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Beyond Deforestation Reductions

Co-Benefits of Conservation Policies in the Brazilian Amazon

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List of Abbreviations and Acronyms

2SLS	two-stage least squares	58
ATT	average treatment effect on the treated	38
BCB	Brazilian Central Bank (*Banco Central do Brasil*)	57
BLA	Brazilian Legal Amazon	2
CAR	Rural Environmental Registry (*Cadastro Ambiental Rural*)	7
CNAE 1	National Classification of Economic Activities Version 1 (*Classificação Nacional de Atividades Econômicas*)	56
CO_2	carbon dioxide	4
CPR	common-pool resources	11
DATASUS	Department of Information and Informatics of SUS (*Departamento de Informação e Informática do SUS*)	89
DETER	Real-Time Deforestation Detection System (*Sistema de Detecção do Desmatamento em Tempo Real*)	7
EU	European Union	12
EUDR	EU Deforestation Regulation	12
e.g.	for example	91
FAO	Food and Agricultural Organization	8
FC	Forest Code (*Código Florestal*)	7
FTE	full-time equivalent	63
GDP	gross domestic product	93
GFW	Global Forest Watch	1
GSC	generalized synthetic control	42
IBAMA	Brazilian Institute of Environment and Renewable Natural Resources (*Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais Renováveis*)	7

IBGE Brazilian Institute of Geography and Statistics (*Instituto Brasileiro de Geografia e Estatística*) 89

ICD-10 International Classification of Diseases Version 10 85

i.e. that is / in other words 92

IFE interactive fixed effects 38

IHS inverse hyperbolic sine 58

INPE National Institute for Space Research (*Instituto Nacional de Pesquisas Espaciais*) 56

IPBES Intergovernmental Platform on Biodiversity and Ecosystem Services 3

IPCC Intergovernmental Panel on Climate Change 4

IV instrumental variable 66

km kilometers 93

km^2 square kilometers 62

LATE local average treatment effect 62

m^2 square meters 89

MMA Brazilian Ministry of the Environment (*Ministério do Meio Ambiente*) 6

MODIS Moderate Resolution Imaging Spectroradiometer 56

$\mu g/m^3$ micrograms per cubic meter 93

NGOs non-governmental organizations 32

PAM Municipal Agricultural Production (*Produção Agrícola Municipal*) 64

PES payment for ecosystem services 12

PL Priority List 7

$PM_{2.5}$ fine particulate matter 93

PMV Green Municipality Program (*Programa Municípios Verdes*) 33

PPCDAm Action Plan for the Prevention and Control of Deforestation in the Legal Amazon (*Plano de Ação para a Prevenção e Controle do Desmatamento na Amazônia Legal*) 5

PPM Municipal Livestock Production (*Pesquisa da Pecuária Municipal*) 64

PRODES Satellite-Based Monitoring of Deforestation in the Brazilian Amazon (*Monitoramento do Desmatamento da Floresta Amazônica Brasileira por Satélite*) 57

Abstract

The worldwide trend in tropical forest loss has been increasing overall since the 2000s. This imposes real social costs, such as through higher greenhouse gas emissions, diminished biodiversity, and harm to livelihoods and health. Market failures drive deforestation beyond the socially optimal level, as private land-use decisions do not internalize the full utility of intact forests or the external costs from clearing. To correct these failures, policymakers have three main mechanisms: (i) raising the private costs of clearing, (ii) increasing the private utility of keeping forests, and (iii) reducing the utility of alternative land uses. Policies that influence these mechanisms may generate additional outcomes beyond avoided deforestation, which have often been overlooked and remain insufficiently studied. This dissertation analyzes the positive secondary effects of conservation policies, the co-benefits, in the Brazilian Amazon to provide a more comprehensive assessment of policy performance and impacts.

The empirical strategy relies on counterfactual impact evaluation, combining high-resolution satellite data on land cover, fires, air pollution, and climate with administrative and socioeconomic records on agricultural production, trade flows, environmental enforcement, and health. Identification draws on quasi-experimental methods, including the generalized synthetic control method for staggered policy rollout, instrumental variables that exploit exogenous variation in satellite-based monitoring effectiveness, and an approach based on differences in spatial discontinuities.

