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

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Forthcoming, Journal of Environmental Economics and Management
This version: Dec 2016

Keywords: protected areas, payments for ecosystem services, incentive-based regulation, poverty and environment, economic development, conservation policy, land use, deforestation, REDD

JEL codes: Q56, Q28, Q58, I38, O13

Acknowledgements: We gratefully thank the Mexican National Forestry Commission; AC RAs Leah Fine, Gus Greenstein, James Liu, and Bonnie Drake; UW Graduate student Carlos Ramirez Reyes; Allen Blackman, Alex Pfaff, Paul Ferraro, Volker Radeloff, Paulo Quadri, members of our departments, and participants at workshops at Harvard University, University of Michigan, Resources for the Future, University of Minnesota, FLARE, BIOECON, and the AERE summer meeting. We thank NSF Grant #1061852, the International Initiative for Impact Evaluation (3ie), the Carnegie Corporation of NY, and our institutions for funding support.

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 had overall neutral impacts on 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 that both direct and incentive-based instruments can be effective, and that policies combining sustainable financing, flexible zoning, and recognition of local economic goals are more likely to achieve conservation without harming 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 (IPCC 2013, Kerr 2013, Pfaff, Amacher and Sills 2013, REDD Desk 2016). Yet in a majority of countries, people also live on the lands important for efforts to reduce emissions from deforestation and forest degradation ("REDD" initiatives). Policymakers working to reduce emissions thus 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) with measures of poverty and population density from 2000-2010 for each locality (Mexican census; CONAPO). We evaluate the impacts of all protected area types in Mexico, both as a group and broken into categories of strict PAs, mixed-use PAs, and biosphere reserves. We compare these estimates to the impacts of Mexico's Federal Payments for Ecosystem Services program, a voluntary conditional cash transfer program that provides payments to selected private and community landowners in exchange for their maintenance of existing land cover.

Our central goal is to compare the performance of all PA types versus PES in the most recent decade, when both policies were key conservation measures. Our identification strategy compares localities with a higher share of land protected by PAs or PES by 2010 with similar localities that did not have these policies during this period. We use differenced outcomes to eliminate level effects and include controls for pre-trends in each outcome, state fixed effects, and multiple geographic and social criteria that determined selection into PAs or PES and could influence outcomes. While this strategy facilitates direct comparison of the two policies, it may still leave out possible unobserved confounders. Because data limitations preclude using a panel analysis that exploits years prior to 2000, 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. In order to separate the effects of conservation policy type from potential differences in outcomes due to the geographic distribution of the policies, we also estimate marginal impacts across comparable levels of deforestation risk. Finally, we test for possible complementarity of the policies and compare their 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 locality protection resulting in an estimated 20-25% reduction in the predicted loss of forest cover in a locality. With respect to development goals, the data show that PES led to small but statistically significant poverty alleviation—a 10-12% increase in the locality poverty alleviation index. Parks as a group had no effect on locality poverty trends, although strict protected areas showed significantly less poverty alleviation than the counterfactual trend. Importantly, none of the park types or PES increased poverty according to absolute measures: localities with conservation policies showed improvements on average across all basic poverty indicators.

Comparing the park types—biosphere reserves, strict PAs and mixed-use PAs—with each other and with PES yields three lessons. First, there are apparent tradeoffs across the individual policies with respect to achieving different social goals. Biosphere reserves, which combine strictly protected core areas with mixed-use buffer zones, generated the most avoided deforestation while PES resulted in the most poverty alleviation. This tradeoff matches theoretical expectations, outlined in the next section, that PAs can offer more complete environmental protection while PES is more likely to alleviate poverty. Second, we find that PES and the biosphere reserves were more successful than strict or mixed-use PAs in achieving environmental conservation without harming livelihoods. Both PES and the biosphere reserves differ from other PA designations because they provide a flexible but still well-defined and enforced approach to conservation. Both also received more substantial financial support during this period. Third, we find that all policies generated avoided deforestation from a mix of low and high cost land. This indicates that there was no clear winner between PAs vs. PES with respect to cost-effectiveness. Overall, our findings suggest 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 PAs and PES

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. Protected areas are a form of direct, involuntary regulation: they work by prohibiting specific agricultural or extractive activities. Backlash against protected areas due to restrictions that may conflict with local livelihood goals (e.g. Adams et al. 2004, West et al. 2006, Brockington et al. 2006) has led conservationists and governments to explore alternative policies. These include more flexibly zoned protected area types and payments for ecosystem services (Ferraro 2001, Ferraro and Simpson 2002, Jack et al. 2008, Wunder et al. 2008, 2014, Pechacek et al. 2013). In contrast to PAs, payments for ecosystem services are incentive-based and generally voluntary: they provide compensation to willing landowners conditional on maintaining a defined land use or fulfilling specific management activities. In addition to being more politically feasible, payments for ecosystem services are an incentive-based mechanism. A central theoretical and empirical finding in the literature on pollution control is that incentive-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).

As they do not suffer from the same self-selection challenge as PES and can cover large contiguous areas, protected areas may, in theory, be sited on lands with either relatively high or low opportunity cost ($c \leq \bar{p}$ or $c \geq \bar{p}$). Prior literature has shown that, empirically, parks tend

[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.

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, many PAs have still been established in high-risk areas, and risk frontiers often move with time. Thus, the actual avoided deforestation generated by PES or PAs in a specific context will depend upon whether each policy is able to enroll land at high risk of deforestation and whether it is 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. PES are voluntary, so barring substantial informational asymmetries or surprise events related to the costs of participation, they 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, Lewis, Hunt and Plantinga 2002, Robalino 2007). 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). However, 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 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 track both social and environmental impacts, assess heterogeneous effects, and compare policy costs.

