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RESEARCH ARTICLE

Agricultural subsidies: cutting into forest conservation?

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Abstract

We examine how agricultural subsidies may induce deforestation and interact with conservation programs by analyzing two large-scale national programs in Mexico that have existed simultaneously for more than a decade: an agricultural subsidy for livestock (PROGAN) and a program of payments for ecosystem services (PES). Looking across the entire Mexican landscape, we exploit the surprises in the timing of enrollment in PROGAN's waves, fluctuations in program payments, and the change in the value of the subsidy induced by inflation and currency fluctuations to identify the impacts of the livestock subsidy on environmental outcomes. We find that PROGAN increased municipal deforestation by 7 per cent. The deforestation effects of PROGAN were smaller in municipalities with higher concentrations of PES recipients. We suggest that livestock subsidies could be better targeted to places with low deforestation risk and high livestock productivity to maximize food production and minimize negative externalities caused by deforestation.

Keywords: agricultural subsidies; deforestation; livestock; payment for environmental services; policy targeting

JEL classification: Q15; Q56; Q28

1. Introduction

There are US$700 billion spent globally by governments on agricultural subsidies each year (OECD, 2020). Under certain conditions, these subsidies may have unanticipated impacts on environmental outcomes, especially native vegetation cover (Pfaff, 1999; Angelsen, 2010; Abman and Carney, 2020). While reducing deforestation has been shown to be a cost-effective way to decrease carbon emissions (Wunder *et al.*, 2008; Busch *et al.*, 2019) and is promoted by international agencies through policies like payments for environmental services (PES), agricultural subsidies may increase opportunity costs of standing forest and reduce the cost-effectiveness of forest conservation. Since agriculture contributes 23 per cent and land-cover change 13 per cent of anthropogenic greenhouse gas emissions (IPCC, 2021), understanding the connections between agricultural policies and environmental outcomes is an essential part of climate change policy (Jaime *et al.*, 2016; Abman and Carney, 2020). It is unclear whether the conflicting goals of these programs may undermine each other, leading to inefficiencies in government expenditures. The question of whether agricultural subsidies encourage or discourage deforestation, and how they interact with conservation efforts, is central to the debate surrounding economic development and the environment.

We study these questions in Mexico, which has a history of both national agricultural subsidies and of cutting-edge conservation efforts. Between 2003 and 2014, the Mexican federal government invested on average 1 per cent of its budget in subsidies to promote agriculture, and simultaneously in payments to mitigate deforestation. In 2020, Mexico was ranked in the top ten countries for deforestation, with about 300,000 hectares of primary forest lost, and the 67th country in terms of deforestation rate (World Resources Institute, 2021). We begin by estimating the impact of a broad-scale, per head livestock subsidy (called PROGAN) on forest loss, and examine the mechanisms that drive its effect. In a second stage, we examine whether PROGAN has enrollment effects on the Mexican PES (and vice-versa), and if enrollment of forests in the PES program ameliorates the impacts of PROGAN on the environment.

To identify the livestock subsidy impacts, we use data from 12 years of both programs across all municipalities with measurable forest cover in Mexico. The 2166 municipalities in our analysis include 40 million hectares of forests in 2000, which corresponds to an area larger than Germany or Japan. Our identification strategy relies on unexpected fluctuations in program payments, arbitrary program rules, surprises in the timing of enrollment, and variation in the value of the real Mexican peso.[1] We use two-way fixed effects to control for municipality and time characteristics. Our treatment measurement is continuous, so identification relies on changes across time in the intensity of enrollment within municipalities. Although it is not possible to test whether changes in enrollment intensity are uncorrelated with unobservables, comparison of pre-program trends between municipalities with high and low subsidy payment intensity suggests that this approach leverages a valid counterfactual. Our identification also relies on the assumption that there are no anticipatory effects. An estimation including the lead of the treatment variable supports this assumption.

Relative to the average municipality-level forest loss rate of 0.29 per cent, we find that increases in subsidy intensity increase municipal forest loss by 7 per cent. Focusing only on years when enrollment is fixed and payment levels vary due to program rules or currency fluctuations, this number is 11 per cent. These findings suggest that the program either encourages extensification directly, or raises the marginal productivity of non-forest uses such that pasture expansion is optimal for producers. The extensification hypothesis is supported by analysis of the choice to plant fodder crops, i.e., crops that would facilitate raising more animals on the same land. Indeed, we find that higher levels of PROGAN enrollment are associated with a decrease in the planting of fodder crops. Further, state-level aggregates of average cattle weight, another proxy for productivity, suggest that the cattle subsidy has not differentially increased productivity in

[1] This last source of variation is similar to the approach used by Richards *et al.* (2012) to examine the impacts of currency fluctuations on soybean expansion and deforestation in Brazil.

states where enrollment is high. While we cannot rule out other mechanisms, we do not find evidence that the program encouraged intensification, and we do observe that it increased deforestation.

We then proceed to examine the interaction of PROGAN with the PES program. The budget of PROGAN dwarfs that of PES.[2] The two programs overlap, but producers do not appear to be substituting one program for the other – not infrequently, they enroll in both. The evidence suggests that enrollment in one program does not affect the choice to enroll in the other, and that the PES program reduces PROGAN's deforestation effect. For a municipality with average participation in PROGAN and without any participation in PES, deforestation rates increased by 11 per cent. However, the increase in deforestation for municipalities with both average PROGAN and PES enrollment was 8 per cent. In a supplementary analysis of land submitted to the PES program, we find that PROGAN generally increases deforestation on unenrolled land, but that the PES payments offset this effect on enrolled land.

To illuminate how the distribution of the programs might be improved, we provide a simple analysis that focuses on the stated goals and targeting criteria of both programs. Since most of the deforestation caused by PROGAN occurred in municipalities with high underlying deforestation risk, a more narrow targeting of the livestock subsidy may reduce its negative impact on standing forests and its interactions with the forest conservation program. Because the PES program is generally targeted to areas with some level of deforestation risk and because PROGAN would generate more benefits in municipalities with greater livestock productivity, we identify the potential set of municipalities where PROGAN could maximize the benefits (i.e., livestock production) and minimize the environmental costs (i.e., deforestation).

The literature shows that programs giving cash transfers (such as PROGAN), rather than physical inputs to increase productivity, may increase the risk of unintended consequences. This is consistent with the finding that cash transfers increase deforestation in some developing country contexts (Alix-Garcia *et al.*, 2013; Heß *et al.*, 2021), although this need not always be the case (Ferraro and Simorangkir, 2020). In addition to speaking to the question of cash transfers and deforestation, this paper contributes to three other strands of literature: environmental effects of agricultural subsidies, payments for ecosystem services, and the interaction between agricultural and environmental policies.

Global policymakers are concerned about the possible effects of agricultural subsidies on the environment (Goodwin, 2011), yet there is limited empirical evidence in low and middle income countries. Estimates of the impact of agricultural policy on environmental outcomes can be divided into two categories: first, changes in crop type or demand for inputs, such as nitrogen fertilizer, that may harm the environment (Hendricks *et al.*, 2014; Weber *et al.*, 2016), and second, increases in agricultural productivity that may affect natural resources (Cohn *et al.*, 2014; Assunção *et al.*, 2017; Abman and Carney, 2020). Analyses focusing on input use and runoff externalities tend to find negative effects on water quality in high income settings, although some work shows that crop insurance can reduce chemical use (Smith and Goodwin, 1996). The Common Agricultural Policy (CAP) was found to have no effect on forest cover in Crete (Lorent *et al.*, 2009). The evidence on the effects of changes in agricultural policies on deforestation in low income settings is mixed. Research in Malawi demonstrates positive effects on

[2]This is not unusual in developing countries. In the Brazilian Amazon and in Indonesia, the subsidies for environmental conservation are exceeded by far by those that promote agriculture (Dempsey *et al.*, 2020).

forest conservation of an agricultural subsidy that delivers inputs to subsistence farmers (Fisher and Shively, 2005; Chibwana *et al.*, 2012; Abman and Carney, 2020). In this case the mechanism for conservation appears to be an increase in both agricultural productivity and income. Increases in agricultural productivity are also associated with decreased deforestation in Brazil (Assunção *et al.*, 2017). However, analysis of the Green Revolution demonstrate conflicting effects for forests: Stevenson *et al.* (2013) found that it saved land from being deforested and Foster and Rosenzweig (2003) that it increased deforestation. We contribute to this work by examining the situation of livestock subsidies in Mexico, a large, middle-income country, which has labor, credit, and transport market frictions shared by many developing countries.