The results show that policy instruments that increase environmental scrutiny—through public disclosure of high-deforestation municipalities, strengthened monitoring and enforcement, and restricted market and credit access for non-compliant production—reduce deforestation and forest-to-pasture conversion. At the same time, the expansion of profitable soybean cultivation shifts towards already-cleared areas, consistent with land-saving intensification rather than at the expense of forests. Local labor markets, wages, and credit availability adjust accordingly to this sustainable transformation, which is associated with productivity gains. Traders mitigate risks along the supply chain by exporting towards destinations less sensitive to imported-deforestation concerns. Conservation policies also deliver substantial health co-benefits: reductions in clearings and burnings lower fine particulate air pollution and avoid associated hospitalizations and deaths.

Taken together, the evidence indicates that well-designed conservation policies can curb deforestation while generating multiple co-benefits. At the same time, both co-benefits and trade-offs are context dependent and likely heterogeneous. Recognizing and explicitly incorporating this heterogeneity into policy design and evaluation can improve the targeting and effectiveness of interventions, help protect vulnerable groups, and ensure a more equitable distribution of gains. Clearly communicating co-benefits and trade-offs could also help strengthen public, economic and political support for forest conservation.

Kurzfassung

Der weltweite Verlust tropischer Wälder hat seit den 2000er-Jahren insgesamt zugenommen und verursacht erhebliche gesellschaftliche Kosten. Dazu gehören zum Beispiel höhere Treibhausgasemissionen, ein Rückgang der Biodiversität sowie Beeinträchtigungen von Lebensgrundlagen und Gesundheit. Marktversagen führt zu übermäßiger Entwaldung, da private Landnutzungsentscheidungen weder den vollen Nutzen intakter Wälder noch die von Waldrodungen ausgehenden externen Kosten berücksichtigen. Um dieses Marktversagen zu korrigieren, verfügen politische Entscheidungsträgerinnen und -träger über drei zentrale Mechanismen: (i) die privaten Kosten von Waldrodung erhöhen, (ii) den privaten Nutzen bestehender Wälder steigern und (iii) den Nutzen alternativer, nicht-waldlicher Landnutzungen senken. Politiken, die diese Mechanismen beeinflussen, können neben vermiedener Entwaldung zusätzliche Wirkungen entfalten, welche bislang häufig übersehen und wenig untersucht wurden. Diese Dissertation analysiert die positiven Begleiteffekte ("*Co-Benefits*") von Naturschutzpolitiken im brasilianischen Amazonasgebiet und ermöglicht dadurch eine umfassendere Bewertung politischer Entscheidungen.

Die empirische Strategie beruht auf kontrafaktischer Wirkungsabschätzung und verknüpft hochauflösende Satellitendaten zu Bodenbedeckung, Bränden, Luftverschmutzung und Klima mit administrativen und sozioökonomischen Informationen zu landwirtschaftlicher Produktion, Handelsströmen, Durchsetzung von Umweltvorschriften und Gesundheit. Die Identifikation stützt sich auf quasi-experimentelle Methoden, darunter das für gestaffelte Politikeinführungen geeignete generalisierte synthetische Kontrollverfahren, Instrumentalvariablen, die die exogene Variation in der Effektivität satellitengestützter Überwachung ausnutzen, sowie einen Ansatz der Differenzen in räumlichen Diskontinuitäten.

Die Ergebnisse zeigen, dass Politiken, die die Einhaltung von Umweltregularien verstärken – etwa durch Aufdeckung und Nennung von Gemeinden mit hohen Entwaldungsraten, durch verstärkte Überwachung und Durchsetzung von Umweltvorschriften sowie durch beschränkten Markt- und Kreditzugang für unzulässige Produktion – die Entwaldung und die Umwandlung von Wald in Weideland verringern. Gleichzeitig verlagert sich die Expansion profitabler Sojabohnenkulturen auf bereits gerodete Flächen, was auf eine flächensparende In-

tensivierung hindeutet. Lokale Arbeitsmärkte, Löhne und Kreditvergabe passen sich an diesen nachhaltigen Strukturwandel an, der mit Produktivitätsgewinnen verbunden ist. Entlang der Lieferketten mindern Händler Risiken, indem sie verstärkt in Absatzmärkte exportieren, die weniger sensibel auf importiertes Entwaldungsrisiko reagieren. Naturschutzpolitiken bringen zudem substanzielle Gesundheitsvorteile: Weniger Rodungen und Brände senken die Luftverschmutzung und Feinstaubbelastung und reduzieren damit einhergehende Krankenhauseinweisungen und Todesfälle.

