.5 share in a protected area. There are 2813 localities in the data that meet this definition of PA border. Of these, 430 localities had at least 5% of area enrolled in PES.

Table 7: Pre-intervention trends for main estimation

Dependent variable:	Forest change (1993-2000)	Poverty alleviation (1990-2000)	Population growth (1995-2000)
	(1)	(2)	(3)
Share PES	-0.0062	-0.0014	0.0249
	(0.1939)	(0.0360)	(0.0165)
Share Strict PA	0.0169	-0.0500*	0.0355
	(0.2110)	(0.0275)	(0.0252)
Share Biosphere reserve	0.0735	0.0227	-0.0150*
	(0.1831)	(0.0292)	(0.0083)
Share Mixed Use PA	-0.1246	-0.0453	0.0013
	(0.1455)	(0.0320)	(0.0117)
N	59535	59535	59535
R^2	0.125	0.049	0.027

* p< .10 ** p < .05 *** p < .01 Robust standard errors, clustered by municipality in parentheses. Regressions include ecoregion dummies, slope and elevation categories, log locality area, logs of the distance to nearest road, to nearest city over 5000, and to nearest urban area, an indicator for overexploited watershed, water availability, majority indigenous municipality, percent common property in the locality, and state dummy variables.

Table 8: Impacts with estimated bounds

Outcomes	**Forest change (2000-2012)**	**Poverty alleviation (2000-2010)**	**Population growth (2000-2010)**
	(1)	(2)	(3)
Share PES	0.2384***	0.1110***	-0.0416**
	(0.0685)	(0.0336)	(0.0200)
β adjusted	**0.2352**	**0.1432**	**-0.0360**
δ	***51.41***	***-1.789***	***5.485***
Share strict PA	0.0519	-0.1061***	0.0197
	(0.0714)	(0.0327)	(0.0387)
β adjusted	**-0.1170**	**-0.0877**	**-0.0366**
δ	***0.3153***	***3.951***	**0.*3676***
Share biosphere reserve	0.3406***	0.0429	0.0118
	(0.0730)	(0.0380)	(0.0109)
β adjusted	**0.3121**	**0.0264**	**0.0303**
δ	***6.802***	***2.460***	***-0.5956***
Share use PA	0.1617***	-0.0388	-0.0489***
	(0.0560)	(0.0415)	(0.0136)
β adjusted	**0.0447**	**-0.0136**	**-0.0500**
δ	***1.332***	***1.499***	***-29.33***
Covariates	Y	Y	Y
State FE	Y	Y	Y
N	59535	59535	59535

* p< .10 ** p < .05 *** p < .01 Robust standard errors, clustered by municipality. The first rows give the covariates from columns 3, 6, 9 of Table 3, panel B for comparison. Below this, coefficients in **bold (β)** indicate the treatment effect that is implied by a coefficient of proportionality equal to one and the ***bold italics (δ)*** give the coefficient of proportionality needed to overturn our results (following Oster 2013).

ONLINE APPENDIX: Robustness checks and additional analysis

Park revenues and heterogeneous impacts

In Table A1 we further explore how park revenues may interact with the share of land protected. We find a significant positive interaction between PA revenues and share PA for forest cover change (column 1), suggesting greater avoided deforestation where parks take in more revenues. In addition, in column 4 we add the risk of deforestation and find that PAs generate more avoided deforestation at higher levels of deforestation risk (coefficient on Risk x share PA = 0.26) and that there is a marginally significant increase in avoided deforestation when park revenues are also high (coefficient on PA revenues x Share PA x Risk = 0.0010). Considering columns 2 and 3, we find that parks with higher revenues are not significantly associated with more poverty alleviation or population growth. However, when we take into account opportunity cost, the sign of the interaction term on PA revenues x Share PA x Low opp cost is positive (but not significant) and there is a significant positive coefficient on population growth (coefficient = 0.0022).

