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Economic and policy mechanisms affecting land use decisions in the Brazilian agricultural frontier

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To Zaira

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Abstract

The sustainable use of land resources is one of the most complex challenges facing the 21st century. As Brazil is one of the countries most endowed with land resources, it can facilitate global food and energy demands while providing vital ecosystem products and services. With half of Brazil's forest in possession of private farmers and landowners, a better understanding of the mechanisms affecting their land use decisions is key to design and implement policies that promote synergies and reduce trade-offs between agriculture and nature conservation. Therefore, the objective of this dissertation is to scrutinize the mechanisms that affect farmers' decisions when they are confronted with a changing economic and policy contexts in the biggest country in South America. The analysis throughout this dissertation focuses on the enormous region covered by the Amazon and Cerrado biomes, as they provide important local and global ecosystemic functions, but have also seen most of the agricultural expansion into forested areas observed in the last and current centuries.

The study is guided by a conceptual framework on agricultural systems to look into three specific mechanisms steering land use decisions: the role of speculation; the production decision response to changes in agricultural markets; and the effect of value-chain and credit restrictions to reduce deforestation and increase land use intensification, such as the 2006 Soy Moratorium. The first analysis uses land market panel data at the regional level and reveals that land users speculate based on planned infrastructure improvements and may also relocate deforestation in response to the establishment of conservation policy. The second analysis uses cross-sectional information at the district level and finds that better economic conditions for agricultural activities do induce land intensification, yet, most of the increments in total production (between 70-99% of the total change) are expected from the expansion of agricultural land into forest and non-forest areas. Moreover, the impact of stronger forest monitoring schemes can result in small changes in production decisions. The third analysis uses information at the farm level between 2001-2017 to corroborate that value-chain and credit restriction policies reduce incentives to deforest, but found little evidence to support an observed policy-driven intensification effect. Moreover, these conservation policy impacts are only observed in years where governmental commitments to enforcing environmental regulations were strongly enforced by the federal government.

To cost-effectively promote the sustainable use of land resources in agricultural frontier areas in Brazil, future policies should consider the implementation of economic and policy mechanisms identified in this dissertation. These findings also suggest that Brazil's

impressive conservation achievements during the 2000s succeeded in large part due to the commitment and collaboration of both government and private sector organizations to ensure that rural development abides by existing environmental regulations. If national political discourses raise expectations of weakening environmental law enforcement, the unique and irreplaceable ecosystems found in Brazil could be irreversibly damaged, creating conditions where the sustainable use of land resources needed for long-term global prosperity are no longer possible.

Zusammenfassung

Die nachhaltige Nutzung von Landressourcen gehört zu den komplexesten Herausforderungen des 21. Jahrhunderts. Als eines der Länder mit den meisten Landressourcen ist, kann Brasilien einen Teil des globalen Nahrungsmittel- und Energiebedarfs decken und gleichzeitig lebenswichtige Produkte und Ökosystemdienstleistungen bereitstellen. Die Hälfte des brasilianischen Tropenwaldes befindet sich derzeit in privatem Besitz. Daher ist ein besseres Verständnis der Mechanismen, die sich auf Landnutzungsentscheidungen auswirken, der Schlüssel zur Konzeption und Umsetzung von Strategien, die Synergien fördern und Zielkonflikte zwischen Landwirtschaft und Naturschutz verringern. Ziel dieser Dissertation ist es, die Mechanismen zu untersuchen, die Landnutzungsentscheidungen beeinflusst haben, während sich der wirtschaftliche und politische Kontext im größten Land Südamerikas wandelte. Die Untersuchung konzentriert sich auf die riesigen Amazonas- und Cerrado-Biome, da sie wichtige lokale und globale Ökosystemfunktionen erfüllen, aber auch am Stärksten von Waldverlust durch landwirtschaftlicher Expansion im letzten und aktuellen Jahrhundert betroffen waren.

Die Dissertation beruht auf einem theoretischen Rahmen für landwirtschaftliche Systeme, in dem drei spezifische Mechanismen zur Steuerung von Landnutzungsentscheidungen vorausgesetzt werden: die Rolle von Spekulation, die Reaktion der Produktionsentscheidung auf Veränderungen auf den Agrarmärkten und die Auswirkungen von Kredit- und Wertschöpfungskettenbeschränkungen (z.B. das Sojamoratorium aus dem Jahr 2006) auf die Verringerung von Abholzung und die Intensivierung der Landnutzung. Die erste Teilstudie verwendet Daten eines Landmarktpanels auf regionaler Ebene und zeigt, dass Landnutzer*innen auf Grundlage geplanter Infrastrukturverbesserungen spekulieren und Abholzung als Reaktion auf Naturschutzgesetze möglicherweise verlagern. Die zweite Teilstudie verwendet Querschnittsdaten auf Bezirksebene und bestätigt, dass bessere Marktbedingungen für landwirtschaftliche Aktivitäten die Intensivierung der Bodennutzung fördern. Der größte Zuwachs der Gesamtproduktion (zwischen 70 und 99% der Gesamtveränderung) ist jedoch mit der Ausweitung der landwirtschaftlichen Nutzfläche auf Wald- und Nichtwaldflächen zu erklären. Darüber hinaus kann die Einführung effektiverer Wald-Monitoringsysteme zu statistisch nachweisbaren Änderungen in Produktionsentscheidungen beitragen. Die dritte Teilstudie stützt sich auf mehrjährige Landnutzungsdaten auf Betriebsebene zwischen 2001 und 2017, und bestätigt, dass Kredit- und Wertschöpfungskettenbeschränkungen Anreize zur Abholzung verringern. Jedoch bleibt

der erwartete Effekt einer politikgetriebenen Intensivierung weitgehend aus. Darüber hinaus sind die Auswirkungen der Naturschutzpolitik nur in den Jahren zu beobachten, in denen die brasilianische Regierung die staatlichen Verpflichtungen zur Umsetzung von Umweltvorschriften nachdrücklich durchsetzte.

Um die nachhaltige Nutzung von Landressourcen an den brasilianischen Agrargrenzen kosteneffektiv zu fördern, sollte die künftige Politik die Umsetzung der in dieser Dissertation identifizierten wirtschaftlichen und politischen Mechanismen in Betracht ziehen. Diese Ergebnisse deuten auch darauf hin, dass Brasiliens beeindruckende Errungenschaften im Bereich des Naturschutzes in den 2000er Jahren zum großen Teil auf das Engagement und die Zusammenarbeit von staatlichen und privaten Organisationen zurückzuführen sind, die sicherstellten, dass sich der Landnutzungswandel stärker an den bestehenden Umweltvorschriften orientierte. Wenn politische Diskurse in Brasilien die Erwartung wecken, die Durchsetzung von Naturschutzgesetzen zu schwächen, könnten die einzigartigen und unersetzlichen Ökosysteme Brasiliens irreversibel beschädigt werden. Dadurch würden Bedingungen geschaffen werden, unter denen eine nachhaltige Nutzung der für den langfristigen globalen Wohlstand erforderlichen Landressourcen nicht mehr möglich wäre.

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ABIOVE	The Brazilian Association of Vegetable Oil Industries (*Associação Brasileira das Indústrias de Óleos Vegetais*)
ANEC	The Association of Cereal Exporters in Brazil (*Associação Nacional dos Exportadores de Cereais*)
BLAR	Brazilian Legal Amazon region
CAR	Rural Environmental Cadaster (*Cadastro Ambiental Rural; Serviço Florestal Brasileiro- Ministério da Agricultura, Pecuária e Abastecimento*)
CONAB	The National Food Supply Company (*Companhia Nacional de Abastecimento; Ministério da Agricultura, Pecuária e Abastecimento*)
C&C	Command and control conservation policy
DETER	The real-time deforestation detection service (*Detecção de Desmatamento em Tempo Real; Ministério da Ciência, Tecnologia e Inovações*)
DNIT	The National Department of Transport Infrastructure (*Departamento Nacional de Infraestrutura de Transportes*; *Ministério da Infraestrutura*)
Embrapa	The Brazilian Agricultural Research Corporation (*Empresa Brasileira de Pesquisa Agropecuária; Ministério da Agricultura, Pecuária e Abastecimento*)
EU	European Union
FC	Forest Code (*Código Florestal brasileiro*)
IBAMA	Environmental Enforcement Agency (*Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais Renováveis; Ministério do Meio Ambiente*)
IBGE	The Brazilian Institute of Geography and Statistics (*O Instituto Brasileiro de Geografia e Estatistica*; *Ministério da Economia*)
INPE	The National Institute for Space Research (*Instituto Nacional de Pesquisas Espaciais; Ministério da Ciência, Tecnologia e Inovações*)
LRA	Legal Reserve Areas
MATOPIBA	Region in four states where soybean production is expanding: southern Maranhão (MA), Tocantins (TO), southern Piauí (PI), and western Bahia (BA)

MSE	Mean squared error
NQ	Normalized quadratic function
PPA	Permanent Preservation Areas
PPCDAm	The Action Plan for Prevention and Control of Deforestation in the Legal Amazon (*O Plano de Ação para Prevenção e Controle do Desmatamento na Amazônia Legal; Ministério do Meio Ambiente*)
RD	Regression discontinuity
REDD+	Reducing emissions from deforestation and forest degradation in developing countries
R&D	Research and Development
SDGs	The Sustainable Development Goals
SICAR	The national system of environmental rural registry (*Sistema Nacional de Cadastro Ambiental Rural; Ministério da Agricultura, Pecuária e Abastecimento*)
SNQ	Symmetric normalized quadratic function
SNUC	The National Protected Areas System (*Sistema Nacional de Unidades de Conservação*; *Ministério do Meio Ambiente*)
SUR	Seemingly unrelated regressions

PART I

Mechanisms affecting agricultural systems and land use decisions

2. Introduction and motivation

At the end of 2015, the United Nations released the 2030 Agenda for Sustainable Development for people, planet, and prosperity (UN 2015) It sets the Sustainable Development Goals (SDGs). They are 17 goals that have as overarching objectives "poverty eradication, changing unsustainable and promoting sustainable patterns of consumption and production, and protecting and managing the natural resource base of economic and social development" (OWG-UN 2014). Yet, reaching desirable outcomes in all fronts is not an easy task (Nilsson *et al* 2016).

Brazil holds great potential to alleviate local and global challenges and in doing so to materialize different SDGs, such as Zero Hunger (SDG 2) and Life of Land (SDG 15), but this depends on how sustainable the South American country uses its land (ICSU 2017). Brazil has a dynamic and increasing agricultural sector, as well as large land endowments with pristine ecosystems. This setting brings trade-offs between agricultural land expansion and forest conservation.[1] Not surprisingly, there has been a struggle to balance these two objectives of agricultural production and preservation of ecosystems for decades in Brazil (Binswanger 1991, Andersen 1996, Margulis 2003, Fearnside 2017). At the beginning of this century, observed reductions of forest conversion in the Brazilian Amazon have been related to market fluctuations and policy instruments in place (Richards *et al* 2012, Arima *et al* 2014, Nepstad *et al* 2014, Cisneros *et al* 2015, Gibbs *et al* 2015). Little is known, however, on the extent to which these mechanisms affect farmers' decision making, specifically, towards more intensification or expansion of agricultural land with consequent deforestation outcomes. This is crucial in the quest to achieve sustainable use of the resource as approximately half of the remaining Brazilian natural vegetation is in private land (Soares-Filho *et al* 2014). The main objective of this thesis is, therefore, to investigate the role of policy and economic mechanisms affecting farmers' land use decisions in the Brazilian agricultural frontier.[2]

The agricultural sector in Brazil can relieve part of the world's demand for food, feed, fuel, and fiber. Global per capita demand for caloric and protein content of crops is expected to double by 2050 compared to its level in 2005 (Tilman *et al* 2011). Particularly, demand for

[1] In this thesis, agricultural land includes cropland and pasture uses which are related to crops and cattle production but also are a mean to secure landownership of the land.
[2] In this thesis, the Brazilian agricultural frontier is understand as a set of different agricultural frontier areas depending on their stage of development and accessibility to markets Schielein and Börner (2018).

soy has increased in the past five decades, and the tendency suggests it will continue such a trend pulled by demands on cooking oil, livestock feed, and biodiesel from higher incomes and dietary changes in emerging economies (Garrett *et al* 2013a). Brazilian agricultural products, like soy and cattle, are highly lucrative while gaining importance in global markets (FAO 2013). Brazil has increased its agricultural and livestock production accordingly in the last decades (FAO 2013). Its agricultural innovations (e.g. lime powder for the acidity of soils) bring also potential south-to-south technological transfers that offer solutions to other regions such as sub-Saharan Africa, in which agricultural productivity is seen as a source for alleviating poverty (Alene and Coulibaly 2009). These increments in agricultural production have been possible, however, not only through innovative technological changes but also through the expansion of agricultural land (Andersen *et al* 2002, Pardey *et al* 2004).

The expansion of agricultural areas in natural landscapes brings losses of ecosystems' products and services, such as those provided by the Amazon or the Cerrado biomes at local and global levels (Ratter *et al* 1997). As an example, carbon sequestration and storage in the Amazon rainforest has gained great attention in recent years by the international community. Estimates show that around 30% of carbon emissions related to deforestation were exported from Brazil at the beginning of the current century (Karstensen *et al* 2013). Therefore, it has major potential for cost-effective ways to mitigate the effects of climate change (Kindermann *et al* 2008). Not surprisingly, the country attracts international climate change-related initiatives such as reducing emissions from deforestation and forest degradation in developing countries (REDD+).

The quest of reducing trade-offs amongst different objectives is also affected by Brazil's political-economic context. Government and private sector initiatives had considerably reduced forest loss in the Amazon rainforest but with important setbacks in the past decade. Reforms to the Forest Code (the environmental law ruling conservation of natural ecosystems in the country) were done in 2012, after strong agricultural lobbying, which took back some conservation achievements from the previous decade (Soares-Filho *et al* 2014). The last two administrations continue to signal reductions in conservation while increasing support to infrastructure and rural development (Fearnside 2017, Soares-Filho and Rajão 2018, Ferrante and Fearnside 2019). Further, private sector initiatives to exclude Amazon deforested land from their value-chain have been partially successful but subversive behavior has decreased this success (Gibbs *et al* 2015, Gibbs *et al* 2016, Merry and Soares-Filho 2017). Moreover, biomes with lower conservation governance, e.g. Cerrado savannah, are hosting the new deforestation hot spots in detriment of water systems regulation and biodiversity endowments (Sawyer 2008).

The presence of competing interests between agriculture and conservation on the use of land is undeniable in the context of Brazil. The call for development is strong, especially in regions like the Brazilian Legal Amazon region (BLAR), which can bring large-scale forest conversion and consequent direct and indirect negative impacts to services and products provided by ecosystems. Moreover, agricultural expansion can lead to negative feedbacks that might reduce the productivity potential of land (Oliveira *et al* 2013). Strong conservation governance has proved to decrease agricultural expansion; however, with different effects across space and time (e.g. spillover and leakage effects) (Lapola *et al* 2010, Arima *et al* 2011, Meyfroidt *et al* 2013). It is both in the national and global interests that the South American country finds a balance between rural development and environmental conservation. Thus, understanding the policy and economic mechanisms behind farmers' land use decisions can inform policymakers about the impacts of different strategies with competing objectives.

This thesis analyzes, therefore, economic and policy mechanisms in Brazil affecting land use decisions in general, and deforestation and agricultural intensification in particular. The remainder of this introductory chapter explains the general framework of this thesis, gives a short summary of agricultural development and forest conservation in the current century in Brazil, and lays down the research questions of interest. The general framework analyzes the effect of policy instruments, environmental governance, and markets on the agricultural system driving different land use configurations. Together the general framework, the Brazilian context, and guiding research questions provide a structure for the analytical chapters, each looking at different economic and policy mechanisms affecting deforestation, intensification, and expansion of agricultural land.

2.1. Conceptual framework

This thesis investigates economic and policy mechanisms affecting farmer's decisions and consequent land use configurations in Brazil, with particular attention on agricultural frontier areas. This analysis is based on an overarching conceptual framework that represents different processes that affect agricultural systems and related land use outcomes. The agricultural system is understood here as a farm (either family or business oriented) with land area and it has as objective the production of crops and/or raising livestock to produce food, feed, fiber, and energy from the Earth's natural resources (Jones *et al* 2016). A major focus of this thesis is to understand the economic decisions taken in the agricultural system that put pressure on forest areas. These decisions reflect potential trade-offs between rural development and environmental protection partially steered by the economic context and policy interventions. The conceptual framework, therefore, builds on previous conceptualizations of land use change related to deforestation but emphasizing the economic rationale behind these decisions (Angelsen and Kaimowitz 1999, Geist and Lambin 2002, Hertel 2018). It represents economic and policy mechanisms driving agricultural systems which can lead to deforestation.

Conversion of forest for pasture or cropland are strategies to pursue an increase in production, or as a mean for land ownership to secure future rents from an expanding agricultural sector (Bowman *et al* 2012, Miranda *et al* 2019). The reason to look into the effects of policy and the economic context on an agricultural system in this thesis is motivated as most deforestation observed worldwide, and in the case of Brazil in particular, is related to agricultural expansion (Geist and Lambin 2002). Together with governance and policies in place, agricultural systems are affected by land suitability across the landscape, social aspects at different administrative levels pushing supply and demand of land resources and its related agricultural products, and technology improvements in the agricultural sector (Kaimowitz and Angelsen 1998, Geist and Lambin 2002, Meyfroidt *et al* 2018). They represent underlying drivers of deforestation affecting agricultural systems and their land use decisions. These drivers are depicted by the green-shaded squares of Figure 2-1. This thesis focuses its analysis on governance and policy underlying drivers (right-bottom square) as sources of change in agricultural decisions.

Underlying drivers are channeled to the system through agricultural and land markets. Prices of farming products, prices of inputs of production, and prices of land affect decisions of farmers which represent the agricultural system depicted in Figure 2-1 (blue-shaded squared area). How sensitive are production decisions to these markets in the short-term depends on the technology available which in turn defines the demand for inputs of production, i.e. the productivity of the system in the use of resources. The response of interest (farmer's sensitivity) to changes in prices or conservation policy is revealed through her or his elasticities of output supply, inputs demand, and land which are represented in Figure 1-1 by ε_p^y, ε_p^x, and ε_p^a, respectively. Particularly, the technology available for production and reflected in the elasticities of the system determine a farmer's decision to intensify or expand her or his available agricultural land. The taken strategy affects land use outcomes represented by the brown-colored area in Figure 2-1. The amount of forestland conversion depends on the extent to which these strategies are performed.

When there exists an increase in land productivity, either through technological progress or through variations in the economic context, the amount of land needed to produce the same amount of output is reduced but at the same time can incentivize land expansion due to higher land profitability (Angelsen and Kaimowitz 2001). In agricultural economics, the first effect of land productivity is known as *Borlaug hypothesis*, after Norman Borlaug the father of the Green Revolution, in which an increase in yields reduces cropland expansion and therefore less deforestation is needed (Green *et al* 2005, Burney *et al* 2010, Phalan *et al* 2011, Stevenson *et al* 2013). The rebound effect of land productivity is named *Jevons paradox*, after the 19th century English economist William Stanley Jevons which observed an increase in aggregate use of coal for energy production following an increase in its productivity. In agricultural economics, this paradox has been adopted to understand a potential increment in the use of land after increases in its productivity for agricultural production (Alcott 2005, Villoria *et al* 2014). The extent to which the Jevons paradox surpassed the Borlaug hypothesis partially depends on underlying drivers of land use change, e.g. policy instruments and governance in place.

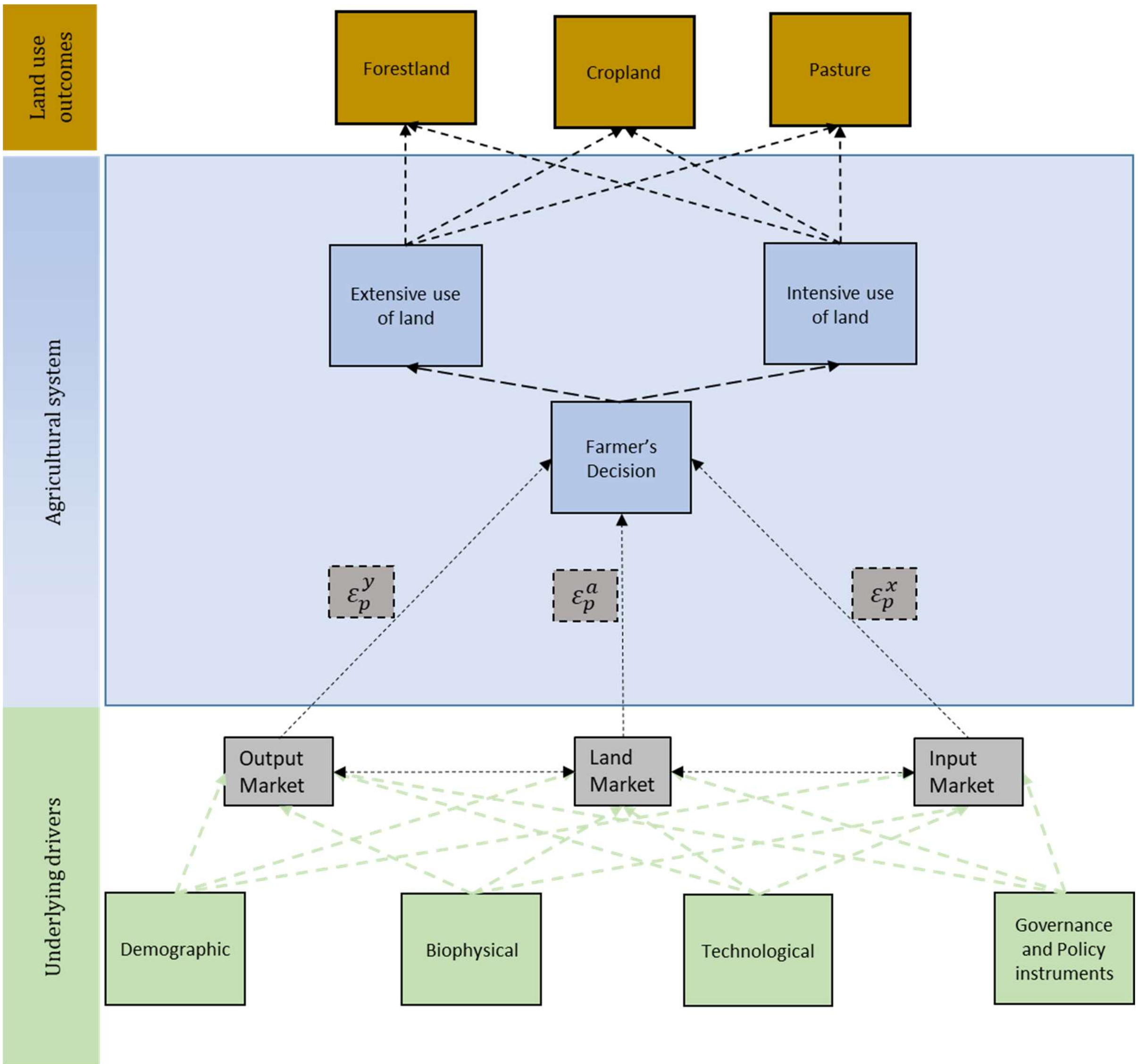

Figure 2-1. Mechanisms driving land use decision in the agricultural system
Note: The figure depicts the relation between agricultural systems (blue-shaded area) and land use configurations (brown-shaded area), as well, as the influence due to different underlying processes (green-shaded area) channeled through agricultural and land markets.

Agricultural productivity and conservation policies are two major aspects affecting agricultural systems and related land use outcomes in the conceptual framework presented above. These two concepts are important to understand how sustainable is agricultural development in Brazil vis-à-vis deforestation. Agricultural productivity is important because it will push the analysis towards knowing the inner elements of production structures; in a peculiar sector with technology embedded with nature and its unpredictable elements (Just and Pope 2001). Conservation policy instruments are important as they can affect agricultural systems and related decisions to reduce, avoid, or stop an environmental loss (Lambin *et al* 2014).

2.1.a. *Agricultural system productivity*

Agricultural productivity growth refers to the economic notion of efficiency. Efficiency increases are defined in two ways: 1) maintenance of a stock of capital with fewer costs and 2) by increasing availability of commodities per unit of time from the stock available (Daly 1974). The first aspect of efficiency overview different strategies to maintain and increase available factors of production. It includes primary (e.g. land) and intermediary (e.g. feed for livestock) inputs. The other aspect highlights technological improvements bringing more products and services with less use of inputs like forestland resources.

Mundlak describes agricultural productivity as "…comparison of changes in aggregate output with changes in aggregate inputs" (Mundlak 2001, p 13). These changes in the use of aggregate inputs are fostered by rearranging the use of factors of production or by technical change (Mundlak 2001). In the view of Nobel laureate, T.W. Schultz, knowledge and investments in human capital (e.g. health and education) foster higher agricultural productivity (Schultz 1980). He also emphasizes that lack of incentives brings an unrealized economic potential for agriculture. This shortage of "right" incentives is mainly affected by price distortions in the form of a lack of information and government policies (Schultz 1980).

Ruttan (2002) complements the work of Schultz by explaining the effect of economic conditions on efficient paths of productivity growth. Furthermore, how they affect the creation of institutions contributing to knowledge. That is, the emergence of innovations is linked to economic conditions (Sunding and Zilberman 2001). This notion is known as induced innovation in which "…productivity growth and factor use … [are] understood in terms of a process of dynamic adjustment to changing relative factor prices…"(Hayami and Ruttan 1970b, p 1124). Binswanger (1974) supports this concept and additionally emphasizes the importance of relative costs of different lines of research (e.g. capital-saving research) thus bringing bias in research efforts. Later, empirical analyzes validated Hayami and Ruttan's work and additionally found a positive influence of knowledge and infrastructure development in innovation implementation (Evenson and Kislev 1975, Antle 1983).

It is important to consider the structure of the technology employed for a better understanding of efficiency in agricultural productivity (Just and Pope 2001). The technology available for agriculture helps to understand policy and market impacts on production decisions and the use of resources. In the context of multi-output technologies, a structure identifies bundles of inputs needed for production. These bundles represent shares of total inputs needed for the production of different commodities. In chapter 3 of this thesis, this is particularly important as knowing the structure implies having an ability to say something about reallocation in the use of land (including forestland) for agricultural production.

2.1.b. Conservation policy instruments

Ecosystems provide society and industries with different products and services at different scales (Constanza *et al* 1997). These natural systems are in a dynamic iterative process with anthropogenic activities and human-wellbeing (MEA 2005). Agriculture is one clear example of these interlinked dynamics (Heal and Small 2002). Benefits for humans obtained from nature's preservation are many times not realized. Often it is due to an existent mismatch between private interests of those in possession of natural resources and the socially desirable provision levels of ecosystems' products and services (Lichtenberg 2002). Securing valuable natural land resources can be achieved through the use of policy intervention (Lambin *et al* 2014).

Börner & Vosti (2013) offer a classification to analyze policy instruments affecting land use and deforestation. They grouped these alternatives into three non-excludable categories. First, enabling measures are those affecting contextual characteristics in which decisions are taken. For example, the presence of property rights of land can secure the rents obtained from investments, and potentially inducing more sustainable practices. The authors consider this group of measures as facilitators for the effectiveness of the other two alternatives. Another group is given by disincentive policies. They influence agents' choices by making unsustainable practices more costly. They are commonly known as command and control (C&C) interventions. The third group is composed of instruments that enhance activities pro-conservation or more sustainable use. The authors named them incentive-based policy instruments. The first two types of interventions have been more prominent in the case of Brazil. They have been related to dramatic reductions in deforestation in the past two decades but it is unclear its long-term success (Arima *et al* 2014, Nepstad *et al* 2014, Soares-Filho *et al* 2014).

Regarding the influence of conservation policy instruments in agricultural activities, Lichtenberg (2002) points out some important aspects that influence the effectiveness of implementation. In some contexts, instruments pursuing environmental quality by inducing technological change or reconfiguration in the use of resources might bring positive or negative spillover effects. A negative spillover, for example, is an increase in the use of pesticides that brings lower water quality. Another aspect is the importance of the scale of analysis in two ways. First, some externalities in the use of resources might not be perceived at local levels of decisionmaking (e.g. climate change-related externalities from deforestation). Second, it is relevant for decision making to know the costs of agricultural production and maintaining environmental quality for small and large scale farmers at different levels of aggregation; then it is possible to identify cost-effectiveness strategies of action. In addition, he steers attention to the issues of hidden information, monitoring efforts, and secondary markets. High monitoring compliance costs and easy creation of secondary markets make it difficult to identify hidden information. In the Brazilian case, monitoring compliance costs together with the spatial location of deforestation influence the enforcement effectiveness of regulatory instruments (Börner *et al* 2014). Additionally, policies aiming to protect forest areas, or to reduce its availability as agricultural inputs, potentially affect the composition in the use of factors of production in farming activities (Ollivier 2012). It exists evidence that such relation might be present in Brazil but up till now, little is known on the level of intensification in the use of land influenced by these policy instruments (Bowman *et al* 2012). Hence, it is key to understand the impacts on agricultural productivity to improve understanding of trade-offs and synergies of such interventions.

2.2. Brazil's agricultural development and forest conservation in the 21st Century

Brazil represents a good example of the complexity to find a balance in the use of land resources. The Southern American country has become since the last two decades a global powerhouse in the production of agricultural commodities such as soy and cattle. These activities, however, have been related to increasing rates of deforestation in highly diverse and ecologically valuable biomes such as the Amazon tropical forest and the Cerrado savannah (Sawyer 2008, Barona *et al* 2010, Macedo *et al* 2012). Local and global demands, together with governance, and policies for development and conservation influence how the production of these commodities shape land configurations (Walker *et al* 2009, Garrett *et al* 2013b).

Production of soy in Brazil started in the second half of the 20th Century and since then it has been expanding both in terms of cultivated area as well as quantity of production (Zalles *et al* 2019). From an area of roughly 0.3 million ha located in the states of Rio Grande do Sul and Paraná in the 1960s, it reached more than 33 million ha in 16 states by 2016 (Martinelli *et al* 2017). One important boost for this expansion is due to high investments in agronomical research that created suitable areas considered previously unproductive (Pardey *et al* 2004).

In terms of global trade at the start of the 21st Century, it produced 23% of the global soybean (oilseeds) exports, just below the United States of America (USA), but since 2018 it holds the lion's share in total world production (reaching 36% in 2020).[3] Ranching activities and pasture expansion have also increased in the BLAR the last half-century (Walker *et al* 2009, Merry and Soares-Filho 2017). The herd size of this region almost tripled between 1987 and 2013 (Merry and Soares-Filho 2017). Brazil has also become the major exporter of livestock, beef and veal worldwide, surpassing the volume of beef and veal exported by Australia (which was the leading exporter in the second half of the 20th century), and livestock from the USA (Vale *et al* 2019, FAO 2020).

Apart from the presence of agricultural incentives to engage in deforestation, the conversion of natural vegetation in Brazil is a quest for claiming land (Hecht 1985, Sauer and Pereira Leite 2012). Asset accumulation in terms of land is common by colonist agents which occupy properties to sell once a land market emerges (Caldas *et al* 2007). During the 1970s and 1980s, small farmers in the south transferred their demand for land to the north in the Amazon contributing to rising land prices and pushing landless settlers and expansive cattle ranching further into frontier areas (Margulis 2003). With the advancement of the agricultural frontier, land prices rise inducing land speculation in a context where land markets do not depend on the presence of formal land titles (Reydon 2011, Barreto *et al* 2013, Holland *et al* 2016). Also, land in the Amazon has been relatively cheap compared to other areas that enhance incentives for conversion (Sauer and Pereira Leite 2012). Land speculation occurs beyond the Amazon biome. In the past two decades, the Cerrado savannah has been subject to this phenomenon in which land is used as a new hub for national and international capital, i.e. pension funds (Borras Jr. *et al* 2012, Sauer and Pereira Leite 2012, WWF 2017).

In terms of environmental regulation and nature conservation, the Forest Code (FC), first enacted in 1934, is the federal law that regulates land use and forest management in private properties in the country. Since its establishment, it mandates farmers to maintain a certain percentage of their farmland as native vegetation. These percentages had major changes in 1965 (Law no. 4.771, 09/15/1965), 1996 (provisional measure no. 1.1511, 06/25/1996), and 2001(provisional measure 2.166-67, 08/24/2001) (Soares-Filho *et al* 2014, Santiago *et al* 2017). By 2001, It required farmers in the BLAR to maintain 80% of their land as native vegetation if they are located in the Amazon biome, 35% for those located in the Cerrado

[3] https://apps.fas.usda.gov/psdonline/app/index.html#/app/home

biome. In the rest of the regions and biomes, a requirement of 20% is needed from each landholder's total property. There are two main types of conservation within farmland in the FC: legal reserve areas (LRA) and permanent preservation areas (PPA). LRA are zones within the farmland that are not geographically defined and are meant for biodiversity conservation in general terms. They are a proportion of farmland that its location is suggested by the landowner and then approved by an environmental agency (Sparovek *et al* 2012). In these areas, productive uses are allowed but only without total removal of natural vegetation (e.g. low impact extraction, undergo regeneration, sustainable forest management) (Ellinger and Barreto 2012). PPAs have as main objective to provide water, soil, biodiversity, landscape conservation, as well as securing the well-being of human populations' (May *et al* 2011). Two different categories are considered: a) Riparian Preservation Areas, i.e. riverside forest buffers; and b) Hilltop Preservation Areas (Soares-Filho *et al* 2014).

Conservation has also been implemented outside private land. Brazil created its first park in 1937 but it was only since the 1970s that the extension and diversity of protected areas significantly increased in the country at different administrative levels (Sparovek *et al* 2012). In 1989 the Brazilian Institute for the Environment and Renewable Resources (IBAMA) within the Ministry of the Interior was created. This institution is held responsible to secure national parks, biological areas, and ecological stations. Likewise, a National Protected Areas Council provides general policies for the creation and management of protected areas (Rylands and Brandon 2005). The National Protected Areas System (SNUC) defines and regulates protected area categories at all levels of government. The types of protected areas in Brazil are *strictly prohibited areas* and *sustainable use areas*. Strictly prohibited areas have biodiversity as a principal objective, and they can be cataloged as national parks, biological reserves, ecological stations, natural monuments, and wildlife refuges. Sustainable use areas relegate biodiversity conservation as a priority and they can be cataloged as environmental protection areas, areas of ecological interest, extractive reserves, fauna reserves, sustainable development reserves, and private natural heritage reserves (Rylands and Brandon 2005, pp 615–6). Also, indigenous reserves are part of the contribution to conservation. By the mid-2000s roughly 12% of Brazil's territory was recognized as this type of reserve (Rylands and Brandon 2005). This represents almost 20% of the Brazilian Legal Amazon region territory (Rylands and Brandon 2005).

The onset of the 21st Century brought historically high rates of deforestation in the Amazon. The government response came, and environmental governance and conservation policies shifted. The first big breakthrough came in 2004 when it was officially launched the Action Plan for Prevention and Control of Deforestation in the Legal Amazon (PPCDAm). In its first phase from 2004 till 2007, IBAMA was restructured and started using nearly real-time satellite information to detect forest conversion to help enforcement efforts. It also increased the extension of areas protected, as well as those of conservation units (Arima *et al* 2014, p 466). A second boost on conservation came from an initiative of the private sector. International attention pointed to the influence of soy production on deforestation which steered major players in the industry to reduce their reputational risk from their value-chain activities (Gibbs *et al* 2015). The Soy Moratorium was established in 2006 which stop any purchase of soy coming from deforested areas after the establishment of the moratorium (Gibbs *et al* 2015). Two years later the federal government joined the efforts to enforce the moratorium in partnership with the private sector. Since its implementation, it was renewed every 2 years and in 2016 it was permanently established (Fearnside 2017, Carvalho *et al* 2019). A similar strategy was done in 2009 for the cattle sector but its achievements have been put into question (Gibbs *et al* 2016, Soares-Filho and Rajão 2018).

In 2008, the government renewed PPCDAM with a second version (PPCDAm-II). This phase targets a reduction of deforestation of 80% by 2020 (relative to 1996-2005 baseline). Restrictions to agricultural credit conditional on environmental performance and the issue of a blacklist of municipalities with a bad environmental record are two additional efforts to stop deforestation that were implemented between 2007 and 2008 (Cisneros *et al* 2015). An additional instrument implemented is the Rural Environmental Registry program (CAR). It was a voluntary registration with state-specific rules and first established in the years 2008 and 2009 in Para and Mato Grosso in the north and mid-west of Brazil, respectively.[4] Through this program, landowners are required to register their property boundaries to the environmental regulatory agency to facilitate monitoring of environmental compliance. It is mandatory in all states since 2012.

Despite these impressive efforts to halt deforestation, environmental protection has been reduced in the last decade. Since 2008 some 12,400 km^2 of protected areas had been degazetted, 31,700 km^2 had been downsizing, and 21,000 km^2 had been threatened by proposals in the National Congress (Ferreira *et al* 2014). A major change in environmental

[4] The state of Mato Grosso was the first one to implement an environmental licensing of rural properties using remote sensing and geographic information systems Azevedo and Saito (2013). In 2000 was established the System for Environmental Lycensing (LAU) as part of the Land Zone Planning System (SLARP) but it only involved CAR until 2009 Azevedo and Saito (2013); Rasmussen *et al* (2016).

governance came with the establishment of the new FC in 2012 (Law no. 12.651, 05/25/2012, and modified by Law no. 12.727 and presidential decree, 7.830, 10/17/2012).[5] The new FC brought important changes: 1) it reduces the requirements of both PPAs and LRAs; 2) it reduces mandatory restoration on critical PPAs of riparian systems; 3) it sets a framework to create markets to offset deficits on legal reserves with the protection of forest surplus in another farm (i.e. similar to a cap-and-trade system); and 4) it offers amnesty to farmers that deforested before 2008 (Soares-Filho *et al* 2014). This amnesty also affected the soy moratorium which in its 2014's renewal adjusted the reference date of compliance from July 2006 to July 2008. Besides agriculture as a threat for conservation, infrastructure investments such as roads, new hydropower dams, and mining projects bring a new challenge towards protecting the environment as they are catalysts for land speculation in the Brazilian Legal Amazon region (Fearnside 2002a, Fearnside 2017). Especially without strong environmental safeguards being in place (Ferreira *et al* 2014). Finally, recent federal administrations have sent signals of reducing efforts of conservation for agricultural development in the Brazilian Legal Amazon region which enhances the likelihood of unpunished subversions of the environmental law (Rochedo *et al* 2018, Ferrante and Fearnside 2019).

2.3. Research question and sub-questions

The evidence so far has shown a strong link between agricultural activities and land use configurations in Brazil that are partially steered by governance and policies in place; however, it exists little empirical evidence on the economic and policy mechanisms driving economic decisions in agricultural systems with its consequent deforestation and land use intensification outcomes. It is important to fill this gap to better inform policymakers about which strategies can bring sustainable land use outcomes, particularly, in managing land resources in regions like the BLAR that hosts valuable ecosystems in three different biomes together with new and consolidated agricultural frontiers with different incentives for forest conversion. Based on the policy relevance of this topic, my interest is to answer the question: ***how do economic and policy mechanisms affect economic decisions in agricultural systems leading to deforestation or land use intensification in Brazilian agricultural frontiers?***

[5] The main reasons for the need of a new law were: i) low compliance with the FC; ii) national and international awareness on consequences from land-use change and its relation with the agricultural sector; iii) high costs of restoration of natural vegetation through planting to achieve full compliance with a pre-2012 FC; and iv) farmer's perception that the FC hinders agricultural development and that conservation should be done mainly in public land Sparovek *et al* (2012).

More specific variants of this general research question are answered in three analytical chapters.

Chapter 2 looks into land market regions to reveal the effect of infrastructure development and conservation policy in agricultural decisions that lead to deforestation. It disentangles land prices in three major components: agricultural rents derived from the use of land, costs from forest conversion, and a speculative component driven by infrastructure improvements and conservation policy implementation. Road infrastructure development is an important driver in Brazilian frontier areas fueling land speculation (Hecht 1985, Fearnside and de Alencastro-Graça 2006, Fearnside 2007). An effective conservation policy can reduce incentives to deforest in areas subject to policy implementation as it makes it more costly to deforest. A negative spillover effect comes from an increase in forest conversion in areas with lower regulations or not subject to policy, i.e. leakage effect; that is, agricultural land is provided from areas with lower expected costs of conversion.