Insgesamt deutet die Evidenz darauf hin, dass gut gestaltete Naturschutzpolitiken Entwaldung wirksam bekämpfen und gleichzeitig vielfältige *Co-Benefits* generieren können. Gleichzeitig sind sowohl *Co-Benefits* als auch Zielkonflikte stark kontextabhängig und vermutlich heterogen. Eine explizite Berücksichtigung dieser Heterogenität in der Gestaltung und Evaluierung von Politiken kann die Zielgenauigkeit und Wirksamkeit von Maßnahmen erhöhen, vulnerable Gruppen besser schützen und zu einer gerechteren Verteilung der Gewinne beitragen. Dadurch könnte auch die öffentliche, ökonomische und politische Unterstützung für den Waldschutz gestärkt werden.

Achievements

The research to Chapter 2 was presented on two occasions—first at the Research Conference on "Sustainability in Global Value Chains" organized by The Research Network Sustainable Global Supply Chains jointly with the United Nations Industrial Development Organization and the Kiel Center for Globalization, held online in December 2021, and second at the 27th Annual Conference of the European Association of Environmental and Resource Economists (EAERE) in Rimini, Italy, June 2022. It was finally published online in December 2023 as **Y. Damm, E. Cisneros, and J. Börner.** 2024. "Beyond Deforestation Reductions: Public Disclosure, Land-Use Change and Commodity Sourcing." *World Development* 175 (March): 106481. ISSN: 0305-750X. https://doi.org/10.1016/j.worlddev.2023.106481. The study gained attention by the United Nations Industrial Development Organization's Industrial Analytics Platform (https://iap.unido.org/), where a summarized version of the article was published in January 2024 (https://iap.unido.org/articles/can-public-scrutiny-make-supply-chains-more-sustainable).

An early version of Chapter 3, "Balancing Growth and Conservation: The Economic Impact of Environmental Enforcement in the Brazilian Amazon", was presented as a poster at the Global Land Programme's 5th Open Science Meeting: Pathways to Sustainable and Just Land Systems held in Oaxaca, Mexico, in November 2024.

Chapter 4 was presented online twice, once at the 26th Annual Conference of the EAERE in June 2021, and once at the 31st International Conference of Agricultural Economists in August 2021. An article with similar content was published in November 2024 as **Y. Damm, J. Börner, et al.** 2024. "Health Benefits of Reduced Deforestation in the Brazilian Amazon." *Communications Earth & Environment* 5, no. 1 (November): 1–9. ISSN: 2662-4435. https://doi.org/10.1038/s43247-024-01840-7. This study attracted considerable media attention. Following a press release by the University of Bonn, media articles were also published by the Deutsche Welle, and Brazilian news outlets O Globo and Agência Pública in November 2024.

Additionally, two co-authored papers were published in collaboration with the research group of Prof. Dr. Thomas Dietz from the University of Münster, focusing on voluntary sustainability standards in the coffee sector.

Publications

Damm, Y., J. Börner, N. Gerber, and B. Soares-Filho. 2024. "Health Benefits of Reduced Deforestation in the Brazilian Amazon." *Communications Earth & Environment* 5 (1): 1–9. ISSN: 2662-4435. https://doi.org/10.1038/s43247-024-01840-7. [xxi, 79]

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Batistic, P. M., T. Dietz, J. Börner, and Y. Damm. 2025. "Explaining Compliance with Voluntary Sustainability Standards: A Case Study from Colombia's Coffee Belt." *Sustainable Development* 33 (1): 1417–1440. ISSN: 1099-1719. https://doi.org/10.1002/sd.3189.

