University of Wisconsin-Madison
Department of Agricultural & Applied Economics

Staff Paper No. 548 August 2010

FOREST CONSERVATION AND SLIPPAGE: EVIDENCE FROM MEXICO'S NATIONAL PAYMENTS FOR ECOSYSTEM SERVICES PROGRAM

By

Jennifer M. Alix-Garcia, Elizabeth N. Shapiro and Katharine R. E. Sims

AGRICULTURAL &

APPLIED ECONOMICS

STAFF PAPER SERIES

FOREST CONSERVATION AND SLIPPAGE: EVIDENCE FROM MEXICO'S NATIONAL PAYMENTS FOR ECOSYSTEM SERVICES PROGRAM

Jennifer M. Alix-Garcia
Agricultural and Applied Economics
University of Wisconsin, Madison

Elizabeth N. Shapiro
Nicholas School of the Environment
Duke University

Katharine R.E. Sims
Department of Economics
Amherst College

WORKING PAPER
August 6, 2010

JEL Codes: Q12, Q24, Q57, R14, O13

Keywords: payments for environmental services, payments for ecosystem services, program evaluation, slippage, leakage, incentive-based mechanisms, Mexico, land use, deforestation

ABSTRACT

Incentive-based programs to reduce deforestation are expected to play an increasingly important role in global efforts to protect ecosystems and sequester carbon but their environmental effectiveness is not clear. We investigate program effectiveness and slippage in the context of Mexico's national payments for hydrological services program, which pays private and communal landowners to maintain forest cover on enrolled lands. To measure program impacts, we use matched controls drawn from the program applicant pool to establish counterfactual deforestation rates in the absence of payments. We find statistically significant but small to moderate avoided deforestation impacts. To examine slippage of deforestation to non-enrolled lands, we develop a model of household land allocation to agriculture or forest in which some households are credit-constrained. We illustrate that payments for forest conservation may result in slippage due to substitution, resulting in increased deforestation on other parcels belonging to program recipients, or due to output price effects, resulting in increased deforestation within markets where there are high levels of program participation. Our data show evidence consistent with slippage through both mechanisms. This suggests that incentive-based mechanisms can work to prevent deforestation but that avoided deforestation should be accounted for at a regional or national level in international schemes to reduce carbon emissions from deforestation and forest degradation.

Acknowledgements: We would like to thank the many people at the Comisión Nacional Forestal of Mexico (CONAFOR) who have helped us gather and interpret data, including but not limited to: Leonel Iglesias Gutiérrez, Sergio Graf, José Armando Alanís de la Rosa, Paola Bauche, Jesús Gutiérrez Cacique, Rodolfo Valdez Garcia, Silvia Martinez, Rigoberto Palafox Rivas, and Carmen Meneses Tovar. We also thank David Saah and Volker Radeloff for assistance with remote-sensing expertise, and Stefano Pagiola, Paul Ferraro, Sami Alpanda, Walter Nicholson, and participants at the 4th World Congress of Environmental and Natural Resource Economists for comments and suggestions. Finally, we would like to thank USAID-Translinks and the International Initiative for Impact Evaluation (3ie) for financial support.
Please send comments to: alixgarcia@wisc.edu, eshapiro88@gmail.com, ksims@amherst.edu.

Introduction

Payments for ecosystem services (PES) are an important form of incentive-based conservation worldwide (Landell-Mills and Porras 2002, Wunder 2007, Pagiola and Platais 2007, FAO 2007, Engel et al. 2008, Jack et al. 2008). These schemes, particularly payments for forest conservation in developing countries, are likely to expand dramatically under proposed international agreements to reduce carbon emissions from deforestation and degradation ("REDD" agreements, see IUCN 2009, UNFCCC 2009). However, there is little rigorous empirical evidence about the environmental effectiveness of existing PES programs (Pattanayak, Wunder and Ferraro 2010).

In the context of reducing deforestation or forest degradation, the goal of PES programs is to induce additional forest conservation ("additionality" or "avoided deforestation") by raising the returns to forested land (see e.g. Ferraro and Simpson 2002, Bond et al. 2009, Pagiola and Zhang 2010). Although this logic is theoretically sound, there is concern that in reality PES programs might not generate additional environmental benefits. One concern is that programs are not effective because they are paying landowners who would have kept land in forest in the absence of payments (e.g. see de Janvry and Sadoulet 2006, Alix-Garcia et al 2008b). A second concern is "slippage" or "leakage": even if forest conservation programs do induce additional conservation on enrolled lands, these benefits may be undermined by new deforestation in other locations (for previous discussions on slippage and leakage or other negative environmental spillovers in the context of land conservation see: Berck and Bentley 1997, Wu 2000, 2005, Wu, Zilberman and Babcock 2001, Chomitz 2002, Lichtenberg 2004, Murray et al. 2004, 2007, Roberts and Bucholtz 2005, 2006, Plantinga and Richards 2008, Lichtenberg and Smith-Ramirez 2010).

In this paper, we study program effectiveness and slippage for an early cohort of Mexico's National Payments for Hydrological Services Program (*Pago por Servicios Ambientales-Hidrológico/*PSAH*)*, which pays landowners to maintain forest cover. Between 2003 and 2009, approximately 2.27 million hectares of land were entered into Mexico's program of payments for ecosystem services (Shapiro 2010), making it one of the largest in the world next to the U.S. Conservation Reserve Program, Costa Rica's PSA program, and China's Sloped Land Conversion Program. We collect and analyze new data on the 2004 PSAH cohort.

We seek to contribute to the existing literature on payments for ecosystem services in two ways. First, we improve upon previous efforts to evaluate the environmental effectiveness of PES programs, by using parcel-level data and drawing control properties from the applicant pool to construct a plausible counterfactual comparison group. Most previous literature on payments for ecosystem services does not measure impacts in comparison to controls or is limited by concerns that the controls used differ significantly on observable and unobservable characteristics (Pattanayak et al. 2010). By using properties from the applicant pool, we ensure that control properties are similar with respect to key unobservable attributes including selection into treatment and the institutional capacity to apply. Rather than use all applicants, we use matching to select controls which are closest in terms of observable characteristics. We estimate impacts using the Abadie-Imbens (2002) bias-adjusted matching estimator or regression to correct for remaining observable characteristics which might be correlated with both program enrollment and deforestation outcomes. We find that the program has reduced the probability of deforestation by 6-11 percentage points, which represents an approximately 24-44 percent decrease in the probability of having any deforestation. Among deforesters, we see a decrease in area deforested of approximately 2-11 percent on average.

Second, we model and test for slippage of deforestation to other areas. In the context of forest-conservation payments, a substitution slippage effect occurs when a landowner who removes one parcel of land from production (enrolling it in the program) shifts the planned production to another parcel within his landholdings. An output price slippage effect occurs if the removal of multiple parcels of land from production or the introduction of payments alters market prices and these changes induce additional deforestation. Whether or not these changes will be spatially close to enrolled lands depends on the size of the relevant markets.

The theoretical basis for slippage through substitution or output price effects has been previously discussed in the literature (Wu 2000, 2005, Wu, Zilberman, and Babcock 2001, Chomitz 2002, Roberts and Bucholtz 2005, 2006). However, existing models are most relevant in the context of well-functioning factor markets and high-income countries. In contrast, future growth in incentive-based forest conservation is expected to occur mainly in developing countries, where credit constraints are common, markets are more localized due to poor infrastructure, and incomes are significantly lower. We therefore construct a simple theoretical model of household land allocation between agricultural and forest production in which some households face a constraint on purchases of a key agricultural input (such as fertilizer or seeds), a situation likely to be relevant in a developing country context. We demonstrate that for credit-constrained households, the introduction of a PES program may perversely increase deforestation in other locations within the household's property through a substitution effect.

We also illustrate that the program could increase deforestation in other locations outside of the household's property through output price effects. The standard channel for such an effect is through an increase in prices resulting from a decrease in supply of the agricultural good. We argue that changes in demand for the agricultural good may also be an important channel in a developing country context. Where baseline incomes are low, the program may result in increased demand for the agricultural good as incomes rise. Although output price effects can be expected to occur regardless of the size of the relevant market, these effects will be detectable only where markets are localized, for instance by poor road infrastructure.

In order to test for results consistent with these predictions, we return to the case of Mexico's program. We find evidence consistent with substitution effects within communal properties in remote areas, lending support to the credit-constraints hypothesis. We also see effects consistent with increased output prices in areas where markets are localized by poor road infrastructure.

The article begins by briefly describing previous literature on the environmental effectiveness of PES and our strategy for analyzing the effects of Mexico's payments for hydrological services program. We then present results from the impact evaluation. Having presented evidence showing a direct program impact, we then discuss previous literature on slippage and present our model of household land allocation. Finally, we test for evidence of slippage consistent with these hypotheses and conclude.

EVALUATING THE ENVIRONMENTAL EFFECTIVENESS OF PES

Previous literature on PES effectiveness

Previous literature addressing the environmental effectiveness of national PES programs indicates small or modest benefits of these programs. Much of this literature focuses on the U.S. Conservation Reserve Program, one of the earliest national-scale programs which directly incentivized individual landowners for conservation actions (Sullivan et al. 2004, Feng et al.

2005, Lubowski et al. 2006). Goodwin and Smith (2003) find significant reductions in soil erosion as a result of the U.S. CRP, using a structural model and county-level data. These effects may be undermined by slippage of production to other areas (see debate between Wu 2000, 2005; Roberts and Bucholtz 2005, 2006), increased production on environmentally vulnerable land (see Lichtenberg and Smith-Ramirez 2010 on evidence regarding conservation payments in Maryland), or increased environmental damage through intensified use of pesticides or fertilizers (Lichtenberg 2004). More optimistically, benefits may be enhanced if programs cause persistent changes in producer behavior (Roberts and Lubowski 2007). Previous studies on China's Sloped Land Conversion Program (Xu et al. 2004, 2005, Uchida et al. 2005) also indicate that programs in China have achieved additional significant soil conservation benefits on the basis of modeling using household surveys on participant behavior and targeting criteria.

Few previous studies directly estimate effectiveness using a comparison group to estimate what would have happened in the absence of the program (Pattanayak et al. 2010). To date, the only evaluations of national PES programs using direct control group comparisons that we are aware of are from Costa Rica (Sills et al. 2007, 2008, Arriagada et al. 2008, Pfaff et al. 2008, Robalino et al. 2007). These studies generally find little or no impacts, possibly because the overall rate of deforestation slowed in Costa Rica around the same time that the program was introduced. Mexico, in contrast, continues to experience significant rates of deforestation (FAO 2005). The small number of studies which discuss Mexico's national payments for ecosystem services program (Muñoz-Piña et al. 2008, Alix-Garcia et al. 2005, 2008a,b, Corbera and Brown 2008) discuss important debates and changes about targeting strategy but do not directly estimate program effectiveness by measuring deforestation and using matched controls. Clearly, additional research is needed which measures impacts relative to a realistic counterfactual. Other ongoing work in this vein includes a study by Honey-Roses et al. (2010) examining the case of private payments in Mexico around a monarch butterfly protection reserve. Preliminary results from that study also suggest effectiveness of conservation payments, particularly in protecting high quality forest habitat. Before describing our strategy for evaluating deforestation impacts using a counterfactual comparison, we briefly describe the administrative structure of Mexico's program of payments for hydrological services.

