Heterogeneous Local Spillovers from Protected Areas in Costa Rica

Juan Robalino, Alexander Pfaff, Laura Villalobos

Abstract: Spillovers can significantly reduce or enhance the net effects of land-use policies, yet there exists little rigorous evidence concerning their magnitudes. We examine how Costa Rica's national parks affect deforestation in nearby areas. We find that average deforestation spillovers are not significant in 0–5 km and 5–10 km rings around the parks. However, this average blends multiple effects that are significant and that vary in magnitude across the landscape, yielding varied net impacts. We distinguish the locations with different net spillovers by their distances to roads and park entrances—both of which are of economic importance, given critical local roles for transport costs and tourism. We find large and statistically significant leakage close to roads but far from park entrances, which are areas with high agricultural returns and less influenced by tourism. We do not find leakage far from roads (lower agriculture returns) or close to park entrances (higher tourism returns). Finally, parks facing greater threats of deforestation show greater leakage.

JEL Codes: Q23, Q24, Q28

Keywords: Heterogeneous effects, Leakage, Protected areas, Spillovers

ACCORDING TO THE WORLD DATABASE ON PROTECTED AREAS, protected areas (PAs) cover 12% of the earth's surface, and establishing such areas is the most common approach to reducing deforestation. Thus, understanding the impacts of

Juan Robalino (corresponding author) is at the University of Costa Rica and Centro Agronómico de Investigación y Enseñanza (robalino@catie.ac.cr). Alexander Pfaff is at Duke University (alex.pfaff@duke.edu). Laura Villalobos is at the University of Gothenburg (laura.villalobos@economics.gu.se). We have received financial support through project Tropi-dry II CRN3 of the Inter-American Institute for Global Change Research, the Tinker Foundation, the NASA Large Scale Biosphere-Atmosphere Experiment in Amazonia, the Latin American and Caribbean Environmental Program, and the Swedish International Development Cooperation Agency through the Environment for Development Initiative of the Department of Economics, University of Gothenburg. We are grateful for the support. For helpful comments, we thank seminar participants at the 2012 AERE conference in Asheville, NC; the 2014 Heartland Workshop on Environmental and Resource Economics in Urbana-Champaign, IL; the Global Land Project; Re-

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PAs on deforestation is important for future conservation policy (see Bruner et al. 2001; Andam et al. 2008; Pfaff et al. 2009; Joppa and Pfaff 2010a; Sims 2010; Blackman, Pfaff, and Robalino 2015; and Robalino et al. 2015). Most analyses to date examine the realized deforestation impacts of PAs only within the borders of those areas. However, it is well known that net forest impacts of PAs can depend significantly on PA impacts outside their borders.[1]

There are numerous hypotheses about how PAs might affect nearby rates of deforestation. Some argue that land-use restrictions, rather than halting development, would only shift it to unprotected areas nearby (Leathers and Harrington 2000; Wu 2000; Fraser and Waschik 2005; Wu 2005; Armsworth et al. 2006; Robalino 2007; Alix-Garcia, Shapiro, and Sims 2012). Just the expectation alone of an increase in land-use restrictions could lead nearby landowners to deforest, in order to lessen the chance of any new such restrictions (Newmark et al. 1994; Fiallo and Jacobson 1995). These hypotheses suggest that PAs could increase deforestation in nearby areas. If the magnitude of such impacts in nearby areas were large, they would constitute spillovers, and such spillovers could fully offset deforestation reductions in PAs.

However, PAs such as national parks might instead decrease the deforestation in nearby areas. Protection could generate incentives for ecotourism near parks, which increases the returns to forest conservation outside of, but near, PAs. In addition, some have argued that PAs increase environmental awareness (Schelhas and Pfeffer 2005), which in turn can lessen deforestation. There is evidence showing that the deforestation decisions of neighbors in Costa Rica reinforce each other, so that private land-use choices to conserve forest can shift incentives for nearby private land use toward additional forest conservation (Robalino and Pfaff 2012). Should PAs generate such spillovers to private land-use choice, then the extent to which they promote conservation may be underestimated.

Empirical estimates of the spillover effects of parks on local forests could reflect combinations of such multiple land-use interactions. Further, the magnitude and sign of the net spillovers may vary across space, generating heterogeneous spillover effects. Our hypothesis is that PAs increase net nearby deforestation in one location but lower it in another, and that both effects are hidden behind an average zero effect. Indeed, a large enough single PA could even generate different net effects around its border.

We examine deforestation spillovers from Costa Rica's national parks from 1986 to 1997, the most recent period during which deforestation rates in Costa Rica were significant. To go beyond prior empirical work (see Andam et al. 2008), we distinguish the forest locations that are near PAs by distances to the nearest road and nearest park

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1. Spillovers generated by other environmental policies, such as a carbon tax, have been documented as well (Baylis et al. 2014).

entrance, both of which are economically important. We expect the existence and intensity of spatial spillovers to vary over space, given that the relevant economic mechanisms are likely to be affected by both transport costs and the proximity to tourism.

High-resolution spatial data for forest parcels allow controlling for parcel characteristics, which are important predictors of both park siting and deforestation patterns, according to the recent literature on PA impacts. Parcels in protected areas differ significantly, on average, from forest left unprotected; Joppa and Pfaff (2009) show this globally. Thus, the forest parcels near PAs also are likely to differ, in relevant characteristics, from unprotected forest parcels to which they are compared in order to estimate spillovers.[2]

We employ matching and regression methods to address the resulting potential biases. In environmental economics, matching strategies have been used for some time for evaluations concerning, for example, the effects of air quality regulations on environmental outcomes (Greenstone 2004) and economic activities (List et al. 2003). More recently, they have been used to identify the causal effects of land-use restrictions and conservation policies on environmental and socioeconomic outcomes (see Andam et al. 2008; Pfaff et al. 2009; Joppa and Pfaff 2010b; Sims 2010; Ferraro, Hanauer, and Sims 2011; Alix-Garcia et al. 2012; Arriagada et al. 2012; Robalino and Pfaff 2012; Canavire-Bacarreza and Hanauer 2013; Robalino et al. 2015; and Robalino and Villalobos 2015).

We start by estimating average impacts for all parcels near PAs. As in Andam et al. (2008), we find that, on average, there are no significant deforestation spillovers for the entire rings of forest immediately surrounding Costa Rica's national parks. However, we argue that this finding hides multiple heterogeneous spillover effects that result from differing influences of transport costs and tourism. Thus, to further test varying spillover effects, we separate forests near protected areas using distances to roads, a factor associated with transport costs, and distances to entrances, a factor associated with tourism.

We find a 10% increase in deforestation close to roads, when far from the entrance within a 0–5 km ring from the border of the PA. Areas within the inner ring, that is, the closest forest, unaffected by tourism and with high agricultural returns, seem to capture pressure emanating from inside the parks. In locations far from an entrance in the more distant 5–10 km rings around the borders of PAs, we find no impact on deforestation rates.