The literature evaluating PES targeted at forested land has generally shown positive impacts on forest conservation, with some heterogeneity. For example, estimates from Mexico show that PES reduced deforestation by 50 per cent in enrolled parcels (Alix-Garcia *et al.*, 2012, 2015), in Costa Rica's, by 11 to 17 per cent (Arriagada *et al.*, 2012), by 50 per cent in Brazil (Simonet *et al.*, 2019), and results from a randomized trial of PES in Uganda estimate a reduction of 50 per cent (Jayachandran *et al.*, 2017). In the United States, the estimated impacts of the American Conservation Reserve Program vary widely, but generally indicate conservation (FAPRI, 2007; Wu and Weber, 2012; Parker and Thurman, 2018).[3] Current evidence suggests that PES conservation impacts are reduced by the spillovers they generate (Wu, 2000; Roberts and Bucholtz, 2005; Alix-Garcia *et al.*, 2012; Robalino *et al.*, 2017), and that unconditional conservation payments may result in poor conservation outcomes (Wilebore *et al.*, 2019). These studies generally do not address other policies that might be influencing outcomes, beyond trying to rule out confounding variation. For example, higher spillovers are found in areas with higher agricultural returns (Robalino *et al.*, 2017). This implies that agricultural policies could increase spillovers by increasing the returns of land. Further, because it is rare that PES payments are indexed and that participants are compensated for the higher opportunity costs caused by agricultural subsidies (Jack *et al.*, 2008), such subsidies may reduce PES effectiveness. Consequently, environmental services may not be provided, either because contracts are not taken up or because compliance is partial (Ajayi *et al.*, 2012). We fill a gap in this literature by estimating whether PES reduces the deforestation impact of the livestock subsidy, and if the programs have enrollment effects on each other. We show that the cattle subsidy increases deforestation, that PES partially offsets this increase in Mexico, and that there are not detectable enrollment effects in either program.

Finally, we also contribute to work that estimates interactions between environmental and agricultural policy. Although there has been much discussion of these types of dynamics in both policy circles (see UNFCCC, 2020; United States Department of State, 2021) and in theory (Angelsen, 2010; Angelsen and Rudel, 2013), less has been established empirically (Lubowski and Rose, 2013). We have identified three analyses of multiple programs related to the analysis we present here. Jaime *et al.* (2016) found negative interactions of a market subsidy on uptake of organic agriculture, making the adoption of sustainable practices more expensive, while Meeks *et al.* (2019) show how biogas subsidies supported the operation of protected areas in the hill region of Nepal. In a related line of inquiry, Moffette *et al.* (2021) examine how two environmental policies interacted to increase cattle productivity, and subsequently reduce deforestation, in the Brazilian Amazon. Our work shows that these programs interact in important ways

[3] Alix-Garcia and Wolff (2014) provide a review of the literature on PES before 2014.

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that are not visible in the analysis of each program individually. In particular, we show that PROGAN raises the background deforestation risk, making PES effects more visible, and that PES serves to reduce the deforestation effects of PROGAN. We also suggest ways in which more judicious targeting of livestock subsidies might reduce their impact on deforestation.

2. Background and data

Here we provide an overview of this paper's trajectory. We begin by describing the data and providing essential programmatic background. In section 3, we examine the direct impacts of the PROGAN livestock subsidy on deforestation throughout Mexico, and then use the information on producer crop planting decisions to suggest a possible mechanism for the producer response. All of this analysis will be conducted at the municipal level, and the overarching strategy is to exploit variation in program take-up and payment amounts controlling for municipality and year fixed effects, with robustness checks that restrict variation to the most exogenous sources, including scheduled changes in payment levels and unanticipated currency fluctuations. In section 4, we use this same approach to understand the interactions between PROGAN and the national PES program. In this section, we begin by providing descriptive statistics of program overlap and take-up, then we study how PROGAN has influenced take-up of PES, and finally, we estimate the joint impact of these programs on deforestation outcomes. Section 5 examines the distribution of each program and compares it with their targeting criteria to determine how PROGAN could be better targeted.

2.1. PROGAN: description and data

PROGAN is a national livestock subsidy that began accepting applications in waves, starting in 2003, with a second wave in 2008 and a third in 2014.[4] According to Álvarez-Macías and Santos-Chávez (2019), the program was inspired by the idea of conditional cash transfers, popular in Mexico due to the rollout of Progresa (an educational conditional cash transfer program) in 1997. The goal of the initial wave (2003–2006) was to improve the productivity of extensive cattle production, and for this reason it was focused in tropical, arid and semi-arid zones where this activity generally occurs. Program managers intended to give applicants yearly payments for the entire duration of the wave after enrollment for the number of animals they owned at the moment of the application. However, payments were unexpectedly disrupted during the first wave, so producers only received payments for four out of five years between 2003 and 2007, and the timing of these varied considerably.

During the second and third waves of the program (2008–2013, and 2014 onwards), the focus was on expanding program coverage to include a broader set of areas and types of production beyond cattle, and on increasing the sustainability of this production. Further, the program required improving vegetation cover and increasing fodder production within their land, although the monitoring of these rules and definitions lacked clarity and guidance (more details included in online appendix A).

To broaden the program, they tried to expand the reach to smaller livestock producers by offering them larger payments starting in 2008. During the first wave, PROGAN subsidized only cattle while the second and third waves added subsidies for dairy cows,

[4]Details on the application process are in online appendix A.

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sheep, goats, and bees. The second and third waves also required commitments to environmental management activities and technical assistance to implement them. However, in practice the support for and monitoring of these commitments was extremely weak (Alvarez and Santos, 2013). In section 2.6, we examine the changing composition of enrollment over the waves included in our analysis. Table B4 (online appendix) summarizes the wave-specific variation in the livestock subsidy coming from changes in program rules and missed payments.

Data on PROGAN enrollment and payments from 2003 to 2015 comes from SAGARPA (Ministry of Agricultural, Livestock, Rural Development, Fish and Food). We use a GDP deflator (World Bank, 2017) to transform monetary values in 2008 real Mexican pesos (MXN). Although the subsidy is available at the individual landowner level, we can not match them to parcels, thus we aggregate them to the municipal level.

Our analysis is focused on livestock (we exclude bee subsidies). Ranchers receive the yearly subsidy as a function of the cattle unit equivalent (referred to as "animal equivalents"), which varies according to the type of animal (e.g., one goat is equivalent to one sixth of a cow, and consequently, the subsidy amount for one goat is one sixth of the amount for a cow) and by a metric called "the pasture coefficient," which measures how many animals a hectare of land can support in a sustainable way. Subsidy amounts are restricted by the pasture coefficient and calculated using the information provided in the National Livestock Registry. We calculate the maximum sustainable livestock for each municipality by tying the calculations for the pasture coefficient to characteristics of the land specific to each municipality (details presented in online appendix B). Total allowed number of cattle unit equivalent subsidized are high and so large ranchers receive a significant portion of the total subsidy.[5] Beyond being given to existing livestock holders and the limitation on subsidies above the amount dictated by the pasture coefficient, PROGAN had no other targeting criteria or conditionalities. It therefore operated as a cash transfer to individuals holding livestock.

During the first wave, the subsidy per animal increased by 100 nominal pesos each year (figure 1a). In 2007, there was no subsidy per animal because this is the unplanned additional year of the first wave. In 2008, the subsidy per equivalent animal dropped by 50 per cent overall, although it started to include a premium for small producers. In 2011, the subsidy represented about 5 per cent of the average value of a cow at the slaughterhouse (SIAP, 2021). Since cattle are typically ready to slaughter after at least 24 months, the subsidy represented on average about 10 per cent of its commercial value at the slaughterhouse (Skidmore *et al.*, 2022). Figure 1c graphs the expected nominal subsidy strictly calculated according to the program rules and compares it with the aggregated payments that were received by producers. Since the first wave was originally planned to occur between 2003 and 2006, with unit payment increasing each year, there is a steep increase in the expected subsidy during those years. The comparison between the expected nominal and the real subsidy reveals two key points. First, the large variation in the real PROGAN subsidy in 2007 results from a combination of increases in payment steps and delayed payments from the beginning of the first wave that were given in later years. Second, the greater variation in the subsidy per animal and uncertainty about

[5] The maximum subsidized cattle unit equivalent is 300 per producer. Starting in the third wave, the maximum was 1000 per registered group of producers. Beginning in 2008, prices were split into two categories. During the second and third waves, owners of 35 or fewer cattle unit equivalents were paid 75 and 70 MXN pesos more per unit than owners with more than 35 units. For simplicity, we excluded this detail from figure 1a.