Appendix Figure 1 also graphs out the marginal impacts of PES and PAs across different values of the opportunity cost proxy. Figure 1a confirms that the biosphere reserves have higher estimated marginal effects across different levels of production revenues. Figure 1b indicates that the poverty alleviation impacts of PES are declining with higher opportunity cost, which would be expected given the fixed PES payment levels. Yet the PAs do not show a consistent pattern for poverty alleviation impacts, again suggesting complicated relationships between revenues generated from tourism or use and the opportunity costs of foregone production.

Forest cover change robustness checks

Table A2 shows robustness to using different specifications for our cover change outcome variable. In order to ensure that our choice to use transformed net forest cover change as an outcome does not drive our result, we test the robustness of our environmental impact estimation to alternate specifications. The columns, in order, show: 1) our baseline outcome (the inverse hyperbolic sine transformation of percent net forest cover change), 2) the inverse hyperbolic sine transformation of hectares forest cover change, 3) a binary variable indicating forest loss greater than 10 hectares, 4) standardized forest loss (values in standard deviations away from the mean), 5) a Winsorized transformation of the data where we replace the top and bottom 5% of the data with the value of the observation in the 95th and 5th percentiles and 6) the inverse hyperbolic sine transformation of gross forest loss. In all cases, our results remain similar to our baseline specification. Note that the signs flip as expected on columns 3 and 6 because these outcomes measure deforestation rather than net forest cover change. The magnitudes are also very similar when we use only gross forest loss rather than net forest loss (column 6), suggesting that the main impact of PES and PAs is through avoided deforestation, not through reforestation.

Specification checks on samples and controls

Appendix Table A3 assesses whether or not the results are sensitive to using different samples of localities or controls to construct the counterfactual. The first panel (a) shows the full sample of all localities in Mexico, regardless of baseline forest cover. The results for forest cover and poverty alleviation of PES are similar in size and significance to our preferred estimation (bolded in Table 3) and none of the pre-trends are statistically significantly different from zero. Panel (b) eliminates controls for the pre-period trends and that are measured in 2000, in order to check robustness to not using controls for poverty and population change that could have been affected by parks in the pre-period. The exception is that we retain 2000 forest cover as a control, since this is a key eligibility requirement for PES, so not including it would create obvious omitted variable bias in the PES estimates. Estimates in panel (b) are again similar to the main results. Panel (c) adds our proxy for opportunity cost at the locality level, thus including an additional control for possible economic opportunities; results are similar to the main specification. Panel (d) uses a subsample which contains only localities with at least 20% and greater than 10 ha baseline forest cover. Panel (e) uses a matched subsample, where the matching algorithm is conducted three times, once for each policy, using the pre-trend, locality area, and percent forest area in 2000 as matching variables. Repeated matches are eliminated from the sample. Finally, panel (f) excludes the smallest 10% of localities. This is done because many localities that occur in densely populated areas tend to be quite small, and the accuracy of the land-use change measurements may be compromised for small localities. Results are similar in panels (d), (e) and (f), except that (d) and (f) do not show a significant decrease in population due to PES. The forest cover impacts of PES are also somewhat smaller for specification (e), with a coefficient of 0.19. Given these results, we do not emphasize the population decrease associated with PES in our findings and we report our main results as an approximately 20-25% change in forest cover.

Timing of treatment robustness checks

We chose to use the share of land protected under either PAs or PES at any point during the years 2000-2010 because it is the most conservative measure of having received any conservation "treatment" during this period. To check that this timing choice is not driving our results, we include three robustness checks in Table A4. Columns 1-3 break the parks variable into the share of land protected before 2000 and the share of land protected between 2000-2010. The results are very similar to the main results in the paper, which reflects the fact that the majority of parks were established before 2000. (5224 localities had some PA protection before 2000 and 1632 had some protection added between 2000-2010.) There are not enough parks in each category established between 2000-2010 in order to break the new parks down into separate types. However, we see that together the results for those new parks are similar in terms of impacts on forest cover (~20%) and they do not appear to have significantly affected poverty or population trends. Columns 4-6 show a similar analysis, breaking the parks into those established before 2005 and between 2005-2010, again with similar results.