Where tourism should have its greatest influence, such as in locations close to park entrances in a 0–5 km ring, we find no leakage at all, even close to roads. Yet, moving 5–10 km out from the park entrance close to roads, we do find an 8.8% increase in the rate

2. Governments pursue specific objectives when siting PAs. They might minimize conflicts with advocacy groups, or target impact (i.e., additionality) by choosing higher deforestation threats (as suggested in Pfaff and Sánchez-Azofeifa 2004), or maximize environmental benefits conditional upon impact (Costello and Polasky [2004], e.g., extend a large literature).

of deforestation. For this second ring, tourism may well not increase private forest returns, and it is even possible that it raises returns to clearing for complementary development, such as hotels for those who pay to see the forest at the park entrance.

We also test whether the deforestation pressure faced by a park affects leakage. For this, we look at a park's differing characteristics that are relevant for deforestation, and find different leakage levels. Leakage is higher when the opportunity cost is high in low-tourism areas. Leakage is significant far from the entrance of flatter parks, which tend to be subject to high deforestation pressure. In parks with steep land, this is not the case. Smaller protected areas, which tend to be in high deforestation pressure areas, generate leakage. Yet, large protected areas, which tend to be in lower pressure areas, do not.

These results show not only the potential importance of spillovers in evaluating PA impact but also the value of delineating specific mechanisms that are likely to underlie spillovers.[3] These mechanisms help to predict the expected spillover effect for a given location. Looking only at impacts inside PAs can be misleading if PAs have positive or negative spillovers in nearby forests, as in Costa Rica. Such information is highly relevant in defining the optimal siting of PAs and other complementary policies.

In section 1, we provide some background concerning both forest conservation in Costa Rica and the estimation of spillovers. We present a simple theoretical framework and a literature review of park leakage and spillover effects in section 2. In section 3, we present our data and empirical approach. We present our results in section 4, and, finally, we present our conclusions in section 5.

1. BACKGROUND

1.1. Deforestation in Costa Rica

While deforestation rates fell significantly by the end of the 1990s, during the late 1970s and early 1980s Costa Rica had one of the highest deforestation rates in the world (Sánchez-Azofeifa, Harriss, and Skole 2001). For example, between 1976 and 1980 the deforestation rate was 3.2% per year (Food and Agriculture Organization 1993), but between 1986 and 1997, the deforestation rate outside protected areas was only about 1% per year (Pfaff et al. 2009). Multiple factors help to explain the observed drop in deforestation rates between the two periods.

One set of factors concerns economics. For instance, beef prices fell while ecotourism activity rose. The profitability of other traditional Costa Rican agricultural products, such as coffee and bananas, also helps to determine where deforestation will occur. Profitability of these agricultural products is greatly affected by transport costs. Hence, roads are an important factor determining deforestation across landscapes, as confirmed empirically in other countries by Chomitz and Gray (1996), Pfaff (1999), and Pfaff et al. (2007). Naturally, another set of key factors involves state interventions. Lower defor-

3. Ando and Baylis (2014) have also noted the importance of spillovers in evaluating PA impact.

estation might result from conservation efforts, including the implementation of PAs, with impacts inside and outside their borders.

1.2. Conservation in Costa Rica

The area under protection in Costa Rica rose greatly between the 1950s and 1990s. Protected areas now cover 25% of the country. The largest category of protected areas is national parks, covering 10% of the country (Pfaff et al. 2009). One characteristic of national parks is that they receive visitors, which in turn generates related economic activities, such as rapidly increasing ecotourism. By 1995, tourism was the country's main source of foreign revenue (Inman et al. 1997), and a significant number of foreign visitors have ecotourism as their main objective, an objective that includes visiting parks.

The public decisions to establish PAs responded to multiple public and private objectives. For instance, the first conservation effort in Costa Rica took place in 1955 with a law that decreed as protected the entire area within 2 km of the crater of any volcano. By 1977, with forest cover reduced to 31% of the territory, the National Park Authority (Servicio de Parques Nacionales) considered the establishment of new PAs an urgent matter. New protected areas were established in order to protect representative portions of all life zones and all major ecosystems (Boza 2015). To this end, the goal was to protect at least 5% of the territory.

With that goal in mind, the government created many additional national parks, yet the specific characteristics of each park differ (Boza 2015). For example, high recreational, cultural, and historical value motivated the foundation of Santa Rosa National Park in 1971, while some of the biggest national parks were created to conserve geologic formations, flora and fauna, habitats and ecosystems, microclimates, life zones, watersheds, and aquifers (Rincón de la Vieja in 1973, Chirripó and Corcovado in 1975, and La Amistad in 1982). Other explicit motives for PAs include preventing the commercial and private exploitation of natural resources (Corcovado in 1975). Braulio Carrillo was created in 1978 to block the expansion of agricultural and real estate activities following ongoing urban growth and the construction of a major road. Finally, some PAs were established to protect specific species, such as the coral reefs in Cahuita in 1970, the turtles in Tortuguero in 1975, and the birds in Palo Verde in 1982 (Boza 2015).

Given these explicit conservation goals, the state also needed to take into account the opportunity costs of PAs. As noted in Pfaff et al. (2009), these opportunity costs could guide protection away from development. In Costa Rica, PAs are located farther from San José, farther from national and local roads, and on steeper lands than are unprotected forests. They are also on less productive lands (Andam et al. 2008). These characteristics are associated both with high costs of transport and agricultural production.

Still, parks in Costa Rica have reduced deforestation significantly on average, even if variably so and on the whole by a smaller magnitude than many might assume (Andam et al. 2008). Around 2% of the forests inside protected areas would have been deforested between 1986 and 1997 without protection, though the effectiveness of the protection depends on location and hence on land characteristics (Pfaff et al. 2009). PAs close to roads, close to San José, and on flatter land avoided significantly more deforestation (Pfaff et al. 2009). What remains undocumented to date—and what we attempt to establish in this paper—is the impact of protection on neighboring forests.

2. THEORETICAL FRAMEWORK AND PRIOR EVIDENCE

2.1. Simple Model of Park Leakage and Spillovers

Following Robalino (2007), we use a von Thünen framework to describe the effects of protection on deforestation in unprotected land. In figure 1, all units of land are presented, in decreasing order, by the relative profitability of clearing. The curve of the relative profitability of clearing is denoted by R_a. As long as profits from clearing are positive, that is, in $[0, f]$, the land will be deforested. Forest will remain when returns are lower than 0, beyond f.

