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PROTECTED AREAS, INDIGENOUS COMMUNITIES, DEFORESTATION AND THE ROLE OF INSTITUTIONS: EVIDENCE FOR THE LOWLANDS OF BOLIVIA

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Protected areas, indigenous communities, deforestation and the role of institutions: Evidence for the lowlands of Bolivia

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Abstract

Protected areas and indigenous communities play a crucial role in controlling deforestation, which is responsible for carbon emissions related to land use and land use change, contributing to global warming and climate change. However, the effectiveness of protected areas is conditioned by their administration and the quality of the institutions in their countries. In this paper, I will analyze the effectiveness of protected areas (Both at the national and sub-national levels) and indigenous territories and the institutions' role in the case of Bolivia's lowlands. I computed deforestation rates for four different periods between 1986 and 2021 to test the impact of institutions on different types of protected areas by using satellite images at 30m resolution and combining them with official data on protected areas and indigenous communities, which includes specific locations and dates of creation. Using a spatial regression discontinuity design, the results show that protected areas at the national level are the most effective in controlling deforestation, particularly after the creation of institutions taking care of them. Departmental protected areas have some impact on preventing deforestation, while municipal ones have no influence. For indigenous communities, there was a significant effect on reducing deforestation with the first reservations created, but the effect wears off over time. Regarding mechanism, being close to cities and routes is a threat for indigenous communities and departmental PAs, while it is an advantage for national PAs. Finally, protected areas and indigenous communities show, in general, greater levels of deforestation when they are exposed to cattle ranching settlements, mines, and oil wells.

KEYWORDS. Deforestation, Indigenous communities, Protected Areas, Institutions

1 Introduction

Deforestation is responsible for carbon emissions related to land use and land use change, which contributes to global warming and climate change (IPCC, 2019). In this context, the protection of forests becomes an essential element for public policy in the world, as the creation of protected areas is the more common conservation policy implemented by developing countries. Besides, according to UN (2014), indigenous peoples are responsible for a significant part of biodiversity and forest preservation.

Protected areas and indigenous communities have been recognized as an effective palliative for deforestation both at cross countries analysis (Blankespoor et al., 2017; Heino et al., 2015; Morales-Hidalgo et al., 2015), as well as for the Amazonia bioma (Nepstad et al., 2006; Walker et al., 2020) or for specific countries like Costa Rica (Andam et al., 2008), Indonesia (Shah and Baylis, 2015), Tanzania (Hall et al., 2022), Bangladesh (Rahman and Islam, 2021), among others. However, the effectiveness of protected areas is conditioned by their administration and the quality of the institutions in their countries. Abman (2018) mentions that countries with effective corruption control are more effective in controlling deforestation because corruption promotes the realization of illegal activities like logging inside reservations and protected areas, increasing deforestation rates. Therefore, the reduction of deforestation does not depend only on the creation of protected areas but also on the government effectiveness and strong political enforcement (Moreira-Dantas and Söder, 2022), macroeconomic stability (Arcand et al., 2008), and sustainable oriented institutions (Bray and Klepeis, 2005). In this paper, I will contribute to this literature by analyzing how different institutions and administration systems of protected areas and indigenous communities in the lowlands of Bolivia contribute to reducing deforestation.

Institutions matter on deforestation not only on the direct effect of protected areas but also by enhancing the threats and drivers of deforestation. An example of this is explained by Bonilla-Mejía and Higuera-Mendieta (2019) by analyzing the case of Colombia, where weak institutions put protected areas and indigenous communities near big cities and routes at risk because they reduce the cost of management and transportation for logging activities. Also, the proximity to other sources of economic activity like mining or cattle ranching can be a source of deforestation for protected areas and indigenous communities, as is mentioned by Müller et al. (2012) for the Bolivian case. However, the study of institutions and deforestation needs to be clarified, and mixed results have been found. While the previous studies have found a positive relationship between institutions and deforestation, others have found that this effect does not exist (Leblois et al., 2017), but also evidence of high rates of deforestation with solid institutions, like the case of Australia (Evans, 2016). The role of institutions and their interaction with protected areas is a question that remains open. In this context, the study of Bolivia plays a key role in this literature, given the variety of protected areas with different types of administration that have evolved over time, which allows for testing of which kind of administration and which institutions are more efficient in controlling deforestation.

Bolivia is an interesting case of analysis for several reasons. First, Bolivia is the 9th country with the world's highest average annual net loss of forest area for 2010-2020, and the second one in the Amazon biome, only surpassed by Brazil (FAO, 2020). Secondly, Bolivia has several types of protected areas (National, departmental, and municipal) with different kinds of administration that have evolved over the years. While in the decade of 1980, several protected areas existed in Bolivia, there was no clear administration and regulations among them; therefore, deforestation was challenging to control. This scenario changed in the decade of 1990 with the creation of the National System of Protected Areas (SNAP) in 1992 and the National Service of Protected Areas (SERNAP) in 1996, where it was specified that the central government would take care of national protected areas, while the departmental and municipal ones would be in charge of local governments. This variety of administration systems is interesting in research analysis because it opens the possibility for heterogeneous effects among them and allows to test separately which one is more effective in controlling deforestation and how different drivers of deforestation are affecting them, which has policy implications for similar contexts.

Finally, since the new constitution was approved in 2009, Bolivia has become a plurinational State, officially recognizing indigenous peoples and providing more autonomy and self-determination rights to the communities. Under this scenario, we should expect to observe more empowerment and control of illegal activities inside indigenous territories. However, indigenous communities face difficulties protecting their habitats because of the extractivist policies implemented by the government that have facilitated deforestation inside indigenous lands (Tockman and Cameron, 2014; López, 2017). A systematic analysis of deforestation in indigenous communities in Bolivia will allow us to assess if their conditions differ from those of the indigenous territories in other countries with fewer rights, less autonomy, and less recognition by the State.

This paper makes three significant contributions. First is the first study that separates different types of protected areas in Bolivia and compares them with indigenous territories. While some studies have pointed out difficulties these territories face because of illegal logging, this is the first study that analyzes which administration system has been more effective to be replicated in other areas. Second, this separation allows the analysis of various mechanisms for different kinds of protected areas or indigenous communities, which helps to develop specific policies for preserving each type of protected area. The mechanisms to be tested are proximity to cities and routes and exposition to other sources of economic activity such as cattle ranching, mines, and oil wells. Finally, it contributes to the literature on the role of institutions in deforestation by pointing out that clear property rights for indigenous communities and centralized institutions taking care of protected areas are effective ways of stopping deforestation.

To test the effectiveness of protected areas and indigenous communities in stopping deforestation, I will implement a spatial regression discontinuity design (RDD) following the non-parametric methods by Calonico et al. (2014) and using the border of indigenous territories and protected areas as the cutoff. The use of RDD to measure deforestation has been recommended by Wuepper and Finger (2023), and the use of borders as identification is being implemented for neighbor countries (Cuaresma and Heger, 2019), for sub-national boundaries in Brazil (Bogetvedt and Hauge, 2017), for indigenous territories in the Brazilian Amazonia (Baragwanath and Bayi, 2020), and for protected areas in Colombia (Bonilla-Mejía and Higuera-Mendieta, 2019).

The RDD relies on the assumption that two land parcels at the border are similar in most unobserved factors, allowing the identification of the local average treatment effect. This requires a continuous variable determining treatment assignment and a clearly defined cutoff. Since distance to the border is a continuous variable that can take negative values outside the protected area and positive values inside, it can be used as the running variable, with the border serving as the cutoff for RDD implementation.

There are two empirical ways of testing the RDD assumptions. The first is to artificially move the cutoff inside and outside the protected area to confirm that discontinuity is observed only in the border (Also known as placebo estimators). The second is to test if spatial cells are comparable at the border regarding observable covariates. To test this assumption, I will implement the Canay and Kamat (2018) permutation test, in which the null hypothesis is that the distribution of covariates is continuous at the cutoff.

I computed deforestation rates for four different periods between 1986 and 2021 to test the impact of institutions on different types of protected areas by using deforestation data provided by MapBiomas Amazon Project (2021) and official data on protected areas and indigenous communities which includes specific locations and date of creation. Finally, I test the effect of different mechanisms reported in the literature, such as distance to cities, access to routes, and exposition to other sources of economic activities like cattle ranching, mines, and oil wells (Killeen et al., 2008; Müller et al., 2014; Peralta-Rivero, 2020).

The results show that protected areas at the national level are the most effective in controlling deforestation, particularly after the creation of institutions taking care of them. Departmental protected areas have some impact on preventing deforestation, while municipal ones have no influence. For indigenous communities, there was a significant effect on reducing deforestation with the first reservations created, but the effect wears off over time. Regarding mechanism, being close to cities and routes is a threat for indigenous communities and departmental PAs, while it is an advantage for national PAs. Finally, protected areas and indigenous communities show, in general, greater levels of deforestation when they are exposed to cattle ranching settlements, mines, and oil wells. These results are consistent with what is being reported in the literature. Some studies have mentioned the difficulty of indigenous peoples in Bolivia to protect their territory because of the extractivist policies implemented by the government (Tockman and Cameron, 2014; López, 2017), while others have pointed out the role of protected areas in stopping deforestation (Paneque-Gálvez et al., 2013), or the limited capability of action that municipal protected areas have (Andersson and Gibson, 2007).

The rest of the document is structured as follows: Section 2 provides the institutional background of Bolivia; Section 3 contains the literature review; Section 4 presents the data sources and descriptive statistics; Section 5 explains the methodology; Section 6 presents the empirical results; and Section 7 concludes.

2 Background

Bolivia is the fifth largest country in Latin America, with a surface slightly above one million square kilometers. Even though Bolivia is primarily recognized as an Andean country and most of the population and big cities are located in the highlands, almost 60% of the surface is covered by tropical forests, which are part of the Amazon bioma that is known as the lowlands of Bolivia. This extensive area is very well known for its biodiversity and the rich ecosystems living there. In this context, protecting the environment is vital for the Bolivian State and the different communities in that area.

The lowlands of Bolivia have been historically considered an uninhabited territory, even though several indigenous communities have lived there for many years (Guiteras, 2011). A colonization process took place in the 20th century and sped up after the agrarian reform of 1953, creating tensions between the indigenous communities and the new settlers (Benavides, 2022). The main reason for the occupation of the lowlands was to expand the economic activities of the Bolivian State through agricultural expansion, which was characterized by colonization and incentives to develop medium and large-scale agriculture (Pacheco, 2006). As a second consequence of the agrarian reform, many protected areas will be created in the upcoming decades at three levels: national, departmental, and municipal. This process was characterized by an almost non-existent legal framework, making it difficult to regulate and protect them Ministerio de Medio Ambiente y Agua (2012). Finally, as the indigenous communities were not included in the reform, in 1982 is created the Confederation of Indigenous Peoples from Bolivia (CIDOB) and, in 1990 took place the first indigenous protest for territory and dignity (Lehm and Lara, 2019).

In this context, two significant changes were implemented in 1992: the creation of the National System of Protected Areas (SNAP) and the first collective titles awarded to indigenous communities. After this point, protected areas and indigenous communities will have different paths and changes in their administration, the first ones always under the control of the state and the second ones claiming autonomy and self-determination.

Müller et al. (2013) highlights indigenous territories' importance in controlling deforestation during this period. Still, it also mentions that indigenous territories are located far from the deforestation frontier, limiting their capability to stop deforestation.

Regarding protected areas, the SNAP grouped all of the existing ones, and the constitution recognized them as a critical element in preserving and managing the natural resources and biodiversity of Bolivia (Farah and Miranda, 2021). However, as Bolivia has several types of protected areas, more precise management of them was necessary. With this purpose, the National Service of Protected Areas (SERNAP) was created in 1998, providing a clear administration for each kind of protected area. Following this, the central government will take care of national protected areas, the departmental protected areas will be in charge of the respective governorship of each department in Bolivia, and the municipal protected areas will be in control of the mayor and council of each municipality (SERNAP, 2007).

The main issue with this context is the unequal distribution of resources and incentives at various subnational levels. While national-level resources are more abundant, managing protected areas at the departmental and municipal levels relies on the available resources, local authorities' willingness to oversee them, and their relationship with the central government. Some departments have created specific institutions to care for protected areas, and others have created networks to improve their management. At the same time, some municipalities say that they need more resources and ask for more help from the government Ministerio de Medio Ambiente y Agua (2012). On top of that, it's been reported that in the past years, it's observed a decrease in the preparation and training of the human capital working in the administration of protected areas and a change in the politics of the government towards a more extractivist philosophy, which increases the risk of deforestation in protected areas of Bolivia (Farah and Miranda, 2021).