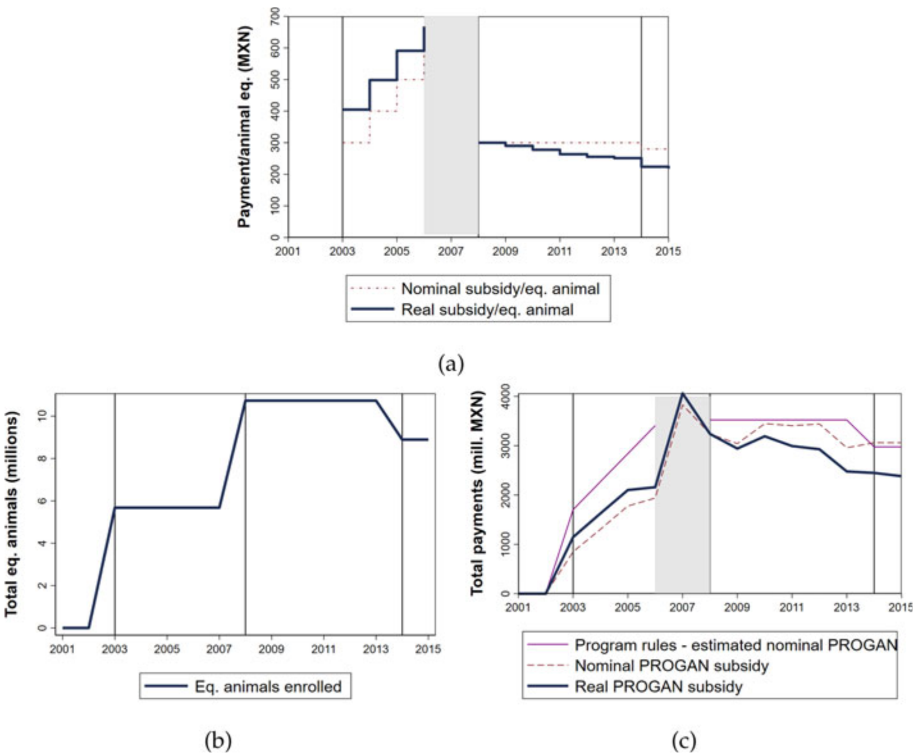

Figure 1. Nominal and real subsidy per equivalent animal in panel (a), variation in equivalent animals enrolled in panel (b), and PROGAN subsidies in panel (c). Vertical lines represent application years for PROGAN (i.e., 2003, 2008, 2014). The grey shadowed area represents the unplanned additional year of the first wave.

timing of payments suggests that the first wave contains more exogenous variation than subsequent waves. Because of this, we implement robustness checks that use only the first wave.

Variation over time in the amount of PROGAN subsidies comes from four sources: changes in (1) the subsidy per animal determined by the program, which varies within enrollment waves; (2) the timing of payments, some of which were unexpected; (3) animals enrolled in 2003, 2008, and 2014 (which varies according to the type of animals eligible and maximum allowed), and (4) the real value of the peso. While the variation from (1), (2), and (4) is arguably exogenous to individual farmer actions, (3) presents identification challenges which we will discuss further below. Figure 1 shows these sources of variation in the livestock subsidy: figure 1a presents (1) and (4); figure 1b presents (3), and figure 1c presents the aggregated subsidy which includes all four sources of variation. The treatment variable we use throughout this analysis is depicted by the thick line in figure 1c.

The number of equivalent animals enrolled in PROGAN is about 5.7 million during the first wave (figure 1b). This initial enrollment occurred in nearly 68 per cent of the municipalities in our sample. Equivalent animals increased by almost 88 per cent for the second wave, with 22 per cent of municipalities enrolling for the first time, and a large

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part of the increase was due to increases in livestock number in the first wave municipalities (figure B6, online appendix). In 2014, the number of equivalent animals decreased by 17 per cent, with new enrollment in only 3 per cent of municipalities. According to conversations with program managers, this is most likely due to a stronger enforcement of the maximum capacity of the land as dictated by the pasture coefficient. Seven per cent of municipalities never had any enrollment in PROGAN.

2.2. PES: description and data

Mexico's PES program, like PROGAN, began in 2003 with a program of payments for hydrological services. Application is voluntary, and payments are only awarded to applicants within eligible zones defined by the government. Eligible zones are determined by geographic characteristics chosen by the federal government and vary year to year. The amount of the subsidy depends on forest type and the government estimates of deforestation risk, and has evolved over time (see online appendix table A2), with decreases in its real value driven by inflation. We examine the arms of the PES program that focus on hydrological services and biodiversity, which are the largest and most durable.[6] Initially, targeting was based almost entirely on eligible zones and existence of forest cover, with properties having higher forest cover receiving priority. Starting in 2006, a point system was implemented to allocate the subsidies to the applicants. Scores are calculated according to a number of criteria that have also evolved over time, such as deforestation risk; surface water scarcity; indicators about whether the property falls in an over-exploited aquifer, a Natural Protected Area, or a municipality with majority indigenous population or with high poverty; and other factors. Applicants with the highest scores are approved until the state budget limit is reached. This means that the threshold of points for acceptance into the program varies by state-year, and by subprogram within state-year. Participants sign a five-year contract, during which payments are conditional upon forest preservation and improvements in forest management (e.g., building fire breaks). Monitoring is done via satellite imagery and follow-up live inspection in the case of suspected violations. Payments are revoked or reduced in the event of contract violations. The evolution of program targeting is discussed in detail in Sims *et al.* (2014) and Alix-Garcia *et al.* (2019).

Data for the federal PES comes from the agency managing the program, the Mexican National Forestry Commission (known by its Spanish acronym CONAFOR). CONAFOR provided digital maps of the parcels, which detail the location and boundaries for all land submitted to the program from 2003 to 2015. For our main analysis, this data is aggregated to the municipality level. An analysis at the parcel level is provided in online appendix F.

Our main treatment variable for the PES program is ratio of recipient hectares per municipal area. We use this rather than the monetary value of payments per municipality hectare because this measure is a more direct proxy for the contract that producers make regarding their deforestation behavior. However, this measure, unlike the treatment variable for PROGAN, does not change with the rate of inflation or the exchange rate. Any effects of the relative value of payments will already be realized in our enrollment levels measure. Figure B5 (online appendix) shows the delineation of

[6]Through the years, there have also been programs for agroforestry, carbon capture, and natural regeneration of the forest. However, the number of properties enrolled in these has been quite small and these programs often did not follow the same rules of operation as for the larger programs.

the eligible zones over time for the PES program and the geographic distribution of the applicants accepted (in black) and rejected (in red). Accepted and rejected parcels tend to be near each other, and they are distributed evenly throughout the country and the eligible zones. Over time, the eligible zones have expanded considerably, so there is significant variation in the number of years in which specific land has been eligible.

The ratio of PES applicant hectares at the municipality level per eligible zone is an outcome of interest to measure whether PROGAN affected the willingness to apply in PES. To control for characteristics of eligible zones land that could confound the willingness to enroll in PES and PROGAN, we calculate the following for each municipality and year: per cent in common property, in natural pasture in 2002, in pasture associated with livestock production, as well as road density and the area of the zones. The National Institute of Statistics and Geography (INEGI) provides data on the baseline pasture (INEGI, n.d.-a) and the road network (INEGI, n.d.-b). The percentage within communal land is from the PHINA (n.d.). Because the zones change from year to year, these aggregates vary across time.

2.3. Sample

In all of our analysis, we restrict ourselves to municipalities that had at least 50 ha of measured forest cover in 2000 (the baseline year of our forest data). The reason for this is that our measure of environmental impact is deforestation, which cannot be detected if there is no measurable forest. We also restrict ourselves to municipalities that had at least 50 ha of eligible zones for the PES program on average over the sample period, although the online appendix shows results with a sample unrestricted by eligible zones. This is because we intend to measure the extent of the competition between agricultural and environmental subsidies. Although the PES program does pay for the conservation of arid ecosystems, many of these may not have forest as measured by our remotely sensed outcome. Because we cannot assess environmental impacts in these areas, we exclude them from our analysis. This restriction limits us to 2,166 municipalities out of the 2457 total municipalities in Mexico. Despite the continuous expansion of the eligible zones across the country, PES payments were only ever allocated in 887 municipalities during our study period.