To date, conservation evaluation has 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 2015), Indonesia (Gaveau et al. 2009, Schwarze and Juhrbandt 2010, Miteva et al. 2015) and Russia (Jones and Lewis 2015). Most studies find that protecting land at higher risk of deforestation produces more avoided 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 and Gutiérrez-Carbonell 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, prior research shows mixed results, with improvements due to PAs found in Costa Rica, Thailand, Uganda, Bolivia, and Indonesia (Andam et al. 2010, Sims 2010, Naughton-Treves et al. 2011, Canavire-Bacarreza and Hanauer 2013, 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 2000 to 2010.

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. Prior studies suggest small but positive poverty reduction in Mexico (Alix-Garcia et al. 2015) and small or neutral livelihood impacts in Costa Rica (Robalino et al. 2014, Arriagada et al. 2015). Research 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, retrospective effects of PA and PES, direct 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 human well-being 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

We face the standard empirical challenge in conservation evaluation, which is that PAs and PES were not randomly sited.[2] In addition, we seek to evaluate and compare two policies with different time frames for establishment and different selection criteria. In light of these constraints, our empirical strategy relies on comparisons of changes in outcomes in the 2000's between localities with different shares of land protected. Specifically, 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-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. Given remaining concerns about unobservable confounders, 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

[2] We experimented with several potential instrumental variables to predict park location based on biodiversity indicators and historical relationships between the Agrarian Reform and the goals of the Cardenas administration (1934-1940). Unfortunately, none had sufficient first stage power.

$\boldsymbol{IHS}(\boldsymbol{\Delta Y}_{imj,1990s})$ is a vector of transformed locality-level values of the pre-trend for each outcome.[3] $\boldsymbol{X}_{imj,2000}$ is a vector of other locality-level covariates related to selection into PAs or PES and likely to influence outcomes. 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. Standard errors are clustered at the municipality level to account for spatial correlation. The next sections explain in more detail the policies of interest and relevant selection criteria, the outcomes data, pre-trend data, and our risk of deforestation index.

Protected areas

Mexico's protected areas system includes federal, state, and locally-designated protected areas. We analyze all available mapped protected areas created before 2010 (Table 1), which together cover 12% of terrestrial territory. Protected areas vary substantially in terms of their stringency, location, and coverage area (Figure 1 and Table 1). Approximately 80% of protected areas are federally managed (WDI 2014, Bezaury-Creel and Gutiérrez-Carbonell 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 (see 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 and Gutiérrez-Carbonell 2009, CONABIO 2012, Chavez 2012, CONANP 2012, 2014). As is common around the world, the legal rules of protected areas

[3] 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 one (from no to full protection).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.

in Mexico often conflict with actual use. Of the localities in our dataset with more than 90% of land in a protected area, 41% of land is under common property ownership. While by constitutional right the government can limit the use of resources on any private property, it faces real challenges in enforcing restrictions, particularly if they limit subsistence use.

Many of Mexico's protected areas have existed for decades (the earliest in 1876), although coordinated management and funding increased dramatically starting in the late 1990's (CONANP 2014). Early parks—mainly established in the 1930's—tended to be close to population centers and focused on watershed conservation and recreational or educational opportunities for urban residents (Wakild 2011, Simonian 1995). A second major push for protection in the 1970's and 80's tripled the area under protection and included the establishment of many of the biosphere reserves (calculations based on data in Bezaury-Creel and Gutiérrez-Carbonell 2009). Unlike the earlier national parks, biosphere reserves were targeted to protect relatively pristine landscapes representing unique and biodiverse ecosystems (Simonian 1995). Biosphere reserves are flexibly zoned, combining core areas with strict protection and buffer zones that allow sustainable use. 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 (Ruiz-Mallén et al. 2015).

Despite the increase in area, Mexican PAs in the 1980's and early 1990's still suffered from lack of resources and a divided management regime. This situation improved in the 1990's with the establishment of CONANP (the National Commission of Natural Protected Areas) and substantial increases in funding.[4] 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 and Gutiérrez-Carbonell 2009). Total spending in 2008 was

[4] 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 and Gutiérrez-Carbonell 2009, p 402-409) while the area protected rose to approximately 7% of land by 2000 (World Development Indicators 2014).

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.[5]

Factors influencing the siting of the protected areas depend on the type of park and the timing of establishment. As mentioned above, biosphere reserves tended 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.

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 as well as smaller programs to support commercial plantations, reforestation, or community forestry.[6] 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 et al. 2015). We include payments from the three major

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

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

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.

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 Yañez-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 Yañez -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 in 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 that 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, measurement of outcomes and pre-trends

Our unit of analysis is the locality, the smallest administrative unit in Mexico. As the 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.[7] 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.[8]

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 data source providing comprehensive wall-to-wall analysis of forest change during this period.[9] 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

[7] 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.

[8] Prior work (Alix-Garcia, Shapiro and Sims 2012) indicates both substitution and output price slippage in deforestation due to the 2004 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.

[9] 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.

locality gained forest, negative if it lost forest. There are inherent limitations with the Hansen et al. data and it is most likely that our estimates of impact are conservative because the data is likely to understate true forest loss.[10] We provide several alternate specifications of this outcome measure as robustness checks in the Appendix.

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[11]). 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 less poverty. We also analyze impacts on each of the individual indicators that are common across years, using 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). We include population density growth as an outcome because of heated prior debates about the effects of parks on population trends (e.g. see Wittemyer et al. 2008 and response letters) and to test whether poverty alleviation impacts might be explained by migration.

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 1993-2000. 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

[10] The data is likely to understate loss of natural forest because it may classify plantations and agroforestry crops as forested areas (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. Because Hansen's data counts forest loss due to timber harvest in addition to forest loss due to conversion to agriculture, it could also overstate apparent deforestation in sustainably managed areas, but this problem is likely to be smaller. Because 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.