Regarding indigenous communities, after obtaining land titles, the indigenous movement approved a law that recognized the indigenous territories and provided some autonomy over their administration (Benavides, 2022). At the beginning of the 2000s, the indigenous movement kept pressuring and protesting against the privatization of water and natural gas, and in 2005, Evo Morales won the republic's presidency as the first Indigenous president in the history of Bolivia. (Makaran, 2007). In 2009, Bolivia changed the constitution and became a 'Plurinational State' by recognizing all the indigenous peoples as pre-existing nations to the Bolivian State as well as adopting all indigenous languages as officials in the Bolivian State (Schavelzon, 2012).

However, this new scenario has yet to imply an improvement for the communities in the lowlands of Bolivia. Tockman and Cameron (2014) points out the contradictions between plurinationalism and an economic expansion based on extracting natural resources and the intention to politically control the indigenous territories. Besides, the plan of the administration of PAs pointed out that the regulation and protection of indigenous communities remained in the communities without any support from the government. This represents a threat to indigenous autonomy, which the constitution should grant, and represents a risk of deforestation for the communities in the lowlands.

From this background, we can identify three important events regarding the institutions taking care of PAs and indigenous communities: the implementation of SNAP in 1992, the creation of SERNAP in 1998, and the change of constitution in 2009. These three years will be used in the empirical analysis to study how implementing new institutions and regulations influenced deforestation control.

3 Literature review

The literature has widely analyzed protected areas and indigenous territories. They are mentioned as one of the most effective policies to control deforestation at cross country and for specific case studies. Morales-Hidalgo et al. (2015) estimate a fixed effects regression at the country-year level and concludes that an increase of 1% in protected areas inside a country is associated with a rise in 0.03% in the total forest area. A similar study is conducted by Blankespoor et al. (2017) by analyzing the effectiveness of protected areas in 64 countries with a two-way fixed effects model. They take ratios of deforestation 10 kilometers inside and outside protected areas and conclude that protected areas are more effective depending on their size, national park status, and management by indigenous peoples.

Regarding country-specific studies, Cuenca et al. (2016) uses matching procedures to study the effect of protected areas in Ecuador and concludes that they reduce deforestation by approximately 6%. Van der Hoek (2017) also analyzes the case of Ecuador using matching methods and shows that deforestation is experienced inside and outside protected areas. However, they are still effective by reducing between 2,600 and 7,800 hectares annually. Regarding indigenous communities, BenYishay et al. (2017) shows that the formalization of indigenous territories in Brazil has not decreased deforestation. De Los Rios (2022) analyzes the overlapping effect of protected areas and indigenous territories in Colombia using matching methods. The conclusion is that the overlap reduces deforestation only in indigenous territories with large sizes and populations.

All previous studies focus on the direct effect of protected areas but take no consideration of possible spillovers outside those territories. Gaveau et al. (2009) study the spillover effects of protected areas in Indonesia. They found that deforestation is lower inside protected and adjacent non-protected areas, accounting for spillover effects. Boillat et al. (2022) study the spatial spillover effects of Bolivia's national protected areas and indigenous territories. They conclude that spillover effects are only observed in national protected areas rather than indigenous communities. A different result is found by Rahman and Islam (2021) when studying protected areas in Bangladesh. They saw a decrease in deforestation inside protected areas but an increase in the adjacent unprotected areas.

However, it is only sometimes clear that protected areas help to reduce deforestation, and heterogeneous effects are reported in the literature. Shah and Baylis (2015) study the effectiveness of seven protected areas in Indonesia and find an average impact of 1.1% less deforestation inside PAs. However, individually analyzing each protected area, the results vary from a decrease of 3.4% to an increase of 5.3% in deforestation. Ferraro et al. (2013) found heterogeneous effects of protected areas in Bolivia, Costa Rica, Indonesia, and Thailand. According to their results, the effectiveness of protected areas is conditioned by the type of protection, with those with strict controls being more effective than the less strict ones.

We have observed so far some different results for direct and indirect effects of PAs in deforestation, which makes it relevant to analyze the mechanisms explaining heterogeneous effects. The first mechanism mentioned in the literature is the location of protected areas, as is shown by Joppa et al. (2008) who make a cross-country analysis in the Amazon and Congo basins and concludes that the positive effect of protected areas on deforestation is because they are located in hardly accessible spots, which makes complicated to differentiate which of the two effects is stopping deforestation. A similar result for indigenous territories in Bolivia is found by Müller et al. (2013), who mentions that large indigenous communities are located in areas with lower deforestation than the rest of the country. Therefore, their relevance in stopping deforestation could be only noticed when the deforestation frontier comes closer.

Demographic characteristics can also be considered an essential mechanism for effectively protecting areas from deforestation. Nepstad et al. (2006) compares inhabited and uninhabited reserves in Brazil. They observe less deforestation in uninhabited reserves than in populated national parks and indigenous territories, highlighting human settlements' relevance to deforestation. Killeen et al. (2007) study historical causes of deforestation in Bolivia and mention migration as one of the most important drivers, mainly in the decade of 1960. Mechanized agriculture and cattle ranching are also cited as more contemporaneous causes of deforestation.

Similar to the previous study, Müller et al. (2012) analyzes the causes of deforestation in Bolivia by using multinomial logit models. They found that the expansion of mechanized agriculture, cattle ranching, and small-scale agriculture are the main drivers of deforestation, being protected areas only effective against mechanized agriculture. Pérez and Smith (2019) also mention industrial-scale agriculture as a direct cause of deforestation in two indigenous communities in Bolivia. The authors also said the construction of roads is a cause of deforestation. Roads are also important because they reduce the transportation cost associated with deforestation. Barber et al. (2014) study the relationship between deforestation and access to highways, navigable rivers, and secondary roads in protected areas in Brazil. The results show an increase in deforestation was higher near routes and rivers. However, the effects are reversed for protected areas, which experienced less deforestation near roads and rivers. This can be understood because transportation networks reduce transportation costs and provide access to resources that help protect national parks against deforestation. Another economic cause of deforestation is illegal mining, which was mentioned by Silva-Junior et al. (2023) as the main factor on indigenous territories in Brazil in the period 2013-2021.

As we can see, different backgrounds and country-specific studies report additional drivers for deforestation. In this context, the role of institutions becomes a critical question because a fragile institutional background can directly and indirectly, affect deforestation. The literature about institutions and deforestation has focused on the quality of government and corruption indexes and management of protected areas and indigenous territories. Regarding the first type, Moreira-Dantas and Söder (2022) makes a cross-country analysis of the relationship between corruption and weak institutions with deforestation for 1992-2015. By running logit models, the authors find that higher government effectiveness and lower corruption perception reduce deforestation's probability. However, Leblois et al. (2017) estimate a two-way fixed effects model to analyze the cross-country drivers of deforestation. They found that agricultural trade is one of the main factors of deforestation, but they found no evidence that institutional quality might influence deforestation. As we saw before, cross-country analysis tends to give contradictory results, which makes it relevant to analyze country-specific studies.

Bogetvedt and Hauge (2017) study the impact of institutional quality in deforestation for 138 Brazilian municipalities in 2002-2004. The authors implement a spatial regression discontinuity design using corruption measures at the municipal level and found no effect of institutional quality on deforestation. On a different setup, Laurance et al. (2011) study the exposition of Papua New Guinea to deforestation because of the impact of weak institutions. The results suggest that because of corruption and weak enforcement capacity, logging operations often violate mandated standards, increasing deforestation in the country. As we can see, the effect of institutions on deforestation is convoluted, and it depends on the characteristics of every context.

Regarding the relationship between institutions and protected areas, in a cross-country analysis, Abman (2018) study the relationship between the rule of law and avoided deforestation from protected areas. The author finds that protected areas were more effective in controlling deforestation in countries with higher levels of corruption control and clear property rights. Similarly, Leverington et al. (2010) analyzes the performance of over 8000 protected areas worldwide and concludes that they are more effective when there are clear property rights and strong governance.

For the case of Bolivia, Andersson and Gibson (2007) analyzes the role of municipal protected areas on deforestation. They explore the case of 30 municipalities in Bolivia's lowlands and conclude that local institutions are effective against unauthorized deforestation. Still, they have no impact on permitted or total deforestation, limiting their capability. One of the most similar studies is the one performed by Bonilla-Mejía and Higuera-Mendieta (2019) that analyzes the effect of protected areas on deforestation under weak institutions. They estimate a spatial regression discontinuity and find that national PAs are only effective near human settlements and in municipalities that provide more public goods and have fewer incidents of violence. On the other hand, collective lands (like indigenous territories) are more effective in remote areas and less developed regions. The relevance of this study is to show that institutional background is essential because it enhances drivers of deforestation, like access to routes and markets for logging trade.

The difference observed in the previous study can be related to property rights as explained by Baragwanath and Bayi (2020) that analyses the case of indigenous communities in the Brazilian Amazon. They found that those communities with full property rights can significantly decrease deforestation, which is not observed in those territories without full property rights. A similar conclusion is found by Araujo et al. (2009) that estimates a two-way fixed effect model for 1988-2000 in nine states of the Brazilian Amazon and concludes that deforestation is more common in regions with insecure property rights. As we saw in the background, many indigenous communities in Bolivia cannot enhance their autonomy, and therefore, they face a situation equivalent to needing full property rights.

There are two important conclusions from this literature review. Regarding mechanism, we can conclude that the absence of solid institutions made protected areas and indigenous communities particularly vulnerable to migration, proximity to cities and routes, and other sources of economic activity such as cattle ranching, mechanized agriculture, and, to a lesser extent, oil, and mining. The second conclusion is that most of the literature focused on indigenous communities or protected areas, with very few studies making distinctions among the different types of protected areas, highlighting this study's relevance.

4 Data

I will use several types of data in this paper. First, I will need data on deforestation rates; second, I will need the shape files of indigenous communities and protected areas. Finally, I will incorporate data on land controls and mechanisms for deforestation.

4.1 Deforestation

Deforestation rates were constructed using the MapBiomas Amazon Project (2021), which provides detailed information on land use and land cover for the lowlands in Bolivia at a 30-meter resolution. The satellite images cover the period 1985 to 2021 and provide 24 classifications of land use. The unit of analysis corresponds to a spatial cell of 100-pixel x 100-pixel[1] with a sample size of 79,168 observation. Each cell's deforestation rates were calculated annually following the methods explained by Silva Junior et al. (2020).

The dependent variable will be the average deforestation rate of a specific cell in the period 1986-2021. I also computed deforestation rates for four periods: 1986-1992, 1993-1998, 1999-2008, and 2009-2021. The first period corresponds to the years before the creation of the National System of Protected Areas (SNAP), a period without institutions taking care of protected areas. The second period tests SNAP's effectiveness and includes the first indigenous territories. The third period incorporates the creation of the National Service of Protected Areas (SERNAP). Finally, in 2009, the new constitution of Bolivia was approved, incorporating more rights and autonomy for indigenous communities.

Another well-known data source utilized in deforestation studies is the annual tree-cover loss from Hansen et al. (2013), which is also at 30 meters resolution but for the period 2000-2021. As there is no data before 2000, it is impossible to test the different periods, so I chose MapBiomas as the primary source of information. However, in the first stage of the research, I used Hansen data, and the results were consistent with what was found using MapBiomass.

4.2 Protected areas

Geo-localization of indigenous communities and protected areas are collected from Geo-Bolivia[2], an initiative depending on the vice presidency of Bolivia. The platform provides access to the specific location of each protected area and indigenous community and a set of variables relative to each one, like area, estimated population, and the year of creation.

I created four dummy variables (one for each of the four territories considered: indigenous communities and national, departmental, and municipal protected areas), taking value 1 if a spatial cell is inside an indigenous or protected area and 0 otherwise. Regarding deforestation, it was computed only for the years in which the cell became a protected area. Finally, to implement the regression discontinuity design, I calculated the distance to the border of the closet protected area and indigenous communities for each cell.

Figure 1 shows deforestation dynamics by kind of land; this means indigenous communities, the three protected areas, and private lands. We observe that private lands have the highest rates of deforestation in most years, being overcome at some points by municipal PAs, which had the lowest deforestation rates at the beginning and the second highest at the end. On the other hand, deforestation rates significantly decreased for national PAs after 1994 and remained at the tiniest levels for the rest of the period. Regarding indigenous communities, there was a clear decrease in the 1990s and an increase in the years after. Finally,

[1]The use of aggregated pixels is widely used in this literature following the recommendations made by Garcia and Heilmayr (2022)

[2]http://geo.gob.bo/portal/

departmental PAs have the most volatile pattern, with a reduction in deforestation until 2003 and an upward trend in the years after that.