This first analytical chapter answers the specific question: *Do land prices convey information on future land conversion in Brazil?* If land prices do convey this information, land markets can reveal information on the extent to which development and environmental protection steer deforestation hot spots. Moreover, land markets can represent a barometer of conservation policy leakage.

Chapter 3 is based on a structural economic model to identify how commodity prices affect economic decisions on agricultural systems with conservation policy in place. The effects of interest are changes in yields and absolute levels of agriculture, and land use allocation due to changes in agricultural prices. The use of the extensive or intensive margin of land is also revealed. This analysis is carried out at a district level. Specifically, it estimates farmers' response by calculating output, input, and land use elasticities to prices. Also, I estimate the respective elasticities to changes in the intensity of environmental fines as a proxy for conservation policy in place.

This second analytical chapter answers the specific question: *do agricultural prices induce market-driven land use intensification in the Brazilian agricultural frontier?* If agricultural prices do induce intensification, the interest is to know which effect then dominates: the Borlaug hypothesis or the Jevons paradox. Moreover, the analysis does account for the effect of conservation policy affecting the dominant effect of land productivity.

Chapter 4 focuses on conservation policy mechanisms affecting agricultural systems and their respective land use outcomes. It uses farmers' information and exploits the biome-targeted assignment rule of two emblematic policies in the 21st Century in Brazil: credit restrictions conditional on compliance with the FC (since 2008); and the Soy Moratorium (since 2006). Production economic theory frames the analysis to set up a regression discontinuity (RD) design. This study is placed in the border between the Amazon and the Cerrado biomes; part of the area named the Arc of Deforestation.[6] The principal advantage of this chapter is that it can single out the effect of those policies only targeted in the Amazon and compares the strategy taken by farmers with those in neighboring Cerrado. Deforestation and land use intensification inside farmland are evaluated after policy implementation.

This third analytical chapter answers the specific questions: *do conservation policy instruments effectively reduced deforestation inside farm areas in the Brazilian agricultural frontier?* If this is the case, *do farmers also intensify the use of already cleared land?* The importance to answer these questions simultaneously is to identify synergies and trade-offs of conservation policy implementation in the 21st Century; in a timeframe in which environmental governance became stronger for some years but shifted back in Brazil.

2.4. Thesis structure

This thesis is organized into three parts. The first introductory part lays down the research motivation, the conceptual framework, the research questions, and offers a brief account of the Brazilian agricultural development and forest conservation in the 21st Century. This first part is given in the present chapter. A second analytical part answers the specific research questions presented above and it is divided into three analytical chapters. Chapter 2 analyze Brazilian land markets to capture the effects of infrastructure development and conservation policy on land speculation, and potential leakage effects (i.e. negative spillover effects of policy implementation). Chapter 3 sets a structural model to analyze the effect of the change in agricultural prices or conservation policy in place on the economic decisions of agricultural systems, and by doing so, it offers an answer on the extent to which the use of the intensive or extensive margin of land is guiding land productivity. Chapter 4 singles out the effect of conservation policy on farmers' decision to deforest and intensify the use of land to

[6] The Amazon Environmental Research Institute (IPAM) defines the Arc of deforestation as „The region where the agricultural border advances towards the forest and also where the highest rates of deforestation of the Amazon are found. It corresponds to 500 thousand km² of land, going from the east and south of the Brazilian state of Pará towards the west, passing through the states of Mato Grosso, Rondônia, and Acre.“ (https://ipam.org.br/glossario/arc-of-deforestation/ consulted in May 2019)

provide some answers on the extent to which policy is achieving its main target of forest conservation together with synergies by inducing an increase in land use intensification. A concluding third part is given in Chapter 6. This chapter gives a general discussion, offers policy recommendations, and sheds some light on potential venues and challenges for future research towards a better understanding of agricultural systems and related land use outcomes in Brazilian agricultural frontiers.

PART II

Analysis of the Brazilian agricultural frontier

3. Land speculation and conservation policy leakage in the Brazilian agricultural frontier: panel fixed-effects analysis at the regional level

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Abstract

The Brazilian Amazon and Cerrado biomes have been subject to strong pressure from agricultural expansion over the past two decades. A common claim is that the associated tree cover loss was partly driven by speculative land acquisition. In this chapter, we analyze the effects of information on planned road infrastructure improvements and changes in conservation policy implementation on expectations of forest conversion. We use a unique land price dataset covering the period from 2001 to 2012. Based on land rent and hedonic valuation theory, we argue that forestland prices convey information on expected future land use. We decompose forestland prices into a conventional forestland rent and a speculative part related to forestland conversion and alternative land use rents. Using a fixed-effect panel, we then assess whether, where, and to what extent changes in conservation policy affect forestland prices over time. Our results confirm that forestland prices contain expectations about converting forestland to agricultural or pasture land. We also find indications that the Brazilian land market conveys information about potential conservation policy leakage and explore this conjecture descriptively using dynamic deforestation hotspot maps.

Keywords: land price, speculation, conservation policy, leakage

3.1. Introduction

Land resources are under pressure to satisfy the global demand for agricultural products (Tilman *et al* 2011, Leblois *et al* 2017). Countries with large amounts of fertile land like Brazil are thus expected to produce additional food, feed, and energy crops (OECD 2015, FAO 2018). However, the production of globally traded commodities such as soy and beef is often associated with the expansion of agricultural frontiers in ecologically sensitive biomes, such as the Amazon and the Cerrado Savannah, at considerable environmental and social costs (McAlpine *et al* 2009, Karstensen *et al* 2013, Nepstad *et al* 2013).

Conversion of natural vegetation at agricultural frontiers is often both a result of productive input allocation decisions and a strategy to secure land claims either for subsistence or to benefit from appreciating land markets (Hecht 1985, Caldas *et al* 2007, Fearnside 2008). Converting forest areas to pasture has long been an effective strategy to secure land ownership. The market price of forestland, therefore, consists not only of the value related to the *current* land uses (e.g., forest-products) but also of *expected revenues* from future land uses, such as pasture (Barreto *et al* 2008, Carrero and Fearnside 2011, Strassburg *et al* 2014). The latter is uncertain and thus an inherently speculative component of the forestland price. Changes in land prices can thus reveal information on the incentives of deforestation and related expectations on future land use change (Margulis 2003, Merry *et al* 2008, Sills and Caviglia-Harris 2009).

In the context of agricultural frontier expansion, the speculative component of the price of forestland constantly changes with new investments into infrastructure making pastures or cropland more profitable (Hecht and Mann 2008, Sauer and Pereira Leite 2012). Similarly, priority shifts in the enforcement of property rights and conservation policies may affect speculative behavior on land markets (Araujo *et al* 2009, Brown *et al* 2016, Azevedo *et al* 2017, Koch *et al* 2017). When governments devise conservation policies to counteract frontier expansion, conservation priorities and enforcement effectiveness tend to vary in space leading to leakage effects (Fearnside 2009, Barona *et al* 2010, Lapola *et al* 2010, Soares-Filho *et al* 2010, Arima *et al* 2011, Gibbs *et al* 2015). Leakage refers to the displacement of land use activities from a region subject to conservation policy enforcement to another region without or with lower levels of enforcement (Lambin and Meyfroidt 2011, Meyfroidt *et al* 2018). If the leakage effect is large, it should be reflected in land markets, with increasing land prices indicating growing demand for land in regions subject to lower levels of conservation policy enforcement.

This paper seeks to shed light on how spatially heterogeneous infrastructure investments and conservation policy enforcement affect the Brazilian land market. We focus on the speculative component of the land price, which contains expectations on the appreciation of low-value forestland after converting it to high-value pasture or cropland. The potential role of speculation as a driver or timely indicator of deforestation has so far rarely been considered explicitly in predictive models of deforestation (Kaimowitz and Angelsen 1998, Busch and Ferretti-Gallon 2017). Uncovering the economic mechanisms driving speculative behavior may thus help policymakers to anticipate future deforestation hotspots.

The remainder of the chapter is structured as follows. In Section 2.2, we develop a theoretical framework that decomposes market prices of forestland into rents, conversion costs, and a speculative component. Section 2.3 provides a background on the study area and documents our empirical strategy. The results are presented in Section 2.4. We find that a reduction in expected travel time from a location in the landscape to the nearest market contributes to an increase in forestland prices, an effect reinforced in our area of study by policy-induced leakage. In Section 2.5, we discuss our findings and policy implications.

3.2. Land prices and speculation

Land rent theory explains how access to markets affects land rents and associated land use patterns (Holland *et al* 2016). According to this theory, land rents are a function of a) distance to sources of trade or relevant markets (Thünian notion), and b) land productivity (Ricardian notion) determined by bio-geophysical factors, such as topography, soil fertility, climate conditions, and agricultural technology (Munroe *et al* 2002).

Figure 2-1 depicts a land rent theory framework for an alternative use of forestland over two time periods t. Yellow lines represent rents of an alternative land use (e.g., pasture), R^P.[7] The straight bold line indicates the first period land rent that depends on distance to market and transport nodes. The agricultural frontier (D_F) is located where land rent becomes zero. The dashed line illustrates the effect of infrastructure improvement in the second period, implying lower transportation costs and therefore a flatter rent curve. The agricultural frontier expands to D''_F. This expansion happens if conversion of forestland to pasture involves negligible costs. If infrastructure investments are accompanied by improved conservation policy enforcement, conversion costs increase (e.g. due to the risk of paying fines) implying a

[7] In the remaining we exemplify alternative uses with pasture. This has been pointed as a major source of deforestation in Brazil and often as the land conversion resulting from speculation (Bowman (2016).

downward shift of the rent curve (Börner *et al* 2014). This leads to a leftward shift of the agricultural frontier ($D_F' < D_F''$ in Fig 1). Fig. 2-1 also shows that infrastructure improvements lead to higher rents from pastures at any location due to travel time savings. Furthermore, land rents beyond the agricultural frontier D_F, but within the frontier D_F', are zero in the first period and become positive in the second period. Here the conversion of forests to alternative land uses increases land rents.

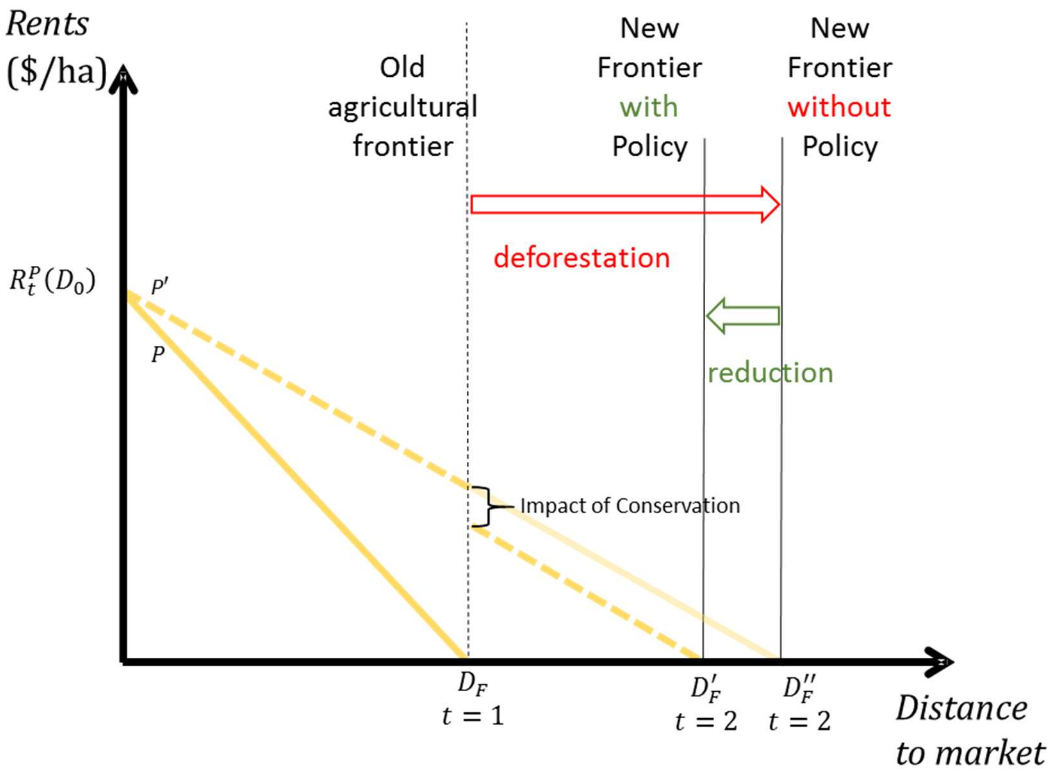

Figure 3-1. Land rents in a two-period model with infrastructure investments and conservation policy enforcement

Note: This graph shows bid-rents for an alternative land use (e.g. pasture). In a first period, at the agricultural frontier, D_F, rents of an alternative land uses are zero and land remains forest. In a second period (half-dashed, half-straight yellow line), a driver of deforestation, e.g. road infrastructure improvement, shifts bid-rents outwards and induces frontier expansion up to D_F''. If conservation policies are implemented effectively, they will reduce the impact of this effect on deforestation by incrementing the cost of converting forestland beyond the old frontier (dashed yellow line).

Standard land rent theory, as summarized in Figure 2-1 can only explain deforestation as a result of changing production incentives (Jepson 2006). To capture speculative behavior we need to expand our perspective to account for land market transactions and expectations.

We use a present value formulation of land prices similar to previous studies to decompose forestland prices in its different components (Shiller 1981, Burt 1986, Tegene and Kuchler 1991, Engsted 1998). Forestland prices can be expressed as follows:

$$P_{it}^{F} = EDR_{it}^{F} + EDR_{it}^{P} - EDCC_{it} \quad \textbf{(3-1)}$$

In eq. (2-1), the price of forestland at location *i* at time *t*, P_{it}^{F}, is the sum of the expected discounted stream of forestland rent, EDR_{it}^{F}, and the discounted stream of rents of the most profitable alternative land use option (e.g. pasture), EDR_{it}^{P}, net of the expected discounted conversion costs, $EDCC_{it}$.[8]

At a given location in the landscape, the market price of forestland thus depends on whether and when conversion occurs. To reflect this probabilistic notion we define the discounted stream of rents from forestland as a function of key components at a given time *t*: the pure forestland rents (R_{it}^{F}), a probability of conversion from forest to pasture at the beginning of time *t* (ρ_t), and a discounted rate (r_t), so that:

$$EDR_{it}^{F} = (1-\rho_0)R_{i0}^{F} + \frac{(1-\rho_0)(1-\rho_1)R_{i1}^{F}}{1+r_1} + \frac{(1-\rho_0)(1-\rho_1)(1-\rho_2)R_{i2}^{F}}{(1+r_1)(1+r_2)} + \cdots \quad \textbf{(3-2)}$$

Pasture rents accrue only after the forestland has been converted to pasture. Under the same assumptions as above, the discounted stream of pasture rents becomes a function of pasture rents (R_{it}^{P}), and the same probability of conversion and a discounted rate as for EDR_{it}^{F} apply, so that:

$$\begin{aligned} EDR_{it}^{P} = {} & \rho_0\left[R_{i0}^{P} + \frac{1}{1+r_1}R_{i1}^{P} + \frac{1}{(1+r_1)(1+r_2)}R_{i2}^{P} + \cdots\right] + \\ & (1-\rho_0)\rho_1\left[\frac{1}{1+r_1}R_{i1}^{P} + \frac{1}{(1+r_1)(1+r_2)}R_{i2}^{P} + \cdots\right] + \\ & (1-\rho_0)(1-\rho_1)\rho_2\left[\frac{1}{(1+r_1)(1+r_2)}R_{i2}^{P} \right. \\ & \left. + \frac{1}{(1+r_1)(1+r_2)(1+r_3)}R_{i3}^{P} \ldots\right] + \cdots \end{aligned} \quad \textbf{(3-3)}$$

To reduce complexity, we assume R_{it}^{F}, R_{it}^{P}, ρ_t, and r_t are constant over time. In addition, expected discounted conversion costs ($EDCC_{it}$) depend also on the probability of conversion and the discount rate but additionally in a cost, τ, which we assumed to be constant in time and space and only accrue at the point of conversion from forest to pasture land, then:

$$EDCC_{it} = \rho\tau + \frac{(1-\rho)\rho\tau}{1+r} + \frac{(1-\rho)^2\rho\tau}{(1+r)^2} + \cdots \quad \textbf{(3-4)}$$

[8] For expositional reasons we use pasture as the only alternative land use in our model.

Substituting eq. (2-2) – (2-4) in eq. (2-1) and all our assumptions combined allow us to construct the current price of forestland as follows (see also section A.1.a):

$$P_{it}^{F} = \frac{(1+r)(-r\rho\tau + (r - r\rho)R_i^F + (1+r)\rho R_i^P)}{r(r+\rho)} \quad \textbf{(3-5)}$$

When conversion probability ρ equals zero, the forestland price is absent of any speculative behavior related to future land conversion, i.e. forestland price depends purely on discounted forestland rents. Further, equation (2-5) emphasizes that even when forestland rents remain unchanged, forestland prices change if the conversion probability, conversion costs, or pasture rents change.

Comparative static analysis of the expression in equation (2-5) (see A.1.a) leads us to the following hypotheses:

H.1) Expected improvements and investments in infrastructure will affect expected net rents from alternative uses and will, thus, increase the forestland price by increasing the probability of conversion.

H.2) Increases in expected conversion cost, for example, due to improved conservation policy enforcement:

a. decrease the forestland price regionally (i.e. land market region), because expected rents from forest conversion are reduced through a lower conversion probability and/or increased conversion costs
b. can increase the forestland price globally (i.e. our study area) if policies focus on sub-regions (i.e. areas inside the Brazilian Legal Amazon region in a land market) and actors in the land market anticipate future policy-induced land scarcity through increased (global) pasture rents (speculation-induced policy leakage).

H.3) Any increase in output prices or decrease in input prices will increase the forestland price through the rent component of forest or pasture.

3.3. Empirical Strategy and Data

Since we cannot directly observe the key components of our theoretical model, we empirically decompose forestland prices according to hedonic theory (first exposed by Rosen 1974) in order to test our hypotheses. In our context, hedonic modeling rest on the assumption that the price of a parcel of land is the sum of the unobserved prices of a bundle of attributes associated with that good (Snyder *et al* 2008). We thus account for heterogeneity in the quality of land and, using panel data, for changes in key attributes that we hypothesized to affect land prices (Chicoine 1981, Sills and Caviglia-Harris 2009), see details in section A.1.a.

Following this notion we can specify a reduced-form model of forestland prices:

$$P_{it}^{F} = \sum_{n=1}^{N} \alpha_n R_{nit} + \sum_{j=1}^{J} \gamma_j S_{jit} + d_t + \mu_i + \epsilon_{it} \qquad \textbf{(3-6)}$$

Here P_{it}^{F} represents forestland prices in region *i* at time *t* as a function of attributes that are averaged at the location, e.g. land market region. R_{it} is a vector of *N* attributes related to forestland and pasture rents and conversion costs. S_{it} is a vector of *J* attributes with influence on the probability of conversion, i.e. our indicators of speculation and conservation policy. In eq. (3-6) α_n and γ_j are vectors of parameters to be estimated. All specifications are estimated as two-way models in log-log form including vectors of time (d_t) and individual (μ_i) fixed effects to capture unobserved year and region specific factors (Baltagi 2016). ε_{it} represents an idiosyncratic error term. In our first specification, we estimate forestland prices by considering attributes that affect land rents and disentangle the effect of speculation. That is, the term S_{it} in eq. (3-6) only has our speculation related variable (*J*=1). In a second specification, we estimate the same model as before but additionally including our proxy for conservation stringency which allows us to test potential leakage effects. In this second specification S_{it} includes two variables affecting the probability of conversion (*J*=2). As robustness check, we use the first lag of all covariates instead of the contemporaneous values for both specifications (see section A.1.f). We point out that our contemporaneous model does not consider the year 2001 so that the results of the contemporaneous and lagged models can be comparable.

Our units of observation are land market regions in the Amazon and Cerrado biomes (61 out of 133 in the whole Brazil), for which average forestland prices were collected between 2001 and 2012 (see Figure A.1.1 in A.1). Land market regions differ in size, number of sample points, and types of land considered, e.g. easy/difficult access Amazon forest or dense/open Cerrado (see also Figure A.1.2 in A.1). During our period of study, major infrastructure investments and forest governance reforms were announced and partially implemented in our study area (Reid and Cabral de Souza Jr. 2005, Nepstad *et al* 2014). First, the federal government published two multiannual development plans between 2000 and 2007, and in

2007, the Ministries of Transport and Defense published a National Plan on Logistics and Transportation (MP 2004, Zioni and Freitas 2015). These plans provide information on expected improvements and constructions in the federal road network. Among these are investments that aim to connect isolated agricultural areas (pavement of highway BR-319 in Amazonas state) or to facilitate exports from well-developed agricultural areas (pavement of highway BR-163 in Mato Grosso and Para). Some studies suggest that these infrastructure projects fueled land speculation and associated forest loss (Fearnside and de Alencastro-Graça 2006, Fearnside 2007). Second, a structural forest governance reform was launched in 2004 with the publication of the Action Plan for Prevention and Control of Deforestation in the Legal Amazon (PPCDAm). By 2016, deforestation in the Amazon biome was 71% lower than in 2004 (INPE 2017), which has been attributed largely to the PPCDAm and accompanying private sector governance measures, such as the soy moratorium (Arima *et al* 2014, Nepstad *et al* 2014, Cisneros *et al* 2015).

To test the hypotheses laid out above, we choose variables that influence the three components of our conceptual framework (eq. (3-5)), i.e. land use rents (e.g. crop prices), conversion costs (e.g. environmental fines), and probability of conversion (e.g., expected improvements in accessibility due to road infrastructure; stringent conservation policy). Summary statistics of our unit of analysis and all variables used in the empirical estimation are presented in Table 3-1. Details on data processing steps are documented in section A.1.b.

The two variables of interest in our analysis are those affecting the probability of forestland conversion component, ρ, as we assume they affect the expectation of land conversion among land market actors. First, we use information on existing and planned roads to calculate *expected accessibility improvements* to relevant markets (i.e. municipality capitals) as a source of speculative behavior**.** We expect land users to adjust their future land rent expectations based on expected road infrastructure improvements, which should be reflected in forestland prices. Second, we construct the variable *Post2004_Conservation* to capture the effect of time and biome-specific changes in conservation governance as follows: *Post2004_Conservation: Dummy PPCDAm x Area share of region outside the Brazilian Legal Amazon x Share of forest area suitable for soy production*; where *Dummy PPCDAm* takes values of 1 for years after 2004 and 0 otherwise. This second variable of interest acts like a treatment effect indicator that identifies agriculturally suitable Cerrado regions as treated from 2005 onwards. Unless there were other significant structural changes affecting any region separately in this particular year, the indicator picks up changes in the behavior of land prices in the Cerrado that were induced by more rigorous conservation policy implementation in the Amazon region (i.e. leakage).

Based on our theoretical model, we expect (1) positive forestland price shifts in target areas of planned infrastructure investments (hypothesis H.1), (2) negative shifts in areas affected by forest governance measures (H.2a), and (3), positive shifts in the presence of conservation policy leakage in regions with comparatively little change in *de facto* governance effectiveness (H.2b).

Table 3-1. Summary of variables and sources used

Variable (units)	Source	Obs	Mean	St. Dev.	Min	Max
Land market region size (km^2)	FNP (http://fnp.com.br/); Own calculation	79	90,588.540	129,045.000	7,147.439	795,965.700
Forestlan price ($R/ha)	FNP (http://fnp.com.br/)	756	467.594	434.631	8.702	2,785.743
Expected Accessibility Improvements (hrs)	Own calculation; DNIT; Hansen *et al* (2013b)	948	-0.497	1.255	-6.659	0.000
Agriculture price index	Own calculation; IBGE (http://sidra.ibge.gov.br/)	948	0.400	0.212	0.054	1.797
Soy aptitude within forest areas (share of region)	Own calculation; Soares-Filho et al.(2016); Hansen *et al* (2013b)	948	0.086	0.117	0.000	0.536
Protected areas (share of region)	Brazilian Ministry of Environment	948	0.077	0.111	0.000	0.509
Cattle density (heads/km2)	Own calculation; IBGE (http://sidra.ibge.gov.br/)	948	0.365	0.270	0.002	1.117
Accessibility (hrs)	Own calculation; DNIT; Hansen *et al* (2013b)	874	4.263	5.320	0.000	24.489
Fines incidence (#/(10 x km2))	IBAMA (http://www.ibama.gov.br)	948	0.031	0.046	0.000	0.557
Districts outside the Brazilian Legal Amazon (share of region)	Own calculation; IBGE	948	0.417	0.487	0.000	1.000
Post2004_Conservation (0/1)	Own calculation	948	0.750	0.433	0	1

3.4. Results

3.4.a. Descriptive analysis

Figure 3-2 below depicts the forestland price dynamics and deforestation rates for land market regions located in different biomes: Amazon forest, Cerrado savannah, and regions with both biomes.[9]

Average forestland prices (upper panel in Figure 3-2) for the three groups were on the rise up to 2004. The implementation of the PPCDAm was accompanied by forestland price reductions across regions (see also Figure A.1.2 in A.1). Yet, forestland prices in the Cerrado clearly rose in subsequent years to levels seven times higher than in the Amazon region in 2012. Note also that land prices were relatively stable in regions with both biomes up until 2010, when they began to rise, and doubled by 2012. This increase coincides with the political debate that led to the reform of the Forest Code and associated amnesties for past forest law offenders (Soares-Filho *et al* 2014).

The lower panel in Figure 3-2 illustrates deforestation rates measured as the percentage change of tree cover in the three types of regions (Hansen *et al* 2013b). After 2004, deforestation rates dropped particularly in regions with historically high levels of forest loss (see Figure 2-4 below and Figure A.1.3 in A.1). Another pronounced reduction in these regions occurred between 2008 and 2009. In these years additional public and private sector initiatives reinforced conservation stringency leading to further reductions in deforestation rates (Arima *et al* 2014, Cisneros *et al* 2015). Meanwhile, deforestation rates remained relatively stable in Cerrado regions.

[9] This last group of regions is located within a highly dynamic area, the so-called “Arc of Deforestation”.

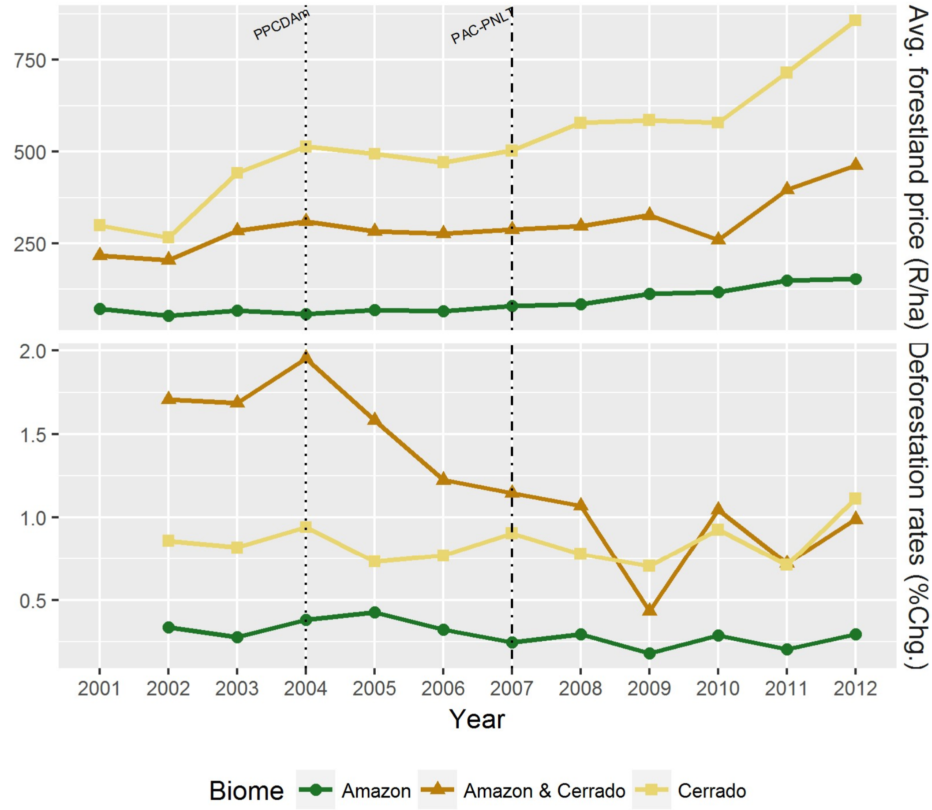

Figure 3-2. Forestland prices and deforestation during the study period
Note: We differentiate three types of regions: within Amazon, within Cerrado, and those overlapping with both biomes. All prices were deflated including the base year 2001. The vertical dotted line refers to the year in which the PPCDAm was implemented. The dotted-straight line represents the year in which a national plan on logistics and transportation was issued.

3.4.b. The speculative component of forestland prices

According to our theoretical model, speculation, represented as an increase in the conversion probability due to market actors' anticipation of land appreciation, will increase forestland prices.

Column 1 in Table 3-2 reports our main results of estimating the respective specification of equation (2-6) considering price attributes that affect rents, conversion costs, and the speculation component of land prices (see A.1.e for extended results). We find that regions with lower average crop prices and high concentration of environmental-related fines tend to exhibit lower forestland prices (as expected by our hypotheses H.3 and H.2a, respectively). Environmental fines are negatively associated with the forestland price, reflecting conversion costs. Due to the log-log specification, we interpret estimated coefficients as elasticities of forestland prices with respect to its corresponding variable (Wooldridge 2013, p 44). Looking at our indicator of speculation (i.e., expected accessibility improvements), the estimated coefficient is significant at the 5% level and positive, i.e. cutting expected travel time from a location to the nearest market by 1% (0.6 minutes) increases the regional forestland price by 1.5%. This finding indicates speculative behavior in land markets hinting toward the future location of agricultural frontiers and corroborates our hypothesis H.1.

Table 3-2. Regression results of speculation and stringent conservation analysis

	*Dependent variable: ln*Forestland price		
	Speculation	Stringent Conservation	Model Component
	(1)	(2)	
*ln*Expected Accessibility Improvements	1.541[b]	1.530[b]	Speculative (ρ)
	(0.760)	(0.740)	
*ln*Crop price index	0.398[b]	0.416[b]	Rents (R)
	(0.197)	(0.197)	
*ln*Soy apptitude	4.697	6.594	Rents (R)
	(4.768)	(4.896)	
*ln*Protected areas	1.108	1.127	Cost of conversion (τ)
	(0.858)	(0.847)	
*ln*Cattle density	0.791	0.979	Rents (R)
	(0.693)	(0.694)	
*ln*Accessibility	0.516	0.268	Rents + Cost of conversion (R+τ)
	(0.963)	(0.960)	
*ln*Fines incidence	-1.459[a]	-1.440[a]	Cost of conversion (τ)
	(0.397)	(0.391)	
*ln*Accessibility x nonBLA♦	-0.587	-0.298	Rents + Cost of conversion (R+τ)
	(1.090)	(1.105)	
Post2004 Conservation		1.677[c]	Stringent Conservation (ρ)
		(0.887)	
Time and regional fixed effects	Yes	Yes	
Observations	682	682	

R^2	0.091	0.100
F Statistic	7.503[a] (df = 8; 602)	7.392[a] (df = 9; 601)

a Significant at 0.01 level. b Significant at 0.05 level. c Significant at 0.1 level. Robust standard errors are given in parentheses. ♦nonBLA refers to the share of area outside the Brazilian Legal Amazon region in a land market region.

3.4.c. Land prices and conservation policies

To explore the effects of regionally focused conservation policy interventions we add the policy shock variable to the model (second column Table 3-2).

Our previous results remain stable and our post-2004 policy indicator is significant at the 10% level and on average positively associated with forestland prices.[10] Our post-2004 conservation variable is associated with an increase in forestland prices by 1.6% on average. Assuming no bias from unobserved variables (see discussion below), this finding speaks to our hypothesis H.2b, i.e. speculation-induced conservation policy leakage to regions and areas that are less controlled or not protected by law. This would primarily affect regions with large reserves of agronomical suitable forestland (e.g. Cerrado areas).

Our results reflect the immanent trade-off between conservation and agricultural development in the Brazilian agricultural frontier. Without increases in environmental law enforcement (here measured in terms of fine incidence), road infrastructure expansion tends to increase land demand, which is associated with deforestation.

3.4.d. Policy relevance and speculation

Figure 2-3 depicts the 2001-2012 average effect of expected improvements in road infrastructure on forestland prices while keeping all other covariate effects constant. Our model thus serves to identify speculation zones that potentially require additional scrutiny by environmental law enforcement agencies. Some of these zones happen to lie outside the Legal Amazon region, where regulations are less stringent. Here the risk of developing into future deforestation hotspots can be comparatively high. Visual comparison with the dynamics of deforestation hotspots after implementation of the PPCDAm (Figure 2-4), confirm this conjecture only for some speculation zones, such as along the BR-163 in the states of Mato Grosso and Pará and in the so-called MATOPIBA region at the eastern border

[10] As mentioned in the Empirical Strategy section, our contemporaneous model does not consider the year 2001. We run an alternative contemporaneous model that includes the year 2001 and found that our conservation policy variable became marginally insignificant (p-value of 0.1145) pointing to limited robustness of this finding. We present this version of the model in Table A.1.3.

of our study area (i.e., the region in four states where soybean production is expanding: southern Maranhão (MA), Tocantins (TO), southern Piauí (PI), and western Bahia (BA)).[11] This observation shows that various factors have to come together for land market speculation to result in deforestation and deserves further research.

In sum, our findings suggest that land market prices in Brazil are not merely governed by expectations on rents and forest conversion costs (hypotheses H.1 and H.2b). Expectations on future infrastructure improvements and conservation policy-driven land scarcity are likely to be priced into today's land market transactions.

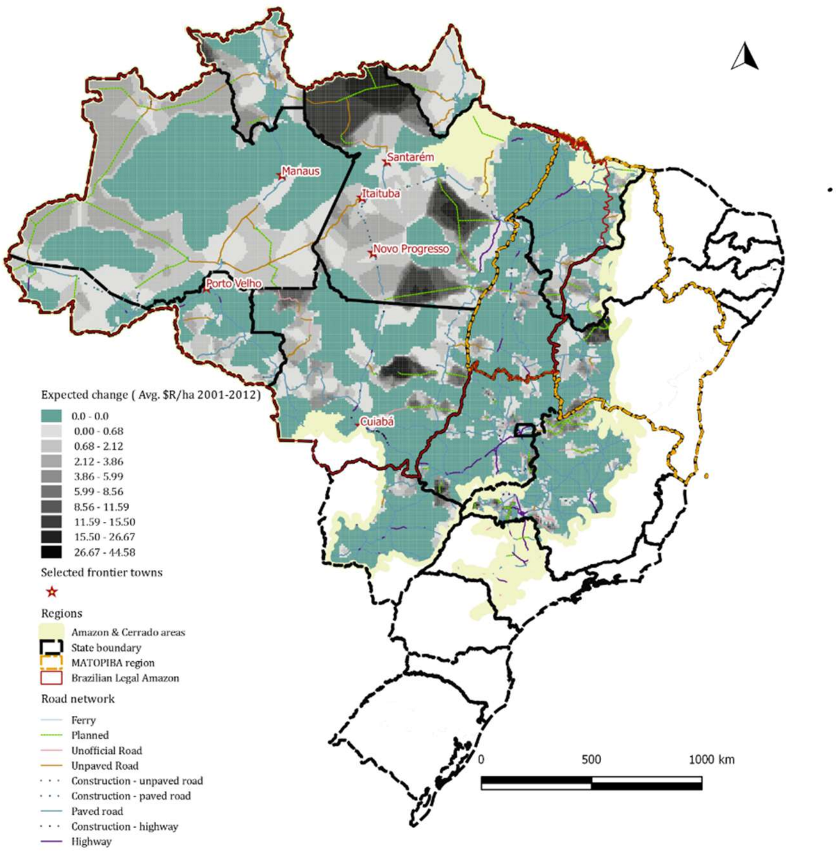

Figure 3-3. Effect of expected changes in accessibility on forestland prices
Note: We use the coefficients estimated in our second model together with a map of expected accessibility improvements at a 10 x 10km raster resolution to obtain percentage changes holding all other factors constant. We then multiply this map with that of average prices in land market regions.

[11] See A.1.g in A.1 for a description on how our deforestation hot spots map is generated.

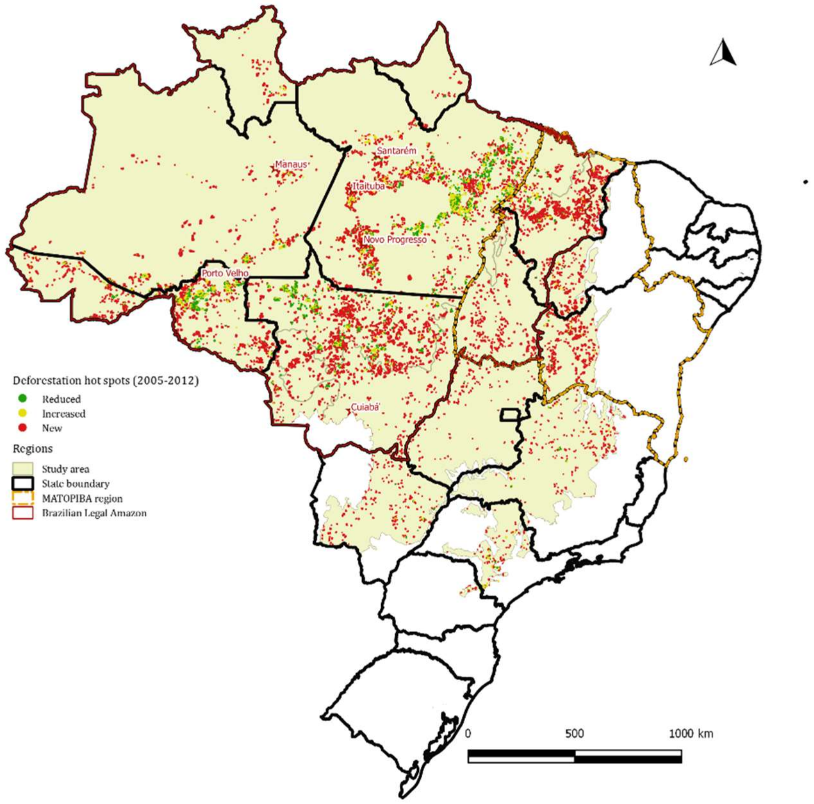

Figure 3-4. Hot spots of forest cover loss (2005-2012)
Note: This figure depicts deforestation hot spots in Amazon and Cerrado biomes between 2005-2012. Three types of hot spots are identified: a) Reduced (green), b) increased (yellow), and c) new (red). High concentration of new hot spots is located in eastern parts of Cerrado in the MATOPIBA region, as well as in Mato Grosso and Pará along the BR-163.

3.5. Conclusion and discussion

We have developed a theory of land market price formation at agricultural frontiers that explains why forestland prices can contain information about future expectations of land market actors. The subsequent empirical analysis using a panel data set of forestland prices and their determinants shows that land markets: (1) convey information about anticipated infrastructure improvements (hypothesis H.1), (2) may indicate conservation policy leakage between regions with heterogeneous levels of legal protection and policy enforcement effectiveness (hypothesis H.2b) - though this finding is less robust to alternative model specifications than the first. This chapter contributes to the debate in indirect land use change (Hertel 2018) by scrutinizing the potential role of land markets both as mechanisms behind land use leakage and as an early warning system to anticipate future deforestation hotspots.

It is worth noting that land market speculation may or not require policy action depending on its social and environmental implications. For example, depending on asymmetries in bargaining power between buyers and sellers, speculative land market transactions may result in suboptimal outcomes for poor smallholders with insecure property rights (Baletti 2012). Moreover, in contexts where deforestation is a means to secure land claims, land market speculation may be associated with irrationally high levels of forest conversion. Speculation thus eventually becomes a mechanism that complements market-based leakage to the extent of neutralizing direct conservation policy effects, as our results seem to suggest for the behavior of forestland prices. Preemptive and spatially targeted policy action may thus sometimes be necessary to counteract potentially negative social and environmental outcomes of land market speculation.