2.4. Deforestation

To analyze the impact of the livestock subsidy on deforestation, we use tree cover loss information from 2001–2014 and forest cover metrics for 2000 from Hansen *et al.* (2013) at the level of the PES parcels and the municipality.[7] An area is defined as forested in 2000 if its canopy cover was greater than 50 per cent, and deforestation is conditional on the area being forested in 2000.

To examine heterogeneity as well as to conduct program targeting analyses, we create a deforestation risk measure at the municipal level. We use a sample of nearly 80,000 $5\,km \times 5\,\mathrm{km}$ grid cells created by the authors to predict the probability of deforestation using pre-program data (i.e., 2001–2002). Rather than choosing a set of variables

[7]The dataset has a spatial resolution of 1 arc-second per pixel, or approximately 30 meters per pixel at the equator. In Mexico, this is an average of 710 m^2 per pixel. Forest cover loss is defined as a stand-replacement disturbance.

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to predict deforestation, we use machine learning. Specifically, we apply a probit lasso estimator, which relies on both regularized estimation and data-driven choices of the regularization parameter. It has an advantage over OLS for prediction in that it helps to minimize overfitting (Ahrens *et al.*, 2018). The choice of a probit also ensures that the predicted value remains between 0 and 1, which is useful when the user is interested in the predicted probability, rather than the marginal effects of particular covariates. Covariates included are baseline forest cover, distance to city, road density, distance to nearest urban area, average and standard deviation of elevation, average and standard deviation of slope, biome indicators, and state indicators (estimation equation and results are presented in online appendix table B3). The deforestation risk measure is equal to the municipal average of the predicted probability of deforestation of each grid cell included in the municipality.

2.5. Livestock intensification outcome

To proxy for intensification of livestock production, we examine a measure of transition from non-fodder crops into fodder production. The planting of fodder crops allows producers to raise more animals on less land, and with proper choice of fodders can improve animal and pasture health, as well as reducing the carbon footprint of raising livestock (Ates *et al.*, 2018). The increase of fodder use and productivity of livestock production was a major focus for PROGAN throughout its various waves (Álvarez-Macías and Santos-Chávez, 2019). The information on fodder crops comes from a government program called PROCAMPO, which began in 1994 as a reaction to the NAFTA agreement (Sadoulet *et al.*, 2001).[8] This program subsidizes continuous agricultural production for around 3 million producers, or approximately 14 million hectares per year. Because additional enrollment in the program was forbidden after 1993 and subsidies do not vary by crop type, producer choice is limited to which crop to produce. If they cease planting, they lose the subsidy. To obtain data on replacement of crop with fodder for each municipality and year, we aggregate the area cultivated in alfalfa, fodders, yellow corn, fodder corn, annual pasture, and forage sorghum, and divide by the total subsidized hectares of the municipality in 1999, the first year of our data. PROCAMPO covered almost all of the agricultural area in 1994 and 58 per cent of the total agricultural area in 2002 (OECD, 2005). Although it is not a census, it does cover a large part of the agricultural activity in the country. Further, it provides a detailed panel on agricultural production that includes producers of all sizes and for all crops.

2.6. Descriptive statistics

Table B5 (online appendix) summarizes a number of key variables according to whether PROGAN enrollment was, on average, above or below the median across all municipalities. Total submitted hectares of PES are lower in areas with high PROGAN and, given the relatively similar percentage of hectares submitted within eligible zones in both types of municipalities, it would appear that the eligible zones are smaller in areas with high PROGAN. We also observe higher deforestation in high PROGAN municipalities, a smaller drop in deforestation over time in areas where PROGAN is more prevalent, and a greater deforestation risk in high PROGAN municipalities. Municipalities with low and high PROGAN intensity have similar baseline forest per

[8]Program background is available in online appendix A.

municipal hectare. PROGAN tends to be higher in municipalities that have less dense population, lower slope and elevation, and also have greater replacement of crop with fodders.

Since the first wave provides a key robustness check to our analysis, we provide additional details about the randomness in payments occurring during this period. Table B6 (online appendix) shows when each of the four payments were provided over the five years. Only 40 per cent of total payments were allocated in the first year of the first wave as expected, with some producers even receiving their first payment in 2007. Overall, 5.1 per cent of the planned payments did not occur in the first wave, 7.6 per cent did not occur in the second wave, and 2.9 per cent additional payments occurred in the third wave (table B4, online appendix). When examining the spatial distribution of payments during the first wave (online appendix table B7), we show that no region benefited disproportionately from earlier payments. In sum, online appendix tables B6 and B7 show that the shifts in the timing of payments during the first wave are beyond the control of producers and occur without any particular spatial pattern.

Municipalities that enrolled in the first two waves had more potential for livestock production as measured by the average municipal pasture coefficient. This potential decreased substantially for the few municipalities that started enrollment in the third wave, and those that never received PROGAN also have relatively lower livestock potential (online appendix figures B7 and B8). Underlying deforestation risk is highest in municipalities that enrolled first, and drops significantly for those beginning in the second wave, increasing slightly for the third wave and never enrolled municipalities. We test statistically how characteristics change by cohort and intensity of enrollment (table B8, online appendix). Overall, municipalities with higher PROGAN enrollment are larger, at lower slope and elevation, with higher deforestation risk and greater potential for animal production. As enrollment progressed, smaller municipalities with somewhat lower deforestation risk and less livestock potential began to enroll.

These wave-specific summary statistics imply that comparisons that exploit the enrollment choice after the first wave may be biased upwards. However, when we limit analysis to the first wave, we observe similar results to the analysis using the entire sample, suggesting that this bias is unlikely to drive our results.

3. Deforestation effects of PROGAN

This section examines the deforestation impact of PROGAN. We begin by briefly discussing our expectations. We then describe the empirical strategy, examine the assumptions that underlie the validity of the strategy, and present the results on deforestation followed by results on intensification.

What is our expectation? A standard land rent framework with higher transport costs for agricultural goods (Angelsen, 2010) yields a familiar set of predictions. Under the simplest assumption of three land uses (forest, pasture, and agriculture), this type of model suggests that landowners would choose to hold forests in areas where the rents to pasture are less than rents to forest, and livestock production would occur in other places. A per head livestock payment increases capital as a function of livestock that is already owned. This effectively lowers the cost of having livestock, and possibly releases producer credit constraints, which we assume to exist in our setting. Because there are no enforced conditionalities, producers might use this income for investment, consumption, or savings.

For credit-constrained producers, releasing the constraint may allow for investments that would have otherwise been impossible, including purchasing more livestock, productivity-increasing technology, or some other investment or consumption good. Even if producers are not credit-constrained, they may believe that enrollment will open up again in the future, and may purchase more livestock in the hopes of receiving greater future payments. This would be a risky choice, given that program renewal was not announced until very close to the next wave. Further, if there is a wealth effect from the transfer that increases the consumption of land-intensive goods, then this could lead to spillover deforestation effects. Increased demand through a wealth effect could also lead to local price increases if markets are not fully integrated, thus intensifying the spillover effect.

Given this set of relationships, if the program induces productivity-increasing investments in livestock by releasing credit constraints, it is possible that PROGAN will leave forests unaffected, since it could increase output on existing pasture. However, if this investment greatly increases the marginal productivity of land in pasture, deforestation is likely to ensue. Deforestation increases might also occur through wealth effects, but would depend upon the income elasticities of consumption of land-intensive versus other goods, and upon the degree of market integration. In a scenario where PROGAN induces additional purchase of animals using the same production techniques or increased consumption of land-intensive goods, deforestation is also likely to increase.