Working papers

Damm, Y., G. M. d. Oliveira, and J. Börner. 2025. *Balancing Growth and Conservation: The Economic Impact of Environmental Enforcement in the Brazilian Amazon.* SSRN Scholarly Paper, 5898282, Rochester, NY. https://doi.org/10.2139/ssrn.5898282. Social Science Research Network: 5898282. [49]

Chapter 1

Introduction

1.1 Motivation and background

Deforestation is an ongoing global phenomenon. Between 2001 and 2024 the world has lost around 5.17 million square kilometers (km^2) of tree cover (Global Forest Watch, GFW, 2014, based on Hansen et al., 2013). Most of it has been converted to agricultural land uses to produce food, feed, or fuels to meet the increasing demands of a still growing global population. But forests also provide important goods and services, and store value as natural capital. For example, forests absorb large amounts of carbon dioxide (CO_2) through photosynthesis—the main greenhouse gas driving climate change—and use it to produce carbohydrates, while releasing oxygen as a byproduct. Deforestation often releases the carbon back into the atmosphere, especially if it is connected to burnings. Tropical forests have already become net carbon emitters instead of net carbon sinks (Baccini et al., 2017; Aragão, Anderson, et al., 2018; Gatti et al., 2021; Qin et al., 2021; Mills et al., 2023).

Tropical forests, such as the Amazon rainforest, are particularly valuable ecosystems. The Amazon is estimated to harbor more than 10% of the world's biodiversity and plays a central role in carbon and water cycles (Science Panel for the Amazon, SPA, 2021). Humid primary forest loss has been progressing at a similar rate as global tree cover loss, with an estimated 0.83 million km^2 lost between 2002 and 2024 (GFW, 2014, based on Turubanova et al., 2018). Addressing the problem of uncontrolled deforestation has been declared a major goal by many world leaders, multinational corporations, and the scientific community (United Nations Climate Summit, 2014). Preventing the Amazon from reaching a "tipping point", a point from which considerable parts of the forest would dieback to a savanna-like state, is critical to the world carbon cycle and climate regulation (C. A. Nobre et al., 2016; SPA, 2021).

One of the major forested countries in the world is Brazil, which covers most of the Amazon rainforest and contained the largest share of tropical forests worldwide in 2020 (GFW, 2014, based on Brandt et al., 2023). Brazil also achieved considerable successes in reducing deforestation from its highest levels end of the 1990s and early 2000s to very low deforestation rates by 2012. For example, almost 27,800 km^2 were deforested in the Brazilian Legal Amazon (BLA) in 2004, while this number fell to around 4,600 km^2 in 2012 (TerraBrasilis, Assis et al., 2019 based on Satellite-Based Monitoring of Deforestation in the Brazilian Amazon, PRODES, cf. Figure 1.1). Since then deforestation has been increasing again in Brazil and worldwide, with 2024 breaking a forest loss record for the Amazon basin region, primarily driven by fires (Global Forest Review, 2025).

This variation in deforestation rates in the Brazilian Amazon, a result of various conservation policies and efforts, makes for an interesting study object. The primary goal of these conservation policies is to prevent forest loss. However, since intact forests provide many beneficial ecosystem services and deforestation can cause various negative external effects, conservation policies can have multiple side effects. These so-called "co-benefits" of conservation policies in the Brazilian Amazon are the main topic of this thesis. The term co-benefits originated in climate science in an attempt to describe the various positive secondary effects, or win-win scenarios, of reducing greenhouse gas emissions (Mayrhofer and J. Gupta, 2016). Some of the most popular and well studied co-benefits of climate policy are public health related, for example from air pollution reductions due to electrification of vehicles, or energy security from the reduced dependence on oil and gas imports (Gao et al., 2018; Karlsson, Alfredsson, and Westling, 2020; Moutet et al., 2025). In conservation science the idea of co-benefits has been gaining popularity in recent years (Raymond et al., 2017; Molina, Costello, and Kaffine, 2024; Helgeson, Al Kajbaf, and Fung, 2025). While the related concepts of externalities and ecosystem services have long been established, co-benefits are a useful term to describe other benefits going beyond ecosystem services, such as social and economic gains, or general benefits that do not necessarily directly impact a third-party utility.

This introductory chapter briefly discusses the bigger political and theoretical picture of the thesis. The following section provides valuable background information with regards to the importance of forests (Section 1.1.1) and the study area, the Brazilian Amazon region (Section 1.1.2). Afterwards, the underlying conceptual framework (Section 1.2) and the research objectives (Section 1.3) are discussed. Finally, Section 1.4 summarizes the main findings, and Section 1.5 discusses the contributions, limitations, directions for future research, and policy implications of the dissertation. Section 1.6 concludes and presents the structure of the dissertation.