A number of caveats apply, which can be addressed in future research, but should be taken into account when interpreting our findings. First, our indicator of expected infrastructure improvements only accounts for primary road expansion and ignores other important planned infrastructure investments, such as in the mining and energy sectors. It is well known from the literature that secondary roads contribute a great deal toward improving accessibility in agricultural frontier development (Arima *et al* 2008, Perz *et al* 2008, Walker *et al* 2011). While this may have led us to underestimate speculation, one should keep in mind that land market actors may not take infrastructure investment plans at face value, given that implementation often lags behind actual plans (Amann *et al* 2016). Second, our policy shock indicator (representing the implementation of PPCDAm) is imperfect in that it captures more than just policy shocks. We can only argue that this policy event has probably dominated land market dynamics in subsequent years, but our results are likely to be simultaneously driven by other unobserved macro-changes. Follow-up research requires land price data at higher spatial resolution (Coomes *et al* 2018) and should focus on directly linking land price dynamics to deforestation patterns.

4. Market-driven land use intensification in the Brazilian agricultural frontier: Cross-sectional analysis of farming production and land use decisions at the district level

Chapter presented at the 25th Annual Conference of the European Association of Environmental and Resource Economists (Virtual format conference). In collaboration with Andrea Zimmermann [a,c] and Jan Börner [a,b].

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Abstract

Tropical forest loss remains a major environmental trade-off vis-à-vis the provision of food, feed, fiber, and bioenergy for an increasingly affluent world population. The effectiveness of agricultural intensification as a key strategy to manage and mitigate this trade-off is, however, contested. Increasing productivity can lead to either land expansion or land-saving depending on context-specific factors, such as land and output market conditions, the type of technology being used, and institutional arrangements including conservation policy. In Brazil, periods of coupling and decoupling between agricultural intensification and deforestation rates have been documented depending, among other factors (e.g. exchange rate fluctuations), on the quality of forest governance. This chapter scrutinizes economic mechanisms behind intensification and land use decisions based on agricultural census data from the Brazilian state of Mato Grosso, a region known for large-scale soy and cattle production with both consolidated and unconsolidated agricultural frontiers taking place in three different biomes. Exploiting the duality properties of the profit function, we estimate a system of equations representing the technology that underlies output supply and factor demand decisions of an average farm in Mato Grosso in 2006, at the onset of Brazil's forest governance reform, which led to a reduction of Amazon forest loss by 80% between 2004 and 2012. We explicitly consider one of the reforms' main implementation mechanisms, i.e. fines for environmental offenses, to consistently estimate output supply and factor demand elasticities, including for demand for forestland. Our results confirm narratives of market-driven intensification in the agricultural frontier, but also suggest sizable expansion at the extensive margin of land. We also find evidence for conservation policy-driven intensification; however, these effects are of small magnitude, especially regarding land use decisions. Hence, unless environmental policies provide effective conservation incentives, a favorable economic context for crop and cattle production, such as increasing global demand for commodities or infrastructure investments that reduce transport costs, is likely to result in further legal and illegal encroachment of cropland and pastures on remaining forests in Brazil's agricultural frontier.

Keywords: duality theory; induce intensification; deforestation

4.1. Introduction

Production of highly demanded farming products has been associated with conversion of forest in different areas across the globe. While the provision of biomass products is essential for humankind's prosperity, it involves trade-offs due to loss of natural areas (DeFries and Rosenzweig 2010). Technological change to increase productivity of land at agricultural frontiers, i.e. intensification, is seen as a solution to reduce pressure on natural vegetation and related ecosystem services; an idea known as *Borlaug hypothesis* (Green *et al* 2005, Burney *et al* 2010, Phalan *et al* 2011, Stevenson *et al* 2013). This idea has been contested, however, because higher profits from cropland relative to other land uses increase farm income, which may lead to an expansion of agricultural areas (Angelsen 2010). This rebound effect is also known as *Jevon's paradox*, as the 19th-Century economist observed a similar effect in aggregate use of coal following an increase in its productivity (Alcott 2005, Villoria *et al* 2014).

At global and regional scales evidence has shown that intensification reduced rates of agricultural expansion, seemingly a support for Borlaug hypothesis (Stevenson *et al* 2013). Yet with its impact being less than expected studies found little evidence to support the idea that intensification brings cropland reductions (Rudel *et al* 2009). Rare examples were in areas in which imports for crops were available or effective conservation governance was in place (Rudel *et al* 2009, Ceddia *et al* 2013). At a local level, case studies suggest ambivalent outcomes in terms of how intensification affects expansion of cropland (Villoria *et al* 2014).

Economics offers an explanation as to why we may observe one or the other phenomenon depending on the local context. In the case of an effective technological change, supply of that product will be expanded with its related increase in output and reduction in prices; yet, whenever demand for the crop produced is elastic, change of prices would not be low enough to deter incentives for expansion due to higher rents of farmland (Angelsen 2010, Hertel 2011). To what degree this phenomenon takes place would be based on opportunities for expansion (extensive margin) or for factor substitution (intensive margin) (Lichtenberg 2002, Hertel 2018). Opportunities differ depending on the spatial distribution of technological improvement or location-specific level of endowments. New agricultural frontier areas are prone to exploit extensive margins, while in consolidated agricultural areas intensive margins are more likely to be observed (Schielein and Börner 2018). This economic explanation helps to understand what evidence at different scales has shown. As global demand for agricultural products is rather inelastic, technological change has reduced rates of cropland expansion at consolidated agricultural frontiers (Rudel *et al* 2009). At new agricultural frontiers, farmers are confronted instead with more elastic demands, and often high land endowments (Villoria *et al* 2014).

Land use intensification (e.g. increase of yields per unit of land) is related to technological and market drivers (Mundlak 2001, Byerlee *et al* 2014). First, technological progress refers to the possibility of producing the same volume of agricultural products using fewer factors of production (e.g. variable inputs) in a fixed piece of land. Market-driven intensification happens when there is a substitution of factors of production (e.g. more fertilizer for land) when relative scarcity brings changes in relative prices (Hayami and Ruttan 1970a). The second channel of market-driven intensification is given by higher product prices and better market opportunities that incentivize different mix production (e.g. land uses more intense in fertilizers or labor for uses extensive in land). The underlying mechanisms of expansion or intensification in the use of land are based on both land endowments and elasticities of substitution between different factors of production.

Brazil has both large land endowments and an evolving agricultural sector which products have gained importance in international markets (FAO 2013, Vieira Filho and Fornazier 2016). This has been possible not only by innovative technological changes in its agricultural industry but also, through expansion of land use in its production (Andersen *et al* 2002, Pardey *et al* 2004). The land use at the intensive or extensive margin has been coupled with changes in the economic context (Walker *et al* 2009, Richards *et al* 2012, Fearnside 2017). In periods of soaring crop prices, land use at the extensive margin is related to higher yields (Garrett *et al* 2013b). In contrast, cropland expansion was reduced at times of less favorable prices (Richards *et al* 2012).

Together with innovations in the agricultural sector, governance is essential to regulate land use at the extensive and intensive margins. Particularly in the South American country, effective conservation governance brought down consistently rates of deforestation in tropical Amazonia from 2004 till 2012 (Arima *et al* 2014, Nepstad *et al* 2014).[12] Mechanisms in place to halt deforestation created an artificial land scarcity for farming production, thus affecting

[12] In 2004, the Brazilian government started a regional initiative, the Action Plan for Prevention and Control of Deforestation in the Legal Amazon (PPCDAm). This plan called for actions on issues such as land tenure, revision of economic incentives in relation with sustainable agriculture and forest management, and guidelines for ensuring the environmental sustainability of infrastructure projects in the transport and energy sectors May *et al* (2011, p 23). It is based in three major pillars: a) tenure regularization; b) monitoring and control; and c) incentives for sustainable production Gebara and Thuault (2013).

the potential to use extensive margins. It is worth noting that an effective implementation needs financial and institutional support that can strengthen its credibility (Rochedo *et al* 2018, Soares-Filho and Rajão 2018). In Brazil after a change in conservation governance in 2004, punishments towards illegal deforestation were credible and later reduced their credibility when negotiations to reform the conservation legal framework started in 2010. The new FC was approved in 2012 (Soares-Filho *et al* 2014, Roriz *et al* 2017, Freitas *et al* 2018).[13] Regarding intensification as a means to reduce deforestation in the country, some studies have warned that without conservation in place, production would rather take place at the extensive margin (Fearnside 2002b, Merry and Soares-Filho 2017). Additional evidence shows that when conservation governance is stronger, agricultural production is decoupled from deforestation and enables synergies between agricultural productivity and conservation objectives (Macedo *et al* 2012, Gollnow and Lakes 2014, Koch *et al* 2019).

This chapter adds evidence on the effects of a changing economic context in farming production, and land use decisions in the Brazilian agricultural frontier, which leads to trade-offs between biomass production and conservation. The question of interest in this chapter is: do agricultural prices induce market-driven land use intensification in the Brazilian agricultural frontier? This general question is further expanded to answer the sub-questions 1) how and 2) to what extent price changes shape production and land use decisions, particularly regarding pressure into forest areas? We analyzed the state of Mato Grosso to tackle these questions. Established and new agricultural frontiers exist in this state (Schielein and Börner 2018). It covers an area of approximately 906,000 km^2 divided into three biomes: tropical Amazon forest (~53%), Cerrado savannah (~40%), and Pantanal (~7%) (Kastens *et al* 2017). Large areas for agricultural activities and the implementation of new technologies have made this state a powerhouse of biomass production hosting 14.7% of total Brazilian planted land by 2009 (Arvor *et al* 2012). Moreover, total cropland area in Mato Grosso went from 41,600 km^2 in the period 2000-2001 to approximately 72,761 km^2 in the period 2006-2007 (Arvor *et al* 2012).

Most empirical models looking at drivers of land use and deforestation in the Brazilian agricultural frontier are based on a reduced-form single equation estimation which cannot identify changes in factor substitution but only individual correlations between products,

[13] In 2010, agricultural lobbyists started to negotiate a new FC that became effective in 2012. In this new law framework some new market mechanisms were established which are seen as positive improvements, yet, amnesty to offences before 2008's deforestation was given setting seed for expectations of low conservation governance.

land, and the economic context (Marchand 2012, Barreto *et al* 2013, Dalla-Nora *et al* 2014). This study aims to fill this gap and explicitly estimate direction and magnitude of factor substitution in agricultural frontier areas due to changes in agricultural markets. A structural economic model is used to infer the underlying technology driving these substitution decisions in the agricultural system. Two systems of equations are estimated using a seemingly unrelated regressions (SUR) econometric model in the empirical analysis. Resulting parameters are used to calculate an average farmer's sensitivity to changes in the economic environment. An important assumption is that land is considered a quasi-fixed input of production such that farmers can reassign different land uses within the boundaries of a short-run fixed total land endowment. The structural economic model is used to derive hypotheses on expected linkages between price changes and production, and related land use decisions. An important premise for this analysis is that conservation policies were effectively implemented in the area of study implying stronger land scarcity effects in the agricultural system. This seems reasonable as this study used cross-section information for the year 2006, a time in which production increase has been partially decoupled from deforestation due to an effective enforcement of conservation (Macedo *et al* 2012, Gollnow and Lakes 2014). Our results show that intensification, but also expected deforestation, is partially induced by agricultural markets. We also show that conservation policy drives intensification while protecting forest areas; however, these effects are of small magnitude in our empirical analysis.

Following this introductory section, the present chapter is organized as follows. The next section presents the structural economic model on which the analysis is based and lays down hypotheses on how the economic context affects the agricultural system in agricultural frontier areas. The third section describes the chosen empirical strategy and the data used in the analysis. A fourth section presents our results. The last section discusses implications for conservation and development policy to tackle trade-offs between agricultural development and nature conservation. It also identifies further avenues of research in order to improve future analyzes of deforestation in the agricultural frontier.

4.2. Theoretical framework and empirical specification

4.2.a. Profit function optimization with land as a quasi-fixed input

A multi-output multi-input short-run profit maximization model is used to analyze Mato Grosso's agricultural system and its sensitivity to changes in agricultural markets. The model frames farming decisions to price changes based on a profit function optimization approach which allows reallocation of land uses within a fixed total land endowment.[14] The model explains a farmer's decision making in two stages (Chambers and Just 1989, p 981, Arnade and Kelch 2007). In a first stage, farmers maximize profits for each crop produced determined by assigned (quasi-)fixed inputs in their production. In a second stage, farmers reallocate their quasi-fixed inputs (e.g., land) optimally given a multiproduct technology. The optimal allocation of quasi-fixed inputs is reached when the shadow prices of these inputs are equal across all different outputs of production (Moore and Negri 1992, Arnade and Kelch 2007).

The farmer's maximization problem is represented by an indirect restricted short-run profit function, $\pi(\cdot)$. This function depends on a vector of expected output prices $\mathbf{p}$ for i outputs; a vector of input prices $\mathbf{w}$ of j inputs; a vector of m fixed inputs, $\mathbf{z}$ (which may include bio-physical characteristics and constraints, and conservation policies); and $\mathbf{L}$, a vector of h land use allocations. Further, $\mathbf{y}$ represents a vector of i outputs and $\mathbf{x}$ a vector of j inputs. $Y(\cdot)$ indicates the output technology set for different configurations of inputs and (quasi-)fixed factors and is assumed to be compact, non-empty, and convex in inputs.

$$\pi(\mathbf{p},\mathbf{w},\mathbf{z},\mathbf{L}) = \max_{\mathbf{y},\mathbf{x}}\{\mathbf{p}'\mathbf{y} - \mathbf{w}'\mathbf{x} \colon \mathbf{y} \in Y(\mathbf{x},\mathbf{z},\mathbf{L})\} \qquad \textbf{(4-1)}$$

This representation becomes more general if one drops the assumption of having one output per land use. This generalization leads to a better representation of a system in which one unit of land can be used for more than one output as in the case of the Brazilian agricultural frontier (Fezzi and Bateman 2011, Brown *et al* 2013). The model considering different land uses, a_k for k in 1, …, h, is represented as:

$$\pi(\mathbf{p},\mathbf{w},\mathbf{z},a_1,\ldots,a_h) = \max_{\mathbf{y},\mathbf{x}}\{\mathbf{p}'\mathbf{y} - \mathbf{w}'\mathbf{x} \colon \mathbf{y} \in Y(\mathbf{x},\mathbf{z},a_1,\ldots,a_h)\} \qquad \textbf{(4-2)}$$

This profit function has certain properties that enable the researcher to make inferences about the underlying technology of production (Chambers 1988). The function should be linearly homogenous of degree one in prices, monotonically increasing (decreasing) in output (variable input) prices, monotonically increasing in total land endowment, and convex in output and input (i.e., netput) prices (Diewert and Wales 1987, Guyomard *et al* 1996). The

[14] This model was first proposed by Chambers and Just (1989) and expanded by Arnade and Kelch (2007), and Fezzi and Bateman (2011) in previous land-use allocation studies.

profit function associated with the optimal land allocation constrained to a total land endowment **L** is:

$$\pi(\mathbf{p},\mathbf{w},\mathbf{z},L) = \max_{a_1,\dots,a_h}\left\{\pi(\mathbf{p},\mathbf{w},\mathbf{z},a_1,\dots,a_h): \sum_{k=1}^{h} a_k = L\right\} \qquad \textbf{(4-3)}$$

As this function maintains the well-behaved properties of a restricted profit function, production and demand can be derived by Hotelling's lemma.[15] Taking the first derivative of the profit function with respect to output price p_i, one obtains the output supply of commodity *i*, y_i. Equivalently, the negative input demand for factor of production *j*, x_j, is obtained by differentiating the profit function with respect to input price p_j. The system of equations for outputs and inputs (netputs) are:

$$y_i(\mathbf{p},\mathbf{w},\mathbf{z},L) = \frac{\partial\pi(\mathbf{p},\mathbf{w},\mathbf{z},L)}{\partial p_i} = \frac{\partial\pi(\mathbf{p},\mathbf{w},\mathbf{z},a_1^*,\dots,a_h^*)}{\partial p_i} \qquad \textbf{(4-4.a)}$$

$$-x_j(\mathbf{p},\mathbf{w},\mathbf{z},L) = \frac{\partial\pi(\mathbf{p},\mathbf{w},\mathbf{z},L)}{\partial w_j} = \frac{\partial\pi(\mathbf{p},\mathbf{w},\mathbf{z},a_1^*,\dots,a_h^*)}{\partial w_j} \qquad \textbf{(4-4.b)}$$

In the previous equations, each a^* represents the optimal land use allocation that solves equation (4-3). The F.O.C. of equation (4-3) with respect to the vector of land uses **a** gives a set of land shadow prices that when equal to each other provide an optimal solution, so that:

$$\frac{\partial\pi(\mathbf{p},\mathbf{w},\mathbf{z},L)}{\partial a_1} = \frac{\partial\pi(\mathbf{p},\mathbf{w},\mathbf{z},L)}{\partial a_k}, \text{for } k = 2,\dots,h \qquad \textbf{(4-5)}$$

Solving for each *a* gives a set of land use functions which depend on netput prices, fixed factors, and total land available, $a_k(\mathbf{p},\mathbf{w},\mathbf{z},\mathbf{L})$, for any *k*. The derivative of any *k* land use function with respect to a netput price p_n is equal to land use *k*'s specific response to price change of commodity *n*, for any *k*=1,…,*h* and *n*=1,…, *i*, *i*+1, …, *i*+*j* (Arnade and Kelch 2007).

From the resulting netput equations (3-4), some properties are derived from the profit function. They have to be continuous in prices as they are the derivatives of a continuous function. The netput equations are monotonic in prices, so that the first derivative to output prices is non-negative and the first derivative to input prices is non-positive. They are homogenous of degree zero in prices, that is, only relative price and not proportional price

[15] If the profit function is differentiable in netput prices the unique profit-maximizing supply and derived-demand functions are as stated in equation (3-4) Chambers (1988, p 126).

changes affect their levels. The first derivatives of the netput equations are symmetric as they are equal to the second derivative of a profit function. As illustration, taking the derivative of product *i* with respect to price *j* is equal as to take the double difference of the profit function with respect to price *i* and price *j*, regardless of the order of differentiation,

$$\frac{\partial y_i(\mathbf{p},\mathbf{w},\mathbf{z},L)}{\partial w_j} = \frac{\partial^2 \pi(\mathbf{p},\mathbf{w},\mathbf{z},a_1,\dots,a_h)}{\partial p_i \partial w_j} = \frac{\partial^2 \pi(\mathbf{p},\mathbf{w},\mathbf{z},a_1,\dots,a_h)}{\partial w_j \partial p_i} = -\frac{\partial x_j(\mathbf{p},\mathbf{w},\mathbf{z},L)}{\partial p_i} \qquad \textbf{(4-6)}$$

4.2.b. Netput and land elasticities

The model disentangles netput responses to changes in prices both when land uses are held fixed, and due to land use reallocation (Chambers and Just 1989, Moore and Negri 1992), such that:

$$\frac{\partial y_i(\mathbf{p},\mathbf{w},\mathbf{z},L)}{\partial p_j} = \left.\frac{\partial y_i(\mathbf{p},\mathbf{w},\mathbf{z},a_1,\dots,a_h)}{\partial p_j}\right|_{\bar{\mathbf{a}}} + \sum_{k=1}^{h} \frac{\partial y_i(\mathbf{p},\mathbf{w},\mathbf{z},a_1,\dots,a_h)}{\partial a_k} * \frac{\partial a_k(\mathbf{p},\mathbf{w},\mathbf{z},L)}{\partial p_j} \qquad \textbf{(4-7)}$$

The same holds for all netputs different from *i* and prices different from *j*. The first element on the right-hand side (RHS) of equation (3-7) represents a compensated change of output's response to p_j with fixed land allocations ($\bar{\mathbf{a}}$). The second element accounts both for changes in output due to land allocation and land allocation readjustments due to a price change of commodity *j*. This decomposition of netput response to prices is used to obtain netput elasticities of the system:

$$\frac{\partial y_i(\mathbf{p},\mathbf{w},\mathbf{z},L)}{\partial p_j} * \frac{p_j}{y_i} = \left\{\left.\frac{\partial y_i(\mathbf{p},\mathbf{w},\mathbf{z},a_1,\dots,a_h)}{\partial p_j}\right|_{\bar{\mathbf{a}}} + \sum_{k=1}^{h} \frac{\partial y_i(\mathbf{p},\mathbf{w},\mathbf{z},a_1,\dots,a_h)}{\partial a_k} * \frac{\partial a_k(\mathbf{p},\mathbf{w},\mathbf{z},L)}{\partial p_j}\right\}\left(\frac{p_j}{y_i}\right) \qquad \textbf{(4-8)}$$

The left-hand side of equation (3-8) represents the *uncompensated* (to changes in land use allocation) netput elasticity of product *i* to p_j. Using the product rule, and representing the netput intensity of product *i* by y_i^l, the first right-hand side term in equation (3-8) can be expressed as:

$$\left.\frac{\partial y_i(\cdot)}{\partial p_j}\right|_{\bar{\mathbf{a}}} * \frac{a_i}{a_i} = \frac{\partial y_i^l(\cdot)}{\partial p_j} * a_i \tag{4-9}$$

We substitute the first RHS term in equation (3-8) by equation (3-9) and we multiple it by the term outside the brackets. The second term in the RHS is multiplied by its respective a_k/a_k. These two terms represent the uncompensated netput elasticity which is expressed by: *i*) a netput intensity elasticity, plus *ii*) the effect on netput levels from land allocation changes multiplied by the response in land use due to price changes (Arnade and Kelch 2007). These steps reformulate equation (3-8) as:

$$\begin{aligned}\frac{\partial y_i(\mathbf{p},\mathbf{w},\mathbf{z},L)}{\partial p_j} * \frac{p_j}{y_i} &= \left(\frac{\partial y_i^l(\cdot)}{\partial p_j} * a_i\right) * \left(\frac{p_j}{y_i} * \frac{1}{a_i}\right) + \left(\sum_{k=1}^{h} \frac{\partial y_i(\cdot)}{\partial a_k} * \frac{\partial a_k(\cdot)}{\partial p_j}\right) * \left(\frac{p_j}{y_i}\right) \\ &= \frac{\partial y_i^l(\cdot)}{\partial p_j} * \frac{p_j}{y_i^l} + \left(\sum_{k=1}^{h}\left(\frac{\partial y_i(\cdot)}{\partial a_k} * \frac{a_k}{y_i}\right) * \left(\frac{\partial a_k(\cdot)}{\partial p_j} * \frac{p_j}{a_k}\right)\right)\end{aligned} \tag{4-10}$$

which is the formula used in the analysis to calculate netput elasticities. To obtain each land use elasticity, the derivative of each $a_k(\mathbf{p},\mathbf{w},\mathbf{z},L)$ with respect to the netput prices is multiplied by the ratio of price to land allocation. For example, the elasticity of land use *k* to a commodity price *j* is:

$$\varepsilon_{p_j}^{a_k} = \frac{\partial a_k(\mathbf{p},\mathbf{w},\mathbf{z},L)}{\partial p_j} * \frac{p_j}{a_k} \tag{4-11}$$

Two main channels induce market-driven intensification: 1) relative scarcity and consequent changes in relative prices brings substitution in factors of production, e.g. fertilizers for land; and 2) when better product prices and market opportunities incentivize a different production mix, e.g. shift production from extensive to intensive uses of land (Hayami and Ruttan 1970a, Hayami 1971, Mundlak 2001, Ruttan 2002, Byerlee *et al* 2014). In the Brazilian agricultural frontier, cattle ranching is related to more extensive use of land relative to crop production; however, both land uses are directly and indirectly related to forest encroachment (Barona *et al* 2010, Arima *et al* 2011, Macedo *et al* 2012). The impact of intensification on forest areas is therefore unclear. First intensification allows production of higher levels of output on the same amount of land which reduces incentives to expand agricultural areas. This higher land productivity can at the same time bring higher profitability in expanding production, and thus deforestation (Angelsen and Kaimowitz 2001, Walker *et al* 2009).

Based on equations (4-10) and (4-11), the following hypotheses are derived and guide the analysis:

H.1) An increase in variable input price reduces intensification reflected by a negative crop intensity elasticity. Cattle production is related to an extensive use of land, therefore it is expected that an increase in input price increases cattle uncompensated and intensity elasticities.

H.2) Better market conditions for crop production, e.g. higher crop prices, induce intensification, and thus it is expected a positive crop and variable inputs intensity elasticities. On the contrary, higher cattle prices are expected to induce extensification, particularly if one observes a positive uncompensated cattle elasticity driven by the land use reallocation effect.

H.3) Market-driven intensification is not reducing incentives to deforest if forestland elasticities are negative with respect to crop and input prices. If also, uncompensated netput elasticities are higher in magnitude than intensity elasticities, land use reallocation due to changes in prices hints to an additional increase in land for agriculture due to better market conditions.

4.2.c. Model specification

Together equations (3-4) – (3-5) represent a system of equations to be solved for optimal netput levels and land use allocations. For the estimation, the system of equations needs to be translated into a unifying framework. This is done by directly specifying the profit function using a determined flexible functional form such as the normalized quadratic (NQ) (Lau 1978, Diewert and Wales 1987). In this chapter, a symmetric normalized quadratic (SNQ) function is applied which is a special case of the NQ function. The SNQ function treats all quasi-fixed factors, inputs, and outputs symmetrically, and can be expressed as (Kohli 1993):

$$\hat{\pi} = \sum_{n=1}^{3} \alpha_n p_n + \sum_{k=1}^{3} \beta_k a_k + 0.5\tau \sum_{n=1}^{3}\sum_{n=1}^{3} \alpha_{nn} p_n p_n + 0.5\theta \sum_{k=1}^{3}\sum_{k=1}^{3} \beta_{kk} a_k a_k$$
$$+ \sum_{n=1}^{3}\sum_{k=1}^{3} \delta_{nk} p_n a_k + \sum_{n=1}^{3}\sum_{s=1}^{4} \mu_{ns} p z'_s + 0.5\theta \sum_{k=1}^{3}\sum_{s=1}^{4} \varphi_{ks} a_k z'_s \qquad \textbf{(4-12)}$$
$$+ 0.5\theta \sum_{s=1}^{4}\sum_{s=1}^{4} \rho_{ss} z'_s z'_s$$

As before the SNQ profit function is in terms of *i*+*j* netput prices (p_n), *h*-1 land uses (a_k), and *m* fixed characteristics plus total land endowment (z'_s). The element θ is the price index used for normalization and its inverse represented by τ (see A.2.a). The elements β,μ,α,φ,δ, and ρ are parameters to be estimated.

If the approximation of a profit function in equation (3-12) holds the theoretical properties described in the previous section, by Hotelling's lemma, the netput equations are:

$$\frac{d\hat{\pi}}{dp_i} = y_i = \alpha_i + \tau \sum_{n=1}^{3} \alpha_{nn} p_n + 0.5\tau^2 \sum_{n=1}^{3}\sum_{n=1}^{3} \alpha_{nn} p_n p_n + \sum_{k=1}^{3} \delta_{nk} a_k$$
$$+ 0.5\tau^{-1} \sum_{k=1}^{3}\sum_{k=1}^{3} \beta_{kk} a_k a_k + \sum_{s=1}^{4} \mu_{ns} z'_s + 0.5\tau^{-1} \sum_{k=1}^{3}\sum_{s=1}^{4} \varphi_{ks} a_k z'_s \qquad \textbf{(4-13)}$$
$$+ 0.5\tau^{-1} \sum_{s=1}^{4}\sum_{s=1}^{4} \rho_{ss} z'_s z'_s$$

If $y_i$is non-negative, it refers to an output per unit of land. If it is non-positive, it refers to an input per unit of land.

The land shadow equations derived from equation (3-12) are:

$$\frac{d\hat{\pi}}{da_k} = \beta_k + \sum_{n=1}^{3} \delta_{nk} p_n + 0.5\theta \sum_{s=1}^{4} \varphi_{ks} z'_s + 0.5\theta \beta_{kk} a_k, \text{for all } k = 1, \dots, h \qquad \textbf{(4-14)}$$

Combining equation (3-14) with the restriction $\sum_{k=1}^{h} a_k = L$ provides a set of reduced-form land equations (Fezzi and Bateman 2011). The system is represented as:

$$a_k = \gamma_k + \sum_{n=1}^{3} \gamma_{nk} p_n + \sum_{s=1}^{4} \omega_{ks} z'_s \text{, for all } k = 1, \dots, h \qquad \textbf{(4-15)}$$

In which the elements γ and ω in equation (3-15) are a combination of the structural parameters in equation (3-12). Parameters obtained with equations (3-13) and (3-15) are used to construct a set of netput and land use elasticities based on equations (3-7)–(3-11), which provide farmers' sensitivity to price changes (Moore and Negri 1992). The chosen model consists of two sets of equations, three structural netput equations, and three reduced-form land use equations.[16] A seemingly unrelated regressions (SUR) is applied to derive the model parameters. The model parameters are then used to calculate netput and land use elasticities. Two sets of equations, as outlined in eqs. (3-13) and (3-15) and adding a stochastic term to each of them, are estimated. The SUR technique assumes a correlation between the error terms across equations (Zellner 1962).

Results should comply with profit maximization duality theory and related properties. By definition, the SNQ profit function is homogenous in netput prices. Symmetry in second derivatives with respect to prices is imposed on the estimations. Convexity in prices is imposed after estimation by using a minimum distance approach, which ensures that the matrix of second derivatives to prices is singular positive semi-definite, which is then always convex in its arguments (Lau 1978, Diewert and Wales 1987, Koebel *et al* 2003). In addition, further restrictions are imposed across land-reduced form equations so that: $\sum_{k=1}^{h} \gamma_{nk} = 0; \sum_{k=1}^{h} \omega_{ks} = 0, \sum_{k=1}^{h} \omega_{kL} = 1$, for all k=1,…, h, and s=1,…,m. The coefficients ω_{kL} represent those related to total land endowment in equation (3-15). For the netput system, convexity in prices is violated, therefore, the preferred specification imposes convexity using a post-estimation approach (Koebel *et al* 2003).

4.3. Agricultural frontier data

The model described above is applied to farmers in the Brazilian agricultural frontier. Information for Mato Grosso is available at a district level from the agricultural census of 2006 and the yearly agricultural and ranching surveys for the years 2004-2006 produced both by the Brazilian Institute of Geography and Statistics (IBGE). Both the census and the survey

[16] An alternative empirical approach is to estimate one structural system of netput and shadow price equations, in which it is assumed that shadow prices are reflected in observed land prices, i.e. each derivative of the profit function with respect to any land use is equal to the average land price faced by farmers Arnade and Kelch (2007). The empirical strategy proposed by Arnade and Kelch (2007) is attractive in the case in which empirical data reflects shadow prices. Then the empirical specification uses one set of structural equations that are enough to recover all parameters from a profit function. In the case of Brazil, heterogeneous land uses are grouped and their average price is reported at the land market level, therefore, it does not allow to homogenize them empirically to shadow prices. We tested the A&K model but the estimation had convergence problems when convexity restrictions were imposed.

data are used in which 139 districts comprising 107,706 farms are considered in the analysis.[17] Table 4-1 provides summary information on the variables used.

Two output levels (crops and cattle), one input level (variable inputs), and four land uses (cropland, pasture, forestland, and other uses) are considered. Farmers commonly decide their production strategies under uncertainty on output prices, and thus, base their decisions on past information. Therefore, expected prices are included as the geometric mean of the previous two years for crops and beef. The crop price is calculated using a value-share-weighted index of crop unit prices at the district level. The cattle price is obtained from micro-region- and state-level information from IBGE and Agrolink, respectively. The variable input price is based on information from 2006, as it is assumed known at the time of making the production decisions. The input price index is based on information from IBGE, the National Food Supply Company (CONAB), and Bowman *et al* (2012). It is weighted using total expenditure shares. The SNQ profit function is normalized using a weighted index of all prices as the numeraire as suggested by Diewert and Wales (1992). All prices are deflated to 2006 and are calculated at a micro-region level. Twenty two micro-regions exist in Mato Grosso state and are used for statistical purposes and defined by IBGE.

[17] Two districts were discarded from the analysis as they did not provide information on crop production, or were outside the Amazon and Cerrado biomes.

Table 4-1. Summary of variables used

Variable (units)	**Source**	**N**	**Mean**	**St. Dev.**	**Min**	**Max**
Farmers *(# p/ district)*	IBGE (http://sidra.ibge.gov.br/)	139	774.863	575.761	63	2,803
pCrops♦ *($R/sack)*	Own calculation; IBGE (http://sidra.ibge.gov.br/)	139	54.070	26.398	26.016	104.382
pCattle *($R/@)*ǂ	Own calculation; IBGE (http://sidra.ibge.gov.br/); Agrolink (https://www.agrolink.com.br/)	139	52.199	3.691	45.520	58.119
pInputs *($R/kg/60)*	Own calculation; IBGE (http://sidra.ibge.gov.br/); CONAB (https://www.conab.gov.br/); Bowman et al.(2012)	139	34.757	25.968	1.942	150.646
qCrops *♦ (sack x 10^{-1})	Own calculation; IBGE (http://sidra.ibge.gov.br/)	139	454.594	1,364.793	0.089	12,107.510
qCattle*** *(heads)*	Own calculation; IBGE (http://sidra.ibge.gov.br/)	139	225.002	186.881	20.540	1,352.804
qInputs * (kg/60 x 10^{-2})	Own calculation; IBGE (http://sidra.ibge.gov.br/)	139	391.067	1,561.582	0.445	15,880.750
Soy aptitude *(share of district)*	Own calculation; Soares-Filho et al (2016), Hansen et al (2013)	139	0.579	0.282	0.000	0.972
Acc. to mkt *(district mean)*	Own calculation; DNIT; Hansen *et al* (2013)	139	4.232	5.384	0.335	37.124
Total farmland* *(ha)*	IBGE (http://sidra.ibge.gov.br/)	139	636.619	753.573	60.946	6,079.921
Env. Fines *(# chg)*	IBAMA (http://ibama.gov.br)	139	-1.604	19.628	-102	43
Cropland* *(ha)*	IBGE (http://sidra.ibge.gov.br/)	139	139.769	399.204	0.479	3,630.112
Pasture* *(ha)*	IBGE (http://sidra.ibge.gov.br/)	139	252.885	303.027	27.586	2,890.375
Forestland* *(ha)*	IBGE (http://sidra.ibge.gov.br/)	139	224.726	252.853	6.639	1,784.854
Other land* *(ha)*	IBGE (http://sidra.ibge.gov.br/)	139	19.238	43.070	0.085	406.034
District area *(ha)*	IBGE (http://sidra.ibge.gov.br/)	139	675,130	630,653	37,134	2,866,577

Note: * These variables are obtained from values available divided by total farmers for each district. Only farmers that own cropland, pasture, or forestland are considered. ♦ A sack is equivalent to approximately 60 kg of product. ǂ One arroba is approximately 15 kg of product.

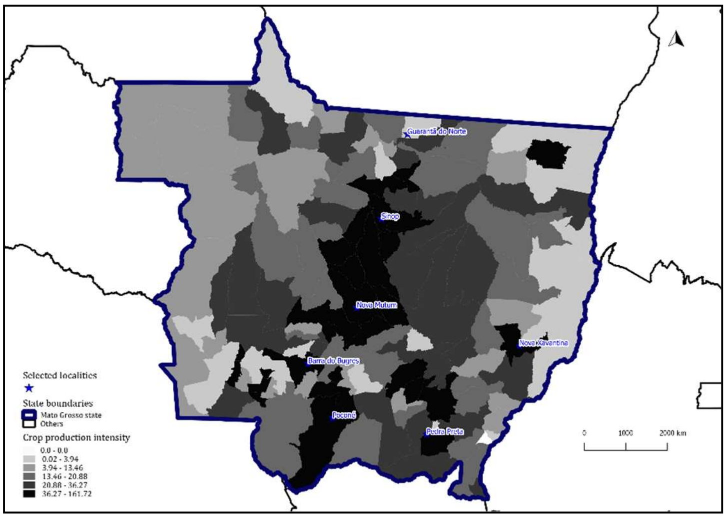
The crop quantity is the ratio between total value of crops and the crop price index. Both perennial and seasonal crops are included such as soy, corn, and cotton–which are major crops in terms of production value. Products excluded are those that do not use cropland for production or information was not available, such as nuts or coffee. Cattle heads' information is directly available in the agricultural census. Fertilizers, herbicides, labor, storage, transportation and machinery, energy, vaccinations, and administrative costs are considered as variable inputs. To obtain inputs demanded, the total expenditure as reported in the census is divided by a price index of inputs described above.

Land use information is obtained from the Brazilian agricultural census. Fifteen land use categories are grouped into four aggregated classifications. Cropland contains areas of permanent and seasonal crops. Pasture refers to natural and planted pastures (in good and bad conditions as reported by the Census). Forestland comprises all areas with natural forest regardless of its protection status. A residual category includes areas for flower plantation, forage plantation, infrastructure, hydrological bodies, degraded land, and land inappropriate for agricultural production.

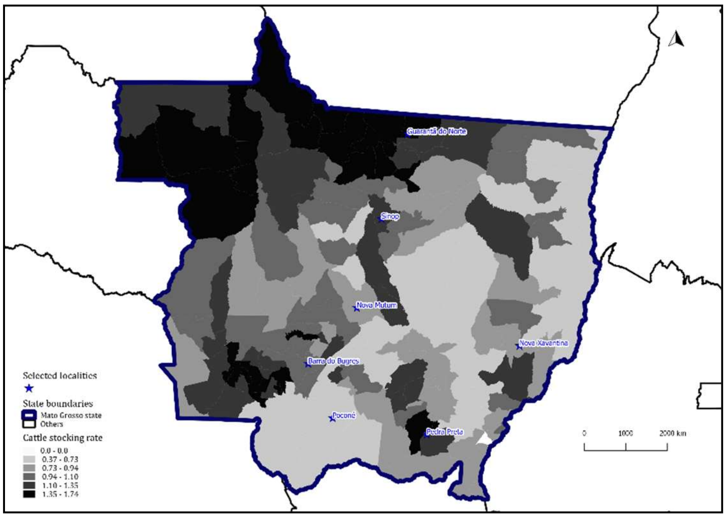
Four fixed factors are included in the analysis. The first one is area potentially profitable for soy production, which is defined as the share of a district's total forestland based on information from Soares-Filho *et al.* (2016) and Hansen *et al.* (2013b). A second factor is an average measure of accessibility to the nearest market, defined here as a municipality capital. It is constructed using information from the National Department of Transport Infrastructure –Brazilian Ministry of Transport (DNIT) and Hansen *et al.* (2013b) which are used to calculate a friction map at a 3km pixel resolution. A least-cost path algorithm is performed for each pixel and the resulting values are averaged for each municipality to obtain the indicator used for the analysis. As mentioned in the introductory section, in the year 2006 an effective conservation policy was gaining presence in Brazil. To control for the effect of the conservation policy, we include the difference in the number of environmental fines issued between year t and $t-1$ in each district. We consider as t the year 2006, therefore we took the fines issued in 2006 minus those issued in 2005 by the Brazilian Institute of Environment and Renewable Natural Resources (IBAMA) to obtain our proxy variable. This variable captures differences in the density of environmental fines across districts as a proxy on the intensity in which conservation policy is implemented. Average farmland endowment in a district is the fourth factor included in the analysis.

Figure 4-1 and Figure 4-2 depict the spatial distribution of production intensities and shares of land in Mato Grosso. As expected, areas of higher cropland share in the center, southeast, and center-west (see map A in Figure 4-2) are also those with higher yields per unit of land (see map A in Figure 4-1). These areas also broadly overlap with areas of high demand for variable inputs (see map C in Figure 4-1). The spatial distribution of cropland, production, and input demand intensities hints to a major use of intensive margins in these regions. High yields are also observed in the southwest, but this might be driven by a general scarcity of land for farm production.

