Patricia G. C. Ruggiero

Impacto de políticas de conservação e ciclos eleitorais sobre áreas protegidas e a cobertura florestal na Mata Atlântica Brasileira

Impacts of conservation policies and electoral cycles on protected areas and forest cover in the Brazilian Atlantic Forest

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Abstract

What strategies really work against the intense loss of native habitat? How can we improve forested areas around the world? The creation of protected areas - natural reserves aiming at preserving biodiversity and ecological relations in pristine environments – has been the main strategy against land surface transformation and the loss of natural capital. Recently, financial incentives have promptly become the most promising and accepted new strategies for conservation, and several experiences were implemented, particularly in the tropics. However, we urge to systematically evaluate the effectiveness of these mechanisms, both old and new, and several researchers have argued that econometric methods, traditionally used in impact evaluation of development, health, and education policies, should be applied in assessing the impacts of interventions for environmental conservation. Our goal in the present study is to estimate the effectiveness of two popular financial incentives against native vegetation loss in Brazil: payments for ecosystem services (PES) and the intergovernmental fiscal transfer based on ecological criteria - the Ecological ICMS. We used a counterfactual approach, which seeks adequate control groups in order to estimate what would have happened in the absence of the intervention – i.e., we search for additional gain or additionality. Payment for ecosystem services programs were evaluated regarding the promotion of additional native forest areas while the Ecological ICMS was assessed for its ability to promote new legally protected areas for conservation. We evaluated two of the oldest and most well established payment for ecosystem services programs in the Atlantic Forest, located in the Cantareira Water System, near the metropolis of São Paulo: the *Produtor de Água* (Water Producer) in Joanópolis and Nazaré Paulista (SP), and the *Conservador das Águas* (Water Conservationist), in Extrema (MG). In total, 83 rural properties contracted by the programs, and 83 properties selected as control units, were mapped using satellite images available in Google since 2003 and evaluated for the presence of regeneration and deforestation in three moments: before the program, at the time of program implementation and after program implementation. We show that PES programs positively affect the area under regeneration inside contracted rural properties, but does not affect deforestation. The average effect on regeneration was 0.69% of property area per year, which means that in 0.69% of the evaluated property area there would be no regeneration of native vegetation were it not for PES programs intervention. This result shows the additional effect of PES programs on regeneration of native forest but also shed light that this is a slow process. Secondly, the effect of the Ecological ICMS mechanism was evaluated regarding the expansion of Conservation Units - protected areas for conservation. We compared 2,481 municipalities from eight states in the Atlantic Forest region, in a panel data (i.e., longitudinal database) covering the period from 1987 to 2016. We show that municipalities under the incentive of the ecological ICMS law, when compared to control municipalities not subject to the same mechanism, present a small increase in area of new conservation units in the first years of law existence. However, this effect decreases over time disappearing about 10 years after the approval of the law. In addition to the temporary effect, we show that protected areas created, particularly by local governments, are preferably Environmental Protection Areas (APA) - a category that imposes few restrictions on land use and promotes little contribution to the conservation of natural resources. In order to estimate the effect of the Ecological ICMS on regeneration and deforestation processes, we accessed a national database on land use change between 1985 and 2017. However, observing deforestation patterns throughout this historical data series, it called our attention the possible existence of electoral cycles affecting deforestation - which gave rise to the third chapter of the thesis. We analyze deforestation (percentage deforested over previous forested area) in a panel with 2,277 municipalities within the Atlantic Forest region, from seven states of the South and Southeast, considering the period from 1991 to 2014. We demonstrate the existence of electoral deforestation cycles, linked to state elections in five observed states and municipal elections in three states. We also show evidence that elections in which the incumbent is losing by a large margin of votes tend to be the events with greater observed deforestation, and that party alignment between mayor and governor parties can also contribute to an increase in deforestation during municipal elections. Financial mechanisms, at least in the way they have been designed to date, are limited as major instruments in achieving the goals of conservation and restoration of native forests. In addition, eventhough there is a major effort to create new mechanisms and increase conservation gains, it is of concern that these gains can be offset by losses due to political motivations, even in the region of the country with one of the most stringent legislations for tropical forests in the world.

Resumo

Quais estratégias realmente funcionam contra a intensa perda de habitat nativo? Como podemos aumentar nossas áreas de florestas? A criação de áreas protegidas - reservas naturais com o objetivo de preservar a biodiversidade e as relações ecológicas de ambientes pristinos – tem sido a principal estratégia contra as transformações na superfície da Terra e a perda de habitat nativo. Recentemente, incentivos financeiros se tornaram rapidamente promissores e populares instrumentos para a conservação ambiental, e diversas experiências foram implementadas, particularmente nas florestas tropicais. No entanto, carecemos de avaliações sistemáticas sobre a efetividade destes mecanismos, antigos e novos, e diversos pesquisadores argumentam que métodos da econometria, tradicionalmente usados na avaliação de políticas de desenvolvimento, saúde e educação, devem ser aplicados na avaliação dos impactos das intervenções para a conservação ambiental. O objetivo do presente trabalho foi estimar a efetividade de dois dos mais populares incentivos financeiros no combate à perda de vegetação nativa no Brasil: os pagamentos por serviços ambientais (PSA) e a transferência fiscal intergovernamental com base em critérios ecológicos - o ICMS Ecológico, concebido e implementado no Brasil de forma inédita. Para estas avaliações, nos baseamos na abordagem contrafactual, que busca adequados grupos de controle com o objetivo de estimar o que haveria ocorrido na ausência da intervenção estudada, isto é, se houve ganho adicional ou adicionalidade. Os programas de pagamento por serviços ambientais foram avaliados quanto a promoção de áreas adicionais de floresta nativa enquanto que o ICMS Ecológico foi avaliado quanto a sua capacidade de promover novas áreas legalmente destinadas à conservação de recursos naturais - as Unidades de Conservação. Avaliamos dois dos mais antigos e bem estabelecidos programas de pagamento por serviços ambientais na Mata Atlântica, localizados no Sistema Cantareira, ao redor da região metropolitana de São Paulo: o Produtor de Água, em Joanópolis e Nazaré Paulista (SP), e o Conservador das Águas, em Extrema (MG). No total, 83 propriedades rurais contratadas pelos programas, e 83 propriedades selecionadas como controle, foram acompanhadas por imagens de satélite disponíveis no Google desde 2003 e avaliadas quanto a presença de regeneração e desmatamento, em três momentos: antes dos programas, no momento da implementação e após um período de operação dos programas. Mostramos que os programas de PSA afetam positivamente a área em regeneração dentro das propriedades rurais contratadas, mas não afeta o desmatamento. O efeito médio sobre a regeneração foi de 0.69% da área da propriedade ao ano, o que significa que em 0.69% da área das propriedades avaliadas não haveria regeneração de vegetação nativa não fosse a intervenção dos programas avaliados. Esse resultado mostra o efeito adicional dos programas de PSA sobre o processo de regeneração de floresta nativa mas nos revela também que se trata de um processo

lento. Na segunda etapa da pesquisa, foi estimado o efeito do mecanismo do ICMS Ecológico sobre a expansão de Unidades de Conservação. Comparamos 2.481 municípios de oito estados na região da Mata Atlântica, em um painel (i.e. base de dados longitudinal) compreendendo o período de 1987 a 2016. Demonstramos que municípios sob o incentivo da lei do ICMS Ecológico, quando comparados com municípios controle não sujeitos ao mesmo mecanismo, apresentam um pequeno aumento em área de novas unidades de conservação nos primeiros anos de existência da lei. No entanto, este efeito decresce ao longo do tempo desaparecendo cerca de 10 anos após a aprovação da lei. Além do efeito temporário, mostramos que as áreas protegidas criadas, particularmente pelos governos locais, são preferencialmente Áreas de Proteção Ambiental (APA) - categoria que impõe poucas restrições ao uso da terra e que traz pequena contribuição à conservação dos recursos naturais. Com o intuito de estimar o efeito do ICMS ecológico sobre processos de regeneração e desmatamento, acessamos uma base de dados nacional sobre mudança no uso da terra no período entre 1985 e 2017. Observando os padrões de desmatamento ao longo desta série histórica, nos chamou a atenção a possível presença de ciclos eleitorais afetando o desmatamento - o que deu origem ao terceiro capítulo da tese. Analisamos o desmatamento (percentual desmatado sobre a área prévia de floresta) em um painel com 2.277 municípios na região da Mata Atlântica, dos sete estados do Sul e Sudeste, considerando o período de 1991 a 2014. Nós demonstramos a existência de ciclos eleitorais de desmatamento, ligados às eleições estaduais, em cinco dos estados observados, e municipais, em três estados. Também mostramos evidências de que as eleições nas quais o incumbente está perdendo por uma ampla margem de votos tendem a ser os eventos com maior desmatamento observado, e que o alinhamento entre o partido do prefeito e do governador também pode contribuir para o aumento do desmatamento durante as eleições municipais. Os mecanismos financeiros, ao menos na forma que foram concebidos até o momento, se mostram bastante limitados como instrumentos majoritários no alcance das metas de conservação e restauração de florestas nativas. Além disso, se existe um grande esforço para que novos mecanismos sejam criados e que se possa ampliar os ganhos em conservação, é preocupante que estes ganhos possam ser compensados por perdas decorrentes de motivações políticas, mesmo na região do país na qual vigora uma das legislações mais rígidas sobre florestas tropicais do mundo.

Contents

Introduction

The Earth's land surface has been going through unprecedented changes and consequent loss of species, making urgent actions that may prevent or reduce those impacts. Historically, and for a long period of time, creating natural reserves was the main strategy adopted to prevent pristine native areas from human driven transformation (Chape et al., 2008). The designation of protected areas (PA) is still considered an essential strategy for conserving biodiversity, but a large scale planning, considering the network of protected areas and the quality and permeability of agricultural or countryside matrix areas, is as crucial to assure the survival of wild species (Mendenhall et al., 2014). In this sense, command and control prescriptive mechanisms are traditionally used to promote a more sustainable landscape planning and natural resource use in private areas (Holling and Meffe, 1996). In addition, alternative strategies especially applying financial incentives were proposed to broaden the tool options for conservation, generating a lot of positive expectations to come from those novel mechanisms (Kemkes et al., 2010). However, how (or if) those strategies contribute to environmental conservation is an urgent question not fully answered. In spite of several recent calls to improve the impact evaluation of conservation tools (Ferraro and Pattanayak, 2006; Pattanayak et al., 2010; Baylis et al., 2015; Le Velly and Dutilly, 2016; Salzman et al., 2018), there are few studies that accomplish this demand up to now. It is thus time to assess the results of those interventions, using consolidated public policy impact methods largely implemented for development, education and health policies (Angrist and Pischke, 2008). This thesis was conceived within this broad scope. Our main objective was to evaluate the effectiveness of recently proposed and largely implemented financial mechanisms to promote environmental conservation, particularly Payments for Ecosystem Services (PES) and the intergovernmental fiscal transfer Ecological ICMS (ICMS-E).

Payments for Ecosystem Services (PES) mechanisms - i.e. paying landowners to conserve natural habitats inside their properties - were proposed in order to get additional areas for conservation on rural landscapes (besides PA) and have been implemented for almost three decades, mainly in the tropics (Ezzine-de Blas et al., 2016). The most antique and studied PES in the world is the Costa Rican program for forest conservation implemented since 1996, where payments promoted the increase of 11-17% of forest inside land properties in eight years (Arriagada et al., 2012). However, data about forest regeneration suggests that PES may have accelerated agricultural land abandonment, which in turn increased forest cover, instead of generating a new conservation behavior outcome (Daniels et al., 2010). The first chapter of this thesis provides the first quantitative assessment of PES contribution to forest conservation across South America, using a counterfactual approach, comparing land properties engaged in PES programs in Brazil with adequate control properties. Besides, our research on PES has markedly one major hairsplitting distinction from all others: we measure forest loss (deforestation) and forest gain (regeneration) pixel-per-pixel separately inside land properties as our main outcome, instead of measuring total forest cover change. This distinction allows to precisely observe the two processes (deforestation and regeneration) separately, which is essential as they may be related to different modes through which the intervention may operate - avoiding deforestation or increasing regeneration. Similarly to what was observed in Costa Rica, we show that results obtained for PES in Brazil were mainly driven by forest regeneration and we suggest that, instead of promoting a new behavior among landowners, this effect may result from a synergy, in this case, with increasing legislation compliance.

Payments for Ecosystem Services aim at enrolling land properties in environmental conservation, consequently producing a diffuse outcome of new small forested areas. The Ecological ICMS is also a financial mechanism; however, it is focused on the public actors and their role to assure ecosystem services for the long term (Ring, 2008). It was conceived by Brazilians, popularized throughout the country in the last 20 years, and has been suggested as innovative and promising for environmental conservation (Farley et al., 2010). The Ecological ICMS is a tax on goods and services that can be understood as "the 'value-added' of local ecological services, i.e. the benefits that cross the boundaries of local jurisdictions, which are of special interest" (Ring, 2008). It is collected from municipalities by the state governments, and then redistributed to local government based on a set of ecological indicators, mainly the proportion of the municipality covered with legal PA (Loureiro, 2002). So, it is expected to promote the increase of PA in the states that have implemented the law (May et al., 2002). Nevertheless, if counterfactual approach evaluations are rare for PES programs, in the case of the Ecological ICMS they are absent. In the second chapter, we evaluated the effect of the ICMS-E law on the promotion of new state and municipal PA in eight different states. We demonstrate that the ICMS-E promotes an increase in state and municipal PA, for both PA categories, restricted and unrestricted, during the first 10 years of law existence. However, after this period, the effect diminishes or disappears, specially when the ICMS-E is expected to work as an incentive and municipal PA are considered. Besides shedding light on the lack of persistence of the ICMS-E as an incentive, our results suggest that the PA expansion, through the increase in their number and area, not necessarily represent an addition in conservation quality or protection category. In both observed context, state and municipal, very unrestricted PA, such as Environmental Protected Areas (Áreas de Proteção Ambiental – APA), are the ones that undergo the higher increase as a percentage of total municipal area. More restricted PA only present significant positive increase when state PA are observed, suggesting that the ICMS-E policy creates different incentives environments for state and municipal governments.

We also planned to investigate the effect of the Ecological ICMS on the increase of forest cover, mainly through regeneration, which is in fact the final conservation goal of these interventions. As we did with forest cover mapping during the investigation of PES, we distinguished forest gain (regeneration) from forest loss (deforestation) pixel-per-pixel, now applied to a huge database generated by the Mapbiomas Project (MapBiomas, 2018) for the Atlantic Forest region. At a first sight, observing general patterns, it called our attention that, while regeneration did not show much evident variation, deforestation seemed to present an interesting pattern. We suspected that deforestation could vary as a consequence of election events, which is a phenomena already described in the political economy literature for economical and social outcomes (Nordhaus, 1975; Shi and Svensson, 2006; Brender and Drazen, 2005) and only recently described for forest resources (Burgess et al., 2012).

So, we dig deep into this investigation and observed that political deforestation cycles related mainly to state elections, but also to municipal elections. A political deforestation cycle was recently observed in the Brazilian Amazon, where deforestation increases 8-10% during municipal elections in municipalities where the incumbent is running for re-elections, (Pailler, 2018). However, it is surprising to see that this process takes place in the Atlantic Forest region, arguably the country region with the most stringent environmental legislation and the most equipped state environmental organizations. So, of concern is that every two years in Brazil presents an opportunity for politically motivated rent-seeking of forest resources, that can offset any inter elections conservation accomplishments.

The next pages detail those stories, presenting our findings and argumentation. Chapter one describes the impact evaluation of two watershed PES programs, among the most well-established programs in Brazil, on forest cover increase inside land properties. In chapter two, we estimate the effect of the intergovernmental fiscal transfer for ecological purposes, the Ecological ICMS, on the expansion of protected areas. Both evaluation methods are based on a counterfactual approach that uses adequate control units to estimate what would have happened in the absence of the intervention (Ferraro, 2009). Finally, chapter three explores the patterns of political cycles linked to deforestation dynamics in the Atlantic Forest region. We conclude comparing the two studied financial mechanisms and the implications of political motivations on conservation outcomes.

Bibliography

Angrist, J. D., Pischke, J.-S., 2008. Mostly Harmless Econometrics: An Empiricistís Companion. Princeton University Press.

Arriagada, R. A., Ferraro, P. J., Sills, E., Pattanayak, S. K., Cordero-Sancho, S., 2012. Do Payments for Environmental Services Affect Forest Cover? A Farm-Level Evaluation from Costa Rica. Land Economics 88 (2), 382–399.

Baylis, K., Honey-Rosés, J., Börner, J., Corbera, E., Ezzine-de Blas, D., Ferraro, P. J., Lapeyre, R., Persson, U. M., Wunder, S., Pfaff, A., 2015. Mainstreaming Impact Evaluation in Nature Conservation. Conservation Letters 9 (1), 58–64.

Brender, A., Drazen, A., 2005. Political budget cycles in new versus established democracies. Journal of monetary Economics 52 (7), 1271–1295.

Chape, S., Spalding, M., Jenkins, M., 2008. The world's protected areas. Prepared by the UNEP World Conservation Monitoring Centre.

Daniels, A., Bagstad, K. J., Esposito, V., Moulaert, A., Rodrigues, C. M., Sep. 2010. Understanding the impacts of Costa Rica's PES: Are we asking the right questions? Ecological Economics 69 (11), 2116–2126.

Ezzine-de Blas, D., Wunder, S., Ruiz-Pérez, M., Moreno-Sanchez, R. d. P., 2016. Global Patterns in the Implementation of Payments for Environmental Services. PLoS ONE 11 (3), e0149847–16.

Farley, J., Aquino, A., Daniels, A., Moulaert, A., Lee, D., Krause, A., Sep. 2010. Global mechanisms for sustaining and enhancing PES schemes. Ecological Economics 69 (11), 2075–2084.

Ferraro, P. J., 2009. Counterfactual thinking and impact evaluation in environmental policy. New Directions for Evaluation 2009 (122), 75–84.

Ferraro, P. J., Pattanayak, S. K., 2006. Money for Nothing? A Call for Empirical Evaluation of Biodiversity Conservation Investments. PLoS Biology 4 (4), e105.

Holling, C. S., Meffe, G. K., 1996. Command and control and the pathology of natural resource management. Conservation biology 10 (2), 328–337.

Kemkes, R. J., Farley, J., Koliba, C. J., 2010. Determining when payments are an effective policy approach to ecosystem service provision. Ecological Economics 69 (11), 2069–2074.

Le Velly, G., Dutilly, C., 2016. Evaluating Payments for Environmental Services: Methodological Challenges. PLoS ONE 11 (2), e0149374–20.

Loureiro, W., 2002. Contribuição do ICMS Ecológico à conservação da biodiversidade no Estado do Paraná. Ph.D. thesis, Engenharia Florestal, Universidade Federal do Paraná.

MapBiomas, 2018. Mapbiomas project collection 3.0 of brazilian land cover & use map series. Accessed: June 28 2018.
URL http://mapbiomas.org

May, P. H., Neto, F. V., Denardin, V., Loureiro, W., May 2002. Using Fiscal Instruments to Encourage Conservation: Municipal Responses to the ‘Ecological’ Value-added Tax in Paraná and Minas Gerais, Brazil. In: Selling Forest Environmental Services. pp. 1–28.

Mendenhall, C. D., Karp, D. S., Meyer, C. F. J., Hadly, E. A., Daily, G. C., Apr. 2014. Predicting biodiversity change and averting collapse in agricultural landscapes. Nature 509 (7499), 213–217.

Nordhaus, W. D., 1975. The political business cycle. The review of economic studies 42 (2), 169–190.

Pailler, S., 2018. Re-election incentives and deforestation cycles in the Brazilian Amazon. Journal of Environmental Economics and Management 88, 345–365.

Pattanayak, S. K., Wunder, S., Ferraro, P. J., 2010. Show Me the Money: Do Payments Supply Environmental Services in Developing Countries? Review of Environmental Economics and Policy 4 (2), 254–274.

Ring, I., 2008. Integrating local ecological services into intergovernmental fiscal transfers: The case of the ecological ICMS in Brazil. Land Use Policy 25 (4), 485–497.

Salzman, J., Bennett, G., Carroll, N., Goldstein, A., Jenkins, M., 2018. The global status and trends of Payments for Ecosystem Services. Nature Sustainability, 1–9.

Shi, M., Svensson, J., 2006. Political budget cycles: Do they differ across countries and why? Journal of public economics 90 (8-9), 1367–1389.

Chapter 1

Payment for ecosystem services programs in the Brazilian Atlantic Forest: effective but not enough

Patricia G. C. Ruggiero[1], Jean Paul Metzger[1], Leandro Reverberi Tambosi[2], and Liz Nichols[3]

[1]Department of Ecology, University of São Paulo, BR

[2]Center for Engineering, Modeling and Applied Social Sciences, Federal University of ABC, BR

[3]Bioscience Department, Swarthmore College, US

1.1 Abstract

Payment for ecosystem services (PES) programs are economic incentives intended to generate conservation benefits on private properties. The global portfolio of PES programs is estimated to represent an annual investment of more than USD $36 billion. Despite this substantial investment, the continued lack of systematic and rigorous impact evaluation of PES has contributed to uncertainty regarding the effectiveness of this tool for conservation. We assessed the effectiveness of two watershed PES programs in promoting native forest conservation in the Brazilian Atlantic Forest. Those two focal programs are among the most well-established PES programs in Brazil, and form part of a larger network of PES programs intended to protect the drinking water supply of over 19 million people in the São Paulo metropolitan region. Using a counterfactual approach, we examined if enrollment in a PES program contributed to the conservation of on-farm native forest cover. With propensity score matching, we identified a set of neighboring, non-enrolled 'control' properties, with similar size, altitude, soil type, demographic density, presence of water sources and forest cover. We then estimated forest cover on enrolled and control properties before and after PES implementation with a difference-in-differences method, modeled as the probability of an observed change in on-farm forest area as a consequence of PES enrollment. We found that PES has a positive effect on forest cover, with PES enrollment over a five-year period associated with an additional 2.8 - 5.6% of farm area coverage in native Atlantic Forest, through forest regeneration. PES enrollment was associated with a non-significant trend toward decreased loss of vegetation. We discuss the implications of these results for understanding the contribution of PES to additionality in forest conservation. While positive, the relatively weak impact of PES on forest regeneration suggests that environmental managers should not count exclusively on PES mechanism to achieve conservation goals.

1.2 Introduction

Payment for ecosystem services (PES) programs are economic incentives for environmental conservation, They are generally defined as "a transfer of resources between social actors, which aims to create incentives to align individual and/or collective land use decisions with the social interest in the management of natural resources" (Muradian et al., 2010). Globally, PES have became a popular complementary strategy to existing conservation strategies like protected areas (Schomers and Matzdorf, 2013; Ezzine-de Blas et al., 2016; Salzman et al., 2018). Despite the extensive investment in PES as a conservation tool, the correlates of conservation success in PES programs remain poorly understood (Ferraro and Pattanayak, 2006; Pattanayak et al., 2010; Naeem et al., 2015; Ezzine-de Blas et al., 2016; Le Velly and Dutilly, 2016).