If a park is established in the interval $[0, p]$, we assume that deforestation cannot occur within that interval. This is an assumption justified by exceptionally low clearing within Costa Rica's PAs. If a park is established in an area where agricultural profits are positive, as within this interval, agricultural production will be reduced and prices of agricultural goods will increase. This will lead to increases in rents in each location (Robalino 2007). This increase in agricultural rents is shown by the curve R'_a. Thus,

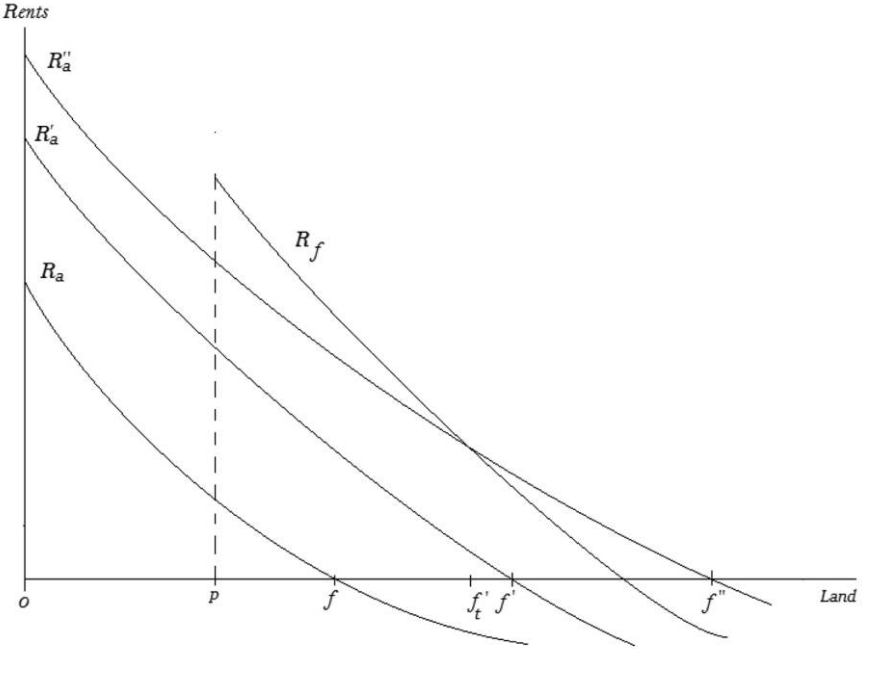

Figure 1. Location of protection, leakage, and spillovers

deforestation will take place in the interval $[p, f']$. The interval $[f, f']$ would not have been deforested without the presence of the park. This is one form of "leakage."

On the other hand, if the presence of a park increases tourism, then the profits from keeping land forested rise to R_f. We assume that the park entrance is located at p. Returns from keeping the forest will decrease as the distance to the entrance of the park increases.[4] In this case, deforestation would not occur in some locations beyond f due to the increases in forest profits. Agricultural products will decrease once more and agricultural rents will increase again, R''_a. The result will be that deforestation will not occur in $[f, f'_t]$, from where the park is located to where forest returns equal agricultural returns. This reduction in deforestation would not have occurred had the park not brought tourism. Parks that bring tourism create a "halo" in unprotected areas. In the figure, we also see that deforestation occurs in the interval $[f'_t, f'']$. Two effects of opposite signs are taking place. The net effect depends on the magnitude of the increase in forest profit, on the magnitude of the final increase in agricultural rents at f, and on the difference in the slopes between agricultural and forest returns.

This simple model has three empirical implications. First, if there are no alternative activities that increase forest returns due to the creation of a park, increased deforestation outside PAs will occur. Protected areas without tourism generate deforestation outside those areas. Second, the locations where such increases in deforestation will take place are the most profitable remaining lands to deforest. Lands with low profits will remain unaffected. Third, if the park increases forest returns, the effects on deforestation outside the PA are ambiguous. Profits favor forests close to the entrance of the park, but also favor agriculture far from the entrance of the park. The sign of the overall effect depends on the magnitude of the increase of forest returns, on the magnitude of the increase in agricultural returns, and on how fast they decrease from the entrance of the park and from the market.

2.2. Previous Empirical Evidence of Deforestation Leakage

As noted above, various hypotheses exist for how parks might affect deforestation in nearby areas. They involve environmental awareness (Schelhas and Pfeffer 2005), displacement of deforestation toward nearby areas (Cernea and Schmidt-Soltau 2006), preemptive clearing to prevent a future expansion of land restrictions (Newmark et al. 1994; Fiallo and Jacobson 1995), and changes in market prices, which could have local and global effects (Armsworth et al. 2006; Robalino 2007).

However, empirical analysis of spillovers, in particular from parks, has been exceptionally limited. Globally, due to changes in market prices, restrictions on timber harvests in one region are expected to increase timber harvests in other regions (Sohngen, Mendelsohn, and Sedjo1999). There is also evidence of large leakage effects from the

4. We assume forest returns fall faster than agricultural returns. If this is not the case, no leakage will take place close to entrance at any distance. This will be tested in the empirical section.

Conservation Reserve Program involving direct payments to farmers in the United States. For every 100 hectares retired under the program, 20 hectares were converted to cropland outside of the program (Wu 2000). Other papers have also shown evidence of leakage in forest carbon sequestration (Murray, McCarl, and Lee 2004; Sohngen and Brown 2004; Chomitz 2007).

For Mexico, there is evidence of leakage from the national ecoservice-payments program (Alix-Garcia et al. 2012). Landowners who had enrolled some land in the program increased deforestation on other property holdings. This effect is stronger in poorer municipalities and with those given less access to commercial banks, where credit constraints are higher (Alix-Garcia et al. 2012).

In the case of Costa Rica, some scholars have explored park deforestation spillovers with regard to their average effects (Andam et al. 2008). Average net effects on nearby forests were seen to be insignificant (Andam et al. 2008), a result that we confirm. Yet, as we show in this paper, averages can mask significant leakage effects in certain particular areas—especially where small changes in deforestation incentives could induce clearing activity, such as forests close to roads within areas where the returns to forests due to tourism are low.

3. DATA AND EMPIRICAL APPROACH

3.1. Data

Using the spatial detail offered by high-resolution data in a GIS (Geographic Information System), we randomly drew 50,000 points, 1 per km^2, from across Costa Rica as our units of analysis.