[11] 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

and 2000.[12] We do not use this data for 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 in treatment across time within localities and truly comparable outcomes data—neither of which is available in this case.[13]

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 gives five year contracts, that means in both cases our policy variables sometimes reflect partial treatment (the most conservative option). In addition, we believe that using the share protected at any point before 2010 correctly conceptualizes PA status as an ongoing treatment, not a one-time event. This reflects the reality of continuing pressure on natural resources and the need to continuously monitor and enforce protected area regulations. In addition, the large increase in investment in parks in the past decade described above may have led to significant impacts during the 2000's of PAs established before that. Yet these impacts will only be reflected in our estimates if we measure policy as the total cumulative share of parks in each locality. In other words, 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 and compare them with the impacts of PES in the same time period. In fact, differencing the outcomes from 2010-2000, including controls for pre-trends, and

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

[13] 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. 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.

incorporating 2000 levels of the outcomes removes pre-2000 level effects from the estimates of PA impact.[14] To test robustness to this choice, we include additional analysis separating parks into those established before and after 2000, and scaling the PES treatment by number of years treated (online Appendix).

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

There is considerable similarity in the types of places where PAs and PES have been applied in Mexico: 15% of localities with some protection have both types and there is substantial overlap in geographic characteristics. At the same time, there are still important differences in their geographic distribution due to history and political feasibility. 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

[14] Lewis, Hunt, and Plantinga (2002) employed a similar strategy in their analysis of the employment effects of public conservation lands in the U.S. We recognize, however, that protected areas established before 2000 may have had prior effects on forest cover or livelihoods that we do not measure. Interestingly, work by Blackman, Pfaff and Robalino (2015) suggests that federal parks did not actually have substantial impacts in the 1990's.

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.[15]

3. RESULTS

Summary statistics

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 a 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.

Avoided deforestation impacts of conservation policies will depend on management and enforcement but also on whether protected lands are at risk of deforestation. 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 by 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 that 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 characteristics have a less clear theoretical relationship with

[15] 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.

deforestation but were associated with higher deforestation risk 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). 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 and Figure 4 present our main estimates of the impacts of PES and PAs during the 2000's. We include specifications with no controls, full controls, and full controls plus state fixed effects. Our preferred specifications, which correspond to equation 1, are shown in bold text. 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.[16] The total land area in the localities analyzed with at least 5% 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

[16] Alix-Garcia, Sims and Yañez -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. Our previous outcome measure was NDVI, or “greenness”, which does not translate directly into forest area. However, estimations of the program’s impact using the outcome from this paper are similar (see appendix to Alix-Garcia, Sims and Yañez -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.

avoided deforestation to date. Yet the popularity of the PES program suggests that it could be scaled up to cover more land area.

Panel B of Table 3 separates 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 that 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 mixed-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.

To better understand the poverty alleviation results, we also examine the changes in individual components of the poverty index (Table 4). For these individual poverty components, the estimated impacts of PES are consistently negative, indicating possible reductions in poverty measures compared to counterfactual localities. PES significantly reduced illiteracy and the percent of people without access to electricity. 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.

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. While there is widespread agreement that international conservation should have a goal of "do no harm," it is not clear whether this should be assessed in terms of absolute changes or relative to the best available counterfactual. 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.

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, we would theoretically like to compare the marginal impacts of different park types with the same types of land (a point made by several others, e.g. Ferraro et al. 2013, Pfaff et al. 2014a). 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.

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 deforestation risk in Figure 5a also indicate that the biosphere reserves were more effective than other types of protection conditional on having the same risk level (although the confidence intervals for the marginal effects overlap). We also 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, conditional on deforestation risk.

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.[17]

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

[17] 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.

reserves may have been able to attract more funding than other protected areas, due to their high profile international status. For instance, Bezaury-Creel and Gutiérrez-Carbonell (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, but this is a key avenue for future research.

It is also possible that communities benefitted 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—a reasonable proxy for tourism potential of the parks (see Table 5).[18] These figures suggest that the strict protected areas actually collected the most revenue, so tourism 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 (Table 5). Biosphere reserves may have benefitted from intermediate levels of tourism combined with being located in lower opportunity cost locations (see additional tests and discussion in the online appendix).

Complementarity between PAs and PES?

An additional important question is whether there is possible complementarity between direct and incentive-based policy. 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, but the interaction terms are negative for forest cover and poverty alleviation and not significantly positive for population. For forest cover, the

[18] 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.

fact that the coefficient is close in magnitude to the main effect for PES indicates that PES would provide no additional avoided deforestation if provided to landowners already fully inside PAs.[19] Interestingly, this lack of environmental complementarity between PAs and PES echoes the results of Robalino et al. (2015) for Costa Rica.

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 results indicate possible positive complementarity near borders for forest cover (the interaction coefficient is 0.47 but not significant) and significant positive complementarity with respect to poverty alleviation (interaction coefficient is 0.556 and significantly different from zero).[20] 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.

Cost effectiveness

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 provide the budgetary expenditures for each program and an analysis of how the opportunity cost of land in each type relates to

[19] 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.

[20] 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.

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.[21] 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 costs of forgone land use, which depend on the forgone profits from the highest value non-forest use.[22] Data on production profits across the country is not available, so we create a proxy based on estimated locality-level production values. The construction of this variable and its limitations are discussed in the appendix. 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 forgone revenues, we show the distribution of predicted production revenues across avoided deforestation estimates for each policy based on land enrolled by 2010 (Figure 6). This figure was generated by using the coefficients from the regressions with polynomial interactions with deforestation risk (Figure 5) to predict the amount of avoided deforestation due to each policy in each locality.[23] Figure 6 thus traces out a log-transformed supply curve for each policy.

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 forgone production revenues is everywhere higher for strict protected areas, which makes sense given their locations closer to population

[21] The exchange rate across this period was approximately 11-12 pesos / USD on average.

[22] For an overview of conceptual issues in assessing conservation costs, see Naidoo et al. 2006. For estimated costs of carbon sequestration for the U.S., see Lubowski, Plantinga and Stavins 2006.