1.1.1 Why forests matter

By now it is well recognized that intact ecosystems provide many valuable goods and services. According to the Millennium Ecosystem Assessment (2005), these can range from provisioning services, such as timber, non-timber products, or medicinal resources, to regulating services, for example climate regulation, water cycling, or flood control, but also include supporting services, which can be soil formation, nutrient cycling, or habitat provision, and even cultural services, such as spiritual, recreational, and aesthetic values. International political recognition of the value and preservation of these services manifested itself in the establishment of the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) in 2012 at the request of the United Nations (UN) General Assembly. In a 2019 report, IPBES notes that "biodiversity and ecosystem functions and services [...] are deteriorating worldwide" (IPBES, 2019, p. 10) and that "[m]ost of nature's contributions to people are not fully replaceable, and some are irreplaceable" (IPBES, 2019, p. 10).

Ecosystem services benefit human life directly and indirectly in many ways. For example, Costanza, de Groot, et al. (2014) estimate that the global value of ecosystem services in 2011 was around 125 trillion United States (US) Dollars (USD), roughly twice as large as global gross domestic product (GDP), but declining at a rate of USD 4.3–20.2 trillion per year between 1997 and 2011 (Costanza, d'Arge, et al., 1997; Costanza, de Groot, et al., 2014). Whenever an activity generates benefits for others that are not reflected in market transactions, economists consider this a case of positive externalities. The environment is therefore the source of countless positive externalities. Ideally, externalities should be compensated as they can otherwise lead to market inefficiencies. A private landowner, for example, could be paid for keeping her land forested and providing ecosystem services. However, many ecosystem services are not directly tangible. Carbon sequestration, which is important in climate regulation, is an indirect benefit to the entire planet. Excluding non-compensating individuals from this service is impossible. Thus, most ecosystem services, such as carbon sequestration, have public goods or common pool resource (CPR) characteristics. Many humans (and non-humans), up to the entire world, can benefit from these services for free or little compensation, as they are non-excludable. This means humanity is free-riding on nature's ecosystem goods and services (Ostrom, 1990).

Forests are particularly valuable in ecological magnitude and economic value, as their contribution spans all four categories of ecosystem services. Among the most important are the many regulating services, which are critical for life on earth. Forest protection is therefore an explicit target of the Sustainable Development Goal (SDG) 15, Life on Land (Sachs et al., 2025). The SDGs were developed by the UN in 2015 with the aim to promote and assess sustainable development until 2030. Indirectly forests and deforestation also play an important role in many other SDGs, for example SDG 2 Zero Hunger or SDG 13 Climate Action (Sachs et al., 2025).

Forests and deforestation play a crucial role in climate change due to their dual role as carbon sinks and sources. The Intergovernmental Panel on Climate Change (IPCC) Special Report on Climate Change and Land estimates that terrestrial ecosystems, including forests, acted as a net carbon sink, removing approximately 6.0 ±3.7 gigatons of CO_2 ($GtCO_2$) per year between 2007 and 2016 (IPCC, 2022). Simultaneously, land-use change—primarily deforestation—emitted about 5.2 ±2.6 $GtCO_2$ per year (IPCC, 2022). However, due to a combination of climate change and continued deforestation, degradation, and burnings, forests are shifting from net sink to net source of carbon. For example, world's tropical forests are estimated to already have been net sources of carbon for the 2003–2014 period (Baccini et al., 2017). Nevertheless, the IPCC report emphasizes that reducing deforestation and enhancing forest conservation and restoration are among the most effective land-based mitigation strategies available, with co-benefits for biodiversity, soil quality, and livelihoods (IPCC, 2022).