A second concern is that PES was received for more years in some localities than others. To address this, we created a variable that multiplied the share of area in each year receiving PES times the number of years receiving payments, to take into account how long each parcel of land received payments. Results using this more nuanced treatment measure are shown in columns 7-9 of Table A4. The sign and significance of the PES results for forest cover change and poverty corroborate the main specification; population change is again not significantly impacted by PES in this robustness check. To interpret the magnitude of the coefficients on share PES * years paid, if we think about a change from none of the locality protected to full protection, for 5 years, this would correspond to a change in forest cover of approximately 22% and in the poverty alleviation index of 9%, results that are similar to the main results in magnitude.

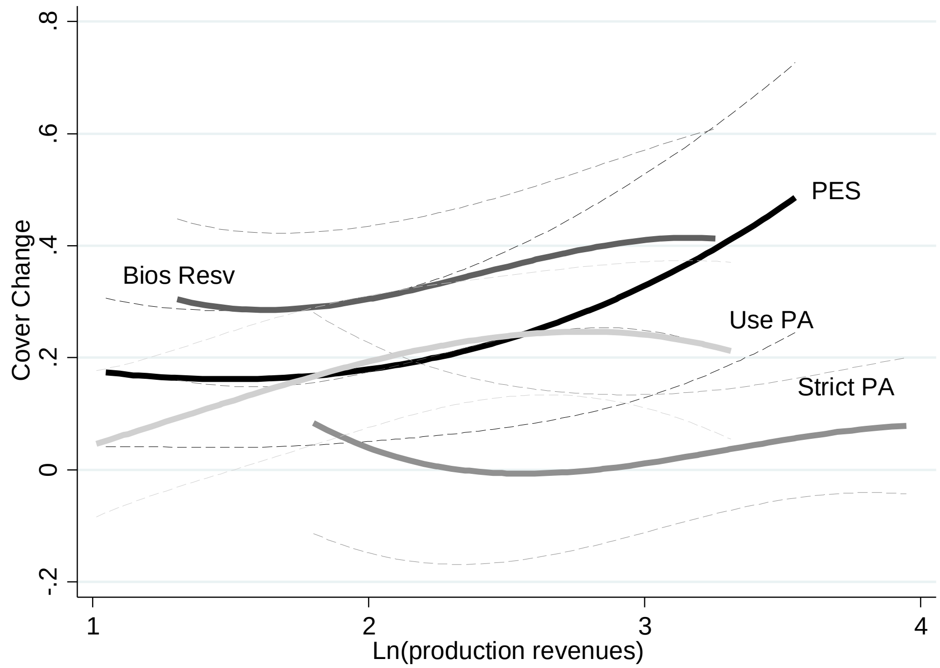
Weighting

Our main estimates use localities as the unit of analysis in order to have a consistent unit across the outcomes. Yet these polygons are generally smaller where population density is higher, which implicitly weights our results towards more highly populated areas. The results are likely to vary somewhat if there is heterogeneity in impacts by locality size and we use different weights (e.g. Solon, et al. 2015). Table A5 presents different estimates with different weights. For the forest outcomes, we re-weight by locality polygon size or area of baseline forest cover. For the poverty and population outcomes, we re-weight by baseline population. We also include un-weighted and weighted regressions that drop the top 1% of observations in terms of area (for forest) or population (for poverty and population outcomes). The reason for this is that weighting can skew results heavily towards outliers when the scales are very different (Solon et al. 2015).