3.1.1. Forest and Sample

We use forest-cover maps for 1986 and 1997 to determine the deforestation over that period. The maps were derived from Landsat satellite images with a 28 × 28 m resolution. They distinguish forests from nonforests and mangroves. Developed by the Tropical Science Center from aerial and satellite pictures, they indicate forest presence or absence at each point. We begin with the 50,000 points just mentioned (see table 1). We then eliminate all points that were not forested in 1986 (23,290) and all points with an uncertain presence of forest (2,759). We also drop 2,864 observations covered by clouds or shadows. That leaves us with 21,087 points (or 42% of our original 50,000 points) under forests in 1986.[5]

5. Observations covered by clouds and shadows, and points where the presence of a forest is uncertain, in principle might generate sample bias. Clouds and shadows are associated with wetter forests. The uncertainty of forest presence is associated with drier places due to the colors of the trees that cannot be distinguished from the color of land used for other purposes. This issue, which is faced by all analyses using satellite data, affects 11% of our sample. However, we are not aware of any reason for which the economic mechanisms outlined in section 2 would not hold

Table 1. Forest and Sample

	Number of Observations	Percentage
Total	50,000	100.00
Drop if there was no forest in 1986	23,290	46.58
Drop if not private land	11,607	23.21
Drop if undefined distance by roads to parks	466	.93
Drop if uncertain about presence of forest	2,759	5.52
Drop if there are clouds or shadows	2,864	5.73
1986 private forests for analysis	9,014	18.03
Ring 1: 0–5 km:	1,253	100.00
Close to entrance:	503	40.14
Close to national roads	125	9.98
Far from national roads	378	30.17
Far from entrance:	750	59.86
Close to national roads	84	6.70
Far from national roads	666	53.15
Ring 2: 5–10 km:	1,486	100.00
Close to entrance:	408	27.46
Close to national roads	92	6.19
Far from national roads	316	21.27
Far from entrance:	1,078	72.54
Close to national roads	190	12.79
Far from national roads	888	59.76
Beyond 10 km:	6,275	100.00
Close to national roads	1,093	17.42
Far from national roads	5,063	80.69
Dropped if close to the entrance (less than 20 km through roads)*	119	1.90

* We consider observations beyond 10 km as untreated. However, if they are relatively close to the entrance (less than 20 km through roads), they might be contaminated and so are dropped from the analysis.

Our focus is nonprotected private forests. Thus, we drop all points inside parks and in public areas where government determines land use, leaving 9,480 observations. Finally, because an important variable is the distance to park entrances via roads (calculated as the distance from the closest road segment to the park entrance), we also

also in these areas. This is certainly something that should be explored empirically in future research.

dropp 466 observations located farther than 5 km from the closest road segment.[6] The number of forest observations remaining is 9,014. Our dependent variable is whether a forest point in 1986 had been cleared by 1997.

3.1.2. National Parks and Nearby Areas

Maps of all protected areas in Costa Rica were digitized by the GIS Laboratory at the Instituto Tecnológico de Costa Rica. We focus on national parks because they cover the most area and they are the strictest type of protection that allows tourism. All PAs included in the analysis were created before 1986. We drop all other PAs and all points within a PA, to analyze only neighboring areas. To determine which points are the neighbors of national parks, we compute the linear distance from each forested point to each national park, and take the minimum distance. This criterion defines our "treated group."

Next, we use this distance to the park to distinguish three sets of observations (see table 1). First, we consider the 1,253 forested points that are within 5 km of the nearest park border (Ring 1). Second, we consider the 1,486 forest points that are between 5 km and 10 km from the nearest park border (Ring 2). Every treated observation will be drawn from these two sets of observations. Finally, we obtain 6,275 observations that are over 10 km from a national park (far from parks). For each test that we perform, we use this last set of observations as controls.

We define the set of observations that are "proximate" to the entrance of a national park as those with an along-the-roads distance of less than 20 km from the nearest park entrance. Within Rings 1 and 2, we split the treated observations into close to entrance (503 observations in Ring 1 and 408 observations in Ring 2) and far from entrance (750 observations in Ring 1 and 1,078 observations in Ring 2).

Finally, we distinguish closer versus farther than 1 km from a national road for each ring separately. In particular, we distinguish the following four groups: (1) close to both entrance and road (125 in Ring 1 and 92 in Ring 2); (2) far from entrance but close to road (84 in Ring 1 and 190 in Ring 2); (3) close to entrance but far from road (378 in Ring 1 and 316 in Ring 2); and (4) far from both entrance and road (666 in Ring 1 and 888 in Ring 2). All are compared with the untreated points. Of the 6,275 observations 10 km or farther from national parks, 1,136 observations are located close to national roads, while 5,139 are located far from national roads.[7]

6. We feel this is the best approach but emphasize that including those observations has no effect on our core results.

7. We dropped 119 observations that are within the 20 km distance via roads but farther than 10 km from parks linearly. These observations would have entered the control group but they might be contaminated by the treatment effect, as they are close to a park entrance via roads.

3.1.3. Parcel Characteristics

We use spatially specific information stored and manipulated within a GIS to obtain characteristics that are helpful in finding untreated points that are similar to the treated. These improved comparisons allow us to better estimate the impacts. We obtain measures of slope, precipitation, elevation, and distances to both rivers and key ocean ports. We also compute distances to San José, population centers, sawmills, and schools. Finally, we compute the fraction of forest in 1986 at the census tract level, as a measure of forest stock in the neighborhood.

3.2. Empirical Approach

In order to determine the impact of national parks on deforestation rates in neighboring areas, we must answer the question, What would the neighboring deforestation rate have been had a park not been established nearby? The simplest estimation strategy to answer this baseline question is to consider the average deforestation rate in untreated forest points, an estimator known as the "naive" estimation (Morgan and Winship 2014). In our case, this would imply comparing deforestation rates in Rings 1 and 2 with deforestation rates beyond 10 km of national parks. This approach is relatively common (Joppa and Pfaff [2010a] list some examples) but clearly inadequate if the treatment group and the untreated group differ in terms of characteristics that also affect deforestation rates.

Table 2 shows such differences. Compared to controls, parcels within 0–5 km of the nearest national park have steeper slopes, more precipitation, higher elevation, a higher census-tract share of forest in 1986, and longer distances to roads, rivers, cities, coasts, sawmills, and schools. In sum, Ring 1 points are more remote and likely to face less deforestation pressure than the average unprotected forest parcel beyond 10 km from a PA (col. 1). Ring 2 also differs from unprotected forests far from PAs but is less remote than Ring 1. The location of these groups of observations can be seen in figure 2.

Table 2 also suggests that the national parks blocked deforestation in Ring 1 but may have increased it in Ring 2. However, such differences in the observed deforestation rates might be caused by the differences in land characteristics and not by proximity to parks. We use matching and regression analysis to compare treated to similar untreated points that do not differ in average land characteristics.

Matching selects the most similar untreated observations as controls. The deforestation rate in the control group is the estimate of what would have happened in areas near parks without the parks. Compared to standard regression, which can be employed after matching, this method imposes fewer assumptions for the functional form that relates land characteristics and deforestation (Rubin 2006). For example, if the treated observations tend to be far from roads, the estimated treatment effect is likely to depend on the functional form assumed for distance to roads (e.g., linear or log-linear). Matching directly reduces the difference in distance to roads between treated and untreated, as shown below, which thereby reduces the effect of functional-form assumptions on the estimates.