Table 1 shows descriptive statistics for the overall period. For the overall sample displayed in row 1, deforestation averages 0.57 percentage points. Regarding different types of land tenure, private lands and municipal PAs have the largest deforestation rates, while national PAs have the smallest, consistent with what was observed before. The last two columns show that private lands have significantly greater deforestation rates than other types of areas. Deforestation is considerably smaller for indigenous communities and national and departmental PAs than in private grounds. Finally, for municipal PAs, the difference in deforestation is not statistically different from the one observed in private lands.

4.3 Controls and Mechanism

Finally, I incorporated information about cities, routes, cattle ranching settlements, mines, and oil well locations to be tested as possible deforestation mechanisms and several geographic characteristics to be used as controls in the regressions.

Location of routes, oil wells, and mines are obtained from the Amazon Network of Georeferenced Socio-Environmental Information (RAISG)[3], which provides detail spatial information for all countries in the Amazonia. The location of cities, towns, and villages comes from the Open Street Map project[4]. Finally, information about cattle ranching settlements comes from the global distribution of livestock provided by Robinson et al. (2014). I computed the distance to the closest city, route, cattle ranching settlements, mines, and oil wells for each cell in the database. I obtained data about elevation above sea level from the Global Solar Atlas published by the World Bank[5]. I used the raster files they provided to compute six indicators of land characteristics such as altitude, slope, aspect, roughness, the Terrain Ruggedness Index (TRI), and the Topographic Position Index (TPI). Finally, I will also incorporate as control the forest cover in 1985, which means before the period of study.

5 Estimation

The main goal of this paper is to test the effect of protected areas on deforestation. There are two econometric challenges to deal with in this context. The first is to look for a proper counterfactual to protected areas because we do not observe the outcome (deforestation) in a treated cell (PA) if that cell wouldn't be treated. The second econometric concern is that PAs are not randomly assigned. Therefore, a

[3]https://www.raisg.org/
[4]https://www.openstreetmap.org/
[5]https://datacatalog.worldbank.org/search/dataset/0037910

simple comparison of cells under the protection of a national park or an indigenous community with other cells without protection is needed. Joppa and Pfaff (2009) shows that protected areas are, on average, located at higher altitudes, steeper slopes, and far from cities and routes. A similar result is provided by Vieira et al. (2019), showing that protected areas in Brazil are located in areas of low intensity of use. Therefore, less deforestation is expected in those areas.

A widely used method to asses this problem is the matching procedure suggested by Joppa and Pfaff (2010), which compares protected and unprotected cells that are similar in their observed characteristics. However, as Dos Santos Ribas et al. (2020) points out, matching methods do not account for unobserved heterogeneities among different spatial cells. Therefore, it does not solve the main bias mentioned above. On the other hand, the two-way fixed effects regressions allow to control by unobserved heterogeneities at the spatial cell and time level (Jones and Lewis, 2015), but they rely on the creation of new protected areas as an identification strategy. At the same time, those existing before the period of analysis are left behind on the identification procedure. Therefore, to control for unobserved land characteristics and to account for the long-term effect of protected areas, I will implement an RDD design to compare cells in the border of PAs (Imbens and Lemieux, 2008; Lee and Lemieux, 2010).

5.1 Parametric setup

Let's define X_i as the distance to the border of a protected area or indigenous territory, taking negative values outside them and positive values inside them. This implies that the cutoff will be at $X = 0$. The treatment will be binary in this case, taking value 1 if a spatial cell is inside a protected area or indigenous community. Therefore, the treatment will be defined as:

$$D_i = \begin{cases} 0, & \text{if } X_i < 0 \\ 1, & \text{if } X_i \geq 0 \end{cases}$$

The main idea is to estimate the Local Average Treatment Effect of D_i on deforestation rate y_i as follows:

$$\begin{aligned} \tau &= \mathbb{E}\left[y_{i(D_i=1)} - y_{i(D_i=0)} | X = \bar{x}\right] \\ \tau &= \mathbb{E}\left[y_{i(D_i=1)} | X = \bar{x}\right] - \mathbb{E}\left[y_{i(D_i=0)} | X = \bar{x}\right] \end{aligned} \quad (1)$$

A first approach to estimate equation (1) is to impose a linear functional form for y_i using D_i and X_i as regressors and including an interaction term as well, which allows changes in the slope of X_i inside and outside the border as follows:

$$y_i = \beta_0 + \beta_1 D_i + \beta_2 X_i + \beta_3 D_i X_i + \varepsilon_i \tag{2}$$

As the literature suggests, adding polynomials of higher order with their respective interactions is possible. However, Gelman and Imbens (2019) suggests using polynomials up to the second degree because the estimations are very sensible to higher polynomial orders, which can distort the estimations. By adding second order polynomial as well as a set of observable covariates Z_i, which includes altitude, slope, aspect, roughness, the Terrain Ruggedness Index (TRI), the Topographic Position Index (TPI), the average annual precipitation, and a set of protected areas dummies, equation (2) becomes:

$$y_i = \beta_0 + \beta_1 D_i + \beta_2 X_i + \beta_3 D_i X_i + \beta_4 X_i^2 + \beta_5 D_i X_i^2 + Z_i'\gamma + \varepsilon_i \tag{3}$$

As RDD relies on identification in the border, it is possible to compute the expected values required to identify equation (1) as follows:

$$\lim_{x \to 0^+} \mathbb{E}\left[y_i | X_i = x, Z_i = z\right] = \mathbb{E}\left[y_{i(D_i=1)} | X_i = 0, Z_i = z\right] = \beta_0 + \beta_1 \tag{4}$$

$$\lim_{x \to 0^-} \mathbb{E}\left[y_i | X_i = x, Z_i = z\right] = \mathbb{E}\left[y_{i(D_i=0)} | X_i = 0, Z_i = z\right] = \beta_0 \tag{5}$$

Note that, in the border, the observable covariates Z are assumed to be equal. By replacing equations (4) and (5) in (1) we get:

$$\tau = \beta_0 + \beta_1 - \beta_0 = \beta_1 \tag{6}$$

Therefore, the local average treatment effect is obtained by estimating β_1 on equation (3).

5.2 Non-parametric estimation

The parametric specification explained before has two problems. The first one is that parametric models impose a functional form for the relationship between the variables, while non-parametric methods rely on the estimation of observation on the border of discontinuity and do not impose a strict functional form.

Besides, as Calonico et al. (2014) points out, confidence intervals calculated with parametric functional form are sensitive to the bandwidth selection for X_i. The authors proposed a robust confidence interval estimators based on mean squared error optimal bandwidth estimation.

According to this procedure, I first estimated the optimal bandwidth and compared the conditional mean of those treated (inside the PA) with those untreated (outside the PA). As this study has four types of PAs, I will use non-protected areas as counterfactual in all cases. This means that the regressions for indigenous territories will not consider the cells identified as national or departmental protected areas. All the regressions will include controls for forest cover in 1985, altitude, slope, aspect, and roughness, the Terrain Ruggedness Index (TRI), the Topographic Position Index (TPI), and dummies for PAs unobserved characteristics.

RDD procedure assumes that the cutoff value is exogenous and cells in the boundary of PAs are identical on observable covariates except for the treatment. There are two empirical ways of testing this hypothesis. The first is to artificially move the cutoff inside and outside the protected area to confirm that discontinuity is observed only in the border[6]. The second is to test if spatial cells are comparable at the border regarding observable covariates. To test this assumption, I will implement the Canay and Kamat (2018) permutation test, in which the null hypothesis is that the distribution of covariates is continuous at the cutoff. Table 2 presents descriptive statistics and the p-value of the permutation test for each protected area. Join tests always fail to reject the null hypothesis, as well as most individual tests; this means that the only difference between cells inside and outside protected areas is the border.

Finally, regarding the test of heterogeneous effects, I will implement separate regressions for those cells near and far from cities, routes, and economic activities. This procedure follows Calonico et al. (2019), which probes that heterogeneous effects can be identified by interacted models only if the covariates do not vary across treatment and control cells, which is not the case in this context because the distance to cities or economic activities differs inside and outside protected areas.

6 Results

In this section, I will present the estimation results in three parts: First, I will describe the baseline estimations, followed by placebo analysis, and finally, I will test the discussed mechanisms for deforestation.

6.1 Baseline results

Regarding baseline results, figure 2 shows the discontinuity in the border for indigenous communities and the three types of protected areas. There is no visible effect for indigenous communities and municipal

[6]This procedure is known as placebo estimators.

PAs, while discontinuity is very strong in national and departmental PAs. This is consistent with the idea that national PAs have better institutions and, therefore, can access to more resources as they depend on the national government. When we move to sub-national units, the resources are more limited. Then, the control over deforestation becomes more complicated, so we see how the effect decreases for departmental areas and disappears for municipal ones. As explained in the literature review, indigenous territories are facing difficulties in enforcing their autonomy and control over their territories.

The specific results are shown in table 3. Columns 1 and 2 report the effects of polynomials of order 1 and 2, respectively, while in column 3, I report second-order polynomial specification with the inclusion of the controls mentioned above. As expected, the effect of national and departmental PAs is negative and significant in the three specifications. Looking at the third column, we see that the local causal effect of national protected areas implies a decrease in deforestation of 1.1 percentage points, which is quite sizeable if we look at the mean of the dependent variable reported at the bottom of table 3 which is 0.59 percentage points. For departmental PAs, the effect reported in column three is a decrease of 0.7 percentage points, which is almost the same as the mean of the dependent variable.

Regarding indigenous communities, the effect is negative but not significant in order 1 polynomial specification, but it becomes significant at 5% level when we move to second order polynomial. Nevertheless, the effect is much smaller, with a coefficient of 0.2 percentage points and a mean of 0.63 percentage points. Finally, municipal PAs is the only one showing positive and significant results, with an increase in deforestation of 0.3 percentage points reported in column three, with a mean of 0.64 percentage points. Again, we see that among protected areas, the nationals are the most effective in controlling deforestation, followed by departmental and municipal. This reinforces the importance of strong institutions in protected areas to prevent deforestation effectively.

However, as explained in Section 2, institutions taking care of protected areas only existed in 1992, although national parks have existed in Bolivia since 1939. The other two relevant moments to analyze are the creation of SERNAP in 1998 and the new constitution approved in 2009. In columns 4 and 5 of table 3, I estimate the before/after effect of the creation of SNAP by using 1992 as the benchmark. In columns 6 to 8, I repeat the exercise for the three remaining periods. It is essential to highlight that indigenous territories were not recognized before 1992, even though there is a longstanding demand for many communities in the lowlands of Bolivia. Finally, all regressions follow the specification of column 3, which means all controls and estimations are made with a polynomial of order 2.

For national PAs, the results reinforce the idea of solid institutions taking care of protection against deforestation. A positive and significant effect of deforestation inside national PAs before the creation of the SNAP is observed, and a negative and significant impact after 1992. When desegregating by periods, the strongest one is between 1992 and 1997, with an effect of 1.5 percentage points, and it reduces the magnitude after that. This result reinforces the need to create SNAP and SERNAP for administrating and providing resources to national PAs.

Departmental PAs show a similar pattern as national PAs, with a non-significant effect on deforestation until 1998 and a negative and significant impact after 1999, when SERNAP was created, and with the most substantial effect observed in 1998-2008, with a decrease of deforestation of 1.7 percentage points. Municipal PAs show mixed results. As explained before, it is the type of protected area with more volatile administration because it depends on the local authorities running the municipality in a specific year. We observe negative effects, then non-significant, then negative again, and positive at the end.

Finally, the case of indigenous territories is also interesting because we observe how the first titles provided to communities in 1992 decreased their impact on deforestation in time. In the first period, the effect was negative and significant. It started reducing its magnitude until it became insignificant after the approval of the new constitution in Bolivia. This is consistent with the demands and complaints of indigenous communities against the extractivist role of the Bolivian State after 2009, which made indigenous communities more vulnerable to external pressures against their forests.

6.2 Placebo Estimations

Placebo estimations play a crucial role in regression discontinuity designs. The idea is to test if the discontinuity is observed only in the cutoff; therefore, we should not observe any effect by artificially moving the order inside and outside the PAs and indigenous territory. To this end, I moved the cutoff 50 kilometers inside and outside the border in ranges of 10 kilometers for the four cases.

The results of this exercise are shown in figure 3, where the X-axis incorporates the distance concerning the border where the new cutoff was set. Then, a value of -20 implies moving the cutoff 20 kilometers outside the border, and a value of 20 means 20 kilometers inside the border, while the baseline is set at a cutoff of 0. Finally, the horizontal red line helps to understand whether the estimations are statistically significant or not.

Ideally, we should observe non-significant results inside and outside the border. This is the case in the four graphs of figure 3. We observe some considerable magnitudes in some cases of national and municipal PAs. However, they are still non-significant, so the placebo exercise confirms the relevance of using the border as the cutoff. The specific results for the regressions are shown in table 4.