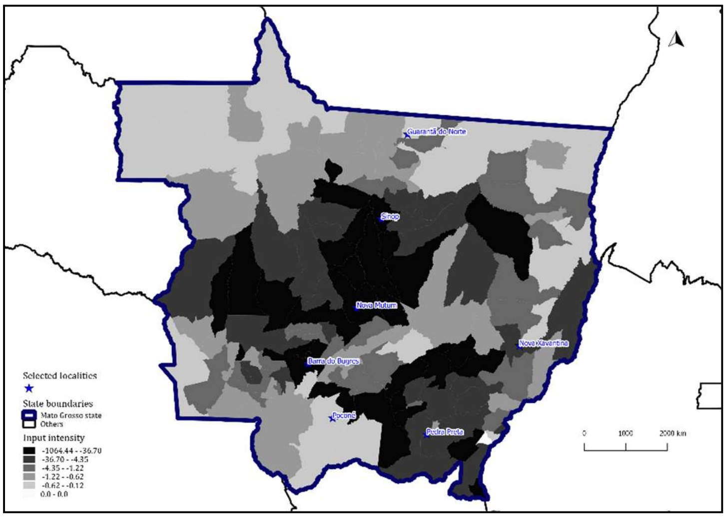
In the northwest, bordering Pará state, high cattle stocking rates are observed (map B in Figure 4-1), where forestland shares are also high (map C Figure 4-2). From there towards the center of the state, competition between pasture and forest for land increases. At the center of the state, in established frontier areas, pasture and cropland compete against each other with cropland often accounting for the lion's share of farmland. This pattern of land use allocation is similar to that of accessibility in which intensive production is located in areas relatively more accessible to municipality capitals, in agreement with a land rent model (see Figure A.2.1 in the A.2 and Miranda et al. (2019)). In the Southwest, the land is less suitable for crop production which might explain the high share of pasture area from total farmland observed.



A)



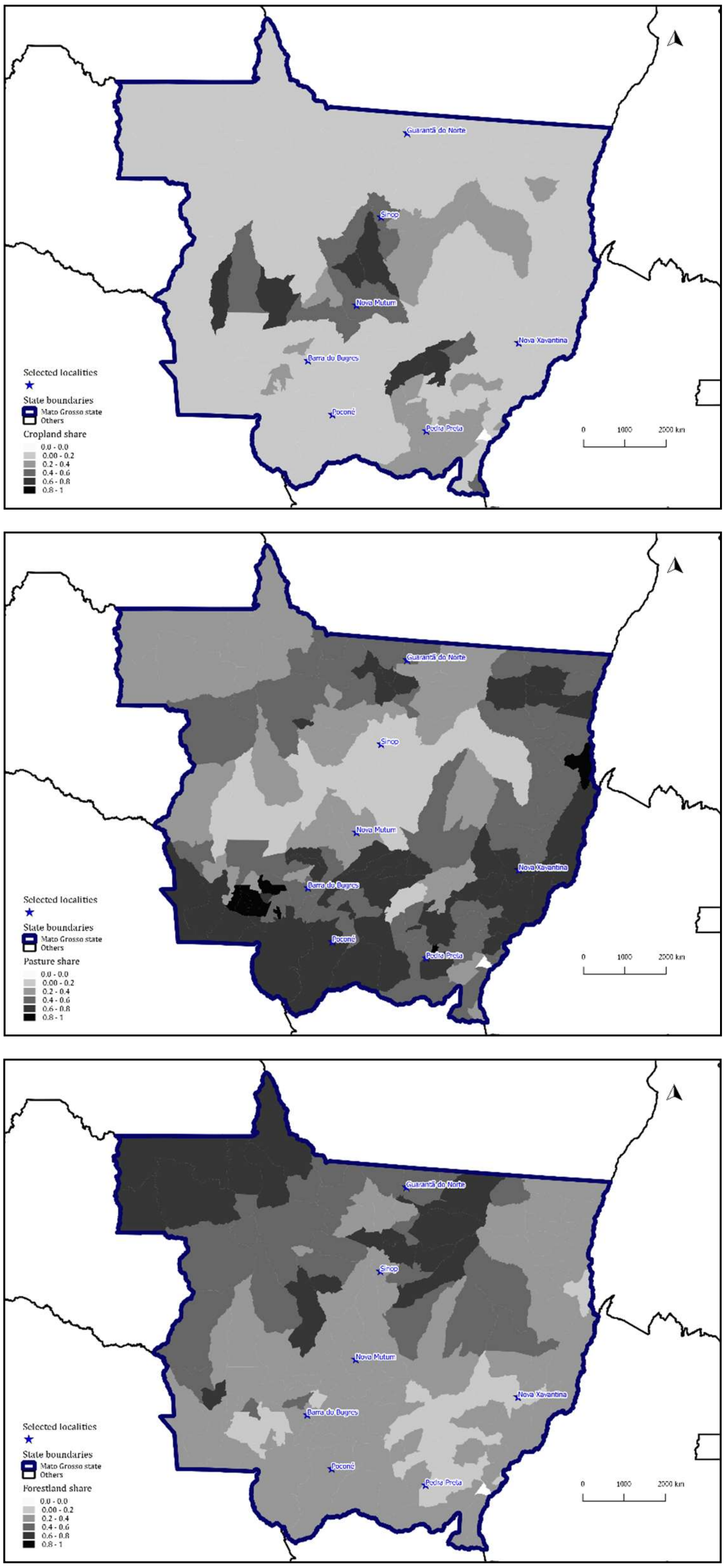
B)



C)

Figure 4-1. Spatial distribution of production and demand intensities in Mato Grosso

Note: The maps depict the average distribution of crop production intensity (A), cattle production intensity (B), and input demand intensity (C) in Mato Grosso's districts. Here intensities are measured as average production or demand per hectare. Selected localities represent important areas of agricultural production and distribution in the state.



A)

B)

C)

Figure 4-2. Spatial distribution of land use shares in Mato Grosso

Note: The maps depict cropland (A), pasture (B), and forestland (C) shares of total farmland in Mato Grosso's districts.

4.4. Farming decisions in the agricultural frontier

4.4.a. Market-driven intensification and land use allocation

The structural operational model presented in the previous section is applied to Brazilian data to answer how and to what extent changes in the economic context shape agricultural production and land use decisions in Mato Grosso. Particularly, the model identifies levels of market-driven intensification in the agricultural system, and translates the impacts of this intensification on expectations of deforestation. A SUR econometric model is applied to the systems of netput and land use equations (see eqs. (3-13) and (3-15) above) using the data described in the previous section. The resulting parameter estimates are shown in Tables A.2.2 and A.2.3 in the annex A.2.c. Based on equations (3-8) – (3-11), relevant parameters are used to calculate uncompensated, intensity, and land use elasticities at mean values. The results are given in Table 4-2 and Table 4-3.

As expected, better economic conditions increase crop and input intensities following the idea of induced innovation in agriculture as laid out by hypotheses H.1 and H.2. In the following, all results are reported for a 10% increase in prices. If crop prices increase, it is expected an increase in crop yields of 1.6% and an increase in input intensity of 0.38%. The effect on cattle intensity reduces just by 0.17%. If input prices increase, there is a reduction of crops for cattle production. An increase in input price reduces yields of production by 1.52%, input intensity by 0.23%, while cattle intensity increases by 0.16%.

Table 4-2. Netput elasticities

	pCrop (TE)	pCattle (TE)	pInputs (TE)	pCrop (YE)	pCattle (YE)	pInputs (YE)	pCrop (LE)	pCattle (LE)	pInputs (LE)
Crops	0.559	-1.822	-0.416	0.160	-0.008	-0.152	0.399	-1.814	-0.264
Cattle	-0.373	1.070	0.190	-0.017	0.001	0.016	-0.356	1.069	0.174
***V*Inputs**	1.291	-2.230	-0.117	0.038	-0.002	-0.023	1.253	-2.228	-0.094

Note: TE refers to total effect or uncompensated elasticities. YE refers to compensated elasticities or yield effect. LE refers to the indirect effect due to land reallocation. The numbers represent a percentage change in supply and demand due to a 1% change of prices. For instance, if the price of crops double, one expects crop yields to increase in 16% by looking at column four in the table.

When land reallocation is allowed, higher uncompensated elasticities are observed. Crop production increases by 5.59%, and reduces by 4.16% for an increase in crop and input prices, respectively. Cattle production increases by 10.7% if the cattle price increases, and by

1.9% for the same increase in input prices. Inputs demand is elastic with respect to output prices. Moreover, it is positively related to crop price and negatively related to cattle price as expected. An increase in crop prices increases inputs' use by 12.9%, and reduces it by 22.3% with a change in cattle price. Inputs' own-price elasticity is negative and inelastic, such that with a 10% increase, inputs demanded are reduced by 1.17%.

Table 4-3. Land use elasticities

	pCrop	**pCattle**	**pInputs**
Cropland	0.633	-1.765	-0.282
Pasture	-0.178	1.641	0.178
Forestland	-0.222	-0.790	-0.045

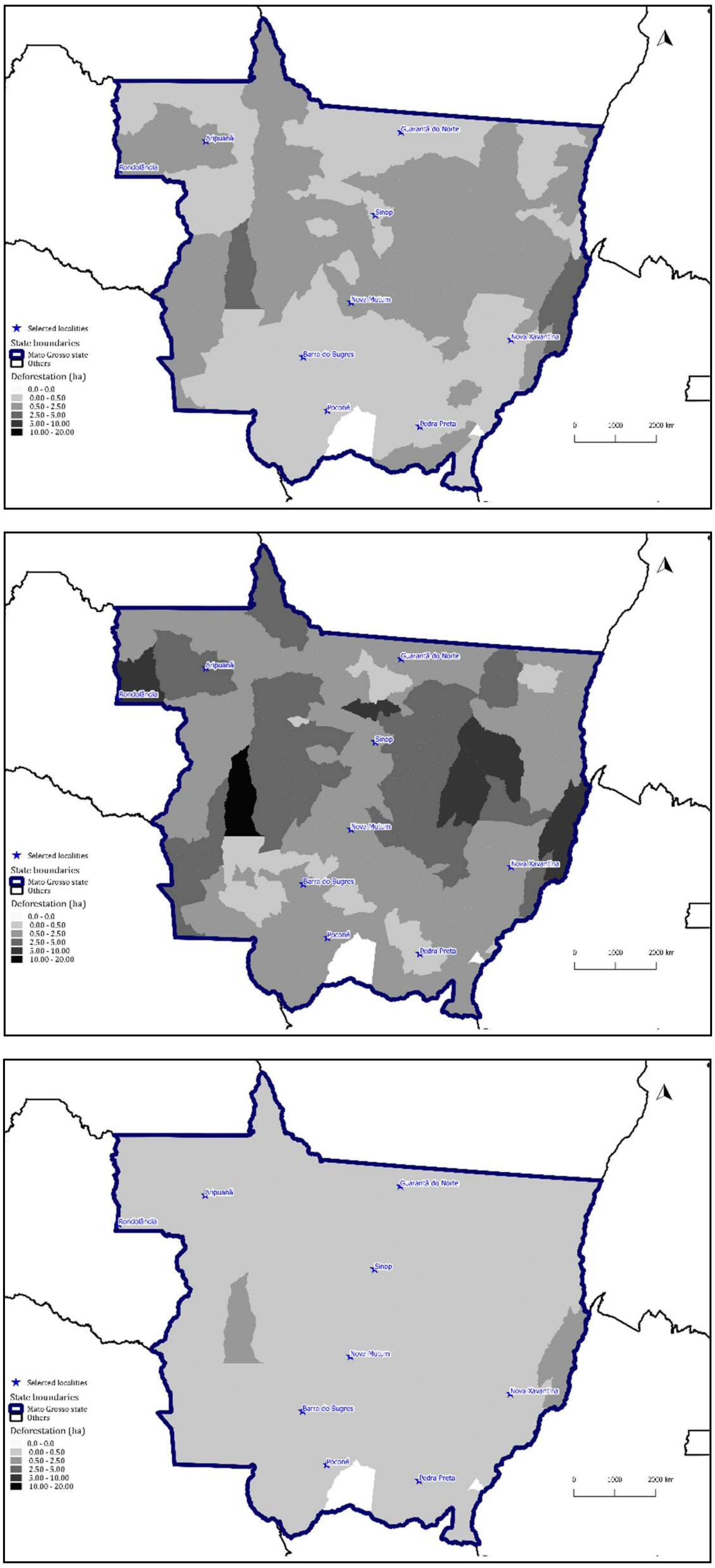
Note: The numbers represent a percentage change in land allocation due to a 1% change of prices. For instance, one percentage change in inputs price increases expected allocation of land into pasture by 0.178%

The results discussed above show the presence of market-driven intensification in Brazilian agricultural frontiers. To understand how this intensification translates into land use decisions, and specifically into pressure on forest areas, land use elasticities are given in Table 3-3. Land use with respect to output and input price changes is rather inelastic, except for cropland and pasture which are elastic with respect to the cattle price. Cropland increases by 6.33% percent for a 10% increase in crop price. A reduction in cropland of 17.65% is expected with a 10% increase in cattle price. A similar change for input price reduces cropland use by 2.82%. Pasture elasticities also show the expected signs. If economic conditions favor crop production indicated, for example, by a 10% increase in prices, farmers reduce land allocated to pastures by 1.78%. Economic conditions in favor of cattle production as expressed by a 10% increase in cattle price will increase land allocated to pasture by 16.41%.

As indicated by these results, the effect of a changing economic environment for crop or cattle production in Mato Grosso contradicts the global perspective of the Borlaug hypothesis, i.e. intensification reduces incentives for forest conversion. Higher crop and cattle prices induce higher forest clearance as shown by the negative elasticities for the use of forestland. The pressure on forests appears to be higher if cattle prices, i.e. a product more extensive in land use, increase. An increase in cattle prices of 10% is expected to reduce forestland allocated within a farm on average by almost 8%. The negative effect on forestland and the positive effect on pasture support the hypothesis that the most important direct threat for forest in the Brazilian agricultural frontier is due to pasture expansion. The same increase in crop prices, reduces the amount of area allocated to forestland by 2.2%. If input prices increase by 10%, forest area is reduced by 0.45%. The small magnitude of the elasticity with respect to input prices hints to a small substitution of inputs for land in the agricultural system.

4.4.b. Expected deforestation

The empirical results on land use allocation point to strong pressure on forest areas in the Brazilian agricultural frontier, however, the effects of a changing economic environment vary across space. Figure 4-3 summarizes the effects of changing agricultural prices by 1% on expected deforestation for an average farmer within each district in the area of study. An increase in crops or cattle prices shows on average a stronger effect on deforestation in regions neighboring areas of established frontiers (darker areas in the center of map A and B). These are also areas in which concentration of land tends to be higher as the number of farmers is smaller than in areas in the northwest or southwest (see fig A.2.3). Northwestern areas are also affected by changing prices, i.e. areas where forestland is relatively more abundant and competition is higher, due to larger numbers of farmers within a district. Forestland is more elastic to changes in cattle than in crop prices (see Table 4-3), therefore areas in map B are darker than areas in map A in the figure. The spatial distribution of expected reduction in forestland due to output and input price changes also offers an explanation on observed deforestation hot spots in the study area (see fig A.2.2). In addition, if input prices increase, this has almost no effect on deforestation (less than half a hectare as presented in map C). Only in some areas in central-west and central-east deforestation in response to input prices is expected to be higher, but only up to 2.5 ha on an average farm. Adding up the average effect for all farmers within a district reinforces these results (fig A.2.4).



A)

B)

C)

Figure 4-3. Expected deforestation from a changing economic context

Note: The maps depict expected deforestation on an average farm within each district due to an increase in 1% in crop (A), cattle (B), and inputs(C) prices in Mato Grosso.

Results should be also interpreted in the context of conservation policy effectively implemented in the Brazilian agricultural frontier in 2006 (Sousa 2016). The change in environmental fines within a district is included as a fixed factor affecting farmers' decisions.

All elasticities obtained from conservation are of small magnitude, nevertheless, all results also point to a conservation policy-driven intensification (see Table A.2.1). Both crop and cattle levels have a positive relation with an increase in fines issued. To reinforce this effect, an increase in fines will deter expansion of cropland and pasture into forest areas. The small effect obtained from these elasticities might be a consequence of the level of compliance with environmental regulations within farmlands, which was lower compared to areas under public administration (Sparovek *et al* 2010).

4.5. Discussion and Conclusions

This chapter provides an empirical assessment of how and to what extent the economic context induces intensification in the Brazilian agricultural frontier. Specifically, it looks at land use for agricultural production at the intensive and extensive margins. Economic theory is used to guide the analysis by exploiting the dual properties of the profit function. The theory is operationalized using an SNQ profit function approach that allows estimating supply, demand, and land allocation responses using information on agricultural transactions.

Relation and sensitivity of decisions due to changes in prices are measured by calculating two sets of elasticities. Supply and demand elasticities are part of one set. The results point to the presence of market-induced intensification in the year of study. Yield and input intensity increase when the economic context is advantageous, i.e. changes happen on the intensive margin. The effect of land reallocation reinforces the effect of higher outputs from land use intensification, i.e. agricultural land use expansion is observed.

The same price effects which induce intensification, also increase expected deforestation. All forest allocation elasticities are negative with respect to a marginal increase of any price. Deforestation is expected to be higher when cattle production is favored. Crop prices have a lower direct effect, yet, are spatially located in areas adjacent to the consolidated agricultural frontier in Mato Grosso. Input prices show a lower effect on deforestation, though it was not equal to zero and was statistically significant (see Table 3-3 above).

Some limitations should be considered when interpreting the results. The data are aggregated at the district level, and thus, the analysis offers an estimate of farmers' sensitivity to the economic context on average. Additionally, information on commodities is highly aggregated, so it does not allow comparing trade-offs in the use of resources among individual crops. An obvious extension to advice policies in specific markets is to disentangle crops' contribution to forest encroachment. This would also allow the identification of additional trade-offs based on observed strategies such as the effect of soy prices on double cropping in the agricultural frontier.

Our results should be viewed under the premise that they reflect an agricultural frontier region in times when conservation governance was *de facto* beginning to be implemented with credible consequences to offenders. Nonetheless, compliance on private farms has not been fully achieved (Sparovek *et al* 2010, Azevedo *et al* 2017). In times of poor conservation, better economic conditions could amplify trade-offs between biomass production and conservation objectives. Moreover, the effect of the conservation policy measures in this analysis is small in magnitude. One reason for this finding is that while the change in governance happened in 2004, its effects in the agricultural system might have been delayed for several years. Furthermore, fines have a highly localized effect on offenders, which can potentially obscure estimations of the effects of conservation on agricultural systems based on aggregate data at the district level (Schmoldt *et al* 1975). Alternative ways to measure the effect of conservation policy are also part of future efforts to understand land use dynamics and its link to conservation policies.

The state of Mato Grosso provides an interesting case study, as it is comprised of different biomes, and exhibits a high level of land use competition in areas with highly productive crop production and extensive cattle ranching. Yet, agricultural expansion and deforestation continue along the arc of deforestation. Due to data availability issues, the results obtained in this study are based on one cross-section and can be affected by unobserved heterogeneity. The analysis is currently being expanded to include other areas of the Brazilian agricultural frontier and newly released census information by the Brazilian federal government.

5. Conservation policy and related land use configurations in the Brazilian agricultural frontier: RD analysis at the farm level

This Chapter is currently a working paper. In collaboration with Jan Börner [a,b]

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Abstract

Sustainable land use is essential to accomplish the targets set in several SDGs, such as "Land for Life" and "Zero Hunger". A common challenge is to balance the conversion of forests to agricultural use with conservation of forests for the provision of environmental products and services. Land intensification is seen as part of the solution, but it can also trigger more expansion if land use policies are not effectively enforced. Agricultural frontier areas with large forest endowments, like in Brazil, are particularly prone to increase expansion when land profitability for agriculture increases, i.e. land is not a scarce resource. Previous studies relate both market forces and enhanced conservation policy implementation to the reduction in the rate of deforestation and land use intensification trends in the Brazilian Legal Amazon region during the past two decades. In this context, we empirically evaluate the impacts and permanence of conservation policy effects on deforestation and intensification at the farm level. We use economic theory to understand a farmer's decision to expand or intensify their agricultural area. In particular, we evaluate biome-targeted policy effects in the Amazon and use farms in neighboring cerrado as a comparison group. We exploit the policy assignment rule to implement a regression discontinuity (RD) design to perform our empirical impact evaluation. In agreement with previous analyzes, we found that, in the short-term, conservation policy has a considerable impact on farm-level deforestation. This success, we argue, is bound to the credibility of implementing these policies. In the long-term, when governance shifts back into pre-conservation status short-term achievements are pushed back. We complement the analysis by looking at the sources of deforestation. We found similar but stronger effects in forest to pasture conversion. Forest to cropland conversion has little or no change due to policies, however, we observe a tendency to increase this type of deforestation after 2012. Land use intensification, measured as the conversion of pasture to cropland, shows little changes during the studied time horizon except in an early period when environmental governance reforms kicked in. In sum, we add empirical evidence that conservation policies reduced forest conversion while promoting land use intensification at the Brazilian agricultural frontier, but these effects are dependent on both farmers' credit constraints and the perceived willingness of the government to enforce environmental regulations.

Keywords: deforestation, induce intensification, conservation policy, RD design

5.1. Introduction

In the quest of finding a balance between development and conservation, Brazil offers an example to analyze how effective environmental governance is in reducing pressure on forest areas while inducing a more intensified agricultural production. Since 2004, the Brazilian government together with the private sector implements different policy instruments to tackle high rates of deforestation, particularly in the tropical Amazon rainforest. Different studies have found that this bundle of policies brought down deforestation rates in the Brazilian Legal Amazon region consistently for eight consecutive years (Soares-Filho *et al* 2010, Arima *et al* 2014, Nepstad *et al* 2014, Assunção *et al* 2015, Cisneros *et al* 2015, Assunção *et al* 2019). While these analyzes are helpful to understand how strong environmental governance reduces deforestation at regional levels and in public forests, it is unclear the effect of these policies on land use decisions taken at the farm level. As farmers' decisions are relevant for biomass production and economic development, and conservation policies are tools to achieve sustainable landscape outcomes, it is important to understand how land-use restrictions affect farmers' decisions in terms of decreasing deforestation, and investment in intensification strategies.

Increasing agricultural productivity is seen as a solution to reduce pressure on forest areas (Spera 2017, Sparovek *et al* 2018). But, intensification may only save forests from agricultural conversion if effective policies are in place, i.e. higher profitability incentivizes land expansion (Merry and Soares-Filho 2017, Soares-Filho and Rajão 2018). In this regard, recent empirical studies have found possible synergies between effective conservation restrictions and policy-driven intensification in Brazil (Oliveira Silva *et al* 2018, Koch *et al* 2019). Moreover, half of the remaining natural vegetation in Brazil is within private areas (Soares-Filho *et al* 2014); thus, it is important to understand the effect of stronger environmental governance on these rural actors.

This chapter empirically evaluates the effect of changes in environmental governance on farmers' land use decisions. One major element of increasing environmental governance is by implementing policies that increase perceived land scarcity such as the conservation policies implemented in Brazil since the last decade. We, therefore, ask the questions: do conservation policy instruments effectively reduce deforestation on private farms at the Brazilian agricultural frontier? If yes, do farmers also intensify the use of already cleared land? We investigate whether, how, and to what extent a shift in environmental governance starting in the mid-2000s with effective implementation of innovative conservation policies affects farmers' land use configurations. We focus the analysis on a time in which a set of policies were implemented that have in common their biome-targeted assignment, i.e. only farmers located in the Amazon biome are subject to these policies. These policies were implemented between 2004 and 2012 when deforestation rates decreased in the Amazon by over 70%, while the production and export of important commodities such as soy and cattle continued to increase (OECD 2015, Zalles *et al* 2019). One important aspect is that the government credibly increased the frequency and severity of punishment for environmental law offenses in the Brazilian Amazon, e.g. government resources were channeled to enforce environmental regulations; more efforts were in place to allow the detection of offenders; additional production restrictions to agriculture were established to those that do not comply with regulations. Since reforms were made to the FC in 2012 (including an amnesty to previous offenders), and after recent signals of the government to favor development over forest protection, agricultural expansion is resurging and the *de facto* enforcement of policies seems questionable.

Our empirical strategy exploits the biome-based characteristic of policy assignment by comparing farms inside the targeted Amazon biome with those in the neighboring Cerrado biome. It is recognized that the agricultural frontier has expanded in these two biomes in the past decades, partially, because new soil management practices allow production in previously unproductive land (Ratter *et al* 1997, Spera 2017).

In agreement with previous studies, we found that conservation policies reduce farm level deforestation. However, we show that these conservation gains dissipate in a strikingly small amount of time. We also detect a policy-induced intensification effect in the first years of implementation, particularly when policies increased costs of conversion instead of reducing the benefits of deforestation. As with deforestation, effects on intensification vanish after some years of implementation. We argue that the expected impacts on deforestation and intensification are due to both credit constraints and the government's perceived willingness to enforce environmental law.

The remainder of this chapter is organized as follows. In the second section, we start with a brief recollection of the environmental governance shifts that took place in the Brazilian agricultural frontier since early this century. It also lays out the theoretical and empirical models behind our analysis. We explain our data sources and processing steps in the third section. A fourth section presents our descriptive and empirical results. The fifth section gives a general conclusion, explains the potential caveats of our analysis, and offers further empirical and methodological research venues.

5.2. Evaluation context and design

5.2.a. Environmental governance shifts in Brazil

The environmental regulation in Brazil started with the enactment of the FC in 1934. In its first decades of existence, the FC had little impact on environmental protection in practice. It was only after a series of presidential decrees starting in the 1990s that the FC was transformed and by 2001 it required farmers to maintain a high percentage of their farmland as vegetation (Soares-Filho *et al* 2014). These percentages varied depending on a farm's location. Farms located within the Amazon biome were required to maintain 80% of their land, while those within the Cerrado biome were only restricted to 35%. For the rest of the farms, only 20% was required.

In the late 1990s and early 2000s, deforestation rates peaked in the Amazon region. As a result, the Brazilian government prepared an action plan to stop this trend in 2003. In 2004, the first phase of the Action Plan for Prevention and Control of Deforestation in the Legal Amazon (PPCDAm) was officially launched. PPCDAm's strategic lines of action deal with: land tenure problems and territorial planning; monitoring, licensing and enforcement; sustainable management of forest and improved use of already-cleared lands; and sustainable infrastructure in the transportation and energy sectors (May *et al* 2011). It is based on three major pillars: a) tenure regularization; b) monitoring and control; and c) incentives for sustainable production (Gebara and Thuault 2013). PPCDAm's first phase went from 2004 till 2008 and included different policy instruments.

Between 2003 till 2010, the federal government adopted the conservation agenda as a priority, therefore human and capital resources were transferred for monitoring efforts. In PPCDAM's first year, Brazil's environmental enforcement agency (IBAMA) was restructured and it began the use of satellite images to identify forest conversion, and therefore aid on-the-ground enforcement efforts. The National Institute for Space Research (INPE) was appointed to create a satellite-based monitoring system. The real-time deforestation detection service (DETER) based on MODIS sensor information at 250 m resolution was introduced as an innovative tool to aid IBAMA's monitoring. Starting in 2004, alerts went off when areas of at least 25 hectares of deforestation were detected nearly in a daily basis.[18] This first shift in environmental governance is related to reduced rates of deforestation in the Amazon biome which went from 27.8 to 19 thousand km^2 in one year. Additionally, the federal government increased the extension of protected areas, as well as those of conservation units during PPCDAm's first phase (Arima *et al* 2014). Also, more fines were issued, which even if little financial resources were recovered, it increases transactional and reputational costs (Börner *et al* 2014, Hummel 2016). We emphasize that this period was particularly important for the conservation agenda as the government sent clear signals of a change in the environmental governance regime.

In 2006 national and international attention focused on how soy production was linked to forest clearance in the Brazilian Amazon. After some protests and NGOs' pressure on retailers in the value chain of production, big players in the industry considered measures to reduce soy sourced from deforested areas. The Brazilian Association of Vegetable Oil Industries (ABIOVE) and the Association of Cereal Exporters in Brazil (ANEC), which purchase 90% of the soy produced in the Amazon together, signed a first Soy Moratorium towards zero-deforestation (Brannstrom *et al* 2012, Gibbs *et al* 2015). Mainly, it stopped any purchase of soy produced in deforested areas after 2006 in the Amazon biome. Those who failed to comply would lose the option of selling their products, but it also didn't give any additional benefit to those that comply with the moratorium (Gibbs *et al* 2015). A similar agreement was signed for the cattle sector in 2009. It has been related to having a reduction on deforestation in the state of Para, however, problems with cattle laundry and increasing demands from regions with little or no restrictions on products sourced from deforested areas cast doubts on the true impact of this agreement (Gibbs *et al* 2016, Merry and Soares-Filho 2017).

In 2008, steps to curve deforestation went further as deforestation jumped by 12% from the previous year (Arima *et al* 2014). To reinforce field-based enforcement, two major policies were set between 2007 and 2008: halt farmers' agricultural credit conditional on environmental performance, and establishment of a list of municipalities with high |deforestation records (Cisneros *et al* 2015). First, the Brazilian Monetary Council in 2008 limited credit to farmers that do not comply with the FC. It stopped access of farmers in 36 counties with the highest deforestation rates to federal agricultural credit and markets (Nepstad *et al* 2014). Moreover, it restricted future credits conditional on demonstrating

[18]http://www.obt.inpe.br/OBT/assuntos/programas/amazonia/deter

environmental compliance (Assunção *et al* 2019). Second, the list, also known as the blacklist, contains those municipalities with important historical deforestation rates. Those jurisdictions inside the blacklist are then confronted with tighter rules on the authorization for forest clearance. Specific administrative targets need to be reached to qualify for removal from this list. It is important to note that although both policies are applied based on the same environmental performance criteria i.e., compliance with the Forest Code, credit restrictions apply in the whole Amazon biome, while blacklisting is targeted in specific municipalities in the Brazilian Legal Amazon region (Cisneros *et al* 2015).

An additional tool implemented to monitor farmers' environmental performance is the environmental rural registry (CAR). CAR provides information on farm boundaries across the country. This registry has been implemented in Para and Mato Grosso states since 2008 and 2009, respectively. It was voluntary at first but in 2012, with reforms to the Forest Code, it has been mandatory for all farmers in rural Brazil (Azevedo *et al* 2017, Jung *et al* 2017).

The government has renewed the PPCDAM with a second version (PPCDAm-II) towards a reduction of deforestation of 80% by 2020 (relative to 1996-2005 baseline). A third phase (2012-2015) focuses on areas with less than 25 ha to improve actions of planning and territorial development and agrarian sustainable productive activities (Gebara and Thuault 2013, p 11). In this regard, the satellite-based monitoring system (DETER) can detect areas of at least 1 ha since 2015. In its latest phase (2016-2020), PPCDAm seeks to implement additional standards and economic instruments for the prevention and combat of deforestation.[19] Also, within the Legal Amazon Region, all states are obliged to develop their action plans in line with PPCDAm actions. The Soy Moratorium has been renewed every 2 years since its first establishment. In 2014's renewal a major change was the modification on the deforestation date considered for the moratorium, from July 2006 to July 2008 (Gibbs *et al* 2015). It is permanently established since 2016 (Fearnside 2017, Carvalho *et al* 2019).

After the implementation of the PPCDAm, deforestation rates in the Amazon biome reached its lower level in 2012. Rates of deforestation have increased since then. This increment has |not reached rates experienced a decade before, yet, environmental governance has reduced gradually in favor of agricultural land expansion for rural development (Fearnside 2017, Carvalho *et al* 2019, Ferrante and Fearnside 2019). As an example, less public funding and support have been given to IBAMA in recent years affecting monitoring and enforcement

[19] http://redd.mma.gov.br/en/legal-and-public-policy-framework/ppcdam

operations (Carvalho *et al* 2019). Successful lobbying to degazette protected areas is another example of how environmental governance shifts in Brazil (Fearnside 2017). We argue later on that these signs from the government reduce farmers' compliance with environmental law, moreover, we keep in mind these shifts in environmental governance when interpreting our results.

Figure 4-1 below depicts the major environmental governance shifts described above in the vertical-dotdashed-black lines, namely, the enactment of the PPCDAm (2004), the establishment of the Soy Moratorium (2006), the imposition of credit constraints (2008), and the promulgation of a new FC (2012). It also depicts the amount of deforestation reported by official numbers in the Amazon (green line) and Cerrado (yellow line) biomes. Finally, Figure 4-1 is contextualized with respect to the different presidential terms in the period of analysis (see the top of the figure) as transition years can be a source of uncertainty in expected environmental governance (Fearnside 2017). Deforestation had an increasing trend in the Amazon biome at the beginning of the century under president Cardoso. The breakthrough in conservation happen during the time of president Lula, as the PPCDAm, the SoyM, and the credit constraints were all issued under his mandate. Environmental governance started to decrease with president Rousseff, and after two years of strong agricultural lobbying, a new FC was proclaimed in 2012. Under president Temer, support for the agricultural sector over the environmental agenda increased.

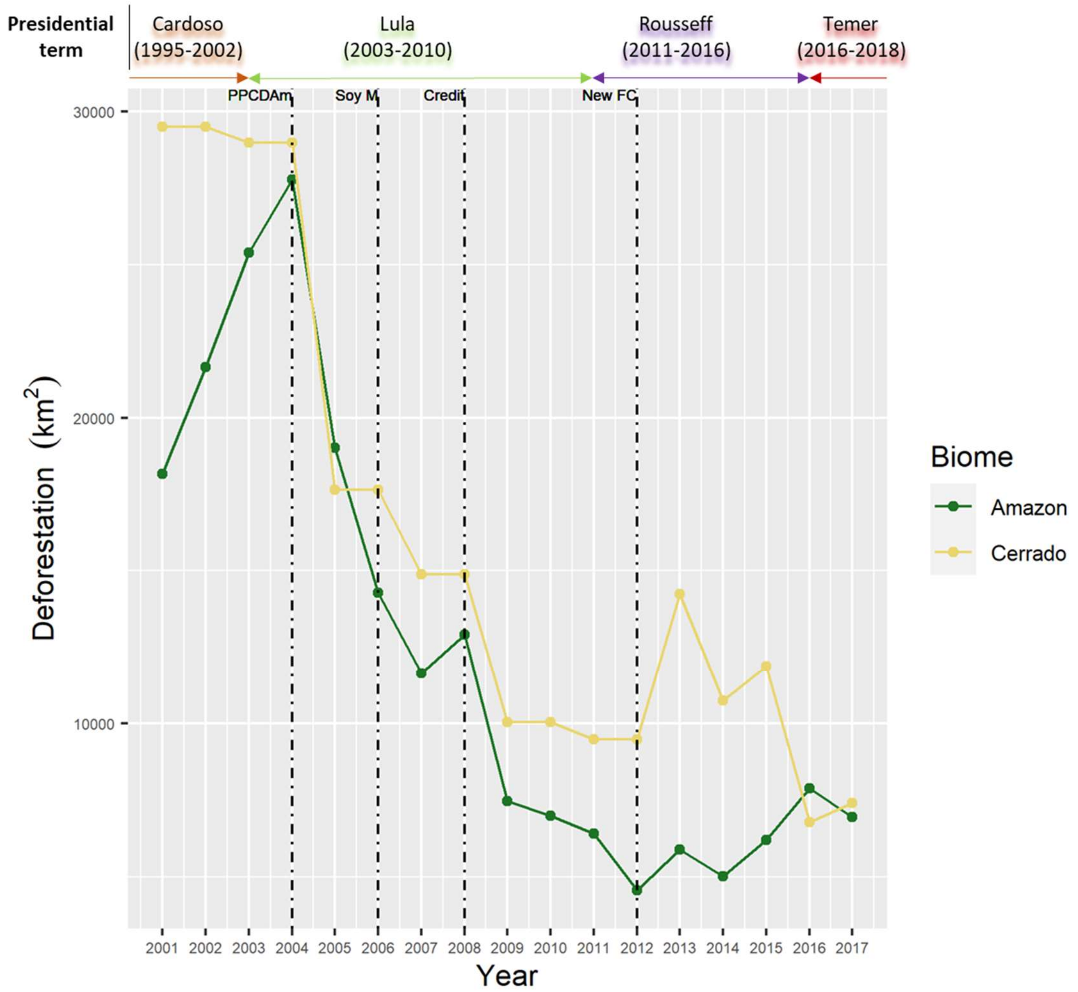


Figure 5-1. Brazil's deforestation and environmental governance in the 21st Century
Note: The lines represent the amount of deforestation reported by the National Space Agency of Brazil (INPE) for the Amazon (green), and the Cerrado (yellow) biomes (http://www.inpe.br/). The vertical-dotdashed-black lines depict the years in which major conservation regulations were put in place: the PPCDAm, the Soy Moratorium, credit restrictions, and the new FC. The different federal executive administrations of the period of analysis are shown at the top of the figure, that is, the period of Fernando Henrique Cardoso (1996-2002), Luiz Inácio Lula da Silva (2003-2010), Dilma Rousseff (2011-2016), and Michel Temer (2016-2018).

Conservation of natural forest is the main objective of the policies presented in this section and depicted in Figure 4-1. From all these different actions the Soy Moratorium and the credit restrictions stand out for their assignment rule, i.e. only farmers located in the Amazon biome are subject to these policies. These two instruments have been related to a great part of deforestation reductions in the Amazon biome (Assunção *et al* 2013, Arima *et al* 2014, Nepstad *et al* 2014, Gibbs *et al* 2015). We exploit this assignment characteristic to set up our empirical evaluation of the effects of conservation on deforestation and land use intensification decisions at the farm level. Moreover, we test the stability of this effect through time. We use land cover information to track annual deforestation, and land use allocation within farmland areas between 2001-2017. This period represents pre- and post-implementation moments in time. We then use this information to implement a regression discontinuity (RD) design to estimate policy impacts on deforestation and intensification decisions.

In the remainder of this section, we explain further the theoretical framework and the empirical strategy adopted.

5.2.b. Policy mechanisms affecting deforestation and intensification

We use an economic model to understand the mechanisms behind conservation policy affecting deforestation to frame our empirical specification. We use the model proposed by Koch *et al* (2019) to analyze a farmer's maximization problem in the short-term in the Brazilian agricultural frontier. She can produce a quantity of output (Y) using her available technology represented by a function f. In our area of study, farmers' production is mostly focused on soy and cattle products. We assume that f is concave in its factors, twice differentiable, and it has positive cross-derivatives to fulfill properties posed by production economic theory (Chambers 1988).

Land available for agricultural use is composed of cropland, pasture, and deforested land, $\{l_c, l_p, l_d\}$. Empirical evidence associates deforestation with pasture and cropland uses in the Brazilian agricultural frontier. Farmers convert forest as an input of production, or as a land reserve which they profit from future higher land prices as the agricultural frontier advances (Hecht 1985, Caldas *et al* 2007, Fearnside 2008). They do so at a physical conversion cost of c. In addition to benefits accrued from agricultural production, this model considers benefits obtained from creating land reserves net to the punishment accrue by offenders due to effective policy implementation ($v = b - \tau$).

One important restriction to simplify the model is that additional land for agricultural production is only sourced through deforested land (l_d), i.e. land already converted in a previous period is fixed ($A_t = l_{ct-1} + l_{pt-1} + l_{dt}$). To focus the analysis on deforestation we maintain the sum of cropland and pasture as fixed at the starting of each year t so that $L = l_{ct-1} + l_{pt-1}$. The maximization decision is constrained by a total budget, B, which includes available government credit, and can be invested in two main things. The farmer's strategy is either to increase land use intensification through investments in capital, K, which includes costs of converting pasture to cropland (e.g. investment on lime-powder), or to deforest (Cohn *et al* 2016, Bragança and Cohn 2019). The interest rate r represents rental costs of capital.