3.1. Empirical strategy

Because agricultural and forest productivity both depend on underlying characteristics of the land, identifying the effect of PROGAN on deforestation is challenging. However, under the assumptions that (1.1) the programmatic adjustments in payment levels, (1.2) the unexpected change in payment timing, and (1.3) fluctuations in the real value of the peso are all exogenous to characteristics of the land; and that (2) producers did not change their behavior in anticipation of enrollment years, we can approach the causal effect of the impact of changes in the value of the livestock subsidy on deforestation. The baseline estimation equation is:

$$Y_{mt} = \beta PROGAN_{mt} + M_m + \theta_t + u_{mt}, \tag{1}$$

where Y_{mt} is the percentage deforestation over baseline forest cover for municipality m and $PROGAN_{mt}$ is the municipal livestock subsidy per hectare. We prefer this measure to the direct measure of PROGAN-funded animals per hectare because the claim for exogeneity is stronger. This amount changes annually, which allows us to examine subsets of the data where no enrollment decisions are being made, and where identification of changes over time comes only from the changes beyond the control of the individual producers. In online appendix tables C9 and C10, we show estimations using only PROGAN-funded livestock per hectare as the treatment metric.

Depending on the specification, M_m is a state or a municipal fixed effect. The first controls for geographic variation across states and the second for time-invariant characteristics of the municipalities. θ_t are year fixed effects and standard errors u_{mt} are clustered at the municipal level. In the fourth specification, we restrict the sample to municipalities that ever had PES enrollment; this sample supports comparability with the parcel level PES analysis, discussed in section 4. In the last specification, we apply the baseline forest area as a weight because these areas vary substantially from municipality to municipality.

The identification of β comes from variation in PROGAN intensity within municipalities across time.[9] The implicit comparison is between municipalities with higher and lower PROGAN intensity. To examine the assumption that trends might be different in places where there was eventually high program enrollment in PROGAN, we apply two tests of differences in trends before 2003. First, we create categories based on the median level of payments over the full duration of the program. Second, we calculate the average of subsidy payments across all years. We interact these with the year trend for the two pre-program years. We observe no differential trends after conditioning on municipality and year fixed effects (online appendix table C1). Figure C1 (online appendix) shows these pretrends visually.

In the results section, we also discuss several robustness checks intended to probe these underlying assumptions, including whether the result is robust to using only the variation in payments caused by adjustments in program rules and changes in the value of the currency during the first wave (2003–2007), a test of anticipatory behavior using leaders, a falsification exercise based on randomizing the treatment multiple times, and analyses of robustness to additional controls.

3.2. Results

Table 1 shows the results of equation (1). Columns (1)–(4) present unweighted regressions and column (5) adds probability weights corresponding to the baseline forest cover area. The estimations show that the livestock subsidy increased deforestation at the municipal level. All coefficients on the livestock subsidy intensity are positive and statistically significant at the 1 per cent level. Results are robust to different specifications including unweighted (column (3)), limiting the sample only to those municipalities that ever had PES enrollment (column (4)), and weighted regression (column (5)). Using specification from column (3), we calculate that the magnitude of the average municipal intensity of PROGAN resulted in a 10 per cent increase in the rate of municipal deforestation. Using the specification from column (5), the effect is 7 per cent.[10]

The results of the impact of PROGAN on deforestation are robust to examining only the years between 2003 and 2007, when variation from number of animals enrolled is excluded (table C2, online appendix). Since the presence of never treated municipalities affords the possibility of finding estimators centered around the average true coefficient (Sun and Shapiro, 2022), we compare these municipalities to those that enrolled in the first wave. Results are robust for all waves (online appendix table C3) and for the first wave exclusively (online appendix table C4). The similar results for the first wave and all waves together implies that treatment effects may be relatively constant across cohorts, which supports the underlying assumptions for identification. We also test whether leads of the treatment variable indicate anticipatory effects of changes in the subsidy (online appendix table C5) and find none. Further, we restrict the sample to municipalities that had eligible zones for the PES program (online appendix table C6) and run the estimation on all municipalities (online appendix table C7). Finally, we conduct a falsification exercise, where we randomize the existing distribution of PROGAN "treatments" – the

[9]The continuous nature of the treatment variables means that we cannot exploit the recent advances in difference in differences estimators (e.g., Callaway and Sant'Anna, 2021).

[10]The average subsidy per ha is 0.01546 kMXN/ha (or 15.46 MXN/ha). We multiply the estimated coefficient from column (3) per the average subsidy, and divide by the pre-mean deforestation of 2002 (i.e., $(1.838 \times 0.01546)/0.29\% = 10\%$). For column (5), we do the same steps but use the weighted PROGAN treatment and the weighted deforestation pre-mean (i.e., $(1.266 \times 0.01912)/0.34\% = 7\%$).

Table 1. Regressions of deforestation on PROGAN subsidy

	Deforestation (%)				
	(1)	(2)	(3)	(4)	(5)
Subsidy/ha (kMXN)	6.3578 (0.3347)	5.6952 (0.3636)	1.8383 (0.5706)	1.2148 (0.4851)	1.2660 (0.4792)
Adjusted R2	0.027	0.144	0.361	0.249	0.501
Observations	30,269	30,269	30,269	12,110	30,269
Year FE	X	X	X	X	X
State FE		X			
Municipality FE			X	X	X
Forest cover weights					X

Note: Years 2001–2014. Unweighted pre-mean of dependent variable is 0.29 %, weighted pre-mean of dependent variable is 0.34 %. Robust standard errors are clustered by municipality. The fourth column includes only municipalities that ever had PES enrollment and the fifth includes weights for baseline forest cover.

vector of PROGAN subsidies associated with a municipality – across all municipalities 1,000 times. This is similar to the process of randomization inference (Athey and Imbens, 2017). Using this method, we find that our estimated coefficient falls very far outside the distribution that would have been expected had PROGAN enrollments been randomly allocated across municipalities (see online appendix figure C2).

Our results may suffer from estimation bias due to different municipality trends that affect deforestation rates and are correlated with enrollment in PROGAN. Online appendix table C8 shows the stability of the results to variables that may affect deforestation rates and be correlated with PROGAN enrollment. We proxy for violence with a panel dataset on murders by municipality, for credit access by the number of bank branches in the municipality, and for migration with flexible state trends since patterns of migration in Mexico are known to be driven by historical persistence (i.e., following old railroad lines and Bracero program recruitment as in Woodruff and Zenteno, 2007) and change very slowly over time (e.g., the recent increase in migration from more southern parts of the country (Riosmena and Massey, 2012)). The state-by-year interactions address not just migration, but also any other year-to-year variation within states that might be related to the capacity of state governments. None of these additional controls change the results.

Pre-program trends and results for the simple measure of animal equivalents enrolled per ha are presented in online appendix tables C9 and C10, respectively. While animals enrolled is a measure of livestock pressure at the time of enrollment, the subsidy measure includes both this initial livestock intensity and variation in the value of the subsidy.

3.3. *Intensification of livestock production*

These estimations demonstrate that the livestock subsidy increased deforestation when the amount of the payments increased. One possible mechanism to explain this effect is extensification. To examine this dynamic, we consider how the introduction of the livestock subsidy modified the crop choices within the long-standing PROCAMPO program. Specifically, we measure the impact of PROGAN on the per cent fodder cultivated on program land between 1999 and 2014. The logic is that planting fodder may allow

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Table 2. Regressions of PROCAMPO fodder (%) on PROGAN intensity

	Fodder (%)			
	(1)	(2)	(3)	(4)
Subsidy/ha (kMXN)	−20.7149 (1.3155)	−2.3503 (0.9466)	−7.8311 (1.5647)	−2.2047 (1.3254)
Adjusted R2	0.008	0.370	0.747	0.713
Observations	34,153	34,153	34,153	34,153
Year FE	X	X	X	X
State FE		X		
Municipality FE			X	X
PROCAMPO weights (1999)				X

Note: Years 1999–2014. Pre-mean of dependent variable is 1.3 %. Robust standard errors are clustered by municipality. The fourth column includes weights for the municipal area of PROCAMPO in 1999.

more livestock to be supported on less land because fodder is of higher nutritional value than extensive pasture. We use a similar specification to equation (1), replacing the deforestation outcome with per cent fodder on PROCAMPO land. As above, the identifying assumption is that changes in PROGAN enrollment are uncorrelated with other factors driving crop choice within PROCAMPO. This is untestable, but we do not find any statistical difference in crop choices during pre-PROGAN years (1999–2002) (table D1, online appendix).

Table 2, specifications (1)–(3), show that PROGAN actually *decreased* the share of fodder on PROCAMPO land. Since there is large variation in PROCAMPO baseline area across municipalities, our preferred specification is in column (4), which contains area weights. We estimate a negative effect significant at the 10 per cent level associated with the subsidy. These findings are similar to the analysis restricted to the impact of PROGAN for 2003–2007 (table D2, online appendix).