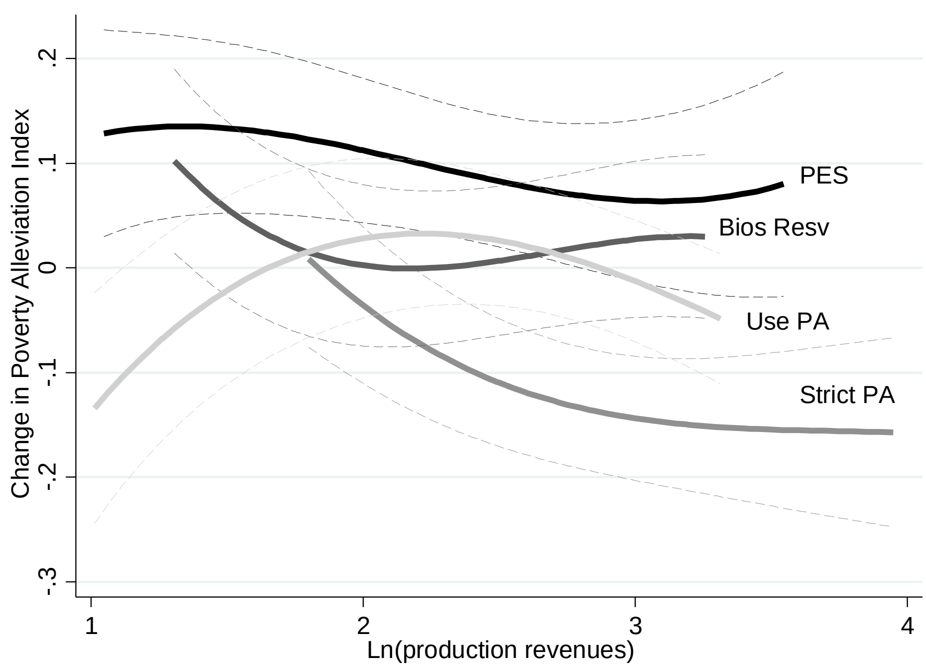
Overall, the results in Table A5 show that weighting can matter, particularly for the magnitude of the forest cover results for biosphere reserves, the population growth results and for the poverty impacts of mixed use areas. However, the core results emphasized in the main text are similar across weighting schemes. In columns 1-5, we observe that when weighting by either Thiessen polygon area or baseline forest size, the point estimate for the forest cover impact of biosphere reserves becomes substantially larger (up to 0.799), and the PES impact somewhat smaller (minimum 0.17). This can be explained by the fact that PAs are often located in very large polygons and that avoided deforestation effects are evidently greater in these large areas. However, when we drop the top 1% largest areas, the weighted regression results are similar to the main effects, except for the coefficient on strict protected areas. Strict protected areas do not significantly impact forest cover change in any of our specifications, but the sign does change with weighting by area. Regarding the poverty results (columns 6-9), we observe that for most of the coefficients, the estimates are quite similar across weighting schemes and subsamples. Use PAs show significant negative effects in the full population weighted specification but this is not robust to eliminating population outliers. Similarly, in columns 10-13 we see large negative coefficients for both PES and use PAs when using population weights, but results are similar to our main results when we drop the population outliers. In all cases, the un-weighted estimates give the most conservative magnitudes of potential impact, thus we retain those as the main results, but have attempted to be transparent about the choice and the implications of weighting.

Appendix Figure 1: Marginal impacts of all policy types by predicted production revenues

a: Forest cover change



b: Change in poverty alleviation index



Coefficients and 95% confidence intervals from main specification with interactions with production revenues, revenues squared and revenues cubed. Outcomes graphed from 10th to 90th percentile of revenues for each policy.