6.3 Mechanisms

As discussed before, the main sources of deforestation that PAs and indigenous communities face are given by migration from highlands, the expansion of mechanized agriculture, the cattle ranching settlements and, to a lesser extent, mines and oil. For the last three mechanism I have precise information about their location, as it was explained in section 4. Distance to populated areas can be used as proxy of the effect and migration, while distance to the main routes of Bolivia is a measure of transportation costs and shows how vulnerable these areas are to external factors.

To do so, I computed several exercises limiting the sample to those spatial cell close or far of the five mechanism explained before. I took different distances as a robustness check to see if there were consistency among the results. In the case of cities, cattle ranching and mines I took intervals of 5 kilometers until a maximum distance of 30 km while for routes and oil wells I took intervals of 10 kilometers until 60 km. The decision was made to ensure, at least, 1% of observations in all specifications. The figures reporting the different effects includes the baseline estimations and the six exercises limiting the sample to a given distance and running the regressions for distances smaller or larger than that benchmark. Again, an horizontal red line shows if the results are significant or not.

Cities

The results for distance to cities are shown in figure 4 and the details can be checked in table 5. Regarding indigenous communities, we can see that being five or less kilometers away from cities has a positive but not significant effect, while been more than 5 kilometers away has a negative and significant impact on deforestation. This effect is robust to changes in the distance, and we can see that with a benchmark of 30 kilometers, the effect of being close is still non significant, but the effect of being more than 30 kilometers away is, in magnitude, bigger than the baseline. We can conclude that for indigenous peoples, being isolated from urban areas helps to protect their territories from deforestation.

Departmental PAs show exactly the same pattern, as indigenous communities, with the effect of reduction on deforestation becoming stronger while we move further away from cities. For municipal PAs, the only remarkable result is with a benchmark of 5 kilometers, where being close to cities increases the effect from 0.3 to 3.6 percentage points. After that, both effects are almost identical to the baseline, so we do not observe heterogeneous effects on the distance to cities.

Finally, for national PAs the situation is different because being close to cities has a negative and significant effects on stopping deforestation (which is in magnitude not different from the baseline result), while being far from cities has a positive but not significant effect. This results is not strange if we follow the idea that national PAs have the strongest institutions, because if those are the areas with more resources then being close to cities help them to access to those resources, while for departmental PAs and indigenous communities without solid institutions, being isolated is what help them to prevent deforestation. These results confirm the relevance of solid institutions for PAs and indigenous territories to be effective in the control of deforestation.

Routes

Figure 5 and table 6 show the results for distance to routes. In the case of indigenous territories the results are not significantly different from the baseline estimations, and no clear pattern is observed. Therefore, distance to cities is not a mechanism of deforestation in indigenous territories. On the other hand, for departmental PAs being far for routes helps to reduce deforestation, while being close have a positive and, in some cases, significant effect on deforestation[7]. Again, isolation is a relevant aspect for departmental PAs.

For national PAs we observe the same effect as for cities, this is being close to routes has a negative, significant and stronger effect on deforestation, although this effect is only observable with the 10 kilometers benchmark, and converge to the baseline result after that. Finally, for municipal PAs, we observe a convex effect on proximity to routes, being the effect positive and significant when the spatial cell is located less than 10, 50 and 60 kilometers from routes.

The effect of routes is not as consistent as cities, but again we observe that proximity to routes is an advantage for national PA (with stronger institutions) and a threat to departmental and municipal PAs (with less solid institutions).

Cattle Ranching

The expansion of cattle ranching settlements is being mentioned in the literature as one of the main causes of deforestation both inside and outside protected areas. The results on the effect of proximity to cattle ranching are shown in figure 6 and table 7. The results for indigenous communities show that being far from cattle ranching helps to reduce deforestation, and being close to them does not reduce it. Departmental PAs present the same pattern as indigenous communities, this is being isolated from other forms of economic activity helps them to protect their forests.

On the other hand, national PAs show a different pattern because we observe than being close to cattle ranching seems to have a bigger effect on deforestation respect to being far from them. In terms of magnitudes, the baseline estimation shows a decrease of 1.1 percentage points while being less than 30 kilometers away from cattle ranching reduce deforestation in 1.5 percentage points and being further than 30 kilometers

[7] The effect can be better understood on the table because of the distortion on the graph.

reduces in 0.7 percentage points. Finally, for municipal PAs, being far from cattle ranching does not change the magnitude respect to the baseline, but it is interesting to note that being close to cattle ranching reduce deforestation in 1 percentage point.

Although the results are inconsistent for all protected areas, we can identify cattle ranching as a threat to indigenous communities and departmental PAs.

Mines

Mining is another source of economic activity that could imply an increase in deforestation. We can see the results in figure 7 and table 8, where the first finding is that proximity to mines is a threat to indigenous communities, while being isolated from them is good in terms of deforestation. When a spatial cell is located less than 30 kilometers from a mine, we observe an increase of 0.4 percentage points, while being located more than 30 kilometers away from the mine reduces deforestation by 0.6 percentage points. However, this effect is not observed when the benchmark is small; it starts to be clear after 15 kilometers.

National PAs are also affected by proximity to mines, with a positive effect of being close and a negative effect of being far, both observable in benchmarks smaller than 15 kilometers. In terms of magnitude, being less than 10 kilometers away from mines generates an increase in deforestation of 3.5 percentage points, while being located more than 10 kilometers reduce deforestation by 1.5 percentage points. After 15 kilometers, both effects converge to the baseline result, but it is clear that mines are also a threat to national PAs.

There is no clear effect on proximity to mines regarding departmental and municipal PAs. Being less than 5 kilometers away from departmental PAs reduces deforestation significantly. However, this effect is not robust to changes in the benchmark, so the result is inconclusive. For municipal PAs, the magnitudes are always around the value of the baseline estimation. Therefore, there is no effect of mines on municipal PAs.

Oil

Finally, figure 8 and table 9 show the results for proximity to oil wells[8] shows this benchmark's results. The results for indigenous communities are counter-intuitive because being closer to oil wells decreases deforestation while being far from them increases it. A similar effect is observed for municipal PAs, but only for the 20 kilometers benchmark, and after that, results converge to the baseline estimation.

Regarding national PAs, we observe after 40 kilometers a negative and significant effect of being far from oil wells, which shows how isolation from economic activities is an advantage against deforestation.

[8] In the graph, I eliminated the effects for 10 kilometers because of the distortion made by the huge confidence interval. Table 9

Finally, results for departmental PAs are inconclusive; in most cases, the coefficients are not different from the baseline result.

7 Conclusions

Protected areas and indigenous territories are widely recognized as critical elements to control and stop deforestation. However, Their role is limited and threatened by several external factors like the expansion of mechanized agriculture and cattle ranching, the proximity to road networks, or mining activities inside their territories. On top of that, weak institutions enhance the threats of deforestation and do not allow protected areas to fulfill their purpose.

This work shows that different management and governance of protected areas and indigenous territories in Bolivia generate different effects on deforestation. By using satellite data to measure deforestation and a spatial regression discontinuity design as an identification strategy, this study shows that national protected areas have more solid institutions administrating them and, therefore, they are more effective in controlling deforestation than departmental and municipal protected areas, in which resources and institutions are weaker. This situation is similar to indigenous communities. These differences are also observed in the mechanisms tested in this study. While being close to cities and routes threatens indigenous communities and departmental PAs, this is an advantage for national PAs. Besides, protected areas and indigenous communities are less effective in stopping deforestation when they are close to cattle ranching settlements, mines, and oil wells.

These results suggest that creating protected areas is insufficient to stop deforestation if no proper institutions take care of them. This situation is, unfortunately, quite common, especially in developing countries with the largest concentration of forest in the world, and needs to be addressed to prevent, control, and stop deforestation.

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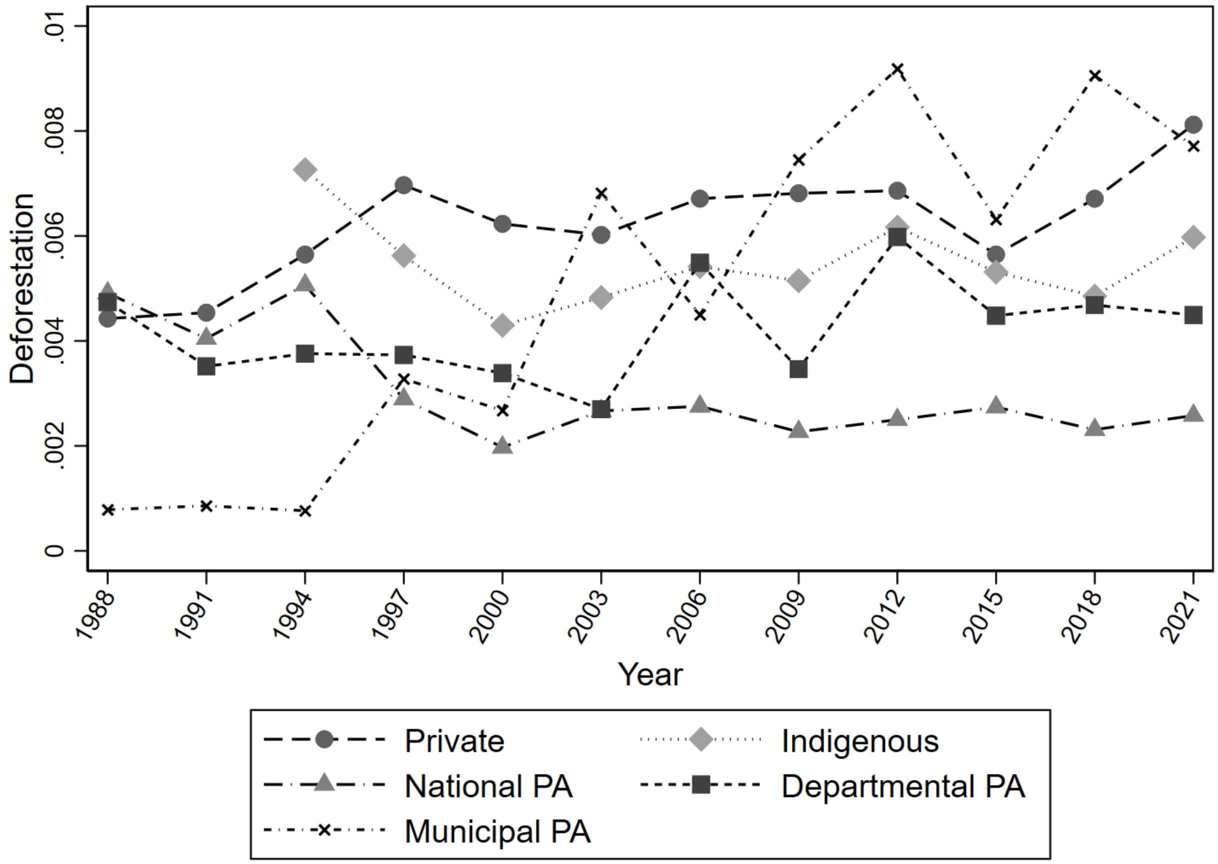