The problem to maximize profits at an output price, p, is given by:

$$L = \max_{l_d, K}\{pf(K, L + l_d) + vl_d - rK; B = K + cl_d)\} \quad \textbf{(5-1)}$$

This problem has the first-ordered conditions (F.O.C.):

$$\begin{aligned} pf^L(K, L + l_d) + v + \lambda c = 0 \\ pf^K(K, L + l_d) - r + \lambda = 0 \end{aligned} \quad \textbf{(5-2)}$$

|Combining the F.O.C with the budget constraint results in an auxiliary function (H) defining an optimal level of l_d (Koch *et al* 2019).[20] Koch *et al* (2019) use this function and apply the implicit function theorem to derive the conservation policy mechanism affecting farmers' deforestation and intensification decisions. They prove that $\frac{dl_d}{dv} > 0$, $\frac{dK}{dv} < 0$, $\frac{dl_d}{dc} < 0$, and $\frac{dK}{dc} > 0$. These important insights of the model relates to the effect of (*de facto*) policy implementation on deforestation decisions. Policy instruments reducing net benefits v (e.g. satellite-based monitoring or a supply-chain intervention) or increasing costs c (e.g. credit restrictions or embargos) result in a reduction of deforestation and more investment in intensification for a constrained specification (Koch *et al* 2019). They further prove that when farmers do not have a constrained budget, the derivative of K with respect to v is positive, i.e. a reduction in v reduces deforestation and investments in capital.

The Amazon biome, host of pristine tropical forest, was the only biome subject to specific policies in times in which environmental governance gain strength at a federal level. From the theoretical model explained above, we derive two hypotheses on the effect of these conservation policies on Amazon farmers **H.1)** If farmers are budget-constrained, an effective conservation policy shock reduces deforestation on private farms and induces intensification. **H.2)** If farmers are not budget constrained (e.g. alternative sources to credit are available), we expect a reduction in both deforestation and intensification.[21]

Hypotheses H.1 and H.2 are dependent on the effectiveness of policy implementation. Credibility is an important element to enable effectiveness, particularly in areas with high

[20] The auxiliary function derived by the Koch *et al* (2019) is $H = pf^{l_d}(B - cl_d, L + l_d) + v + (r - pf^K(B - cl_d, L + l_d))c$ where the price (p) is normalised to 1.

[21] One would expect some increase on intensification as resources are reallocated from land to capital investments when land becomes more expensive relative to capital in production. This leads to more product produce per unit of land. In our analysis we could only measure intensification due to land use reallocation from an extensive use (pasture) to a more intensive land use (cropland); therefore, our hypotheses focus on this type of intensification.

speculative behavior, such as the Brazilian agricultural frontier. Two additional hypotheses follow. **H.3)** If farmers expect lower benefits from land expansion due to clear government signs to enforce conservation regulations, we expect deforestation reductions as proposed in our hypotheses H.1 and H.2 to materialize. In their first phase, new regulations have greater potential to introduce behavioral changes, as individuals are confronted with a new context, i.e. we expect a low likelihood for subversions. This potential is maintained when government actions and discourse support the environmental agenda, and it is accompanied |by a systematic reduction in deforestation. **H.4)** If farmers expect no punishment due to scarce or null government action to enforce conservation regulations, we expect hypotheses H.1 and H.2 to not materialize.

The theoretical model is operationalized by exploiting the biome-targeted assignment quality of conservation policies. All else equal, farms in the Amazon biome would be subject to lower deforestation incentives than those in Cerrado areas where biome-targeted policies do not apply. We compare deforestation decisions between farmers located in one or the other side of the border for pre- and post-implementation years. We explain our empirical approach below.

5.2.c. RD applied to the agricultural frontier

We use private farms' location to operationalize the economic model presented above. As we are interested in biome-targeted policies that only apply to farmers in the Amazon, we used information from farmers in neighboring Cerrado as controls to investigate the effect of a policy instrument, τ, on deforestation. We assume, therefore, that farmers in one or the other side of the biome border would have similar characteristics except for the biome-targeted policy effect, a similarity that increases the more proximate they stand to the border. If this is the case, and one knows the exact location of each farmer, we can invoke the properties of a sharp RD design and implement it in a geographic setting (Keele *et al* 2017).

We use private farm's locations to construct a running variable in which at zero distance to the border represents the threshold between being subject or not to a biome-targeted policy. The running variable determining treatment is used to set up an RD design. This approach has as main features that all units i are a) assigned treatment based on an observed running variable X_i, and b) the conditional probability of treatment effect changes discontinuously at a known cutoff value c dependent on the running variable (Lee and Lemieux 2010, Cattaneo and Escanciano 2017).

In a sharp RD design, the probability of being subject to policy changes from zero to one at the cutoff (Sekhon and Titiunik 2017). In our case, locations on the Amazonia side are assigned as treated and those on the other side of the border as our controls. The geo-referenced location of farmers is known, which allows the implementation of a generalization of a sharp RD design (Keele *et al* 2017). This framework considers two potential outcomes $\{Y_{i1}, Y_{i0}\}$ for each farmer. In our study Y_{iT} represents a farmer's deforestation measured as the |annual amount of forest converted into other land uses. We use annual area of pasture converted into cropland to identify intensification strategies related to conservation policy. As complementary indicators, we use the annual amount of forest converted into cropland, pasture, or a residual category in a previous year. We consider treated status with $T = 1$ and zero otherwise. The observed outcome for n observed farmers is given by:

$$y_i = TY_{i1} + (1 - T)Y_{i0}, \text{for all } i = 1, \dots, n \qquad \textbf{(5-3)}$$

so that,

$$y_i = \begin{cases} Y_{i1} \; if \; T = 1 \\ Y_{i0} \; if \; T = 0 \end{cases} \qquad \textbf{(5-4)}$$

We use equations (4-3) and (4-4) to estimate y_i as a regression function dependent on distance to the border between biomes, $\varphi_T(x) = E[Y_{iT}|X_i = x]$. We only observe treated values for $x \geq c$ and untreated for $x < c$, therefore we cannot compare both statuses for any farmer at any value of x. A sharp RD design exploits the continuity property of the running variable. We assume therefore that it exists a neighborhood or bandwidth of farmers at δ distance of the border in which their potential outcomes are not abruptly different from those at c; units in the neighborhood $[c, c + \delta]$ are assumed to be very similar except for their treatment status (Cattaneo *et al* 2019a). These assumptions allow us to measure the difference in outcomes at the Amazonas-Cerrado border, B, that is, the policy effect is estimated at the cutoff:

$$\begin{aligned} \tau(c) = \varphi_1(c) - \varphi_0(c) = E[Y_{i1} - Y_{i0}|X = c] = E[y_i|X = c] \\ = \lim_{x \downarrow c} E[y_i|X = x] - \lim_{x \uparrow c} E[y_i|X = x] \end{aligned} \qquad \textbf{(5-5)}$$

The last equality in (4-5) is given by the continuity properties of $\varphi_T(x)$ on c (Hahn *et al* 2001, Cattaneo *et al* 2019a). The reduced-form treatment effect is unobservable but it is approximated as a function of two limits that depend on factors observed in the data (Cattaneo *et al* 2019b). We assume that the treatment effect in (4-5) is equal at any point along the border, $b_i \in B$, to simplify the analysis. This assumption simplify the analysis, as heterogeneous effects are not considered, i.e. we do not estimate any set of effects τ_i at any point $b_i \in B$ along the border (Keele and Titiunik 2014).

One common situation encountered when using spatially explicit data in an RD is that of compound treatment effects, in which two or more policies are implemented simultaneously for some moments in time with the same assignment rule (Keele *et al* 2017). This compound effect does not allow us to measure the effect of a single policy. Within our time framework, between 2006-2008 all policy effects can be attributed to the soy moratorium as it was the only biome-targeted policy at the time. Since the establishment in 2008 of conditional credits based on environmental regulations, we would have a compound effect. Even, we expect some additional compound effect from the implementation of the rural land registration, CAR, in the states of Mato Grosso and Para. In our analysis, we are interested in the combined effect of these policies (i.e. signs of a change in environmental governance) on deforestation and intensification outcomes over several years, more than evaluating the effect of a single policy.[22] Also, these policies have been associated with strong effects on the observed reductions in forest conservation in the Amazon biome. If we instead were interested in the effect of a single policy, additional steps to single out the effect would be needed, e.g. outcome transformations similar to standard difference-in-difference (Eggers *et al* 2018).

The continuity-based framework presented above relies on the assumption of continuity of the potential outcomes as functions of the score. This assumption allows us to compare units marginally above or below the cutoff to estimate the average treatment effect at the cutoff (Cattaneo *et al* 2019b). We use the R-software package *rdrobust* to empirically approximate equation (4-5) (Calonico *et al* 2015). We choose our bandwidth in different steps to make comparisons across years. First, we explore a data-driven generated algorithm to calculate the bandwidth that minimizes the mean squared error (MSE) of the estimation for all observations within 100 km distance to the Amazon-Cerrado biome border in each year (Imbens and Kalyanaraman 2012, Calonico *et al* 2014). This method optimizes the bandwidth value by considering the trade-off between correcting for bias and the variance it brings but leads to heterogeneous bandwidth choices across years which are also too large for conventional confidence intervals to be valid (Thoemmes *et al* 2017, Cattaneo *et al* 2019a).

[22] To our knowledge, only conservation policies are a source of such a compound effect in the area and time frame of the analysis.

The resulting annual average MSE bandwidth is 20 km. We conducted falsification tests on different predetermined covariates to compare the data-driven optimal bandwidth. These tests use different distance to the border to support our design and choice of bandwidth (see A.3.c). In all tests, farmers in the Amazon and the Cerrado are statistically different at 20 km (see A.3.c). We found that at 5 km distance to the border farmers in both sides present stronger similarities based on the covariates included (i.e. small coefficient estimations with high p-Values), therefore it was our final bandwidth of choice. This is an undersmoothing procedure which justifies the use of conventional inference calculations (Keele and Titiunik 2014). This bandwidth then drives the analysis in a polynomial function of order p, which in our empirical implementation equals 1. We weigh observations based on their location vis-à-vis the biome border using a triangular kernel density function. As mentioned before, we conduct falsification tests on four different predetermined covariates: total land endowment, forest endowment in $t-1$, geographic accessibility to markets, and density of fines in the previous two years to t. Due to the high correlation between the land and forest endowment variables (see Figure A.3.2 in the appendix), we do not use land endowment as a covariate in the final RD empirical estimation.

An advantage of our design is that it uses a natural boundary to set up the evaluation, instead of an administrative boundary to use spatial information and apply it in an RD approach (Wuepper *et al* 2020). The assumption that individuals are equal and the only observed differences are due to policy is weakened with the use of higher administrative boundaries, e.g. individuals in different countries might be subject to considerably different institutional, political and economic contexts despite their geographical closeness. Also, our approach compares policy impacts on deforestation and intensification outcomes over time in a unified empirical framework, and we interpret these results dependent on the different shifts in environmental governance experienced in Brazil.

5.3. Data sources and processing

Our analysis uses two main sources of information. First, the national system of environmental rural registry (SICAR), provided by The Brazilian Agricultural Research Corporation (Embrapa), offers a unique electronic dataset to relate land use changes with decision-making at the farm level. [23] This information is rich in its spatial detail but it has farm boundary inconsistencies such as overlaps and duplicates. The Brazilian NGO,

[23]http://www.car.gov.br/publico/imoveis/index.

IMAFLORA, in partnership with the Royal Institute of Technology in Stockholm and the Federal Institute for Education, Science, and Technology of Sao Paulo corrected these inconsistencies and therefore provide a spatially consistent dataset.[24] We exploited the spatial richness of this dataset to explore land use configurations over time. An important assumption is that the CAR boundaries used, do represent the same farms along the period of study. This is not unrealistic as the number and size of farms have been rather stable in the past decades (Ferreira Filho and Vian 2016).

The second major source of information comes from MapBiomas. MapBiomas is an initiative supported by universities, NGOs, and private companies that provides with annual temporal series of land cover and land use maps of Brazil. MapBiomas tracks land use over time by employing satellite-images (Landsat 5, 7, and 8) for all Brazilian biomes (MapBiomas 2019). They use satellite information from 1998 till 2018 at 30 x 30 m resolution pixels. MapBiomas was established in 2015 and since then it has released 4 versions of information, each update expanding the time-frame with higher accuracy on land use detection. We use its fourth version to create statistics of land use within farm boundaries. We grouped MapBiomas' land cover classifications into four major land use categories: forestland, cropland, pasture, and a residual category. First, we consider forestland as natural forest formations including savannahs. Cropland includes annual, perennial, and semi-perennial crops, and forest plantations. Pasture includes grassland formations, pastures, and pasture-agricultural mosaics. The residual category, others, includes wetlands, other non-forest natural formations, non-vegetated areas (i.e., dunes, urban infrastructure, mining), and water bodies.

We calculate the outcomes and running variables for each farm boundary which centroids are located within 100 km Euclidean distance to the biome border in the states of Rondonia, Mato Grosso, Pará, Tocantins, and Maranhão. Our period of study is from 2001 till 2017 as these are the years for which land use changes can be captured. Also, it represents a period before and after conservation policies were implemented. Most reductions between 2004 and 2007 have been related to contraction of agricultural profits due to a reduction in global soy prices and foot-and-mouth disease outbreaks in the Brazilian cattle sector (Arima *et al* 2011, Richards *et al* 2012, Bowman 2016) and the second part of this period by conservation policies (Nepstad *et al* 2014, Fearnside 2017, Soares-Filho and Rajão 2018). If our empirical strategy remains valid, then all effects from land suitability, markets, and non-biome targeted

[24] Available at http://maps.csr.ufmg.br/

conservation policies would be accounted for, remaining only the effect of biome-targeted policies in our analysis. Earlier studies suggested biome-targeted policies (i.e.the soy moratorium and conditional credits) to have had strong effects on deforestation in the Amazon biome, while deforestation remains untamed in the Cerrado biome (Gibbs *et al* 2016, Assunção *et al* 2019, Soterroni *et al* 2019, Zalles *et al* 2019).

We need to identify an outcome variable that captures changes in deforestation to evaluate whether, how, and to what extent conservation policy reduces this behavior at the farm level. We measure deforestation as *gross* deforestation (i.e. conversion of primary forest) instead of *net* deforestation (i.e. accounting also for forest regeneration and/or tree plantations). Gross deforestation has been recommended as a more accurate target for pristine forest conservation, as net deforestation captures tree plantation and regeneration of previously cleared or degraded areas but masks deforestation of natural forest in other areas (Brown and Zarin 2013, Fearnside 2017). We calculate gross deforestation as the sum of forest area for a given farm in year $t-1$ that is classified as either cropland, pasture, or other uses in year t.

We break down deforestation into its sources to construct complementary outcome variables, that is, we decompose the effect of conservation policies into conversion for cropland or pasture. From our four land use categories, we combine pasture and other land uses to create our additional outcome variables. In exploratory analyzes, we found that most of our residual category is composed of areas labeled as “non-forest natural formations” which are often classified as pasture (and in few cases as cropland) after some years. That is, we identified it as a transition category area between forest and agricultural land. In the remainder of this section when we refer to the forest to pasture conversion, it includes forest conversion to our residual category.

Our intensification outcome is measured as pasture in year $t-1$ that was converted into cropland in year t. We choose pasture to cropland as it is one of the available strategies to intensify agricultural systems (Cohn *et al* 2016, Bragança and Cohn 2019). Intensification is seen as one alternative to decouple agricultural expansion from deforestation in Brazil (Macedo *et al* 2012, Gollnow and Lakes 2014, Sparovek *et al* 2018); however, additional studies warn that beneficial outcomes related to intensification are only realized with an effective conservation policy in place (Merry and Soares-Filho 2017, Soares-Filho and Rajão 2018). In this regard, our setup allows us to test the effect of conservation policies on intensification strategies.

The running variable in the empirical analysis is dependent on farmers' distance to the biome border between the Amazon and the Cerrado. Information on biome boundaries is obtained from the Brazilian Institute of Geography and Statistics (IBGE). We use this information to extract boundary lines intersecting both biomes as the border. We then calculate the euclidean distance from each CAR polygon's centroid to the nearest point at the border to identify the running variable in the analysis. The running variable takes a value of 0 at the border, negative values for those farmers located in the Cerrado, and positive values for those located in the Amazon.

The processing steps above result in a data set of 1,864,469 observations (≈109,000 farmers/year). Variations across years come from some observations engaging in deforestation in some years and not in others due to complete loss of forest cover. Our sample covers a farmland area of 46.16 million hectares.[25] From all the farms in our dataset, those located in the Amazon represent 48% of our observations and hold 52% of the total area represented in our sample (see A.3.a). It is important to clarify that our sample does not cover all existing farms, as it only covers those registered in the CAR system; therefore, deforestation is not captured in both non-registered farms and public land in this study.

As mentioned above, we assume that biome-targeted conservation policies are the only sources of difference in the expected outcome variables as all other mechanisms are already controlled in the analysis. This assumption seems less likely in the case of the initial years of PPCDAM in which satellite-aided enforcement was implemented in the Brazilian Legal Amazon, a region that includes the whole biome border and our sample of farmers located at both sides of the border. In the case of the soy moratorium and the conditional credits, they are both biome-targeted policies. If farmers cannot differentiate their status, meaning they cannot differentiate between Cerrado or Amazon biome, the evaluation would be noisy. We assume that farmers know their biome type. All biomes in Brazil were officially recognized by the Ministry of the Environment (due to ratification of the Convention on Biological Diversity) in 1996 (Aguiar *et al* 2016). Before the PPCDAm was first established, compliance with environmental protection in the Brazilian Legal Amazon region requires farmers to maintain 80% and 35% of farmland as natural forest in the Amazon and Cerrado biomes, respectively. This differential in thresholds for compliance was established by 2001 and later relaxed in 2012 with changes in the FC (Soares-Filho *et al* 2014). Such restriction in

[25] Sweden has a total area of aprproximately 45 million ha.

the use of land was established before biome-targeted policies were implemented. Moreover, whenever biome-targeted policies affect agricultural profitability coming from deforestation, incentives are in place for farmers in the affected biome to reduce deforestation and potentially to increase their productivity through intensification. We argue therefore that all estimated differences in outcomes between farmers at one or the other side of the border since the establishment of biome-targeted environmental policies represent the total policy effects of interest.

We compare our estimations on deforestation and intensification related outcomes to investigate whether, how, and to what extent biome-based conservation policies effectively reduce deforestation while promoting intensification in the use of land at the farm level. We interpret our results based on our economic model and empirical specification presented in the previous section. In the next section, we present our descriptive and empirical results.

5.4. Results

This section presents the results of our analysis on the impact of conservation policy instruments on deforestation and land use intensification decisions in the Brazilian agricultural frontier. We conducted falsification tests to validate our approach on four different predetermined covariates: total land endowment, forest endowment in $t-1$, accessibility to markets, and density of fines in the previous two years to t. We present these tests in section A.3.c in Appendix A.3. In the remainder of this section, we show our results from a descriptive analysis followed by our results from our RD design for the years 2001-2017.

5.4.a. Deforestation and intensification at the biome border

The amount of deforestation represented by the farmers on our total sample at 100 km distance to the biome border is depicted in Figure 4-2. Bars are stacked on the total amount of forest conversion to cropland or pasture. Each block of stacked bars shows the amount of deforestation per year for a) all farms, b) only farms in the Amazon biome, and 3) only farms in the Cerrado biome (left-hand side, middle, and right-hand side, respectively).

In our area of study, deforestation inside farms sums to approximately 110,000 square kilometers in the Amazon biome from 2001 till 2017. Total deforestation in the Cerrado mount to 135,000 square kilometers in the same period. This is around 51% of total deforestation reported for these biomes in the same years by official numbers.[26] The peak years of deforestation in all farms analyzed were during the first five years. Since 2006, total deforestation was reduced drastically and maintained on average at 12,836 km^2. This represents a reduction of 29.4% to average annual deforestation in the previous years (18,191 km^2). We consistently observed greater deforestation inside farms in the Cerrado biome, except for the year 2003 in which those in the Amazon account for more area deforested. Since 2009, deforestation in Cerrado farms has been even more pronounced.

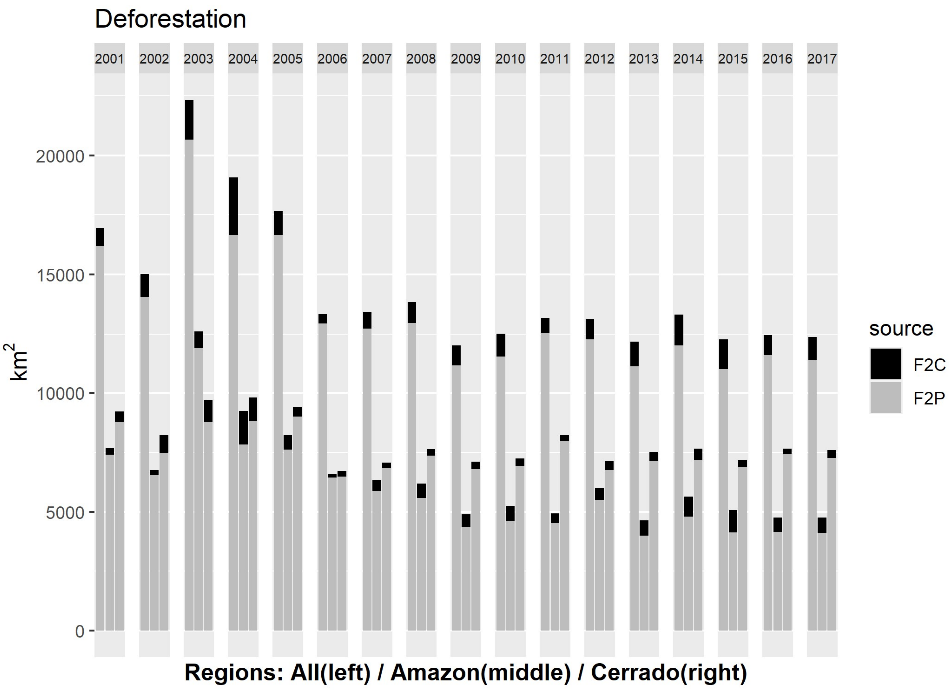


Figure 5-2 Deforestation in private farms at the Amazon-Cerrado biome border

Note: Stacked bars represent the amount of gross deforestation in farms identified at 100 km distance from the biome border. The figure depicts deforestation from 1) all farms (left bar), 2) only those farms located in the Amazon biome (middle bar), and 3) only those located in the Cerrado biome (right bar) for each year. F2C and F2P correspond to forest areas in a previous year transformed into cropland or pasture, respectively, in the year of reference.

As expected, most deforestation within farmland is due to pasture conversion which amounts to 77.7% (190,468 km^2) of total forest conversion in our sample. Except for the year 2003, Cerrado farmers have converted more areas to pasture than those located in the Amazon with a stronger difference since 2008 (see also fig A.3.8). Farmers in the Amazon carry out more forest to cropland conversion than those on the other side of the biome border after 2003. Only in 2006, forest to cropland conversion is less in farms located in the Amazon biome (see also FigureA.3.9).

[26] The amount of deforestation reported between 2001 and 2017 is approximately 208.2 and 272.2 thousand km^2 for the Amazon and the Cerrado biomes, respectively (http://www.inpe.br/).

We depict the amount of pasture to cropland conversion in Figure 5-3. Our sample shows that farmers reduced intensification jointly with deforestation from 2004, and reaching the lowest amount of area converted in 2006. It coincides with our results in which all forest conversion for agricultural use plummeted in that same year (Figure 4-2 above). Hence, if there exists an effect of conservation on deforestation is combined with a reduction in intensification, as proposed by our H.2. From 2007 till 2014, expansion of cropland into pasture areas permanently increases when we consider all our sample of farmers (left bars in each year in Figure 5-3.), while gross deforestation remained rather stable (see Figure 4-2), which corresponds to our first hypothesis (H.1). This intensification tendency is observed in both biomes, but more strongly to farmers in the Amazon. By 2014, the rate of intensification was slightly above 6,000 km^2 and decreased to half this amount three years later.

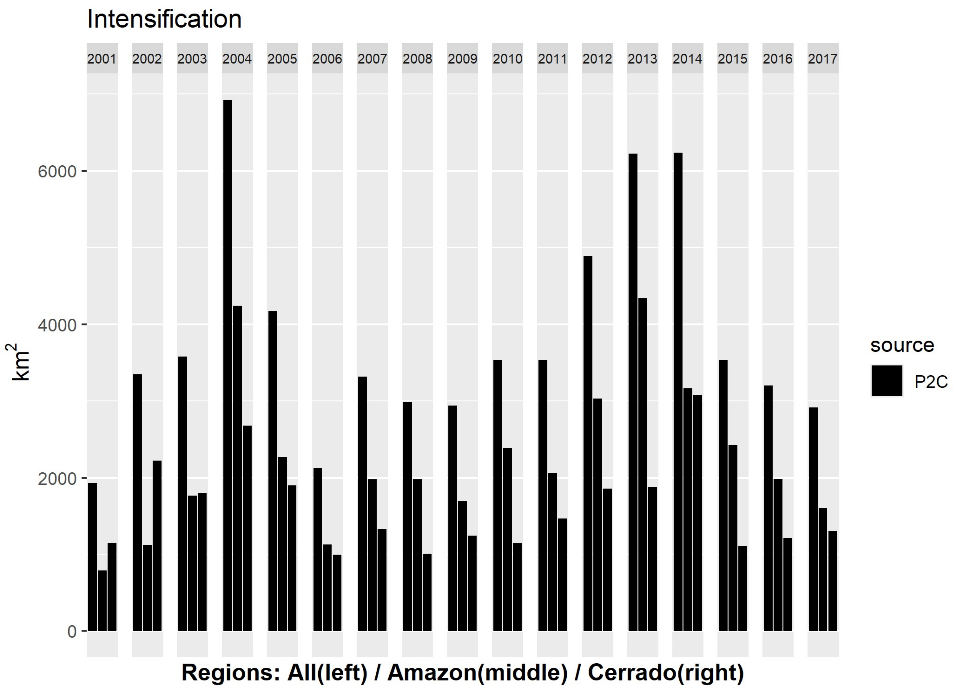


Figure 5-3. Pasture to cropland conversion in private farms at the Amazon-Cerrado biome border

Note: Bars represent the amount of land use intensification in farms identified at 100 km distance from the biome border. The figure depicts intensification from 1) all farms (left bar), 2) only those farms located in the Amazon biome (middle bar), and 3)only those located in the Cerrado biome (right bar) for each year. P2C corresponds to pasture areas in a previous year transformed into cropland in the year of reference.

These descriptive results show reductions in deforestation at the same time in which conservation policies were implemented, particularly when biome-targeted policies were established. This suggests, in line with previous analyzes, that important reductions in deforestation coincide with times of stronger environmental governance in Brazil. Moreover, our descriptive results suggest a policy-induced intensification effect. To disentangle the effect of conservation policies on land use configurations, particularly of biome-targeted policies, we now turn to our RD approach.

5.4.b. Policy effect on land use configurations

In this section, we present different effect estimations of biome-targeted policies on deforestation and intensification decisions. We compare farmers subject to policies to those in the Cerrado as a control group. Figure 4-4 shows our RD point estimates for the deforestation outcome and the green lines represent 95% level confidence intervals (see also Table A.3.2 in the Annex). The average difference in deforestation is -0.45 hectares between the years 2001 to 2005. From 2006 till 2012 this difference increase to 1 ha on average. In the last years of analysis, the difference is again shortened with -0.68 ha. The highest differences both in magnitude and statistical significance are observed in the years 2006 and 2007 (-2 and -1.7 hectares, respectively). Besides the years 2006 till 2009, we do not observe over time a systematic reduction in deforestation due to biome-targeted conservation policies in place. These empirical results are suggestive of our hypothesis H.3 up to 2009, in which the expected risk of punishment by offenders is arguably higher in the early stages of policy implementation. These effects did not remain after 2011 which suggests our fourth hypothesis, i.e conservation policy effect on reducing deforestation is not realized.

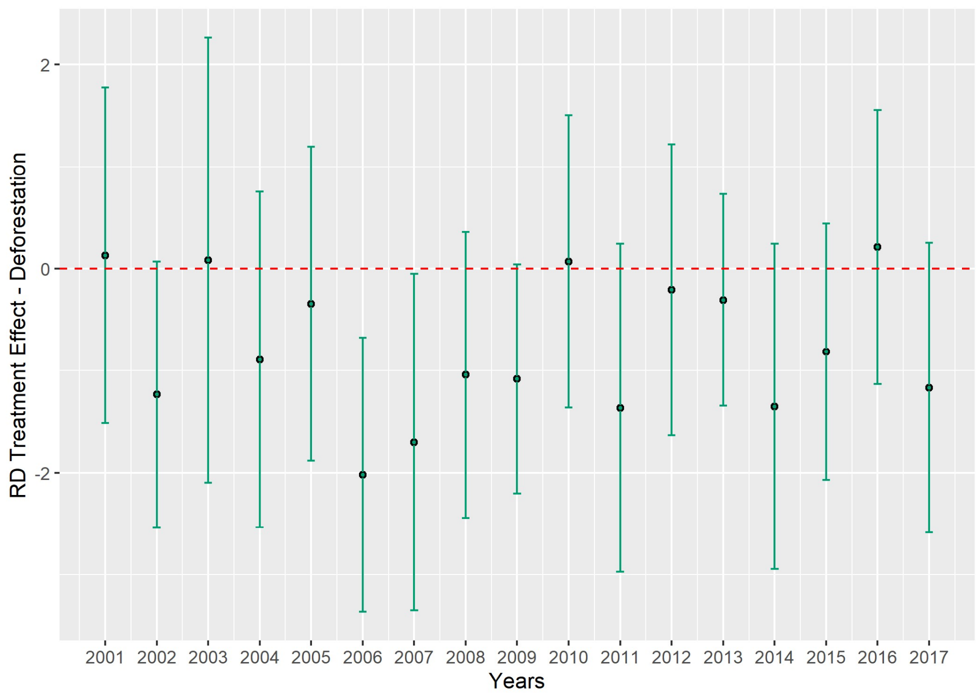
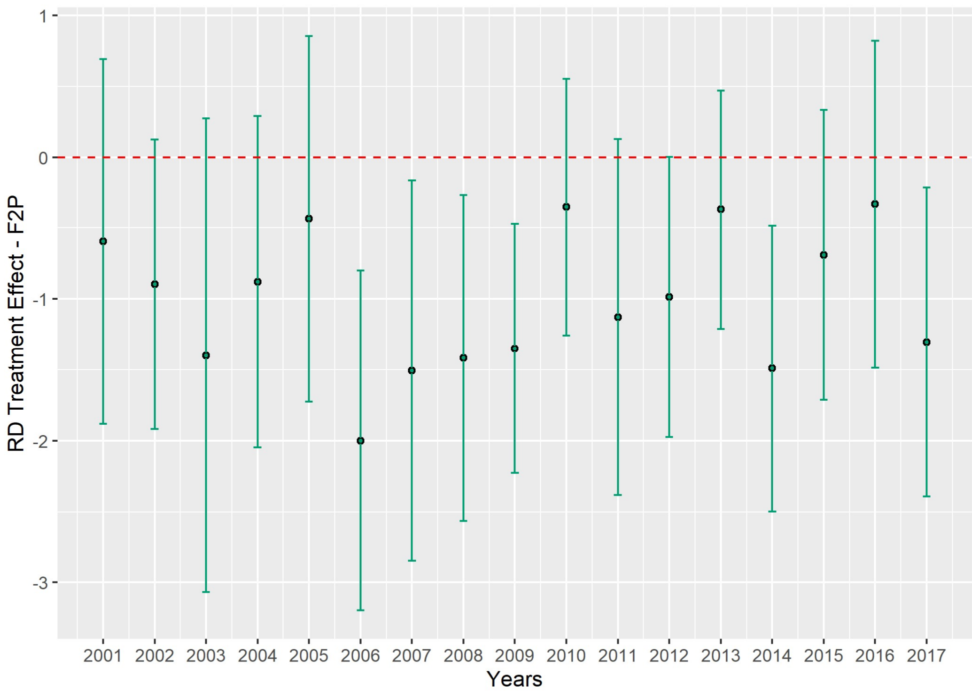


Figure 5-4. Deforestation – RD estimation (2001-2017)
Note: Estimated differences of *gross* deforestation measured in hectares. The lines depict 95% confidence intervals for the point estimates.

We break down forest conversion into pasture or cropland expansion to further analyze the causal mechanisms affecting farmers' land use decisions. Our estimations are depicted in Figure 4-5 (see also Tables A.3.3 and A.3.4). In panel A) we show the effects on forest converted into pasture for a given year. We see a systematic negative difference at a 95% confidence level from 2006 till 2012. These differences are stronger both in magnitude and significance in comparison to our analysis of total deforestation regardless of its agricultural use above. Only in 2010, the estimated impact is not significant and in 2011 it is significant at a 90% level. The average difference in conversion for these seven years is -1.25 ha (≈2% less deforestation than the average from 2001 till 2005). In the succeeding years, this gap in deforestation is reduced to the same difference as in the years before policy implementation (-0.84 ha). Forest to pasture conversion is frequently associated with the creation of land reserves to obtain gains other than those related to agricultural production. Our results confirm that conservation policy reduces deforestation (as proposed in our hypotheses H.1 and H.2). In panel B) we report estimates of the difference in forest conversion to cropland. We observe small differences in magnitude with no statistical significance between farmers on one or the other side of the border in our period of study, except for the year 2003, a time in which world soy prices were on the rise and dropped sharply after peaking in 2004 (Martinelli *et al* 2017). The average differences in conversion are calculated at 0.21 ha (2001-2005), 0.00 ha (2006-2012), and -0.10 ha (2013-2017). Our results suggest that Cerrado farmers have engaged in more conversion of forest to cropland (in 11 out of the 17 years of study); however, this difference is decreasing and even starting to reverse in recent years (from 2013-2017).

A)



B)

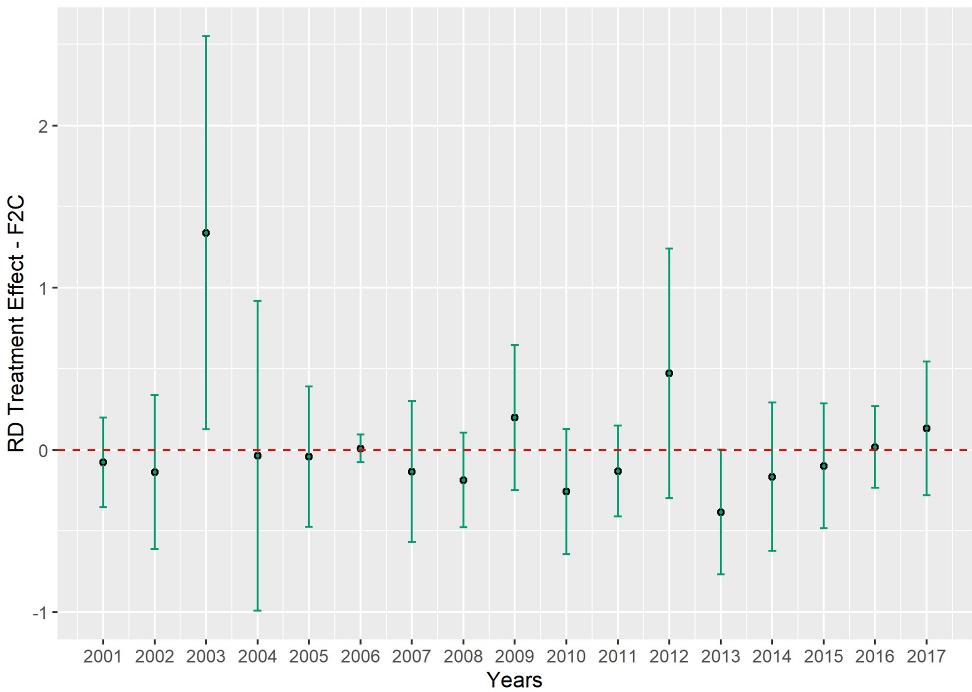


Figure 5-5. Sources of deforestation – RD estimation (2001-2017)
Note: Estimated differences of A) forest to pasture (F2P) or B) forest to cropland (F2C) conversion are measured in hectares. The lines depict confidence intervals for the estimation points at 95% level.

Figure 4-6 focuses on intensification effects measured as pasture to crop conversion (see also Table A.3.5). We estimate average effects of -0.25 ha from 2001 to 2005, -0.37 ha from 2006 till 2012, and -0.05 ha from 2013 till 2017. Like with our deforestation and forest to pasture outcomes, the year 2006 shows the largest (statistically significant at 95% level) effect on intensification with -0.40 ha. These results suggest a policy effect as described in our second hypothesis (H.2) in which the policy effect on deforestation and intensification move in tandem. By 2008, it reached a statistically significant (at 90% level) difference of -0.84 ha. Between 2009 and 2012, we found that this difference is reduced to -0.25 ha on average, which suggests a policy-driven intensification effect. If this gap reduction is due to an increase in intensification in the Amazon more than that in the Cerrado, this result would speak our first hypothesis (H.1), i.e., conservation policy bites on budget-constrained farmers and a policy-driven intensification effect follows. However, we do not find a statistically significant effect, i.e. no systematic differences due to policy are observed across these four years. Similar to the forest to cropland conversion (see Figure 4-5(B)), we estimate a reduction in the difference of conversion between farmers in the Amazon and Cerrado after 2012, i.e. no policy-driven intensification effect. These results are an indication of our second hypothesis, in which intensification follows deforestation trends.

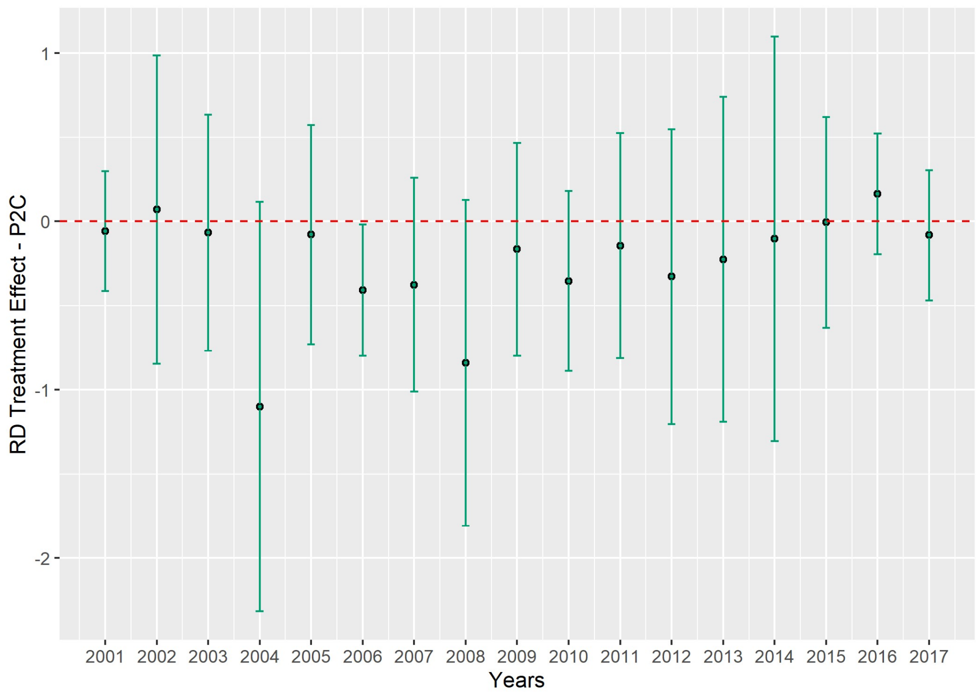


Figure 5-6. Intensification – RD estimation (2001-2017)
Note: Estimated differences of pasture to cropland (P2C) conversion measured in hectares. The lines depict confidence intervals for the estimation points at 95% level.

Comparing our various results, we observe signs of effective policy implementation for the early years since biome-targeted policies were established. This supports our third hypothesis, in which a perceived willingness of the government to enforce environmental regulations exists. From 2003 till 2010, Brazil's federal government was strongly committed to reducing rates of forest conversion in the Amazon. Legislative debates to reform the Forest Code began in 2010, the same year a new presidential term started in Brazil (see Figure 4-1 above). Since the enactment of the new FC, the federal government has gradually reduced its support to forest conservation, reducing government spending in many sectors, and recently focuses increasingly on boosting agricultural production and infrastructure expansion. This observation aligns well with observed results suggesting increasing evidence for our fourth hypothesis in which perceived willingness from the government to enforce environmental regulations is small.