Table 2. Land Characteristics and Group Mean Differences

		Treated, 0–5 km		Treated, 5–10 km	
	Untreated (Mean)	Mean	*t*-stat*	Mean	*t*-stat*
Dependent variable:					
Deforestation rate	13.42	10.61	−2.70	14.87	1.47
Control variables:					
Slope (percentage)	44.85	64.93	7.66	55.01	4.19
Precipitation (mm)	3.30	3.73	15.15	3.67	14.02
Elevation (m)	.35	.75	27.08	.43	6.53
Dist. to local roads (km)	.78	1.01	8.34	.99	8.15
Dist. to national road (km)	3.90	4.35	3.96	3.69	−2.11
Dist. to rivers (km)	1.42	1.61	4.80	1.25	−4.83
Dist. to capital city (km)	105.70	104.01	−1.14	116.42	7.81
Dist. to Pacific coast (km)	52.30	50.45	−1.44	55.70	2.80
Dist. to Atlantic coast (km)	110.23	104.99	−2.49	96.75	−6.79
Dist. to towns (km)	2.82	3.36	9.51	3.10	5.49
Dist. to sawmills (km)	18.34	22.28	11.55	22.06	11.49
Dist. to schools (km)	15.21	14.32	−2.98	13.37	−6.58
Percentage of forest 1986	52.17	58.86	9.08	55.05	4.15

* Test of means against untreated.

One important condition within matching is that the characteristics of the parcel we use reflect park siting determinants and, therefore, characteristics of surrounding areas. As documented in Pfaff et al. (2009) and discussed in section 1.1, parks tend to be located in relatively remote areas with relatively low opportunity costs. We control for distance to the capital city, roads, towns, and sawmills to account for these opportunity costs. As we also discussed in section 1.1, all volcanoes are protected. Thus, we control for altitude, which in Costa Rica is highly correlated with the presence of volcanoes. Additionally, we control for precipitation that, combined with altitude, determines the type of ecosystems and the presence of flora and fauna. As we also mentioned in section 1.1, flora and fauna are important factors in determining park location. However, we acknowledge that there could be factors that affect park location that are missing from the available variables in the data, such as the quality of institutions, as we discuss in section 5.

Matching requires a definition of *similar*. One is the distance in the characteristics' space between any two points,[8] known as covariate matching (Abadie and Imbens

8. That is, $((x_1-x_2)'\ V\ (x_1-x_2))^{½}$ where x_1 and x_2 are land characteristics for any two observations and V is a positive-definite weight matrix.

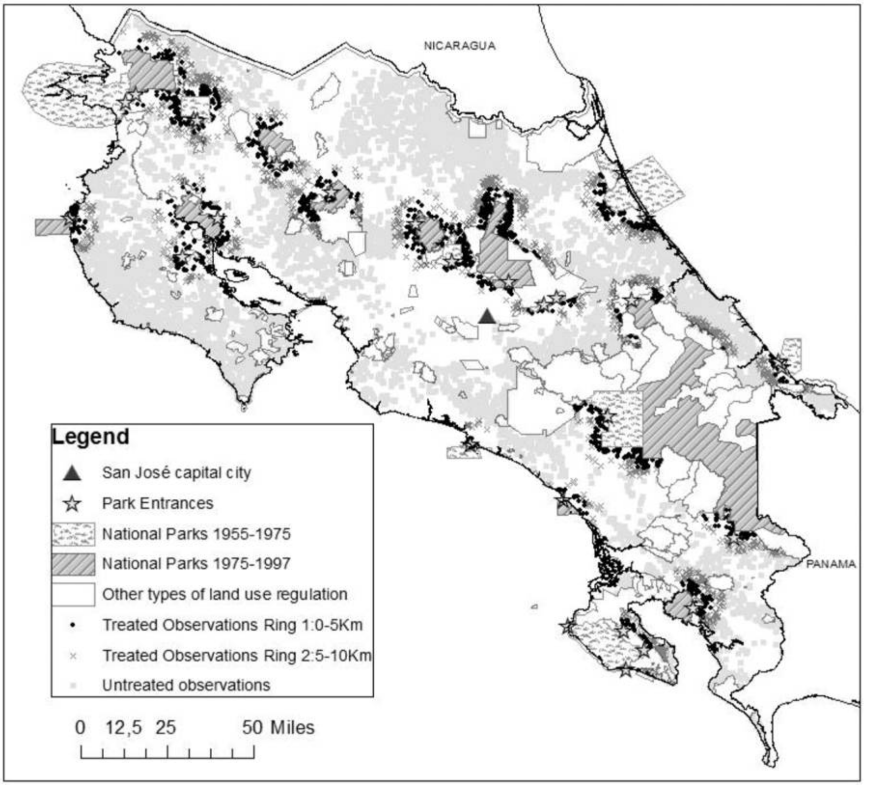

Figure 2. National parks and location of treated and untreated observations

2006). One advantage of this strategy compared to other matching estimators is that the standard errors are consistently estimated (Abadie and Imbens 2006). In table 3, we show the number of covariates that are different between treated and untreated groups both before and after matching, using a 5% significance level. Covariate matching reduces the number of unbalanced covariates for each test we perform.[9]

In sum, we aim at testing the impact of park proximity on nearby private forests. Our counterfactual in all cases is what would have happened in those private locations if the park had not been created. We estimate these effects using the observed deforestation rate for the most similar unprotected forest far from parks. For each ring of private forest near a park, we test overall and heterogeneous effects by considering (1) forest close to and far from park entrances, (2) forest close to and far from roads,

9. We also tested propensity score matching but covariate matching achieved better balances. Thus, we choose to focus on covariate matching.

Table 3. Matching Balances: Number of Statistically Different Covariates at 5% Significance Level before and after Matching

	Before Matching	After CVM
Ring 1: 0–5 km:	11	2
Close to entrance:	10	0
Close to roads	6	0
Far from roads	10	0
Far from entrance:	13	2
Close to roads	5	0
Far from roads	12	2
Ring 2: 5–10 km:	12	0
Close to entrance:	9	0
Close to roads	5	0
Far from roads	10	0
Far from entrance:	9	0
Close to roads	5	0
Far from roads	12	0
Rings 1 and 2:	11	2
Close to entrance:	9	0
Close to roads	6	0
Far from roads	10	0
Far from entrance:	11	2
Close to roads	5	0
Far from roads	10	2

and (3) all these heterogeneous effects for parks with different characteristics, always separately testing Ring 1 and Ring 2.

4. RESULTS

4.1. Local Spillovers from Protected Areas

We test first whether there are deforestation spillovers on average near national parks. The naive estimator (first two columns in table 4) reflects different mean deforestation rates for treated and untreated observations.[10] Lower deforestation rates are found in the 5 km ring, in particular close to park entrances and far from roads. In the second ring, overall we find higher deforestation far from park entrances, although this difference is not statistically significant. Still, as noted, land characteristics can explain variations in deforestation rates between the treated and the untreated observations. Thus, from this alone we cannot conclude that parks cause these differences in deforestation rates.