One reason why clear links between PES implementation and conservation outcomes are rare may be study design. Many studies either measure forest conservation outcomes in ways that obscure important underlying socio-ecological processes, or use designs that fail to demonstrate additionality (i.e. conservation outcomes would not have happened without the PES intervention). For example, a large number of studies that evaluate PES impact consider net forest change as the main indicator of avoided deforestation (Sánchez-Azofeifa and Boomhower, 2007; Honey-Rosés et al., 2011; Arriagada et al., 2012; Robalino and Pfaff, 2013; Yang et al., 2013; Jayachandran et al., 2017; Goh and Yanosky, 2016). This is problematic, as changes in net forest cover may be driven by avoided deforestation or alternatively through forest regeneration - processes that are incentivized by distinct PES program activities. Furthermore, a lack of net change in forest cover does not necessarily mean a lack of change in the landscape, since it is possible to swap between forested and deforested states without an overall change in land use proportions (Pontius Jr and Millones, 2011). Evaluations of PES programs for forest conservation outcomes should thus ideally be based on spatially explicit tracking of the distinct land use transitions linked to both deforestation and regeneration processes.

A second key component of PES-mediated forest conservation evaluations is the demonstration of additionality. This requires a counterfactual approach capable of inferring causality to policy interventions, particularly where randomized treatments are not possible (Engel et al., 2008; Ferraro, 2009; Jones and Lewis, 2015; Wunder, 2015; Le Velly and Dutilly, 2016). Despite this importance, remarkably few PES impact evaluations incorporate counterfactual approaches (Salzman et al., 2018). For example, Calvet-Mir and colleagues (2015) recently reported that only seven of 26 peer-reviewed evaluations of the effectiveness of 24 different PES projects incorporated some kind of counterfactual approach. These studies often employ matching methods that use strategic subsampling from among treated and potential control cases to create treatment and control assignments that are similar in all respects except the intervention of interest (Rubin, 1973; Rosenbaum and Rubin, 1983; Morgan, 2006). A smaller group of studies combine counterfactual approaches with the differences-in-differences method, that allows further comparison before and after the intervention (Honey-Rosés et al., 2011; Arriagada et al., 2012; Clements et al., 2013; Costedoat et al., 2015). The importance of demonstrating additionality is especially crucial given the challenge of distinguishing PES-mediated outcomes from outcomes generated by broader socio-economic-ecological trends (Velly and Dutilly 2016). For example, between 1962 and 1981, increased forest regeneration near São Paulo, Brazil was driven by both decreased demand for charcoal and by increased land abandonment. Similarly, deforestation between 1981 and 2000 was driven both by road infrastructure improvement and by expansion in agriculture demands (Teixeira et al., 2009). The distinction between the effect of these broader socio-economic processes and the policy intervention itself on forest conservation outcomes requires a counterfactual approach.

In this study, we use a spatially explicit and counterfactual approach to evaluate the contribution of PES programs to the regeneration and avoided loss of native vegetation near São Paulo, Brazil. We define forest conservation outcomes in three distinct ways, as: i) net forest change for each property, ii) the percentage of each property under regeneration, and iii) vegetation loss rate. To our knowledge, this is the first evaluation of the conservation impact of PES in South America to explicitly detect vegetation loss and regeneration with a counterfactual approach, as well as the first quantitative assessment of PES program effectiveness within Brazil's Atlantic Forest, despite more than 20 programs operating in the region over the past two decades (Guedes and Seehusen, 2011; Calvet-Mir et al., 2015; Grima et al., 2016).

1.2.1 Study region

The study region is located within the threatened Atlantic Forest, which hosts one of the world's most diverse tropical forest biota (Mittermeier, 2004). Following over 500 years of European colonial history, the Atlantic Forest's estimated original 1,500,000 km2 extent has been radically transformed, with less than 16% of forest remaining, much of which is relegated to small remnants of secondary growth forest (Ribeiro et al., 2009). Given that only approximately 1% of the Atlantic Forest is formally protected (Ribeiro et al. 2009), programs like PES that incentivize conservation actions on private lands are a critical component of regional conservation strategies.

1.2.2 PES Programs

We assessed the effectiveness of two watershed PES programs, Produtor de Água (PA), in the municipality of Joanópolis (São Paulo state), and Conservador das Águas (CA) in the municipality of Extrema (Minas Gerais state), which are among the most well-established PES programs in the Brazilian Atlantic Forest (Fig. 1.1) (Guedes and Seehusen 2011). The PA program was launched in 2006 with most contracts established between 2011 and 2013 (Viani and Bracale, 2015) and the CA program was launched in 2005 with most contracts established after 2007 (Richards et al., 2015).

Both programs share the goal of improving water quality and quantity in the Cantareira water supply system through soil and forest conservation activities (Table 1.1; (Zanella et al., 2014;

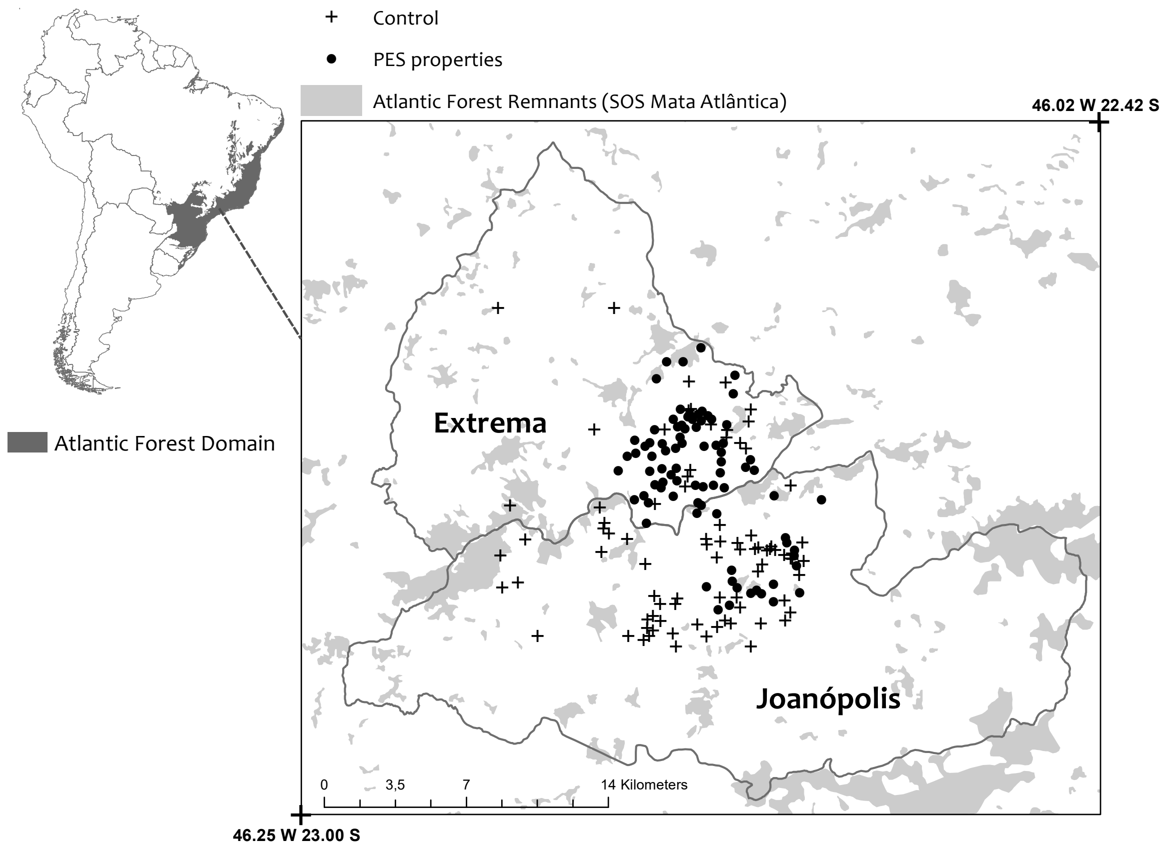

Figure 1.1: Historical extent of Atlantic Forest domain, and study region, encompassing implementation areas for the *Conservador das Águas* (municipality of Extrema, Minas Gerais State) and *Produtor de Água* (municipality of Joanópolis, São Paulo State) PES programs. Grey areas in the study region represent remaining Atlantic Forest fragments. Dots (control) and crosses (treatment) represent the centroids of studied properties.

Richards et al., 2015; de Água, 2012)). The two programs are located within the Cantareira water supply system (see Table A1, Supplementary Material) in adjacent municipalities (Joanópolis, 37,429 ha and Extrema, 24,457 ha), that have land-uses dominated by pasture, eucalyptus plantations and forest remnants (Whately and Cunha, 2007). The Cantareira system is one of the world's largest public water systems, servicing over 5.4-8.8 million people in the São Paulo Metropolitan Region (Dias, 2016). The principal activities incentivized by both programs include the maintenance of forested areas, as well as active forest restoration (Richards et al., 2015; Viani and Bracale, 2015)(Table 1.1). The PA and CA programs have two main differences. Regarding their execution structure, PA is operated by the São Paulo State Government in partnership with The Nature Conservancy and CA is operated by the Extrema municipal government. Regarding program rules, payments are calculated based on total property area in the CA program and based only on contracted area in the PA program.

Table 1.1: Structure of the two studied PES programs.

	Conservador das Águas (CA)	Produtor de Água (PA)
Program area (municipality and state)	Extrema (Minas Gerais state)	Joanópolis and Nazaré Paulista (São Paulo State)
Launch year	2005	2006
Period under operation	2007 to present	2009 - 2015
Executers	Municipal government	State Government and The Nature Conservancy
Eligibility	Rural properties $\geq$ 2ha, inside priority basins Land title documents current	Properties in the rural area and inside priority basins Land title documents current
Number of properties	74	26
Payment to landowner	Based on opportunity costs, considering whole property area	Based on opportunity costs, considering contracted area
Incentivized management practices	Soil conservation Forest conservation Forest regeneration Improvement of waste disposal Register of private reserves	Soil conservation Forest conservation Forest regeneration

1.2.3 Data collection

We considered all properties that enrolled in the PA or CA programs between 2007 and 2013, and that had valid contracts through December 2014 (PA) or February 2015 (CA) as treatment properties. We considered the date of PES contract establishment as the 'start' of a given PES program. We delimited control properties from a subset of properties registered in the Environmental Rural Registry (in Portuguese, "Cadastro Ambiental Rural" – CAR(MMA 2016) that geographically overlapped with the PA or CA program area. Property boundary information was obtained through government or PES sources (Table A1,Supplementary Material). As properties smaller than 2 hectares are not eligible for the CA program, we limited all control properties from the Extrema municipality to an area $\geq$ 2 ha.

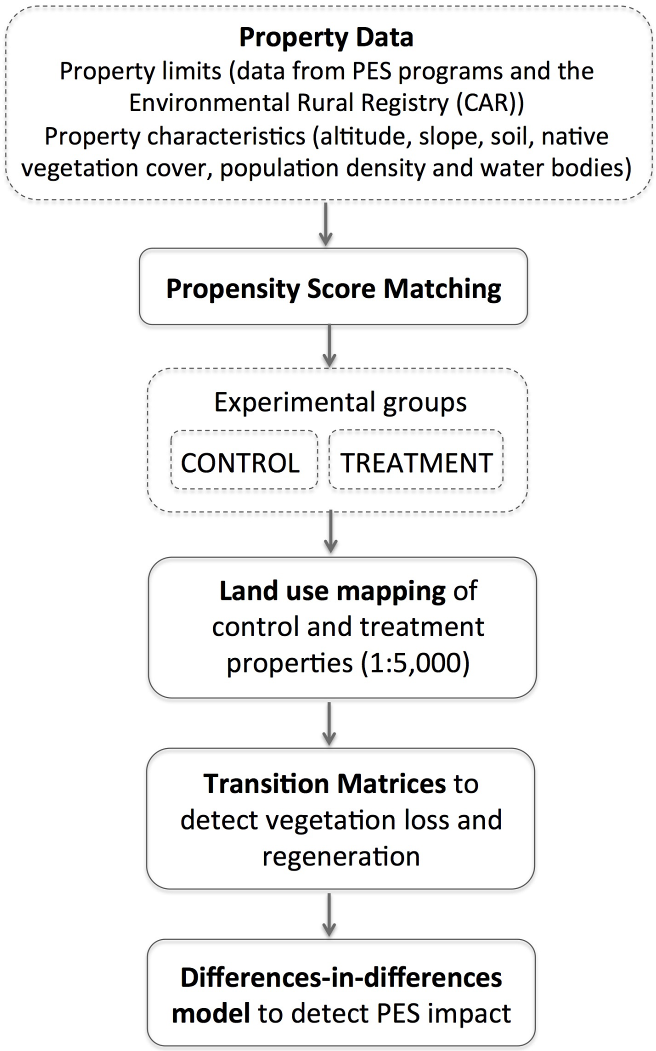

Figure 1.2: Main methodological steps used to detect PES impact on vegetation loss and regeneration. Dashed line boxes refer to data while continuous line boxes refer to methods.

Matching control and treatment areas

In order to arrive at a set of control properties, we controlled for a set of covariates potentially related to both the probability of property enrollment in a PES program, and to conservation outcomes. PES program technical officers report that they actively offered PES enrollment to properties with forest remnants and freshwater springs (Maria Fernanda Gioni - TNC, personal communication). Thus, we included the number of freshwater springs and native forest cover extent at the implementation period as covariates. We also included property size, altitude, slope and soil type as they influence opportunity costs related to agricultural suitability. (Fig. A2; Supplementary Material).

We estimated propensity scores (the estimated probability of the treatment as a function of variables that predict treatment assignment; (Morgan, 2006), using an additive logistic regression model (Rosenbaum and Rubin, 1983; Morgan, 2006). We then matched on propensity scores using the nearest-neighbor method, without replacement, and with a ratio equal to 1 (i.e. matching each treated property to one control property), after testing for both optimal and Mahalanobis matching methods (Rosenbaum and Rubin, 1983, 1985; Imbens, 2015). Based on an assessment of covariate balance quality, we defined a caliper width (a distance which is acceptable for any match) equal to 0.2 of the standard deviation of the propensity score (Fig. A1, Supplementary material).

We started with 492 properties, divided into n = 100 treatment (n = 74 from CA and n = 26 from PA) and n = 392 control. The use of a 0.2 caliper resulted in 83 treated properties successfully matched with control properties, effectively balancing both treatment and controls for the observed covariates (Table A2, Supplementary Material) and substantially improving covariate balance (Fig. S1, Supplementary material).

Transition matrix of land use change

We used a transition matrix approach to calculate pixel-level (2 x 2 m) conservation gains through vegetation regeneration or avoided loss on both treatment and control properties. Transition matrices provide simple and spatially-explicit estimates of land use conversion, considering both loss (initial vegetation loss and deforestation) and gain (initial vegetation and forest regeneration) processes for each unit. This permits a quantification of the probability of change or permanence of each transition, and thus an overall description of landscape dynamics (Lambin et al., 2003).

We considered the change in land use before and after the PES intervention, using the differences-in-differences method. We defined the moment before PES implementation as T- (i.e., the first observation in time), the moment of PES implementation as T0, and the moment after PES intervention as T+. The year T- was 2003 for all properties, but T0 and T+ varied from property to property. Consequently, for T0, we used images obtained closest to the contract year for each property. T0 varied from 2007 - 2012, and T+ varied from 2014 - 2015.

We classified land use and land cover at each time period (T-, T0 and T+) for each property (treatment and control) at a 1:5,000 scale using Google Earth online available images. All Google Earth images were exported and georeferenced based on a 2007 SPOT image provided by the São Paulo Secretariat for the Environment (source: Instituto Geográfico Cartográfico – IGC, São Paulo). Landscapes were classified into four units: initial vegetation (shrub vegetation, typical of initial forest succession, and active restoration), forest (intermediate to advanced second growth forest, $>$ 15–20 years old), eucalyptus plantation and other agricultural uses (primarily cattle pasture). We used these four units to compare landscape changes between the period before PES implementation ($Tbefore$ = T- to T0) and after implementation ($Tafter$ = T0 to T+) using transition matrices for each property, calculated at a 4 m2 resolution.

1.2.4 Data analysis

To detect effect of PES programs on forest cover without separating the process of regeneration and vegetation loss, we first investigated net forest change. We calculated the annual net forest change (NFC) as the simple difference in the percentage of forest per total property area before and after the PES intervention:

$$NFC = \frac{\frac{F2-F1}{A}}{Y} * 100 \tag{1.1}$$

where $F2$ is forest (ha) in a property at the end of an observation period ($Tbefore$ or $Tafter$) and $F1$ is forest (ha) at the beginning of the period (T- or T0, respectively), A is the total property area and Y the number of years over which changes were evaluated. The annual net forest change was calculated for all observed properties.

We also calculated the annual rate of net forest change (RFC) based on the formula proposed by Puyravaud (2003):

$$RFC = \frac{\log F2 - \log F1}{Y} * 100 \tag{1.2}$$

where $F2$ is forest (ha) in a property at the end of an observation period ($Tbefore$ or $Tafter$) and $F1$ is forest (ha) at the beginning of the period (T- or T0) and Y the number of years over which changes were evaluated, considering only properties that had native forest at T- and T0.

To detect effects of PES on regeneration and avoided loss separately, we considered two main types of transition. Regeneration represents both transitions of agricultural land to initial vegetation (initial vegetation regeneration) and transitions of initial vegetation to forest (forest regeneration). We did not observe transitions of agricultural land to forest, given that $Tbefore$ or $Tafter$ are too brief (7 and 5 years) for forest development and establishment. $Regeneration$ for both initial vegetation and forest were calculated as a percentage of the total property area:

$$Regeneration = \frac{\frac{R}{A}}{Y} * 100 \tag{1.3}$$

where R is the farm area (ha) under regeneration (initial vegetation or forest) in the period ($Tbefore$ or $Tafter$), A is the total property area, and Y is the number of years over the period ($Tbefore$ or $Tafter$). We calculated the percentage of initial vegetation regeneration and forest regeneration for all observed properties.

Vegetation loss is both transitions of initial vegetation to agricultural land use (initial vegetation loss) and transitions of forest to any other unit (deforestation). We calculated vegetation $Loss$, for both initial vegetation and forest, as a percentage of the total initial vegetation in the property:

$$Loss = \frac{\frac{V_L}{V_I}}{Y} * 100 \tag{1.4}$$

where V_L is the lost quantity (hectares) of vegetation (initial vegetation or forest) in the period ($Tbefore$ or $Tafter$), V_I is the initial quantity of vegetation (initial vegetation or forest) in the property (at T- and T0), and Y is the number of years over which the loss is observed, considering only properties that had native vegetation (initial vegetation or forest) at T- and T0.

Finally, we calculated total vegetation increase rate considering all treatment properties together, in order to evaluate the rate of vegetation gain across the PES implementation area. This total forest increase rate (TFR) was calculated also based on the annual rate of change (Puyravaud 2003), but considered both initial vegetation and forest regeneration together:

$$TRF = \frac{\log(I_2 + F_2) - \log(I_1 + F_1)}{Y} * 100 \tag{1.5}$$

where I2 and F2 are total initial vegetation and forest (ha) respectively in the PES implementation area at the end of an observation period (Tbefore or Tafter), I1 and F1 are total initial vegetation and forest at the beginning of the period (T- or T0), and Y is the number of years over which changes were evaluated. We used this total increase rate to estimate the potential conservation outputs of these two regional PES programs to the entire study region. To do this we assumed that: i) a conservative forest restoration goal would be 33% of the study region under forest cover in order to maintain biological integrity similar to conserved forest areas (Banks-Leite et al. 2014), ii) PES is the only conservation intervention applied to obtain forest regeneration targets iii) all rural properties could be enrolled without attrition, iv) no change in deforestation trends will occur, and v) these PES programs will maintain the observed rate of program result (total forest increase rate) for the first 20 years, and then sustain a 1% increase per year.

We modeled net forest change (Eq.1 and 2), percentage of regeneration (Eq. 3) and rate of vegetation loss (Eq. 4) as a function of PES enrollment (treatment or control), time (before or after PES) and their interaction (treatment : time). This permitted us to distinguish between cases when forest conservation outcomes occurred independently of PES enrollment (e.g. if control and treatment properties demonstrated conservation gains over time), from cases when PES enrollment was associated with additional forest conservation gains (e.g. if conservation gains differed between treated and control properties after PES implementation, but not before). We used generalized linear mixed models (incorporating property ID as a random effect) and appropriate errors (Gaussian distributed errors for net forest change – percentage and rate, and gamma for all other models). We estimated average treatment effect on the treated (ATT) from the difference in means of the fitted values of the proposed models. All analyses were conducted in R version 3.3.1. (R Core Team, 2016), using the Matchit package for propensity score matching and balancing (Ho et al., 2011), and the glmmADMB package for mixed models (Bates et al., 2014).

1.3 Results

We found that PES enrollment resulted in an increase in net forest change, measured as either the percentage of forest per total property area (Eq. 1) or the rate of forest change per property (Eq. 2), resulting in net forest gain among PES farms (Fig. 1.3; Table 1.2). Average annual net forest change on enrolled farms increased from 0.43% (± 0.80) per farm before implementation to 0.98% (± 1.81) after implementation. Over approximately five years of program implementation, this resulted in an additional 2.8 - 5.6% of total farm area under Atlantic Forest coverage. In contrast, average annual net forest change among control properties declined following implementation (before: 0.38% ± 1.25; after 0.01% ± 1.21). Similarly, the annual rate of net forest change on enrolled properties increased following implementation (before: 4.00% ± 9.41; after 6.68% ± 13.63), while the average annual rate of net forest change among control properties declined (before: 3.52% ± 7.65; after - 4.98% ± 39.42). Both the percentage and rate of net forest cover change declined marginally over time (Table 1.2).

We found essentially a significant influence of PES on forest regeneration (Fig. 1.4; Table 1.2). The average proportion of farm area under forest regeneration increased after implementation for enrolled farms (before: 0.43% ± 0.74; after 0.80% ± 0.50), yet declined for control farms (before: 0.47% ± 0.96; after 0.15% ± 1.50). Regeneration of initial vegetation increased among all properties over time (Table 1.2). We also detected a (non-significant) trend in PES enrolled farms having a higher increase in average proportion of farm area under initial vegetation regeneration relatively to control farms (treatment before: 0.61% ± 1.03; after 1.20% ± 1.42; control before: 0.34% ± 0.56; after 0.74% ± 1.78).