5.5. Discussion and Conclusion

In this chapter, we study whether, how, and to what extent biome-targeted conservation policies affect farmer's land use decisions. We focus on the direct outcomes of private land users' decisions to deforest or intensify land use. Conservation policies are assumed to affect these decisions via credit access constraints and temporal variation in the credibility of the federal government's commitment to enforcing environmental regulations. We observe that in times of strong political support for forest conservation, the effect on reducing deforestation is also strong. In times when agricultural and infrastructure developments are priorities regardless of environmental protection, we observe small or no effects on deforestation. Intensification has moved in tandem with deforestation in most of the period we analyzed. When credit restrictions were implemented in 2008 and up to 2012, however, we observe a reduced difference in intensification between farmers in one versus the other side of the biome border which suggests a policy-driven intensification effect. In the same years, the difference in the conversion of forest to pasture remained significantly negative. Our results suggest that the joint implementation of value-chain restrictions and credit constraints based on environmental performance induce synergies between the reduction of forest conversion together with land use intensification, although we only observed it for a short period (2006-2012). Besides, these results suggest that win-win situations are achieved when the right incentives are tackled by a policy, as when the credit restrictions affect budget-constrained farmers (constrained, for instance, in the case in which it would be profitable for a farmer to invest but she cannot raise the funds).

In the second decade of analysis, the success in reducing deforestation and potential synergy effects in intensification described in the first decade disappeared. We attribute it to the lack of credibility in the willingness of the government to enforce environmental regulations. Since 2010, strong lobbying was done in the Brazilian Congress to reform the Forest Code which was later approved in 2012. One important change for farmers is a given amnesty to offenders before July 2008. Moreover, less budget has been assigned to institutions monitoring and enforcing environmental law in the last decade (Carvalho *et al* 2019). In the view of our results and as deforestation in Brazil is a profitable activity to do, we expect (*other things being equal*) increases in deforestation unless the government shows a renewed commitment to forest protection through action (see also Figure A.3.1 for observed levels of forestland reserves and compliance with conservation through time).

Our empirical results should be interpreted with some caveats in mind. First, we make an empirical analysis that uses a natural border as an identification strategy for our RD approach which reduces confounding problems from taking coarser administrative borders (e.g. nation-states), but it is not exempt from the problem of compound treatment effects (Keele and Titiunik 2014). Therefore we cannot single out the effects of individual conservation policy strategies, such as the soy moratorium and the conditional credits, but rather measure their compound effect. We can only observe the effect of the soy moratorium for the years 2006 and 2007, as the credit restrictions were implemented in 2008. It is important to emphasize that different rules from the Forest Code apply in one or the other side of the biome border, which explains differences in absolute land use; yet, our interest is to investigate how these differences change over time. Therefore, our results must be interpreted as the marginal effect of the policy instruments that impose a discontinuity in land use incentives at the biome border. One area of improvement is to implement a difference-in-design approach which is similar to that of difference-in-difference in the use of panel data (Eggers *et al* 2018). This is an alternative approach that disentangles compound policy effects in the interpretation of an RD method; however, it increases in complexity with both more policies to disentangle and years analyzed.

Second, we use information from the SICAR database and assume that those polygons represent individual farmers that have existed throughout our period of analysis. We cannot test this as we do not have information on the year of establishment. We also do not know when polygon areas were declared for inclusion in the rural registry database. We might be capturing some settlement processes during the years of analysis which can reduce the power of our estimations if the differences estimated systematically differ at the biome border. Evidence has shown, however, that amount of farms in the Brazilian Legal Amazon region has been rather stable in the past decades (Ferreira Filho and Vian 2016). An additional problem is that some polygons in our database might be owned by the same person which masks land concentration and its effect on decisions at the agricultural frontier. Moreover, previous studies have shown that the CAR did have a short-term increasing effect and medium-term negative effect on deforestation (Azevedo *et al* 2017, Jung *et al* 2017). This can give an additional compound effect on our estimations as it was only implemented in Mato Grosso (with both Amazon and Cerrado biomes) and Para (only within the Amazon biome) from 2008-2009 till 2012. Private farms' information on farm establishment and their time of registration can be added as previous researchers have done by collecting surveys for some segments of the Brazilian Legal Amazon region; however, to collect this information for our sample of 139,000 farmers remains a challenge (Gibbs *et al* 2016).

Third, we are estimating average treatment effects at the border of the Amazon and Cerrado biomes but we cannot generalize our results to the rest of the biomes. Also, we did not disentangle spatially heterogeneous treatment effects. The Brazilian Legal Amazon region hosts diverse agricultural activities with different incentives to deforest. Regarding external validity, our results might not apply in other parts of the BLAR, however, our sample at 100 km was responsible for half the total amount of deforestation and with similar trends of change as the official numbers reported in both biomes (see descriptive results above). Therefore, we believe our results might apply also in other areas of the Brazilian Legal Amazon region, and they illustrate the effect of biome-targeted conservation on those farmers pushing the agricultural frontier. An additional improvement is to divide the analysis in different grids along the border to capture spatial heterogeneous effects of conservation policies.

Besides these considerations, we expect our general conclusions on farmers‘ land use configuration decisions to remain valid. Our hypotheses H.3 and H.4 related to the credibility on the conservation agenda determine when and where hypotheses H.1 and H.2 (related to incentives to deforest and/or intensify) materialize. Therefore, expect to find similar tendencies, though perhaps changes in magnitude, if the considerations above could be addressed through methodological refinement and better data.

In sum, our research has used economic theory with quantitive methods to test the impact of conservation policies and stronger environmental governance on farmers‘ deforestation and intensification decisions. Our results are in line with previous studies that effective policy implementation reduces deforestation, and also suggest that they can induce intensification. All these changes are only possible if the government sends the correct signals to the Brazilian population, and farmers in particular, about its willingness to protect forest areas. Moreover, it can lead to synergies with agricultural development if the policies in place do tackle the right incentives (e.g. as with the credit constraints). With the current political and governance context, our results suggest that deforestation in the Amazon and Cerrado will continue untamed. Also, if the speculative mechanism behind deforestation becomes stronger, for example, through more investments in infrastructure or fewer investments in environmental enforcing, levels of deforestation can reach similar peak levels experienced in the mid-1990s and mid-2000s. Finally, our analysis adds empirical evidence to the argument that a rural sustainable development in Brazil cannot be reached simply through R&D for intensification while sparing land from conversion. Instead, intensification, especially in the land abundant and export-oriented Brazilian agricultural system, can result from the introduction of technology-driven intensification options under an effective environmental governance regime.

PART III

Conclusion

6. Conclusion

6.1. Drivers of land use in the Brazilian agricultural frontier

Brazil faces significant challenges in managing the substantial growth the agricultural sector has undergone in the last few decades, which has partially expanded into tropical forest areas in the Brazilian Legal Amazon. This region, which includes both portions of the Amazon and Cerrado biomes, hosts the agricultural frontier areas that were analyzed in this dissertation. The conceptual framework, guiding the empirical analysis, focuses on the economic and policy mechanisms driving decisions in the agricultural system with consequent levels of deforestation and land use intensification. The analysis is divided into three analytical chapters looking into key mechanisms emphasized in the conceptual framework in Chapter 1.

The first analytical chapter (Chapter 2) investigates the effect of underlying mechanisms affecting expectations on land use decisions in agricultural systems reflected in land markets. In particular, it looks at the effect of infrastructure development and conservation policy on land speculation in the Amazon and Cerrado biomes. Road infrastructure investments affects land prices, which reveal information on future forestland conversion. The results of this chapter also suggest that effective implementation of conservation policies reduce the speculative pressures on land prices in areas subject to policy; however, we did identify the potential for conservation policy leakage in land markets that fall outside of the policy's regional jurisdiction.

The second analytical chapter (Chapter 3) investigates the mechanisms affecting agricultural systems through the information revealed by agricultural markets. The sensitivity of the agricultural system to agricultural prices and conservation policy indicators is analyzed by estimating the output, input, and land elasticities of the system. The study looked at agricultural systems in Mato Grosso, one of the most dynamic states in terms of agricultural development and forest conversion. Cross-sectional agricultural market dynamics were found to induce changes in land productivity, accompanied by sizeable land use at the extensive margin. As expected, increases in crop prices induce stronger land use intensification effects than that of cattle or variable inputs of production. The results suggest not only the presence of market-driven intensification, but also of policy-driven intensification pressures. The effect of conservation policy enforcement, proxied as fine density at the district level, is nevertheless highly inelastic for all elasticities calculated.

The third analytical chapter (Chapter 4) evaluates the impact of conservation policy affecting agricultural systems by looking at changes in farm-level land use mixes over time. More specifically, it investigates the effect of value-chain governance measures and credit restrictions, which are only in affect the Amazon biome, on deforestation and land use change. Although these conservation policies aim chiefly at stopping forest conversion, the results in this chapter suggest a potentially beneficial but of small magnitude policy-driven land use intensification effect. In order to identify how policies effect the drivers of deforestation, we disaggregate whether forestlands are transformed into pasture or croplands after being deforested. Moreover, pasture to cropland conversion is used as a proxy to evaluate land use intensification decisions. At the onset of policy implementation, deforestation is reduced in farm boundaries that are subject to the aforementioned biome-targeted conservation policies. The reduction of deforestation is stronger for forest converted into pasture compared to that converted into cropland. Regarding a policy-driven intensification effect, the results suggest that when the policy affects budget-constrained agents, as seen with credit restrictions, then it is more likely to induce an intensification strategy. The analysis also points out that government signals to enforce environmental regulations are key to achieve desirable reductions of deforestation based on policy implementation. Since 2012, when the Brazilian Forest Code was reformed and with recent administrations cutting public expenditure for conservation policy enforcement, positive policy impacts were no longer observed in our analysis.

6.2. Policy implications

Sustainable land use remains an unaccomplished conservation objective partially due to continuous deforestation associated with agricultural production across the globe (Geist and Lambin 2002, Busch and Ferretti-Gallon 2017, Leblois *et al* 2017). Technological change, translated into an increase in agricultural productivity, is seen as one part of the solution if certain conditions enable land use at the intensive and not at the extensive margin (Byerlee *et al* 2014, Villoria *et al* 2014, Hertel 2018). Conservation policies are a complementary element of development to the extent that they effectively reduce forest conversion (Lambin *et al* 2014). However, their effectiveness is subject to the quality of (environmental) governance and can be diminished by potential leakage effects. This thesis empirically investigates these aspects in Brazil with important implications for policy design and implementation in agricultural frontier areas in the Amazon and Cerrado biomes.

First, infrastructure can have different impacts on rural development and forest conservation (Asher and Novosad 2020, Kaczan 2020). In Brazil, anecdotal evidence suggests that land speculation fueled by infrastructure improvements, such as roads, can lead to increases in deforestation. The empirical analysis in Chapter 2 supports this argument and warns that the anticipation of improvements to infrastructure can lead to speculative forest conversion. It also shows that stronger environmental governance and monitoring efforts can reduce this effect if implemented properly. Our study recommends that infrastructure projects require specific safeguards to avoid potential negative impacts of rural development. For instance, protected areas are considered a powerful instrument in reducing deforestation in Brazil, which can also act as a shield against plans for future infrastructure improvements (Soares-Filho *et al* 2010, Nolte *et al* 2013). It is important to note that these infrastructure development safeguards should be designed with a long-term perspective in mind; otherwise, they may only postpone speculative land transformation due to eventual expected improvements to infrastructure (Fearnside 2017). In terms of potential leakage effects, measures to protect forests should also cover other areas threatened by further agricultural frontier expansion across different biomes. Despite the potential reductions in deforestation that may result in the implementation of the aforementioned schemes and policies, there are strong pressures to downgrade, degazette, or downsize existing protected areas in recent years (de Marques and Peres 2015). Additionally, the results of this chapter suggest that land price changes can act as an early-warning system of potential deforestation hot-spots, thereby serving a complementary monitoring tool.

Second, there are challenges in promoting agricultural productivity for food, feed, fiber, and fuel production but without undermining ecosystem functions and biodiversity through the expansion of the agricultural frontier into natural areas. Subsidies for agricultural production can help to create incentives for intensification at a first glance, e.g. support for the use of inputs of land-intensive soy production. Chapter 3 uses the example of Mato Grosso, to highlight that higher agricultural prices can increase production by intensification, but also in great part by increased agricultural expansion into forest and non-forest areas. It is relevant for policy design to acknowledge that technological change alone does not reduce pressure on forest areas and that it may even increase pressure in areas with vast forest resources like in Brazil. Thus, support for technological change in agriculture must be accompanied by restricting land supply together with effective environmental policy enforcement. A mandatory forest reserve inside private farms is part of the environmental law in Brazil, however, it has been often been subject to loopholes and subversions (Soares-Filho *et al* 2014). Market-based instruments are potential alternatives to costly command and control policies that rely on effective law enforcement. They can come in the form of subsidies for land to be maintained as forest (e.g. climate change mitigation payments such as REDD+), or as taxes either targeting direct use of forest areas (e.g. land-use tax) or on the externalities derived from its use (e.g. Pigouvian tax) (Souza-Rodrigues 2019). However, most of the examples of market-based instruments have proven to be exceedingly expensive to implement (Cunha *et al* 2016). Moreover, unless the aforementioned land use restrictions are *de facto* implemented, they would be insufficient to halt the expansion of land transformation at the extensive margin of Brazilian agricultural systems.

Third, the importance to halt deforestation lays not only in national governments' actions but also on private firms and related international trade intermediaries that source their products from frontier areas. The analysis conducted in Chapter 4 supports previous anecdotal and empirical evidence that value-chain related instruments can and do reduce incentives for deforestation at the farm level and that the focus on value-chain and trade-related instruments is of great importance in Brazil. In 2019, great steps were taken to reach a trade agreement between the MERCOSUR region (comprising Brazil, Paraguay, Argentina, and Uruguay) and the European Union (EU). This is not a trivial agreement for global sustainability objectives, as approximately 20% of soy and 17% of beef imports to the EU from Brazil are linked to deforestation in the Amazon and the Cerrado biomes (Rajão *et al* 2020). Improvements in traceability can be possible through stronger technological cooperation and knowledge sharing in South-South and North-South cooperation initiatives (Kehoe *et al* 2020). This could also increase the option to effectively ban commodities that fail to comply with environmental standards and international agreements (e.g., the Paris Agreement tackling climate change), and move towards a more sustainable use of land resources. Additionally, scaling-up these measures beyond the Amazon biome has the potential to address leakage effects (within and across national borders) into areas with high biodiversity and climate change mitigation potential, such as the Cerrado, the Pantanal, or the Gran Chaco (Kehoe *et al* 2020).

Fourth, the willingness of the Brazilian government to enforce environmental laws and implement conservation policies is crucial to achieving desirable sustainable land use outcomes for economic development and the environment. Early in the 21st century, illegal deforestation was rampant in the Brazilian Amazon. Institutional capacities and progressive environmental legislation were *de jure* in place, but lacked effective coordination and implementation on the ground. Correspondingly, at the time, macroeconomic contextual factors represented the main drivers of forest loss. The results of this dissertation note a regime shift in enforcing environmental laws only two years after a new government took control of the country beginning in 2003. Amazon deforestation rates peaked in 2004 and then dropped by more than two thirds until 2012, when the legal basis for forest law enforcement was watered down and political changes, as well as economic downturn, increasingly resulted in public spending cuts. The 2012 reform of the Forest Code thus marked the second turning point in deforestation rates.

Fifth, this thesis clearly shows how these macro-level changes translate into farm-level decisions through economic mechanisms that can offer some lessons learned when analyzing other agricultural frontier areas. The first mechanism stems from the expectations of higher rents from land conversion due to expected improvements in the road network, which is reflected by higher land prices in the affected area. Notably, speculation-fueled deforestation can occur before infrastructure is built and can occur even when plans do not eventually materialize. A similar trend is observed in other South American agricultural frontiers. Due to insatiable global demand for soy and cattle, frontiers at the Gran Chaco in Argentina, Paraguay, and Bolivia are also subject to speculation pressures (Caldas *et al* 2015, le Polain de Waroux *et al* 2018). Different improvements to the primary road network in South America are in the pipeline which can lead to important impacts on forest conversion (Vilela *et al* 2020). If higher expected rents are linked to road improvements, deforestation is expected to increase even before infrastructure plans are materialized. As suggested in this thesis, higher land prices can potentially serve as an early-warning system for detecting future deforestation hot spots, which can be targeted to reduce deforestation. The second mechanism identified in this thesis stems from changes in agricultural prices and consequent increases in agricultural profitability, which we measure and explicitly calculate through the elasticities of the agricultural system. The analysis on the Brazilian agricultural frontier shows that even in circumstances that lead to market-driven agricultural intensification, increments in production at the extensive margin can lead to a Jevons paradox situation. This result can serve as a warning for other frontier areas that promote more intensification without considering adverse effects on forest areas. For instance, countries in the Congo Basin envision increased oil palm production due to rising regional and global demand for palm oil, in a region in which smallholders with low-yield technologies make up the majority of the oil palm producers (Ordway *et al* 2019). Intensification could be a reasonable way to meet demand through increased output while deterring forest encroachment. However, there is still a risk of a Jevons paradox, as agricultural profitability is expected to increase in areas with vast forest areas. Therefore, it is important to ensure that effective conservation policies are in place while increasing the productivity of agricultural systems (Phelps *et al* 2013, Ordway *et al* 2019).

In sum, societies across the globe face extraordinary challenges to secure a better world for present and future generations, while maintaining ecosystems vital for economic and social development in the 21st Century. Brazil has several low-hanging fruits at its disposal to overcome part of these challenges. The economic and policy mechanisms identified in this thesis have policy implications to increase the likelihood of win-win outcomes. Moreover, the outcomes of the analytical chapters provide examples for future considerations in the design and implementation of forest conservation strategies in agricultural frontier regions in Brazil and other countries. Yet, a key element for success in ensuring the sustainable use of land resources is the commitment from government agencies (at every level) to implement and enforce environmental regulations combined with the cooperation and collaboration from the private sector to adhere to these regulations.

6.3. Future research needs

Additional issues should be considered in future empirical research of assessing agricultural systems as drivers of land use change and subsequent policy recommendations. First, trade-offs concerning local livelihoods and the labor market may exist when agriculture productivity changes come together with unsustainable use of forestland resources e.g. increments in labor opportunities that attract more immigration to frontier areas, which in turn increase forest conversion (Angelsen and Kaimowitz 2001). Quantifying these trade-offs is beyond the scope of this thesis but it is important to acknowledge them. Efficiency gains that affect the value of factors of production (e.g. labor, land, or capital) will also bring changes in the income distribution of the population, which can exacerbate land concentration and income inequality (Hayami and Godo 2005). Additionally, non-timber forest products are recognized as sources of income for smallholders and indigenous groups, which would also be lost with increased deforestation in the Amazon rainforest (Shanley *et al* 2011, Duchelle *et al* 2012). These tradeoffs represent potential elaborations of issues beyond the scope of this thesis.

Second, up-to-date and more disaggregated data can improve the analysis performed in this thesis, particularly in capturing the heterogeneity of economic and policy mechanisms affecting land use decisions across space and time. Publicly available land use datasets are continuously improving, can be offered at high resolutions, and with more powerful algorithms to identify different land use processes. For instance, identifying land uses such as double cropping across space would help to link these areas to farm boundaries and improve the analysis of mechanisms driving intensification in agriculture. For the specific Brazilian case, there is new and publicly available census data, which can be used to update the analyses and improve the methods implemented in Chapters 2 and 3. More complex methods to identify spatial heterogeneity of conservation policy impacts were not performed in the analysis of Chapter 4, but they represent a straightforward extension of the analysis, i.e. divide the RD analysis in several RD analyzes for different segments of the Amazon-Biome border.

Finally, the support of both public and private institutions is crucial to provide researchers with relevant and up-to-date social, economic, and land use information to assist in identifying mechanisms affecting land use decisions in agricultural frontier areas. This is important in the telecoupled world in which we are living today. Value-chain processes and new trade agreements can bring benefits and challenges in which national and international firms, together with national governments, can help in tracing the impacts of export-oriented commodities on the environment in production regions. Moreover, this flow of information can increase transparency so that independent evaluations can be conducted to hold agricultural production accountable to environmental standards. Without this flow of information, empirical research will fall short in providing sound solutions that lead to more sustainable use of land resources at the scale needed for long-term global prosperity.

Appendix

A.1. Appendix – Chapter 2

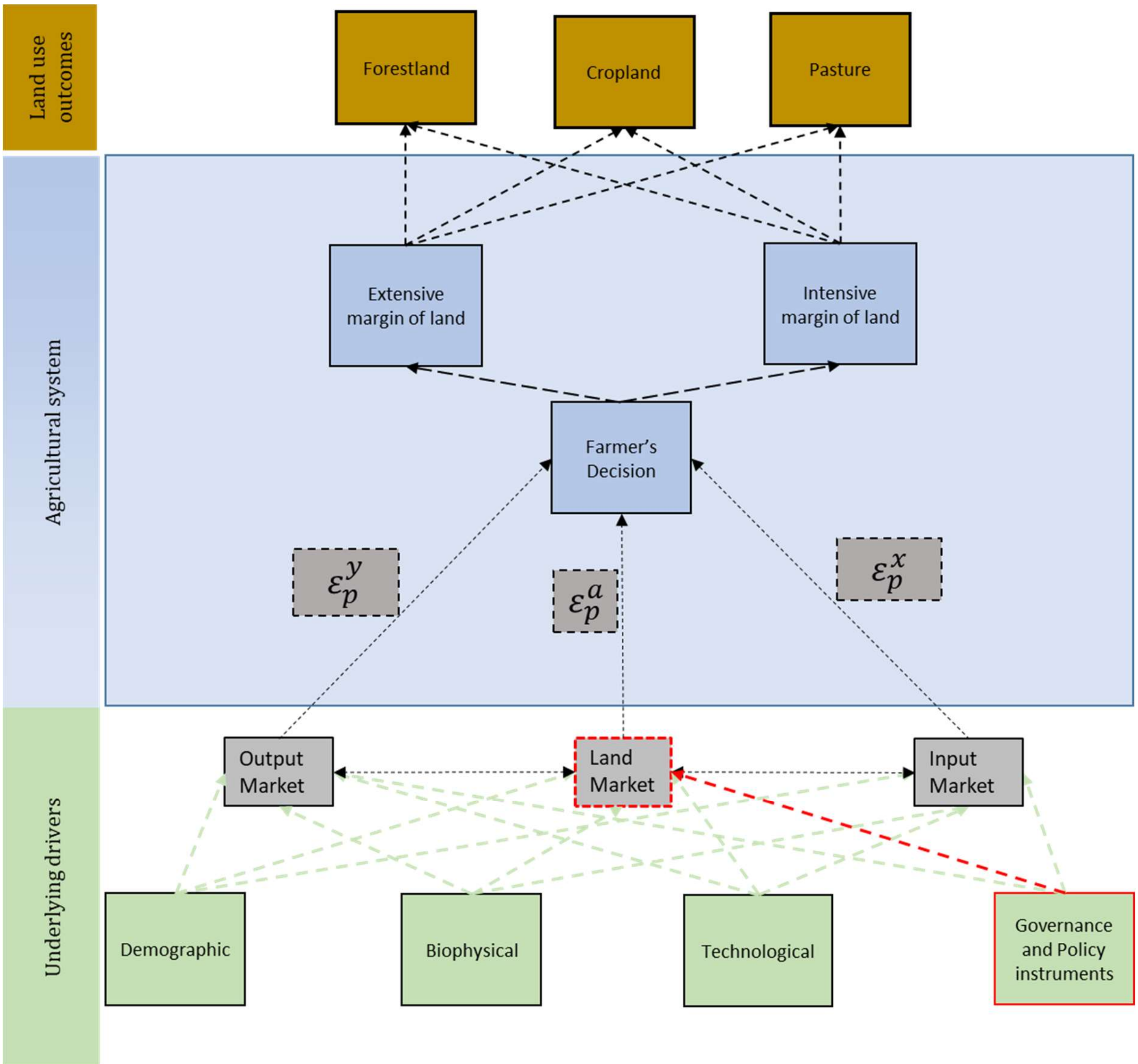


Figure A. 1. Mechanisms affecting land use decisions analyzed in chapter 2

A.1.a. Forestland price model

Forestland can be converted to the most profitable alternative use, e.g. pasture, which may give higher rental incomes (rent), according to the von Thünen model (Angelsen 2010). As the conversion of forestland is in uncertain future, expected rents under alternative land use patterns need to be considered. Hence, the price of forestland, P_{it}^F at location *i* at time *t* is the sum of the expected discounted stream of forestland rent, EDR_{it}^F, and expected discounted stream of pasture rent, EDR_{it}^P, net of the expected discounted conversion costs $EDCC_{it}$.

$$P_{it}^F = EDR_{it}^F + EDR_{it}^P - EDCC_{it} \tag{A.1-1}$$

Forestland rent

If land is forest at time *t*, its rent equals the marginal productivity (e.g. of timber and non-timber forest production) net of transportation costs. Let R_{it}^F be the pure forestland rent at time *t*; let ρ_t denote the probability that the forestland is converted to an alternative land use at the beginning of time *t*; and let r_t denote the discount rate, then:

$$EDR_{it}^F = (1-\rho_0)R_{i0}^F + \frac{(1-\rho_0)(1-\rho_1)R_{i1}^F}{1+r_1} + \frac{(1-\rho_0)(1-\rho_1)(1-\rho_2)R_{i2}^F}{(1+r_1)(1+r_2)} + \cdots \tag{A.1-2.a}$$

$$EDR_{it}^F = (1-\rho_0)\sum_{t=0}^{\infty}\left(\prod_{s=0}^{t}\frac{1-\rho_s}{1+r_s}\right)R_{it}^F \tag{A.1-2.b}$$

Hence, forestland rent income at time *t*, R_{it}^F, occurs only if the forest had not been converted to an alternative land use at time *t*.

Assuming that ρ_t, r_t and R_{it}^F are constant over time, this simplifies to:

$$EDR_{it}^F = (1-\rho)R_i^F\sum_{t=0}^{\infty}\left(\frac{1-\rho}{1+r}\right)^t = \frac{(1+r)(1-\rho)R_i^F}{r+\rho} \tag{A.1-3}$$

Thus, if conversion probability ρ is zero, we have the standard expression $EDR_{it}^F = \frac{R_i^F}{r} + R_i^F$, i.e., expected discounted annual forestland rents.

Pasture rent

Pasture rent income R_{it}^P accrues on original forestland only after conversion to pasture, thus:

$$EDR_{it}^P = \rho_0\left[R_{i0}^P + \frac{1}{1+r_1}R_{i1}^P + \frac{1}{(1+r_1)(1+r_2)}R_{i2}^P + \cdots\right] +$$

$$(1-\rho_0)\rho_1\left[\frac{1}{1+r_1}R_{i1}^P + \frac{1}{(1+r_1)(1+r_2)}R_{i2}^P + \cdots\right] +$$

$$(1-\rho_0)(1-\rho_1)\rho_2\left[\frac{1}{(1+r_1)(1+r_2)}R_{i2}^P+\frac{1}{(1+r_1)(1+r_2)(1+r_3)}R_{i3}^P\dots\right]+\cdots \quad \textbf{(A.1-4)}$$

where the first line expresses the expected value if the land is converted at *t*=0, the second line the expected value if the land is converted at *t*=1, and so on. Assuming again that ρ_t, r_t and R_{it}^P are constant over time, re-arranging the sum gives:

$$EDR_{it}^P=\sum_{t=0}^{\infty}\rho\left(\frac{1}{1+r}\right)^t R_i^P+\sum_{t=1}^{\infty}\rho(1-\rho)\left(\frac{1}{1+r}\right)^t R_i^P+\sum_{t=2}^{\infty}\rho(1-\rho)^2\left(\frac{1}{1+r}\right)^t R_i^P+\dots$$

$$=\rho R_i^P\sum_{s=0}^{\infty}(1-\rho)^s\sum_{t=s}^{\infty}\left(\frac{1}{1+r}\right)^t \quad \textbf{(A.1-5)}$$

Which can be simplified to:

$$EDR_{it}^P=\frac{\rho R_i^P(1+r)^2}{r(r+\rho)} \quad \textbf{(A.1-6)}$$

In case of immediate conversion ($\rho=1$), the standard expression, $EDR_{it}^P=\frac{R_i^P}{r}+R_i^P$, for the land price holds, i.e. expected discounted annual rents of pasture.

Conversion cost

Conversion costs τ are assumed to be constant and only occur at the time of the conversion from forestland to pasture. They can contain explicit taxes and costs of conversion as well as implicit costs, such as fines from command-and-control based conservation policies:

$$EDCC_{it}=\rho\tau+\frac{(1-\rho)\rho\tau}{1+r}+\frac{(1-\rho)^2\rho\tau}{(1+r)^2}+\cdots$$

$$=\rho\tau\sum_{t=0}^{\infty}\left(\frac{1-\rho}{1+r}\right)^t=\frac{(1+r)\rho\tau}{r+\rho} \quad \textbf{(A.1-7)}$$

Price of forestland

Using the calculations above, the price of forestland, including the value and cost of converting the land to pasture, is:

$$P_{it}^F=\frac{(1+r)\left(-r\rho\tau+(r-r\rho)R_i^F+(1+r)\rho R_i^P\right)}{r(r+\rho)} \quad \textbf{(A.1-8)}$$

Note that without conversion probability ($\rho=0$), we would get the usual discounted forestland rents for the price of forest: $EDR_{it}^F=\frac{R_i^F}{r}+R_i^F$. Hence, this forestland price depends purely on (discounted) forestland rents.

Comparative static analysis

To derive testable hypotheses, we calculate the total derivative of forestland prices after some key components. Here with the assumption that land prices capture expected rents from forest conversion. The alternative would be that land conversion is not reflected in forestland prices and is only determined by rental prices. Based on our theoretical model and the objective in this chapter we are interested in two main components. First, the conversion probability ρ is considered in our model as the main source of a speculative behavior. If high expectations exist of ripping profits from conversion to pasture, then ρ would be close to or equal to 1. However, in a more stringent conservation scenario, we would expect a smaller likelihood of pasture conversion even if it might be profitable. Taking the derivate of the forestland price with respect to ρ (and without indexes to ease explanation), we obtain:

$$\frac{dP^F}{d\rho} = -\frac{(1+r)\left(r\tau+(1+r)R^F-(1+r)R^P\right)}{(r+\rho)^2} \qquad \textbf{(A.1-9)}$$

The expression is always positive if $R^P - \frac{r\tau}{1+r} \geq R^F$. Hence, whenever the pasture rent net of the discounted conversion costs is higher than the forestland rent (i.e. it would be profitable to convert the land); higher protection stringency (lower ρ) leads to lower forestland prices.

The second component of interest is the effect of a change in conversion costs τ. Taking the derivate of the relative forest price after the conversion costs, we get:

$$\frac{dP^F}{d\tau} = -\frac{(1+r)\rho}{(r+\rho)} < 0 \qquad \textbf{(A.1-10)}$$

which is always negative. Thus, higher conversion costs such as areas with dense vegetation, or subject to *de facto* collected fines, reduce the price of forestland.

Table A.1.1 summarizes our hypotheses that forestland prices convey information on future land conversion.

Table A.1. 1. Hypotheses summary

	Impact on forestland price	
Relevant components	*Anticipation of land conversion*	*No anticipation of conversion*
Conversion probability ρ	+	0
Conversion costs τ	-	0

Note: The signs for the anticipation case partially depend on the assumption that (pure) forestland rents are (sufficiently) lower than pure rents from pasture which is a common case for consolidated frontier areas as well as land that is expected to become part of a consolidated frontier area.

Leakage

The comparative static analysis focused so far only on the determinants of forestland prices in region *i* due to changes in policies or parameters in region *i*. Leakage describes any side-effects of a policy in region *i* that spills over to region *j* (Lambin and Meyfroidt 2011, Meyfroidt *et al* 2018). Relevant mechanisms of leakage are through changes in output prices of agricultural products, and comparatively lower costs of conversion or higher conversion probabilities. To see this in our framework, consider a change in conversion probability ρ_i or conversion cost τ_i in region *i* that reduces effectively total available land for alternative uses in region *i*. This, in turn, leads to reduction in farming products in region *i* and, thus, also in the aggregate over all regions. Because output prices for farming products are inversely related to production quantity, output prices increase, implying also higher rents for alternative land uses $R_{j \neq i}$ for all other regions. This impact is expected to be stronger if region *i* is a large region such that any region-specific policies have a significant impact on output prices. The same changes in conversion probability or conversion costs in region *i* will give other regions a comparative advantage to supply land due to relative lower conversion costs and higher probability of conversion (e.g. due to lower or no conservation law enforcement). Based on our model's hypothesis we expect that these mechanisms will contribute to higher forestland prices in areas outside region *i*.

Hedonic valuation

In our empirical strategy, we aim to test the hypothesis that forestland prices convey information on future land conversion. To achieve our objective we decompose land prices by evaluating the relative importance of attributes influencing forestland prices based on hedonic theory (Rosen 1974, Chicoine 1981, Snyder *et al* 2008). The fundamental hedonic equation is $P^F = h(Z)$. In which $h(\cdot)$ represents a functional relationship between the forestland price and its different attributes, z_i (eq. (2-6) in Chapter 2). One can derive the marginal implicit price, $\partial P^F / \partial z_i = \partial h(Z) / \partial z_i$, for each attribute which represents the additional value people would pay for a small change in the specific attribute (Shultz and King 2001, Snyder *et al* 2008, Ma and Swinton 2011).

This approach is also attractive as we cannot observe the key components of the theoretical model. We then investigate the effect of some relevant attributes that strongly influence those relevant components. Thus our strategy evaluates the transformation on levels of certain attributes that influence land prices (Sills and Caviglia-Harris 2009).

Following this notion, we specify a reduced-form model of forestland prices (see eq. (2-6) in Chapter 2). We estimate various specifications using a two-way fixed effects model (Baltagi 2016). In each specification, the dependent variable is the logarithmic transformation of average forestland prices. The first specification only considers those attributes that affect land rents and disentangles the effect of speculation measured by expected improvements in accessibility. In a second specification, we include our proxy for stringent conservation policy which allows us to capture leakage effects. As robustness check, we estimate the model using the first lag of all covariates instead of the contemporaneous values. The results of robustness checks are presented in Table A.1.4.

A.1.b. Data sources and processing

Forestland price

We use forestland selling prices as the dependent variable in all model specifications. This information is obtained from a survey applied to brokers, appraisers, and agricultural technicians two times per year by Informa Economics FNP (http://fnp.com.br/), an agribusiness consulting company. The information is divided in 133 land markets or micro-regions for all Brazil per year (exclusion of Sao Paolo and Rio de Janeiro regions due to highly distorted housing prices). These regions are defined by FNP to account for the highly heterogeneous Brazilian territory in terms of main municipality characteristics such as 1) access to markets of outputs and inputs, 2) types of land use, 3) climate, and 4) types of soil. This country division also looks into more specific characteristics such as intensity of farming activities, proximity to rivers, vegetation, average rainfall, amongst others. Besides this homogenous clustering of space, we find differences in the area size of each land market, in the number of observations sampled per region, and in the types of land considered, e.g. easy/difficult access Amazon forest or dense/open Cerrado. In the original information, three types of land are distinguished: cropland, pastureland, and natural land cover (forestland). We have access to an aggregated database with regions as the unit of observation, in which the median, mean, minimum, and maximum value of land was reported. The values of forestland prices throughout our period of study ranged between 8.7 and 2,785 $R/ha deflated at 2001 values. We use the mean value of forestland per region in our empirical model. We deflate this value to 2001 levels using Brazil's General Price Index- Internal Availability (IGP-DI). Finally, we apply a logarithmic transformation to land prices.

From 133 land market regions in the country (excluding Sao Paolo and Rio de Janeiro regions), we consider 79 that lie within the borders of the Amazon and the Cerrado biomes.

For only 61 of the 79 regions, forestland prices were consistently reported. The timeframe of our study (2001 to 2012) allows us to observe potential shifts in land speculation and conservation policy leakage before and after our policy shock of interest.

Rent indicators

A set of regional level attributes are used to predict rents from conversion of forestland. First, we create an aggregate revenue-weighted ***crop price index*** that includes all seasonal and temporal crops using information from the municipality annual agricultural survey of Brazil (Produção Agrícola Municipal - PAM in Portuguese). We use 2001 as base year. We include this attribute as we expect it to influence average rents of converted forestland. Second, we use the number of ***environmental fines*** issued by the Brazilian Institute for the Environment and Renewable Natural Resources (IBAMA). We assume that a higher density of fines would affect potential conversion costs and thus land conversion decisions. Third, we use the normalized area of ***protected areas*** within a region, using protected area coverage data that we obtained from the Brazilian Ministry of the Environment. Protected areas increase the scarcity of land for agriculture, which we expect to increase land prices.

Average ***accessibility from forest to major markets*** within a region is included to account for location impacts on rents (a discussion on its calculation is offered in the Speculation section below). We also include the proportion of ***soy suitable area*** on forestland in a region (Soares-Filho *et al* 2016). Soy suitability area considers bio-physical characteristics like slope, soils, and climate zones that favor soy production in particular, and agriculture in general (Gibbs *et al* 2015). We consider it as a proxy to control for *in situ* potential rents and the impact of agriculture development within a region. For the same reason, we control for cattle density in a region using data on cattle herd sizes from the Annual Ranching Survey (Produção Pecuaria Municipal - PPM in Portuguese).

Speculation indicator – Expected accessibility improvements

In Brazil, construction of new roads has been linked to land conversion and land speculation since the beginning of Amazon colonization (Hecht 1985, Pfaff 1999, Soares-Filho *et al* 2006, Fearnside 2008). To measure related expectations we construct an indicator of ***expected accessibility improvements***, by combining two accessibility maps with a 10 x10km resolution for each year. The first map measures travel distance for each pixel to the nearest district capital (i.e., relevant market), using information on existing roads in a given year. The second map updates travel times from the first map assuming that currently planned new roads and road improvements had materialized (see Table A.1.2). We then calculate expected accessibility improvements as the difference between the second and the first map.

Data processing involves different steps. We create a friction map using information on land cover, roads, water bodies, and the effect of the slope. Land cover is classified as areas with primary forest, secondary forest, and non-forest with data from Hansen *et al* (2013a, 2013b). Their tree cover (vegetation over 5 meters in height) map for the year 2000 is compared with yearly areas of tree cover losses (both anthropogenic and non-anthropogenic) between 2001 and 2012. The original data at a 30 x 30m pixel resolution is aggregated to a10 x10km resolution for computational reasons. Each new grid cell is classified as primary (60% +), secondary (30% - 60%), or non-forest (30% -) based on its percentage of forest pixels from a total of 6,400 in each grid cell. Historical road network information is obtained from Brazil's National Department of Transport Infrastructure (DNIT). Accessibility to roads in hours is calculated using a Knight's move algorithm from standard GIS software.

Table A.1. 2. Speed per road type used in accessibility maps

Type of road	Description	Factual Speed	Expected Speed
Planned Road	Road that has been planned and its construction is expected	NA	81 km/hr
Unofficial road	Road created from opening natural land without following the official norms for construction. Usually are not available all year	30 km/hr	30 km/hr
Unpaved road	Constructed with official norms. Available throughout the year	48 km/hr	48 km/hr
Improvement – construction of unpaved road	From unofficial or planned to unpaved road	30 km/hr	48 km/hr
Paved road	Roads that had been paved	81 km/hr	81 km/hr
Improvement – pavement of road	From unpaved or planned to paved road	50 km/hr	81 km/hr
Highway	Having two or more lines per direction	110 km/hr	110 km/hr
Improvement – Construction of highway	From paved or planned to highway	81 km/hr	110 km/hr

Note*:* The speeds were used in the creation of two different friction maps. The friction maps are one of the main inputs to create accessibility maps. The other input was a map identifying the municipality capitals as markets of interest. Our assumption on the speeds were made by consulting staff from IBAMA and their answers were based on travel-logs during their field operations.