10. We use clustered standard errors in all naive estimates.

Table 4. Initial Estimates of National Park Impact on Nearby Deforestation

	Naive[a]		OLS[b]		ATT with OLS[a]		Covariate Matching[b] with Abadie & Imbens (2006)		Covariate Matching[a,b]	
	Ring 1: 0–5 km	Ring 2: 5–10 km	Ring 1: 0–5 km	Ring 2: 5–10 km	Ring 1: 0–5 km	Ring 2: 5–10 km	Ring 1: 0–5 km	Ring 2: 5–10 km	Ring 1: 0–5 km	Ring 2: 5–10 km
Overall effect	–.0280*	.0145	.0071	.0199	.0192	.0206	.0079	.0080	.0073	.0079
	[.015]	[.017]	[.013]	[.015]	[.013]	[.015]	[.011]	[.011]	[.017]	[.018]
Far from park entrance	–.0137	.0255	.0186	.0211	.0231	.0205	–.0001	.0059	.0029	.0066
	[.019]	[.020]	[.016]	[.018]	[.015]	[.017]	[.014]	[.013]	[.024]	[.021]
Close to park entrance	–.0515***	–.0173	–.0017	.0065	.0233	.0144	.0081	.0186	.0083	.0166
	[.018]	[.024]	[.016]	[.023]	[.016]	[.021]	[.013]	[.018]	[.019]	[.027]
Far from roads	–.0424**	.0034	.0038	.0148	.0189	.0157	.0070	.0065	.0061	.0069
	[.017]	[.018]	[.013]	[.017]	[.013]	[.016]	[.011]	[.012]	[.018]	[.020]
Close to roads	.0387	.0579*	.0280	.0420	.0390	.0413	.0435	.0235	.0421	.0227
	[.038]	[.032]	[.040]	[.033]	[.038]	[.032]	[.027]	[.028]	[.040]	[.034]

Note. Standard errors in square brackets. ATT = average treatment effect on the treated.
[a] Clustered standard errors at the census tract level.
[b] The control variables used are shown in table 1. Distances as controls are in logs.
* Significant at the 10% level.
** Significant at the 5% level.
*** Significant at the 1% level.

We include land characteristics in estimations using ordinary least squares (OLS) and covariate matching (CVM) to isolate the effect of nearby parks. For the OLS specification, we estimate the average treatment effect for all observations, as well as the average treatment effect on the treated, which is directly comparable with the CVM estimator. We clustered the standard errors at the census tract level,[11] and for CVM we also present the robust standard errors proposed by Abadie and Imbens (2006). We do not find any significant effects in either ring, whether or not we distinguish subsets by the distance to entrances or roads. This zero effect result confirms previous average spillover estimates for Costa Rica (Andam et al. 2008) and is robust to the estimation strategy.

However, this zero average effect might blend effects of different significance, magnitude, and even sign. As discussed, national parks might reduce deforestation in nearby areas under some conditions, yet raise it under other conditions. Thus in principle, the average findings in table 4 could be the result of blending overlapping and offsetting heterogeneous effects.

4.2. Heterogeneous Local Spillovers per Returns from Agriculture and Tourism

We expect greater deforestation leakage as the difference between the returns to agriculture and to forest conservation increases. A powerful determinant of agricultural returns is the distance to the nearest road. A powerful determinant of complementary touristic activities, which can raise returns to forests, is likely to be proximity to the entrance of a park. Those factors are combined in table 5.

We expect more deforestation in locations near parks when not affected by tourism, while at the same time close to roads. We find large and significant leakage effects under exactly these conditions in Ring 1, within 5 km of parks (see the first row under the Ring 1 columns). The sign of this leakage result is robust to the different strategies used. The magnitude ranges between 8.59% and 14.67%, and five out of six estimates are statistically significant.

Moreover, the forces generating that leakage seem to be absorbed in the initial ring around the parks. In the first row under the Ring 2 columns we show that there are no significant effects in Ring 2, even when close to roads (low transport costs) and far from entrances (low tourism). In the second row of table 5, we show that no impacts are found far from roads in either ring.

We might also expect that even for Ring 1 close to roads, leakage could be offset by tourism. Table 5 shows this result in the third row under the Ring 1 columns. However, if we remain close to roads but move away from the entrance (the third row under the Ring 2 columns), we again see some evidence of leakage. These spillover

11. To estimate the clustered standard errors for the CVM estimator, we run a regression with the treated and matched control observations, following Alix-Garcia et al. (2012).

Table 5. Additional Matching Estimates of National Park Impact on Nearby Deforestation

	Ring 1: 0–5 km						Ring 2: 5–10 km					
	OLS	OLS (ATT)	CVM	CVM Trimmed[a]	OLS after CVM	OLS after CVM Trimmed[a]	OLS	OLS (ATT)	CVM	CVM Trimmed[a]	OLS after CVM	OLS after CVM Trimmed[a]
	Clustered[b] Standard Errors		Robust Standard Errors		Clustered[b] Standard Errors		Clustered[b] Standard Errors		Robust Standard Errors		Clustered[b] Standard Errors	
Far from entrance:												
Close to roads	.0859	.0761*	.1039**	.1327***	.1086*	.1467**	.0269	.0256	–0.0145	–.0313	–.0135	–.0254
	[.067]	[.046]	[.043]	[.048]	[.060]	[.074]	[.042]	[.040]	[.035]	[.042]	[.047]	[.057]
Far from roads	.0110	.0209	–.0114	–.0289	–.0088	–.0250	.0192	.0189	.0072	–.0217	.0087	–.0211
	[.017]	[.016]	[.015]	[.019]	[.025]	[.029]	[.019]	[.018]	[.014]	[.015]	[.023]	[.025]
Close to entrance:												
Close to roads	–.0008	.0142	.0161	.0187	.0151	.0175	.0632	.0738**	.0882**	.1682**	.0825*	.1318
	[.041]	[.037]	[.030]	[.046]	[.044]	[.069]	[.047]	[.036]	[.041]	[.074]	[.042]	[.085]
Far from roads	.0084	.0154	.0125	.0310*	.0121	.0307	.0060	.0066	.0083	–.0149	.0075	–.0153
	[.019]	[.016]	[.014]	[.016]	[.019]	[.019]	[.030]	[.025]	[.019]	[.021]	[.031]	[.024]

Note. Standard errors in square brackets.
[a] Using only observations in the interval of 0.1 and 0.9 of the propensity score as suggested by Crump et al. (2009).
[b] Clustered standard errors at the census tract level.
* Significant at the 10% level.
** Significant at the 5% level.
*** Significant at the 1% level.

estimates are large and represent important increases in deforestation rates, with magnitudes that range from 6.32% to 16.82%. This effect could reflect other elements of tourism, such as complementary hotel infrastructure.