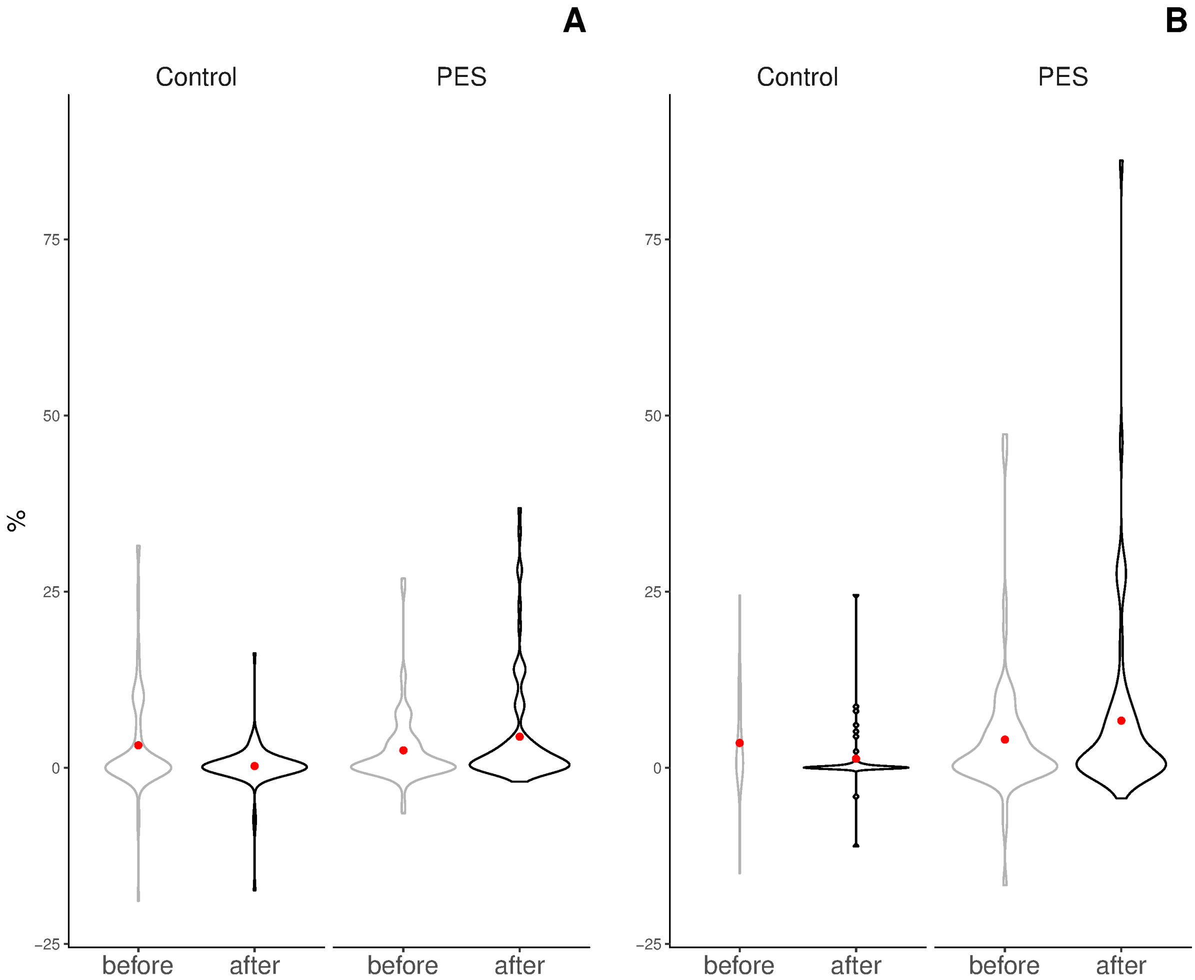

Figure 1.3: Violin plots of annual net forest change (A) and the annual rate of net forest change (B) for PES and control properties, before and after PES implementation. Red points indicate mean values.

PES enrollment was secondarily associated with a (non-significant) trend toward decreased vegetation loss (Fig. 1.4; Table 1.2). While average forest loss was similar among enrolled and control properties before PES implementation (enrolled: 1.26% year^{-1} ± 3.00; control: 1.27% year^{-1} ± 3.03), forest loss on enrolled properties fell after implementation (0.39% year^{-1} ± 1.17), and remained constant on control properties (1.21% year^{-1} ± 4.28). The average loss of initial vegetation significantly declined for both enrolled (1.50% to 1.27%) and control (1.40% to 0.59%) properties over time (Table 1.2).

Finally, considering all properties together, the total rate of increase of native vegetation

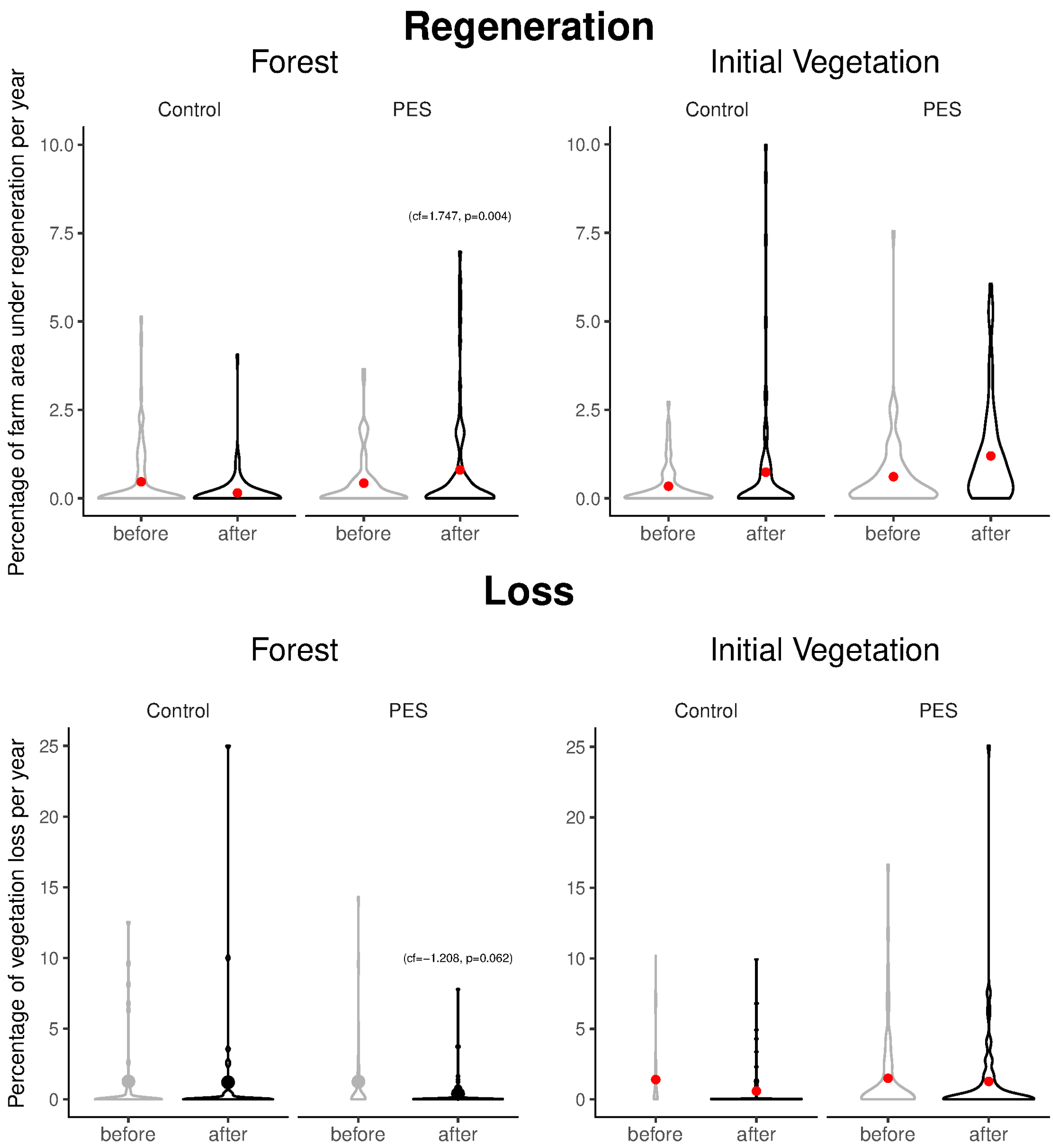

Figure 1.4: Violin plots for vegetation regeneration and loss, separately for forest and initial vegetation, PES and control properties, for times before and after PES implementation. Red points indicate mean values.

(including both forest and initial vegetation) in the PES contracted area increased following PES implementation (before: 1.3% year^{-1}; after: 3.9% year^{-1}).

Following approximately 5 years of program implementation, the two CA and PA programs together contributed to an additional 37.4 hectares under native vegetation regeneration across all enrolled farms. Considering our initial assumptions (see methods), to achieve the target of 33% of native forest cover in these municipalities using only these two programs would require approximately 180 years.

Table 1.2: Model estimates coefficients for all conservation outcomes: net forest change, rate of net forest change, and percentage of vegetation regeneration and loss, for both forest and initial vegetation. Values between parentheses are standard errors. ATT denotes average treatment effect on the treated.

Outcome	PES	Time	PES:Time	ATT
Net forest change	0.057	-0.364+	0.906**	0.91
	(0.20)	(0.20)	(0.28)	
Rate of net forest change	0.487	-8.506+	11.190*	11.19
	(3.98)	(4.45)	(5.63)	
Forest regeneration	-0.084	-1.120**	1.747**	0.69
	(0.43)	(0.43)	(0.61)	
Initial vegetation regeneration	0.580	0.768**	-0.095	0.19
	(0.36)	(0.36)	(0.50)	
Forest loss	0.931	0.043	-1.208+	-0.62
	(0.68)	(0.51)	(0.65)	
Initial vegetation loss	0.544	-0.918*	0.722	0.35
	(0.51)	(0.39)	(0.52)	

1.4 Discussion

We demonstrate that PES has a positive effect on forest cover, through a counterfactual assessment of the effectiveness of PES programs for vegetation loss and regeneration outcomes (Ferraro 2009, Velly and Dutilly 2016). PES enrollment increased both the percent of forest area per farm, and the rate of increase of forest per farm, driven by forest regeneration. We estimate that PES enrollment in the study region would result in approximately 2.8-5.6% of total farm area in additional forest cover over a period of five years.

Positive influences of PES on net forest cover have been previously reported by studies observing farm-level PES effects (Honey-Rosés et al. 2011, Arriagada et al. 2012, Costedoat et al. 2015). For example, Arriagada et al. (2012) observed an increase of net forest cover from 11 to 17% of the mean area enrolled in Costa Rica's national PES program (Pagos por Servicios Ambientales) over an eight years period. Similarly, an evaluation of ejidos (i.e. community private lands) under PES contracts in Mexico estimated an additional increase in net forest cover between 12 and 14.7% of forest over a five-year period (Costedoat et al. 2015). However, the evaluation of net forest cover may confuse two different underlying processes that may drive similar observed net forest changes - avoided deforestation and forest regeneration. A separate assessment of the influence of incentivized PES activities on these distinct processes is particularly important because they accrue over distinct timescales, from as short as days (deforestation) to multiple years (regeneration).

We detected a positive effect of PES on forest regeneration, totaling an additional average 0.69% of farm area per year covered with forest for enrolled properties. This result is encouraging, given that participants in both programs were required to retire a portion of their farms for either natural regeneration or active restoration. However, we did not detect a significant influence of PES on initial vegetation regeneration, which was unexpected, given that this regeneration occurs relatively rapidly. We attribute some of this null effect to an artifact of the temporal availability of the mapping products used to classify the T0 period. Some initial vegetation regeneration following PES enrollment may have been included in the "before" period, rather than the "after" period, reducing the overall observed influence of PES. We expect this effect to have occurred only for initial vegetation and not forest regeneration, as initial vegetation recruitment occurs over a shorter timespan.

While we found that PES strongly incentivized new forest regeneration, it showed a non-significant trend on deforestation avoidance. Deforestation rates across all farms (both control and treatment) both before and after PES implementation (Fig. 3) were remarkably low. This is relatively unsurprising given the low deforestation rates in the Cantareira region . Other studies in regions with low regional deforestation rates have also reported minimal, measurable influences of PES intervention on avoided deforestation (Sanchez-Azofeifa et al. 2007, Robalino and Pfaff 2013).

We know of only two other studies designed to allow separate assessments of the influence of PES on forest loss and forest regeneration outcomes. In the San Juan – La Selva Biological Corridor, Morse and colleagues (2009) reported that PES reduced the rate of deforestation (from 1.43% year-1 to 0.01% year-1) while the rate of net gain of total forest cover was similar (0.5% year-1 before PES and 0.6% year-1 after PES). In the Osa Peninsula, Costa Rica, Sierra and Russman (2006) did not explicitly evaluate distinct deforestation and forest regeneration outcomes, but did observe different land uses inside farm area. They found that total forest cover was not different among PES enrolled and non-enrolled farms, suggesting that PES had no effect on pre-existing forest area. However, Daniels et al (2010) pointed out that data from both studies similarly show that PES increased forest regeneration through charral regeneration, even total forest cover was not significantly different. This shows the importance of observing these outcomes separately. Along the San Juan – La Selva Biological Corridor, PES implementation was associated with nearly twice more charral (early secondary forest) transitioning to forest classes than returning to pasture (Morse et al. 2009, Daniels et al. 2010). In the Osa Peninsula, PES-enrolled farms supporting more than four times as much as forest (charral) compared to non-PES farms (11.2 vs. 2.5% respectively) (Sierra and Russman 2006). The authors further suggest that the strong PES-mediated forest regeneration seen across Costa Rica may be associated with broader socio-economic trends. The well-documented transition from an agrarian to a service-based economy in Costa Rica lead to widespread agricultural abandonment (Sierra and Russman 2006, Daniels et al. 2010), and the financial incentives associated with PES implementation may have contributed positively to landowner decisions to abandon agricultural activities, leading to enhanced forest regeneration (Sierra and Russman 2006, Daniels et al. 2010).

We posit that a similar synergy between PES program implementation and broader socio-economic context may occur in our study region; specifically a synergy between PES, environmental regulation, and regional abandonment of the eucalyptus industry, resulting in native vegetation cover gains. In recent decades, government agencies and civil society have increased pressure on farmers and agricultural companies to restore native vegetation in compliance with the Brazilian National Forest Code (law 12.651/2012) (Richards et al. 2015, Brancalion et al. 2016). For small to medium sized farms (e.g. the majority of properties in our study), compliance with environmental regulation is hindered by both the opportunity costs of removing land from agricultural activities, and the high costs of active forest restoration (Brancalion et al. 2012). The financial incentives offered by PES activities likely enabled some property owners to comply with existing environmental regulation (Gonçalves 2013, Richards et al. 2015). The combination of revenue generated by PES and existing regulation, operating together in tandem, may be an important pre condition for PES success (Salzman et al. 2018). Additionally, while the majority of the Cantareira region remains primarily agricultural (Richards et al. 2015), several property owners report a regional slow-down in eucalyptus production. We observed that the majority of farm area undergoing recruitment to initial vegetation regeneration came principally from previous areas of eucalyptus cultivation (Figs. A2 and A3, Supplementary Material). This further supports the idea that PES success for vegetation regeneration outcomes may be partially due to the broader context of declining opportunity cost.

Our results shed light on new ways of considering additionality in PES programs that should be taken into account during PES design and implementation phases. We suggest an expansion of the concept of additionality to include two critical components: legal and temporal. Legal additionality could be defined as additional conservation results above and beyond the existing and enforceable legal mandates that are already expected to produce given conservation outcomes. For example, if PES in the Cantareira study region primarily acts as a cost-effective mechanism for property owners to get into compliance with existing Forest Code legislation, conservation gains through PES may provide relatively little legal additionality. Temporal additionality (also called the permanence problem) focuses on the longevity of conservation outcomes (Pagiola et al. 2016). For example, PES may contribute to additional conservation outcomes during the course of program enrollment, but that are eventually lost when a property declines to renew a contract, or a PES program closes. Such temporal effects may further interact with the natural timeline of forest regeneration. For example, while mature Brazilian Atlantic Forest is protected by law (Atlantic Forest law 11.428/2006), younger Brazilian Atlantic Forest still within the initial phases of regeneration is not protected, and may be legally cut down when a given PES contract finishes. Consequently, the most robust forest conservation outcomes will come from PES programs with contracts sufficiently lengthy to protect initial vegetation regeneration until it reaches a legally protected mature forest status (after ca. 8-15 years for the Atlantic forest).

While we found that PES has a positive effect on conservation outcomes, we also point that this conservation gains come slowly. We estimated that Produtor de Água and Conservador das Águas programs would require approximately 180 years to reach a minimally adequate amount of regional forest cover to support biological integrity (33%; Banks-Leite et al, 2014) across the implementation area. This estimate is conservative, both because we likely underestimated the influence of PES enrollment on regeneration (as some regeneration may have been accounted for the "before" period), but also because this estimate considers a fairly optimistic PES enrollment scenario, with no loss of contracted properties and a sustained rate of forest increase for the first 20 years of program execution. Thus, the PES mechanism can contribute to conservation gains on private properties but will require additional and complementary conservation interventions to achieve robust conservation gains.

1.5 Conclusion

We found that while PES has a significant additional positive impact on forest cover in the Atlantic Forest of Brazil, the overall conservation impact is relatively low and conservation gains may be considered vulnerable. Over a five-year period, we observed PES-driven forest gain through tropical forest regeneration in the Cantareira region. Despite these gains, these positive impacts of PES may be vulnerable to both a lack of legal and or temporal additionality. In either case, any reduction of the observed conservation gains may be problematic, given the low overall observed impacts of PES. While PES do create conservation gains, we caution against the sole reliance on PES to reach native vegetation cover targets, and stress that PES effectiveness should only be considered as complementary strategy to other conservation policies.

1.6 Acknowledgments

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Bibliography

Arriagada, R. A., Ferraro, P. J., Sills, E., Pattanayak, S. K., Cordero-Sancho, S., 2012. Do Payments for Environmental Services Affect Forest Cover? A Farm-Level Evaluation from Costa Rica. Land Economics 88 (2), 382–399.

Bates, D., Máchler, M., Bolker, B., Walker, S., 2014. Fitting linear mixed-effects models using lme4.

Calvet-Mir, L., Corbera, E., Martin, A., Fisher, J., Gross-Camp, N., 2015. Payments for ecosystem services in the tropics: a closer look at effectiveness and equity. Current Opinion in Environmental Sustainability 14 (C), 150–162.

Clements, T., Rainey, H., An, D., Rours, V., Tan, S., Thong, S., Sutherland, W. J., Milner-Gulland, E. J., 2013. An evaluation of the effectiveness of a direct payment for biodiversity conservation: The Bird Nest Protection Program in the Northern Plains of Cambodia. Biological Conservation 157 (C), 50–59.

Costedoat, S., Corbera, E., Ezzine-de Blas, D., Honey-Rosés, J., Baylis, K., Castillo-Santiago, M. A., 2015. How Effective Are Biodiversity Conservation Payments in Mexico? PLoS ONE 10 (3), e0119881–20.

de Água, A. P. P., 2012. Manual operativo do produtor de água. Brasília: Agência Nacional das Águas.

Dias, N., 2016. O Sistema Cantareira e a crise da Água em São Paulo falta de transparência, um problema que persiste. Technical report.

Engel, S., Pagiola, S., Wunder, S., 2008. Designing payments for environmental services in theory and practice: An overview of the issues. Ecological Economics 65 (4), 663–674.

Ezzine-de Blas, D., Wunder, S., Ruiz-Pérez, M., Moreno-Sanchez, R. d. P., 2016. Global Patterns in the Implementation of Payments for Environmental Services. PLoS ONE 11 (3), e0149847–16.

Ferraro, P. J., 2009. Counterfactual thinking and impact evaluation in environmental policy. New Directions for Evaluation 2009 (122), 75–84.

Ferraro, P. J., Pattanayak, S. K., 2006. Money for Nothing? A Call for Empirical Evaluation of Biodiversity Conservation Investments. PLoS Biology 4 (4), e105.

Goh, T. Y., Yanosky, A., 2016. Payment for Ecosystem Services works, but not exactly in the way it was designed. Global Ecology and Conservation 5, 71–87.

Grima, N., Singh, S. J., Smetschka, B., Ringhofer, L., 2016. Payment for Ecosystem Services (PES) in Latin America: Analysing the performance of 40 case studies. Ecosystem Services 17 (C), 24–32.

Guedes, F. B., Seehusen, S. E., 2011. Pagamentos por serviços ambientais na mata atlântica: lições aprendidas e desafios. Brasília: MMA 272.

Ho, D. E., Imai, K., King, G., Stuart, E. A., 2011. MatchIt: Nonparametric Preprocessing for Parametric Causal Inference. Journal of Statistical Software 42 (8), 1–28.

Honey-Rosés, J., Baylis, K., Ramírez, M. I., 2011. A Spatially Explicit Estimate of Avoided Forest Loss. Conservation Biology 25 (5), 1032–1043.

Imbens, G. W., 2015. Matching Methods in Practice. Journal of Human Resources 50 (2), 373–419.

Jayachandran, S., de Laat, J., Lambin, E. F., Stanton, C. Y., Audy, R., Thomas, N. E., Jul. 2017. Cash for carbon: A randomized trial of payments for ecosystem services to reduce deforestation. Science 357 (6348), 267–273.

Jones, K. W., Lewis, D. J., 2015. Estimating the Counterfactual Impact of Conservation Programs on Land Cover Outcomes: The Role of Matching and Panel Regression Techniques. PLoS ONE 10 (10), e0141380–22.

Lambin, E. F., Geist, H. J., Lepers, E., 2003. Dynamics of land-use and land-cover change in tropical regions. Annual review of environment and resources 28 (1), 205–241.

Le Velly, G., Dutilly, C., 2016. Evaluating Payments for Environmental Services: Methodological Challenges. PLoS ONE 11 (2), e0149374–20.

Mittermeier, R. A., 2004. Hotspots revisited. Cemex.

Morgan, S. L., 2006. Matching Estimators of Causal Effects: Prospects and Pitfalls in Theory and Practice. Sociological Methods & Research 35 (1), 3–60.

Muradian, R., Corbera, E., Pascual, U., Kosoy, N., May, P. H., 2010. Reconciling theory and practice: An alternative conceptual framework for understanding payments for environmental services. Ecological Economics 69 (6), 1202–1208.

Naeem, S., Ingram, J. C., Varga, A., Agardy, T., Barten, P., Bennett, G., Bloomgarden, E., Bremer, L. L., Burkill, P., Cattau, M., Ching, C., Colby, M., Cook, D. C., Costanza, R., DeClerck, F., Freund, C., Gartner, T., Goldman-Benner, R., Gunderson, J., Jarrett, D., Kinzig, A. P., Kiss, A., Koontz, A., Kumar, P., Lasky, J. R., Masozera, M., Meyers, D., Milano, F., Naughton-Treves, L., Nichols, E., Olander, L., Olmsted, P., Perge, E., Perrings, C., Polasky, S., Potent, J., Prager, C., Quetier, F., Redford, K., Saterson, K., Thoumi, G., Vargas, M. T., Vickerman, S., Weisser, W., Wilkie, D., Wunder, S., Mar. 2015. Get the science right when paying for nature's services. Science 347 (6227), 1206–1207.

Pattanayak, S. K., Wunder, S., Ferraro, P. J., 2010. Show Me the Money: Do Payments Supply Environmental Services in Developing Countries? Review of Environmental Economics and Policy 4 (2), 254–274.

Pontius Jr, R. G., Millones, M., 2011. Death to Kappa: birth of quantity disagreement and allocation disagreement for accuracy assessment. International Journal of Remote Sensing 32 (15), 4407–4429.

Puyravaud, J.-P., 2003. Standardizing the calculation of the annual rate of deforestation. Forest Ecology and Management 177 (1-3), 593–596.

R Core Team, 2016. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.
URL https://www.R-project.org/

Ribeiro, M. C., Metzger, J. P., Martensen, A. C., Ponzoni, F. J., Hirota, M. M., 2009. The Brazilian Atlantic Forest: How much is left, and how is the remaining forest distributed? Implications for conservation. Biological Conservation 142 (6), 1141–1153.

Richards, R. C., Rerolle, J., Aronson, J., Pereira, P. H., Gonçalves, H., Brancalion, P. H. S., 2015. Governing a pioneer program on payment for watershed services. Stakeholder involvement, legal frameworks and early lessons from the Atlantic Forest of Brazil. Ecosystem Services 16 (C), 23–32.

Robalino, J., Pfaff, A., 2013. Ecopayments and Deforestation in Costa Rica: A Nationwide Analysis of PSA's Initial Years. Land Economics 89 (3), 432–448.

Rosenbaum, P. R., Rubin, D. B., 1983. The central role of the propensity score in observational studies for causal effects. Biometrika 70 (1), 41–55.

Rosenbaum, P. R., Rubin, D. B., 1985. Constructing a Control Group Using Multivariate Matched Sampling Methods That Incorporate the Propensity Score. The American Statistician 39 (1), 33–38.

Rubin, D., 1973. Matching to remove bias in observational studies. Biometrics 29, 159–183.