Mexico's PSAH Program

Our analysis focuses on Mexico's payments for hydrological services program (PSAH), which is designed to incentivize the increased production of hydrological services through forest conservation. Mexico's program focuses on the conservation of existing forest cover (avoided deforestation) which is likely to be a feature of future programs under international climate agreements (e.g. CONAFOR 2009). Under the PSAH program, five-year renewable contracts are signed with both individual and communal landowners. Payments are made annually and verification of forest cover through satellite image analysis or ground visits is conducted annually on approximately half of all enrolled properties (McAfee & Shapiro 2010). Lands where clearing is detected are removed from the program and payments are revoked. Payment rates were originally based on approximate calculations of the average opportunity cost of land conversion from forest to maize crops. Rates are fixed, with slightly higher per hectare payments for cloud forest than for other forest types. Participation is targeted to sites with a potential demand for hydrological services, i.e., in overexploited watersheds and upstream from population centers > 5,000 (Shapiro 2010).

MEASUREMENT AND EMPIRICAL STRATEGY

Deforestation indicator

Measuring deforestation at a national scale is difficult, which explains why previous analyses of Mexico's program have not directly measured deforestation on enrolled properties. We use data from a recently constructed annual forest cover change monitoring data.[1] This dataset covers all of Mexico and relies on MODIS satellite images (250 m). We use data from the period 2003-2006. The classification of deforestation is based on changes in the dry season Normalized Difference Vegetation Index (NDVI) values across time. Although NDVI change is the best available indicator of changes in forest cover, we caution that there may be errors in this indicator of forest loss because of weather shocks which change NDVI, because changes may reflect degradation rather than deforestation, or because very small areas of deforestation are missed. We note that these errors are in the dependent variable and are unlikely to be correlated with the treatment conditional on regional controls. Finally, since the mandate of the Monitoreo Forestal was to measure deforestation, our dependent variable is censored. We correct for this potential censoring problem by using tobit regression in the data analysis.[2]

We calculate deforestation within the boundaries of the parcels enrolled in the PSAH program and within the boundaries of the control properties. In order to measure localized spillovers, we also construct measures of deforestation for surrounding areas. Our analysis focuses on an early cohort (2004) of Mexico's program; we choose the 2004 cohort so that sufficient time has passed in order to reasonably measure deforestation rates.

---

[1] "Monitoreo Forestal de Mexico." Further description available at CONAFOR's website: http://148.223.105.188:2222/gif/snif_portal/index.php?option=com_content&task=view&id=2&Itemid=3

[2] Practically speaking, we do not believe that afforestation is a major concern. Mexico was not a net afforester during this period. In fact, FAO's 2005 Global Forest Resources Assessment places Mexico in 13th place in the world in terms of net forest loss over the period 2000-2005.

Constructing a comparison control group for impact analysis

Evaluation of the PSAH presents the standard identification problem which is that we do not know how recipients would have behaved had they not received payments. In order to construct a plausible counterfactual, we rely on a comparison to controls drawn from the applicant pool. We use two types of controls. First, we use applicants from 2004 that were rejected on the basis of administrative or geographic details. Second, we use recipients of payments in a future (2006) round of the program. We combine both the rejected parcels and the future enrollees as one category of “non-recipients.” Figure 1 illustrates the location of 2004 participants and controls.

Enrolled and control parcels from the applicant pool are likely to be similar with respect to key unobservable characteristics including selection into treatment and institutional capacity to apply. However, they may still be different with respect to other characteristics, which could bias estimates. Considering the rejected applicants, we note that the parcels we use are only those which have boundary shapefiles and so have already passed through a first round of screening to ensure most paperwork is in place (i.e. have demonstrated reasonable institutional capacity). Therefore the main reasons for rejection in the group we draw controls from are having part of the parcel outside of the program eligible zones (defined by watershed boundaries) or not having a high enough percentage of land in forest within the submitted property. Conditional on appropriate controls for region, market access, and baseline land use, these differences are unlikely to be correlated with deforestation. We use spatial overlay analysis to construct a full set of geographic controls for each property designed to capture such potential differences.

Considering the future enrollees, we argue that they constitute a reasonable control group because 2004 was an early year of the program. Anecdotal and case study evidence (Shapiro

2010) suggests that many landowners simply did not know about the program in 2004.[3] If they did know about the program and planned to enroll, that would most likely bias the results towards finding no effect: in order to enroll in the program, households must have existing forest cover, so households who plan to enroll in the future would be less likely to deforest if they anticipate future enrollment.

In addition, rather than use all applicants, we use matching to select controls which are closest in terms of observable characteristics and geographic region. We then estimate impacts using the Abadie-Imbens (2002) bias-adjusted matching estimator or regression to correct for remaining observable characteristics. We acknowledge that despite these efforts, this is not an ideal identification strategy because there may still be remaining unobservable differences between the groups. However, we believe that using matched controls from the applicant pool and parcel-level data represents a significant advance considering the limitations of previous research on the effectiveness of PES programs.

Pre-processing using matching

As shown in Table 1, a comparison between all recipient parcels from the 2004 PSAH cohort and all potential control parcels indicates only moderate differences in characteristics across these two groups. There are more ejidos amongst the recipients, who are also found in areas of higher slope and elevation. Enrollees have lower amounts of semi-deciduous forest and more selva (tropical wet forest). They also tend to be somewhat closer to the nearest road.

[3] If this assumption is correct, there should be no significant differences in deforestation behavior between the rejected applicants and future enrollees. To test this we took the future enrollees as a "pseudo-treatment" group and we checked whether the estimated causal effect of the program is in fact zero. The logic for this test follows that of Heckman, Ichimura, and Todd (1997). We use both regression and matching estimators to assess the differences in impact on these two groups, and find no significant effect.

Calculations of the normalized difference in means (a metric independent of sample size) of these two samples reveal no differences greater than 1/3 of a standard deviation.

Following Ho et al. (2007), we use matching to "pre-process" the data in order to select the best controls from among the applicant pool. We match parcels based upon area enrolled, slope, elevation, proportion of the forest which is semi-deciduous, proportion in selva, and road density. Exact matching on regional and tenure--communal vs. private property--variables is also included. [4] We restrict the estimation sample to enrolled parcels which have reasonably good matches, as defined by taking the best 80% of matches.[5] Table 2 shows the comparison across observable characteristics in recipient and non-recipient groups when we consider the best 80% of the matches. The balance across groups with respect to observable characteristics is significantly improved compared to the full sample (Table 1); there are no significant differences in control characteristics between recipient and non-recipient groups in the matched subsample shown in Table 2.

Matching estimation

The matching estimation framework appeals to the concept of "potential outcomes". In particular, for each property $i = 1, \ldots, N$, we suppose that we can observe deforestation if the

[4] Region 1 is comprised of the states in the north and west of the country, including Baja California, Chihuahua, Coahuila, Durango, Sinaloa, and Sonora. Region 2 includes Aguascalientes, Guanajuato, Hidalgo, Nayarit, Nuevo Leon, Queretaro, San Luis Potosi, Tamaulipas, and Zacatecas. Region 3 houses the central states: Colima DF, Jalisco, Michoacan Morelos, Mexico, Puebla and Tlaxcala, and region 4 all of the southern states. Summary statistics by region are given in Table 5.

[5] There are several metrics that can be applied to establish the nearest neighbor within the sample; we use both the Mahalonobis metric and the inverse sample standard errors. The estimator calculates the difference between the predicted outcomes for properties that are in and out of the program. In our case, we are particularly interested in the average treatment effect among the treated, where we use only one match (the nearest neighbor) in calculating the effect. In other words, the treatment effect is the average difference between the properties receiving the program and their nearest match. This is similar to restricting a sample according to propensity score values, where the standard in the literature is to eliminate the 10% most and 10% least likely to be receiving a program (Crump et al. 2009). We also restricted the sample to the best 90% matches, with no significant differences in results.

landholder does not receive the program ($Y_i(0)$) and if the landholder does receive the program ($Y_i(1)$). Clearly only one of these two events is actually realized. W_i indicates the actual state of property i. Define $W_i = 1$ if the property participates in the program and zero otherwise. We are interested in the difference between these two outcomes: $\tau = E[Y_i(1) - Y_i(0)]$. In our case, we are particularly interested in the average treatment effect among the treated, where we use only one match (the nearest neighbor) in calculating the effect:

$$\hat{\tau} = \frac{1}{N_1} \sum_{i:W_i=1}^{N} \left(Y_i - \hat{Y}_i(0) \right),$$

Where

$$\hat{Y}_i(0) = \left\{ Y_j \qquad if\ W_i = 1, \right.$$

with j indicating the observation in the control group that is closest to the treated observation i. Because matches in finite samples can never be exact, we use the adjustment proposed by Abadie and Imbens (2002) and described in Abadie, Drukker, Herr, and Imbens (2004). The adjustment begins by estimating deforestation only for the control group using a simple ordinary least squares regression. The “missing outcome” is then estimated by taking the deforestation level of the closest match plus the difference between predicted deforestation for the treated match and its best (closest) match, where the prediction is based on the coefficients estimated using the control group in the first step. Matching is conducted with replacement.

Regression estimation

In order to be able to understand how program impacts vary with respect to observable characteristics, we also estimate effects using regression on the best 80% matches. We postulate a relationship between deforestation and program participation as follows:

$$\ln(1 + d_i) = \propto + \tau W_i + \beta' X + u_i,$$

where the dependent variable is the natural log of 1 plus the amount of deforestation in kilometers squared, which is a function of receiving payments, W (equal to one if the property was enrolled in the program and zero otherwise), plus geographic control variables which might be correlated with both deforestation (X) and the probability of enrollment.[6] Because the dependent variable is censored, the equation is estimated using a Tobit.

RESULTS: PROGRAM IMPACT ON ENROLLED LAND

This section presents estimates of the impact of the program on land enrolled in the program. It begins with results from the matching estimator and then uses regression to examine impact heterogeneity.

Matching estimator

Table 3 shows the estimated program impacts applying the Abadie and Imbens (2002) matching estimator with adjustment. Part (a) illustrates estimates based on the full sample and part (b) shows the estimates for only the restricted sample of the data. Columns (1)-(3) use the Mahalonobis metric and columns (4)-(6) inverse of sample standard errors. In both the full and the restricted samples, the program shows a small but significant decrease in deforestation in enrolled properties. In order to understand how censoring may affect the estimates, columns (2) and (5) show the impact of the program on the probability of deforestation, and columns (3) and (6) on deforestation amongst those in the sample with positive forest loss ("deforesters"). The results from the matched sample suggest that the program significantly reduced the probability of deforestation by 10-11 percentage points. Given that the probability of any deforestation in the

[6] As robustness checks, we have used total area deforested and percent area deforested as dependent variables. In all cases we find similar results.

control group is approximately 0.25, this represents an approximately 44 percent reduction. In addition, the program reduced the area indicated with deforestation among deforesters (without adjusting for the likelihood of being in this group) by around 10-11 percent.

Post-matching regressions and impact heterogeneity.