In sum, we split the sample of forest areas near parks into subsets, using proxies for factors that are likely to be correlated with the returns to agriculture and tourism. We find that leakage from parks can be significant when close to roads. Tourism can reduce leakage, but impacts are not fully eliminated, as increases in deforestation simply take place farther away.

4.3. Robustness

In table 6, we test whether the results are sensitive to the choice of the thresholds defining close and far from roads and park entrances. If we move the threshold that defines proximity to roads by 50 meters, our results do not change. Within Ring 1 far from park entrances, effects are large and significant when close to roads using this definition as well (panel A, first two columns, first three rows). If we change the definition of proximity via roads to park entrances by 1 km, again we still find large and significant effects (panel A, third to sixth columns, first row). The results still hold even when we combine these tests (panel A, third to sixth columns, first to third rows).

We perform the same tests for Ring 2 close to the entrance (panel B). Changes in the definition of proximity to park entrances do not affect the results. When we increase the definition of proximity to roads by 50 meters, results do not change. However, significance is lost when we test reductions of 50 meters in the definition of proximity to roads. This might be explained simply by a reduction in the sample size. The sign of the effect is still positive and magnitudes are high.

We also test proximity to park entrances as a continuous variable. Using the same samples we have used in the previous robustness tests, we use a continuous distance variable from the park instead of a discrete treatment variable. As expected, deforestation spillovers fall as distance from the park increases, both for Ring 1 far from the entrance but close to roads (see panel A, seventh and eighth columns, first row) and for Ring 2 close to entrances (see panel B, seventh and eighth columns, fourth row). We also tested different distances to roads (see seventh and eighth columns, second, third, fifth, and sixth rows). Only for the robustness test of proximity to roads in Ring 1 far from the entrance does the continuous treatment lose significance for clustered standard errors. However, we still get the same magnitude and sign.

4.4. Heterogeneity by Park Characteristics

Finally, we test whether these local spillover effects vary when the park characteristics differ. For instance, steeper parks facing less pressure to clear forestland might have different spillover effects than parks on relatively flat lands. Larger and smaller parks might also have different spillovers. These are empirical questions. In theory, the magnitude and sign of impacts will depend on how much productive land is protected, on the characteristics of nearby land, and on the presence of tourism.

Table 6. Distance and Threshold Robustness Tests

	Discrete Treatment						Continuous Treatment	
	CVM	OLS after CVM	CVM	OLS after CVM	CVM	OLS after CVM	OLS after CVM	
	Robust	Clustered[a]	Robust	Clustered[a]	Robust	Clustered[a]	Robust	Clustered[a]
	More Than 20 km in Roads		More Than 21 km in Roads		More Than 19 km in Roads		More Than 20 km in Roads	
A. Ring 1: 0–5 km, far from entrance:								
Close to roads (1 km)	.1039**	.1086*	.1051**	.1097*	.1019**	.1061*	–.0442**	–.0442*
	[.043]	[.060]	[.044]	[.062]	[.042]	[.059]	[.021]	[.026]
Close to roads (1.05 km)	.0956**	.1013*	.0921**	.0995*	.0970**	.1025*	–.0341*	–.0341
	[.040]	[.055]	[.041]	[.056]	[.039]	[.054]	[.020]	[.024]
Close to roads (.95 km)	.1036**	.1084*	.1020**	.1080*	.1023**	.1066*	–.0438**	–.0438
	[.044]	[.063]	[.044]	[.064]	[.042]	[.061]	[.022]	[.028]
	Within 20 km in Roads		Within 21 km in Roads		Within 19 km in Roads		Within 20 km in Roads	
B. Ring 2: 5–10 km, close to entrance:								
Close to roads (1 km)	.0882**	.0825*	.0788*	.0728*	.0814**	.0757*	–.0701***	–.0701***
	[.041]	[.042]	[.041]	[.042]	[.041]	[.042]	[.021]	[.026]
Close to roads (1.05 km)	.0910**	.0851**	.0865**	.0804**	.0867**	.0808**	–.0650***	–.0650**
	[.038]	[.040]	[.038]	[.040]	[.039]	[.040]	[.020]	[.026]
Close to roads (.95 km)	.0639	.0598	.0589	.0543	.0561	.0519	–.0823***	–.0823***
	[.041]	[.040]	[.042]	[.040]	[.041]	[.040]	[.021]	[.028]

Note. Standard errors in square brackets.
[a] Clustered standard errors at the census tract level.
* Significant at the 10% level.
** Significant at the 5% level.
*** Significant at the 1% level.

In the first two columns of table 7, we present the estimates of these heterogeneous effects for places where the conditions generate more leakage, which is far from the entrances and close to roads in Ring 1. Flatter parks have higher leakage effects (panel A). We also find that smaller parks have higher leakage effects than larger parks (panel B). As documented in Pfaff et al. (2009), smaller parks tend to be located in areas with a high threat of deforestation. Taken together, these results are consistent with the model presented in section 2, as flatter and smaller parks tend to have higher opportunity costs and greater levels of deforestation threat.

Close to roads and close to the entrance in Ring 2 are conditions where we also found leakage (see cols. 3 and 4 in table 7), although perhaps for different reasons, such as complementary tourism infrastructure. For this leakage location, we find opposite results compared to Ring 1 far from entrances, where a tourism mechanism is irrelevant. Here, larger parks with steeper lands have higher leakage effects than smaller and flatter parks (panels A and B). Significant tourism infrastructure might play a greater role for these parks. The entrances of the larger parks with steeper lands tend to be found in spots with relatively easier access. Therefore, these areas have higher opportunity costs than other spots within those parks.[12] This could explain why we find higher leakage in Ring 2 near park entrances than in Ring 1 far from park entrances. Such within-park differences are smaller in small parks.

Older parks also differ from newer ones in terms of local deforestation spillovers (table 7, panel C). As explained in section 1.1, older parks protect volcanoes (e.g., Poás and Irazú) and other areas with high recreational, cultural, and historical value (e.g., Santa Rosa and Manuel Antonio). Tourism activities are highly consolidated all around old parks. We even have negative coefficients, though statistically insignificant, far from the entrance in Ring 1. However, we do find leakage in Ring 2 close to entrances for older parks, a result that is again consistent with considerable tourism infrastructure. These are areas located at some distance from those parks, where deforestation does not spoil tourism directly yet provides easy access to parks. In contrast, newer parks generate significant leakage effects for Ring 1, when close to roads and far from the entrance.

5. DISCUSSION

Motivated by the observation that spillovers can significantly reduce or multiply the effects of land conservation policies, we empirically examined how national parks in Costa Rica affect the deforestation rates in forested lands near them. We used the most similar parcels that are far from parks as counterfactual comparisons in order to estimate spillover impacts. We employed the definition of similarity embedded

12. Indeed, a simple test showed that when inside the park, land near entrances is significantly closer to roads than land far from entrances.