Table A1: Park revenues, risk, and opportunity cost

Outcomes	Forest change (2000-2012)	Poverty alleviation (2000-2010)	Population growth (2000-2010)	Forest change (2000-2012)	Poverty alleviation (2000-2010)	Population growth (2000-2010)
	(1)	(2)	(3)	(4)	(5)	(6)
Share PES	0.2518***	0.1168***	-0.0396*	0.2170***	0.0627	0.044
	(0.0685)	(0.0334)	(0.0202)	(0.0649)	(0.0880)	(0.0500)
Share PA	0.2352***	-0.0266	-0.0191	0.3444***	0.0075	-0.0666*
	(0.0449)	(0.0285)	(0.0125)	(0.0567)	(0.0881)	(0.0387)
PA revenues x Share PA	0.0009**	-0.0005***	0.0009	0.0006**	-0.0001	0.0073***
	(0.0004)	(0.0001)	(0.0012)	(0.0003)	(0.0008)	(0.0014)
Risk of deforestation				-0.5551***		
				(0.0831)		
Risk x Share PES				0.1051		
				(0.0762)		
Risk x Share PA				0.2642***		
				(0.0503)		
PA revenues x Share PA x Risk				0.0010*		
				(0.0005)		
Low opp cost					0.2569***	0.0078
					(0.0310)	(0.0162)
Low opp cost x Share PES					-0.0255	0.0396
					(0.0362)	(0.0265)
Low opp cost x Share PA					0.015	-0.0211
					(0.0298)	(0.0190)
PA revenues x Share PA x Low opp cost					0.0001	0.0022***
					(0.0003)	(0.0004)
N	59535	59535	59535	59535	59535	59535
R^2	0.338	0.192	0.059	0.34	0.192	0.06

* p< .10 ** p < .05 *** p < .01 Robust standard errors, clustered by municipality in parentheses. Specifications in columns (1)-(3) include the same covariates as Table 3 columns 3, 6, 9 (state fixed effects and full controls). The "low opportunity cost" variable is the opposite of the opportunity cost variable (predicted locality production revenues).

Table A2: Robustness to different forest cover change outcome variables

Dependent variable:	**Percent net forest cover change**	**Hectares net forest cover change**	**Deforest > 10 ha (0/1)**	**Standardized forest loss**	**Windsorized percent net forest cover change**	**Percent gross forest loss**
	(1)	(2)	(3)	(4)	(5)	(6)
Share PES	0.2384***	0.3864***	-0.0795***	0.0966**	0.6213***	-0.2379***
	(0.0685)	(0.1149)	(0.0273)	(0.0465)	(0.1731)	(0.0678)
Share strict PA	0.0519	0.1079	-0.0192	-0.0119	-0.0058	-0.0616
	(0.0714)	(0.1189)	(0.0240)	(0.0342)	(0.2010)	(0.0732)
Share biosphere reserve	0.3406***	0.5084***	-0.1201***	0.1111**	0.8554***	-0.3408***
	(0.0730)	(0.1261)	(0.0308)	(0.0527)	(0.1708)	(0.0786)
Share use PA	0.1617***	0.2910***	-0.0667***	0.1152***	0.3814***	-0.1662***
	(0.0560)	(0.1057)	(0.0248)	(0.0426)	(0.1322)	(0.0565)
N	59535	59535	59535	59535	59535	59535
R^2	0.338	0.346	0.351	0.252	0.375	0.534

* p< .10 ** p < .05 *** p < .01 Robust standard errors, clustered by municipality. Columns 1, 2, 5, and 6 are inverse hyperbolic sine transformed. Specifications include the same covariates as Table 3 columns 3, 6, 9 (state fixed effects and full controls).