Post-2004 Conservation

Since the mid-2000s, a stringent environmental governance was implemented in Brazil, particularly to protect the Amazon Biome.[27] To test the impact of this structural break in conservation governance, we include a proxy for stringent conservation policy which allows us to analyze potential leakage effects. First, we create a time dummy with before and after 2004 effects. We interact this with the share of districts within a land market region that are outside the Brazilian Legal Amazon region, and with our measure of soy suitability described above to capture a stronger pressure to convert more productive land. We expect that stringent conservation in the Amazon will pressure land resources strongly in less protected biomes. Based on our theoretical model, we would observe leakage effects if the data shows positive forestland price shifts in areas not subject to stringent conservation policy, i.e. areas outside the Brazilian Legal Amazon region.

[27] We test also the impact of protected areas but no variation prove to be significant, in contrast with other studies that was significant in explaining pasture and cropland values (see Cohn *et al* (2016)).

A.1.c. Area of study

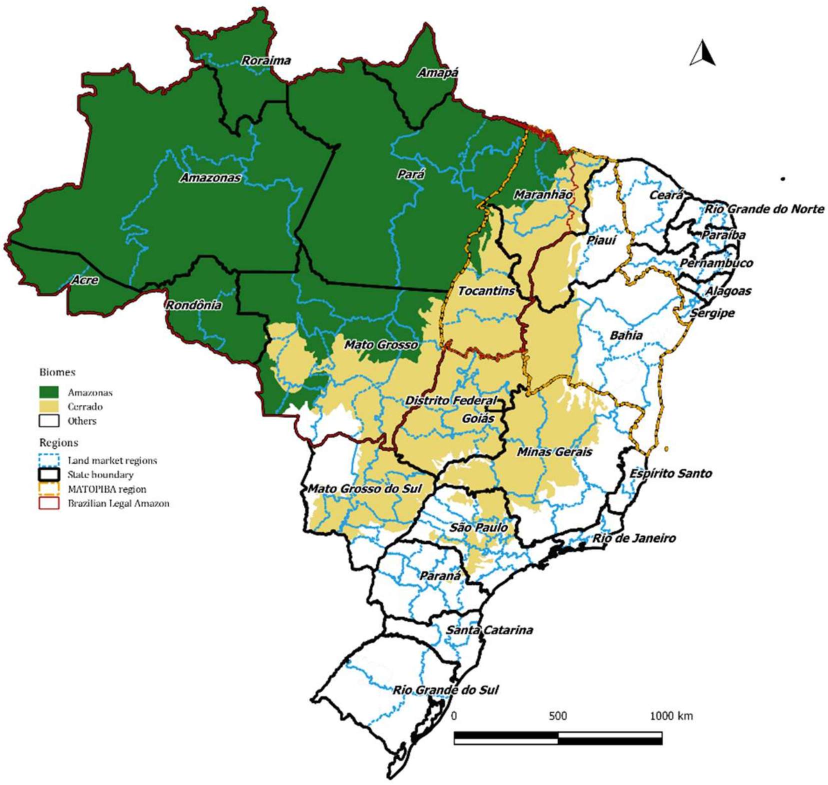


Figure A.1. 1. Area of study

Note: This map draws those regions of interest: Amazon and Cerrado biomes and the Brazilian Legal Amazon region (PPCDAm's area of influence). It also shows the MATOPIBA region, a cluster of states that has been highlighted as a very dynamic frontier area in recent years.

A.1.d. Forestland price – Average percentage change (2001-2004 & 2005-2012)

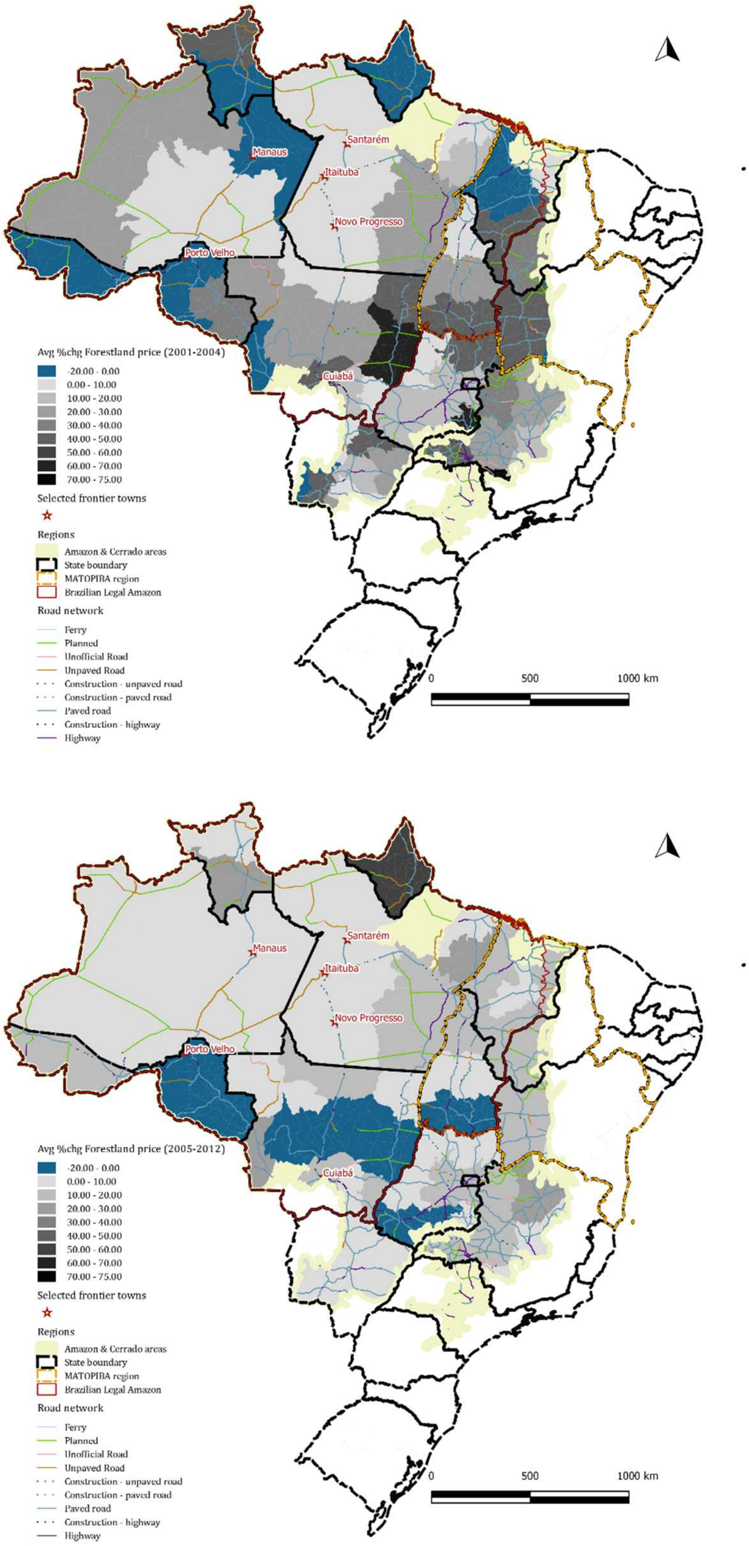


Figure A.1. 2 Average percentage change – Forestland prices
Note: The upper map shows the average percentage change on forestland prices for the period 2001-2004 for each land market region. The lower map does it for the period 2005-2012. Note an average reduction in states like Mato Grosso and increases in northern parts of the MATOPIBA region after change in conservation governance in 2004.

A.1.e. Contemporaneous models with full sample

Table A.1. 3. Regression results of speculation and stringent conservation analysis (full sample)

	*Dependent variable: ln*Forestland price	
	Speculation (1)	Stringent Conservation (2)
*ln*Expected Accessibility Improvements	1.510[b]	1.484[b]
	(0.723)	(0.700)
*ln*Crop price index	0.425[b]	0.451[b]
	(0.207)	(0.210)
*ln*Soy apptitude	1.115	3.123
	(3.846)	(3.893)
*ln*Protected areas	1.019	1.027
	(0.874)	(0.861)
*ln*Cattle density	0.881	1.042
	(0.689)	(0.700)
*ln*Accessibility	0.701	0.414
	(0.888)	(0.888)
*ln*Fines incidence	-1.654[a]	-1.629[a]
	(0.413)	(0.410)
*ln*Accessibility x nonBLA[♦]	-0.804	-0.477
	(1.002)	(1.024)
Post-2004 Conservation		1.513
		(0.958)
Time and regional fixed effects	Yes	Yes
Observations	744	744
R^2	0.083	0.091
F Statistic	7.507[a] (df = 8;663)	7.340[a] (df = 9; 662)

[a] Significant at 0.01 level. [b] Significant at 0.05 level. [c] Significant at 0.1 level.Robust standard errors are given in parentheses. [♦]nonBLA refers to share of districts outside the Brazilian Legal Amazon region in a land market region

A.1.f. Lagged models

Table A.1. 4.Regression results of robustness check models

	*Dependent variable: ln*Forestland price	
	Speculation	Stringent Conservation
	(1)	(2)
*ln*Expected Accessibility Improvements	1.692[b]	1.656[b]
	(0.717)	(0.695)
*ln*Crop price index	0.348	0.321
	(0.246)	(0.251)
*ln*Soy apptitude	4.836	6.626
	(4.014)	(4.038)
*ln*Protected areas	1.997	2.012
	(3.167)	(3.142)
*ln*Cattle density	0.438	0.477
	(0.687)	(0.686)
*ln*Accessibility	0.704	0.418
	(0.974)	(0.978)
*ln*Fines incidence	-1.166[a]	-1.169[a]
	(0.388)	(0.383)
*ln*Accessibility x nonBLA[♦]	-0.642	-0.336
	(1.071)	(1.083)
Post-2004 Conservation		1.639[c]
		(0.863)
Time and regional fixed effects	Yes	Yes
Observations	682	682
R^2	0.100	0.109
F Statistic	8.348[a] (df = 8; 602)	8.164[a] (df = 9; 601)

[a] Significant at 0.01 level. [b] Significant at 0.05 level. [c] Significant at 0.1 level. Robust standard errors are given in parentheses. [♦]nonBLA refers to share of districts outside the Brazilian Legal Amazon region in a land market region.

A.1.g. Hot spot maps

To create hot spot maps of deforestation presented in the chapter (see Figure 2-4) and Figure A.1.3 below, we follow Harris et al (2017), but using a different temporal analysis of change. We identified deforestation hot spots by 1) identifying each year's levels of spatial clustering of deforested areas, and 2) how these spatial clustering change over the period of analysis. First, we used 30 x 30 m resolution pixels to identify incidence of forest cover loss within areas of approximately 2.5 x 2.5 km^2 (Hansen *et al* 2013a, 2013b). We applied a Getis-Ord Gi* statistic for each area to spatially identify the level of clustering of forest cover loss (Getis and Ord 1992, Harris *et al* 2017). We obtained a Z-score and a p-value for each area observation which represent metrics of spatial clustering. A higher Z-score means stronger clustering, and we used a p-value < 0.05 as our threshold of significance to classify an area as a hot spot (i.e. observations with higher p-values are considered a "not hot spot"). As in Harris *et al* (2017), we did not consider cold spots of deforestation, i.e. negative Z-score values. To identify degree of intensity through time, we compared the Z-scores for each pair of years. If the score in the second year was higher, we coded our temporal measure as 1; as -1 if it was smaller, and 0 otherwise. We added these values for the whole period to obtain our metric of temporal frequency. Finally, we identified five types of hot spots by looking at their intensity and frequency over time: 1) reduced (≥75% of years classified as a hot spot and a negative temporal measure), 2) intermittent (>0% and <75 % years classified as a hot spot), 3) constant (≥75% of years classified as a hot spot and temporal measure=0), 4) increased (≥75% of years classified as a hot spot and a positive temporal measure), and 5) new (if any of the last two years were classified as a hot spot).

Below we present two maps of forest cover loss hot spots as additional visualizations on conversion and conservation policy leakage in our period of analysis. The upper map in Figure A.1.3 shows the type of hot spots identified in the period 2001-2004. A high concentration of increased and new hot spots are clearly observed in Mato Grosso state in the South-west part of the Brazilian Legal Amazon region. The lower map shows hot spots for the period 2005-2012 where an increased presence of hotspots is observed in the so called "MATOPIBA" region at the eastern border of our study region.

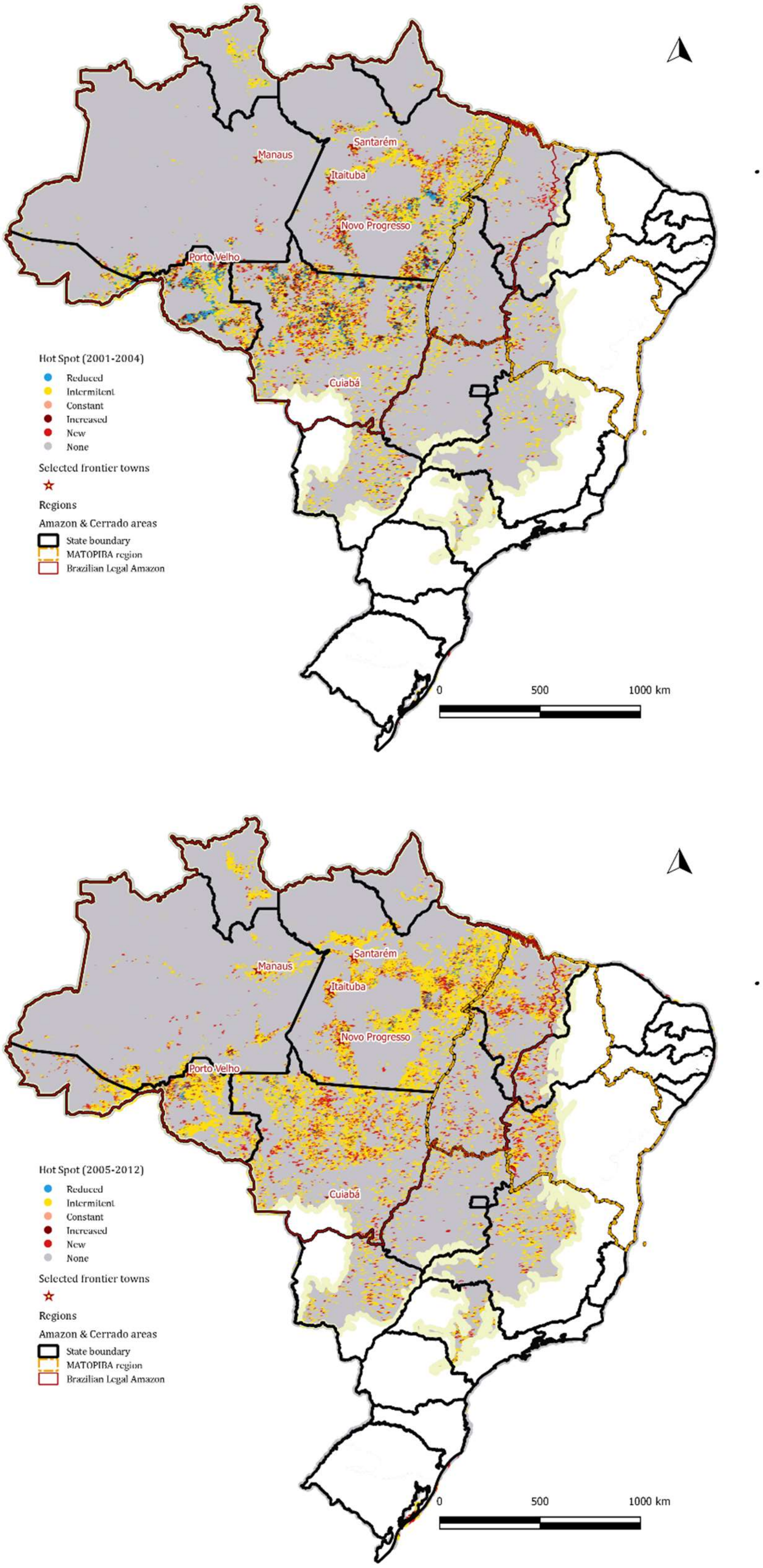


Figure A.1. 3. Identified hot spots of forest cover loss (2001-2004 & 2005-2012)
Note*:* Upper figure shows types of deforestation hot spots identified between 2001 and 2004. The lower figure shows changes in the period 2005 to 2012.

A.2. Appendix – Chapter 3

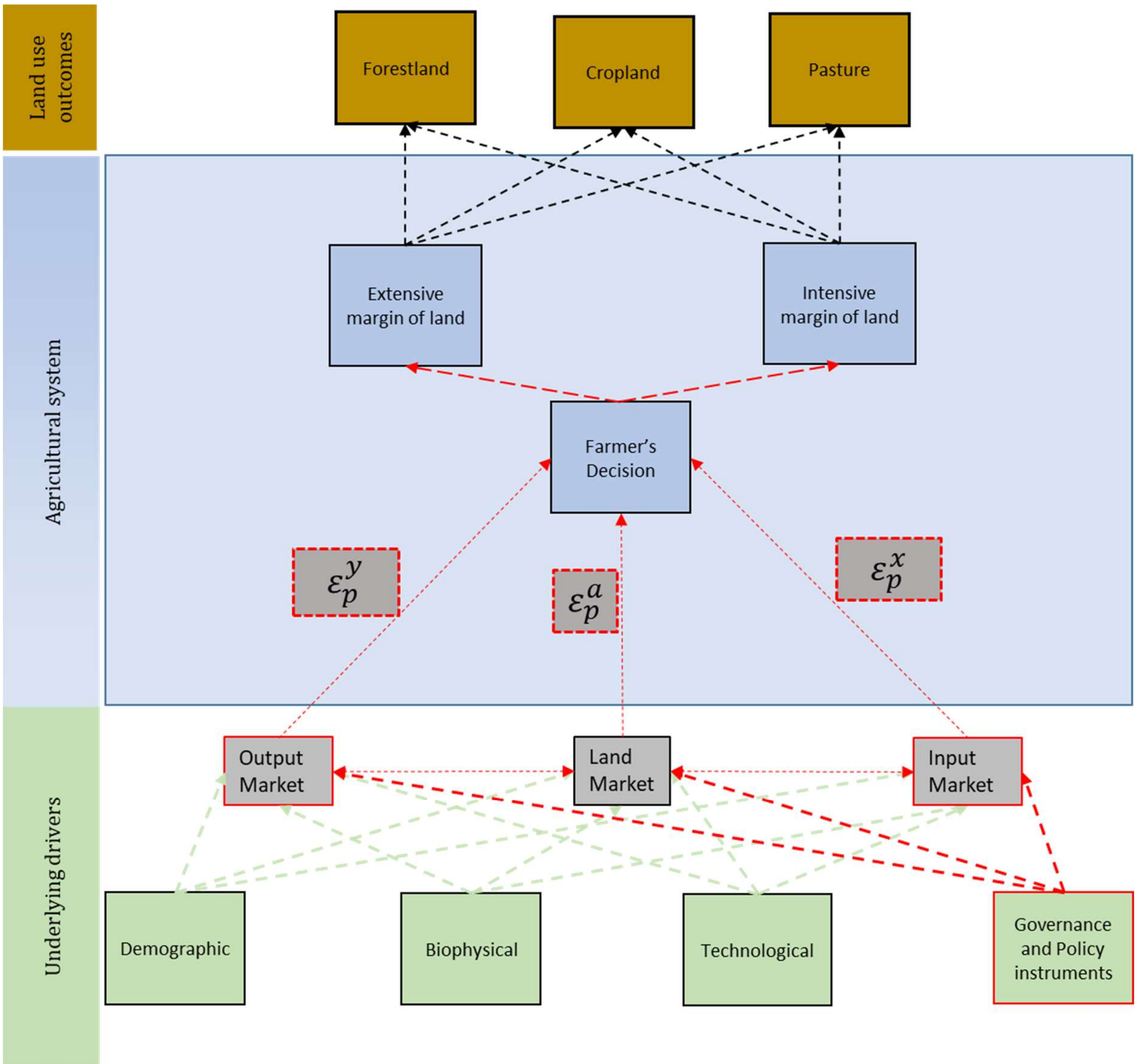


Figure A. 2. Mechanisms affecting land use decision analyzed in chapter 3

A.2.a. Additional empirical issues

To implement the empirical specification, we used the free-access R software for statistical analysis. The netput system of equations is implemented using the package MicEconSNQP (http://www.micecon.org/). It allows normalizing prices using a weighting approach suggested by Diewert and Wales (1992). This R package imposes symmetry restrictions, and homogeneity is achieved by definition of the SNQ profit function. Convexity in prices can be imposed following Koebel et al. (2003). In the empirical analysis, convexity was violated and therefore imposed. Related statistics were calculated using a jackknife bootstrap estimation.

The package *systemfit* was employed to estimate the land use system of equations (Henningsen and Hamann 2007). The package allows implementing a SUR model and imposing land constraint restrictions.

A.2.b. Conservation policy in the model

Table A.2.1 presents netput and land elasticities with respect to environmental fines. The estimated relationship points to a conservation policy-driven intensification effect. If the number of fines from one year to another increases, the results show a positive relation with crop and cattle production. To reinforce this effect, land elasticities are negative for cropland and pasture expansions, while positive for forestland, indicating successful preservation. These results are in agreement with previous claims that conservation does affect productivity, and even could enhance it (Merry and Soares-Filho 2017, Soares-Filho and Rajão 2018).

Table A.2. 1. Netput and land elasticities to environmental fines

		Environmental fines
Netputs	**Crops**	0.010
	Cattle	0.008
	VInputs	-0.029
Land Uses	**Cropland**	-0.005
	Pasture	-0.001
	Forestland	0.005

A.2.c. Parameters estimated

Table A.2. 2. Netput system estimations

Parameter	Estimate♦	Std. Err.	*t*-Value	Pr(>|t|)	Estimate‡ (Convex)
α_1	84.02326	202.0924	0.415767	0.677827	90.17077
α_2	79.4519	91.23815	0.870819	0.38443	67.89193
α_3	-274.261	296.948	-0.9236	0.356311	-273.289
α_{11}	65.86738	38.35265	1.717414	0.086759	84.88532
α_{12}	25.43643	23.2516	1.093965	0.274698	2.518062
α_{13}	-91.3038	44.68198	-2.04341	0.041735	-87.4034
α_{22}	-45.2454	27.67623	-1.63481	0.102958	0.074696
α_{23}	19.80899	23.95395	0.826961	0.408804	-2.59276
α_{33}	71.49482	59.67838	1.198002	0.2317	89.99614
δ_{11}	8.47505	9.201739	0.921027	0.35765	8.68983
δ_{12}	7.280931	9.39703	0.774812	0.438957	7.520474
δ_{13}	6.387271	9.036558	0.706826	0.48013	6.58411
δ_{21}	2.166458	4.179039	0.518411	0.604489	2.175511
δ_{22}	3.779341	4.268562	0.88539	0.376534	3.796386
δ_{23}	2.998459	4.104263	0.730572	0.465513	2.949564
δ_{31}	12.73067	12.33633	1.031965	0.302777	13.10213
δ_{32}	18.64295	12.5743	1.482624	0.139044	19.03109
δ_{33}	15.40872	12.11491	1.27188	0.204232	15.73042
μ_{11}	-23.6534	493.6753	-0.04791	0.961812	-32.6806
μ_{12}	11.85229	29.07874	0.407593	0.683813	11.76149
μ_{13}	-689.685	903.3948	-0.76344	0.4457	-709.828
μ_{14}	-0.60482	-5.48819	0.110205	0.912308	-0.41339
μ_{21}	71.02773	224.4932	0.316391	0.751888	75.60552

μ_{22}	5.005275	13.18393	0.37965	0.704428	5.261403
μ_{23}	-303.936	410.3541	-0.74067	0.459375	-303.507
μ_{24}	0.410686	-2.49053	-0.1649	0.869115	0.70918
μ_{31}	751.6367	675.5354	1.112653	0.266596	759.8774
μ_{32}	44.84252	39.88159	1.124392	0.261592	44.8462
μ_{33}	-1720.09	1209.304	-1.42238	0.155778	-1753.44
μ_{34}	-2.09938	-7.57324	0.27721	0.781777	-1.94971
β_{11}	0.088557	0.42467	0.208531	0.834932	0.074783
β_{12}	0.086013	0.419907	0.204838	0.837814	0.07206
β_{13}	0.084244	0.387779	0.217247	0.828139	0.071258
β_{22}	0.082645	0.416362	0.198494	0.84277	0.068469
β_{23}	0.089039	0.382817	0.23259	0.816211	0.076054
β_{33}	0.072252	0.361887	0.199652	0.841865	0.059663
φ_{11}	-21.676	31.0927	-0.69714	0.486163	-22.6582
φ_{12}	-1.62892	3.787727	-0.43005	0.667414	-1.67507
φ_{13}	-8.19357	40.29092	-0.20336	0.838968	-6.86744
φ_{14}	-0.7028	-0.6662	1.054941	0.292156	-0.67209
φ_{21}	-31.0208	31.01442	-1.00021	0.317879	-31.9825
φ_{22}	-1.41864	3.584187	-0.39581	0.692482	-1.46115
φ_{23}	-8.0918	39.80121	-0.20331	0.839011	-6.7558
φ_{24}	-0.62026	-0.67076	0.924714	0.355731	-0.59008
φ_{31}	-24.115	29.52796	-0.81668	0.414647	-24.8844
φ_{32}	-1.10302	3.489286	-0.31612	0.752097	-1.12641
φ_{33}	-7.76488	36.91505	-0.21034	0.833517	-6.51322
φ_{34}	-0.52665	-0.68423	0.769703	0.441978	-0.49508
ρ_{11}	-988.307	2264.666	-0.4364	0.662805	-978.88
ρ_{12}	-88.684	99.68894	-0.88961	0.374267	-89.8665

ρ_{13}	2812.736	2973.08	0.946068	0.344745	2900.841
ρ_{14}	10.41893	-13.25	-0.78634	0.432185	9.925131
ρ_{22}	-3.92398	5.943788	-0.66018	0.509557	-3.87916
ρ_{23}	126.773	346.58	0.365783	0.714741	129.9884
ρ_{24}	0.192481	-1.43346	-0.13428	0.893258	0.120544
ρ_{33}	756.3472	3817.396	0.198132	0.843053	629.2028
ρ_{34}	56.36105	-66.2344	-0.85093	0.395369	53.31083
ρ_{44}	0.020843	0.274883	0.075826	0.939599	0.029333

Note: These parameters represent estimations from the system represented by equation (3-13) in the chapter. ♦Represents estimation without imposing convexity in prices. ǂRepresents estimations after imposing convexity in prices as proposed by Koebel *et al.* (2003). Two outputs (1=crops, 2=cattle), variable inputs (3), and their respective prices are included. Three land uses are included (1=cropland, 2=pasture, 3=forestland), a residual category can be inferred through the restrictions. Three fixed characteristics plus total land endowments are also included in the estimation (1= soy aptitude, 2= accessibility, 3= land endowment, 4= change in environmental fines).

Table A.2. 3. Land use system estimations

Parameter	Estimate	Std. Err.	*t*-Value	Pr(>|t|)
γ_1	-16.935	331.7606	-0.05105	0.959308
γ_2	-125.794	318.4603	-0.39501	0.692996
γ_3	173.7288	127.0665	1.36723	0.172131
γ_4	-30.9995	35.2779	-0.87872	0.37995
γ_{11}	1.637462	0.893518	1.8326	0.067421
γ_{21}	-4.72669	6.389666	-0.73974	0.459784
γ_{31}	-1.13325	0.916966	-1.23587	0.217053
γ_{12}	-0.83213	0.857697	-0.9702	0.33239
γ_{22}	7.950844	6.133504	1.2963	0.195435
γ_{23}	1.295266	0.880204	1.47155	0.141733
γ_{13}	-0.9232	0.342223	-2.69765	0.007205
γ_{23}	-3.40177	2.447283	-1.39002	0.165104
γ_{33}	-0.29045	0.351204	-0.82702	0.408597
γ_{14}	0.117869	0.095013	1.24057	0.21531
γ_{24}	0.177617	0.679448	0.26141	0.79387
γ_{34}	0.128434	0.097506	1.31719	0.18834
ω_{11}	201.9702	89.88923	2.24688	0.025057
ω_{12}	-2.13883	4.088997	-0.52307	0.601144
ω_{13}	14.08176	2.813992	5.00419	7.64×10^{-7}
ω_{14}	0.466068	1.096599	0.42501	0.670999
ω_{21}	-286.661	86.28555	-3.32224	0.000954
ω_{22}	-10.2795	3.925069	-2.61894	0.009072
ω_{23}	2.472954	2.701178	0.91551	0.360339
ω_{24}	0.181274	1.052636	0.17221	0.863338
ω_{31}	82.94741	34.42815	2.40929	0.016322

ω_{32}	12.65002	1.566112	8.07734	4.44 x10^{-15}
ω_{33}	4.470279	1.077777	4.14768	3.91 x10^{-05}
ω_{34}	-0.63444	0.420004	-1.51054	0.131499
ω_{41}	1.743307	9.558407	0.18238	0.85535
ω_{42}	-0.23167	0.434805	-0.5328	0.59439
ω_{43}	-20.025	0.299227	-66.9224	< 2 x10^{-16}
ω_{44}	-0.01291	0.116607	-0.11069	0.9119

Note: These parameters represent estimations from the system represented by equation (3-15) in the chapter. Land uses considered are 1=cropland, 2=pasture, 3=forestland, and 4= other land uses.

A.2.d. Supporting maps

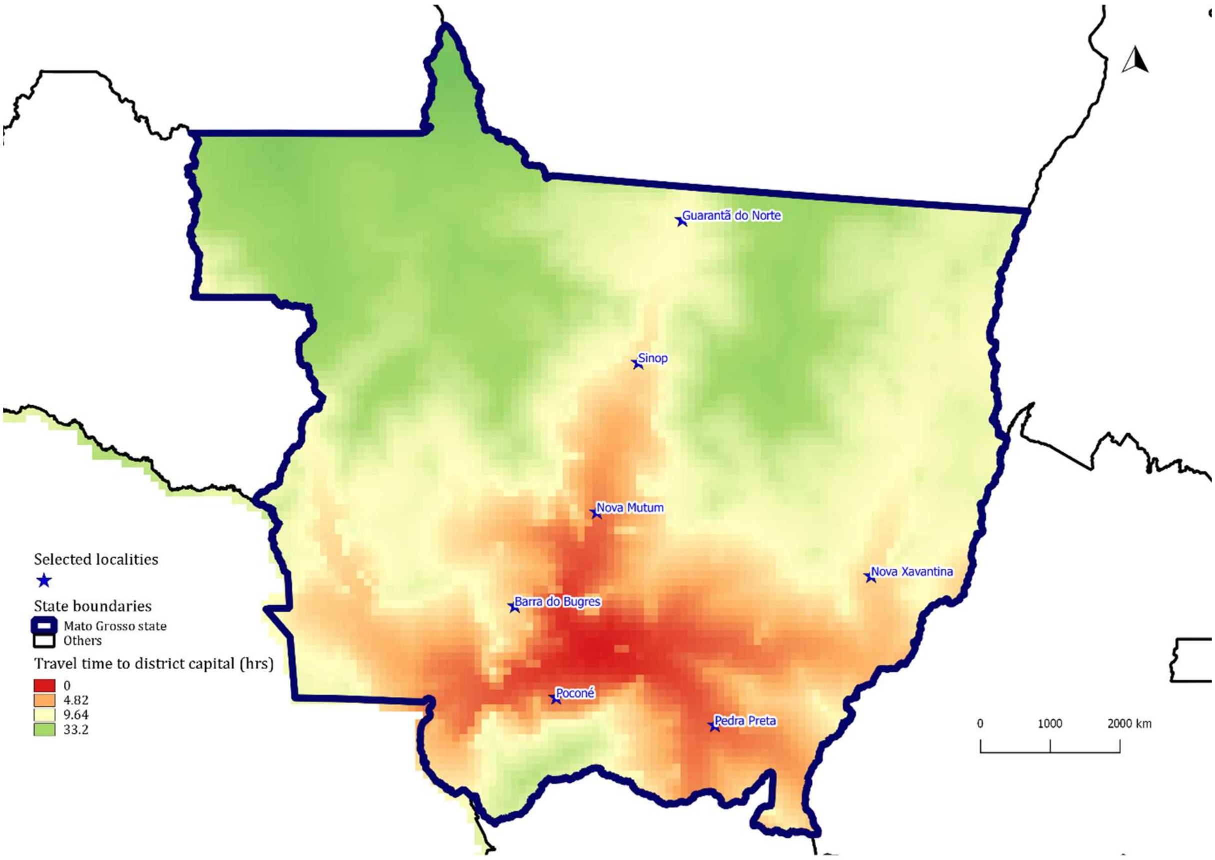


Figure A.2. 1. Travel time to district capitals in Mato Grosso
Note: This map shows resulting calculations from an accessibility model considering a friction map calculated for 2006 and as sources all district capitals in the country. The units for travel time are expressed in hours.

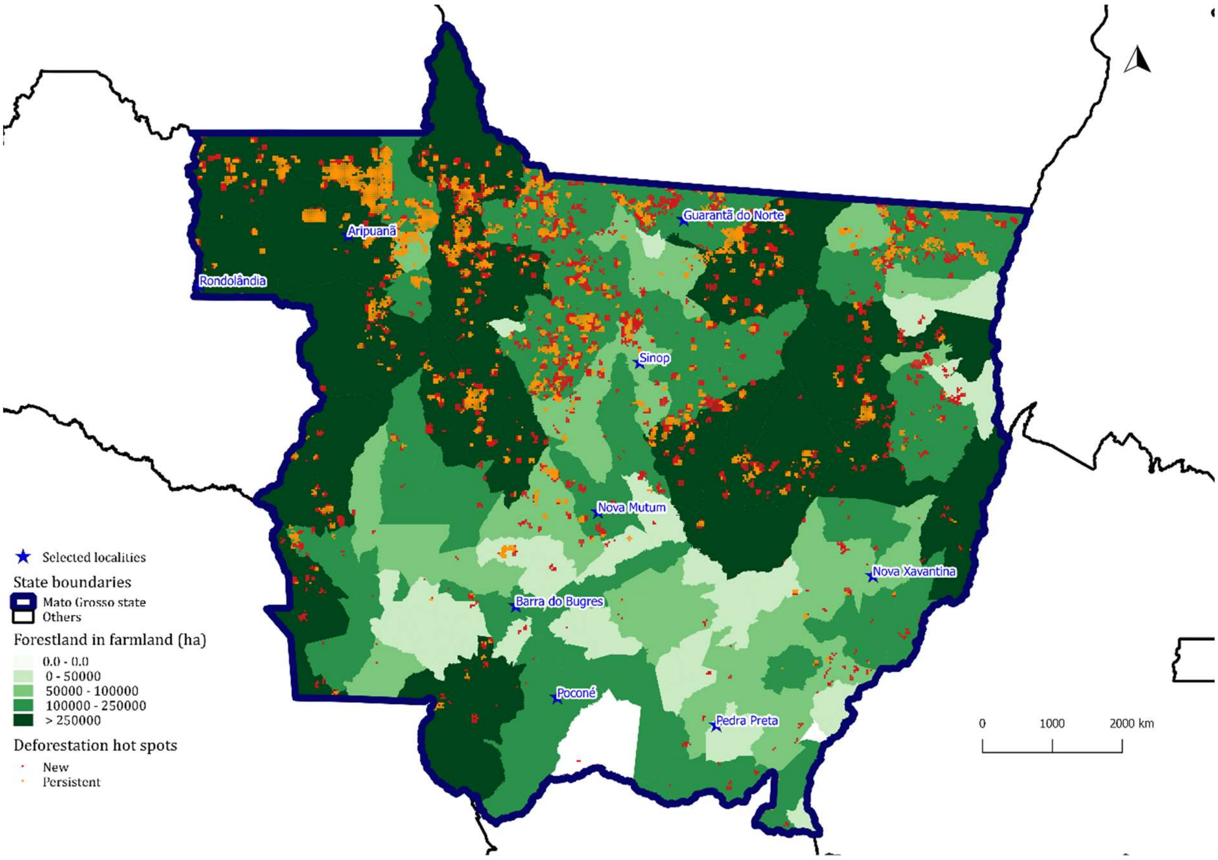


Figure A.2. 2. Farms' forest and deforestation hot spots in Mato Grosso
Note: This map shows amount of forest in total farmland area within a district and deforestation hot spots in Mato Grosso for the year 2006.

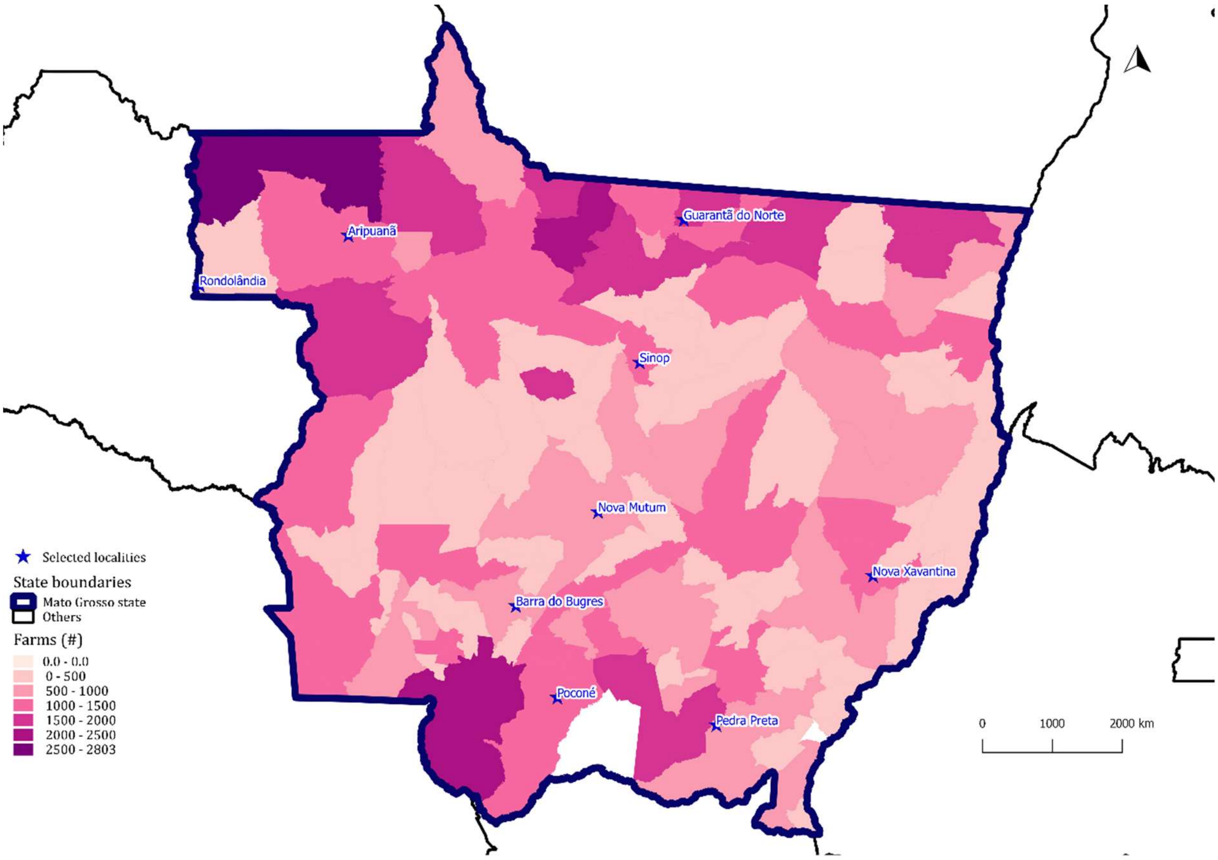
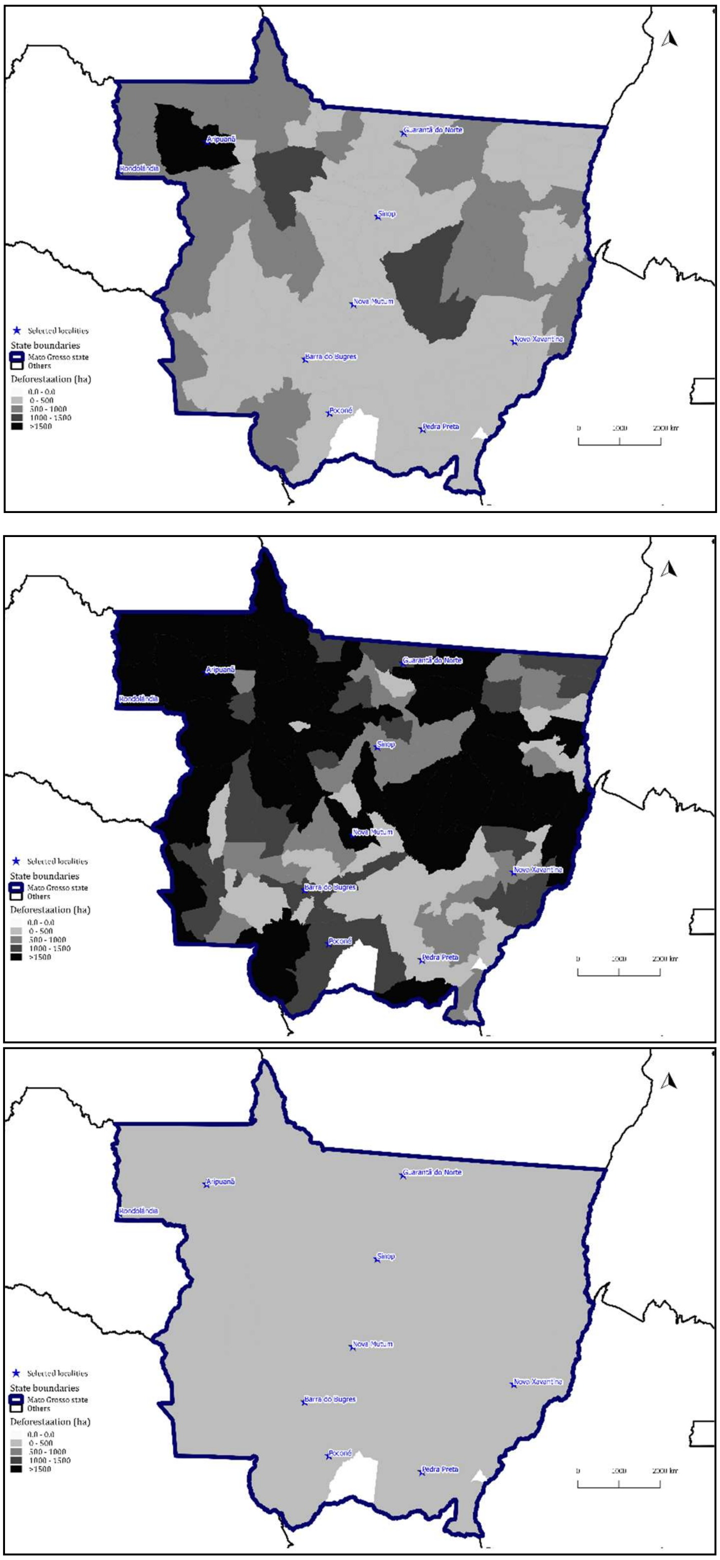


Figure A.2. 3 Total farms per district in Mato Grosso



A)

B)

Figure A.2. 4 Expected deforestation in total farmland in Mato Grosso
Note: The maps depict expected aggregated deforestation on total farmland within each district due to an increase in 1% in crop (A), cattle (B), and inputs(C) prices in Mato Grosso. This figure takes the effect of an average farm within a district (see Figure 3 in the chapter) and multiplies it by the total number of farmers in a district.