Salzman, J., Bennett, G., Carroll, N., Goldstein, A., Jenkins, M., 2018. The global status and trends of Payments for Ecosystem Services. Nature Sustainability, 1–9.

Sánchez-Azofeifa, G Arturo, P. A. R. J. A., Boomhower, J. P., 2007. Costa rica's payment for environmental services program: intention, implementation, and impact. Conservation biology 21 (5), 1165–1173.

Schomers, S., Matzdorf, B., 2013. Payments for ecosystem services: A review and comparison of developing and industrialized countries. Ecosystem Services 6 (C), 16–30.

Teixeira, A. M. G., Soares-Filho, B. S., Freitas, S. R., Metzger, J. P., 2009. Modeling landscape dynamics in an Atlantic Rainforest region: Implications for conservation. Forest Ecology and Management 257 (4), 1219–1230.

Viani, R. A. G., Bracale, H., 2015. Produtor de Água no PCJ. Technical report.

Whately, M., Cunha, P., 2007. Cantareira 2006. Um olhar sobre o maior manancial de água da Região Metropolitana de São Paulo. Tech. rep.

Wunder, S., 2015. Revisiting the concept of payments for environmental services. Ecological Economics 117, 234–243.

Yang, W., Liu, W., Viña, A., Luo, J., He, G., Ouyang, Z., Zhang, H., Liu, J., 2013. Performance and prospects of payments for ecosystem services programs: Evidence from China. Journal of Environmental Management 127 (C), 86–95.

Zanella, M. A., Schleyer, C., Speelman, S., 2014. Why do farmers join Payments for Ecosystem Services (PES) schemes? An Assessment of PES water scheme participation in Brazil. Ecological Economics 105 (C), 166–176.

Supplementary material

A.1 Detailed study site information

Similarly to the rest of the Cantareira region, both Extrema and Joanópolis municipalities are occupied mostly by pasture, eucalyptus plantations and forest remnants (Whately and Cunha 2007). They are divided by a mountain range named Serra do Lopo, with altitudes varying from 900 to 1700 meters, and mainly covered by Montane Atlantic Forest (Yamamoto 2009). In 2010, Extrema had 28,599 residents from which 2,576 were rural, while Joanópolis was a less dense municipality with 11,768 inhabitants, with no data for rural residents (IBGE 2012). They have similar Human Development Index (IDH) (0.699 for Joanópolis and 0.732 for Extrema, in 2010) (PNUD) and similar Educational Human Development Index (both were 4.2 in 2009); however Extrema shows a constant increase of this last index over time (INEP 2009). Extrema has a more active local government and a more intense economical activity with a GDP per capita equivalent to U$ 39 in 2015, while Joanópolis has almost three times fewer enterprises, presenting a GDP per capita equivalent to U$ 3.8 (IBGE 2017). As regard to the rural area, Extrema has 10,888 ha (45% of its total area) occupied by 481 land properties, while Joanópolis has 32,730 ha (87% of its total area) occupied by 381 land properties (IBGE 2007).

A.2 Covariate data sources

Mean altitude inside each property was calculated from a digital elevation model based on ASTER images with 30 x 30 m resolution. For type of soil data, we used the soil map for São Paulo state that also includes the municipality of Extrema produced by the Cartographic Geographic Institute of São Paulo (Instituto Geográfico Cartográfico – IGC). We defined soil type for each property as a unique class, choosing the soil class in which the major part of the property area was located. Population density data was obtained from 2000 IBGE census. All studied properties were classified according to population density sectors from IBGE. For percentage of vegetation inside the properties, we used the map produced by IPÊ (Instituto de

Pesquisas Ecológicas) with automatic classification based on the 2007 SPOT Image. It is worth noting that percentages of forest cover for T0 based on IPÊ mapping and our manual mapping differ (Table 2 and Fig.3), probably as a consequence of mapping method and scale. The number of springs inside each property was obtained using the drainage map of the Piracicaba-Capivari-Jundiaí basin, provided by the Brazilian National Water Agency (Agência Nacional de Águas - ANA).

A.3 Propensity score matching choices

In order to achieve a good balance in the propensity score matching, we evaluated different propensity score models and different caliper values. Calipers impose a maximum distance accepted between the treated unit and its possible control units (Stuart 2010), such as control units that fall beyond the caliper distance are not considered for matching. We ran propensity score matching with the nearest neighbor method for 6 different possible models, varying the covariates and, for each model, we varied calipers from 0.05 to 1.00, with a 0.01 interval (Figure S1). For each procedure, we observed the standardized difference in means, i.e. the difference in means for each covariate divided by the standard deviation of the full treated group, which is a common numerical balance diagnostic (Stuart 2010). As suggested by Stuart (2010), we considered that the closer the standardized difference in means (SD Mean diff in Table 2) is to zero, the better the balance and that balance would be acceptable when the absolute standardized difference in means was less then 0.25. At the same time, we observed the number of matched treated, and unmatched treated, according to different models and calipers. As we had a small sample of 100 treated properties, we considered that an adequate balance in our case would be the best balance of the covariates with the less lost of treated samples. We visually selected the propensity score model and the caliper value based on Figure S1. Our propensity score model had PES as a function of land farm total area (property size), altitude, population density, type of soil, percent of native vegetation inside property and the number of water sources inside property. We chose a caliper value of 0.20.

Tabela A.1: Geographic information and its sources for treated and control properties.

Material	Source
Limits of the Produtor de Água (PA) enrolled properties	The Nature Conservancy and the São Paulo Secretariat for the Environment (SMA)
Limits of the Conservador das Águas (CA) enrolled properties	Environmental Secretariat of the Extrema City Hall
Control units group from the São Paulo state	Cadastro Ambiental Rural (CAR) or Environmental Rural Registry obtained from the DataGeo database (datageo.ambiente.sp.gov.br) of the São Paulo Secretariat for the Environment (SMA)
Control units group from the Minas Gerais state	Cadastro Ambiental Rural (CAR) or Environmental Rural Registry provided by the Minas Gerais Environmental State Foundation (FEAM)

Tabela A.2: Differences among PES enrolled and control properties before and after matching across the covariates, i.e. the characteristics related to the probability of a property being assigned to treatment (enrolled into one of the PES programs studied). When good balance is achieved with matching, we expect that differences between PES enrolled and control values to approximate zero. eQQ: mean value of differences in empirical quantile functions; SD Mean diff: standardized mean difference. Value between parenthesis are standard deviations.

Variable	Mean treatment (PES)	Mean control	SD mean diff.*	eQQ mean*
Area(ha)				
Unmatched	34.24 (44.68)	13.38 (20.44)	0.47	0.23
Matched	25.20 (29.92)	22.53 (34.31)	0.06	0.09
Altitude(m)				
Unmatched	1064.87 (113.30)	1006.72 (132.07)	0.51	0.17
Matched	1050.07 (104.73)	1039.84 (119.13)	0.09	0.04
Population density (hab/km2)				
Unmatched	13.26 (5.58)	71.15 (298.82)	-10.37	0.19
Matched	13.46 (5.52)	13.82 (6.58)	-0.06	0.06
Argissolos				
Unmatched	0.87 (0.34)	0.39 (0.49)	1.14	0.24
Matched	0.85 (0.36)	0.85 (0.35)	0	0
Cambissolos				
Unmatched	0 (0)	0.01 (0.12)	-	0.01
Matched	0 (0)	0 (0)	0	0
Latossolos				
Unmatched	0.13 (0.34)	0.59 (0.49)	-1.37	0.23
Matched	0.14 (0.35)	0.14 (0.35)	0	0
Forest cover (percentage)				
Unmatched	18.54 (22.50)	19.86 (20.09)	-0.06	0.06
Matched	19.37 (23.99)	19.19 (18.70)	0.01	0.06
Number of springs				
Unmatched	0.59 (1.05)	0.17 (0.46)	0.40	0.07
Matched	0.33 (0.65)	0.35 (0.71)	-0.02	0

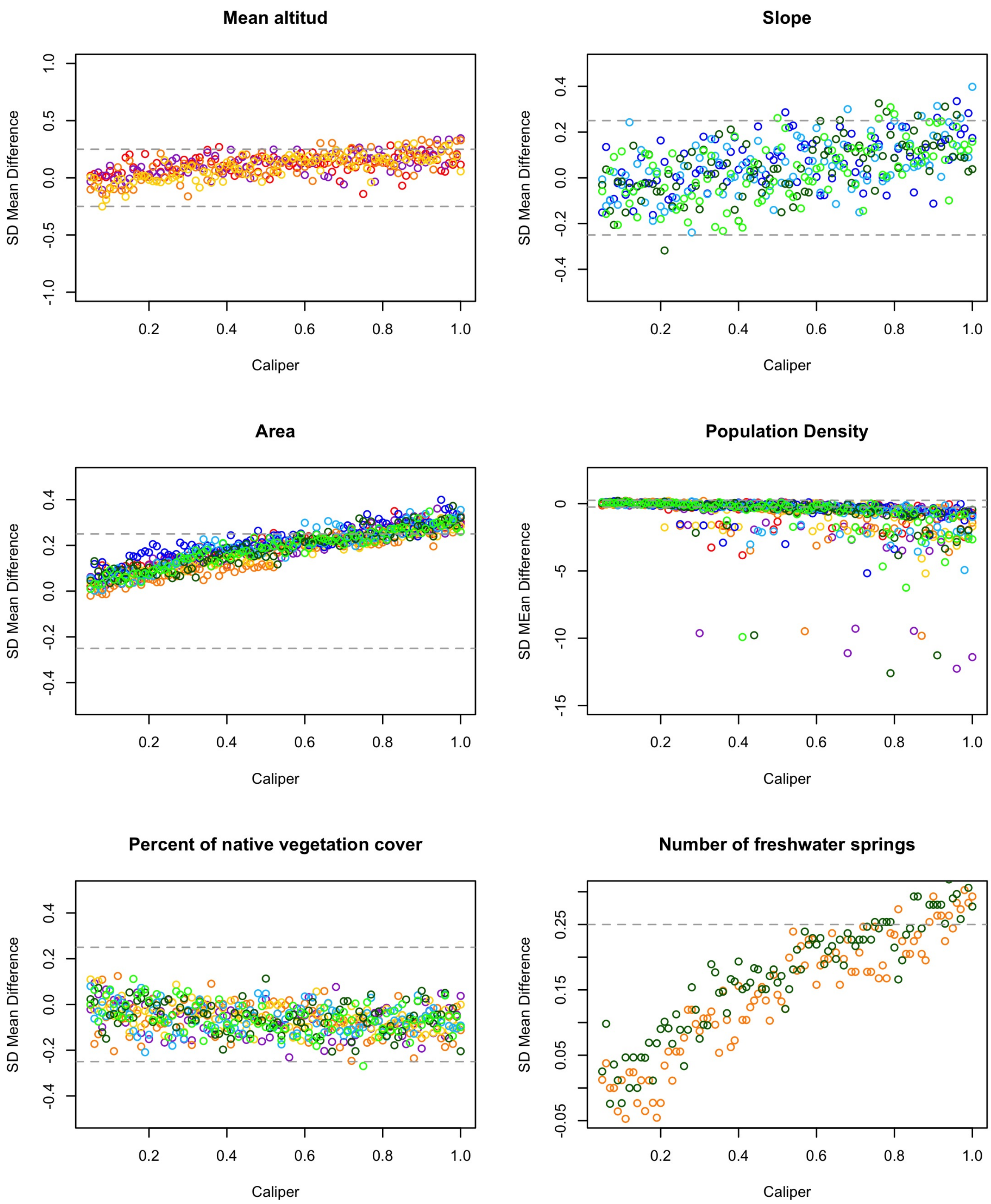

Figura A.1: Standardized difference in means (difference in means between matched treated and control divided by the standard deviation of the full treated group) for each covariate and 6 different propensity scores models, with calipers (calipers define a tolerance level for judging the quality of matches) varying from 0.05 to 1.00, and 0.01 intervals.

Chapter 2

The Brazilian intergovernmental fiscal transfer for conservation: an incentive with limited effects

Patricia G. C. Ruggiero[1], Alex Pfaff[2], Liz Nichols[3], Paula Pereda[4], and Jean Paul Metzger[1]

[1]Department of Ecology, University of São Paulo, BR

[2]Sanford School of Public Policy, Duke University, US

[3]Bioscience Department, Swarthmore College, US

[4]School of Economy, Administration and Accountability, University of São Paulo, BR

2.1 Abstract

The Brazilian Ecological intergovernmental fiscal transfer (ICMS-E) is a conservation incentive intended to stimulate the creation and improved management of protected areas, through the insertion of ecological criteria for redistributing added-value tax revenue to local governments. The ICMS-E was conceived to work as a compensatory mechanism for the opportunity costs imposed by protected areas (PAs). The proportion of local municipal area covered with state and/or municipal PA is used as the main criteria for fiscal transfer redistribution. The ICMS-E currently acts not only as a compensation for existing PAs, but as an incentive for local governments to voluntarily expand PAs, particularly municipal PAs. We combined PA data (e.g. year of creation and proportion occupied in the municipality) and ICMS-E law presence into a longitudinal dataset with 2481 municipalities, from eight states in the Atlantic Forest region, from 1987 to 2016. We estimated the effectiveness of ICMS-E legislation on the creation of new state and municipal protected areas, differentiating the effect on very unrestricted Brazilian Environmental Protected Area (APA) category (close to IUCN category V). Excluding the unrestricted APAs, we found that the percent of the municipality area covered with both state and municipal PAs, increased over time for the first 10 years of the ICMS-E law – an effect strong for state areas (State: 0.031% ± 0.010 ; Municipal: 0.018%, ± 0.005). However, for unrestricted APAs, we found a larger expansion in the municipal, than state PAs (State: 0.160%, ± 0.044; Municipal: 0.351% ± 0.038). Our results suggest that ICMS-E legislation does promote the creation of new PAs, but the rate of creation declines sharply over time, with the majority of new PAs created within the first 10 years. Additionally, in response to ICMS-E intervention, state institutions create varied PAs, while municipal local governments create preferably unrestricted APAs. The distinct effect on state and municipal PAs indicates that ICMS-E mechanism operates through different incentives for state and municipal governments, generating legitimate conservation action in the state level and being vulnerable to opportunistic political motivation in the municipal level.

2.2 Introduction

Intergovernmental fiscal transfers are public finance instruments to assure public goods, being also used to internalize spatial externalities (Ring, 2008). Brazil has implemented an innovative fiscal transfer intended to stimulate the creation and improved management of protected areas (PA), through the redistribution of an added-value tax revenue to local governments based on ecological criteria. The ICMS (in Portuguese *Imposto sobre a Circulação de Mercadorias* *e Serviços*) is a tax on the circulation of goods and services, similar to the "value-added" tax in other countries. It is collected from municipalities by the Brazilian state governments, and then part of these revenues is redistributed to local government, as an intergovernmental fiscal transfer, based on a set of indicators (Ring, 2008; Young, 2005). The Ecological ICMS (hereafter ICMS-E) uses ecological indicators to allocate ICMS revenues among local governments, i.e. municipalities (Loureiro, 2002) (Figure 1). The ICMS-E has been described as a promising mechanism for environmental conservation, to address the fact that people in one location providing ecosystem services can receive resources generated by beneficiaries elsewhere - a similar motivation for payment for ecosystem services mechanism (Farley, et al. 2010, Ring 2008, Young 2005). Beyond Brazil, only Portugal has implemented a similar fiscal transfer for biodiversity conservation (Santos et al., 2012).

As a loss of economic opportunities usually occur where PAs are implemented (Venter et al., 2014), the ICMS-E was conceived to work as a compensatory mechanism that municipalities receive for hosting protected areas (Loureiro 2002, May, et al. 2002). The main criteria for the ICMS-E is the proportion of the municipality with protected areas (PAs) (federal, state or municipal, depending on each state law definition), weighted by PA category. Because the total amount of ICMS-E available is divided by all the municipalities that compete for this resource in a given year (Loureiro, 2002), the amount received by each municipality hosting a PA is also dependent on the total extension of PAs in the state, and thus on the number of municipalities that can benefit from the ICMS-E. This means that the total absolute amount of money available to be redistributed among the participant municipalities varies depending on (i) economic growth that determines the total ICMS collected and (ii) the percentage of the total ICMS assigned to ecological criteria, defined by a state law; while the relative value paid for PA unit area, varies according to (iii) total PA accumulated area in the state and (iv) total number of municipalities hosting those PAs. Therefore, while absolute values received by municipalities per additional area for environmental conservation may increase as economy growth increases, relative values proportionally decreases as new PAs are created in the state.

In Brazil, national PAs are gazetted by federal government institutions, while state government institutions are responsible for the design and implementation of state PAs and the ICMS-E. Because the ICMS-E effectively pays municipalities for the creation of new protected areas, it may act to reduce local resistance by municipal government and local residents to the creation of state PAs. When the criteria for receiving the ICMS-E is also triggered by the presence of municipal protected areas, i.e. municipalities receive for the protected area they create, the ICMS-E can stimulate local government to voluntarily expand its protected area (Sauquet et al., 2014). Thus, the ICMS-E is expected to increase the area under environmental protection either through reduced resistance to the creation of new state protected areas (compensation) or increased local government initiatives for creating new municipal areas (incentive) (Fig. 2.1). However, it would be naïve not to consider that there are other efforts towards the creation of new PAs in Brazil. The total area of federal PAs increased enormously from the 30's up to now. Between 1980 and 2009, 90% of the total existent PA area was gazetted because of a range of factors, including the creation of the National Plan for Conservation Units (Plano do Sistema de Unidades de Conservação do Brasil), stronger environmental institutions and higher management capacity (Drummond, et al. 2011), established democracy (Abman, 2018), international support (MMA, 2010; The World Bank and Development, 2012) and an increasingly engaged civil society (Oliveira, 2005).

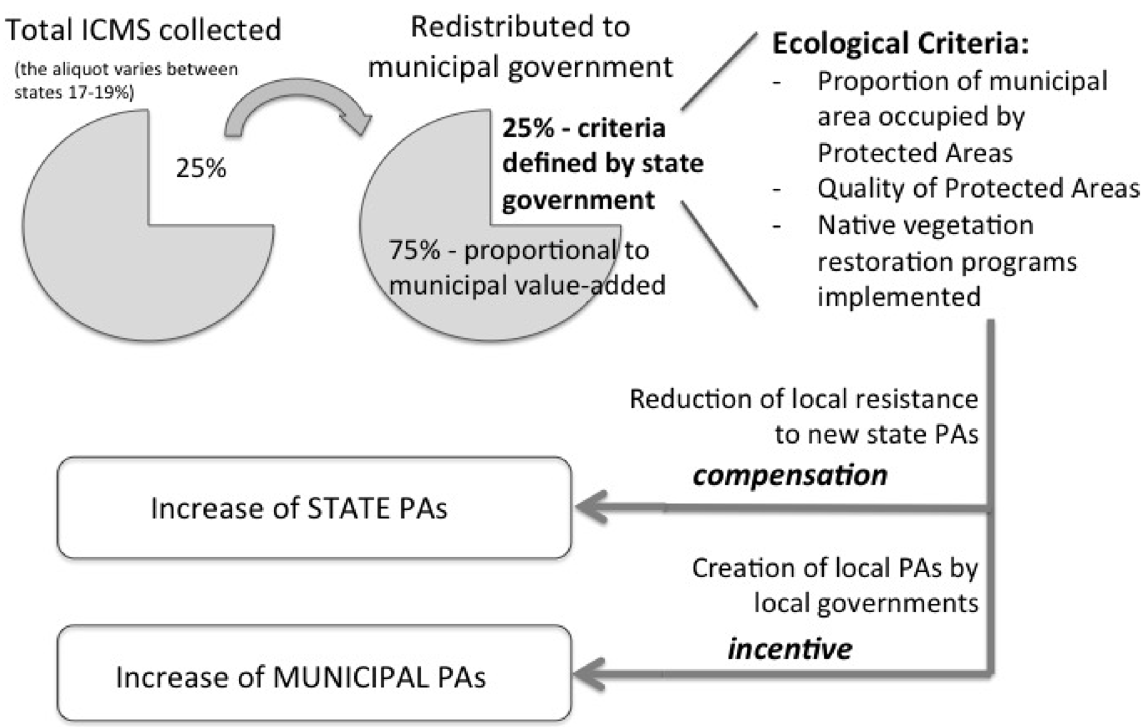

Figure 2.1: The Ecological ICMS redistribution mechanism and two possible ways it is expected to impact nature conservation.

Estimates of the positive impacts of the ICMS-E on the promotion of new PAs are mixed. Loureiro (2002) and May et al. (2002) note that the expansion of the number of both state and municipal PAs in the states of Paraná and Minas Gerais over the period 1991-2000 parallel the implementation of the ICMS-E policy in these states. Against this described correlation, Silva

Junior et al (2013) detected no significant effect on the creation of new PAs in a comparison of Pernambuco state (ICMS-E since 2012) with the control group of Bahia, Alagoas and Paraíba states (da Silva Júnior et al., 2013). This contradictory observations about the effect of the ICMS-E may be a consequence of different methodological approaches. A careful monitoring of indicators is necessary, but assuming that when an indicator improves, it means the intervention is "working" might be misleading as indicators may vary due to other possible factors correlated with the timing and the location of interventions (Ferraro, 2009). The counterfactual approach uses econometric tools to estimate what would have occurred without the intervention and overcome the problem of possible existing confounding factors, in a way that one can attribute the observed effect to the intervention and not to other plausible explanations (Baylis et al., 2015; Ferraro, 2009). It is largely used in impact studies of health and education policies (Angrist and Pischke, 2008) and is pointed as the adequate approach to environmental policy evaluation also (Baylis et al., 2015; Ferraro, 2009).

Here we evaluate the effectiveness of ICMS-E legislation on the area under legal protection, using a counterfactual approach. Focusing on the Atlantic Forest region in south and southeastern Brazil, we contrasted municipalities from states that implemented the Ecological ICMS with comparable control states, with a dataset of 2481 municipalities. We explored three main questions: i) Does the ICMS-E influence the creation of new state PAs, when it is expected to work as a compensatory resource?; ii) Does the ICMS-E influence the creation of new municipal PAs, working as an incentive to local governments?; iii) If present, does the effect of the ICMS-E persists over time as more and more PAs are created? We hypothesized that the expected increase in new PAs occur through different processes for state or municipal protected areas. While local governments have no action on the creation of state PAs, they still benefit from the ICMS-E revenue when these new areas are created. So, state institutions can "bargain" and/or face less resistance from local municipalities to their PA proposals. In this case, the ICMS-E revenue is understood as a compensatory resource for local municipalities hosting state areas for conservation - the original concept of the ICMS-E. On the other hand, when municipal areas are accounted in the calculus of the ICMS-E, local managers can voluntarily set aside areas for conservation in order to benefit from the ICMS-E resource sharing. In these cases, the existence of the ICMS-E law can be considered an incentive for the creation of new municipal PAs. The crucial difference is that, in the case of state PAs, the PA proposer is not the one benefitted by the ICMS-E revenue, while in the case of municipal PAs, the proposer is the same who benefits from the ICMS-E generated revenue. Finally, we expect that the effect of the ICMS-E law diminishes over time either due to a reduction of the relative value paid by the ICMS-E as beneficiaries increases, or due to lack of space for new PAs.