Using regression post-matching (Table 4) similarly suggests that the program has reduced deforestation. The point estimates are negative, indicating that the program has reduced deforestation, and the results are significant. Calculating the marginal effects (bottom of Table 4), the post-matching regression indicates that the program has reduced the probability of deforestation by 6 percentage points (or approximately 24 percent) and has reduced deforestation among deforesters by approximately 2 percent.

There is significant heterogeneity in these program effects, however. As shown in Table 4, the marginal effects calculated for region 2 suggest that the program decreased the probability of deforestation by 12 percentage points, and the area deforested among deforesters by 5.3%. The effects are smaller and only marginally significant in regions 3 and 4, and not significant in region 1. These results imply that the largest program impacts are found in the northeast and north central states. We do not find heterogeneity in impacts by land tenure arrangements, however. Column 3 in Table 4 includes an interaction term between ejido and payments; there is no significant difference in the effects by property type.

Finally, we examine whether effects are different for those properties which are more isolated, using an interaction term between road density and receipt of payments (Column 4). We observe that the program appears to have reduced deforestation only in places with higher road density. At the mean level of road density (marginal effects shown in Table 4), the program does not have a statistically significant impact. However, at the 90th percentile of road density, the marginal effects are significant at the 3% level and negative. To better understand the pattern of marginal effects by road density, Figure 2 graphs out the estimated program impact coefficient by different road densities. We see that the program only decreases deforestation significantly once there are more than 900 kilometers of roads in the 50 kilometer buffer (.36 km of roads per square kilometer) around the treated community (just over 60% of the sample are in areas with less than this road density). One reason we might observe this is that on the ground monitoring of the program becomes increasingly difficult as communities become more remote. Another possible explanation is that the pressures on forests may be lower in remote communities, so remote communities from the applicant pool are less likely to deforest when they do not receive payments.

In sum, the results indicate that the program has had a modest direct effect of reducing deforestation, with larger impacts occurring on enrolled properties in the center and northeast of the countries and in areas with good infrastructure. These direct impacts, however, may be undermined by slippage of deforestation through changes in recipient behaviors on un-enrolled land within the same properties or through changes in prices which induce market-wide effects. The next section presents a theory to formalize the mechanisms through which such impacts might occur.

SLIPPAGE EFFECTS IN A DEVELOPING COUNTRY

The possibility of spillovers due to changes in production behavior poses a serious problem for efforts to conserve forests or other land (Berck and Bentley 1997, Wu 2000, Wu, Zilberman and Babcock 2001, Murray et al. 2004, 2007, Wear and Murray 2004, Fraser and Waschik 2005, Gan and McCarl 2007, Plantinga and Richards 2008). Concerns about slippage related to incentive-based conservation were raised by Wu (2000) in this journal in an analysis of the U.S. Conservation Reserve Program and debated in a subsequent exchange (Roberts and Bucholtz 2005, 2006, Wu 2005). This debate distinguished between two types of slippage: substitution effects and price increases in output markets. In the context of forest-conservation payments, a substitution effect occurs when a landowner who removes one parcel of land from production (enrolling it in the program) shifts the planned production to another parcel within his landholdings. An output price effect occurs if the removal of multiple parcels of land from production or the introduction of payments alters market prices and these changes induce changes in production (potentially across all landholdings in the market).

Roberts and Bucholtz (2005) noted that substitution effects are unlikely to occur in developed country settings because markets should function well to reallocate resources, making the returns to each parcel within a landholding independent of production choices on others. In a developing country setting, however, substitution may be an important slippage channel because of rigidities in land, credit, or labor markets. Indeed, Uchida et al (2009) find evidence of labor market effects in China as a result of its “Grain for Green” program. Considering deforestation more generally, Zwane (2007) suggests that increases in income may relax credit constraints and increase deforestation in Peru. In addition, Robalino and Pfaff (2009) find evidence for spatial spillovers in deforestation decisions in Costa Rica as a result of strategic interactions unlikely to exist in the context of well-developed markets. We further investigate this question using the economic production framework described in the next section.

ECONOMIC PRODUCTION FRAMEWORK

In order to illustrate the potential changes in production behavior induced by a conservation payments scheme, we develop a simple model of household production and land allocation.[7] In this model, households have endowments of land and capital. They seek to maximize consumption, which is a function of the total value of their production. Households choose how much land to allocate to two types of production, agriculture and forest. Agricultural production requires a variable input (representing seeds, fertilizer, etc.) which households must purchase. In order to purchase this input, households can either use their endowments (existing liquid wealth) or borrow. Households therefore also choose how much of the variable input to purchase and how much to borrow.

We formalize the model by assuming that the total area of land that is managed by a household is fixed at T.[8] The amount of land devoted to agriculture is denoted as a and to forest f, so that: $T = a + f$. Agricultural production is a function of land in agriculture, a, and the quantity of the variable input, n:

(1) $y^a = a^\alpha n^\varphi$

For simplicity, assume that if either argument of the production function is equal to zero, then production is equal to zero and that $\alpha + \varphi < 1$. We assume that forest production is a simple linear function of the amount of land in forest:

(2) $y^f = \beta f$

[7] The set-up of an input purchase is similar to Guirkinger and Boucher (2007) which models how agricultural productivity depends on credit constraints.

[8] The assumption of a land constraint is more reasonable for areas where most land is formally or informally claimed, rather than frontier situations, where family labor may be the main constraint (e.g. Pagiola and Holden 2001). We argue the land constraint better reflects the areas of Mexico where the PSAH is operational.

Households do not directly consume their products but sell these products in nearby markets, for prices p^a and p^f respectively. We assume competitive markets, so individual producers take these prices as exogenous.

Household allocation of land subject to credit constraints

Households start with an endowment of liquid wealth, denoted *K.* They can spend some or all of this endowment on the variable input, n. For simplicity, we assume that they can consume remaining capital. They also have the option of borrowing a sum B from the bank to purchase more of the variable input and thus boost production; we also assume that they can consume leftover borrowed funds. The cost of borrowing is r. At the end of each period, the household must pay back $(1+r)B$ when agriculture is harvested.

Borrowing is constrained because households have a limited amount of collateral to back their loans. We assume that this collateral is represented by the total value of their land (they cannot borrow against their cash endowment). The borrowing constraint is thus:

(3) $B \leq p^l T$

where p^l is the price of land.

At the start of the period, households can purchase productive inputs (n). Their purchase of productive inputs is limited by the amount of working capital ("cash-in-advance") that they have:

(4) $p^n n \leq K + B$

Consumption is limited by total production plus endowments, which gives the constraint: $c \leq p^a a^\alpha n^\varphi - p^n n + p^f \beta f + K - rB$. Assuming that utility only depends upon consumption, households seek to maximize consumption subject to the working capital and borrowing constraint. The maximization problem can then be written:

(5) $\max_{a,f,n,B} [p^a a^\alpha n^\varphi - p^n n + p^f \beta f + K - rB] - \gamma_1(a + f - T)$

$$-\gamma_2(B - p^l T) - \gamma_3(p^n n - B - K)$$

Given this maximization problem, the first order conditions are:

FOC1. a: $p^a \alpha a^{\alpha-1} n^\varphi - \gamma_1 = 0$

FOC2. n: $p^a \varphi a^\alpha n^{\varphi-1} - p^n - \gamma_3 p^n = 0$

FOC3. f: $p^f \beta - \gamma_1 = 0$

FOC4. B: $-r - \gamma_2 + \gamma_3 = 0$

FOC5. γ_1: $a + f = T$

FOC6. $\gamma_2(B - p^l T) = 0, \gamma_2 \geq 0, B \leq p^l T$

FOC7. $\gamma_3(p^n n - B - K) = 0, \gamma_3 \geq 0, p^n n \leq B + K$

By definition, the land constraint always binds. However, the borrowing and capital constraints may or may not. Although there are a variety of cases that one could consider, we restrict ourselves to discussing the implication of these conditions for households which have at least some land in agriculture. Among these households, there are those who choose to borrow and those who do not, and among borrowers, some households which are constrained borrowers and others which are not. We consider these cases in order.

Non-borrowing households

If households do not borrow, then they chose only the allocation of land and the amount of the variable input. We note the standard result that the optimal choice of land use is the one that equates the marginal product of agricultural land with the marginal product of forest land: $p^a \alpha a^{\alpha-1} n^{\varphi} = p^f \beta$. The optimal *n* is determined by FOC2, which, when rearranged, yields: $n^{nb} = \left(\frac{p^a \varphi a^{\alpha}}{p^n}\right)^{\frac{1}{1-\varphi}}$. This gives an expression for the optimal amount of agricultural land for non-borrowing households:

(6) $$a^{nb} = \left(\frac{\alpha}{p^f \beta}\right)^{\frac{1-\varphi}{1-\alpha-\varphi}} \left(\frac{\varphi}{p^n}\right)^{\frac{\varphi}{1-\alpha-\varphi}} (p^a)^{\frac{1}{1-\alpha-\varphi}}$$

Unconstrained borrowing households

For some households, the endowment of K will not be sufficient to purchase the optimal amount of the variable input, *n*. For households which do require borrowing in order to purchase inputs, but are not limited by their endowments, $\gamma_2 = 0$ and $\gamma_3 > 0$. This implies that $\gamma_3 = r$, and the optimal purchase of the input is $n^{ub} = \left(\frac{p^a \varphi a^{\alpha}}{p^n(1+r)}\right)^{\frac{1}{1-\varphi}}$. The optimal amount of agriculture is:

(7) $$a^{ub} = \left(\frac{\alpha}{p^f \beta}\right)^{\frac{1-\varphi}{1-\alpha-\varphi}} \left(\frac{\varphi}{p^n(1+r)}\right)^{\frac{\varphi}{1-\alpha-\varphi}} (p^a)^{\frac{1}{1-\alpha-\varphi}}$$

Comparing expressions (6) and (7) we see that households which borrow will have less land in agriculture, since borrowing effectively raises the cost of agricultural activity.

Constrained borrowing households

Finally, among for households which are constrained in borrowing, $\gamma_2 > 0$ and $\gamma_3 > 0$. For these households, input use will be limited by the household's land and cash endowments: $n^{cb} = \frac{p^l T + K}{p^n}$. This implies:

$$(8) \qquad a^{cb} = \left(\frac{\alpha}{p^f \beta}\right)^{\frac{1}{1-\alpha}} \left(\frac{p^l T + K}{p^n}\right)^{\frac{\varphi}{1-\alpha}} (p^a)^{\frac{1}{1-\alpha}}$$

For households with the same land endowment, *T*, but varying cash endowments, *K*, we can therefore conclude that less land is put into agriculture relative to forestry as the cash endowments decrease. In other words, $a^{nb} > a^{ub} > a^{cb}$.

Introducing PES payments

We introduce payments for forest services into this setting in a very stylized manner, but one which we think captures the key decisions being made at the household level. We specify that each household can put land in the program, denoted as S. For each unit of land in the program, the household earns a per unit payment, p^s. This payment, $p^s S$, comes to the household as cash which the household can then use to buy additional inputs or can consume. We assume that once land is enrolled in the program, it cannot be used for forest production.[9] The optimization problem is thus modified as follows:

$$\max_{a,f,n,B} [p^a a^\alpha n^\varphi - p^n n + p^f \beta f + K - rB + p^s S] - \gamma_1(a + f + S - T)$$

$$-\gamma_2(B - p^l T) - \gamma_3(p^n n - B - K - p^s S)$$

[9] The contracts for the PSAH do limit forest production, including extractive activities or grazing, in the enrolled land. The assumption that households can put as much land as they want in the program is a simplification because in reality they can only enroll land which was already in forest use. There is also a maximum limit on the number of hectares which can be enrolled but that only binds for very large properties.