Further descriptive evidence that intensification did not occur as intended can be found in online appendix B. These figures show trends in average cattle weight over time using aggregated state data from the Service of Agri-food and Fishery Information (SIAP, 2021), in states with above versus below median levels of PROGAN enrollment. Cattle weight is a proxy for productivity, since increases in slaughter weight are associated with better feed and genetics (Terry *et al.*, 2021). The data show flat cattle weight in both high and low PROGAN states, with a simultaneous increase in 2014. We also do not observe differential trends across states in heads slaughtered, cattle price, or production. Trends for pork, a potential placebo, also show no differences across high and low PROGAN states, and exhibit similar time trends to cattle.

We therefore find no evidence that PROGAN increased intensification. There are two caveats to this. First, producers might be planting fodder on their own PROCAMPO plots and selling it to neighboring municipalities, so spillover effects might undermine identification. Second, producers may be intensifying in other ways, for example, by importing feed, improving livestock rotations, or planting fodder on non-PROCAMPO land. There are also other potential mechanisms that might drive the observed increase in deforestation. First, higher rents per hectare as a result of the program could induce land use change. A complementary dynamic might occur if the cash payments allowed producers to engage in production for which they did not have capital prior to the program.

Finally, increases in income resulting from the program could result in deforestation via increased consumption of land-intensive goods. We thus take this evidence as suggestive.

4. Agricultural subsidies and PES

This section examines the interaction of PROGAN with Mexico's national PES program. We begin with a brief discussion of how the programs might affect each other. Then we use data to describe the interaction of both programs, and we confirm that each is not measurably affecting enrollment in the other. Finally, we examine the interactions of the two programs on deforestation.

We established above that increases in the PROGAN subsidy values increase deforestation. This suggests that producers are using the subsidies either to augment the number of their livestock or that they are making other investments that favor pasture and agriculture over forest. It is also possible that there is a wealth effect that is driving increased demand for non-forest uses of land. The subsidy does not appear to finance fodder that would allow for the production of more livestock on the same amount of land. So, what happens in the presence of the PES incentive?

PES gives payments that are conditional on preserving forest, and therefore raise the value of standing forest. By themselves, they should help conserve forest in places where it otherwise might have been replaced by other uses. Whether or not this incentive interacts directly with PROGAN depends upon producers' land holdings. There are three possibilities for program interactions for those that hold forested property. Producers with small amounts of forest might have to make a choice about whether or not to enroll in the PES (or PROGAN) program at all. If it is simply more profitable to engage in their agricultural or livestock investment activities on forested land than to enroll it in the PES, they may do so, thus decreasing enrollment in PES. The process could also work in reverse, resulting in lower PROGAN enrollment in areas with high PES potential.

For producers with larger amounts of forest than the PES program minimums, PROGAN could increase the appeal of deforestation on land that is not already enrolled in the PES program, provided that land can support livestock. In this case, the additional deforestation would not take place on land submitted to the PES program but this activity would still undermine conservation by increasing deforestation in general. If producers already have land enrolled in the PES, they may break the program rules and deforest it. This would directly undermine the goals of the PES, and is the third potential outcome (analyzed in online appendix F).

To summarize, the interaction of the programs may change enrollment in the PES program, increase deforestation on non-PES land, or induce deforestation on enrolled PES land. Since PES is also a cash transfer program, if the main available investment activities involve deforestation, then the additional capital from PES may, perversely, support future deforestation. The next section describes the distribution and overlap of the land enrolled in the two programs.

4.1. Program growth and competition

Enrollment in PROGAN and PES increased substantially over our study period. The quantity of subsidies paid out to PROGAN participants dwarfs those paid for PES. The programs have less overlap in some areas than in others. Figures 2a,b compare PES applications and payments to PROGAN in municipalities with below and above median

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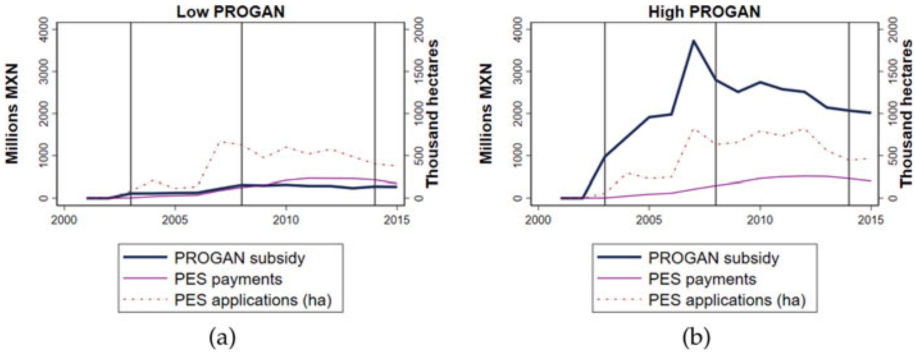

Figure 2. Total payments for PROGAN, total payments for PES, and area of land applying for PES. Vertical lines represent application years for PROGAN (i.e., 2003, 2008, 2014). Municipalities with below median levels of PROGAN are in panel (a), and above median municipalities in panel (b).

PROGAN participation. In municipalities with low PROGAN participation, PES payments increase more over time, whereas in those with high PROGAN participation, PES payments are relatively lower. Further, online appendix table B5 shows that the average area enrolled in PES is nearly twice as high in low PROGAN municipalities.

Figure A1 (online appendix) shows the geographic intensity of the PROGAN subsidy in 2008 and all PES recipients over the course of our study. While many PES recipients are located in low PROGAN municipalities in the center of the country, the map shows big overlaps, particularly in coastal municipalities. There is also evidence that these two programs operate within the very same beneficiary household or community. A 2016 survey of over 850 PES applicants between 2011 and 2014 (Alix-Garcia *et al.*, 2019) found that 36 per cent of livestock-owning households in villages that had applied for PES had received a payment from PROGAN in the previous year. Therefore, although we observe that municipalities with lower PROGAN tend to have higher PES, there are still geographies with considerable presence of both programs. The next section examines more rigorously whether there is substitution in enrollments.

4.2. Enrollment effects

If PROGAN decreases enrollment in PES, then it may indirectly reduce the amount of deforestation the program is able to avoid. This section examines the effect of the livestock subsidy on willingness to enroll in PES between 2004 and 2015. The dependent variable is the total number of hectares that apply to the PES program divided by the total hectares in eligible zones for each given year. We include characteristics within PES eligible zones to control for changes in the quality of land available for PES participation.

Generally speaking, the results presented in table 3 show that there is no statistically significant impact of PROGAN on PES enrollment, regardless of the level of controls and specification. The point estimates are large, so we cannot rule out all levels of enrollment effects. Bias might enter into the estimation through simultaneity or because there is an omitted variable driving changes in the willingness to enroll in PES that is correlated with the choice of the number of animals to enroll during the allowable enrollment years. The latter is less likely with municipal fixed effects, but might occur if there are trends, such as technological change, that drive investment in livestock differentially across municipalities with higher and lower PROGAN enrollment. To address the issue

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Table 3. Regressions of PES submitted on PROGAN enrollment

	PES submitted (% of eligible zones)				
	(1)	(2)	(3)	(4)	(5)
Subsidy/ha (kMXN)	−6.0139 (1.5916)	−2.9789 (1.8162)	−3.0356 (1.8735)	−4.6521 (3.6486)	−3.2131 (3.3019)
Adjusted R2	0.029	0.036	0.037	0.064	0.066
Observations	18,708	18,708	18,708	18,708	18,708
Year FE	X	X	X	X	X
State FE		X	X		
Municipality FE				X	X
El. zones controls			X		X

Note: Years 2004–2015. Mean of DV in 2004 is 7.3. The panel is unbalanced. Robust standard errors are clustered by municipality. Controls included in columns (3) and (5) are eligible zones-specific. They include road network, the percentage within communal land, the percentage of the municipality that is located in an eligible zone, the baseline characteristics of pasture (both natural pasture and pasture associated with livestock production).

of strategically increasing herd size prior to the year of enrollment, we include estimates with leads (table E1, online appendix). Although there are some specifications that indicate weakly significant simultaneous impacts, they are are not consistent, and the sum of the lead and contemporaneous terms are small and not statistically different from zero. We also run a regression with a balanced panel where the dependent variable is the percentage of municipal area submitted to PES (table E2, online appendix). To exploit only the exogenous variation in PROGAN stemming from programmatic adjustments in payment levels, random delays in payment timing, and changes in the value of the peso, we restrict the sample to the years 2004–2007, when no new PROGAN enrollment decisions were made (table E3, online appendix). Column (4) shows that there may have been effects up to a reduction of 4 per cent in the hectares submitted per eligible zone, although this effect disappears when we add the eligible zones controls (column (5)). If PROGAN reduces enrollment in the PES, we would be understating the interactions on the deforestation side.