2.3 Methods

We considered eight states in the Atlantic Forest region (Table B1, Supplementary material): the first states to implement the ICMS-E (Paraná, Minas Gerais and São Paulo), control states in the same regions (South and Southeast), where ICMS-E was not implemented (Santa Catarina and Espírito Santo) and the adjacent states in order to build a robust database (Rio Grande do Sul, Mato Grosso do Sul and Rio de Janeiro). We considered the six states that implemented the ICMS-E (Paraná, São Paulo, Minas Gerais, Rio Grande do Sul, Mato Grosso do Sul and Rio de Janeiro) as treatment for the analysis of state PAs, and five as treatment for the analysis of municipal PAs. In this case, we assigned São Paulo as control because it does not include municipal PAs in the calculus of the ICMS-E redistribution.

2.3.1 Empirical approach

Globally, two main factors can influence the impact of the ICMS-E on the creation of new PAs: i) the relative pay-off value per additional PA area, and ii) remaining municipal area available for new PAs. We created two variables to distinguish the effect of these two factors. First, we obtained an approximately estimated relative pay-off value based on the original ICMS-E formula (see Supplementary Material). This revenue *share* variable values indicate the relative division of the ICMS-E revenue in each year in each state, being inversely proportional to the pay-off value per additional PA unit area. We used a two year lag for the revenue *share* variable because the ICMS-E revenue division is based on the proportion of PAs existent two years before. We expect that as more municipalities join the group of ICMS-E beneficiaries, the less attractive the ICMS-E resource is, with potentially a dampening effect on the probability of new PAs over time.

Secondly, municipalities can run out of area for new PAs over time, creating an upper-limit on the number of PA categories that can not overlay. So we calculated the accumulated percentage of total municipality area covered with PAs, for each individual municipality in each year, to account for remaining area available for new PAs. Regarding PA categories, we consider crucial to distinguish the very unrestricted *Environmental Protected Areas (APAs)* from other PA categories. Based on the National System for Conservation Units (in Portuguese *Sistema Nacional de Unidades de Conservação – SNUC*), APAs are very extent areas, with consolidated human occupation, encompassing public and private lands, which implementation does not displace resident population and allows economical activities, such as agriculture and mining (Federal Law 9985/2000). Among PA categories, it is the most unrestricted one and can overlay with any other type of PA. This latter aspect is important because existing PAs are not a preclusion for APA creation. As PA accumulated percentage increases, space for new PAs decreases, so we call this variable *lack of space* for new PAs and we calculated it separately for: i) all PAs except APAs and ii) for APAs exclusively. We expect that as the accumulated area of PAs in a municipality increases, the probability of creating new PAs will decline.

We tested if ICMS-E legislation positively affect the creation of new PAs separately for state and municipal PAs, for APAs exclusively and other PAs. To do this, we used a panel data regression of the form:

$$\begin{aligned} y_{m,t} =& \theta_t + \beta_1\, law_{m,t} + \beta_2\, share_{m,t} + +\beta_3\, law.share_{m,t} + \beta_4\, space_{m,t} + \beta_5\, X_{m,t} + \\ & \alpha_m + \epsilon_{m,t} \end{aligned} \tag{2.1}$$

where y is the percentage of total municipality covered with new PAs (state or municipal) in each municipality m in each year t; *law* identifies if a municipality is under ICMS-E law in t year; revenue *share* and lack of *space* are independent variables; X are the control variables; θ denotes time effect and ϵ is the error term. As we expect that revenue *share* is only relevant when ICMS-E law is implemented, *share* variable is interacted with *law* variable.

There are several reasons that may cause different states to present different treatment effects, including institutional trajectory, investments in technical resources, and political will concerning the environment. To detect separate effects for each state we included in the regressions the two-way interaction between *law* and state identity ST.

$$\begin{aligned} y_{m,t} =& \theta_t + \beta_1\, law * ST_{m,t} + \beta_2\, share_{m,t} + \beta_3\, space_{m,t} + \beta_4\, X_{m,t} + \\ & \alpha_m + \epsilon_{m,t} \end{aligned} \tag{2.2}$$

Finally, we expect that the effect of the ICMS-E law diminishes over time either because the incentive diminishes due to the ICMS-E revenue share increase or due to lack of space for new PAs. To test this hypothesis, we included the categorical variable of the year of ICMS-E implementation (year 1, year 2, year 3 etc), as well as its two-way interaction with ST to detect whether the law effect over time differs among states. As per year effect showed a marked difference between the first initial years and the following years, we ran the regressions for these two periods separately.

Additionally, because expanded area under agricultural land use on rural lands reduces the chances of new protected areas for conservation, we controlled for both agricultural and cattle beef production. All models were conducted in R (Version 3.4.2.) (R Core Team, 2017), using the PLM package (Version 1.6-6) for panel regression analyses (Croissant et al., 2008).

2.3.2 Protected Areas

We gathered data about protected areas from two sources: i) the National Conservation Units Register (Cadastro Nacional de Unidades de Conservação) organized by the Federal Ministry of the Environment and available in its website (MMA, Ministério do Meio Ambiente, n.d.); and ii) a survey of municipal reserves conducted by the *Ambiental 44 Informação e Projetos em Biodiversidade Ltda*, under the supervision of Luiz Paulo Pinto and with the support of the NGO SOS Mata Atlântica. The National Conservation Units Register comprises all the categories of protected areas in Brazil - Strictly Restricted(SR), Sustainable Use (SU) and Indigenous Land (IL) - and from the three administrative levels - federal, state and municipal -, since the creation of the first Protected Area in Brazil in the 30's. While federal and state reserves are well documented in the national federal register, municipal reserves are poorly registered. Municipal reserves are created by local government (municipality), through municipal decrees. The *Ambiental 44 Informação e Projetos em Biodiversidade Ltda* has detailed information about the municipal protected areas in the Atlantic Forest region and is probably now the best and most updated source for municipal PA. From both sources, we obtained the following information about federal, state and municipal protected areas: category, year of creation, locality (municipalities where they are located) and extension of the reserve (area). Then, we calculated the percent of the total municipality area covered by each PA for 2481 municipalities in the Atlantic Forest region belonging to eight studied states (31 municipalities created between 1995 and 2010 were excluded from the database).

2.3.3 Control variables

We searched for data at the *Instituto Brasileiro de Geografia e Estatística (IBGE)* (National Institute of Geography and Statistics) through its online database SIDRA (IBGE, n.d.), linked to the Federal Ministry of Planning, Development and Management. For agricultural yield, we used a derivative variable calculated by the weighted mean of the produced quantity and current average price paid to the producer, according to periods of harvest and commercialization of each product (freight, taxes and charges are not included in the price) (IBGE, Instituto Brasileiro de Geografia e Estatística, 2017b). In this case, we updated values from their original currencies

(Cruzados, Cruzados Novos, Cruzeiros, Cruzeiros Reais and Reais) to corrected US dollars, using dollar quotation for Brazilian currencies for the last day of each year, provided by the Brazilian Central Bank, and inflation information (based on Consumer Price Index), provided by the Bureau of Labor Statistics, US government. Cattle beef production was based on the effective herd in each municipality divided by total municipal area (density of cattle). In this case, data was also obtained at the *Instituto Brasileiro de Geografia e Estatística (IBGE)* (National Institute of Geography and Statistics) (IBGE, Instituto Brasileiro de Geografia e Estatística, 2017a).

2.4 Results

2.4.1 State and Municipal PAs

Following the general trend for PA creation between 1980 and 2009, we found that both state and municipal governments increased the areal extent of new PAs over time, with unrestricted APAs being larger than other PA categories (Fig. 2).

We found that the Ecological ICMS law has a positive and significant effect on the creation of new state and municipal PAs during the first 10 years of law, after when the effect is no longer significant or even positive (Table 2.1). During the first decade of the law, the effect is positive and significant for both state and municipal PAs, despite category (Fig. 2.3). For all PAs except APAs, the effect of the ICMS-E law on new state PAs is twice the effect on new municipal PAs. On the other hand, for unrestricted APAs the effect of the ICMS-E law is almost twice higher for new municipal APAs than for new state APAs (Table 2.1).

Table 2.1: Regression coefficients and standard errors (inside parentheses) of the ICMS-E effect on new state and municipal PAs, as the percentage of municipal area, per period of time.

	State		Municipal	
DV: New PAs (municipal area percentage)	All PAs, except APAs	exclusively APAs	All PAs, except APAs	exclusively APAs
Year law 1 to 10	0.031**	0.160***	0.018***	0.351***
	(0.010)	(0.044)	(0.005)	(0.038)
Year law 11 to 25	-0.005	0.015	0.009	-0.042
	(0.014)	(0.061)	(0.006)	(0.045)

***p $<$ 0.001, **p $<$ 0.01, *p $<$ 0.05, .p $<$ 0.1

The revenue share variable has a little negative effect on the creation of new state and municipal unrestricted APAs, whether there is ICMS-E, and a very little negative effect, on other state PAs, even in the absence of ICMS-E (Table 2.2). The accumulated area of restricted and unrestricted PAs play both a significant role on the creation of new PAs, showing a negative effect most of the time. This means generally, as expected, that as the total accumulated

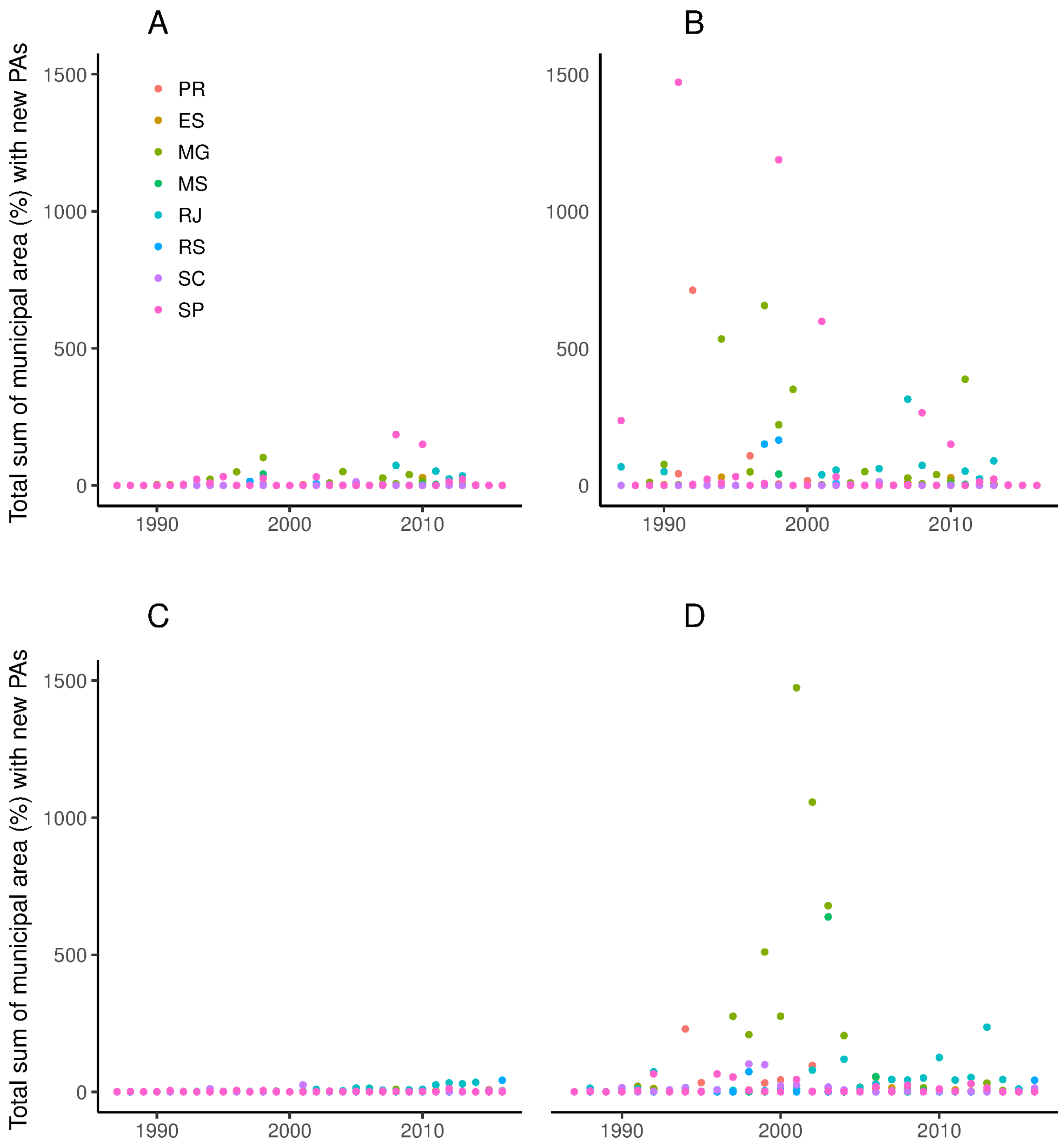

Figure 2.2: The creation of new PAs per year, as the total sum of individual municipal area percentages covered with new PAs, over time for state and municipal PAs (A – all state Pas, except APAs; B – state APAs only; C – all municipal PAs, except APAs and D – municipal APAs only).

PA area increases in the municipality, the probability of new PAs significantly decreases. The exception is in the case of state unrestricted PAs, in which the accumulated restricted PA area (*lack of space for PAs*), except APAs, has a small but positive effect, while accumulated area of unrestricted PAs (*lack of space for unrestricted APAs*) has a large and negative effect (Table 2.2). Agricultural production yield and cattle density have a significant but very small effect in

Table 2.2: Regression coefficients and standard errors (inside parentheses) of the ICMS-E effects on new state and municipal PAs, as the percentage of total municipal area.

	State		Municipal	
DV: New PAs (municipal area percentage)	All PAs, except APAs	exclusively APAs	All PAs, except APAs	exclusively APAs
ICMS-E law	0.018	0.222***	0.008	0.354***
	(0.011)	(0.049)	(0.006)	(0.051)
Share	< -0.001**	< 0.001	< 0.001**	< 0.001
	(< 0.001)	(< 0.001)	(< 0.001)	(< 0.001)
Lack of space for PAs, except APAs	-0.037***	0.028***	< 0.001	-0.018***
	(0.001)	(0.005)	(< 0.001)	(0.005)
Lack of space for unrestricted APAs	-	-0.113***	-	-0.007***
	-	(0.001)	-	(0.001)
Agricultural Production	< 0.001**	< 0.001**	< 0.001	< 0.001.
	(< 0.001)	(< 0.001)	(< 0.001)	(< 0.001)
Cattle density	< 0.001	-0.001*	< 0.001	< 0.001
	(< 0.001)	(< 0.001)	(< 0.001)	(< 0.001)
Share : ICMS-E law	< 0.001*	< 0.001**	< 0.001	< 0.001**
	(< 0.001)	(< 0.001)	(< 0.001)	(< 0.001)

***p < 0.001, **p < 0.01, *p < 0.05, .p < 0.1

the case of new state PAs, and no effect on new municipal PAs.

2.4.2 Ecological ICMS effect per state

Table 2.3: Regression coefficients and standard errors (inside parentheses) of the ICMS-E effects on new state and municipal PAs, as the percentage of total municipal area, for each state separately.

	State		Municipal	
DV: New PAs (municipal area percentage)	All PAs, except APAs	exclusively APAs	All PAs, except APAs	exclusively APAs
Mato Grosso do Sul (MS)	0.078.	1.089***	0	0.719***
	(0.042)	(0.187)	(0.021)	(0.169)
Minas Gerais (MG)	0.040*	0.212**	0.003	0.552***
	(0.015)	(0.066)	(0.006)	(0.053)
Paraná (PR)	-0.002	0.357***	-0.004	-0.018
	(0.020)	(0.090)	(0.009)	(0.078)
São Paulo (SP)	0.043*	0.018	-	-
	(0.016)	(0.070)	-	-
Rio de Janeiro (RJ)	0.203***	0.574***	0.147***	0.610***
	(0.025)	(0.112)	(0.012)	(0.102)
Rio Grande do Sul (RS)	-0.039**	-0.078	-0.001	-0.199***
	(0.014)	(0.064)	(0.007)	(0.058)

***p < 0.001, **p < 0.01, *p < 0.05, .p < 0.1

The effect of the ICMS-E law differs among states (Table 2.3 and Fig. 2.3). Generally, similar to when all states are observed together, the ICMS-E has a higher and significant effect on new unrestricted PAs. Exceptions are São Paulo that shows a significant and positive effect on state restricted PAs, and Rio Grande do Sul where the ICMS-E has a null or negative effect on new PAs. It is worth noticing that the effect of ICMS-E law on new restricted PAs is only significant in the case of state PAs; only Rio de Janeiro shows a positive and significant effect of the ICMS-E law on new restricted municipal PAs (Table 2.3).

The ICMS-E effect on all PAs (restricted and unrestricted considered together) for state and municipal PAs for each year of law existence, showed a similar ephemeral effect for individual states, as observed for all states together (Fig. 2.3). The main exceptions are Rio Grande do Sul that shows no positive effect of the ICMS-E on new PAs and Minas Gerais that does not show a decrease in state PAs. It is worth noticing that Rio de Janeiro is the only state that has less then 10 years of law implementation.

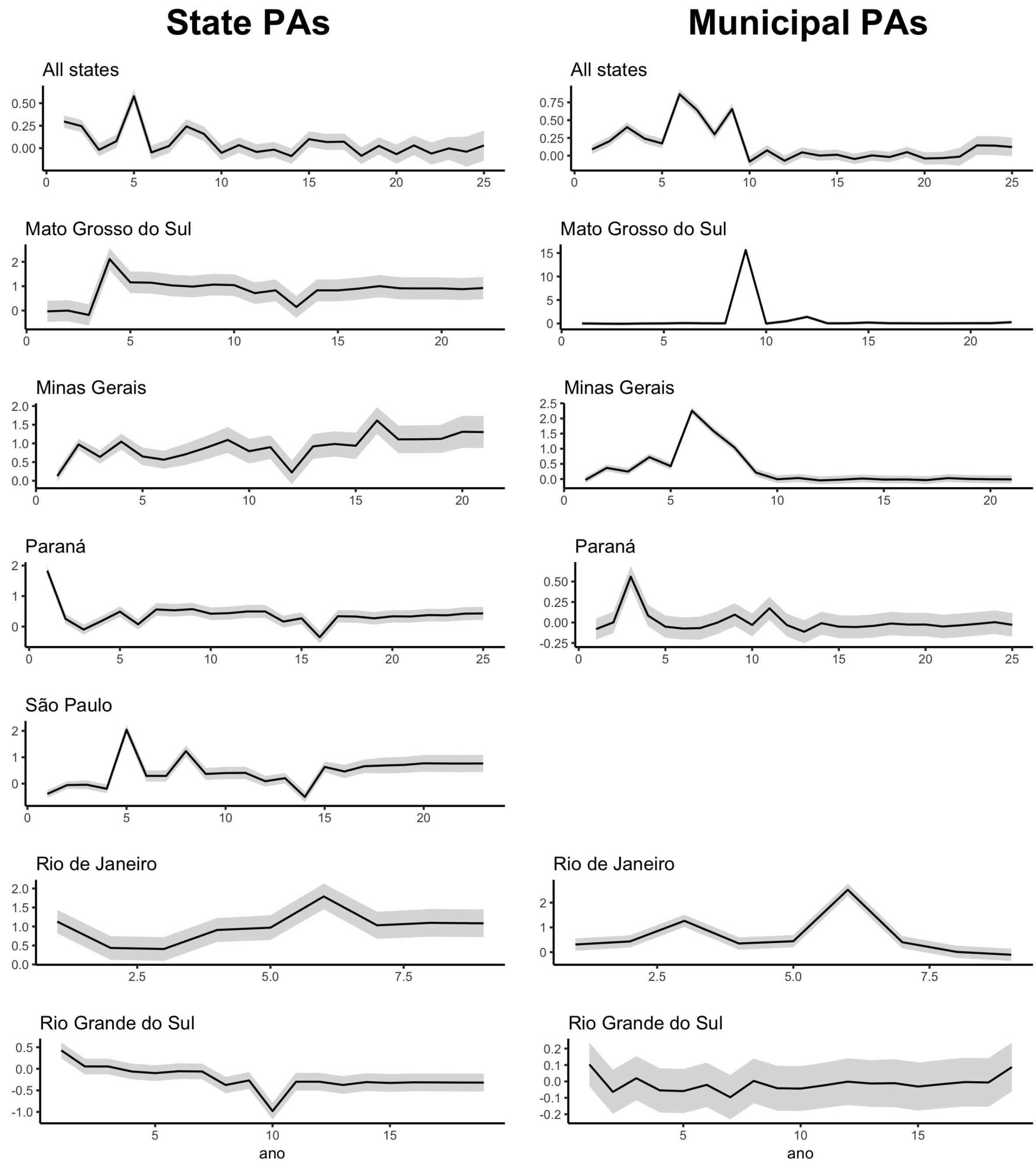

Figure 2.3: Regression coefficients and their standard deviation for each year of law implementation in each state, for state and municipal PAs. Note that y axis have different values in each state, as the observed effects have different magnitudes.

2.5 Discussion

Our results demonstrate that the ICMS-E intervention promotes the creation of new PAs, principally in a category of protection that imposes very little restrictions to land use (*Environmental Protected Areas - APAs*), for which the overall effect is more than five times higher than the effect for other PAs (Strictly Protected and Sustainable Use). Besides, the ICMS-E also differentially affects state and municipal PAs, promoting more PAs in the case of state PAs when APAs are ignored, and corroborating the idea that the mechanism operates through different processes regarding state and municipal governments. The overall effect of the ICMS-E, and particularly the effect on new municipal PAs, is present in the initial years right after the implementation of the law, but largely disappears after 10 years. We found that this decline in the creation of new PAs is affected by reduced available area for further conservation, rather than by the reduction of ICMS-E revenue share.

Although the ICMS-E has an overall positive effect on the creation of new PAs, it is not equal for all PA categories, and for state and municipal PAs. In four of the six observed states, the unrestricted APAs undergo a larger increase in extension as a consequence of the ICMS-E law, when compared to other PAs. Usually, ICMS-E state laws define different weights for different PA categories, being the more restricted PAs the ones with higher weight. However, because of major land use restrictions, creating restricted PAs usually requires more technical studies, stakeholder's participation and involvement, taking more time, eventually many years, from the conception to the creation of the new area. Often, PA proponents face local population or economical sectors resistance to their proposals (Pedlowski et al., 1999; Vivacqua and Vieira, 2005), which we expect to be stronger insofar as the PA category proposed is more restricted. The ICMS-E is structured in a way - with relative resource sharing increasing as new PAs are created - that stimulates a race for its resource right after the law signature: who gets first benefits from a larger part of the resource, at least for a few initial years. Each year a new share occurs considering new municipalities in the group of beneficiaries. In this scenario, PAs, such as the *Environmental Protected Area (APA)* category, that impose very few land use restrictions and encounter lower resistance from locals are easily and rapidly proposed and created. Although APAs have a low weight in the ICMS-E formula, they might allow a certain municipality to move from zero ICMS-E revenue to some additional ICMS income. If we consider a mayor with a four years term, he/she can plan an APA in its first year of the term and still increase its ICMS total revenue right before the next elections. This possibly explains why APAs show an almost 20 times higher increase in extension compared to other municipal PAs.

Regarding state PAs, unrestricted APAs also increase due to the ICMS-E intervention. However, other state PAs undergo a larger and significant increase during the first years of the ICMS-E law, when compared to municipal PAs. State government - the proponents of state PA - are not the ones benefitting from the ICMS-E so there is no particularly interest in proposing quickly and easily accepted PAs. On the contrary, instead of approving the very unrestricted APAs, state government environmental institutions may take the opportunity of the ICMS-E receipt as a chance to approve and implement other PAs, both strictly protected and sustainable use, certainly valuable for environmental conservation (Pfaff et al., 2014; **?**). Human modified landscapes and their elements, such as agricultural mosaics or tree monoculture, may be of important conservation value complementing the goals of rich biodiversity and less untouched protected areas (Tabarelli et al., 2010). In this context, $APAs$ may be conceived as buffer areas near or around other PAs, as part of a state strategy to combine different PA categories in order to improve overall conservation outcomes. New unrestricted $APAs$ overlay with already existent PAs would explain why the accumulated area of other PAs has a small but positive effect on new state APAs.