The first order conditions become:

FOC1p. a: $p^a \alpha a^{\alpha-1} n^{\varphi} - \gamma_1 = 0$

FOC2p. n: $p^a \varphi a^{\alpha} n^{\varphi-1} - p^n - \gamma_3 p^n = 0$

FOC3p. f: $p^f \beta - \gamma_1 = 0$

FOC3ap. S: $p^s - \gamma_1 + \gamma_3 p^s = 0$

FOC4p. B: $-r - \gamma_2 + \gamma_3 = 0$

FOC5p. γ_1: $a + f + S = T$

FOC6p. $\gamma_2(B - p^l T) = 0, \gamma_2 \geq 0, B \leq p^l T$

FOC7p. $\gamma_3(p^n n - B - K - p^s S) = 0, \gamma_3 \geq 0, p^n n \leq B + K + p^s S$

Non-borrowing households

When households do not borrow, the linearity of the forest and PES production functions results in a corner solution as the price of payments increases. When $p^f \beta > p^s$, households would simply keep land in forest and the policy would have no effect on land use. When $p^s > p^f \beta$, however, households would choose to enroll all land previously in forest in the program. For those households who do choose to enroll in the program, the optimal choice of *n* is the same as in the no-policy case. Expression (6) becomes:

(6p) $$a^{nb,p} = \left(\frac{\alpha}{p^s}\right)^{\frac{1-\varphi}{1-\alpha-\varphi}} \left(\frac{\varphi}{p^n}\right)^{\frac{\varphi}{1-\alpha-\varphi}} (p^a)^{\frac{1}{1-\alpha-\varphi}}$$

Therefore, in the case where $p^f \beta = p^s$, the expressions with and without the policy are equal so there is no change in the amount of land in agriculture. When the PES price exceeds the effective price per unit for forest land, however, then more land is put in to forest (less in agriculture) under the program. This is of course the intended effect of the program--to induce additional land to be in forest use by raising the returns to forest.[10]

Unconstrained borrowing households

If households need to borrow to finance inputs, but are not constrained in their borrowing, $\gamma_2 = 0$ and $\gamma_3 > 0$, then optimal land in agriculture is given by:

(7p) $$a^{ub,p} = \left(\frac{\alpha}{(1+r)p^s}\right)^{\frac{1-\varphi}{1-\alpha-\varphi}} \left(\frac{\varphi}{p^n(1+r)}\right)^{\frac{\varphi}{1-\alpha-\varphi}} (p^a)^{\frac{1}{1-\alpha-\varphi}}$$

Note that households will enroll land previously in forest in the program if $(1+r)p^s > p^f\beta$. This suggests that households who are borrowing constrained would require less compensation in order to be induced to enroll at least some land in the program.[11] We also see that compared to expression (7) above, a will be smaller as long as $(1+r)p^s > p^f\beta$. So when the policy is introduced, the amount of land devoted to agriculture will generally decrease for households who borrow but are unconstrained in the amount borrowed. This is due to the same logic as above--the program increases the relative returns to having land in forest.

It is also possible that for households who were just on the margin of borrowing, the cash availability from the program could push them from being unconstrained borrowers to being non-borrowers (unconstrained in input use). In this case, the introduction of the program could actually increase land in agriculture (mathematically, $a^{nb,p} > a^{ub}$).

[10] In reality, forest takes time to regenerate, so the goal of the program is to induce additional land to remain in forest. We could think of this as corresponding to a dynamic framework where the price of agriculture relative to forest is increasing over time so more land is being put in agriculture each period in general.
[11] An interesting implication which unfortunately we cannot test without household data but certainly deserves investigation.

Constrained borrowing households

For households whose borrowing was constrained before the program, if they are still constrained with the policy in place and choose to put land in the program, then their purchase of inputs will be: $n^{cb,p} = \frac{p^l T + K + p^s S}{p^n}$ where S is a choice variable. Expression (8) from above becomes:

(8p) $a^{cb,p} = \left(\frac{\alpha}{(1+\gamma_3)p^s}\right)^{\frac{1}{1-\alpha}} \left(\frac{p^l T + K + p^s S}{p^n}\right)^{\frac{\varphi}{1-\alpha}} (p^a)^{\frac{1}{1-\alpha}}$

From expression (8p) compared to expression (8), we see that the program affects both the price ratio relative to agriculture (denominator of first term) and the cash-in-advance constraint (numerator of second term). As the price of payments increases, this will tend to decrease the amount of land in agriculture, but will also increase the cash in advance constraint. The net impact on land use, compared to the situation without the program, will depend on how these terms grow relative to each other as the payment offered by the program increases. (In other words, we are comparing: $\frac{(p^l T+K)^{\varphi}}{p^f \beta} vs. \frac{(p^l T + K + p^s S)^{\varphi}}{(1+\gamma_3)p^s}$).

Let us consider the case where the program pays just enough to induce landowners to switch from forest production to the program when it is introduced. Then for a small number, ε, we have: $(1+\gamma_3)p^s = p^f \beta + \varepsilon$. It will therefore be the case that the program will increase land in agriculture when: $\frac{p^f \beta + \varepsilon}{p^f \beta} = 1 + \frac{\varepsilon}{p^f \beta} < 1 + \left(\frac{p^s S}{p^l T + K}\right)^{\varphi} = \frac{(p^l T + K + p^s S)^{\varphi}}{(p^l T + K)^{\varphi}}$. This is likely to be true as long as there is some land enrolled, since $S \leq T$ and $\varphi < 1$.

In other words, if the program just barely compensates for the opportunity cost of forest production, it will lead to an <u>increase</u> in the amount of land in agriculture for credit constrained households. However, as the difference between program payments and returns to forestry increase, the increase in relative returns to forested land will dominate and land in agriculture will decrease. Another way to see this is if we solve completely for *a* in terms of parameters and simplify:

$$(9)\ a^{cb,p} = \frac{\alpha}{\varphi+\alpha}\left[T + \frac{p^{l}T+K}{p^{s}}\right]$$

We see that for constrained borrowers, as the price of payments increases, it becomes optimal to put less land in agriculture. Also, as before, it is possible that the payments will shift credit-constrained households into the category of unconstrained borrowers, which would result in an unambiguous increase in their agricultural production.

In sum, then, individual households operating under no constraints will reduce their land in agriculture – thereby increasing their forest area – if the relative prices are high enough to induce enrollment in the PES program. Households which are unconstrained borrowers will also reduce their land in agriculture, increasing forest cover, unless it is the case that the payments from the program allow them to cease borrowing altogether. But constrained households are likely to increase their agricultural production, thereby reducing forest area, relative to their behavior without the program. At a household level, therefore, we expect to observe slippage through substitution where households require borrowing in order to purchase agricultural inputs and are credit-constrained.

Output price slippage effects

We are also interested in potential slippage through price effects, which requires consideration of the aggregate market effects of the program. Output price effects could happen through either supply or demand channels. On the supply side, if households remove land from agricultural production, this may reduce supply of agricultural goods and therefore increase prices.[12] On the demand side, increases in income as a result of the program may increase consumption of land-intensive goods and therefore demand for resource intensive products.[13]

Price slippage effects--supply shifts

For simplicity, we assume that all households have the same amount of land, *T*, but differ in their endowments of *K*. The total number of households is *N* and they fall into the three categories as above, depending on their endowments of *K*. Then total land used for agricultural production within a given geographic area can be written as:

(10) $A = N_0 a^{nb} + N_1 a^{ub} + N_2 a^{cb}$

This gives us a corresponding market supply for agricultural goods of:

(11) $S_a = S(\vec{p}, T_o, K, N_0, N_1, N_2; \alpha, \beta, \varphi)$

The standard price spillover effect is assumed to occur from a reduction in this supply due to an increase in the price of agriculture as additional land is put in forest use. However, given the changes described above, our model implies that it would be possible to see either an increase or a decrease in the supply of agriculture, depending on how many households fall into each category above.[14] Therefore we expect output price slippage effects but cannot rule out the fact

---

[12] Fraser and Waschik 2005 use a CGE model to simulate policies for agricultural land retirement in Australia and find that this type of slippage may significantly reduce conservation benefits.

[13] See Alix-Garcia, McIntosh, Sims, and Welch 2010 for an example of this effect in the context of poverty alleviation programs. However, other forces may counter this trend. Wealth may also increase demand for forest products (Foster and Rosenzweig 2003), facilitate the development of stronger institutional mechanisms for conservation, or increase investment in greener technologies.

[14] We focus on changes in demand for the agricultural good, but if enough households enroll forest in the program rather than use it for forest production, this would increase the price of forested goods. In general equilibrium and with an unlimited program budget, we would expect that the price of forest goods would rise until the marginal returns to forest land were equal to the price of the payments with some land in each use.

that they could go in either direction. We note that the magnitude of these effects should depend on the elasticity of the corresponding demand, as previously demonstrated by Chomitz (2002).

Price slippage effects – demand shifts

A second potential channel for output price effects is through the demand side of the market. This is a standard insight from agricultural household modeling (see Singh, Squire and Strauss 1986), where household profits determine household consumption. In order to consider changes in demand for agricultural goods, we must make assumptions regarding the composition of household consumption (c). Let us assume that the consumption good includes both agricultural and forest goods. Household demand for these products is a function of their prices, a household preference parameter π, and household income I, which is composed of profits from agriculture and forest, and is therefore a function of land and other asset endowments. We can then write each household's demand for agricultural and forest products as:

(12) $d_i\left(p_i, p_j, I; \pi\right)$

where $i, j \in a, f, i \neq j$.

Total market demand is the sum of all of the individual demand functions and is:

(13) $D_i(\vec{p}, I(\vec{p}, N_0, N_1, T_0, K); \pi) = \sum_{k=1}^{N} d_k$.

This illustrates a second channel for output price effects: increases in household income I as a result of the program could lead to increases in demand for consumption goods. If these goods are agricultural, this will increase p^a in our model, which raises the marginal product of agriculture[15]. In response, profit-maximizing households will tend to shift their allocation of land towards agriculture, increasing deforestation.

Price slippage effects and market size

Since price effects are mediated through markets, they are likely to be stronger in areas which have high transportation costs and thus localized markets. Within the context of our model, an isolated market is effectively one which has a low N, a feature which decreases the elasticities of supply and demand in our model. A lower N essentially concentrates the impact of decreased supply or increased demand, which could lead to a large local price effect and hence a large local program spillover[16]. Previous literature on price slippage effects is limited by the fact that developed country markets are quite large and so output price effects are difficult to observe empirically. However, in a developed country case, high transportation costs mean that markets may be more localized and it would be possible to observe output price effects.

EMPIRICAL ANALYSIS OF SPILLOVERS

We now turn to the Mexican case to test empirically for evidence consistent with the above theoretical predictions. We begin with our analysis of substitution slippage and finish with a suggestive analysis of price slippage.