Further tests reveal no evidence of reverse causality. We examine this in a regression of PROGAN enrollment on PES area enrolled (table E4, online appendix) and on PES area submitted (table E5, online appendix) for the years 2004, 2008 and 2014 using the PES eligible zones as an instrument for PES enrollment. These results suggest that PROGAN did not affect the willingness to enroll in PES and, similarly, that PES did not affect the willingness to enroll in PROGAN. This means that if the estimations in the next section show interactions between the two programs, they are unlikely to occur due to reduced enrollment in the PES program or due to simultaneity in the enrollment choice.

4.3. Deforestation effects: interactions of PROGAN and PES

Our main estimation for understanding interactions between PROGAN and PES measures deforestation at an aggregated level. This both reduces the noise that arises from imperfect satellite measures of deforestation and implicitly accounts for spillover effects that might occur within a municipality. However, its weakness is that the measure of the PES treatment is very small in magnitude at a municipal level – ranging only from zero

Table 4. Regressions of deforestation on PES and PROGAN

	Deforestation (%)						
	(1)	(2)	(3)	(4)	(5)	(6)	(7)
Subsidy/ha (kMXN)			1.8342 (0.5664)	1.1584 (0.4632)	2.1765 (0.5988)	1.8341 (0.4504)	2.0019 (0.4215)
Enrolled PES/ha (%)	−0.1713 (0.0782)	−0.1297 (0.0714)	−0.1422 (0.0779)	−0.1151 (0.0718)	−0.0151 (0.0793)	−0.0153 (0.0690)	0.2372 (0.1791)
Subs./ha (kMXN) ×					−18.6229	−15.1916	−25.2848
Enr. PES/ha (%)					(5.5178)	(4.8375)	(7.4173)
Adjusted R2	0.362	0.250	0.363	0.250	0.363	0.251	0.503
Observations	30,269	12,110	30,269	12,110	30,269	12,110	30,269
Year FE	X	X	X	X	X	X	X
Municipality FE	X	X	X	X	X	X	X
Forest cover weights							X
Municipalities ever PES		X		X		X	

Note: Years 2001–2014. Unweighted pre-mean of dependent variable is 0.29 %, weighted pre-mean of dependent variable is 0.34 %. Robust standard errors are clustered by municipality. Columns (2), (4), and (6) include only municipalities that ever had PES enrollment and the seventh column includes weights for baseline forest cover.

to 1.18 per cent of the municipal area. In online appendix F, we examine the subset of land that ever applied for the PES program. This reveals the impact of the PES program on land that was submitted for enrollment, which gives insight into the probable location of the deforestation effects induced by PROGAN.

As in equation (1), we examine regressions with state or municipal fixed effects combined with year effects and standard errors clustered at the municipality level. The dependent variable is the per cent deforestation over baseline forest cover, and we are interested in the interaction between the PES program, measured as the per cent of the municipal area enrolled in the program, and the PROGAN subsidy. In some estimations we restrict the sample to municipalities that ever had PES enrollment, and in others we apply the baseline municipal forest area as weight.

Table 4 shows the results of interactions between the two programs. Although the effect of the PES alone is not our main focus, we include two estimations that show just the PES effect (columns (1) and (2)), and two columns without interactions between the programs. This table and tables with just state effects are available in the online appendix. All columns are unweighted with the exception of column (7). The even columns restrict the sample to only those municipalities where there was ever PES enrollment.

Higher levels of PES are associated with lower deforestation in both the main and restricted samples. Including the PROGAN subsidy does not substantially change the independent PES impact. However, adding the interaction term between the two programs does: these terms are all negative and statistically different from zero. The effect of PES where there is zero PROGAN enrollment is not statistically different from zero, although the impact of PROGAN where there is no PES is positive and statistically different from zero in most specifications.

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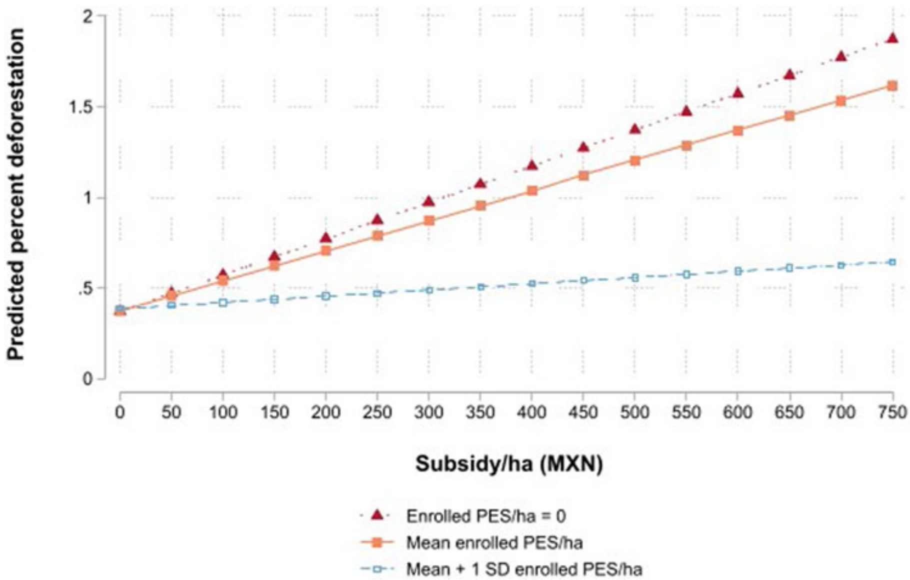

Figure 3. Marginal interaction effects of PROGAN and PES on deforestation. Marginal effects are calculated according to the specification presented in column (5) of table 4. The PROGAN subsidy/ha label represents the full range of the program. Marginal effects on deforestation are depicted for: no PES enrollment, PES/ha enrollment at the average, and given a one standard deviation increase from the average enrolled PES/ha.

A one standard deviation increase in PROGAN increases deforestation by 1.838*0.154 = 0.283 percentage points (column (3), table 1). The effect of a one standard deviation increase of PES alone (column (1), table 4), is $-0.171 * 0.367 = -0.063$. The sum of these is an increase in deforestation of 0.22 percentage points. The interaction term (column (5)) shows that deforestation resulting from PROGAN is lower in municipalities where there is higher PES. However, this effect was not sufficiently large on average to eliminate the increase in deforestation from the livestock subsidy. For a municipality without PES, the average livestock subsidy increased the deforestation rate by 11 per cent.[11] In municipalities with average PES coverage, the effect of the average livestock subsidy was 3 percentage points lower (extracted from column (7), weighted average PES = 0.022). To visualize these results, we present the interaction effects of PROGAN and PES on deforestation in figure 3. A one standard deviation increase in PES would result in a reduction of deforestation nearly compensating the deleterious effects of PROGAN.

The results are robust to restricting the analysis to the first PROGAN wave. In this case, the deforestation effect of the subsidy and the deterrent effect of PES were about 50 per cent bigger in the first round than for our whole study period (online appendix table E6).

Online appendix F details our analysis of all the land that was submitted to the PES program over this period at the parcel level. Using rejected land as a counterfactual for land that was enrolled, we estimate the interaction between the municipality aggregate

[11] We multiply the average subsidy per ha (0.01546 kMXN/ha) by the estimated coefficient from column (5) by the average subsidy, and divide by the year 2002 deforestation: $(2.17 \times 0.01546)/0.29\% = 11\%$.