Table A3a: Robustness checks with different samples or controls

Dependent variable:	Outcomes (2000's)			Pre-trends (1990's)		
	Forest change	**Poverty alleviation**	**Population growth**	**Forest change**	**Poverty alleviation**	**Population growth**
(a) Full sample	**(1)**	**(2)**	**(3)**	**(4)**	**(5)**	**(6)**
Share PES	0.1774**	0.1414***	-0.0646***	0.0148	0.0168	0.0105
	(0.0731)	(0.0346)	(0.0206)	(0.1828)	(0.0401)	(0.0175)
Share Strict PA	0.0511	-0.0383	-0.0446	0.2156	0.0057	-0.0373
	(0.0508)	(0.0251)	(0.0456)	(0.1584)	(0.0251)	(0.0266)
Share Biosphere Reserve	0.2191***	0.0303	-0.0005	0.0503	-0.0014	-0.0119
	(0.0515)	(0.0403)	(0.0186)	(0.1719)	(0.0261)	(0.0085)
Share Use PA	0.1459***	-0.0636*	-0.0540***	-0.0982	-0.0298	-0.0164
	(0.0450)	(0.0353)	(0.0177)	(0.1261)	(0.0266)	(0.0119)
N	105632	105632	105632	105632	105632	105632
R^2	0.392	0.223	0.08	0.106	0.046	0.033
(b) Controls only for geographic characteristics and 2000 forest cover						
Share PES	0.2419***	0.1016***	-0.0391*	0.0457	-0.0112	0.0162
	(0.0722)	(0.0377)	(0.0203)	(0.1954)	(0.0368)	(0.0167)
Share Strict PA	0.0508	-0.0894***	0.0128	0.0311	-0.0527*	0.0331
	(0.0718)	(0.0318)	(0.0365)	(0.2102)	(0.0275)	(0.0253)
Share Biosphere Reserve	0.3440***	0.0361	0.0137	0.0772	0.022	-0.0156*
	(0.0744)	(0.0365)	(0.0100)	(0.1820)	(0.0291)	(0.0082)
Share Use PA	0.1567***	-0.0268	-0.0501***	-0.123	-0.0456	0.001
	(0.0571)	(0.0347)	(0.0127)	(0.1450)	(0.0319)	(0.0117)
N	59535	59535	59535	59535	59535	59535
R^2	0.333	0.09	0.05	0.125	0.049	0.027
(c) Controlling for predicted locality production revenues as a proxy for opportunity cost						
Share PES	0.2384***	0.1110***	-0.0416**	0.1572	0.0200	0.0082
	(0.0685)	(0.0336)	(0.0200)	(0.1075)	(0.0366)	(0.0162)
Share Strict PA	0.0519	-0.1061***	0.0197	0.1678	-0.0302	0.0201
	(0.0714)	(0.0327)	(0.0387)	(0.1150)	(0.0295)	(0.0244)
Share Biosphere Reserve	0.3406***	0.0429	0.0118	-0.0238	0.0100	-0.005
	(0.0730)	(0.0380)	(0.0109)	(0.1029)	(0.0293)	(0.0134)
Share Use PA	0.1617***	-0.0388	-0.0489***	0.0231	-0.0259	-0.0138
	(0.0560)	(0.0415)	(0.0136)	(0.0900)	(0.0314)	(0.0120)
N	59535	59535	59535	59535	59535	59535
R^2	0.338	0.193	0.059	0.63	0.232	0.201