A.3. Appendix – Chapter 4

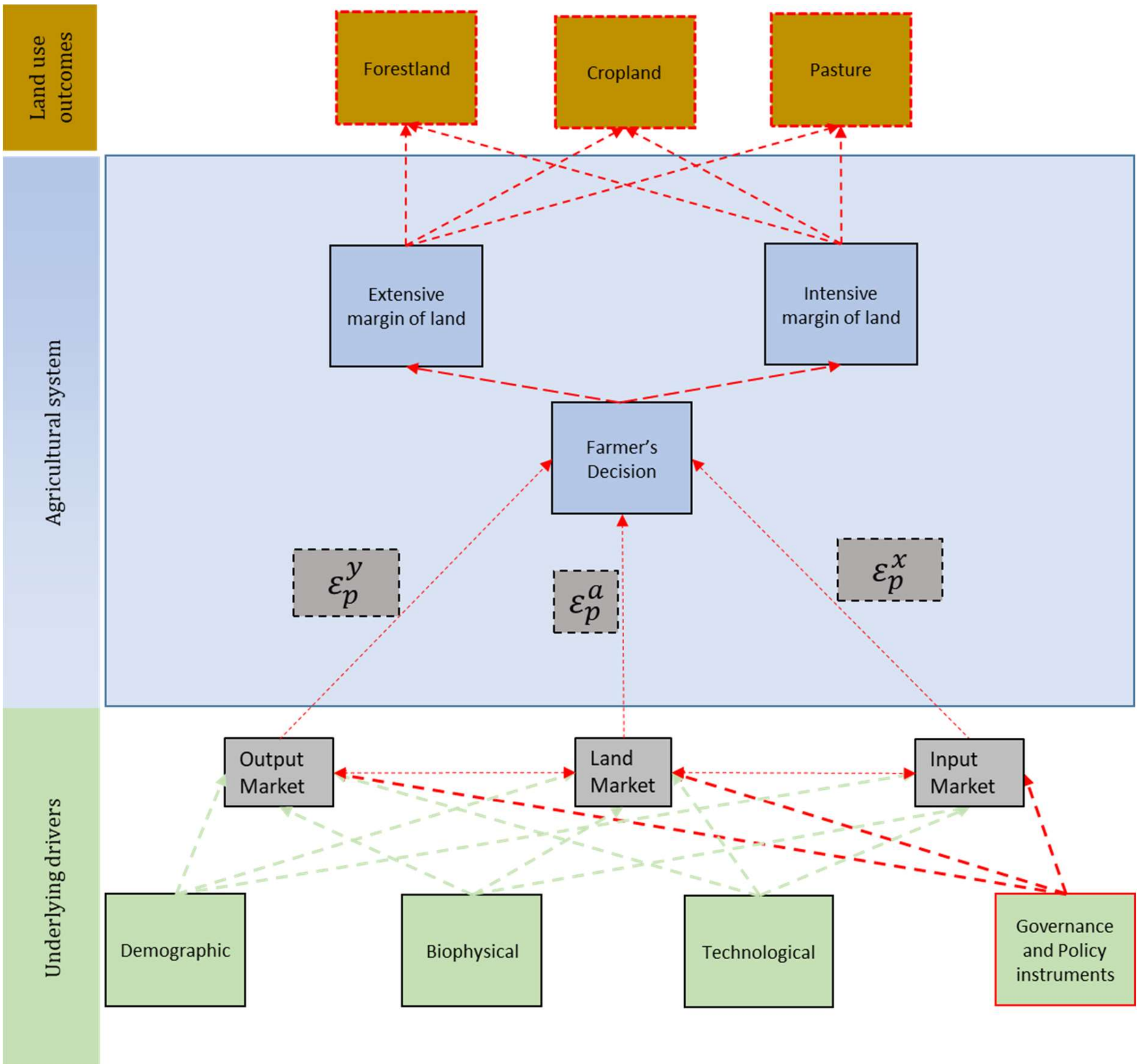


Figure A. 3. Mechanisms affecting land use decision analyzed in chapter 4

A.3.a. Summary statistics of our sample at 100 km from the biome border

Table A.3. 1. Summary of variables and sources used

Variable (units)	Source	Obs	Mean	St. Dev.	Min	Max
Deforestation (ha)	Own calculation; MapBiomas (https://mapbiomas.org/); SICAR (http://www.car.gov.br/publico/imoveis/ index); Imaflora *et al.* (2019) ().	1,864,469	0.131	0.685	0.000	91.369
F2P (ha)	Same as deforestation.	1,864,469	0.102	0.566	0.000	90.392
F2C (ha)	Same as deforestation.	1,864,469	0.009	0.156	0	31
P2C (ha)	Same as deforestation.	1,864,469	0.035	0.461	0	239
Dist. to Border - Amazon(km)	Same as deforestation + IBGE(http://sidra.ibge.gov.br/).	889,420	44.561	28.490	0.011	99.997
Dist. to Border - Cerrado(km)	Same as deforestation + IBGE(http://sidra.ibge.gov.br/).	975,049	-44.224	28.245	-100.000	-0.137
Assignment	Same as deforestation + IBGE(http://sidra.ibge.gov.br/).	1,864,469	0.477	0.499	0	1
Forest in t-1 (ha)	Same as deforestation.	1,864,469	198.060	895.440	0.090	70,369.710
Land endowent (ha)	Same as deforestation.	1,864,469	395.898	1,471.449	0.065	146,909.800
Fine intensity index	Own calculation; IBAMA (http://www.ibama.gov.br).	1,864,469	86.363	154.893	-0.000	2,640.169
Access to markets (hrs)	Own calculation; DNIT; Hansen *et al* (2013b), Miranda *et al* (2019).	1,864,469	1.995	2.593	0	28

A.3.b. Compliance and forest stocks

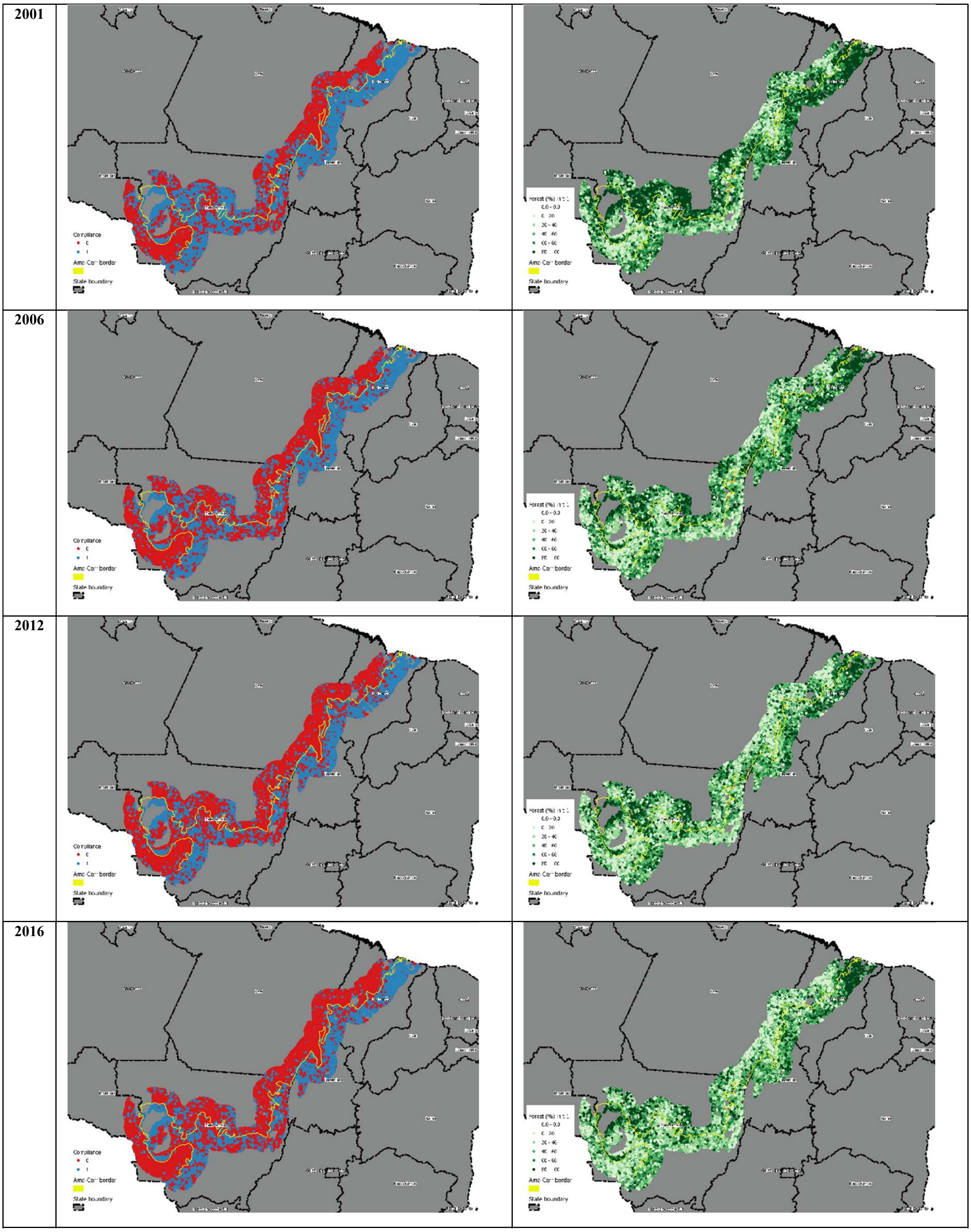


Figure A.3. 1. Farmers' compliance and their stock of forestland at the Amazon-Cerrado border.

Note: Figures depict centroids of our sample of farmers at 100 km distance. On the left, we depicted their compliance based on the percentage of forest to total land endowment that is observed. We consider as compliant those farmers in the Amazon with 80% or more of their total land as forest in a previous year, and with 35% or more for those in the Cerrado. In red are colored those centroids with less share of forest as per mandate of the Forest Code before 2012. The figures in the right column show centroids and their percentage of forest to total land endowment. We show these figures for the years 2001, 2006, 2012, and 2016.

A.3.c. Falsification tests

Figure A.3.2 shows a correlation matrix of the different predetermined covariates employed in our falsification tests (see below) and our RD estimations from Chapter 4. The variables employed are geographic access to markets, total land endowment, the intensity of environmental fines in the area, and the total forest area in $t-1$. Correlation coefficients determine the color intensity and size of the circles. As expected we observe a strong correlation between forest and total land endowments. Correlation between the other variables is rather small or non-existent. Based on these correlations, our RD estimations only use as predetermined covariates the forest endowment, accessibility, and fine intensity. The accessibility variable was obtained from Miranda *et al* (2019). The fine intensity index was calculated using spatial points of information from fines issued by IBAMA. We use these points to calculate a kernel density distribution using a distance of 3 km from each point as the bandwidth of reference. We did this for the years 2000 till 2017. For each year t, we sum the number of fines in the previous two years and use this to create our final index which was our final predetermined covariate on fines.

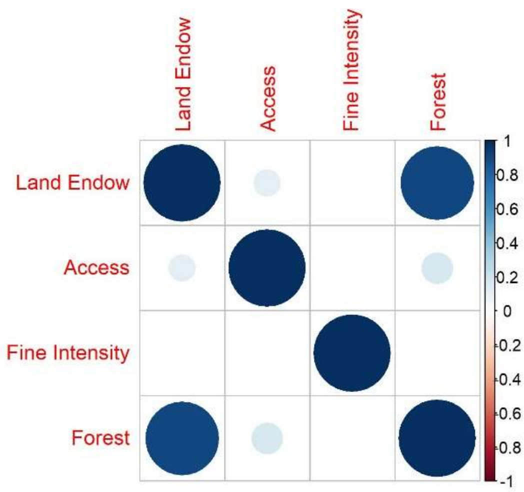


Figure A.3. 2. Correlation matrix of predetermined covariates.
Note: Color intensity and size of circles is determined by the correlation coefficients.

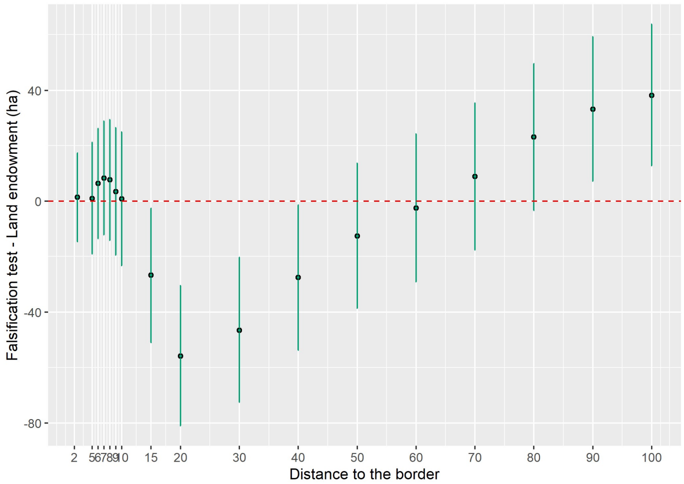


Figure A.3. 3. Falsification test – Land endowment.

Note: The points represent the RD estimations at different distances to the biome border. Confidence intervals at 95% level are depicted by the green lines.

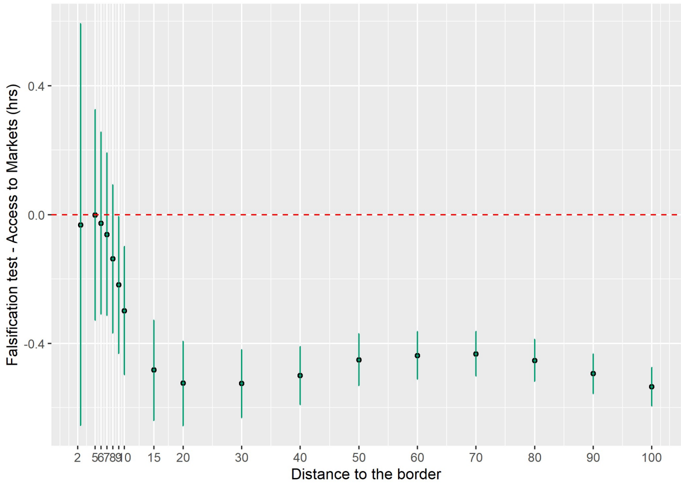
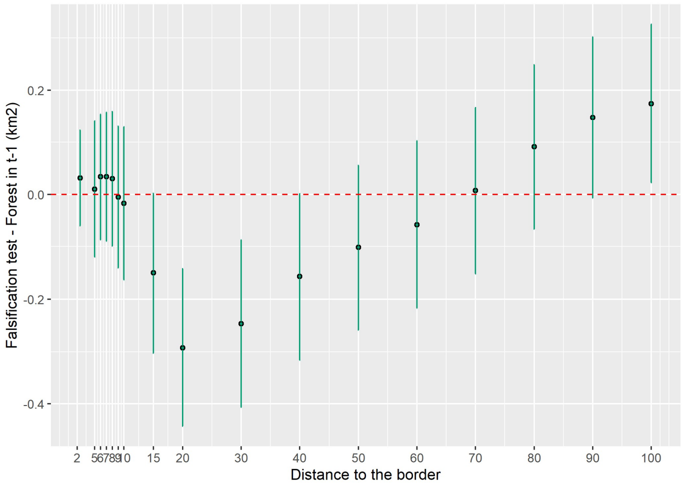


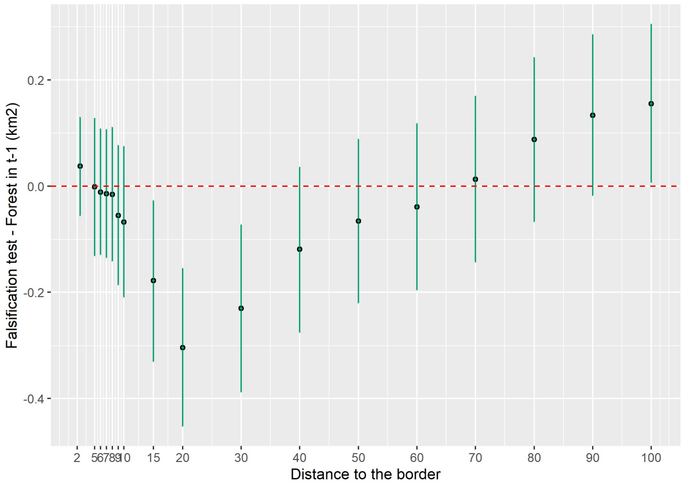
Figure A.3. 4. Falsification test – Accessibility to markets.

Note: The points represent the RD estimations at different distances to the biome border. Confidence intervals at 95% level are depicted by the green lines.

A)



B)



C)

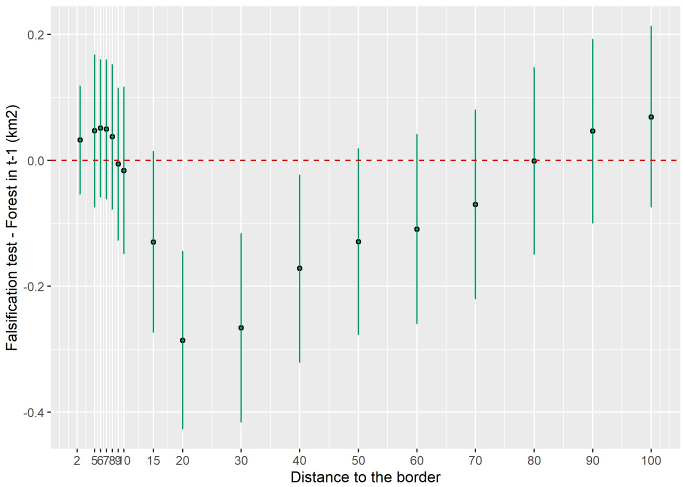


Figure A.3. 5. Falsification test – Forest endowment in t-1.

Note: The points represent the RD estimations at different distances to the biome border. Confidence intervals at 95% level are depicted by the green lines. From top to bottom, the figures depict the years 2004, 2008, and 2012.

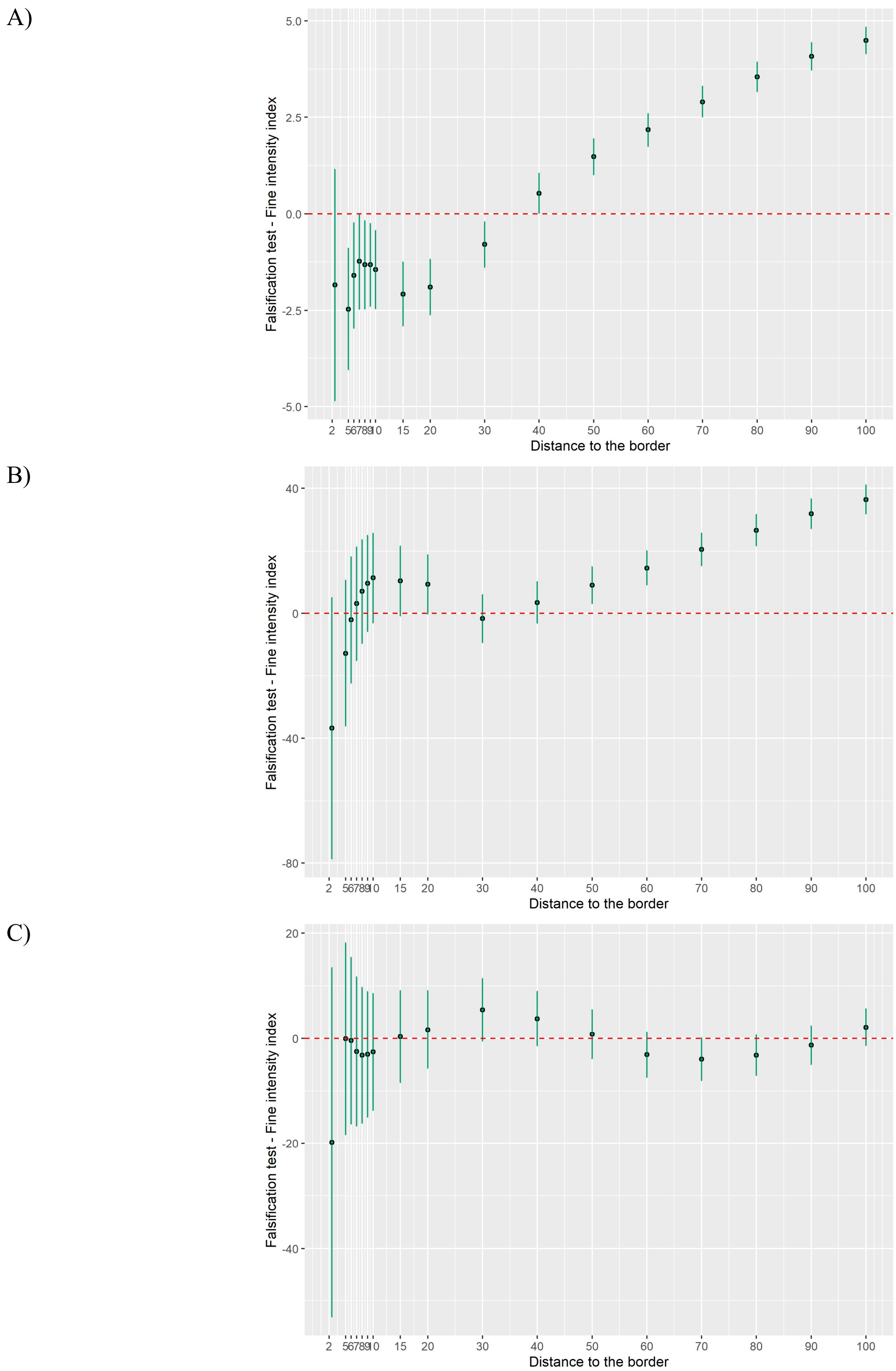
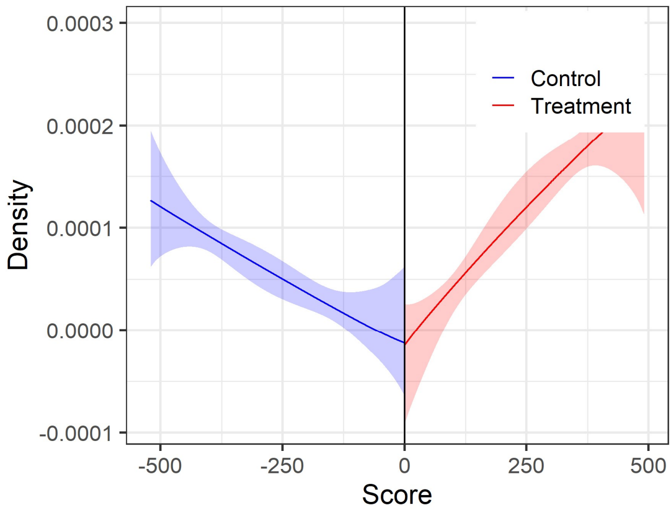


Figure A.3. 6. Falsification test – Fine intensity index.
Note: The points represent the RD estimations at different distances to the biome border. Confidence intervals at 95% level are depicted by the green lines. From top to bottom, the figures depict the years 2004, 2008, and 2012.

A)



B)

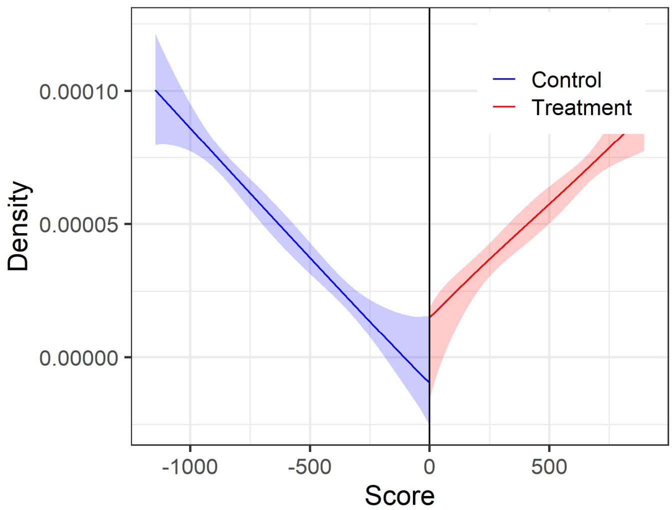


Figure A.3. 7. Score variable density distribution test.
Note: Panel A) is at 2km and panel B) at 5 km.

A.3.d. Supporting graphs

Sources of deforestation

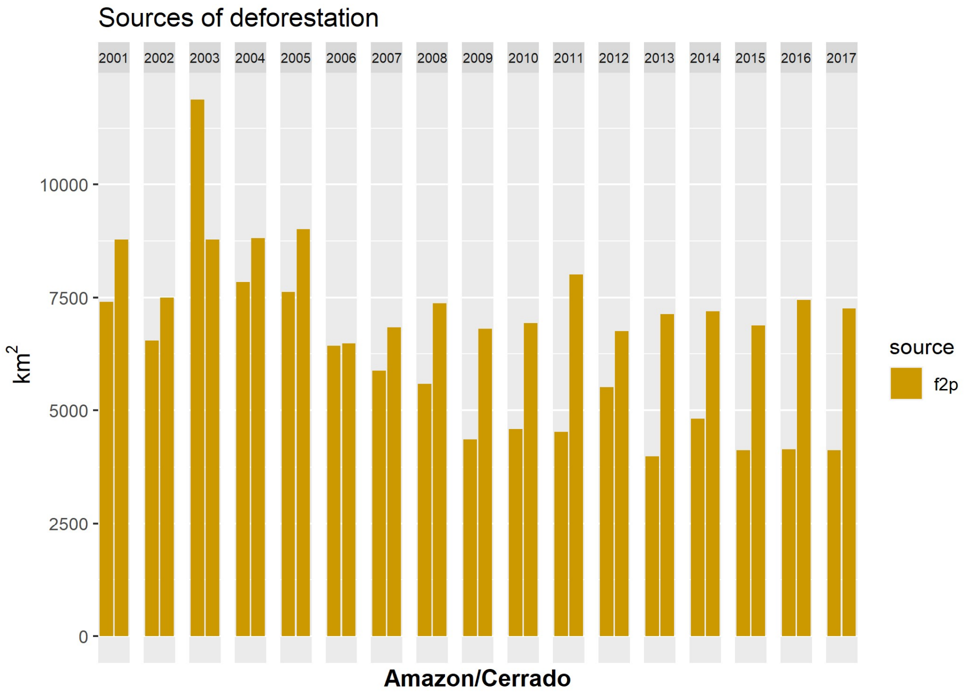


Figure A.3. 8. Forest to pasture conversion at the Amazon-Cerrado biome border
Note: Bars represent the amount of forest to pasture (F2P) conversion in farms identified at 100 km distance from the biome border.

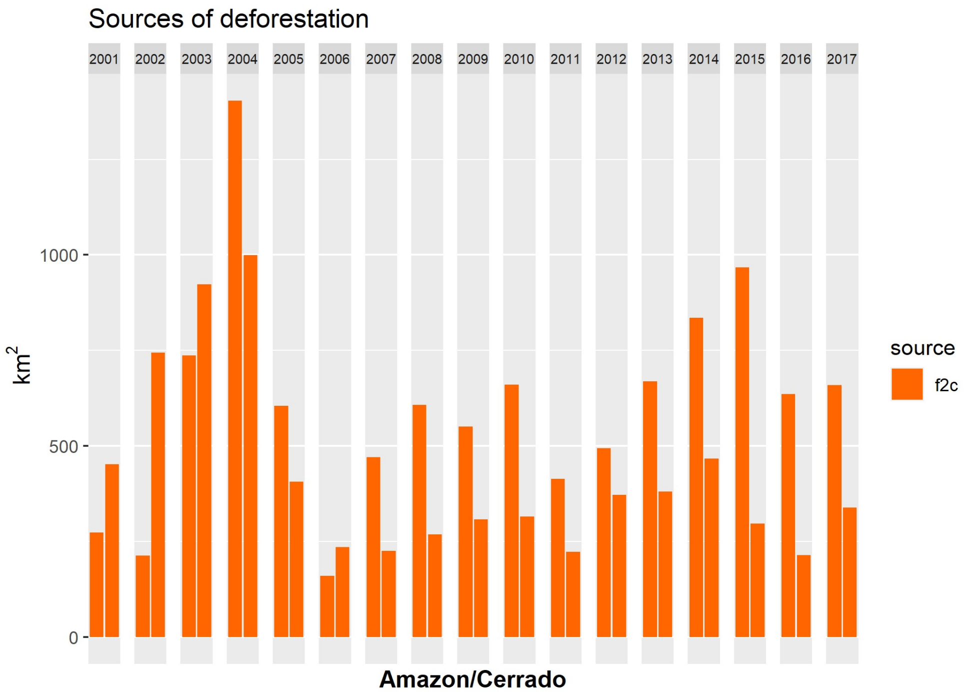


Figure A.3. 9. Forest to cropland conversion at the Amazon-Cerrado biome border
Note: Bars represent the amount of forest to cropland (F2C) conversion in farms identified at 100 km distance from the biome border.

A.3.e. Supporting tables

Table A.3. 2. RD estimations – Deforestation

Year	Amazon	Cerrado	Coeff	p-Value	CILower	CIUpper
2001	2.938	2.807	0.131	0.876	-1.513	1.775
2002	1.824	3.059	**-1.235**	**0.064**	-2.540	0.070
2003	1.671	1.590	0.082	0.941	-2.099	2.262
2004	0.558	1.448	-0.890	0.288	-2.534	0.753
2005	1.452	1.795	-0.343	0.662	-1.880	1.195
2006	2.777	4.796	***-2.019***	***0.003***	-3.361	-0.677
2007	1.901	3.601	***-1.700***	***0.044***	-3.351	-0.048
2008	2.960	4.001	-1.041	0.146	-2.444	0.361
2009	2.587	3.667	***-1.080***	***0.059***	-2.201	0.041
2010	3.249	3.178	0.071	0.922	-1.364	1.506
2011	1.380	2.746	**-1.366**	**0.096**	-2.974	0.242
2012	1.951	2.158	-0.208	0.775	-1.634	1.218
2013	2.344	2.649	-0.306	0.563	-1.343	0.731
2014	3.334	4.684	**-1.350**	**0.097**	-2.946	0.247
2015	1.663	2.477	-0.815	0.204	-2.072	0.442
2016	1.689	1.475	0.214	0.755	-1.129	1.557
2017	3.063	4.229	-1.167	0.107	-2.587	0.254

Note: The second and third columns show the estimations from our polynomial function for Amazon and Cerrado farmers, respectively. The fourth column shows the coefficients of interest and the fifth column its p-Value. The last two columns show the confidence intervals. Numbers in bold and italics are significant at a 95% confidence level, and those only bold are significant at a 90% level.

Table A.3. 3. RD estimations – Forest to Pasture (F2P)

Year	Amazon	Cerrado	Coeff	p-Value	CILower	CIUpper
2001	1.422	2.017	-0.595	0.364	-1.880	0.690
2002	1.484	2.380	**-0.896**	**0.086**	-1.917	0.126
2003	1.637	3.035	-1.398	0.101	-3.069	0.273
2004	1.142	2.020	-0.878	0.141	-2.047	0.290
2005	1.177	1.612	-0.436	0.507	-1.723	0.852
2006	1.967	3.965	***-1.998***	***0.001***	-3.198	-0.799
2007	1.369	2.876	***-1.507***	***0.027***	-2.846	-0.168
2008	2.031	3.449	***-1.418***	***0.015***	-2.564	-0.271
2009	2.206	3.554	***-1.349***	***0.003***	-2.225	-0.472
2010	2.382	2.734	-0.353	0.445	-1.257	0.551
2011	1.499	2.626	***-1.127***	***0.078***	-2.382	0.128
2012	2.343	3.328	***-0.985***	***0.051***	-1.973	0.002
2013	2.216	2.586	-0.370	0.388	-1.210	0.470
2014	2.115	3.606	***-1.490***	***0.004***	-2.496	-0.485
2015	1.557	2.246	-0.689	0.186	-1.710	0.331
2016	1.541	1.874	-0.333	0.571	-1.485	0.819
2017	2.102	3.407	***-1.305***	***0.019***	-2.391	-0.218

Note: The second and third columns show the estimations from our polynomial function for Amazon and Cerrado farmers, respectively. The fourth column shows the coefficients of interest and the fifth column its p-Value. The last two columns show the confidence intervals. Numbers in bold and italics are significant at a 95% confidence level, and those only bold are significant at a 90% level.

Table A.3. 4. RD estimations – Forest to Cropland (F2C)

Year	Amazon	Cerrado	Coeff	p-Value	CILower	CIUpper
2001	-0.091	-0.014	-0.077	0.585	-0.352	0.199
2002	-0.249	-0.111	-0.138	0.568	-0.613	0.336
2003	-0.921	-2.258	***1.337***	***0.031***	0.126	2.548
2004	-1.374	-1.339	-0.035	0.942	-0.990	0.920
2005	-0.571	-0.528	-0.043	0.845	-0.476	0.390
2006	-0.056	-0.064	0.008	0.851	-0.078	0.094
2007	-0.323	-0.189	-0.135	0.543	-0.569	0.299
2008	-0.186	0.001	-0.187	0.210	-0.479	0.105
2009	-0.479	-0.678	0.199	0.381	-0.247	0.646
2010	-0.581	-0.324	-0.257	0.192	-0.644	0.129
2011	-0.366	-0.236	-0.131	0.359	-0.410	0.149
2012	-1.113	-1.584	0.471	0.230	-0.298	1.240
2013	-0.672	-0.288	***-0.384***	***0.050***	-0.769	0.001
2014	-0.785	-0.619	-0.166	0.476	-0.623	0.290
2015	-0.643	-0.544	-0.099	0.615	-0.484	0.286
2016	-0.710	-0.727	0.017	0.893	-0.234	0.269
2017	-0.228	-0.360	0.132	0.529	-0.279	0.543

Note: The second and third columns show the estimations from our polynomial function for Amazon and Cerrado farmers, respectively. The fourth column shows the coefficients of interest and the fifth column its p-Value. The last two columns show the confidence intervals. Numbers in bold and italics are significant at a 95% confidence level, and those only bold are significant at a 90% level.

Table A.3. 5. RD estimations – Pasture to Cropland (P2C)

Year	Amazon	Cerrado	Coeff	p-Value	CILower	CIUpper
2001	-0.133	-0.075	-0.058	0.749	-0.414	0.298
2002	-0.893	-0.963	0.070	0.881	-0.846	0.986
2003	-1.095	-1.028	-0.067	0.852	-0.770	0.636
2004	-2.197	-1.095	**-1.102**	**0.076**	-2.318	0.114
2005	-0.885	-0.807	-0.078	0.815	-0.728	0.573
2006	-0.491	-0.082	***-0.409***	***0.040***	-0.799	-0.019
2007	-0.798	-0.421	-0.377	0.244	-1.012	0.258
2008	-0.016	0.824	**-0.840**	**0.089**	-1.807	0.127
2009	-0.670	-0.504	-0.166	0.608	-0.799	0.468
2010	-0.355	-0.001	-0.354	0.194	-0.887	0.180
2011	-0.401	-0.257	-0.144	0.672	-0.813	0.524
2012	-1.259	-0.931	-0.328	0.463	-1.204	0.548
2013	-1.192	-0.967	-0.225	0.648	-1.191	0.741
2014	-0.136	-0.033	-0.103	0.866	-1.304	1.097
2015	-0.479	-0.474	-0.005	0.987	-0.631	0.621
2016	-0.449	-0.612	0.163	0.374	-0.196	0.523
2017	-0.574	-0.493	-0.081	0.681	-0.468	0.306

Note: The second and third columns show the estimations from our polynomial function for Amazon and Cerrado farmers, respectively. The fourth column shows the coefficients of interest and the fifth column its p-Value. The last two columns show the confidence intervals. Numbers in bold and italics are significant at a 95% confidence level, and those only bold are significant at a 90% level.

A.4. Appendix – R packages used

corrplot	https://www.rdocumentation.org/packages/corrplot
extractexact:	https://www.rdocumentation.org/packages/exactextractr
dplyr	https://www.rdocumentation.org/packages/dplyr
ggplot2:	https://www.rdocumentation.org/packages/ggplot2
lwgeom:	https://www.rdocumentation.org/packages/lwgeom
plm:	https://www.rdocumentation.org/packages/plm
plyr:	https://www.rdocumentation.org/packages/plyr
psych:	https://www.rdocumentation.org/packages/psych
raster:	https://www.rdocumentation.org/packages/raster
rdrobust:	https://www.rdocumentation.org/packages/rdrobust
rddensity	https://www.rdocumentation.org/packages/rddensity
reshape	https://www.rdocumentation.org/packages/reshape
rgdal:	https://www.rdocumentation.org/packages/rgdal
rgeos:	https://www.rdocumentation.org/packages/rgeos
sf:	https://www.rdocumentation.org/packages/sf
stargazer:	https://www.rdocumentation.org/packages/stargazer
systemfit:	https://www.rdocumentation.org/packages/systemfit
tictoc:	https://www.rdocumentation.org/packages/tictoc
xtable:	https://www.rdocumentation.org/packages/xtable

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