Measurement and identification of substitution effects

Substitution effects by definition should occur within the landholdings of the owner who enrolls in the program. As above, we use both matching and regression with matched samples in

[15] Evidence from evaluations of Progresa suggest that this is the case (Gertler et al 2006).
[16] See Alix-Garcia, McIntosh, Sims, and Welch (2010) for further elaboration on this point.

order to test for increases in deforestation close to the enrolled parcels, compared to deforestation close to control parcels. We separate our analysis between private and common properties. This is done for two reasons. First, because we know the boundaries of the ejidos, we know which land is most likely to provide potential substitutes for the land enrolled in the program when the recipients are ejidos. For common properties, we calculate deforestation within the boundary of the ejido but outside of the property enrolled (or seeking to be enrolled) in the program. We also construct buffers of 1, 2, and 5 km around the ejido property, since community members may also control land which is nearby but not within the official ejido boundary. For the private properties, we do not know the property boundaries. We calculate deforestation within 1, 2, and 5 km buffers around each privately enrolled parcel under the assumption that other private landholdings are contiguous with the enrolled property. The second reason for separating the analysis between private and common properties is that we might theoretically expect different effects either due to differences in market rigidities or collective action issues within the common properties.

Results: substitution

The results indicate possible slippage effects within communal properties which are more remote. As shown in the top panel of Table 5, we do not find evidence of significant slippage effects within ejidos or in 1 or 5 km buffers around ejidos if we assume homogeneity in treatment effects. However, if we allow slippage effects to vary by road density, we do find significant within-property slippage at low road densities (Table 5, column 5). Figure 3 maps out these changes in impact coefficient estimates according to road density. The solid line indicates the estimated coefficient of the program impact as it varies with road density, and the dashed lines are confidence intervals around this estimate. Figure 3 indicates that as road infrastructure improves, spillovers within communal property decrease and may in fact become negative.[17]

Comparing these results to the estimates of direct program impact yields two insights. First, we observe significant spillovers where we did *not* observe significant direct avoided deforestation impacts of the program: in areas with poor infrastructure. This lends support to the credit constraints hypothesis because we observe spillovers where the program does not appear to be reducing production but could have relaxed credit-constraints. Second, the magnitudes of the spillover effects suggest that we should be concerned about the issue of slippage through substitution, at least in some areas. For instance, at the 25th percentile of road density (409 kilometers) the spillover effect on the probability of deforestation is 0.19 (with standard error .07), and on the area deforested 0.21 (with standard error .08). These numbers are significantly larger than the average impact of the program (-.06 on the probability of deforestation and 2 percent on the area deforested), suggesting that substitution spillovers could swamp the program impact in these very remote areas.

Finally, a similar analysis of deforestation in 1 and 5 km buffers around each private property (Table 5, bottom panel) does not find similar evidence of substitution spillovers. It is difficult to know whether this is due to differences in behavior across property types or because the buffers around the enrolled private properties are a noisy measure of lands that might be used by private landowners who have enrolled forest, and are therefore inadequate to capture a substitution effect that might be occurring within private properties. Ideally, we would have detailed information on the boundaries of individual properties enrolled in the program. This

[17] The combination of road density and treatment effect ceases to be significantly positive at the mean road density of 863 kilometers.

would be an important type of information for future PES programs in Mexico or elsewhere to collect as part of the application process.

Identifying price spillovers

We next present a simple test for slippage through output price effects (Table 6). In order to examine potential price spillovers, we calculate the number of hectares of land enrolled in the PSAH within a 50 km radius of each property in our sample. This gives us a proxy for the degree of the reduction in the supply of land and/or the magnitude of the total payments going to a particular area. All else equal, where there are more surrounding properties enrolled in the program, we expect that price increases for agricultural goods would be larger and we would expect to see a greater increase in deforestation. The ability to observe this effect, however, depends on markets being sufficiently localized to prevent price changes from being distributed through the entire national market. We expect that output price effects would only be observable where markets are relatively localized due to having low access to infrastructure (low road density).

Results: spillovers

Table 6 shows regressions of deforestation in the buffer zones of 1 km and 5 km around all properties as a function of surrounding enrollment and other controls.[18] We leave out within-ejido measurements, as the results of the previous section suggest that impacts there may be confounded with substitution effects. We measure surrounding enrollment in the 50 km radius using both the total area enrolled and a binary measure of high or low surrounding enrollment (defined by being in the top 10% of area enrolled). We choose this second measurement with the

[18] We also divided the sample between ejidos and private properties in order to analyze the 1 and 5 km buffers. This separation had no effect on the results, so we present here the pooled sample for these buffers.

idea that price effects may only be detectable where there are very large amounts of the land in the program – i.e., beyond some threshold.

We find that having more surrounding area enrolled in the program is significantly related to increases in deforestation in all buffer zones and that the effects are smaller as road density increases. Figure 4 maps out the changes in this impact according to road density for the 5 km buffer and defining treatment as being surrounded by the top 10% of area enrolled in the program. The x-axis measures road density, so the graph shows how the impact of having substantial enrollment surrounding a community varies with infrastructure. We see that as infrastructure quality increases, detectable spillover effects are dissipated. As a robustness check, we also test whether this effect is different in recipient and non-recipient communities, and fail to reject the null hypothesis that they are the same[19].

The bottom panel of table 6 shows marginal effects calculated at road densities at the 25th and 75th percentile. For the interaction term where the surrounding area enrolled is continuous, the marginal effect is calculated for a one standard deviation change in area enrolled. In this case, the impacts are only significant for the 5 kilometer buffer. In the case where the sample is divided between "most enrollment" (top 10%) and the rest, the price spillover effect is never detectable in road dense areas. For less connected areas, it is both detectable and large, compared to the direct impact of the program. At the 25th percentile, the marginal increase in deforestation for a one standard deviation change in area enrolled is an increase in the probability of deforestation of 4 percentage points and in area deforested of 6.9 percent.

We caution that this test is imperfect, as it is possible that high enrollment of area in the program is correlated with some other unobserved characteristic of a region that also increases

[19] Results available upon request.

deforestation. Table 7 shows summary statistics for all the properties, with the sample separated into those with low surrounding enrollment (lowest 90%), and very high enrollment. From this table we observe that although the two samples are similar in important deforestation determinants like slope, elevation, and road density, their regional distribution is quite different – most of the properties in the “high surrounding enrollment” category are found in region 4, the southeast, which is also a tropical zone, as indicated by the relatively higher proportion of tropical forest in the sample. Because of the nature of our dependent variable, and the potential confounds in the identification of the price effect, we take the price spillover results as merely suggestive and deserving of further exploration.

CONCLUSION

This article has analyzed the impact of Mexico’s payments for hydrological services program using recipients enrolled in an early cohort of the program. We find that the program has significantly reduced the probability of deforestation by approximately 6-10 percentage points and has reduced the area deforested among deforesters by 2-11 percent. Given the small number of studies which have evaluated national PES programs using rigorous comparisons, this is a valuable piece of evidence indicating small to moderate program effectiveness. There is also considerable heterogeneity in these effects. In particular, the program seems to be more effective in generating avoided deforestation where road infrastructure is good and it seems to be most effective in the northeast and central states of Mexico. Future work should seek to better understand the mechanisms driving differences in program effectiveness across regions and to understand how program impacts have changed over time. Given that 2004 was one of the early years of implementation and that later cohorts increased targeting to areas at higher risk of deforestation, future cohorts are likely to demonstrate greater avoided deforestation impacts.

With respect to slippage, our results indicate deforestation spillovers through both price and substitution mechanisms. Substitution mechanisms appear to be at work only in remote ejidos. This is consistent with the notion that substitution should only occur where there are significant credit, land or labor market rigidities, as suggested by Roberts and Bucholtz (2005) and demonstrated in our model. Individuals living in ejidos are generally poorer than private land owners, so the increased deforestation that we observe in unenrolled lands of participating ejidos may occur as a result of the relaxation of a credit or endowment constraint on production. Price mechanisms are likely to be ubiquitous across both developed and developing countries but can only be detected when markets are localized, which is the case in parts of Mexico. We find that deforestation is more likely when more land in the surrounding 50 km radius has been enrolled in the PSAH program, and there is no difference in this effect between enrolled and unenrolled pieces of land. Unfortunately, we cannot differentiate between a price spillover caused by general wealth effects which result in increased demand versus one caused by a reduction in supply through decreasing land available through agriculture--this is also an area deserving of future research.

The combination of within-ejido and remote-property price spillovers does indicate some lessons for the use of PES in schemes to reduce carbon emissions from deforestation and forest degradation. Given the evidence that spillovers may occur as individuals increase their production as program payments loosen credit constraints, one possibility for limiting this type of displacement is to use a contract which forbids land conversion in the entire area of the ejido, rather than for a subset of land (this has been suggested by Schwarze et al. 2002 among others).

However, this would fundamentally change the nature of the contracts, would likely require higher payments, and still does not address the problem of spillovers through output price effects beyond property boundaries. The problem of output price effects is not in any way likely to be unique to Mexico and suggests that REDD designers should consider embedding PES programs in larger national systems which track overall deforestation at a regional or national scale rather than attempting a project-based approach. This lends support to the "national inventory approach" to REDD suggested by Plantinga and Richards (2008).

Finally, much of the uncertainty in our analysis results from the paucity of readily usable satellite imagery measuring forest cover. The evaluation of land use change is subject to error due to seasonality and topography. CONAFOR has established an excellent monitoring program which shows the power and utility of geographic information systems, but even with outstanding staff and top of the line equipment, monitoring presents a considerable challenge. Continued improvements in technology, lower cost imagery, and technical training for individuals responsible for monitoring deforestation are absolutely essential for future analysis and design of PES programs.

TABLES AND FIGURES

Table 1: Summary statistics for recipients and non-recipients (full sample)

Variable	Recipients	Non-Recipients	Test for difference	Normalized difference
Enrolled area	9.35	7.04	2.26	-.006
Proportion ejidos	0.67	0.58	2.69	
Ln(Average slope of enrolled area)	2.44	2.33	2.11	0.11
Average elevation of enrolled area	2092	1878	3.40	0.13
Proportion enrolled area semideciduous	0.20	0.32	4.03	-0.35
Proportion enrolled area selva	0.33	0.27	2.39	0.22
Road density in 50 km radius	5.82	6.48	1.71	0.18
Proportion in region 1	0.21	0.32	3.55	
Proportion in region 2	0.15	0.22	2.35	
Proportion in region 3	0.37	0.25	3.45	
Proportion in region 4	0.27	0.21	2.15	
Proportion with deforestation	0.22	0.23	0.17	
Ln(1+area deforested)	0.05	0.07	1.36	-0.21
Observations	352	462		

Recipients are from the 2004 PSAH cohort. Non-recipients are rejected applicants from 2004 or future recipients of the program (2006).