Table A3b: Robustness checks with different samples or controls

	Outcomes			Pre-trends		
	Forest change	**Poverty alleviation**	**Population growth**	**Forest change**	**Poverty alleviation**	**Population growth**
(d) 20% baseline forest cover						
Share PES	0.2499***	0.1157***	-0.0276	0.0226	-0.0132	0.019
	(0.0672)	(0.0353)	(0.0197)	(0.2067)	(0.0359)	(0.0165)
Share Strict PA	0.0282	-0.1114***	0.0073	-0.0782	-0.0301	0.0045
	(0.0923)	(0.0425)	(0.0376)	(0.2319)	(0.0295)	(0.0233)
Share Biosphere Reserve	0.3814***	0.0375	0.0032	0.0179	0.0323	-0.0172*
	(0.0919)	(0.0400)	(0.0114)	(0.2052)	(0.0345)	(0.0097)
Share Use PA	0.1548**	-0.0559	-0.0473***	0.005	-0.0523	-0.0005
	(0.0628)	(0.0480)	(0.0116)	(0.1548)	(0.0374)	(0.0148)
N	42056	42056	42056	42056	42056	42056
R^2	0.365	0.193	0.058	0.126	0.049	0.028
(e) Matched on 93-00 deforestation, locality area, and baseline % forest within treatment types						
Share PES	0.1927***	0.1038***	-0.0202	-0.1781	0.0134	0.0143
	(0.0610)	(0.0345)	(0.0199)	(0.1812)	(0.0350)	(0.0168)
Share Strict PA	0.020	-0.1103***	0.0245	-0.039	-0.0292	0.0203
	(0.0818)	(0.0380)	(0.0353)	(0.2221)	(0.0294)	(0.0268)
Share Biosphere Reserve	0.3688***	0.0132	0.0105	0.1243	0.0282	-0.0122
	(0.0675)	(0.0345)	(0.0105)	(0.1665)	(0.0282)	(0.0093)
Share Use PA	0.1756***	-0.0302	-0.0532***	-0.2372*	-0.0355	-0.0007
	(0.0515)	(0.0392)	(0.0142)	(0.1439)	(0.0300)	(0.0127)
N	18052	18052	18052	18052	18052	18052
R^2	0.374	0.188	0.063	0.127	0.045	0.047
(f) Dropping smallest 10% of localities						
Share PES	0.2513***	0.1037***	-0.0327**	0.1124	-0.0262	-0.003
	(0.0716)	(0.0379)	(0.0152)	(0.1588)	(0.0366)	(0.0111)
Share Strict PA	0.0326	-0.1068***	0.005	0.0209	-0.0316	0.0457*
	(0.0730)	(0.0310)	(0.0364)	(0.2106)	(0.0291)	(0.0252)
Share Biosphere reserve	0.3578***	0.0496	0.0024	0.0946	0.0217	-0.0092
	(0.0733)	(0.0382)	(0.0095)	(0.1596)	(0.0291)	(0.0078)
Share Use PA	0.1630***	-0.0325	-0.0328***	-0.133	-0.0417	-0.0008
	(0.0568)	(0.0422)	(0.0109)	(0.1490)	(0.0329)	(0.0108)
N	53583	53583	53583	53583	53583	53583
R^2	0.338	0.189	0.067	0.128	0.048	0.036

* p< .10 ** p < .05 *** p < .01 Robust standard errors, clustered by municipality. Specifications described in the Appendix text.

Table A4: Robustness checks about timing of treatment for Parks and PES

Dependent variable	Forest change (2000-2012)	Poverty alleviation (2000-2010)	Population growth (2000-2010)	Forest change (2000-2012)	Poverty alleviation (2000-2010)	Population growth (2000-2010)	Forest change (2000-2012)	Poverty alleviation (2000-2010)	Population growth (2000-2010)
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)
Share PES	0.2379***	0.1121***	-0.0420**	0.2343***	0.1109***	-0.0416**			
(2003-2010)	(0.0683)	(0.0336)	(0.0202)	(0.0683)	(0.0336)	(0.0201)			
Share PES each							0.0431***	0.0206**	-0.0074
year * years paid							(0.0152)	(0.0088)	(0.0050)
Share Strict PA	0.0541	-0.1123***	-0.0202						
(2000)	(0.0677)	(0.0378)	(0.0417)						
Share Bios. Reserve	0.3907***	0.0419	-0.0001						
(2000)	(0.0822)	(0.0391)	(0.0116)						
Share Use PA	0.1597***	-0.0423	-0.0482***						
(2000)	(0.0600)	(0.0475)	(0.0172)						
Share New PA	0.1962**	-0.0118	0.0362						
(2000-2010)	(0.0799)	(0.0536)	(0.0327)						
Share Strict PA				0.0425	-0.0972***	-0.0096			
(2005)				(0.0721)	(0.0348)	(0.0385)			
Share Bios reserve				0.3722***	0.0372	-0.0018			
(2005)				(0.0802)	(0.0383)	(0.0115)			
Share Use PA				0.1732***	-0.0336	-0.0446***			
(2005)				(0.0577)	(0.0455)	(0.0166)			
Share New PA				0.2510***	-0.0118	0.0349			
(2005-2010)				(0.0807)	(0.0706)	(0.0415)			
Share Strict PA							0.0559	-0.1043***	0.019
(2010)							(0.0716)	(0.0327)	(0.0387)
Share Bios Reserve							0.3492***	0.0467	0.0103
(2010)							(0.0725)	(0.0378)	(0.0109)
Share Use PA							0.1644***	-0.0376	-0.0494***
(2010)							(0.0560)	(0.0415)	(0.0136)
R^2	0.338	0.193	0.059	0.339	0.193	0.059	0.338	0.193	0.059
N	59535	59535	59535	59535	59535	59535	59535	59535	59535