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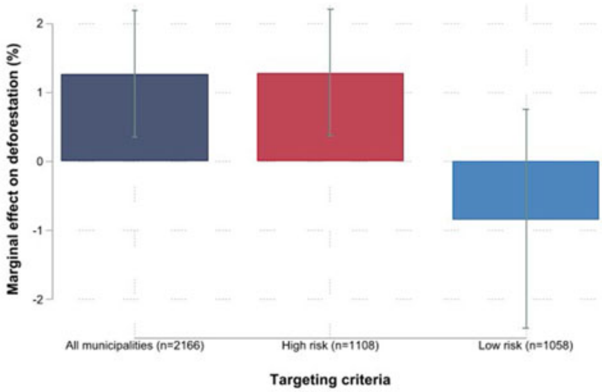
measures of PROGAN and enrollment in PES at the parcel level. The estimates show that PROGAN increases the probability of deforestation on land that was submitted but not enrolled in the program. The evidence demonstrates that on enrolled land, PES payments served to offset this increase in the probability of deforestation, but were not sufficient to produce additionality beyond that. The combination of results implies that deforestation from PROGAN is occurring both on unenrolled PES land and also on forest that was never submitted to the program.

5. Can PROGAN be targeted to support livestock and also limit deforestation impacts?

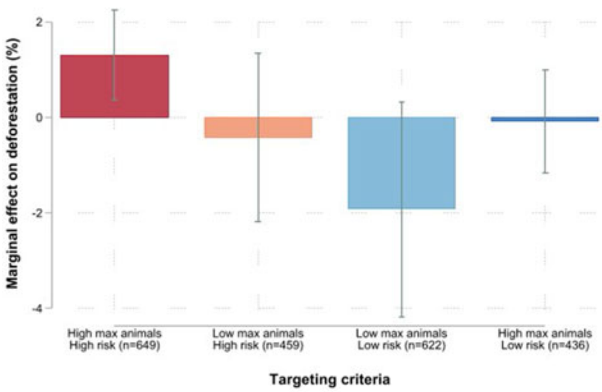
Because these programs may be working at cross-purposes in terms of environmental outcomes, this begs the question of whether it is possible to adjust the structure of PROGAN in order to minimize its deforestation footprint while also supporting livestock owners. There are many possible structural changes – for example, PROGAN could promote specific technologies rather than giving lump-sum transfers per livestock head. Adjustments of agricultural policy to limit environmental impacts have become increasingly popular in high income countries. Reforms in the United States (1996, 2002) and in the European Community (1992, 2003) provide examples of changes in regulation where farm programs have contained environmental conditionalities (e.g., conserving wetlands, highly erodible lands, etc.) (Baylis *et al.*, 2008, 2022; Claassen *et al.*, 2017). This section discusses the question of targeting. In particular, we examine whether a different spatial distribution of PROGAN could direct payments areas with high pasture potential and low deforestation risk.

Without knowing the policymaker's objective function, it is impossible to know what kind of program distribution would be optimal. We provide here a simple analysis founded on the stated goals and targeting criteria of both programs. The current structure of PROGAN payments is based upon the maximum sustainable number of animal equivalents per hectare. If we assume that a higher number according to this measure indicates greater potential livestock productivity, and if the goal of the program is to increase livestock productivity in a sustainable way under the same budget, more PROGAN payments should be given to places with a higher maximum. If most forests provide similar environmental benefits, PES payments should be targeted to places where deforestation risk is higher, since those are areas where environmental services are threatened and additionality in avoided deforestation is greatest. Indeed, it has been found in other work that the program is more effective in areas at higher deforestation risk (Alix-Garcia *et al.*, 2015). The question is whether or not these programs are currently distributed according to these criteria, and if there is a space where deforestation risk is low and maximum animal equivalents are high.

Figure G1 (online appendix) shows the relationship between program distribution and variables that proxy for these priorities. For the former, we exploit the pasture coefficients from SAGARPA to calculate an average maximum sustainable number of livestock per hectare within the municipality and for the latter we use the average predicted deforestation risk aggregated to the municipal level. In both cases, the relationship between recipients per hectare and these targeting criteria is positive. However, there is significant noise around these relationships. There is a large number of places with high potential for livestock and low deforestation risk that could be good targets for PROGAN (figure G2, online appendix), and also municipalities with very high concentration of PROGAN payments (above the 75th percentile) and relatively low livestock potential.



(a)



(b)

Figure 4. The linear combination that examines the optimal targeting of PROGAN based on whether the municipality is above or below the average deforestation risk, as well as above or below the average maximum sustainable animals. Linear combinations for panel (a) come from table 1 for the first bar and from column (2) of online appendix table G1 for the other bars. Linear combinations for panel (b) come from column (6) of table G1.

If we sum up total PROGAN allocations, over 62 per cent of them went to municipalities with below median livestock potential.

Do environmental outcomes change in municipalities with high deforestation risk or animal potential? Figure 4 examines this possibility by estimating the interaction between PROGAN, maximum sustainable animals, and deforestation risk. The outcome of interest is municipal deforestation and the specification parallels the municipality regressions, with municipal and year fixed effects. There are two main findings here. First, all of the deforestation induced by PROGAN is taking place in municipalities with high risk of deforestation. Second, payments in areas with high livestock potential and low deforestation risk do not increase deforestation, and there are quite a few municipalities that fall into this category. There are also areas with high livestock potential and high deforestation risk. Whether or not these areas should continue being prioritized by PROGAN depends upon the policymakers' objective function. PES payments alongside PROGAN payments may induce producers to sustainably intensify livestock production

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in areas with high deforestation risk. Without higher PES payments in these places, however, additional avoided deforestation is unlikely. Policymakers would be trading off conservation for livestock producer welfare goals.

The FAO's evaluation of PROGAN in 2013 notes that there is a lack of complementarity between PROGAN and other programs run by SAGARPA and other ministries charged with natural resource conservation, which offers low hanging fruit for improving program management (Alvarez and Santos, 2013). These small targeting changes could represent a simple programmatic improvement that could reduce PROGAN's environmental impact.

6. Conclusion

This paper examines the interactions between agricultural subsidies and conservation. The agricultural subsidy increased municipal deforestation rates by 7 per cent on average. We also illustrate how the subsidy interacts with a conservation program. Producers are taking advantage of both types of supports – enrollment in one does not preclude enrollment in the other. We are limited in our ability to observe substitution within properties due to the fact that our livestock subsidy data is not sufficiently spatially explicit. However, the results that we have suggest that cash transfers for livestock have a deleterious effect on forest outcomes, particularly in forest that is not enrolled in the PES program.

The combination of the increase in deforestation rate, a decrease in the measurable proportion of fodder planted, and no change in the weight trends of cattle associated with greater PROGAN intensity suggests that the subsidy is not encouraging intensification. It is not obvious that intensification will lead to better environmental outcomes, since it may lead to an increase in pasture land due to higher marginal productivity (Villoria *et al.*, 2014). However, it is fairly certain that extensification increases deforestation. Although we cannot rule out the possibility that the program increased productivity of cattle production, it is also not controversial to point out that accompanying future livestock supports with training in sustainable animal production may achieve superior results in terms of both productivity and environmental impact.

The environmental subsidies do help limit some of the impacts of PROGAN, but this sort of program dueling is probably inefficient, and could be avoided by more careful targeting of both livestock subsidies and environmental supports. Adjusting targeting to municipalities or areas with lower deforestation risk and higher livestock potential would reduce its environmental impact and might even end up encouraging production in areas where producers have higher return livestock production systems. The targeting of the PES program uses eligible zones in a way that could easily be replicated for PROGAN – areas with high animal production potential and low deforestation risk could be prioritized in the distribution of payments. However, without any conditionality or training in sustainable production methods, the second outcome may not occur (Bragança *et al.*, 2022). Certainly more complicated targeting schemes could also be created. For example, one might create a set of targeting criteria that directly use metrics of animal production potential and deforestation risk at a property level. The PES program does such specific property-level targeting within eligible zones using a points scheme for each application. However, refined targeting stategies also come with increased administrative costs. Given limited administrative resources and a goal of distributing as much support as possible to rural areas, even a simple approach could result in significant improvements.

We would be remiss if we did not mention that improving the sustainability of meat production is a key ingredient in the constellation of policies to support climate change mitigation. Many experts propose demand-side interventions to reduce meat consumption (Willett *et al.*, 2019), however, supply side interventions are likely to play a substantial role. Our analysis raises questions about the benefits of non-targeted agricultural subsidies, and suggests limiting policies that increase the extensive agriculture rent as part of next actions to limit climate change and as part of the next REDD+ agenda (Angelsen and Rudel, 2013). In a world where many of the places contributing large shares of carbon emissions from land use are poor, thinking carefully about policy designs that respect livelihoods as well as conservation outcomes is the main path to sustainability.

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