[23] We limit the y-axis at 4000 hectares because there is very little land enrolled with these high values.

centers and more frequently in the agriculturally productive center of the country. For PES, mixed-use PAs, and biosphere reserves, the distributions of forgone revenues are fairly similar – confidence intervals overlap for all of these categories. For all three policies, about 60% of the avoided deforestation comes from land of relatively low value: less than 800 pesos (~70 USD) of revenue per hectare. Most of the remaining avoided deforestation for these three policies (about 35%) comes from land in localities with estimated production values under 2000 pesos (~175 USD). Biosphere reserves, due to their substantially larger area, contribute more avoided deforestation at any level of revenue, but a somewhat greater share comes from land in localities with higher predicted production revenues. Although we should interpret the results with caution because production revenues are only a proxy for opportunity cost, Figure 6 illustrates an important point: PES is not necessarily more cost-effective simply because it is an incentive-based rather than command and control conservation mechanism.

4. ROBUSTNESS CHECKS

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

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. We find no significant differences ($p < .05$) in pre-trends for forest change, poverty, or population for any of the treatments (see Appendix). Next, we conducted a variety of robustness checks to assess sensitivity 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 using different measures of the forest cover variable (see 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 use the fact that changes in the coefficients of interest when new covariates are introduced provide information about the possible impacts of omitted variables (Murphy and Topel 1990, Altonji, Elder and Taber 2008, Oster 2013). The estimation of these impacts depends on the ratio of the covariance between the omitted variable and the treatment variable compared to the covariance between the observables and the treatment variable, or the "coefficient of proportionality." This is an unknown parameter about which assumptions must be made. 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).[24] Both are shown in Table 7, 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 one. 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 to inform 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 find that both conserved forest, with similar estimates of avoided deforestation by share of land protected. Localities with land in both PAs and PES also showed average absolute gains in all poverty indicators, although relative gains were highest in response to PES and biosphere reserves. We believe these results are good

[24] 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).

news for policymakers facing a choice between PAs and PES to achieve REDD, as they indicate that both can achieve conservation while not impoverishing local communities. The results also indicate that PAs deserve as much attention as PES when considering possible cost-effective conservation solutions. While the literature on direct versus incentive-based mechanisms for pollution control emphasizes the cost-effectiveness of market-based solutions, our findings illustrate that this result cannot be assumed to hold for land conservation.

Our analysis also illustrates likely tradeoffs inherent to PAs vs. PES and raises multiple questions for future work. Specifically, we find that a type of protected area—biosphere reserves—achieved more environmental impact, while payments for ecosystem services produced more poverty alleviation. Yet both PES and the biosphere reserves came closer to "win-win" solutions for forests and livelihoods than the other park types. Assessing what they have in common, we note that both PES and biosphere reserves explicitly recognize the need to improve local livelihoods. PES directly 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. Finally, both have received national and international attention and sustained funding in the past decade. These common elements suggest that it may be less important whether conservation instruments are direct or incentive-based than whether they are well-funded and combine enforceable protection with zoning that allows for 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