Table 7. Splits by Park Characteristics in Areas Close to Roads: Slope, Size, and Years after Creation

	Ring 1: 0–5 km, Far from Entrance		Ring 2: 5–10 km, Close to Entrance	
	CVM, Robust (1)	OLS after CVM, Clustered[a] (2)	CVM, Robust (3)	OLS after CVM, Clustered[a] (4)
Entire sample	.1039***	.1086*	.0882***	.0825*
	[.043]	[.060]	[.041]	[.042]
A. By slope:				
Steeper by 50%:	.0182	.0206	.1422*	.1187
Standard error	[.061]	[.083]	[.075]	[.074]
Number of treated obs.	42	42	45	45
Flatter by 50%:	.1246***	.1575	−.0144	−.0071
Standard error	[.059]	[.099]	[.038]	[.041]
Number of treated obs.	42	42	47	47
B. By size:[b]				
Larger by 50%:	−.0049	.0136	.2027***	.1900***
Standard error	[.059]	[.084]	[.076]	[.071]
Number of treated obs.	42	42	42	42
Smaller by 50%:	.1726***	.1758***	.0186	.0182
Standard error	[.060]	[.088]	[.042]	[.038]
Number of treated obs.	42	42	50	50
C. By years after creation:[c]				
Older by 50%:	−.0764	−.0572	.1508***	.1392***
Standard error	[.054]	[.080]	[.058]	[.054]
Number of treated obs.	42	42	52	52
Newer by 50%:	.2295***	.2312***	−.0281	−.0292
Standard error	[.065]	[.087]	[.061]	[.055]
Number of treated obs.	42	42	40	40

[a] Clustered standard errors at the census tract level.

[b] We use the median for the split. Larger parks are those with more than 12,000 hectares, and smaller parks are those with less than 12,000 hectares.

[c] We use the median for the split. Older parks were established before 1975, and newer parks were established between 1975 and 1986.

* Significant at the 10% level.

** Significant at the 5% level.

*** Significant at the 1% level.

in covariate matching, which generated the best balance of treated and controls across the parcel characteristics.

We significantly extended the existing literature by using economic rationales concerning the returns from agriculture and tourism as a basis for splitting nearby forested lands into subsets. We expect these groups to have different net returns to forest (versus clearing) and thus to differ in the magnitude and sign of the spillover. There are multiple possible mechanisms by which land-use interventions could affect the factors that determine the net returns to forest clearing. Considering them yielded various theoretical predictions, which we validated empirically.

On average, we found insignificant net spillover effects within both 0–5 km and 5–10 km of parks, when controlling for land characteristics using matching and regression methods. However, averages blend heterogeneous spillover park impacts for different subsets of nearby forested lands, defined according to the distance to roads (critical for agricultural returns) and to park entrances (critical for tourism and thus forest returns). Spillovers close to park entrances are insignificant—in areas associated with higher tourism—but we found large increases in deforestation (around 9%) near roads in the areas less exposed to tourism. Further, we again found leakage when moving away from the entrances toward areas where the immediate tourism returns are lower and the returns to clearing for agriculture and for tourism infrastructure increase. When looking across all the parks, these heterogeneous spillover impacts results are quite robust.

We further extended the existing literature by separating parks into groups in multiple ways that may meaningfully characterize different settings that could raise or lower such significant spillovers. For instance, the leakage effect that we found far from the entrance near roads is higher if the parks in question are in lower-sloped (higher opportunity cost) forestland. Also, older and larger parks, associated with higher tourism, generate more leakage near entrances except in Ring 2.

As discussed in the empirical strategy, identification relies upon successfully controlling for all characteristics that are correlated with both park location and deforestation. However, one set of characteristics that is not available in our data concerns the organizational capacity and political capital of communities and their leaders. Communities with strong organizational capacity and political capital might affect the decision process for park location and orient that process toward either tourism or agricultural employment. To the extent that such factors are not correlated with deforestation, these omitted variables do not bias our estimates. However, they could in principle affect the rate of tourism and agricultural development and thus affect deforestation. If other covariates do not capture some of these effects, our estimates could be biased. The sign of the bias will depend on whether strong communities will attract parks and tourism or reject parks in favor of agriculture. For our period of analysis, agriculture was an important source of income for rural Costa Rica while tourism was not. Thus, parks may be near communities unable to advocate effectively for development (which yields deforestation). That would bias impact estimates for spillovers toward not finding increases in deforestation near parks.

Without additional information, unfortunately we cannot comment on how enforcement affects spillover effects. However, in the case of Costa Rica, we do not expect protection enforcement to vary substantially by type of protected area. Still, this is a dimension to be explored in future work on spillovers, given the importance of the variation in enforcement across types of protected areas found in other leading tropical forest countries (see Joppa and Pfaff 2010a; Nelson and Chomitz 2011; Pfaff et al. 2013; Pfaff, Robalino, Herrera, and Sandoval 2015; and Pfaff, Robalino, Sandoval, and Herrera 2015).

We acknowledge the fact that even though we are using state-of-the-art measures of deforestation, better metrics of forest loss are needed to detect forest degradation. Binary measures of deforestation indicate the presence of forests, but there could be some underlying forest loss. When metrics of forest degradation can be utilized, estimates of carbon leakage will be improved, which will be highly relevant in the context of reducing emissions from deforestation and forest degradation (REDD) policies. Similarly, better land-use data are needed to avoid having observations with an uncertain presence of forests due to clouds or shadows. Future research should test whether results would change significantly if parcels covered by clouds and in places where the presence of forests is certain are included in the analysis.

Additionally, we have considered only protected areas. There are many other interventions in land management that may well generate significant spillovers from private behavior. These changes in private behavior may affect other private behavior, shifting regional equilibria (Robalino and Pfaff 2012). For instance, payments for ecosystem service programs are growing rapidly. These payments could influence land use outside the program through various mechanisms, such as the new option value of possible future payments. Certification of logging concessions, for instance by the Forest Stewardship Council, is another type of intervention that is growing rapidly and may feature spillovers through mechanisms such as optimization across multiple concessions by logging firms.

Protected areas, which, as in Costa Rica or the Brazilian Amazon, can represent interventions on a very large scale, could have additional spillover effects on the optimal public behavior of other parts of the government. Herrera (2015), for instance, finds that the establishment of protected areas affects both private migration and the location of new roads. That possibility raises the potential for game-theoretic interactions between agencies pursuing very different frontier agendas.

We find evidence of heterogeneous spatial spillovers that depend on the location of roads and the presence of tourism. Therefore, it is critical to consider the impacts outside the boundaries of an intervention when designing future policy, particularly as the attention to the impact evaluation of conservation and development interventions increases. While clearly it was natural and understandable to start with impacts inside those boundaries, improving our understanding of spillovers can help us develop more accurate estimates of program impact and help protected-area managers be more vigilant of specific nearby areas where deforestation is likely to occur.

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