Figure 1: Evolution of deforestation by land tenure

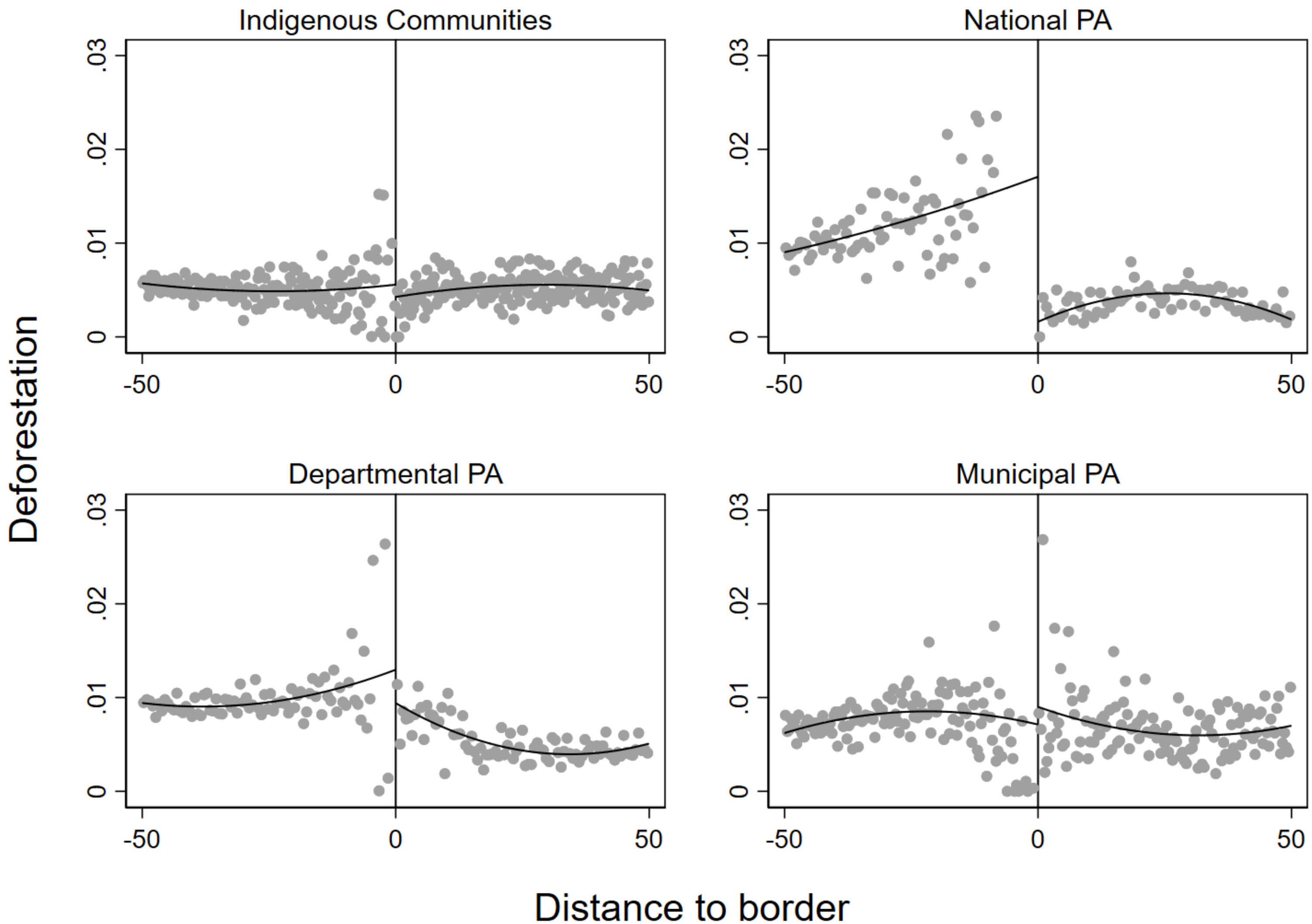

Figure 2: RDD estimations

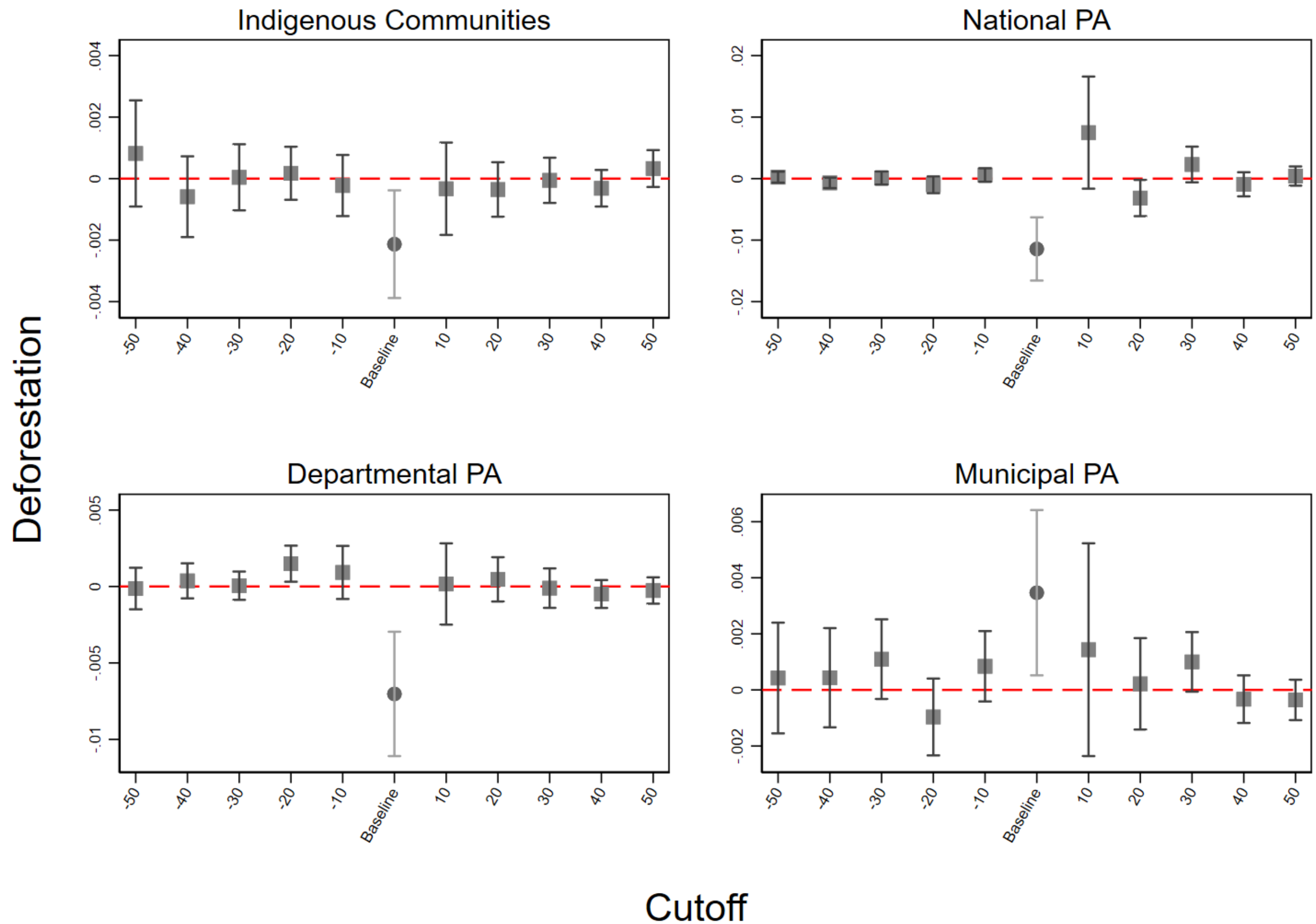

Figure 3: Placebo estimations

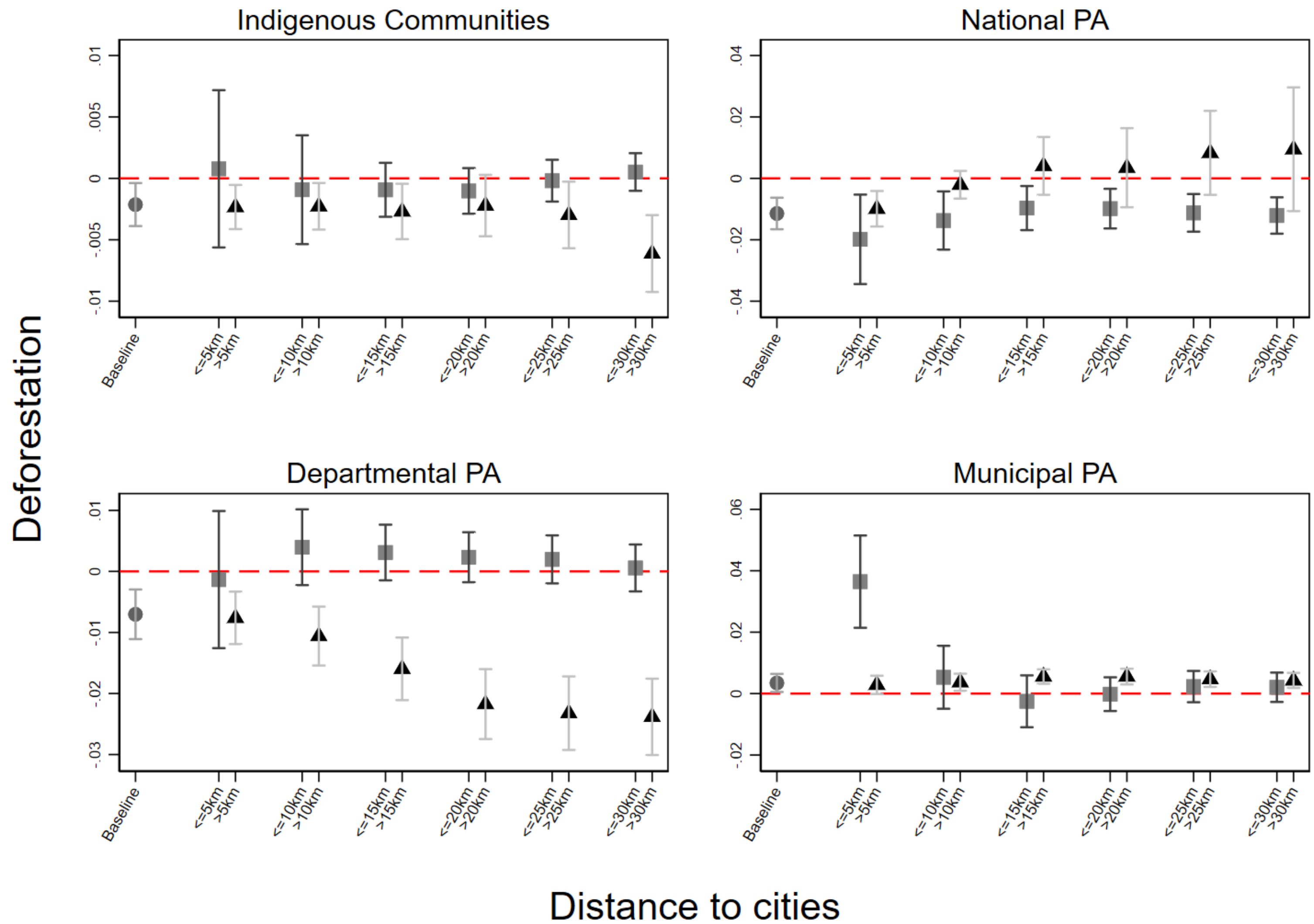

Figure 4: Distance to cities

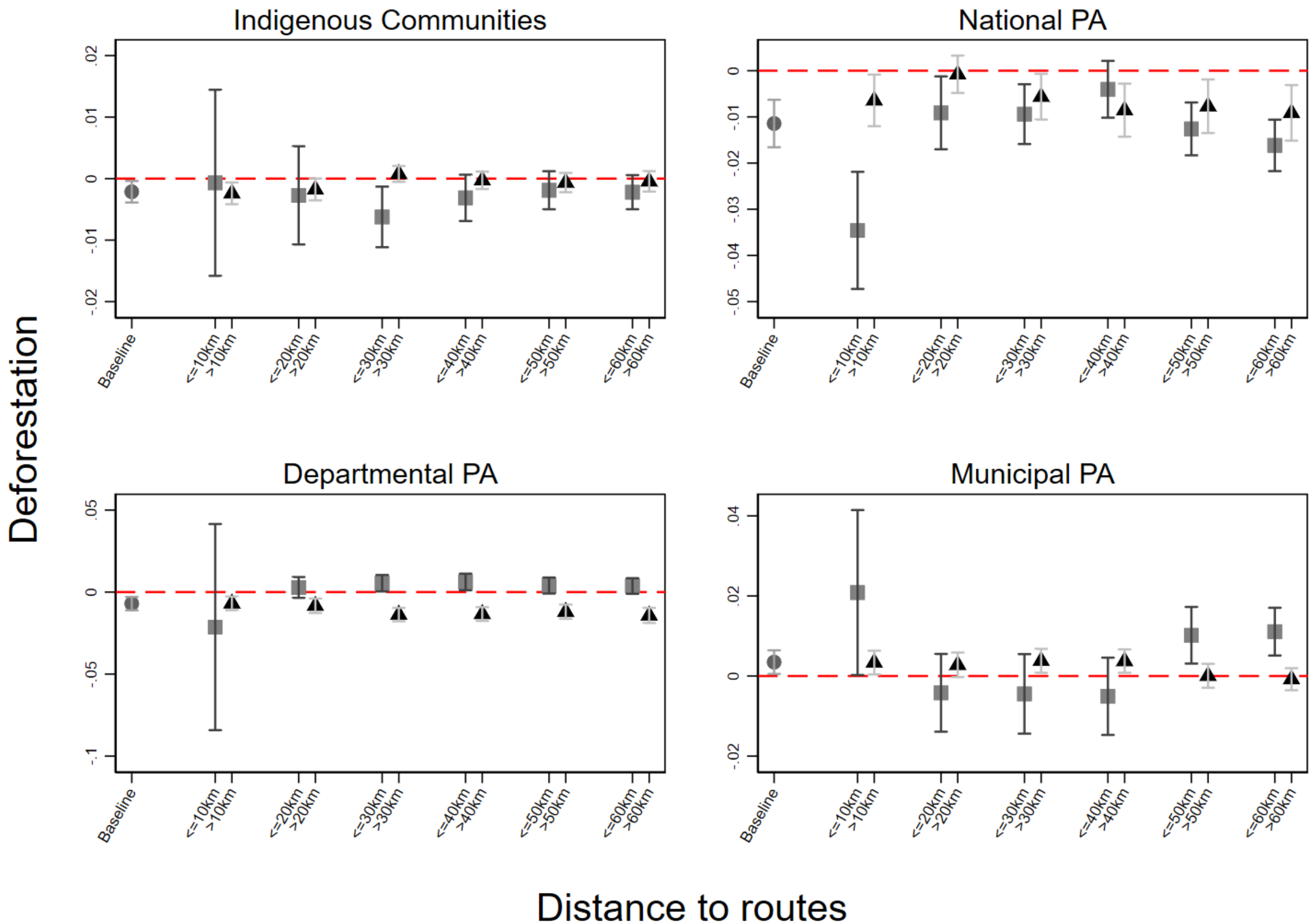

Figure 5: Distance to routes

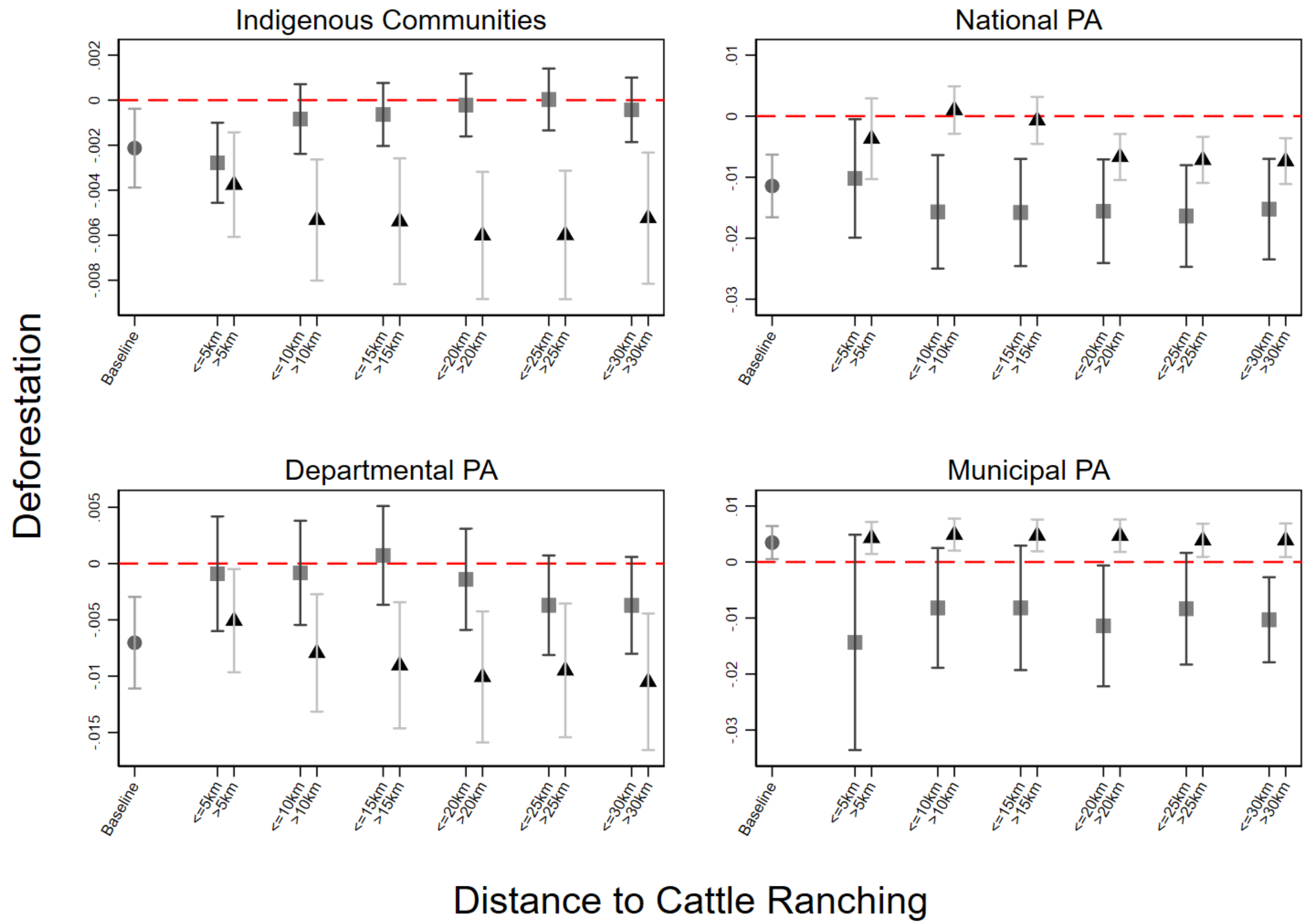

Figure 6: Distance to cattle ranching

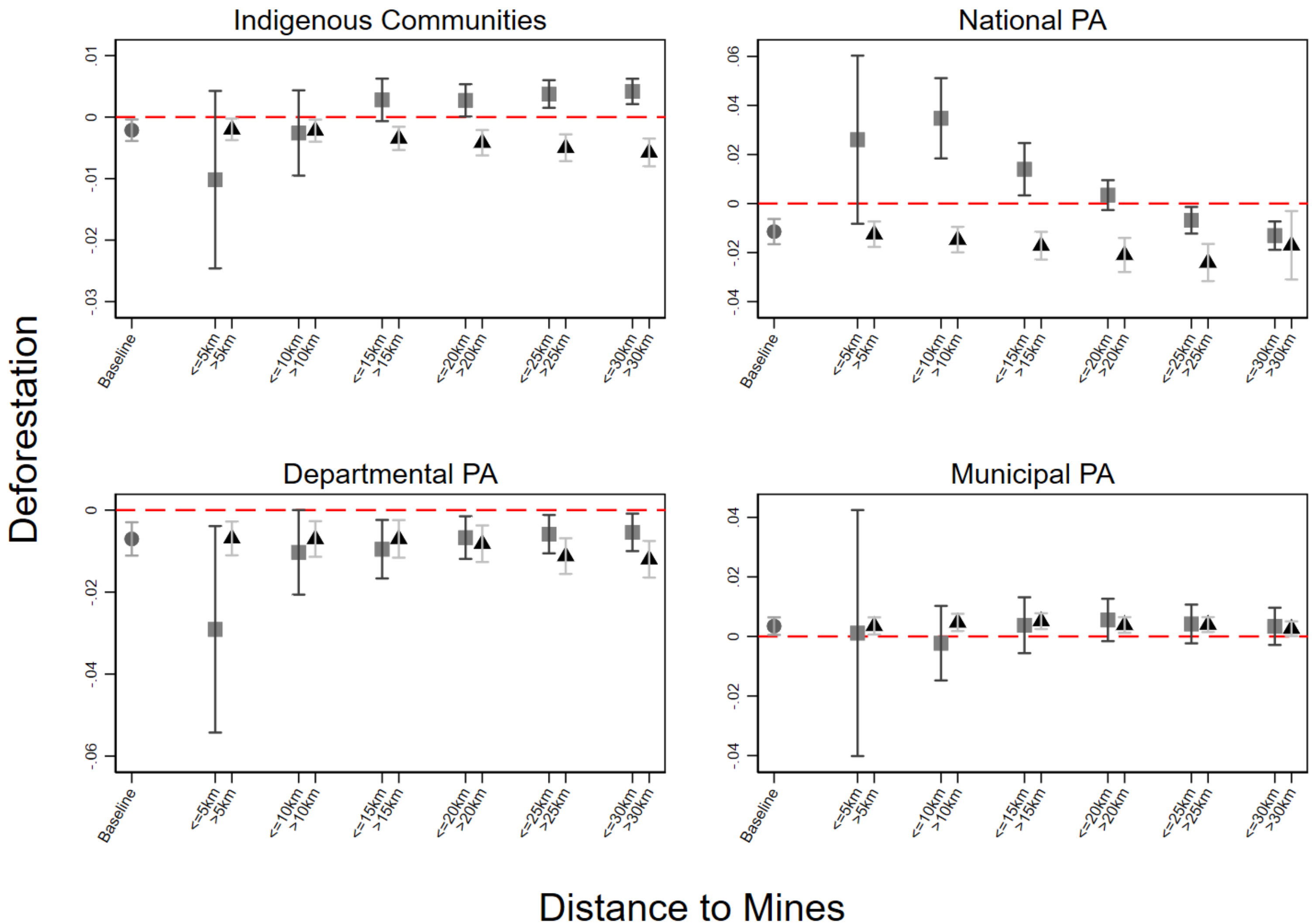

Figure 7: Distance to mining

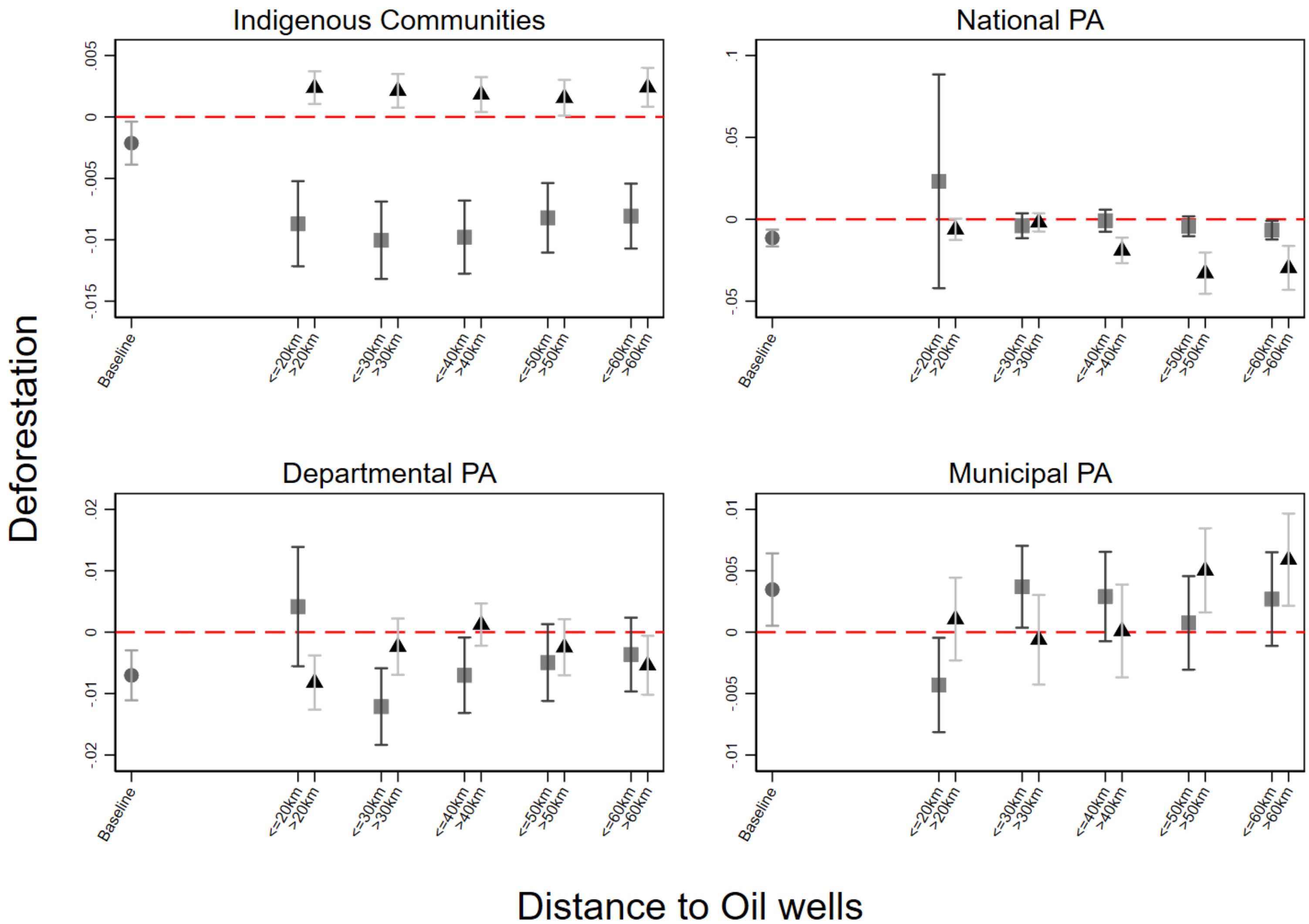

Figure 8: Distance to oil wells

Table 1: Descriptive statistics

	Mean (%)	**Standard Deviation (%)**	**N**	**Area (Km2)**	**Difference**	**P-value**
Bolivia	0.566	0.963	79168	680211	-	-
Private Lands	0.646	1.038	49987	429470	0.217	0.000
Indigenous Communities	0.537	0.841	9560	82383	-0.109	0.000
National PA	0.265	0.652	9230	78342	-0.381	0.000
Departmental PA	0.494	0.83	5847	50688	-0.152	0.000
Municipal PA	0.627	1.353	7870	68026	-0.019	0.154

Notes: The first row refers to the overall sample. For private lands, the comparison is against all kinds of PAs. Indigenous territories and PAs are compared with private lands.