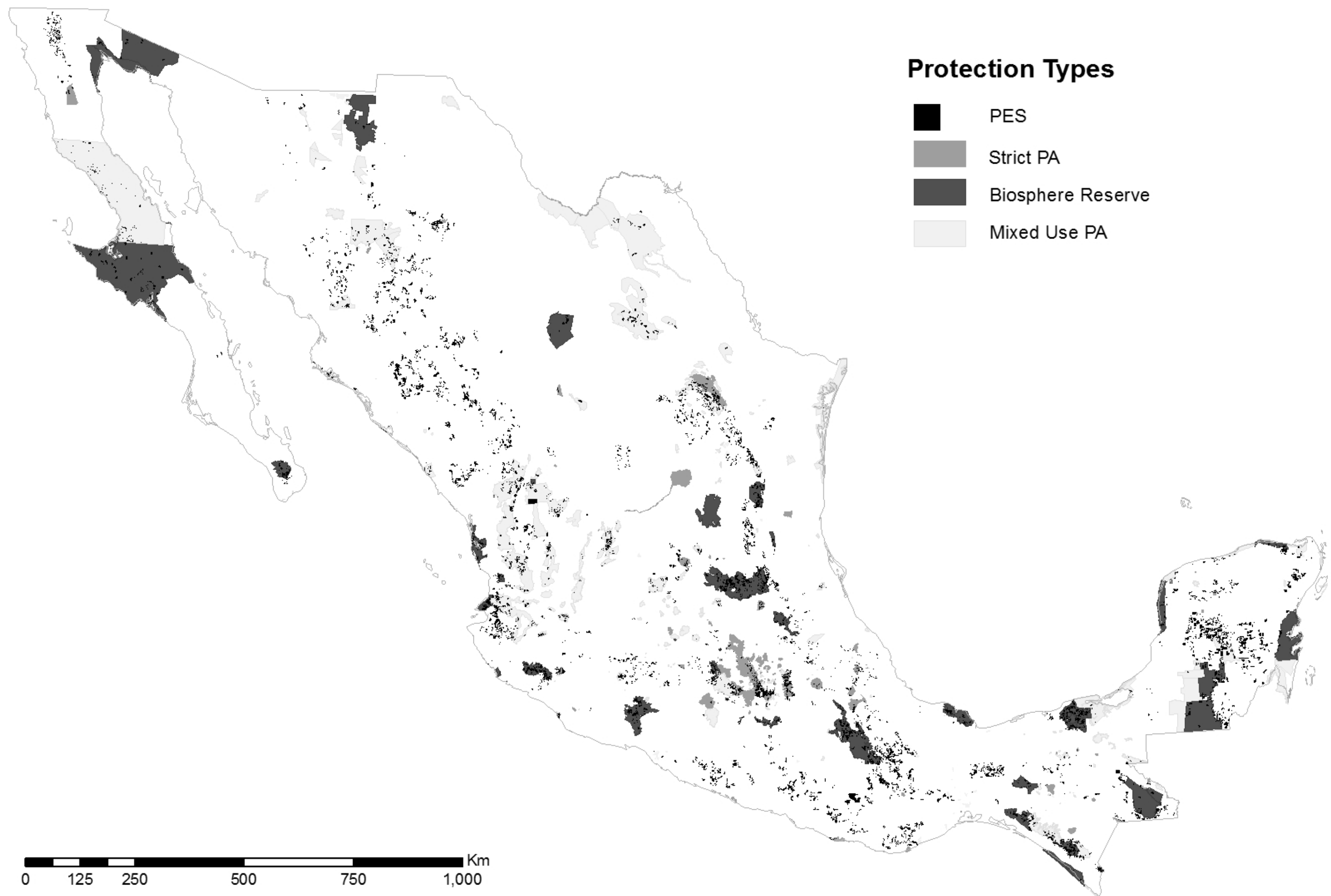
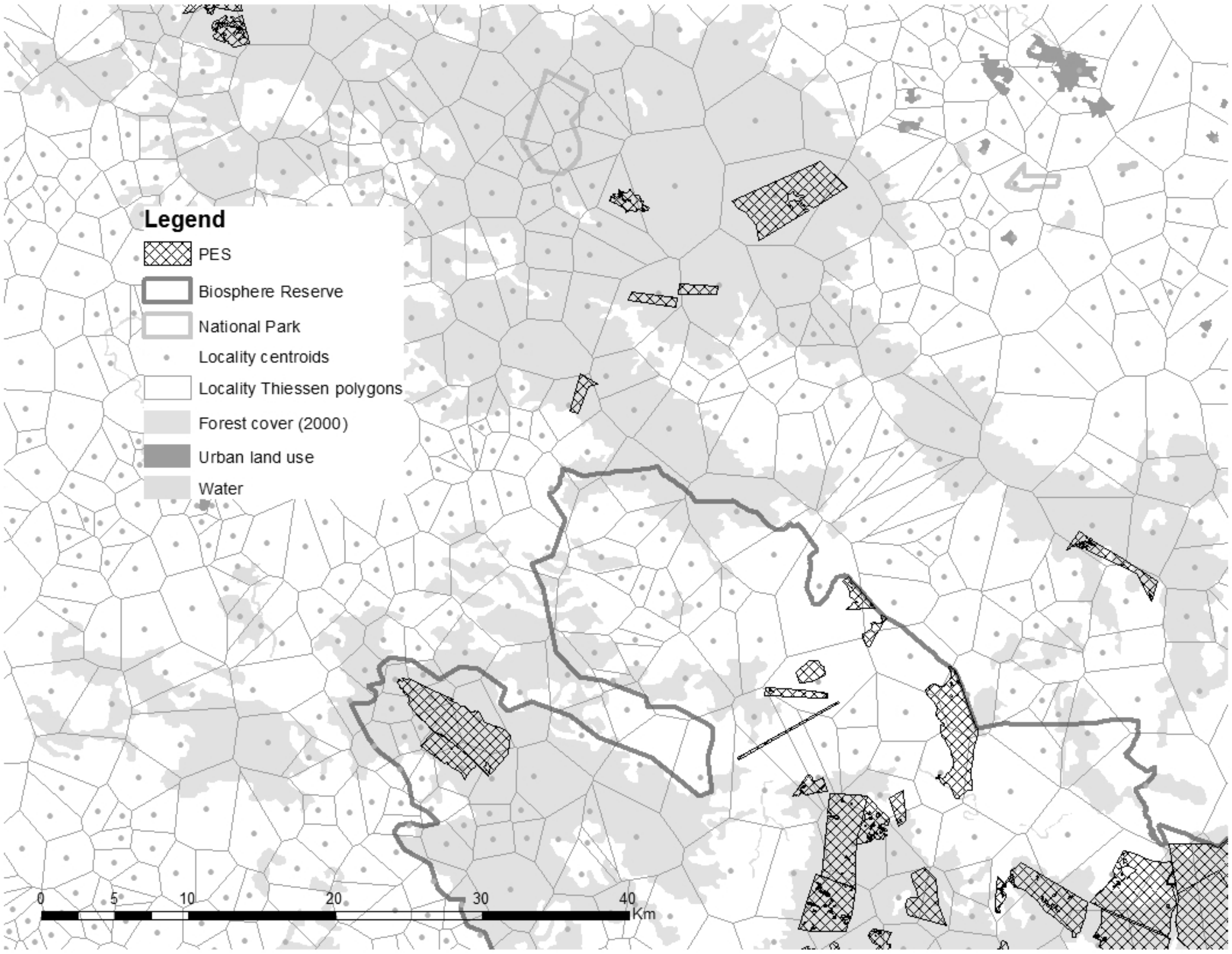
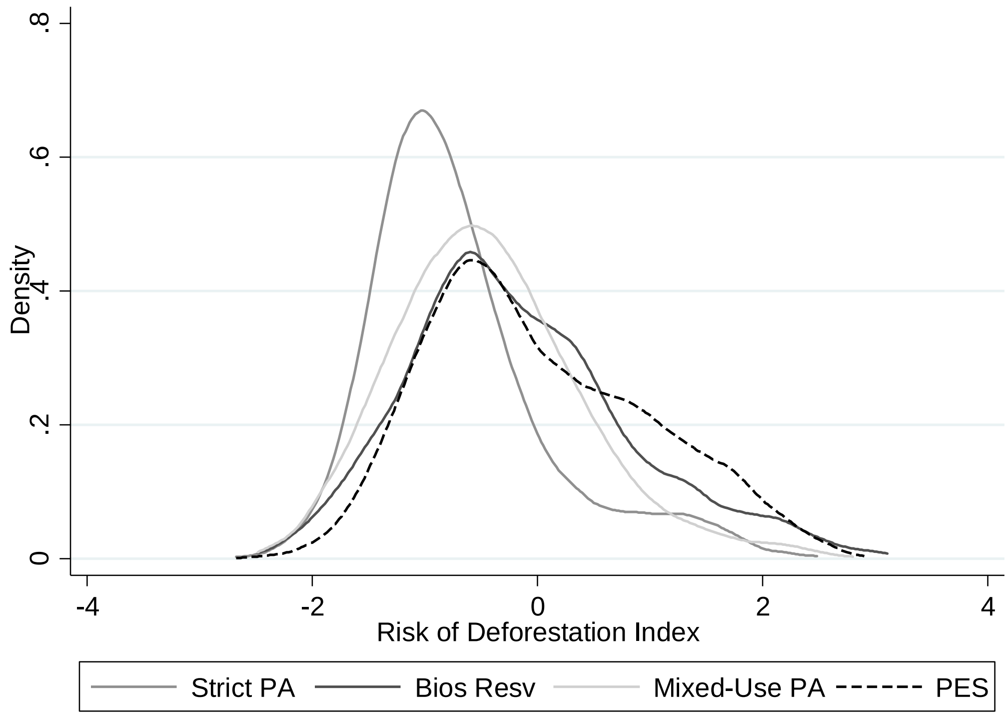