Table 2: Summary statistics for recipients and non-recipients (best 80% matches)

Variable	Recipients	Non-Recipients	Test for difference	Normalized difference
Enrolled area	5.53	5.78	0.40	-.002
Proportion ejidos	0.65	0.60	1.46	
Average slope of enrolled area	2.46	2.45	0.216	0.01
Average elevation of enrolled area	2.19	2.11	1.25	0.06
Proportion enrolled area semideciduous	0.19	0.24	1.55	-0.16
Proportion enrolled area selva	0.32	0.28	1.34	0.13
Ln(road density)	6.70	6.64	1.32	0.08
Proportion in region 1	0.22	0.27	1.69	
Proportion in region 2	0.16	0.22	1.99	
Proportion in region 3	0.36	0.33	0.98	
Proportion in region 4	0.26	0.18	2.52	
Proportion with deforestation	0.22	0.25	0.55	
Ln(1+area deforested)	0.04	0.07	2.45	
Observations	341	315		

Recipients are from the 2004 PSAH cohort. Non-recipients are rejected applicants from 2004 or future recipients of the program (2006).

Table 3: Estimates of program impact on deforestation rates: matching estimator (ATT)

a. Full sample

	Mahalonobis metric			Inverse sample standard errors		
	Ln(1+area deforested)	Deforest (0/1)	Ln(1+area defor)| Deforest > 0	Ln(1+area deforested)	Deforest (0/1)	Ln(1+area deforested)| Deforest > 0
	(1)	(2)	(3)	(4)	(5)	(6)
Treatment effect	-0.0438**	-0.122***	-0.0638	-0.0447**	-0.104**	-0.0501
	(0.0208)	(0.0419)	(0.0643)	(0.0213)	(0.0436)	(0.0642)
Observations	817	817	186	817	817	186

b. Best 80% matches

	Mahalonobis metric			Inverse sample standard errors		
Dependent variable	Ln(1+area deforested)	Deforest (0/1)	Ln(1+area defor)| Deforest > 0	Ln(1+area deforested)	Deforest (0/1)	Ln(1+area deforested)| Deforest > 0
	(1)	(2)	(3)	(4)	(5)	(6)
Treatment effect	-.0488***	-.105**	-.1136**	-.0478***	-.1052**	-.0989*
	(-3.169)	(-2.459)	(-1.973)	(-3.096)	(-2.345)	(-1.944)
Observations	656	656	160	656	656	160

Standard errors in parentheses; *** p<0.01, ** p<0.05, * p<0.1
Estimations based on differences between nearest neighbors adjusted for remaining differences in observables (following Abadie and Imbens 2002). Columns 1-3 use matches based on the Mahalonobis metric and Columns 4-6 use matches based on inverse sample standard errors. Matches are based on area enrolled, slope, elevation, proportion of the forest which is semi-deciduous, proportion in selva, road density, region and tenure (private property vs. ejido).

Table 4: Post-matching regression estimates (matched sample)

	(1)	(2)	(3)	(4)
Recipient	-0.092**	0.036	-0.103	0.564
	(0.045)	(0.096)	(0.092)	(0.474)
Recipient x region 2		-0.284*		
		(0.149)		
Recipient x region 3		-0.121		
		(0.121)		
Recipient x region 4		-0.144		
		(0.130)		
Recipient x ejido			0.0142	
			(0.106)	
Recipient x ln(rd density)				-0.095
				(0.070)
Area	0.006**	0.006**	0.006**	0.012***
	(0.003)	(0.003)	(0.003)	(0.003)
Ln(average slope)	-0.179***	-0.179***	-0.179***	-0.169***
	(0.037)	(0.037)	(0.037)	(0.038)
Average elevation	0.0201	0.014	0.020	-0.005
	(0.042)	(0.042)	(0.042)	(0.042)
Proportion of area in semideciduous forest	0.073	0.073	0.073	0.055
	(0.075)	(0.076)	(0.075)	(0.077)
Proportion of area in selva	0.099	0.094	0.100	0.074
	(0.065)	(0.065)	(0.065)	(0.066)
Ln(road density)	-0.014	-0.0080	-0.014	0.090
	(0.065)	(0.065)	(0.0645)	(0.071)
Ejido	0.307***	0.307***	0.302***	
	(0.058)	(0.058)	(0.071)	
Observations	656	656	656	656
Pseudo R-squared	0.145	0.151	0.145	0.0956
Selected marginal effects (at mean)				
Pr(d>0)				
Recipient	-.063**	.036	-.071	.391
	(.031)	(.096)	(.063)	(.312)
Recipient x region 2		-.284		
		(.149)**		
Recipient x ln(rd density)				-.067
				(.049)
Ln(d|d>0)				
Recipient	-.020**	.025	-.022	.134
	(.009)	(.067)	(.020)	(.121)
Recipient x region 2		-.148		
		(.054)**		
Recipient x ln(rd density)				-.022
				(.016)

Standard errors in parentheses; *** p<0.01, ** p<0.05, * p<0.1. Dependent variable = ln(1+area deforested). All regressions also include regional dummies and a constant.

Table 5: Substitution effects

	Matching estimation+			Regression with 80% best matches++					
	Within	1 km	5 km	Within	Within	1 km	1 km	5 km	5 km
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)
Ejidos									
Recipient	0.069	-0.016	-0.022	0.122	2.791**	-0.078	0.319	-0.066	1.288
	(0.08)	(0.04)	(0.08)	(0.11)	(1.06)	(0.07)	(0.68)	(0.10)	(0.97)
Recipient x ln (road density)					-0.403*		-0.059		-0.202
					(0.16)		(0.10)		(0.14)
Pseudo R-squared				0.053	0.063	0.032	0.033	0.031	0.033
Observations	411	505	505	370	370	459	459	459	459
Private properties									
Recipient		-0.018	0.178			-0.092	0.744	0.015	2.245
		(0.03)	(0.10)			(0.10)	(1.20)	(0.17)	(2.18)
Recipient x ln (road density)							-0.126		-0.336
							(0.18)		(0.33)
Pseudo R-squared						0.099	0.100	0.062	0.064
Observations		312	312			280	280	280	280

All regressions include a constant (not shown). Standard errors in parentheses. *** p<0.01, ** p<0.05, * p<0.1.
Dependent variable is ln (1+area deforested).
+ Estimation based on the same methodology as results in Table 3. ++ Regression results are partial results. Other covariates include ln(road density), area enrolled, ln(average slope), proportion of enrolled area in semideciduous forest, proportion of enrolled area in selva, and regional dummies.

Table 6: "Price" spillovers

	1 km		5 km	
	(1)	(2)	(3)	(4)
Km^2 in PSAH program	0.008**		0.019***	
within 50 km buffer†	(0.004)		(0.006)	
Km sq x road density	-0.001**		-0.003***	
	(0.001)		(0.001)	
Top 10% area enrolled		1.919**		4.978***
		(0.746)		(1.190)
Top 10% x road density		-0.287**		-0.736***
		(0.113)		(0.180)
Area enrolled	0.003*	0.003*	0.003	0.003
	(0.002)	(0.002)	(0.003)	(0.003)
Ejido	0.285***	0.285***	0.308***	0.306***
	(0.060)	(0.060)	(0.089)	(0.089)
Ln(1+avgslope)	-0.193***	-0.193***	-0.366***	-0.360***
	(0.037)	(0.037)	(0.057)	(0.057)
Prop. of enrolled area in	-0.029	-0.022	-0.227*	-0.207*
semi-deciduous forest	(0.077)	(0.077)	(0.119)	(0.119)
Prop. of enrolled area in	0.098	0.091	0.190*	0.189*
selva	(0.072)	(0.071)	(0.112)	(0.109)
Ln(road density)	0.206***	0.150**	0.486***	0.383***
	(0.073)	(0.061)	(0.113)	(0.094)
Region 2	-0.019	-0.008	-0.058	-0.078
	(0.099)	(0.096)	(0.151)	(0.146)
Region 3	-0.040	-0.038	-0.017	-0.040
	(0.104)	(0.102)	(0.156)	(0.153)
Region 4	-0.004	-0.008	-0.043	-0.072
	(0.084)	(0.082)	(0.130)	(0.128)
Constant	-1.340***	-0.994***	-2.262***	-1.575***
	(0.462)	(0.384)	(0.717)	(0.591)
Observations	817	817	817	817
Pseudo-R-squared	0.0648	0.0665	0.0447	0.0488
Marginal effects of program on Pr(d>0)				
At 25% rd density	.015	.110	.040	.189
	(.019)	(.058)*	(.018)**	(.055)***
At 75% rd density	-.031	-.065	-.023	-.085
	(.019)	(.058)	(.018)	(.054)
Marginal effects of program on Ln(d|d>0)				
At 25% rd density	.009	.068	.069	.329
	(.012)	(.036)*	(.031)**	(.097)***
At 75% rd density	-.019	-.041	-.040	-.148
	(.012)	(.036)	(.032)	(.095)

† Measured as square km in columns 1,3, and 5; measured as a dummy variable for being in the top 10th percentile of surrounding area enrolled for columns 2,4, and 6. Marginal effects for columns 1, 3, and 5 are calculated using a 1 standard deviation increase in enrolled area within 50 kilometer buffer.

Table 7: Summary statistics by PSAH enrollment in surrounding 50 km

Variable	Lowest 90% surrounding enrollment	Highest 10% surrounding enrollment	Test for difference
Enrolled area	7.81	12.36	2.86
Proportion ejidos	0.59	0.83	4.42
Ln(Average slope of enrolled area)	2.37	2.47	1.24
Average elevation of enrolled area	1980	1922	.59
Proportion enrolled area semideciduous	0.28	0.21	1.51
Proportion enrolled area selva	0.28	0.43	3.46
Ln(Road density in 50 km radius)	6.55	6.58	0.40
Proportion in region 2	0.21	0.05	3.57
Proportion in region 3	0.31	0.18	2.69
Proportion in region 4	0.20	0.51	6.61
Observations	724	93	