Table 2: Test for continuous distribution of covariates

	Mean	SD	P-value		Mean	SD	P-value
Indigenous terriotires				**National PA**			
Forest cover (1985)	0.602	0.41	0.011	Forest cover (1985)	0.678	0.399	0.01
Altitud	311.234	234.45	0.013	Altitud	293.203	165.977	0.008
Slope	0.628	1.193	0.048	Slope	0.692	1.264	0.042
Aspect	163.77	58.485	0.597	Aspect	158.921	61.811	0.371
Roughness	29.976	54.277	0.048	Roughness	32.747	58.212	0.01
TPI	-0.012	3.686	0.075	TPI	-0.132	3.918	0.004
TRI	10.018	18.212	0.062	TRI	10.664	18.847	0.007
Precipitation	130.202	59.358	0.002	Precipitation	145.712	71.879	0.000
Joint Test			0.808	Joint Test			0.997
Departmental PA				**Municipal PA**			
Forest cover (1985)	0.709	0.363	0.006	Forest cover (1985)	0.602	0.412	0.028
Altitud	220.618	114.335	0.092	Altitud	225.231	103.907	0.000
Slope	0.334	0.599	0.609	Slope	0.37	0.569	0.000
Aspect	168.075	57.528	0.221	Aspect	171.406	55.557	0.037
Roughness	16.333	26.891	0.597	Roughness	18.206	26.066	0.003
TPI	-0.031	1.981	0.180	TPI	-0.03	1.856	0.079
TRI	5.505	9.152	0.314	TRI	6.091	8.607	0.006
Precipitation	139.2	48.785	0.030	Precipitation	128.213	35.996	0.238
Joint Test			0.837	Joint Test			0.387

Notes: Columns 1 and 2 report the mean and standard deviation of land characteristics between 50 kilometers from the border. Column 3 reports the p-value for the null hypothesis of continuity of the distributions of covariates at the cutoff.

Table 3: Baseline Results

	Indigenous Territories							
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
	P1	P2	Controls	1986-1992	1993-2021	1993-1998	1999-2008	2009-2021
Coefficient	-0.001	-0.005**	-0.002**	-	-0.003***	-0.004***	-0.002**	-0.002
	(0.001)	(0.002)	(0.001)	-	(0.001)	(0.001)	(0.001)	(0.001)
Observations	59,547	59,547	59,197	-	59,197	66,277	64,402	59,197
Mean of dep. var	0.0063	0.0063	0.0063	-	0.0066	0.0066	0.0063	0.0067

	National PA							
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
	P1	P2	Controls	1986-1992	1993-2021	1993-1998	1999-2008	2009-2021
Coefficient	-0.014***	-0.023***	-0.011***	0.005***	-0.010***	-0.010***	-0.008***	-0.007**
	(0.002)	(0.006)	(0.003)	(0.001)	(0.002)	(0.003)	(0.002)	(0.003)
Observations	59,217	59,217	58,804	72,857	58,804	67,788	64,009	58,804
Mean of dep. var	0.0059	0.0059	0.0059	0.0050	0.0062	0.0063	0.0059	0.0062

	Departmental PA							
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
	P1	P2	Controls	1986-1992	1993-2021	1993-1998	1999-2008	2009-2021
Coefficient	-0.004***	-0.006*	-0.007***	-0.002	-0.007***	-0.000	-0.018***	-0.005***
	(0.001)	(0.003)	(0.002)	(0.002)	(0.002)	(0.002)	(0.004)	(0.002)
Observations	55,834	55,834	55,469	72,431	55,469	63,208	59,748	55,469
Mean of dep. var	0.0063	0.0063	0.0063	0.0049	0.0066	0.0066	0.0063	0.0067

	Municipal PA							
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
	P1	P2	Controls	1986-1992	1993-2021	1993-1998	1999-2008	2009-2021
Coefficient	0.005***	0.010***	0.003**	-0.005***	0.004**	-0.011	-0.008***	0.007***
	(0.002)	(0.003)	(0.002)	(0.002)	(0.002)	(0.007)	(0.003)	(0.003)
Observations	57,857	57,857	57,522	68,615	57,522	58,905	57,636	57,522
Mean of dep. var	0.0064	0.0064	0.0064	0.0050	0.0068	0.0068	0.0065	0.0068

Notes: Dependent variable is the average percentage of deforestation observed in a spatial cell between 1986 and 2021. The controls include forest level in 1985, altitude, slope, aspect, roughness, the Terrain Ruggedness Index (TRI), and the Topographic Position Index (TPI). Columns 4 to 8 uses a polynomial of order 2 and the whole set of controls. All specifications include protected areas fixed effects. Standard errors in parenthesis are estimated using the nearest neighbor variance estimator. *, **, ***, are significance at the 10%, 5% and 1% level, respectively.

Table 4: Placebo estimations

	Indigenous Territories										
	Baseline	+10km	-10km	+20km	-20km	+30km	-30km	+40km	-40km	+50km	-50km
Coefficient	-0.002**	-0.000	-0.000	-0.000	0.000	-0.000	0.000	-0.000	-0.001	0.000	0.001
	(0.001)	(0.001)	(0.001)	(0.000)	(0.000)	(0.000)	(0.001)	(0.000)	(0.001)	(0.000)	(0.001)
Observations	59,197	59,197	59,197	59,197	59,197	59,197	59,197	59,197	59,197	59,197	59,197

	National PA										
	Baseline	+10km	-10km	+20km	-20km	+30km	-30km	+40km	-40km	+50km	-50km
Coefficient	-0.011***	0.007	0.001	-0.003**	-0.001	0.002	0.000	-0.001	-0.001	0.000	0.000
	(0.003)	(0.005)	(0.001)	(0.002)	(0.001)	(0.001)	(0.001)	(0.001)	(0.000)	(0.001)	(0.000)
Observations	58,804	58,804	58,804	58,804	58,804	58,804	58,804	58,804	58,804	58,804	58,804

	Departmental PA										
	Baseline	+10km	-10km	+20km	-20km	+30km	-30km	+40km	-40km	+50km	-50km
Coefficient	-0.007***	0.000	0.001	0.000	0.001**	-0.000	0.000	-0.000	0.000	-0.000	-0.000
	(0.002)	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)	(0.000)	(0.000)	(0.001)	(0.000)	(0.001)
Observations	55,469	55,469	55,469	55,469	55,469	55,469	55,469	55,469	55,469	55,469	55,469

	Municipal PA										
	Baseline	+10km	-10km	+20km	-20km	+30km	-30km	+40km	-40km	+50km	-50km
Coefficient	0.003**	0.001	0.001	0.000	-0.001	0.001*	0.001	-0.000	0.000	-0.000	0.000
	(0.002)	(0.002)	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)	(0.000)	(0.001)	(0.000)	(0.001)
Observations	57,522	57,522	57,522	57,522	57,522	57,522	57,522	57,522	57,522	57,522	57,522

Notes: Dependent variable is the average percentage of deforestation observed in a spatial cell between 1986 and 2021. All specifications use a polynomial of order 2 and the whole set of controls. All specifications include protected areas fixed effects. Standard errors in parenthesis are estimated using the nearest neighbor variance estimator. *, **, ***, are significance at the 10%, 5% and 1% level, respectively.

Table 5: Distance to cities

	<=5km	>5km	<=10km	>10km	<=15km	>15km	<=20km	>20km	<=25km	>25km	<=30km	>30km
Indigenous Territories												
Coefficient	0.001	-0.002**	-0.001	-0.002**	-0.001	-0.003**	-0.001	-0.002*	-0.000	-0.003**	0.001	-0.006***
	(0.003)	(0.001)	(0.002)	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)	(0.002)
Observations	3,138	56,059	9,116	50,081	15,998	43,199	22,473	36,724	28,374	30,823	33,522	25,675
Mean of dep. var	0.0109	0.0060	0.0092	0.0058	0.0085	0.0055	0.0082	0.0051	0.0080	0.0048	0.0078	0.0044
National PA												
Coefficient	-0.020***	-0.010***	-0.014***	-0.002	-0.010***	0.004	-0.010***	0.003	-0.011***	0.008	-0.012***	0.009
	(0.007)	(0.003)	(0.005)	(0.002)	(0.004)	(0.005)	(0.003)	(0.007)	(0.003)	(0.007)	(0.003)	(0.010)
Observations	2,617	56,187	7,808	50,996	13,956	44,848	19,860	38,944	25,277	33,527	30,147	28,657
Mean of dep. var	0.0113	0.0056	0.0096	0.0053	0.0089	0.0049	0.0084	0.0046	0.0082	0.0042	0.0079	0.0038
Departmental PA												
Coefficient	-0.001	-0.008***	0.004	-0.011***	0.003	-0.016***	0.002	-0.022***	0.002	-0.023***	0.001	-0.024***
	(0.006)	(0.002)	(0.003)	(0.002)	(0.002)	(0.003)	(0.002)	(0.003)	(0.002)	(0.003)	(0.002)	(0.003)
Observations	2,588	52,881	7,705	47,764	13,800	41,669	19,608	35,861	24,923	30,546	29,792	25,677
Mean of dep. var	0.0114	0.0061	0.0096	0.0058	0.0089	0.0054	0.0085	0.0051	0.0082	0.0048	0.0079	0.0044
Municipal PA												
Coefficient	0.036***	0.003*	0.005	0.004***	-0.003	0.006***	-0.000	0.006***	0.002	0.005***	0.002	0.004***
	(0.008)	(0.002)	(0.005)	(0.001)	(0.004)	(0.001)	(0.003)	(0.001)	(0.003)	(0.001)	(0.002)	(0.001)
Observations	2,819	54,703	8,353	49,169	14,863	42,659	21,081	36,441	26,786	30,736	31,909	25,613
Mean of dep. var	0.0108	0.0062	0.0092	0.0060	0.0087	0.0057	0.0083	0.0054	0.0081	0.0050	0.0079	0.0047

Notes: Dependent variable is the average percentage of deforestation observed in a spatial cell between 1986 and 2021. All specifications use a polynomial of order 2 and the whole set of controls. All specifications include protected areas fixed effects. Standard errors in parenthesis are estimated using the nearest neighbor variance estimator. *, **, ***, are significance at the 10%, 5% and 1% level, respectively.