Figure 2: Illustration of locality Thiessen polygons



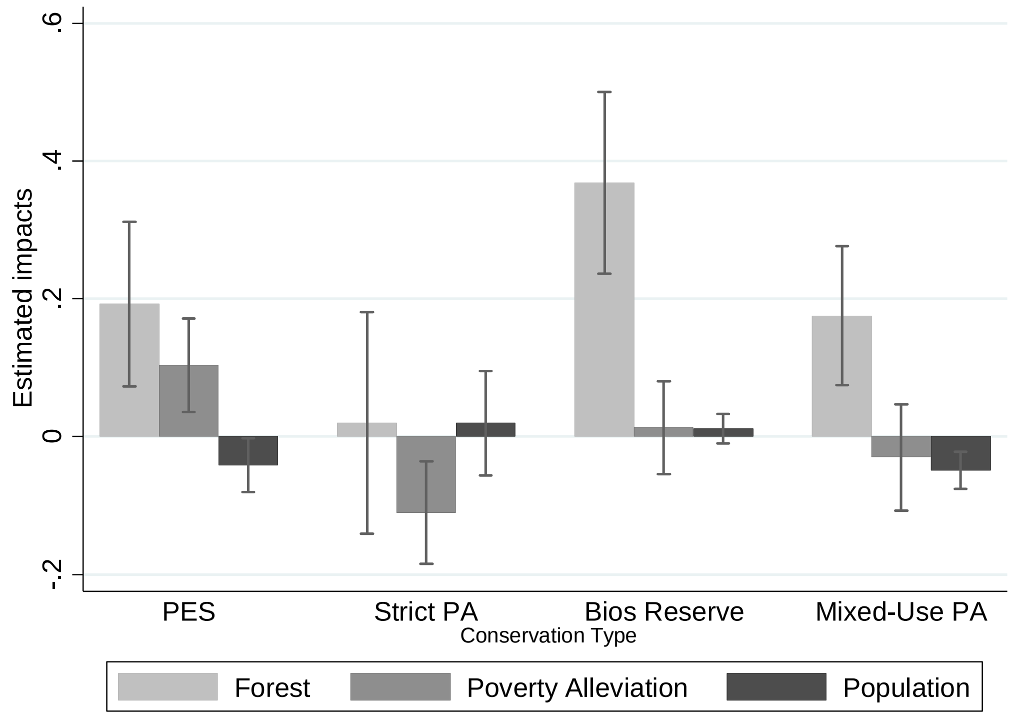
Dots indicate locality centroids; grey lines show Thiessen polygons for each locality (unit of analysis).

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



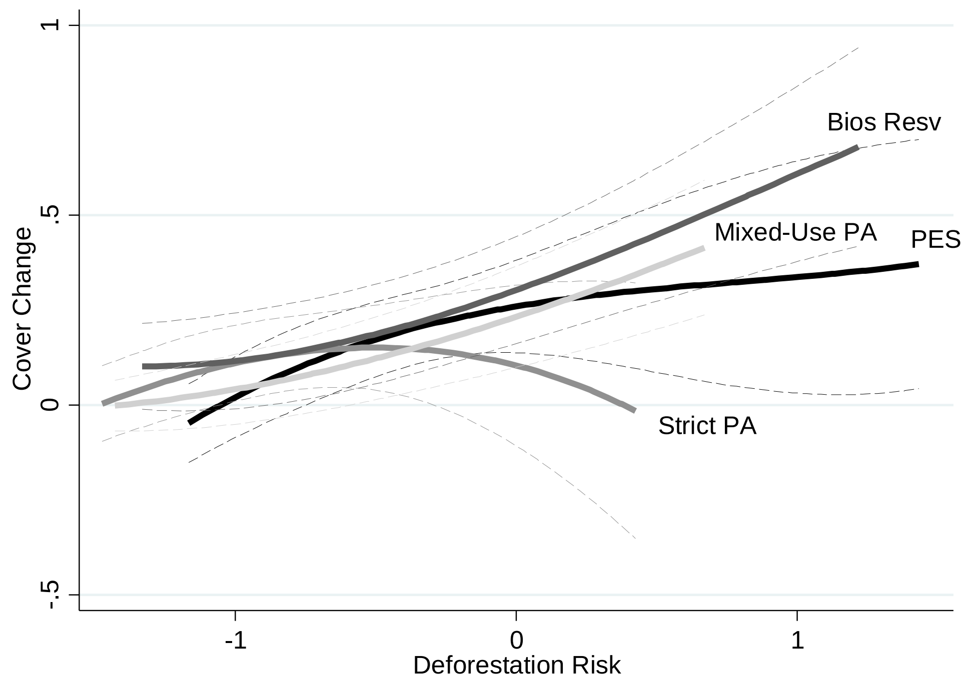
Kernel density distributions of risk of deforestation index by protection type (bandwidth = .2) Predicted risk of deforestation is based on net forest cover change for all non-treated localities as described in the text; index values are normalized to mean zero and SD = 1.