Figure 1: Location of recipients and control properties

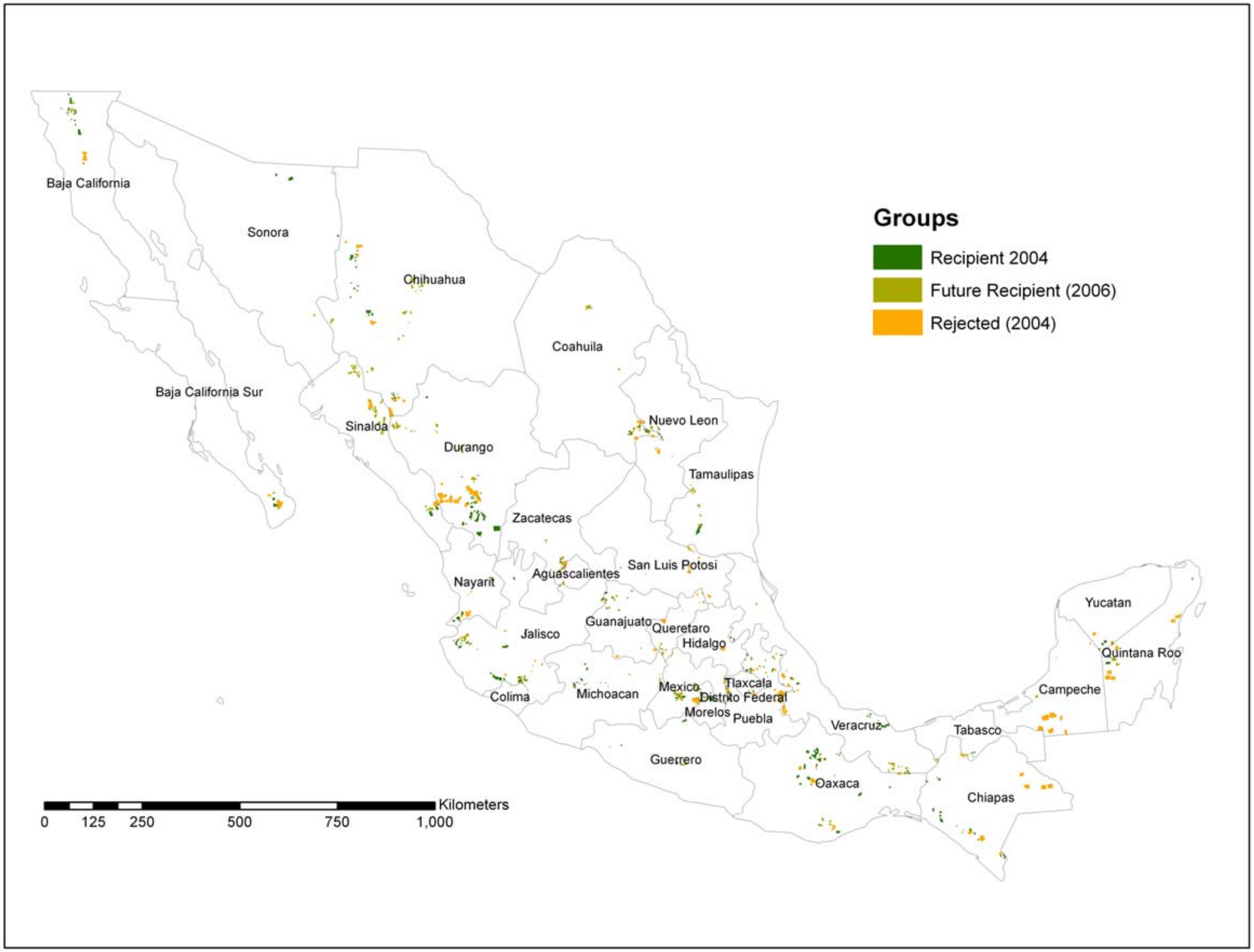
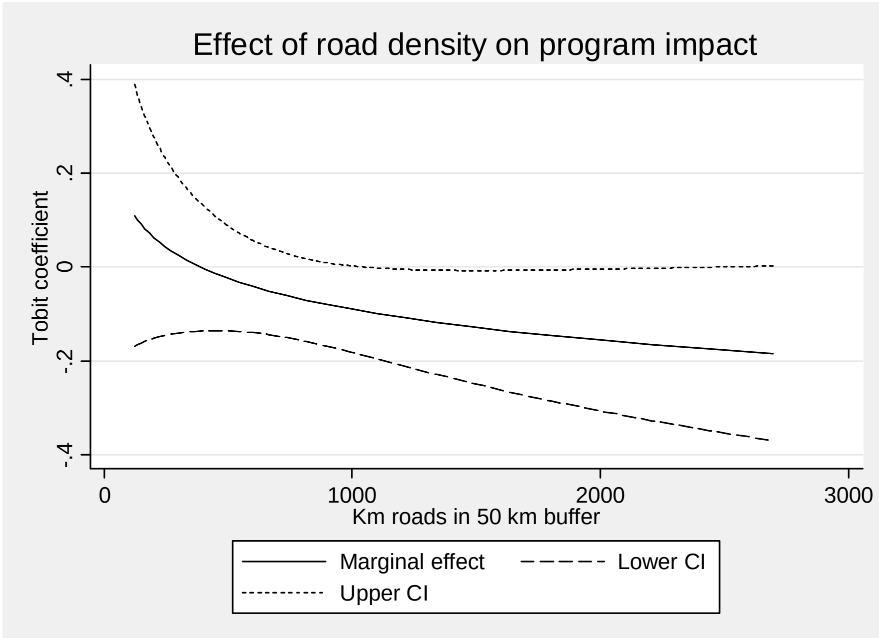
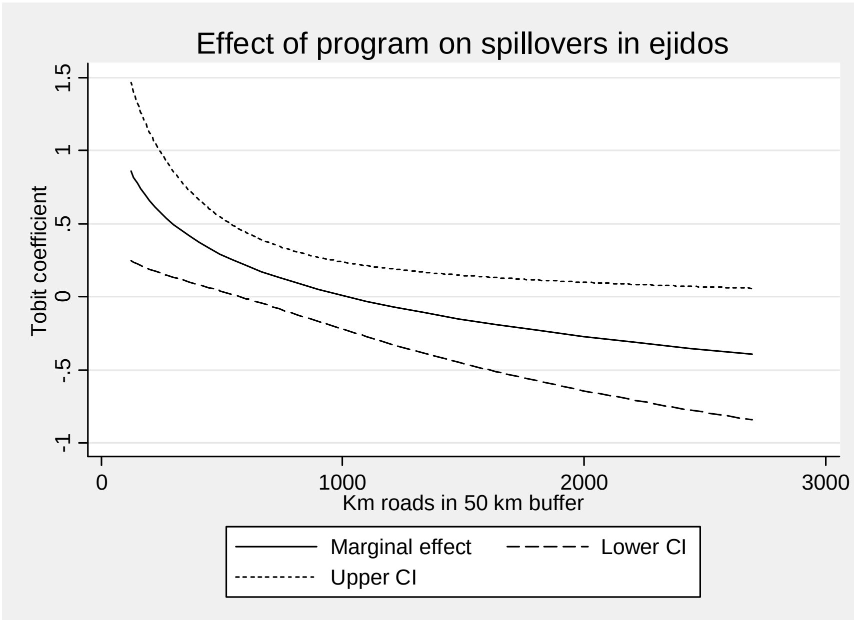


Figure 2: Heterogeneity in program impacts by road density



Tobit coefficients by road density based on regression in Table 4, Column 4.

Figure 3: Within ejido slippage effects by market access (road density)



Tobit coefficients based on regressions in Table 5, Column 5.

Figure 4: Output price slippage effects by road density

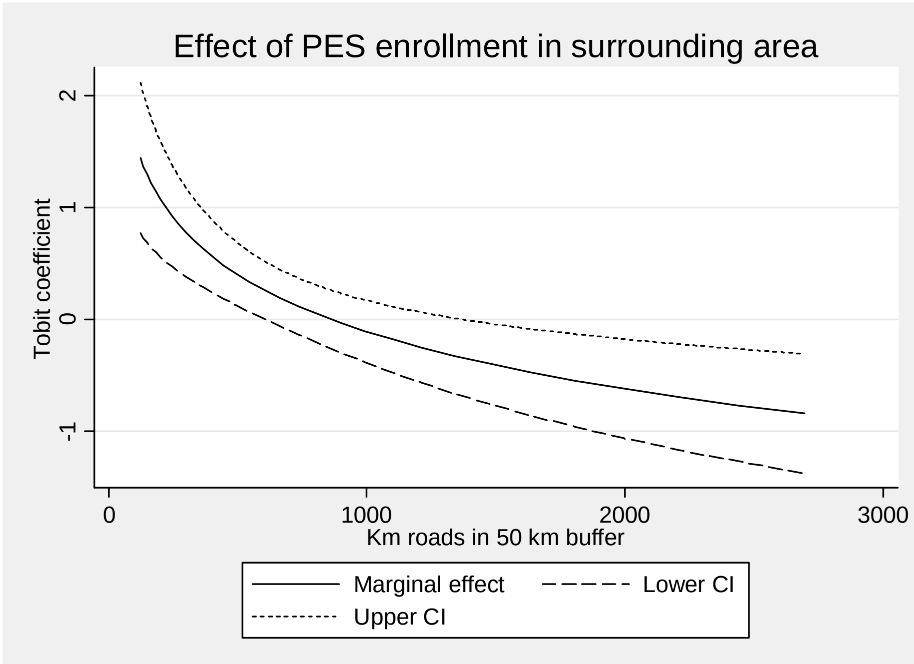


Figure shows coefficients based on regression in Table 6, Column 4.

REFERENCES

Abadie, A. and Imbens, G. (2002) “Simple and Bias-Corrected Matching Estimators for Average Treatment Effects” NBER Working Paper No. T0283

Abadie, A., Drukker, D., Herr, J., and Imbens, G. (2004) “Implementing matching estimators for average treatment effects in Stata” *The Stata Journal* 4 (3): 290-311.

Alix-Garcia, J., A. de Janvry, E. Sadoulet, and J.M. Torres. (2005) *An Assessment of Mexico’s Payment for Environmental Services Program*. A report for the United Nations Food and Agriculture Organization (FAO).

Alix-Garcia, J., de Janvry, A., Sadoulet, E., and Torres-Rojo, J-M. (2008a) “Lessons learned from Mexico’s payment for environmental services program.” In *Managing Environmental Services in Agricultural Landscapes* eds David Zilberman, Randy Stringer, Leslie Lipper, and Takumi Sakuyama, Springer Press.

Alix-Garcia, J., de Janvry, A., and Sadoulet, E. (2008b) “The Role of Deforestation Risk and Calibrated Compensation in Designing Payments for Environmental Services.” *Environment and Development Economics*, 13(3): 375-394

Alix-Garcia, J., McIntosh, C., Sims, K., and Welch, J. (2010) “Development and Deforestation in Mexico: Impacts Using the Discontinuity in Eligibility for Oportunidades” Working paper, Department of Agricultural and Applied Economics, University of Wisconsin, Madison.

Arriagada, R., E. Sills, S.K. Pattanayak and P. J. Ferraro. (2008) "Private landowners, public payments, and forest cover in Costa Rica: evaluating the impact of payments for ecosystem services." Working Paper, North Carolina State University. Presented at the annual meetings of the European Association of Environmental and Resource Economics, Gotenburg, Sweden.

Berck, P. and W.R. Bentley. (1997) "Hotelling's theory, enhancement, and the taking of the Redwood National Park" *American Journal of Agricultural Economics*, 79 (2): 287-298

Bond, I., Grieg-Gran, M., Wertz-Kanounnikoff, S., Hazlewood, P., Wunder S. and A. Angelsen (2009) "Incentives to sustain forest ecosystem services: A review and lessons for REDD" International Institute for Environment and Development, UK.

Chomitz, K. (2002). “Baseline, leakage and measurement issues: how do forestry and energy projects compare?" *Climate Policy* 2: 35-49

Corbera, E., and K. Brown. (2008) “Building Institutions to Trade Ecosystem Services: Marketing Forest Carbon in Mexico.” *World Development* 36 (10): 1956-1979.

CONAFOR. (2009) “México impulsa acciones contra el cambio climático a través del pago por servicios ambientales.” in C. N. F. d. México, editor. Comisión Nacional Forestal de México, Zapopan, Jalisco.

Crump, R.K., Hotz,V.J, Imbens,G., and Mitnik, O. (2009) “Dealing with Limited Overlap in Estimation of Average Treatment Effects.” *Biometrika*, 96:187–99.

de Janvry, A. and Sadoulet, E.. (2006) “Making Conditional Cash Transfers More Efficient: Designing for Maximum Effect of the Conditionality.” *World Bank Economic Review*, 20:1-29.