As expected, the estimated revenue relative share, that accounts for the effect of the ICMS-E allotment, has a significant effect on the creation of new PAs, but this effect is very reduced and limited over time. This result suggests that the relative allotment of the ICMS-E resource, although significant, may not be the main factor responsible for the decreasing effect of the ICMS-E law on the creation of new PAs, concerning the period and states observed. There are feasible explanations for this observed result. First, new PAs, created as a consequence of the ICMS-E intervention, may be located in lands with very low opportunity costs. Usually, PAs are located in places with higher elevations, steeper slopes and greater distances to roads and cities (Joppa and Pfaff, 2009), where land has very low opportunity costs and are under low economic pressure (Joppa and Pfaff, 2011). With ICMS-E intervention, municipalities searched for these low opportunity cost areas for PA creation in order to quickly benefit from the ICMS-E revenue. Once these low costs PAs are created, the marginal cost of new PAs increases as space reduces and the remaining areas have higher opportunity costs. The observed negative effect of the accumulated area of PAs means that the higher the area covered with PAs in the municipality, the lower the chance of new PAs creation. Secondly, land opportunity costs may have raised across the observed period, creating new economical incentives for land use transformation, both rural and urban, that surpasses the ICMS-E attractiveness, even if pay-off value per additional PA hectare also increases. Either because low opportunity costs lands are preferred for PA creation or new economical incentives arises, we suggest that absolute values paid by the ICMS-E intervention did not meet or exceed real land opportunity costs, at least during the observed period (from 1989 to 2016). In this case, the created new PAs may not bring an additional contribution to natural resources protection, as those areas were not facing deforestation pressure. Globally, protecting the cheapest lands will increase the number of threatened vertebrates covered in PAs by only 6%, from 2014 to 2020, even if we manage to achieve the Aich target for conservation, defined in the Convention for Biological Diversity – 13 to 17% of the world's surface inside protected areas (Venter et al., 2014). Thus, we urge to create conservation incentives and strategies that allow to increase the contribution of more valuable lands, and their imperiled species, to global biodiversity conservation. Regarding the very unrestricted APA areas, the lack of contribution can possibly accrue to these new areas being unable to refrain activities threatening natural resources.

The general patterns of PA creation were also observed in the states separately, but we detected differences among them. One exception is Rio Grande do Sul that does not show an effect of the ICMS-E. In this case, state government possibly did not properly divulge the ICMS-E mechanism among municipalities and local governments have not been aware of the possible benefit related to it (Banker, 2011). In Rio Grande do Sul, municipal governments need to actively register their municipal PAs in the State System of Conservation Units (created by the state decrees 34,256/92 and 38,814/98), so that PAs can be considered for the ICMS-E benefit (Aude Lovatto and Marçal Rocha, 2016). Lovatto et al (2016) found that several municipalities hosting PAs are not receiving the ICMS-E revenue as they should be. It is also possible that our database misses some municipal PAs also absent in the Rio Grande do Sul state register system. Paraná and Rio de Janeiro are illustrative cases regarding the ICMS-E impact over time. Paraná was the first state to implement the ICMS-E and has 25 years of observed policy impact, while Rio de Janeiro was the last state in our sample to implement the ICMS-E, with only 9 observed years of law existence. If per year effect is not evidenced, one could conclude that the ICMS-E effect is stronger in Rio de Janeiro that shows positive and significant increase of area for all types of PAs. In Paraná, when the whole 25 years period is considered, the ICMS-E effect on new PAs is non significant because the initial increase of PAs is offset by the following decrease of new PAs creation. Nevertheless, observing per year effect, the ICMS-E in Paraná actually shows an increase of municipal PAs during the initial years of the ICMS-E (Fig. 2.3), similar to the observed effect in Rio de Janeiro and other states.

We observe that the ICMS-E law creates different incentive environments for state and municipal governments. Mato Grosso do Sul, Minas Gerais and São Paulo show a significant increase in all state PAs, not only APAs, and this effect persists after the first decade of the ICMS-E law signature. In these cases, state government political will towards the creation of new PAs might be the main driver for PA proposals and expansion in the state. The ICMS-E resource does not work exactly as a compensation, as it does not face real land opportunity costs, but certainly works as an additional and positive benefit that state government negotiates in exchange for PAs implementation. Thus, in the case of state PAs, we suggest the ICMS-E is a secondary driver for PA improvement, operating in synergy with a major driver - the state government conservation plans execution. Political will and strength of state government towards conservation action is an unobservable variable in our study that may explain the observed persistence of new state PAs over time.

The ICMS-E is certainly a benefit for municipalities hosting PAs, working as a complementary strategy and increasing the chance of success of other actions towards PA creation, but it is not a strong incentive for an exclusive voluntary action. As a benefit, the ICMS-E revenue mechanism does not generate conflict between state government political motivations and state environmental goals; it supports state government institutions in the implementation of their conservation plans, possibly lowering local resistance to new PA proposals. In a similar way, the ICMS-E revenue could align with civil society common interests and support different communities to organize and improve environmental conservation and common resource management, at the state and local levels. On the other hand, as a financial incentive able to raise local government receipt, the ICMS-E revenue promotes the competition between political and environmental goals at the local level. As a consequence, it is more subjected to political cycles and their consequent short-term decisions framed by opportunistic political motivation.

2.6 Conclusion

Generally, our results corroborate the expectations that the ICMS-E promotes the expansion on new PAs (May et al., 2002), but at the same time they shed light on the persistence and quality of this impact. When ICMS-E is expected to work as an incentive to local government to increase the number and extension of local PAs, it shows a lack of persistence: the initial years witness an increase in PA area but this effect disappears approximately one decade later. The race to expand PA areas in municipalities competing for the ICMS-E initial resources and political motivations to promptly increase local government receipt lead to several large unrestricted local new APAs. The same volatile behavior is not observed when ICMS-E is expected to work as a compensatory resource for state PAs. In this case, the intervention acts in synergy with state conservation plans execution, increasing the chances of well succeeded PA proposals. Thus, the ICMS-E mechanism creates distinct incentive environments and has a differentiated effect upon state and municipal governments. It stimulates environmental legitimate action in the former case, and political motivated action in the latter. Hence, the overall ICMS-E impact is of higher quality and more persistent in the state case, which indicates that it can be a useful tool to be applied in conjunction with other conservation strategies.

2.7 Acknowledgments

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Bibliography

Abman, R., 2018. Rule of law and avoided deforestation from protected areas. Ecological Economics 146, 282–289.

Angrist, J. D., Pischke, J.-S., 2008. Mostly Harmless Econometrics: An Empiricistís Companion. Princeton University Press.

Aude Lovatto, P. M., Marçal Rocha, J., 2016. ICMS Ecológico como ferramenta de proteção ambiental: análise da aplicação no estado do Rio Grande do Sul. Ciência e Natura 38 (2).

Banker, G., 2011. Prefeituras desconhecem o ICMS Ecológico. Jornal do Comércio.
URL http://www.jornaldocomercio.com/site/noticia.php?codn=67516

Baylis, K., Honey-Rosés, J., Börner, J., Corbera, E., Ezzine-de Blas, D., Ferraro, P. J., Lapeyre, R., Persson, U. M., Wunder, S., Pfaff, A., 2015. Mainstreaming Impact Evaluation in Nature Conservation. Conservation Letters 9 (1), 58–64.

Croissant, Y., Millo, G., et al., 2008. Panel data econometrics in r: The plm package. Journal of Statistical Software 27 (2), 1–43.

da Silva Júnior, L. H., Pedrosa, B. M. J., da Silva, M. F., 2013. Avaliação dos impactos do icms socioambiental na criação de unidades de conservação e unidades de tratamento de resíduos sólidos em pernambuco: Uma análise a partir do método de diferençasem-diferenças. Revista Econômica do Nordeste 44 (2), 557–572.

Ferraro, P. J., 2009. Counterfactual thinking and impact evaluation in environmental policy. New Directions for Evaluation 2009 (122), 75–84.

IBGE, I. B. d. G. e. E., n.d. Sistema ibge de recuperação automática.
URL https://sidra.ibge.gov.br/home/pimpfbr/brasil

IBGE, Instituto Brasileiro de Geografia e Estatística, 2017a. Tabela 3939 - efetivo dos rebanhos, por tipo de rebanho.
URL https://sidra.ibge.gov.br/tabela/3939

IBGE, Instituto Brasileiro de Geografia e Estatística, 2017b. Tabela 5457 - área plantada ou destinada à colheita, área colhida, quantidade produzida, rendimento médio e valor da produção das lavouras temporárias e permanentes.
URL https://sidra.ibge.gov.br/tabela/5457

Joppa, L. N., Pfaff, A., 2009. High and Far: Biases in the Location of Protected Areas. PLoS ONE 4 (12), e8273–6.

Loureiro, W., 2002. Contribuição do ICMS Ecológico à conservação da biodiversidade no Estado do Paraná. Ph.D. thesis, Engenharia Florestal, Universidade Federal do Paraná.

May, P. H., Neto, F. V., Denardin, V., Loureiro, W., May 2002. Using Fiscal Instruments to Encourage Conservation: Municipal Responses to the 'Ecological' Value-added Tax in Paraná and Minas Gerais, Brazil. In: Selling Forest Environmental Services. pp. 1–28.

MMA, B. M. o. t. E., 2010. Fourth national report to the Convention on Biological Diversity Brazil.

MMA, Ministério do Meio Ambiente, n.d. Cadastro nacional de unidades de conservação.
URL http://www.mma.gov.br/areas-protegidas/cadastro-nacional-de-ucs

Oliveira, J. A. P. d., 2005. Enforcing protected area guidelines in brazil: What explains participation in the implementation process? Journal of Planning Education and Research 24 (4), 420–436.

Pedlowski, M., Dale, V., Matricardi, E., 1999. A criação de áreas protegidas e os limites da conservação ambiental em rondônia. Ambiente & sociedade (5).

Pfaff, A., Robalino, J., Lima, E., Sandoval, C., Herrera, L. D., 2014. Governance, Location and Avoided Deforestation from Protected Areas: Greater Restrictions Can Have Lower Impact, Due to Differences in Location. World Development 55 (C), 7–20.

R Core Team, 2017. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.
URL https://www.R-project.org/

Ring, I., 2008. Integrating local ecological services into intergovernmental fiscal transfers: The case of the ecological ICMS in Brazil. Land Use Policy 25 (4), 485–497.

Santos, R., Ring, I., Antunes, P., Clemente, P., 2012. Fiscal transfers for biodiversity conservation: The Portuguese Local Finances Law. Land Use Policy 29 (2), 261–273.

Sauquet, A., Marchand, S., Féres, J. G., 2014. Protected areas, local governments, and strategic interactions: The case of the ICMS-Ecológico in the Brazilian state of Paraná. Ecological Economics 107 (C), 249–258.

Tabarelli, M., Aguiar, A. V., Ribeiro, M. C., Metzger, J. P., Peres, C. A., 2010. Prospects for biodiversity conservation in the atlantic forest: lessons from aging human-modified landscapes. Biological Conservation 143 (10), 2328–2340.

The World Bank, I. B. f. R., Development, 2012. Expanding financing for biodiversity conservation: Experiences from latin america and the caribbean.

Venter, O., Fuller, R. A., Segan, D. B., Carwardine, J., Brooks, T., Butchart, S. H., Di Marco, M., Iwamura, T., Joseph, L., O'Grady, D., et al., 2014. Targeting global protected area expansion for imperiled biodiversity. PLoS Biology 12 (6), e1001891.

Vivacqua, M., Vieira, P. H. F., 2005. Artigo: Conflitos socioambientais em unidades de conservação. Política & Sociedade 4 (7), 139–162.

Young, C. E. F., 2005. Financial Mechanisms for Conservation in Brazil. Conservation Biology 19 (3), 756–761.

Supplementary material

B.1 Ecological ICMS

The ICMS-E is currently employed across 16 states in Brazil, and its implementation differs in important ways across states. The ICMS-E was first implemented in Paraná state in 1992, after several municipalities articulated frustration to state government about the economic impacts of land use restrictions related to protected areas created within their boundaries (Loureiro, 2002). While the Paraná government originally constructed the ICMS-E as a compensatory mechanism to adjust for these costs, such payments may additionally stimulate the gazetting of new Pas in order to qualify for ICMS-E payments (Loureiro, 2002).

Following the Paraná example, São Paulo and Minas Gerais States approved the Ecological ICMS-E law, with slight differences in each state. The São Paulo legislation excludes municipal PAs in the calculus of the proportion covered with protected areas in the municipality (São Paulo state law no. 8,510/1993). Minas Gerais has broadened the scope of redistributiong criteria for the ICMS-E including, for instance, municipal accomplishments in health and education, in a way that the ICMS-E law in Minas Gerais has the nickname "Robin Hood Law". Finally, it is worth noting, for instance, that the percentage of the ICMS assigned to ecological accomplishments varies among states (Table 1).

B.2 Ecological ICMS general formula

The ICMS-E formula varies among states. However, generally, the ICMS-E allotment is calculate using the following rationale developed by the state environmental institution of Paraná (Loureiro, 2002). The following summarized description is based on Loureiro (2002).

For each municipality hosting at least one PA, hence participating in the ICMS-E benefit, the proportion of its total area occupied by each PA is multiplied by a conservation factor in a way that more restricted PAs have higher values than less restricted PAs. It is named the Biodiversity Basic Conservation Coefficient (CCB):

Table B.1: ICMS-E information for eight studied states in the Atlantic Forest region. Atlantic Forest percent refers to the percentage of state area inside the Atlantic Forest region

State	ICMS decree year	Percent of ICMS assigned to ICMS-E	State area (hectares)	Atlantic Forest percent
Espírito Santo	-	-	4,609,503	100
Minas Gerais	1995	1	58,651,979	47
Paraná	1991	5	19,930,768	99
Santa Catarina	-	-	9,573,618	100
São Paulo	1993	0.5	24,822,624	69
Rio de Janeiro	2007	1.125	4,377,783	100
Rio Grande do Sul	1997	7	26,876,641	52
Mato Grosso do Sul	2000	5	35,714,473	18

Source: (TNC, The Nature Conservancy, n.d.) and (SOS MA, 2018)

$$CCBij = \frac{Auc}{Am} * Fc \tag{B.1}$$

where CCB is the coefficient for municipality i and protected area j, Auc is the protected area extent (hectares), Am is the total municipal area (hectares) and Fc is the conservation factor. The conservation factor Fc is 1 for strictly protected PAs, 0.5 for sustainable use PAs and 0.08 for the very unrestricted *Environmental Protected Areas (APAs)*.

Then, the Biodiversity Conservation Coefficient for each municipality is:

$$CCBMi = \sum_{j=1}^{n} CCBi \tag{B.2}$$

where $CCBM$ is the total coefficient for municipality i, and CCB is the coefficient for municipality i and protected are j up to n existent protected areas in the municipality. The $CCBMi$ is the sum of all $CCBij$ obtained for the municipality.

Finally, in each year, a municipal factor FM is calculated for each municipality based on the following fraction:

$$FMi = 0.5 * \frac{CCBMi}{\sum_{i=1}^{n} CCBM} * 100 \tag{B.3}$$

where 0.5 is the percentage of the ICMS-E assigned to protected areas. In the case of Paraná, half of the ICMS-E is redistributed for PA coverage, according to these formulas. The ICMS-E revenue each municipality receives is proportionate to this municipal factor FM. In the case of Paraná, the CCB coefficient is also multiplied by a quality factor indicating that the PA has improved its quality or not in the specific year.

B.3 Revenue share variable

Based on the above formula, we calculated the sum of all $CCBM$ for each state in each year:

$$RSys = \sum_{i=1}^{n} CCBM \tag{B.4}$$

RS is the revenue *share* variable for year y and state s, and $CCBM$ is the total coefficient for each municipality i in the state. As this is the denominator of equation 3, it represents the relative division of the ICMS-E revenue executed in each year.

Bibliography

Loureiro, W., 2002. Contribuição do ICMS Ecológico à conservação da biodiversidade no Estado do Paraná. Ph.D. thesis, Engenharia Florestal, Universidade Federal do Paraná.

SOS MA, S. M. A., 2018. Desmatamento na mata atlântica é o menor registrado desde 1985. Accessed: August 13 2018.
URL `https://www.sosma.org.br/projeto/atlas-da-mata-atlantica/dados-mais-recentes/`

TNC, The Nature Conservancy, n.d. ICMS Ecológico. Accessed: January 17 2017.
URL `http://www.icmsecologico.org.br/site/`

Chapter 3

Political cycles drive deforestation in the Brazilian Atlantic Forest

Patricia G. C. Ruggiero[1], Alex Pfaff[2], Liz Nichols[3], Marcos Rosa[4], and Jean Paul Metzger[1]

[1]Department of Ecology, University of São Paulo, BR

[2]Sanford School of Public Policy, Duke University, US

[3]Bioscience Department, Swarthmore College, US

[4]Department of Geography, University of São Paulo, BR

"Uma eleição é feita para corrigir o erro da eleição anterior, mesmo que o agrave."

Carlos Drummond de Andrade

(O avesso das coisas – Aforismos.

1997. Record, 4a. edição.)

3.1 Abstract

Political electoral incentives may be a significant driver of deforestation, yet have largely been neglected in the scientific literature. Although the Brazilian Atlantic Forest region benefits from an exceptional amount of conservation regulation and finance, we demonstrate that deforestation linked to political election cycles occurs across the region. We combined fine-scaled annual deforestation data from 1991 to 2014 for 2,277 municipalities in seven Brazilian states, and information on state and municipal election results to explore the temporal relationship between deforestation and political cycles. We found a significant increase in annual municipality deforestation rates during state election years (0.027% ($\pm$ 0.008), or roughly an addition of 4,000 hectares of deforestation due to election in the observed region. Municipal elections also contributed to deforestation in at least three states. Deforestation is particularly exacerbated when incumbents face a very probable defeat; a trend visible for all state elections (0.109% $\pm$ 0.013), and for municipal elections in two states (Rio de Janeiro 0.343 $\pm$ 0.121 and Rio Grande do Sul 0.117 $\pm$ 0.025). Our results also indicate that the overall effect of municipal elections on deforestation may be mediated by party alignment between municipal and state governments. We demonstrate that elections are an important driver of tropical deforestation, even in relatively strong environmental governance contexts and recommend enhanced research and monitoring of the impacts of political cycles on tropical forest dynamics in conservation science and practice.

3.2 Introduction

For decades, conservation biologists and economists have studied the multiple drivers of deforestation, historically focusing on economic factors, institutions, national policies and remote influences driving agricultural expansion, wood extraction and infrastructure extension (Geist and Lambin, 2002). Critically missing from this body of work is role of political elections on deforestation. Elections create conditions that generate politically motivated decisions, and are generally related to what is referred to as political cycle - public intertemporal choice, where decisions are made within a political framework (Nordhaus, 1975). Within this framework, policy instruments related to economic and social outcomes, such as inflation and unemployment, are often manipulated cyclically, as a consequence of these political constrains (Shi and Svensson, 2006; Brender and Drazen, 2005). However, only recently it has been demonstrated that political cycles affect natural resources (Burgess et al., 2012). Understanding underlying political drivers of deforestation is particularly critical in the tropics, where deforestation and forest degradation contributes to 7-14% of the total global carbon emissions from human activities (Harris et al., 2012), and threatens a majority of the world's biodiversity (Venter et al., 2014).

Several political motivations may mediate cyclic outcomes in electoral processes, including the need for voter support, to obtain funds for campaign and also corruption. Because elected officials already in an established position of power may have more facile access to levers of political power, the relationship between campaign needs and corruption may be more clear for political incumbents (i.e. those seeking re-election). Political decisions have their highest weight right before elections and lowest right after, as voters have a decaying memory of past events (Nordhaus, 1975). Thus, seeking voter support through political decisions is more common when elections are close, while impopular decisions tend to be taken in the first year of the term (Nordhaus, 1975). During elections, candidates need funds for campaign so rent-seeking, i.e. rent that is obtained through the change of rules, and not through production of real goods and services (Black et al., 2017), might be one of an incumbents' strategies to raise campaign funds (Samuels, 2008). Finally, upcoming elections increase the effective discount rate of local officials, and in case an incumbent may lose their position, they might choose to take advantage of this current position rather than to leave it for later (Burgess et al., 2012). Thus, distinct from rent-seeking and opportunism, corruption activity - i.e. obtaining private gains from public office through bribes, extortion and embezzlement of public funds (Black et al., 2017), is related to political cycle, especially when it is unlikely that misbehavior will be detected (Pereira et al., 2009).

Political cycles and forest resources may be linked by the liquidation of forest products (leading to deforestation) as a mode of either voter support, rent-seeking or corruption(Burgess et al., 2012; Pailler, 2018). For example, Burgess et al. (2012) demonstrated a clear "political logging cycle" in Indonesia, in which illegal logging estimated from 2001 to 2008 increased by 29% two years prior to the election, by 42% in the year before the election, then fell by 36% during the election year, without resuming thereafter. The authors posit these deforestation patterns to be driven efforts to transform forest resources into voting support or rent-seeking, either by permitting illegal logging or intentionally reducing enforcement. Pailler (2018) recently reported on a similar election-natural resource link in the Brazilian Amazon, where deforestation increased 8-10% in municipal election years, when an incumbent mayor runs for re-election in comparison to municipalities where the incumbent is not disputing the election ("no incumbents" condition; Pailler, 2018). Although the author did not measure illegal deforestation directly, highly corrupt incumbents showed approximately 50% more deforestation when compared to municipalities with no incumbents running for re-election. It is worth noticing that Brazil and Indonesia produced the highest carbon emissions from deforestation between 2000 and 2005, accounting for 55% of total emissions from tropical deforestation (Harris et al., 2012).

While it was initially advocated that these opportunistic cycles are more evident and significantly larger in developing countries (Shi and Svensson, 2006), the magnitude of these cycles seems better related to the presence of "new democracies" and their consequent new electoral systems (Brender and Drazen, 2005). A possible explanation for this is the increased ability of voters to distinguish between pre-electoral manipulations and official's competence, as independent media and civil society develops, raising transparency, and resulting opportunists incumbents being punished in the ballot instead of rewarded (Brender and Drazen, 2005; **?**).

Brazil seems potentially vulnerable to political deforestation cycles, given its relatively new democracy with national and state elections reestablished in 1986, and extensive forest resource coverage. Brazil also has a notoriously corrupted electoral and campaign financing system (Watts, 2017; Samuels, 2008). Within Brazil, the Atlantic Forest is one of the richest forests in the world under one of the most globally stringent forest legislations, regarding the National Forest Act (Law 12,651/2012) and the Atlantic Forest law (Law 11,428/2006). It is also the most populated region with the strongest economy in the country, encompassing states that concentrate 52.7% of the national GDP - São Paulo (32.6%), Rio de Janeiro (11.2%) and Minas Gerais (9.3%). State institutions in this region have extensive personal, financial and technical resources to implement environmental policies and to promote legislation enforcement. State governments host Environmental Secretariats who are in charge of: i) the concession of state licenses (for instance, to deforestation and to industry settlement) (federal law 11,428/2006), ii) management of state protected areas and iii) the implementation of state environmental policies and programs. The state government also commands the forest police, linked to state civil police, who is mainly responsible for environmental surveillance. Although municipalities are not explicitly mentioned in the Brazilian constitution as responsible for the environment, they can legislate on local interests (Banerjee et al., 2009), once municipal legislation does not conflict with state and federal legislation. Depending on how organized municipalities are, they may present an Environmental Secretariat and an Environmental Committee usually in charge of municipal environmental programs and municipal conservation units management. In most cases, municipalities do not have those institutions and have little action towards the environment, particularly regarding forest and biodiversity conservation.