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

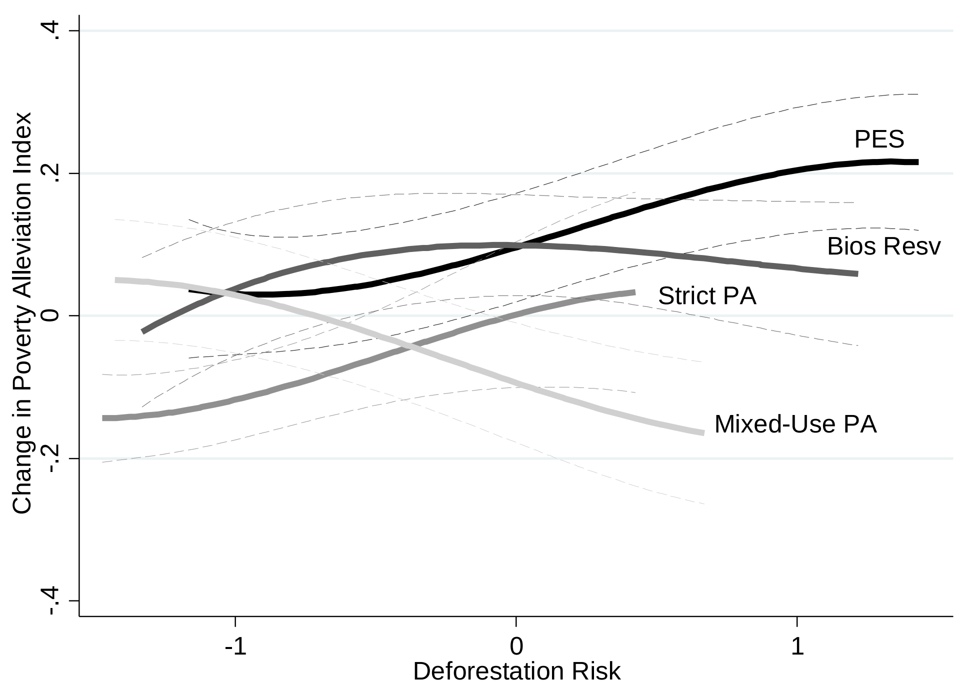


Coefficients and 95% confidence intervals from main specification (equation 1 and Table 3 columns 3, 6, 9). A "win-win-win" situation for forests, poverty alleviation, and population would have positive values for all estimated coefficients.

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

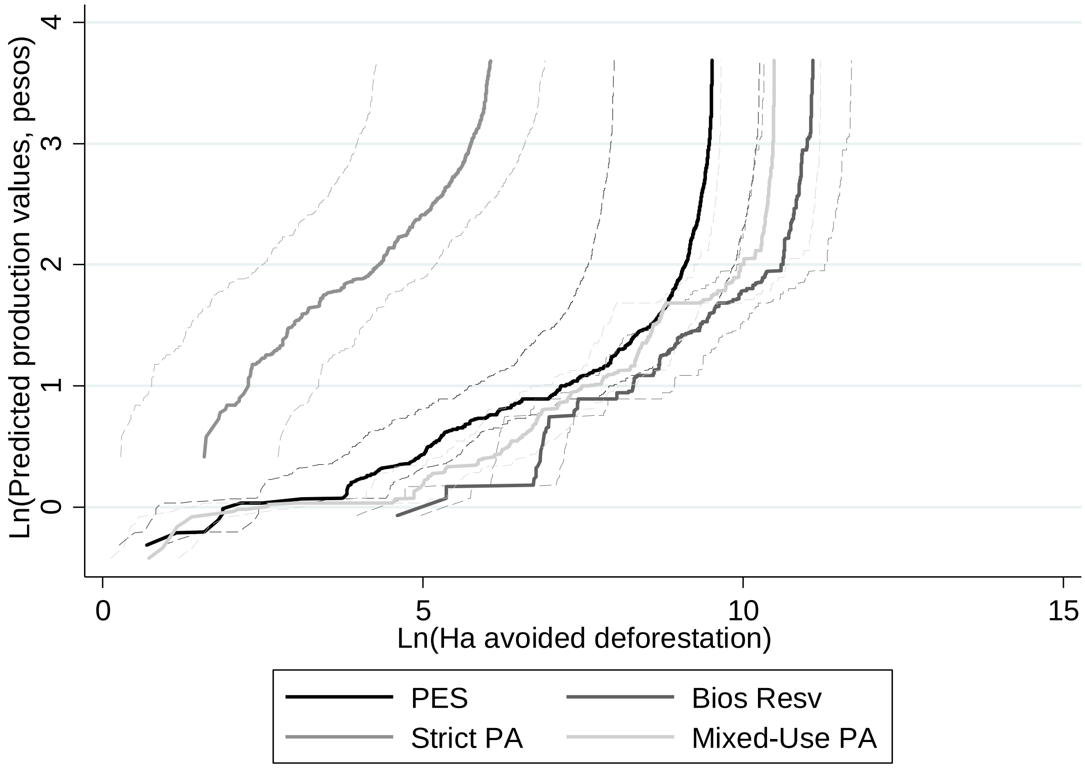


b: Change in poverty alleviation index



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

Figure 6: Locality production revenues vs. cumulative avoided deforestation



For each policy, graph shows the relationship between predicted locality production revenues and estimated avoided deforestation. Confidence intervals are from a clustered bootstrap procedure with 1000 iterations.