Engel, S., Pagiola, S. and S. Wunder (2008) "Designing Payments for Environmental Services in Theory and Practice: An Overview of Issues." *Ecological Economics* 64(4): 663-674.

FAO (2005) *Global Forest Resources Assessment 2005.* http://www.fao.org/forestry/fra/fra2005/en/

FAO. (2007) *The Global Environmental Facility and Payments for Ecosystem Services. A Review of current initiatives and recommendations for future PES support by GEF and FAO programs, b*y P. Gutman & S. Davidson, WWF Macroeconomic for Sustainable Development Program Office. Payments for Environmental Services from Agricultural Landscapes- PESAL Papers Series No.1 Rome.

Feng, H., Kling, C.L., Kurkalova, L.A., Secchi, S. and P.W. Gassman. (2005) ”The Conservation Reserve Program in the Presence of a Working Land Alternative: Implications for Environmental Quality, Program Participation, and Income Transfer." *American Journal of Agricultural Economics*, 87(5): 1231-1238.

Ferraro, P.J. and Simpson, D. (2002) "The Cost-Effectiveness of Conservation Payments." *Land Economics*, 78(3): 339-353.

Foster, A. D. & Rosenzweig, M. R. (2003) “Economic Growth And The Rise Of Forests”, *The Quarterly Journal of Economics* 118(2), 601–637.

Fraser, I. and R. Waschik (2005) “Agricultural Land Retirement and Slippage: Lessons from an Australian Case Study”. *Land Economics* 81 (2): 206 – 226.

Gan, J. and B. McCarl (2007) “Measuring transnational leakage of forest conservation”. *Ecological Economics* 64: 423-432

Gertler, P. J., Martinez, S. & Rubio-Codina, M. (2006), "Investing Cash Transfers to Raise Long Term Living Standards", World Bank Policy Research Working Paper No. 3994.

Guirkinger, C. and Boucher, S. (2008) “Credit Constraints and Productivity in Peruvian Agriculture." Working Paper, Dept. of Agricultural and Resource Economics, UC Davis.

Goodwin, B. and Smith, V. (2003) “An ex post evaluation of the conservation reserve, federal crop insurance, and other government programs: program participation and soil erosion” *Journal of Agricultural and Resource Economics* 28(2): 201-216.

Ho, D., Imai, K., King, G. and E. Stuart (2007) "Matching as Nonparametric Preprocessing for Reducing Model Dependence in Parametric Causal Inference." *Political Analysis* 15(3): 199-236

Honey-Roses, J., Baylis, K. and I.R. Ramirez. (2010) "The contribution of payments for environmental services versus regulation in preventing deforestation: a case study of the Monarch Butterfly Conservation Fund." Abstract, University of Illinois, Urbana-Champaign.

Heckman, James J., Hidehiko Ichimura, and Petra E. Todd (1997) "An Econometric Evaluation Estimator: Evidence from Evaluating a Job Training Programme” *Review of Economic Studies*, 64(4): 605onom

International Union for Conservation of Nature. (2009) “Reducing emissions from deforestation and forest degradation (REDD) in developing countries.” Position paper for UNFCCC Climate Change Talks, June 1-12, 2009 at Bonn.

Jack, K., Kousky, C., and Sims, K.R.E. (2008) “Designing Payments for Ecosystem Services: Lessons from Previous Experience with Incentive-based Mechanisms” *Proceedings of the National Academies of Sciences*, 105: 28, 9465-9470.

Landell-Mills, N. and I. T. Porras (2002) *Silver Bullet or Fools' Gold? A Global Review of Markets for Forest Environmental Services and their Impacts on the Poor.* London, Instruments for Sustainable Private Sector Forestry Series. International Institute for Environment and Development.

Lichtenberg, E. (2004) "Are Green Payments Good for the Environment?" *Agricultural and Resource Economics Review* 33, 138-147.

Lichtenberg, E. and Smith-Ramirez, Ricardo (2010) "Slippage in Conservation Cost Sharing." Working paper, Department of Agricultural and Resource Economics, University of Maryland, College Park.

Lubowski, R.N., Bucholtz, S., Claassen, R., Roberts, M.J., Cooper, J.C., Gueorguieva, A. and R. Johansson. (2006) "Environmental Effects of Agricultural Land-Use Change: The Role of Economics and Policy." USDA Economic Research Service, Research Report 25.

McAfee, K. & E.N. Shapiro. Forthcoming. “Payment for ecosystem services in Mexico: Nature, neoliberalism, social movements and the state.” *Annals of the Association of American Geographers, July 2010.*

Muñoz-Piña, C., Guevara, A., Torres J.M. and J. Braña. (2008) "Paying for the hydrological services of Mexico's forests: Analysis, negotiations and results." *Ecological Economics* 65:4 725-736

Murray, B.C., McCarl, B.A. and H. Lee (2004). "Estimating Leakage from Forest Carbon Sequestration Programs." *Land Economics*, 80(1): 109-124.

Murray, B., Sohngen, B., and M. Ross (2007). “Economic consequences of consideration of permanence, leakage and additionality for soil carbon sequestration projects.” *Climate Policy* 80: 127 – 143.

Pagiola, S. and Holden, S. (2001) "Farm household intensification decisions and the environment." In *Tradeoffs or synergies? Agricultural intensification, economic development and the environment,* edited by Lee, D. R. and Barrett, C. B.

Pagiola, S., Arcenas, A., and Platais, G. (2005) “Can payments for environmental services help poverty? An exploration of the issues and evidence to date in Latin America*.” World Development* 33 (2): 237-253.

Pagiola, S. and Zhang, W. (2010) "Using PES to implement REDD." Conference paper for the 4th World Congress of Environmental Economists, Montreal, CA.

Pattanayak, S.K., Wunder, S. and P.J. Ferraro (2010) "Show me the money: do payments supply ecosystem services in developing countries?" Forthcoming, *Review of Environmental Economics and Policy.*

Pfaff, A. Robalino, J. and G.A. Sánchez-Azofeifa, G. 2008. Payments for environmental services: empirical analysis for Costa Rica. Working Paper Series SAN08-05. Terry Sanford Institute of Public Policy. Duke University, Durham, NC. 25p.

Plantinga, A. and Richards, K. (2008) "International Forest Carbon Sequestration in a Post-Kyoto Agreement" Discussion Paper 08-11, The Harvard Project on International Climate Agreements.

Robalino, J., Pfaff, A., Sanchez, A., Alpízar, F., León, C. and Rodríguez CM., (2007) "Deforestation Impacts of Environmental Services Payments: Costa Rica's PSA Program 2000-2005" *Resource for the Future and EfD Working Paper Series* DP24

Robalino, J. and Pfaff. A. (2009) "Contagious Development: Neighbor Interactions in Deforestation" Submitted working paper.

Roberts, M. J. and S. Bucholtz. (2005) "Slippage in The Conservation Reserve Program or Spurious Correlation? A Comment" *American Journal of Agricultural Economics* 87: 244-250

Roberts, M. J. and S. Bucholtz. (2006) "Slippage in The Conservation Reserve Program or Spurious Correlation? A Rejoinder" *American Journal of Agricultural Economics* 88: 512-514

Roberts, M.H. and R. Lubowski (2007) "Enduring Impacts of Land Retirement Policies: Evidence from the Conservation Reserve Program." *Land Economics* 83(4): 516-538.

Schwarze, R., J.O. Niles, and J. Olander. (2002) Understanding and managing leakage in forest-based greenhouse-gas-mitigation projects. *Philosophical Transactions: Mathematical, Physical and Engineering Sciences* 360 (1797), Carbon, Biodiversity, Conservation and Income: An Analysis of a Free-Market Approach to Land-Use Change and Forestry in Developing and Developed Countries (Aug. 15, 2002): 1685-1703. The Royal Society.

Shapiro, E.N. (2010). *Political Economy and Community-Level Impacts of the Mexican Federal Payment for Ecosystem Services Programs*. Doctoral dissertation, Environmental Science, Policy & Management, University of California, Berkeley.

Sills, E. O., Pattanyak, S. K., Ferraro, P.J., Arriagada, R., Ortiz, E. and Cordero, S. (2007) "Testing Pigou: Private Provision of Public Goods" *Working paper, presented at AERE*, Mystic CT.

Sills, E., Arriagada, R., Ferraro, P., Pattanayak, S., Carrasco, L., Ortiz, E., Cordero, S., Caldwell, K. and K. Andam. (2008) "Impact of the PSA Program on Land Use." Prepared for: Ecomarkets: Costa Rica's Experience with Payments for Environmental Services, Gunars Platais and Stefano Pagiola, editors

Singh, I., Squire, L., and Strauss, J. (1986) *Agricultural Household Models: Extensions, Applications, and Policy*. Johns Hopkins University Press for the World Bank.

Sullivan, P., Hellerstein, D., Hansen, L., Johansson, R., Koenig, S., Lubowski, R., McBride, W., McGranahan, D., Roberts, M., Vogel, S. and S. Bucholtz. (2004) "The Conservation Reserve Program: Economic Implications for Rural America." Agricultural Economic Report No. AER-834, USDA Economic Research Service.

Uchida, E., Rozelle, S. and J. Xu. (2009) "Conservation Payments, Liquidity Constraints, and Off-Farm Labor: Impact of the Grain-for-Green Program on Rural Households in China."

Uchida, E., Xu, J. and S. Rozelle (2005). "Grain for Green: Cost-Effectiveness and Sustainability of China's Conservation Set-Aside Program."

United National Framework Convention on Climate Change (UNFCCC). (2009) "Copenhagen Accord" Conference of the Parties Fifteenth Session, Dec 18, 2009, Copenhagen.

Wear, D.N. and B.C. Murray (2004). “Federal Timber Restrictions, Interregional Spillovers, and the Impact on U.S. Softwood Markets”. *Journal of Environmental Economics and Management* 47 (2): 307 - 330

Wu, J. (2000) “Slippage Effects of the Conservation Reserve Programs” *American Journal of Agricultural Economics* 82: 979-992.

Wu, J. (2005) “Slippage Effects of the Conservation Reserve Programs: Reply” *American Journal of Agricultural Economics* 87: 251-254.

Wu, J., Zilberman, D. and B.A. Babcock (2001) "Environmental and Distributional Impacts of Conservation Targeting Strategies." *Journal of Environmental Economics and Management*. 41: 333-350.

Wunder, S. (2007) “The Efficiency of Payments for Environmental Services in Tropical Conservation” *Conservation Biology* 21(1): 48-58.

Xu, Z., M. T. Bennett, R. Tao and J. Xu (2004) "China's Sloping Land Conversion Program Four Years on: Current Situation and Pending Issues" *International Forestry Review* 6(3-4): 317-326.

Xu, Z., J. Xu, X. Deng, J. Huang, E. Uchida and S. Rozelle (2005) "Grain for Green versus Grain: Conflict between Food Security and Conservation Set-Aside in China " *World Development* 34(1): 130–148.

Zwane, A. P. (2007) "Does poverty constrain deforestation? Econometric evidence from Peru" *Journal of Development Economics* 84(1), 330–349.