We investigated the existence of a political deforestation cycle related to state and municipal elections in the Brazilian Atlantic Forest region (Fig. S1). We explore three main hypotheses. First, we hypothesize that election years witness higher deforestation than no election years, possibly as a consequence of a political deforestation cycle. Although political deforestation cycles have been described for local elections in the Amazon and in Indonesia, we consider that a political cycle related to state elections is possible since state government has the main responsibilities with respect to the environment and play a stronger regulatory and enforcement role in forest management and deforestation control than federal and municipal institutions in this region. Second, rent-seeking behavior expected to take place in election years may be enhanced, weaken or even only exist depending on certain conditions regarding the election dispute. For instance, in close elections, the value of an additional bad behavior may increase at the margin, leading to corruption increase (Pereira et al., 2009). Thus, we tested if election competitiveness, i.e. close races (small difference of votes between opponents) and safe races (large difference), influence deforestation outcomes. Third, political alignment generates distortions on the allocation of intergovernmental transfers, resultant of officials benefitting political allies or punishing their opponents (Brollo and Nannicini, 2012). Parties governing the state office might co-opt municipal elections as a mechanism to consolidate influence via local governance. Extending this rationale to deforestation process, we expect the alignment between state and municipal parties to enhance deforestation in this municipalities during municipal elections.

3.3 Data and Methods

3.3.1 Study region

The Atlantic Forest is one of the most biodiverse and endangered forests in South America, with excepcional high levels of species endemism and plant diversity per unit area, being higher than most of the Amazon forests (Joly et al., 2014). The Atlantic Forest has been largely degraded as a result of a long history of human occupation and land use transformation. It is the most populated region with the strongest economy in Brazil - 70% of the Brazilian population lives in the Atlantic Forest and is responsible for 80% of the national GDP (MMA, Ministério do Meio Ambiente, n.d.). This dense and old human occupation threatens the survival of Atlantic Forest's unique biodiversity through the promotion of an intense forest loss and fragmentation across the whole region, with fragments smaller than 100 hectares corresponding to 40% of the remaining vegetation (Ribeiro et al., 2009). Similar to other tropical forests in the world, deforestation in the Atlantic Forest has contributed to the increase of carbon loss and consequent climate change, however fragmentation processes contribute to an additional 9-24% of the annual carbon loss due to deforestation (Pütz et al., 2014).

3.3.2 Data

We gathered deforestation information at the municipal level and election results, for state and municipal elections, to build a longitudinal database with 2277 municipalities across seven states located inside the Atlantic Forest region limit (Minas Gerais, Rio de Janeiro, Espírito Santo, São Paulo, Paraná, Santa Catarina and Rio Grande do Sul) for the period from 1991 to 2014.

Deforestation

We used land cover data from MapBiomas project 3$^{\text{rd}}$ collection (MapBiomas, 2018). MapBiomas provides annual, cloud-free and automatically classified land cover data based on Landsat images, at a 30m spatial resolution, covering 1985 to 2017. We discarded the three initial and final years of the data series due to possible mapping inaccuracy (Marcos Rosa, pers. comm.). We considered a total of 2,277 municipalities from seven states - Minas Gerais, Espírito Santo, Rio de Janeiro, São Paulo, Paraná, Santa Catarina and Rio Grande do Sul. We obtained forest transition matrices from the annual mapping product, distinguishing deforested area at the pixel scale. Transition matrices are useful to properly quantify changes in a landscape (Lambin et al., 2003), specially because a lack of net change does not necessarily indicate a lack of change on the landscape since swapping may occur (Pontius Jr. et al., 2004). We represented deforestation as the percentage lost of previous forested area, per municipality.

Elections

The elected term of all Brazilian federal, state, and municipal officials is four years, and re-election is allowed for one additional term. Presidential and state elections occur in the same year, with municipal elections offset by two years (Fig. S1). State elections and municipalities with more than 200,000 population may have a two-round election in case no candidate receives more than 50% of valid votes. Electoral data for the whole of Brazil is available online at *Tribunal Superior Eleitoral* (National Electoral Office). We collected state and municipal election results for all municipalities, from 1991 to 2014 (TSE, n.d.). We obtained the percent of valid votes for each candidate always from the first round of the election, and used these election results as a proxy of race competitiveness. In those cases where a second election round occurred, we collected the identity and party of the final winner. The first two observed municipal elections (1992 and 1996) were not considered for party alignment analysis because information about municipal leaders in office between 1988 and 1992 are diffuse and not available from a single accessible source. For each of the 30 municipalities created between 2000 and 2010, we searched for individual information about municipality dismemberment to obtain the government in power in this period.

Control variables

Previous studies on deforestation patterns have suggested that productive rural activities and human population growth may play an important role in deforestation outcomes **??**. Data about agricultural production, cattle breeding and human population was obtained from the *Instituto Brasileiro de Geografia e Estatística (IBGE)* (National Institute of Geography and Statistics) through its online database SIDRA (IBGE, n.d.). Data from 1990 up to 2014 was collected and transformed to current US dollars. We updated values from its original currency (Cruzados, Cruzados Novos, Cruzeiros, Cruzeiros Reais and Reais) to corrected US dollars, using dollar quotation for Brazilian currencies for the last day of each year, provided by the Brazilian Central Bank, and inflation information (based on Consumer Price Index), provided by the Bureau of Labor Statistics, US government. For the same period, data for the number of cattle per municipality was also obtained and divided by the total municipal area to generate an average cattle density. Human population was divided by the total municipal area to obtain an average of population density. Squared population density was also used to capture possible decreasing or increasing marginal effects of this variable over deforestation. Annual average precipitation per municipality, obtained from CHIRPS through the Columbia University database (IRI, International Research Institute for Climate and Society, n.d.), was used to control for possible Landsat mapping residual variation related to differences between dry and rainy years.

3.3.3 Empirical approach

In order to investigate the existence of a political deforestation cycle in the Atlantic Forest, we first searched for general patterns regarding the political cycle years. We defined a categorical variables named *cycle* with three levels, stipulating if the election year pertain to a municipal, or state election, or an inter-election period.To test the idea that election years result in significantly higher deforestation than no election years, we used the general equation:

$$y_{m,t} = \theta_t + \beta_1\, cycle_{m,t} + \beta_2\, X_{m,t} + \alpha_m + \epsilon_{m,t} \tag{3.1}$$

where y is the deforestation outcome expressed as the percentage of previous existent forest in each municipality m in each year t; *cycle* is the categorical variable identifying the occurrence of election in t year; X are control variables; θ denotes time effect and ϵ is the error term.

To test for the effect of race competitiveness, we used election results as a proxy of pre-contest party expectations of success. It is important to note that while in many studies, the incumbent is considered the individual candidate regardless of party affiliation, here the incumbent is considered the party regardless of candidate, given the frequency of party switching by Brazilian candidates. In the case of state elections, we also considered the party as an incumbent when it is part of a coalition running for election. Additionally, term-limits apply to individual candidates, but not parties. We calculated the voting difference between the two candidates as the fraction of votes received by the incumbent party minus the fraction of votes received by their strongest opponent, in the first round of the election (*margin* variable). When the *margin* of votes is positive, the incumbent wins the election and stays in power; when the *margin* of votes is negative, the party in power changes. We found this did not occur in approximately 9% of all observed election events, when the result in the second round was the opposite of the first round. We classified *margin* variable into four *range* categories: close win (0 $<$ margin $<$ 10), close loss (-10 $<$ margin $<$ 0), large win (margin $>$ 10) and large loss (margin $<$ -10). We used non election years as the reference level for *range* variable. We then regressed deforestation as a function of *range* variable using the general equation (1), for state and municipal elections separately.

To test our third hypothesis that municipal elections have an effect on deforestation when there is an alignment between the party in the state office and the party running for re-election in the municipality, we used a binary variable that identifies any election event with party alignment, for state and municipal elections separately. In this case, we run panel regressions only for municipal elections considering the general equation (1) but adding election years as a control variable to account for the effect of elections alone, as follows:

$$y_{m,t} = \theta_t + \beta_1\, friend_{m,t} + \beta_2\, election_{m,t} + \beta_3\, X_{m,t} + \alpha_m + \epsilon_{m,t} \tag{3.2}$$

where $friend$ is the party alignment and *election* identifies municipal election year.

Finally, as the observed region is geographically extensive, and inclusive of a range of cultural and administrative differences, we also ran all panel regressions for each state separately. We controlled all models for municipality fixed effects and time trend (using PLM package function argument), as deforestation rates may vary across time (Fig. S2). All procedures were conducted in R (Version 3.4.2.) (R Core Team, 2017), using the PLM package (Version 1.6-6) for panel regression analyses (Croissant et al., 2008).

3.4 Results

3.4.1 Political cycle years

For all municipalities together, total deforestation significantly increases during state elections, when compared to non election years (Table 3.1). However, a mixed pattern is observed when panel regressions are executed for states separately. State election years show significant increase in deforestation in Paraná and Santa Catarina, while municipal elections increase deforestation in Rio Grande do Sul and Santa Catarina (Table 3.2). Overall deforestation in election years is not significantly different from non election years in the other states. We estimate a total impact of state elections and municipal elections to be respectively 4,117 and 3,855 hectares per election year, across the whole observed region (Table 3.1).

Table 3.1: Regression coefficients (standard errors) of municipal and state election years, compared to non election in-between years, on total deforestation, for all states together. Control variables coefficients are also presented below.

DV: Total deforestation	In % of previous forest	In hectares
Municipal elections	-0.004 (0.008)	1.693. (0.909)
State elections	0.027*** (0.008)	1.808* (0.916)
Control variables:		
Population density	< 0.001 (< 0.001)	0.003 (0.006)
Squared population density	< 0.001 (< 0.001)	< 0.001 (< 0.001)
Agricultural production	< 0.001** (< 0.001)	< 0.001** (< 0.001)
Cattle density	-0.001*** (< 0.001)	-0.092*** (0.025)
Precipitation	-0.001*** (< 0.001)	-0.068*** (0.020)
Time trend	-0.008*** (0.001)	-0.522*** (0.062)

***p < 0.001, **p < 0.01, *p < 0.05, .p < 0.1

Table 3.2: Regression coefficients (standard errors) of municipal and state election years, compared to non election in-between years, on total deforestation, for states separately. All models have control variables (not shown).

DV: Total deforestation (% of previous forest)	State	Municipal
Minas Gerais (MG)	0.008 (0.017)	-0.018 (0.017)
Rio de Janeiro (RJ)	0.053 (0.036)	0.049 (0.035)
Espírito Santo (ES)	-0.046 (0.046)	0.037 (0.045)
São Paulo (SP)	0.021 (0.026)	-0.029 (0.025)
Paraná (PR)	0.061*** (0.018)	0.005 (0.017)
Santa Catarina (SC)	0.041* (0.018)	0.041* (0.018)
Rio Grande do Sul (RS)	-0.006 (0.009)	0.022** (0.008)

***$p < 0.001$, **$p < 0.01$, *$p < 0.05$, .$p < 0.1$

3.4.2 Race competitiveness

Both state and municipal election events show positive and negative values of *margin* equally distributed (Fig. S2, A and B). In municipal election events, the incumbent wins with 100% of the votes in approximately 15% of the events (Fig. S2B), from which 93% of the cases it was only one candidate running for the election. For most of state elections, deforestation presents higher and significant values when incumbent is losing by a large margin of votes (>10) (Table 3.3). In this case, the analysis of race competitiveness evidenced the effect of state elections in Minas Gerais, São Paulo and Rio de Janeiro, which was not clear when overall election years were analyzed. For municipal elections, losing by a lot is also a strong condition raising deforestation in Rio de Janeiro and Rio Grande do Sul.

3.4.3 Party alignment

Party alignment showed an effect on total deforestation increase during municipal elections in the state of Rio Grande do Sul, where municipalities with incumbents from a political party currently in power at the state government level are associated with almost five times more deforestation for municipalities with no alignment (Table 3.4).

Overall, we observed low annual deforestation percentages across all the Atlantic Forest region, with mean values for municipalities of less than 1% in all states (Table C1 and Fig. C2, Supplementary material).

Table 3.3: Regression coefficients (standard errors) of the effect of race competitiveness on total deforestation, for all states together and states separately. All models have control variables (not shown).

DV: Deforestation	State				Municipal				
(% of previous forest)	Losing close	Winning close	Losing by a lot	Winning by a lot	Losing close	Winning close	Losing by a lot	Winning by a lot	No incumbent
All states	0.012	-0.057**	0.109***	0.003	0.016	0.001	-0.017	0.003	-0.022*
	(0.016)	(0.020)	(0.013)	(0.029)	(0.024)	(0.023)	(0.022)	(0.019)	(0.010)
Minas Gerais (MG)	-0.023	-	0.153***	-0.052*	0.027	-0.053	-0.054	0.061	-0.034.
	(0.034)		(0.027)	(0.021)	(0.048)	(0.047)	(0.042)	(0.039)	(0.020)
Rio de Janeiro (RJ)	0.547***	-	-0.061	-0.104*	0.089	0.261*	0.343***	0.002	-0.040
	(0.073)		(0.051)	(0.049)	(0.121)	(0.117)	(0.089)	(0.085)	(0.040)
Espírito Santo (ES)	-	-	-0.138*	0.019	-	-	-0.515***	-0.261.	
			(0.059)	(0.057)			(0.154)	(0.153)	
São Paulo (SP)	-	-0.117*	0.204***	0.023	0.038	-0.133	-0.096	-0.074	-0.009
		(0.052)	(0.055)	(0.029)	(0.086)	(0.082)	(0.061)	(0.055)	(0.030)
Paraná (PR)	0.015	-0.025	0.156***	0.066.	0.017	0.105.	0.005	0.020	-0.037.
	(0.037)	(0.026)	(0.026)	(0.039)	(0.060)	(0.057)	(0.052)	(0.043)	(0.019)
Santa Catarina (SC)	-0.047	0.011	-0.064	0.090***	-0.017	0.029	0.021	0.037	0.038.
	(0.038)	(0.036)	(0.039)	(0.024)	(0.042)	(0.044)	(0.057)	(0.044)	(0.021)
Rio Grande do Sul (RS)	-0.031**	-	0.021.	-	0.034	0.063**	0.117***	0.023	0.003
	(0.010)		(0.012)		(0.021)	(0.020)	(0.025)	(0.019)	(0.010)

***$p < 0.001$, **$p < 0.01$, *$p < 0.05$, .$p < 0.1$

Table 3.4: Same party in municipal elections effect on deforestation for the whole observed period.

DV: Deforestation (% of previous forest)	Municipal elections	Party alignment
Minas Gerais (MG)	0.008 (0.019)	0.039 (0.054)
Rio de Janeiro (RJ)	0.090* (0.036)	0.205 (0.142)
Espírito Santo (ES)	0.091. (0.048)	-0.046 (0.126)
São Paulo (SP)	-0.039 (0.026)	-0.016 (0.090)
Paraná (PR)	0.055** (0.019)	0.024 (0.041)
Santa Catarina (SC)	0.026 (0.023)	0.022 (0.039)
Rio Grande do Sul (RS)	0.015 (0.013)	0.070** (0.022)

***$p < 0.001$, **$p < 0.01$, *$p < 0.05$, .$p < 0.1$

3.5 Discussion

Our results suggest that political deforestation cycles exist in the Brazilian Atlantic Forest. The deforestation cycle is strongest for state elections, but also occurs in municipal races, and the dynamics of these cycles demonstrate key differences across the observed states. We also found that deforestation is significantly enhanced when the incumbent is losing the election dispute, usually by a large difference in the margin of votes. Finally, a party alignment between mayor and state government in office may significantly enhance deforestation during municipal elections, as demonstrated for the state of Rio Grande do Sul.

To disclose and understand the effect of both state and municipal elections on deforestation in our region, we need to jointly analyze results from election years and race competitiveness, as the latter is essentially another way of looking to elections, in a more detailed fashion. For instance, if close elections increase deforestation while safe elections have the opposite effect, one offsets the other, hiding the real dispute effect, when overall election years are observed. So, using the margin of votes to decouple election events certainly helps to elucidate the electoral process effect. When observing election years, the states from southern Brazil – Paraná, Santa Catarina and Rio Grande do Sul – showed higher deforestation values. Paraná and Santa Catarina showed an effect of state elections on deforestation, while Santa Catarina and Rio Grande do Sul presented an effect of municipal elections, regardless of race competitiveness. However, for the southeastern states with the strongest economies in the country – São Paulo, Minas Gerais and Rio de Janeiro – the effect of elections, both state and municipal, are only evidenced when race competitiveness is observed. São Paulo, Minas Gerais, Rio de Janeiro and possibly Rio Grande do Sul also present an state election effect, while Rio de Janeiro show a municipal election effect on deforestation. Espírito Santo did not present a clear pattern, neither for state nor municipal elections.

Political deforestation cycles has been already demonstrated for the Brazilian Amazon (Pailler, 2018). Different from the Amazon, where forest resources are still abundantly available with dynamic timber extraction economy and advancing agricultural frontiers, the Atlantic Forest region has an old and dense human occupation that has already modified region's natural environment (Joly et al., 2014). Agricultural frontiers for most of the studied region are well established since the 90's; however, opening area for agricultural expansion still undergoes (Freitas et al., 2010) within a smaller extension when compared to the Amazon. In addition, the Atlantic Forest has among the most stringent environmental laws applied to tropical forests (Law 11,428/2006). This ecological-socio-economic and legal context of the Atlantic Forest states explains the low values of deforestation.

In the Atlantic Forest region, state governments are the main actors in charge of emitting native vegetation logging permits, that are only delivered in cases of public utility and social interests, particularly when it regards vegetation under advanced or secondary regeneration stages (federal law 11.428/2006). In addition, although the federal government institution IBAMA is responsible for national deforestation policies and control, state institutions, such as Environmental Secretaries and Environmental Police, also actively monitor and invigilate illegal environmental activities, including deforestation. Certainly São Paulo, Minas Gerais and Rio de Janeiro, but also Paraná and Santa Catarina, have stronger state governments with more institutional, financial and technical resources available for environmental police and surveillance. Thus, it is plausible that state elections are linked to deforestation in these states, once they have strong control over forest governance. In the Amazonian region, states strongly depend on federal resources and federal institutions to implement state responsibilities, particularly regarding the environment. If state control is debilitated, municipal governments have more opportunity to dispose of forest resources in their favor. This explains the existence of a deforestation cycle related to municipal elections in the Amazon, described by Pailler (2018). However, in our study, municipal elections affect deforestation in states with strong state governance, which in the case of Rio de Janeiro and Santa Catarina that present also a political deforestation cycle related to state elections.

Deforestation accrued in elections may be the result, or a side effect, of a more complex net of exchanged support and benefits, legal or illegal. Burgess et al. (2012) suggested that reducing environmental enforcement and allowing people to illegally deforest can directly praise voters, both timber sellers and agricultural producers, in Indonesia. In the Brazilian Amazon, municipalities with corrupt incumbents show 40 - 60% higher deforestation, when compared to other municipalities (Pailler, 2018). A large number of Brazilian parties have weak organization, lack of coherent programmatic positions and no electoral national organizations, and rely on expensive campaigns to attract voters (Samuels, 2008). Rent-seeking and corruption activities seems to be an established strategy to obtain campaign funds (Watts, 2017). In Brazil, local governments largely engage in corruption activities(Ferraz and Finan, 2011; Pereira et al., 2009), but also state elections have already been shown as the scene for vote brokerage electoral strategy - paying local brokers to generate an increase in votes to the incumbent governor (Gingerich et al., 2014).

There is some controversy in the literature whether re-election incentives enhances opportunistic behavior. When running for re-election, incumbents may reduce misbehavior to avoid being punished by voters in the ballot, in comparison to lame-duck mayors facing a shorter political horizon (Ferraz and Finan, 2011). However, when chances of bad behavior being detected is low and information on candidates' corruption is not available to voters, re-election incentives politicians to fiercely engage in corruption, particularly when races are competitive, as demonstrated for the state of Pernambuco, Northeast Brazil (Pereira et al., 2009). Extending this rationale, deforestation could also be expected to increase during close races, while safe races could lack motivation for rent-seeking. Close races, winning or losing, only affected deforestation in Rio de Janeiro, in both election levels, and in Rio Grande do Sul, in municipal elections.

Consistently to our expectations, winning by a lot of votes is the situation where deforestation during elections is smaller or not different from years with no elections, except for Paraná and Santa Catarina state elections (Table 3.3). On the other hand, when the incumbent is losing by a lot of votes, deforestation increases significantly in three and two states, respectively during state and municipal elections. Safe elections, when the incumbent is certain about losing the position, is possibly related to parties taking the opportunity to get funds for future electoral campaigns. As Burgess et al. (2012) highlighted, the incumbent effective discount rates might increase when they expect to lose the re-election, in other words the opportunity to collect bribes from the selling trees in the future may end and so they may wish to sell them now. We suggest the expectation of losing is an important condition for opportunistic behavior - when losing close, the opportunistic behavior has a higher marginal value because one additional misbehavior may bring the victory (Pereira et al., 2009); when losing by a lot, the incumbent discount rate increases as the party is certainly leaving the office and the opportunity to bribe will end (Burgess et al., 2012).

State governments can manipulate deforestation in order to win municipal elections aiming at keeping important allies in municipal offices. Municipalities in Brazil are more likely to receive voluntary transfers from state governments if the city mayor belongs to the same party as the governor, and those transfers influence preference of voters(Bugarin, 2006). Federal transfers to municipalities can also be operated for political reasons - they significantly benefit municipalities run by a mayor belonging to the same party as the president, and they not only praise voters and strengthen important allies, but punish municipalities run by the opposition (**?**). We suggest that a similar mechanism can operate with deforestation. Rio de Janeiro showed a marked increase in deforestation when municipal and state parties are the same in municipal elections; but it is not significant. We detected this strategy in Rio Grande do Sul, where the effect of party alignment is significant while the effect of municipal elections alone is not. This result indicates that the overall effect of municipal elections can be attributed to party alignment condition and that the increase in deforestation during local elections may be in fact mediated by the state government.

The apparent evidence that the links between elections and deforestation occur for both national/state and mayoral elections, is of concern meaning that every two years in Brazil presents an opportunity for forest lost due to politically motivated rent-seeking and corruption. It is also possible that the conservation gains accomplished in the intra-election cycles are offset by these election-deforestation drivers. The political literature describes that political cycles get less intense as society experiences a learning process in democracy with accumulated electoral events, but exclusively when information is processed and made available to voters (Brender and Drazen, 2005; Akhmedov and Zhuravskaya, 2004). This highlights the importance of reliable institutions monitoring deforestation annually and making this data available to voters, so that they can have the choice to punish instead of inadvertently reward this type of behavior.

3.6 Conclusion

We demonstrate that there is a political deforestation cycle in the Brazilian forested region with arguably the most stringent environmental regulation, the majority of the population, and high conservation investments (Joly et al., 2014). Jointly with another recent study (Pailler, 2018), our results show that incentives generated by both national/state and municipal elections affect forest resources, representing a high frequency and intense pressure over this natural and already very threatened resource. As a consequence, forests are also subjected to candidate's and party's political strategies, such as rewarding political allies with deforestation increase. These links are a gap in current conservation practice, which instead broadly focuses on environmental legislation and enforcement.