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. Municipal 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 PAs: 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 Reserve	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 Mixed-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
Mixed-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). The poverty index is -1*poverty alleviation index. 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 one 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: 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: Additional information and robustness checks

Test for parallel pre-trends

Table A1 shows results from regressions of the change in forest cover, poverty and population in the 1990's on the share in PES and PAs by 2010, with controls for all time-invariant covariates. There are no significant differences at the $p<.05$ level for any of the treatments. Strict PAs show marginal significance indicating slower growth in poverty alleviation in the pre-period; biosphere reserves show possibly meaningful forest protection (~7%) and slower population growth (~1.5%, marginally significant) in the pre-period. These motivate inclusion of controls for pre-trends and 2000 levels, as discussed in the main text.

Park revenues and heterogeneous impacts

In Table A2 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 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 forgone production.

Forest cover change robustness checks

Table A3 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 of 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 95^{th} and 5^{th} 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 A4 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 controls 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 A5. Columns 1-3 separate 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 to separate the new parks into all 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, separating 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 A5. 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 if we use different weights (e.g. Solon, et al. 2015). Table A6 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).

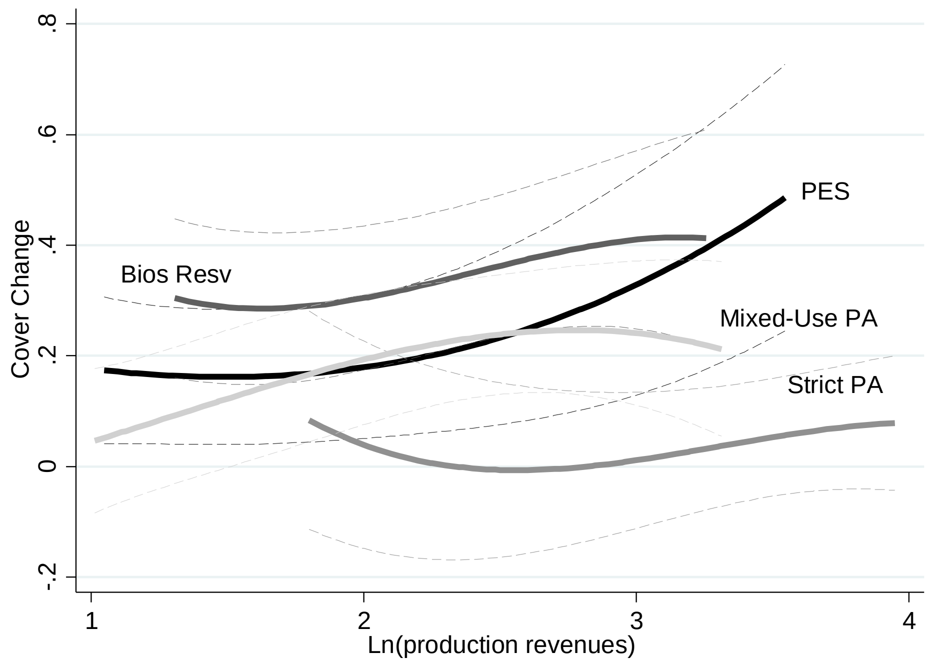
Overall, the results in Table A6 show that weighting can matter, particularly for the magnitude of the forest cover results for biosphere reserves, the population growth results, and the poverty impacts of mixed-use areas. However, the core results emphasized in the main text are similar across weighting schemes. In columns 1-5, we observe that when weighting by either locality area (Thiessen polygon) 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 localities 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. Mixed-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 mixed-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.

Construction of the estimated locality production values

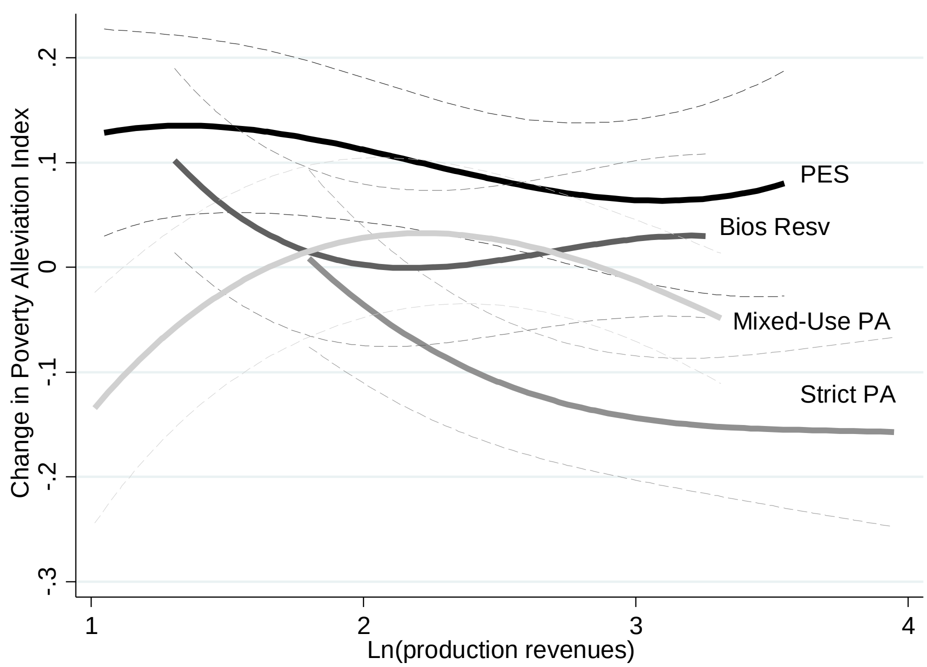
To create the locality predicted production revenues measure, we start with data on production revenues for principal crops and livestock at the municipal level from 2003, normalized by the non-forested area of the municipality (to give an approximate measure of revenues per hectare in agricultural or pastoral use).[25] 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 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.

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

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.

Appendix Table A1: 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 A2: 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 A3: 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 Mixed-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 A4a: 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 Mixed-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 Mixed-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 Mixed-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 A4b: 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 Mixed-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 Mixed-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 Mixed-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 A5: 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 Mixed-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 Mixed-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 Mixed-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 A6: 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)
Mixed-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.