Table 6: Distance to Routes

	Indigenous Territories											
	<= 10km	> 10km	<= 20km	> 20km	<= 30km	> 30km	<= 40km	> 40km	<= 50km	> 50km	<= 60km	> 60km
Coefficient	-0.001	-0.002***	-0.003	-0.002*	-0.006**	0.001	-0.003	-0.000	-0.002	-0.001	-0.002	-0.000
	(0.008)	(0.001)	(0.004)	(0.001)	(0.003)	(0.001)	(0.002)	(0.001)	(0.002)	(0.001)	(0.001)	(0.001)
Observations	1,793	57,404	5,722	53,475	10,574	48,623	15,605	43,592	20,687	38,510	25,367	33,830
Mean of dep. var	0.0127	0.0061	0.0115	0.0057	0.0107	0.0053	0.0098	0.0051	0.0090	0.0048	0.0083	0.0048

	National PA											
	<= 10km	> 10km	<= 20km	> 20km	<= 30km	> 30km	<= 40km	> 40km	<= 50km	> 50km	<= 60km	> 60km
Coefficient	-0.035***	-0.006**	-0.009**	-0.001	-0.009***	-0.006**	-0.004	-0.009***	-0.013***	-0.008***	-0.016***	-0.009***
	(0.006)	(0.003)	(0.004)	(0.002)	(0.003)	(0.003)	(0.003)	(0.003)	(0.003)	(0.003)	(0.003)	(0.003)
Observations	1,689	57,115	5,425	53,379	10,022	48,782	14,798	44,006	19,658	39,146	24,204	34,600
Mean of dep. var	0.0131	0.0057	0.0115	0.0053	0.0107	0.0049	0.0097	0.0046	0.0090	0.0043	0.0082	0.0042

	Departmental PA											
	<= 10km	> 10km	<= 20km	> 20km	<= 30km	> 30km	<= 40km	> 40km	<= 50km	> 50km	<= 60km	> 60km
Coefficient	-0.021	-0.007***	0.003	-0.008***	0.005**	-0.014***	0.006**	-0.013***	0.004	-0.012***	0.004	-0.014***
	(0.032)	(0.002)	(0.003)	(0.002)	(0.003)	(0.002)	(0.003)	(0.002)	(0.002)	(0.002)	(0.002)	(0.002)
Observations	1,606	53,863	5,130	50,339	9,436	46,033	13,823	41,646	18,271	37,198	22,250	33,219
Mean of dep. var	0.0133	0.0061	0.0117	0.0058	0.0110	0.0054	0.0101	0.0051	0.0093	0.0048	0.0087	0.0047

	Municipal PA											
	<= 10km	> 10km	<= 20km	> 20km	<= 30km	> 30km	<= 40km	> 40km	<= 50km	> 50km	<= 60km	> 60km
Coefficient	0.021**	0.003**	-0.004	0.003*	-0.004	0.004**	-0.005	0.004**	0.010***	0.000	0.011***	-0.001
	(0.011)	(0.002)	(0.005)	(0.002)	(0.005)	(0.002)	(0.005)	(0.001)	(0.004)	(0.002)	(0.003)	(0.001)
Observations	1,635	55,887	5,252	52,270	9,764	47,758	14,426	43,096	19,093	38,429	23,403	34,119
Mean of dep. var	0.0131	0.0062	0.0114	0.0059	0.0108	0.0056	0.0100	0.0052	0.0095	0.0049	0.0089	0.0048

Notes: Dependent variable is the average percentage of deforestation observed in a spatial cell between 1986 and 2021. All specifications use a polynomial of order 2 and the whole set of controls. All specifications include protected areas' fixed effects. Standard errors in parenthesis are estimated using the nearest neighbor variance estimator. *, **, ***, are significance at the 10%, 5% and 1% level, respectively.

Table 7: Distance to cattle ranching

	Indigenous Territories											
	<= 5km	> 5km	<= 10km	> 10km	<= 15km	> 15km	<= 20km	> 20km	<= 25km	> 25km	<= 30km	> 30km
Coefficient	-0.003***	-0.004***	-0.001	-0.005***	-0.001	-0.005***	-0.000	-0.006***	0.000	-0.006***	-0.000	-0.005***
	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)
Observations	6,859	52,338	10,872	48,325	13,437	45,760	15,683	43,514	17,825	41,372	19,876	39,321
Mean of dep. var	0.0046	0.0065	0.0054	0.0065	0.0058	0.0064	0.0061	0.0064	0.0063	0.0063	0.0065	0.0062

	National PA											
	<= 5km	> 5km	<= 10km	> 10km	<= 15km	> 15km	<= 20km	> 20km	<= 25km	> 25km	<= 30km	> 30km
Coefficient	-0.010**	-0.004	-0.016***	0.001	-0.016***	-0.001	-0.016***	-0.007***	-0.016***	-0.007***	-0.015***	-0.007***
	(0.005)	(0.003)	(0.005)	(0.002)	(0.004)	(0.002)	(0.004)	(0.002)	(0.004)	(0.002)	(0.004)	(0.002)
Observations	6,114	52,690	9,478	49,326	11,639	47,165	13,525	45,279	15,301	43,503	16,966	41,838
Mean of dep. var	0.0049	0.0060	0.0058	0.0059	0.0063	0.0058	0.0065	0.0057	0.0068	0.0056	0.0069	0.0055

	Departmental PA											
	<= 5km	> 5km	<= 10km	> 10km	<= 15km	> 15km	<= 20km	> 20km	<= 25km	> 25km	<= 30km	> 30km
Coefficient	-0.001	-0.005**	-0.001	-0.008***	0.001	-0.009***	-0.001	-0.010***	-0.004	-0.009***	-0.004*	-0.010***
	(0.003)	(0.002)	(0.002)	(0.003)	(0.002)	(0.003)	(0.002)	(0.003)	(0.002)	(0.003)	(0.002)	(0.003)
Observations	6,175	49,294	9,592	45,877	11,834	43,635	13,887	41,582	15,827	39,642	17,679	37,790
Mean of dep. var	0.0048	0.0065	0.0057	0.0064	0.0061	0.0064	0.0064	0.0063	0.0066	0.0062	0.0067	0.0061

	Municipal PA											
	<= 5km	> 5km	<= 10km	> 10km	<= 15km	> 15km	<= 20km	> 20km	<= 25km	> 25km	<= 30km	> 30km
Coefficient	-0.014	0.004***	-0.008	0.005***	-0.008	0.005***	-0.011**	0.005***	-0.008	0.004**	-0.010***	0.004**
	(0.010)	(0.001)	(0.005)	(0.001)	(0.006)	(0.001)	(0.006)	(0.001)	(0.005)	(0.002)	(0.004)	(0.002)
Observations	6,368	51,154	10,145	47,377	12,724	44,798	15,037	42,485	17,268	40,254	19,294	38,228
Mean of dep. var	0.0047	0.0067	0.0054	0.0067	0.0058	0.0066	0.0061	0.0066	0.0062	0.0065	0.0063	0.0065

Notes: Dependent variable is the average percentage of deforestation observed in a spatial cell between 1986 and 2021. All specifications use a polynomial of order 2 and the whole set of controls. All specifications include protected areas fixed effects. Standard errors in parenthesis are estimated using the nearest neighbor variance estimator. *, **, ***, are significance at the 10%, 5% and 1% level, respectively.

Table 8: Distance to mines

	Indigenous Territories											
	<= 5km	> 5km	<= 10km	> 10km	<= 15km	> 15km	<= 20km	> 20km	<= 25km	> 25km	<= 30km	> 30km
Coefficient	-0.010	-0.002**	-0.003	-0.002**	0.003	-0.003***	0.003**	-0.004***	0.004***	-0.005***	0.004***	-0.006***
	(0.007)	(0.001)	(0.004)	(0.001)	(0.002)	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)
Observations	2,097	57,100	5,436	53,761	9,141	50,056	13,130	46,067	17,214	41,983	21,267	37,930
Mean of dep. var	0.0078	0.0062	0.0075	0.0062	0.0073	0.0061	0.0072	0.0060	0.0071	0.0060	0.0070	0.0059

	National PA											
	<= 5km	> 5km	<= 10km	> 10km	<= 15km	> 15km	<= 20km	> 20km	<= 25km	> 25km	<= 30km	> 30km
Coefficient	0.026	-0.012***	0.035***	-0.015***	0.014**	-0.017***	0.003	-0.021***	-0.007**	-0.024***	-0.013***	-0.017**
	(0.017)	(0.003)	(0.008)	(0.003)	(0.005)	(0.003)	(0.003)	(0.004)	(0.003)	(0.004)	(0.003)	(0.007)
Observations	2,025	56,779	5,375	53,429	9,076	49,728	13,060	45,744	17,138	41,666	21,162	37,642
Mean of dep. var	0.0077	0.0058	0.0074	0.0057	0.0072	0.0056	0.0070	0.0055	0.0069	0.0055	0.0068	0.0053

	Departmental PA											
	<= 5km	> 5km	<= 10km	> 10km	<= 15km	> 15km	<= 20km	> 20km	<= 25km	> 25km	<= 30km	> 30km
Coefficient	-0.029**	-0.007***	-0.010*	-0.007***	-0.010***	-0.007***	-0.007**	-0.008***	-0.006**	-0.011***	-0.005**	-0.012***
	(0.013)	(0.002)	(0.005)	(0.002)	(0.004)	(0.002)	(0.003)	(0.002)	(0.002)	(0.002)	(0.002)	(0.002)
Observations	1,966	53,503	5,166	50,303	8,704	46,765	12,482	42,987	16,316	39,153	20,048	35,421
Mean of dep. var	0.0076	0.0063	0.0074	0.0062	0.0074	0.0061	0.0073	0.0060	0.0072	0.0059	0.0071	0.0059

	Municipal PA											
	<= 5km	> 5km	<= 10km	> 10km	<= 15km	> 15km	<= 20km	> 20km	<= 25km	> 25km	<= 30km	> 30km
Coefficient	0.001	0.004**	-0.002	0.005***	0.004	0.005***	0.006	0.004***	0.004	0.004***	0.003	0.003**
	(0.021)	(0.001)	(0.006)	(0.001)	(0.005)	(0.001)	(0.004)	(0.001)	(0.003)	(0.001)	(0.003)	(0.001)
Observations	1,985	55,537	5,165	52,357	8,671	48,851	12,428	45,094	16,283	41,239	20,102	37,420
Mean of dep. var	0.0082	0.0064	0.0081	0.0063	0.0082	0.0061	0.0081	0.0060	0.0079	0.0059	0.0078	0.0057

Notes: Dependent variable is the average percentage of deforestation observed in a spatial cell between 1986 and 2021. All specifications use a polynomial of order 2 and the whole set of controls. All specifications include protected areas fixed effects. Standard errors in parenthesis are estimated using the nearest neighbor variance estimator. *, **, ***, are significance at the 10%, 5% and 1% level, respectively.

Table 9: Distance to oil

	Indigenous Territories											
	<= 10km	> 10km	<= 20km	> 20km	<= 30km	> 30km	<= 40km	> 40km	<= 50km	> 50km	<= 60km	> 60km
Coefficient	-0.013***	0.001**	-0.009***	0.002***	-0.010***	0.002***	-0.010***	0.002**	-0.008***	0.002**	-0.008***	0.002***
	(0.003)	(0.001)	(0.002)	(0.001)	(0.002)	(0.001)	(0.002)	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)
Observations	2,384	56,813	7,003	52,194	12,027	47,170	16,707	42,490	20,989	38,208	24,759	34,438
Mean of dep. var	0.0108	0.0061	0.0102	0.0058	0.0096	0.0054	0.0091	0.0052	0.0086	0.0050	0.0083	0.0049

	National PA											
	<= 10km	> 10km	<= 20km	> 20km	<= 30km	> 30km	<= 40km	> 40km	<= 50km	> 50km	<= 60km	> 60km
Coefficient	0.058	-0.008***	0.023	-0.006*	-0.004	-0.002	-0.001	-0.019***	-0.004	-0.033***	-0.007**	-0.030***
	(0.039)	(0.003)	(0.033)	(0.003)	(0.004)	(0.003)	(0.003)	(0.004)	(0.003)	(0.006)	(0.003)	(0.007)
Observations	2,104	56,700	6,308	52,496	11,029	47,775	15,674	43,130	19,996	38,808	23,800	35,004
Mean of dep. var	0.0116	0.0057	0.0108	0.0053	0.0100	0.0049	0.0093	0.0046	0.0086	0.0045	0.0082	0.0043

	Departmental PA											
	<= 10km	> 10km	<= 20km	> 20km	<= 30km	> 30km	<= 40km	> 40km	<= 50km	> 50km	<= 60km	> 60km
Coefficient	-0.149	-0.009***	0.004	-0.008***	-0.012***	-0.002	-0.007**	0.001	-0.005	-0.002	-0.004	-0.005**
	(0.159)	(0.002)	(0.005)	(0.002)	(0.003)	(0.002)	(0.003)	(0.002)	(0.003)	(0.002)	(0.003)	(0.002)
Observations	2,052	53,417	6,170	49,299	10,724	44,745	14,947	40,522	18,598	36,871	21,632	33,837
Mean of dep. var	0.0113	0.0061	0.0107	0.0058	0.0102	0.0054	0.0097	0.0051	0.0092	0.0049	0.0088	0.0047

	Municipal PA											
	<= 10km	> 10km	<= 20km	> 20km	<= 30km	> 30km	<= 40km	> 40km	<= 50km	> 50km	<= 60km	> 60km
Coefficient	-0.005	0.004**	-0.004**	0.001	0.004**	-0.001	0.003	0.000	0.001	0.005***	0.003	0.006***
	(0.009)	(0.002)	(0.002)	(0.002)	(0.002)	(0.002)	(0.002)	(0.002)	(0.002)	(0.002)	(0.002)	(0.002)
Observations	2,126	55,396	6,461	51,061	11,492	46,030	16,472	41,050	21,062	36,460	25,051	32,471
Mean of dep. var	0.0110	0.0063	0.0101	0.0060	0.0094	0.0057	0.0088	0.0055	0.0084	0.0053	0.0082	0.0051

Notes: Dependent variable is the average percentage of deforestation observed in a spatial cell between 1986 and 2021. All specifications use a polynomial of order 2 and the whole set of controls. All specifications include protected areas fixed effects. Standard errors in parenthesis are estimated using the nearest neighbor variance estimator. *, **, ***, are significance at the 10%, 5% and 1% level, respectively.