3.7 Acknowlegments

We demonstrate that there are political deforestation cycles in the Brazilian forested region with arguably the most stringent environmental regulation, the majority of the population, and high conservation investments (Joly et al., 2014). Jointly with another recent study (Pailler, 2018), our results show that incentives generated by both national/state and municipal elections affect forest resources, representing a high frequency and intense pressure over this natural and already very threatened resource. As a consequence, forests are also subjected to candidate's and party's political strategies, such as rewarding political allies with deforestation increase. These links cannot be any more disregarded, and should be considered in future policy and conservation practices, in addition to other environmental conservation mechanisms.

Bibliography

Akhmedov, A., Zhuravskaya, E., 2004. Opportunistic political cycles: test in a young democracy setting. The Quaterly Journal of Economics, 1301–1338.

Banerjee, O., Macpherson, A. J., Alavalapati, J., 2009. Toward a policy of sustainable forest management in brazil: a historical analysis. The Journal of Environment & Development 18 (2), 130–153.

Black, J., Hashimzade, N., Myles, G., 2017. A dictionary of economics. Oxford University Press.

Brender, A., Drazen, A., 2005. Political budget cycles in new versus established democracies. Journal of monetary Economics 52 (7), 1271–1295.

Brollo, F., Nannicini, T., 2012. Tying Your Enemy's Hands in Close Races: The Politics of Federal Transfers in Brazil. American Political Science Review 106 (04), 742–761.

Bugarin, M. S., 2006. Political budget cycles in a fiscal federation: The effect of partisan voluntary transfers.

Burgess, R., Hansen, M., Olken, B. A., Potapov, P., Sieber, S., 2012. The political economy of deforestation in the tropics. The Quarterly Journal of Economics 127 (4), 1707–1754.

Croissant, Y., Millo, G., et al., 2008. Panel data econometrics in r: The plm package. Journal of Statistical Software 27 (2), 1–43.

Ferraz, C., Finan, F., 2011. Electoral accountability and corruption: Evidence from the audits of local governments. American Economic Review 101 (4), 1274–1311.

Freitas, S. R., Hawbaker, T. J., Metzger, J. P., 2010. Effects of roads, topography, and land use on forest cover dynamics in the Brazilian Atlantic Forest. Forest Ecology and Management 259 (3), 410–417.

Geist, H. J., Lambin, E. F., 2002. Proximate Causes and Underlying Driving Forces of Tropical Deforestation. BioScience 52 (2), 143–150.

Gingerich, D. W., et al., 2014. Brokered politics in brazil: An empirical analysis. Quarterly Journal of Political Science 9 (3), 269–300.

Harris, N. L., Brown, S., Hagen, S. C., Saatchi, S. S., Petrova, S., Salas, W., Hansen, M. C., Potapov, P. V., Lotsch, A., 2012. Baseline map of carbon emissions from deforestation in tropical regions. Science 336 (6088), 1573–1576.

IBGE, I. B. d. G. e. E., n.d. Sistema ibge de recuperação automática.
URL `https://sidra.ibge.gov.br/home/pimpfbr/brasil`

IRI, International Research Institute for Climate and Society, n.d. The IRI/LDEO Climate Data Library.
URL `https://iri.columbia.edu/resources/data-library`

Joly, C. A., Metzger, J. P., Tabarelli, M., 2014. Experiences from the Brazilian Atlantic Forest: ecological findings and conservation initiatives. New Phytologist, n/a–n/a.

Lambin, E. F., Geist, H. J., Lepers, E., 2003. Dynamics of land-use and land-cover change in tropical regions. Annual review of environment and resources 28 (1), 205–241.

MapBiomas, 2018. Mapbiomas project collection 3.0 of brazilian land cover & use map series. Accessed: June 28 2018.
URL `http://mapbiomas.org`

MMA, Ministério do Meio Ambiente, n.d. Área da mata atlântica é habitada por 70% da população brasileira.
URL `http://www.mma.gov.br/informma/item/9818-area-da-mata-atlantica-e-habitada\-por-70-da-populacao-brasileira`

Nordhaus, W. D., 1975. The political business cycle. The review of economic studies 42 (2), 169–190.

Pailler, S., 2018. Re-election incentives and deforestation cycles in the Brazilian Amazon. Journal of Environmental Economics and Management 88, 345–365.

Pereira, C., Melo, M. A., Figueiredo, C. M., 2009. The corruption-enhancing role of re-election incentives? counterintuitive evidence from brazil's audit reports. Political Research Quarterly 62 (4), 731–744.

Pontius Jr., R. G., Shusas, E., McEachern, M., 2004. Detecting important categorical land changes while accounting for persistence. Agriculture, Ecosystems & Environment 101 (2-3), 251–268.

Pütz, S., Groeneveld, J., Henle, K., Knogge, C., Martensen, A. C., Metz, M., Metzger, J. P., Ribeiro, M. C., de Paula, M. D., Huth, A., 2014. Long-term carbon loss in fragmented Neotropical forests. Nature Communications 5, 1–8.

R Core Team, 2017. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.
URL `https://www.R-project.org/`

Ribeiro, M. C., Metzger, J. P., Martensen, A. C., Ponzoni, F. J., Hirota, M. M., 2009. The Brazilian Atlantic Forest: How much is left, and how is the remaining forest distributed? Implications for conservation. Biological Conservation 142 (6), 1141–1153.

Samuels, D., 2008. Money, elections and democracy in Brazil. Latin American Politics and Society 43 (2), 27–48.

Shi, M., Svensson, J., 2006. Political budget cycles: Do they differ across countries and why? Journal of public economics 90 (8-9), 1367–1389.

TSE, T. S. E., n.d. Tribunal superior eleitoral - eleições anteriores.
URL `http://www.tse.jus.br/eleicoes/eleicoes-anteriores/eleicoes-anteriores`

Venter, O., Fuller, R. A., Segan, D. B., Carwardine, J., Brooks, T., Butchart, S. H., Di Marco, M., Iwamura, T., Joseph, L., O'Grady, D., et al., 2014. Targeting global protected area expansion for imperiled biodiversity. PLoS Biology 12 (6), e1001891.

Watts, J., 2017. Operation car wash: Is this the biggest corruption scandal in history? The Guardian.
URL `https://www.theguardian.com/world/2017/jun/01/brazil-operation-car-wash-\is-this-the-biggest-corruption-scandal-in-history`

Supplementary material

C.1 Brazil's Democracy

Brazil is a relatively new democracy, with national and state direct elections reestablished in the 1980s, after 21 years of a military government. During the military government, elections were indirect for the positions of president of the republic, state governors and mayors of state capitals, hydromineral resorts and municipalities considered areas of national security (for example, municipalities bordering other countries). A system of universal and direct voting returned in 1986 for state and federal elections, and 1988 for local (municipal) elections. The elected term of all federal, state, and municipal officials is four years, and re-election is allowed for one additional term. Presidential and state elections occur in the same year, with municipal elections offset by two years (Fig. C.1).

C.2 Federal environmental governance

The current Brazilian constitution, approved in 1988, defines that both federal and state governments are in charge of developing and implementing environmental legislation. The federal government conceive and implement national policies and programs through the activity of its three main organizations: the *Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais Renováveis* (IBAMA), the *Instituto Chico Mendes de Conservação da Biodiversidade* (ICMBio) and the Ministry of the Environment (MMA). Both IBAMA and ICMBio are under the supervision of the Ministry of the Environment. IBAMA is the largest environmental organization in the country and is responsible for controlling the production and commercialization of natural products, monitoring and surveilling native forest cover (with fines application), authorizing national environmental licenses, among other less visible activities. The ICMBio is responsible for biodiversity monitoring and conservation and is also in charge of the national protected areas management. Finally, the Ministry of the Environment is responsible for the definition and implementation of national policies and programs.

C.3 About MapBiomas

MapBiomas project is an annual mapping of soil coverage and land use initiative in Brazil that involves a collaborative network of specialists in biomes, land uses, remote sensing, geographic information systems (GIS) and computer science, using cloud processing and automated classifiers developed and operated from the Google Earth Engine platform (MapBiomas, 2018). MapBiomas aim at the development of an automated methodology for the production of annual detailed mappings of the whole Brazil for long historic series (from 1985 to 2017) that allows the unprecedented monitoring of changes in land use and occupation. Consequently, the dynamics of deforestation, forest fragmentation and regeneration can be analyzed and understood in novel ways. The MapBiomas project has four main characteristics. All land use classification is based on Landsat images. Secondly, a pixel-per-pixel processing generates a 30m spatial resolution information, without degradation. Processing is also organized by Landsat images (1: 250,000) and biome teams are responsible for the classification of natural formations, while transversal teams are responsible for the classification of pasture, agriculture, urban areas and coastal zones. Finally, all cloud processing is automated, classification is supervised and there is no manual editing of final product.

C.4 Data description

Deforestation as the percentage of previous forest for all municipalities is summarized in Table C.1 and Figure C.1.

Tabela C.1: Annual deforestation, as the percentage of previous forest, summary statistics per state (1991-2014).

	No. observed municipalities	Mean	St.dev.	Median	Min	Max
Minas Gerais (MG)	564	0.56	0.89	0.31	0	15.38
Rio de Janeiro (RJ)	91	0.38	0.75	0.15	0	10.20
Espírito Santo (ES)	78	0.56	0.90	0.29	0	16.65
São Paulo (SP)	485	0.57	1.18	0.26	0	37.04
Paraná (PR)	396	0.49	0.76	0.27	0	14.30
Santa Catarina (SC)	293	0.43	0.70	0.25	0.01	22.92
Rio Grande do Sul (RS)	370	0.23	0.34	0.14	0	13.73

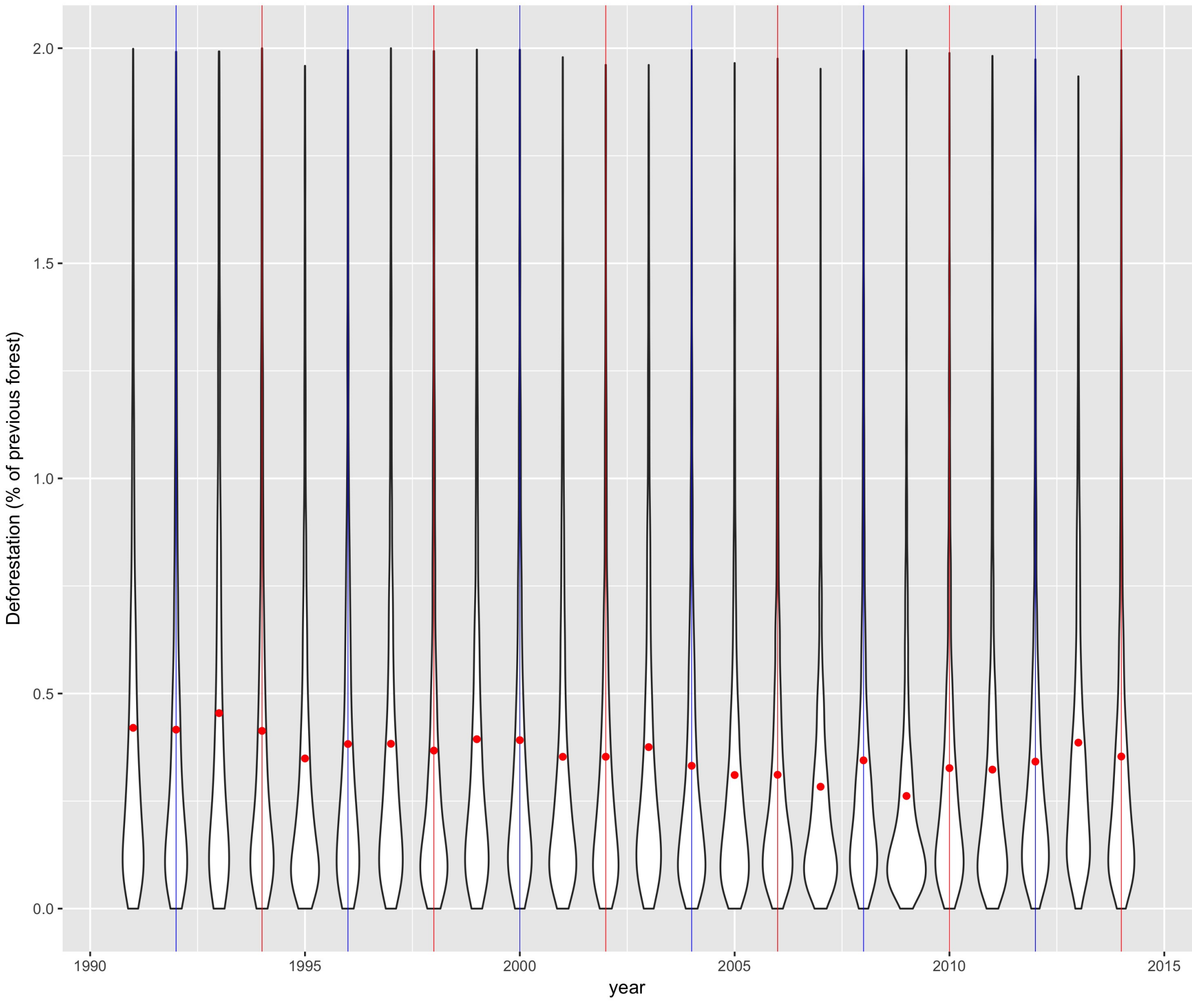

Figura C.1: Violin plots of annual municipal deforestation, as the percentage of previous existing forest (red lines are state election years and blue lines are municipal). Red points indicate mean values. Outliers with values higher then 2% were excluded for visualization purposes.

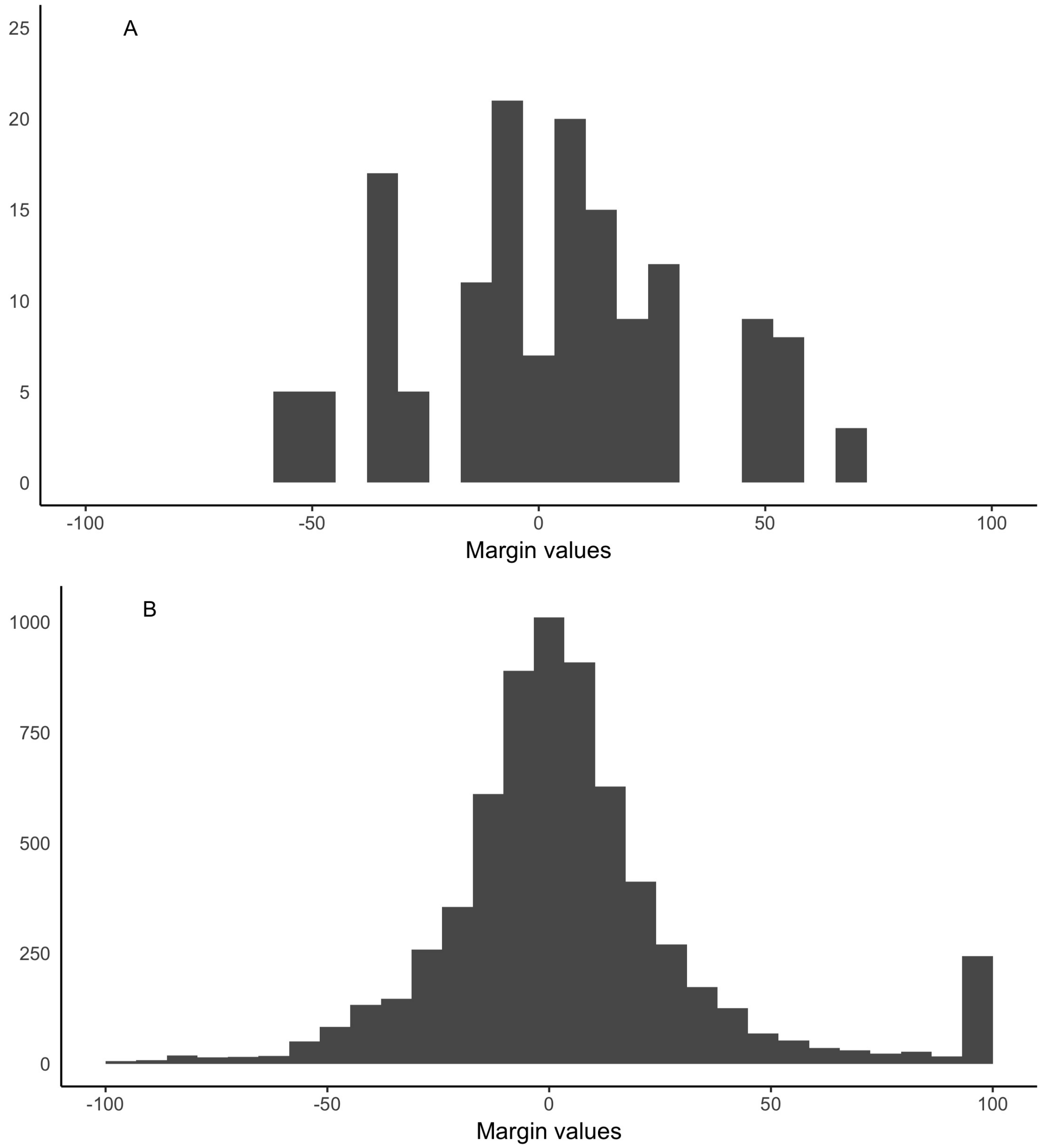

Figura C.2: Frequency of *margin* of votes values for municipal elections events.

Bibliografia

MapBiomas, 2018. Mapbiomas project collection 3.0 of brazilian land cover & use map series. Accessed: June 28 2018.
URL http://mapbiomas.org

Conclusion

The proposal of financial mechanisms for conservation generated high expectations concerning their potential to overcome difficulties faced in the implementation of traditional protected areas and command and control instruments. As a consequence, particularly payment for ecosystem services (PES) programs were widely implemented in the tropics and in developing countries by governments and private organizations (Ezzine-de Blas et al., 2016). In Brazil, the intergovernmental fiscal transfer for conservation (Ecological ICMS) was also rapidly popularized among many states, and then implemented in Portugal. Both mechanisms have a similar time of existence: the first PES Program, as we currently conceive this strategy, was implemented in Costa Rica in 1996 and the first Ecological ICMS law was approved in Brazil in 1991. More than two decades later, we begin to understand how these instruments operate and what are their restrictions and caveats.

We found that the impact of the evaluated financial mechanisms is smaller and has less additionality then expected. Our results demonstrate that PES programs in the Cantareira region have a positive effect on the increase of forest cover inside land properties, mainly through forest regeneration. PES programs offered technical and financial assistance to restore areas that landowners are obliged by law to restore, in exchange of landowners setting aside a larger area for restoration. It means that, considering PES concept strictly, the studied programs may partially lack legal additionality, as new regenerated areas were already under legal jurisdiction for restoration. The Ecological ICMS also shows a positive effect on the creation of new protected areas (PA), both state and municipal. However, similarly to PES initiatives, it may lack additionality as most of the expanded PAs are very unrestricted areas for conservation. Besides, this effect is temporal disappearing after one decade, specially in the case of municipal PAs.

Although both mechanisms are financial incentives for conservation, they are focused on different actors, and work through entirely different processes and administrative structures. PES programs focus on the behavior of private landowners and thus act on private land. So, PES programs executers need personal technical resources to contact landowners, present program options and, most of the time, to convince the landowner to engage in PES activities.

Additionally, PES implementation faces the lack of other rural public policies, such as land regularization and agricultural technical assistance, and encounters demands for both legal compliance and rural productive systems improvements. The Ecological ICMS is focused on the behavior of government actors, state and municipal (local), and operates through an entirely different structure. Once the ICMS-E law is negotiated and approved in the state, the mechanism is easily implemented. State structure and procedures to collect and redistribute the ICMS value-added tax are already present in all Brazilian states, as it is a national defined policy. In order to implement the ICMS-E mechanism, state government only needs to monitor the creation of new PAs, specifically its extent in the municipality, and to incorporate the ICMS-E formula in the redistribution procedures of the ICMS revenue.

Thus, PES programs have a high implementation, operation and transaction costs and, particularly in developing countries, end up incorporating several policy demands into program structure. This leads to very expensive programs with low cost-benefit relations (Fendrich, 2017). On the contrary, an intergovernmental fiscal transfer mechanism, such as the Ecological ICMS, needs minimal additional resources to be implemented. In this sense, PES has very week automaticity - i.e. the extent to which a tool utilizes an existing administrative structure (Salomon, 2002) , while the Ecological ICMS is a very automatic mechanism. On the other hand, PES may have higher coerciveness - i.e. the extent to which a tool restricts or forces individual or group behavior as opposed to merely encourage it (Salomon, 2002) , while the Ecological ICMS has clearly low coerciveness, similar to other tax policies (Kemkes et al., 2010). Those are among the aspects that need to be considered in future evaluations and practical experiences, since there may be several contexts and policy adjustments that can reduce overall policy cost-effectiveness. A complete policy evaluation needs to combine several methods and discuss aspects from causal theory to operation costs to properly inform decision makers and environmental managers.

We explored some possible theory behind the causal relations between interventions and their outcomes. We have contributed to the general questions of what works and why it works (or not); however, future research should increase investigation of the underlying assumptions of those environmental policies (White, 2009). With unexpected and unprecedented results, we showed that politically motivated decisions increase forest loss in political deforestation cycles, related to both municipal and state elections in the Atlantic Forest region. These Atlantic Forest political cycles are not as intense as they are in the Amazon region (Pailler, 2018); however, they indicate that, even with one of the most stringent environmental legislation for tropical forests, this natural and already threatened resource is subjected to politically motivated decisions, which can offset environmental conservation efforts in a cyclically zero-sum output. Both PES programs and the Ecological ICMS certainly interact with the political framework that constrains government officers. Those interactions can be crucial and mediate the impact of these interventions. The Ecological ICMS intervention clearly generates different incentives to municipal and state government, when it interacts with these political constrains. Mayors choose to implement very unrestricted PAs, instead of more laborious ones, possibly in order to quickly benefit and take advantage of the ICMS-E revenue before the upcoming elections.

Our results indicates that vital areas for conservation should not depend on financial incentives to be implemented, as already discussed by other researchers (Kemkes et al., 2010). Notwithstanding, financial mechanisms are possible tools to be operated in synergy with other policy initiatives, working as a complement or as an alternative in specific contexts. Finally, political incentives cannot be neglected as they certainly interact with those mechanisms and may significantly influence their outcomes.

Bibliography

Ezzine-de Blas, D., Wunder, S., Ruiz-Pérez, M., Moreno-Sanchez, R. d. P., 2016. Global Patterns in the Implementation of Payments for Environmental Services. PLoS ONE 11 (3), e0149847–16.

Fendrich, A. N., 2017. Análise de custo-eficácia dos pagamentos por serviços ambientais em paisagens fragmentadas: estudo de caso de são paulo. MSc dissertation, Escola de Engenharia de São Carlos, Universidade de São Paulo.

Kemkes, R. J., Farley, J., Koliba, C. J., 2010. Determining when payments are an effective policy approach to ecosystem service provision. Ecological Economics 69 (11), 2069–2074.

Pailler, S., 2018. Re-election incentives and deforestation cycles in the Brazilian Amazon. Journal of Environmental Economics and Management 88, 345–365.

Salomon, L. M., 2002. The tools of government. A guide to the new governance. Oxford University Press.

White, H., 2009. Theory-based impact evaluation: Principles and Practice. International Initiative for Impact Evaluation. Working Paper 3.