* p< .10 ** p < .05 *** p < .01 Robust standard errors, clustered by municipality. Same controls as Table 3 in main text. Columns 1-3 use the share in old parks by the year 2000 for specific categories and in new parks established between 2000-2010 for any category; Columns 4-6 use the share in old parks by the year 2005 for specific categories and in new parks established between 2005-2010 for any category; Columns 7-9 retain the original definitions of shares for parks but measure the PES treatment variable as the share of each area in PES in a given year before 2010 x the number of years payments were received.

Table A5: Weighted and un-weighted regressions

Dep. variable:	Forest change (2000-2012)					Poverty alleviation (2000-2010)				Population growth (2000-2010)			
Sample	Full	Full	Full	99%	99%	Full	Full	99%	99%	Full	Full	99%	99%
Weights	None	Locality area	Forest area	None	Forest area	None	Pop	None	Pop	None	Pop	None	Pop
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)	(11)	(12)	(13)
Share locality in PES 2010	0.238***	0.171**	0.202**	0.233***	0.317***	0.111***	0.096**	0.112***	0.080**	-0.042**	-0.960**	-0.041**	-0.063
	(0.069)	(0.083)	(0.100)	(0.069)	(0.078)	(0.034)	(0.048)	(0.033)	(0.036)	(0.020)	(0.444)	(0.020)	(0.076)
Strict PA share	0.052	-0.082	-0.140	0.054	-0.205	-0.106***	-0.192***	-0.103***	-0.099***	0.020	-0.195	0.051	0.044
	(0.071)	(0.113)	(0.164)	(0.071)	(0.140)	(0.033)	(0.059)	(0.033)	(0.028)	(0.039)	(0.674)	(0.034)	(0.090)
Biosphere reserve share	0.341***	0.653***	0.799***	0.296***	0.342***	0.043	0.053	0.042	0.024	0.012	0.053	0.011	0.018
	(0.073)	(0.124)	(0.124)	(0.074)	(0.069)	(0.038)	(0.040)	(0.038)	(0.039)	(0.011)	(0.147)	(0.011)	(0.040)
Use PA share	0.162***	0.218***	0.285***	0.159***	0.147**	-0.039	-0.188***	-0.037	-0.042	-0.049***	-0.565	-0.046***	-0.110*
	(0.056)	(0.069)	(0.102)	(0.057)	(0.057)	(0.041)	(0.047)	(0.042)	(0.029)	(0.014)	(0.376)	(0.012)	(0.063)
Covariates	yes	yes	yes	yes	yes	yes	yes	yes	yes	yes	yes	yes	yes
N	59535	58876	59535	58479	58479	59535	59535	59288	59288	59535	59535	59288	59288
r2	0.338	0.375	0.411	0.336	0.404	0.193	0.411	0.193	0.282	0.059	0.426	0.063	0.074

* p< .10 ** p < .05 *** p < .01 Robust standard errors, clustered by municipality. Table shows the results of weighting the data to account for potential heterogeneity in impacts (e.g. Solon, Haider, and Wooldridge 2015). Columns 1, 6, and 10 repeat the estimates from Table 3 columns 3, 6, and 9. Columns 2, 3 and 5 weight observations by the area of the locality and by the area of baseline forest cover (2000). Columns 4 and 5 drop the top 1% of outliers in terms of forest area size. Columns 7. 9, 11 and 13 weight observations by the locality population at baseline (2000). Columns 8, 9, 12 and 13 drop the top 1% of outliers in terms of population size.