The UN Framework Convention on Climate Change recognized forest's greenhouse gases removal potential already in 2005 by discussing and later implementing a voluntary payment mechanism to reduce emissions from deforestation (RED) (Cadman, 2019). Later this scheme was improved to also include reduced emissions from forest degradation while acknowledging the role of conservation and sustainable management of forests, among others (REDD+). The main idea behind REDD+ is to create financial incentives for forest conservation and reduced deforestation, particularly in developing countries (Wunder, Schulz, et al., 2024). This is a crucial financial mechanism for the protection of valuable tropical forests, which are mostly located in developing countries. Tropical forest conservation is now more important than ever, with 2024 hitting a new record high in tropical forest loss,[1] as global forest cover declines continuously (GFW, 2014, based on Turubanova et al., 2018).

1.1.2 The Brazilian Amazon

The country of Brazil is the largest recipient of REDD+ payments, receiving more than USD 1.4 billion of all tracked disbursements until 2014 (Silva-Chávez, Schaap, and Breitfeller, 2015).[2] On the one hand, Brazil still had by far the largest remaining tropical tree cover worldwide in 2020 in absolute terms.[3] On the other hand, it

[1] 67,300 km^2 (GFW, 2014, based on Turubanova et al., 2018).

[2] Financial pledges to Indonesia are larger than for Brazil, but tracked disbursements are much lower (approx. USD 400 million, Silva-Chávez et al., 2015).

[3] 5.41 million km^2s vs. 1.69 million km^2 in Democratic Republic of Congo in second place (GFW, 2014, based on Brandt et al., 2023).

has been loosing tropical forest at a faster rate than the global average during the 2002–2024 period.[4]

Most of Brazil's tropical tree cover is located in the Amazon rainforest, in the northern and northwestern regions of the country. The Amazon is arguably the most globally significant tropical forest in size, ecological function, and symbolic value. Covering over 5.4 million km^2 in 2001 across nine countries, the Brazilian portion comprises roughly 60% of the total Amazon biome (Malhi, Roberts, et al., 2008; Santos, Salomão, and Veríssimo, 2021). The Amazon is a biodiversity hotspot, hosting more than 10% of the world's terrestrial species (SPA, 2021; Flores et al., 2024). It plays a vital role in regional and global climate regulation, including moisture recycling and rainfall patterns across South America and even North America via atmospheric teleconnections[5] (Werth and Avissar, 2002; Avissar and Werth, 2005; SPA, 2021; C. Smith, Baker, and Spracklen, 2023). The trees act as a biotic pump, transporting water from the soil through the roots into the leaves and finally the atmosphere by evapotranspiration (A. D. Nobre, 2014; SPA, 2021). The so-called "flying rivers" are estimated to transport more water than the Amazon River itself, distributing this moisture as rainfall southwards as far as northern Argentina (A. D. Nobre, 2014; SPA, 2021). Without these flying rivers, many regions in South America between around 15–35 degrees south latitude—which include the megacities São Paulo and Buenos Aires—could face severe drying and desertification risk, similar to other regions around the world at these latitudes, such as in southern Africa or Australia (A. D. Nobre, 2014).

Historically, the Amazon has also acted as a major carbon sink, but it is now becoming a net source due to widespread degradation and deforestation (Aragão, Anderson, et al., 2018; Gatti et al., 2021; Qin et al., 2021). Almost 20% of the Amazon has already been deforested (SPA, 2021; Veríssimo, 2023). At 40% total deforested area, the Amazon is estimated to pass a tipping point, beyond which significant portions may undergo irreversible dieback and transition into a savannah-like ecosystem, with drastic consequences for the regional and global climate (C. A. Nobre et al., 2016; SPA, 2021). Preserving ecosystems and reducing carbon emissions from land-use change is also an important component of Brazil's contribution to the Paris Climate Agreement (Government of Brazil, 2024). Ecosystem restoration and reforestation is expected to contribute to net-zero emissions by 2050 through nature-based carbon removals (Government of Brazil, 2024). Re-elected former President Lula committed to end deforestation by 2030 at the 27th Climate Conference of the Parties, formalized by a new Action Plan for the Prevention and Control

[4] 10% vs 8%, respectively (GFW, 2014, based on Turubanova et al., 2018).

[5] Atmospheric teleconnections refer to large-scale patterns of climate variability that link weather and climate conditions across distant regions of the globe. These connections arise due to the way atmospheric circulation systems, e.g. jet streams, interact and redistribute energy and moisture over thousands of kilometers (Feldstein and Franzke, 2017).

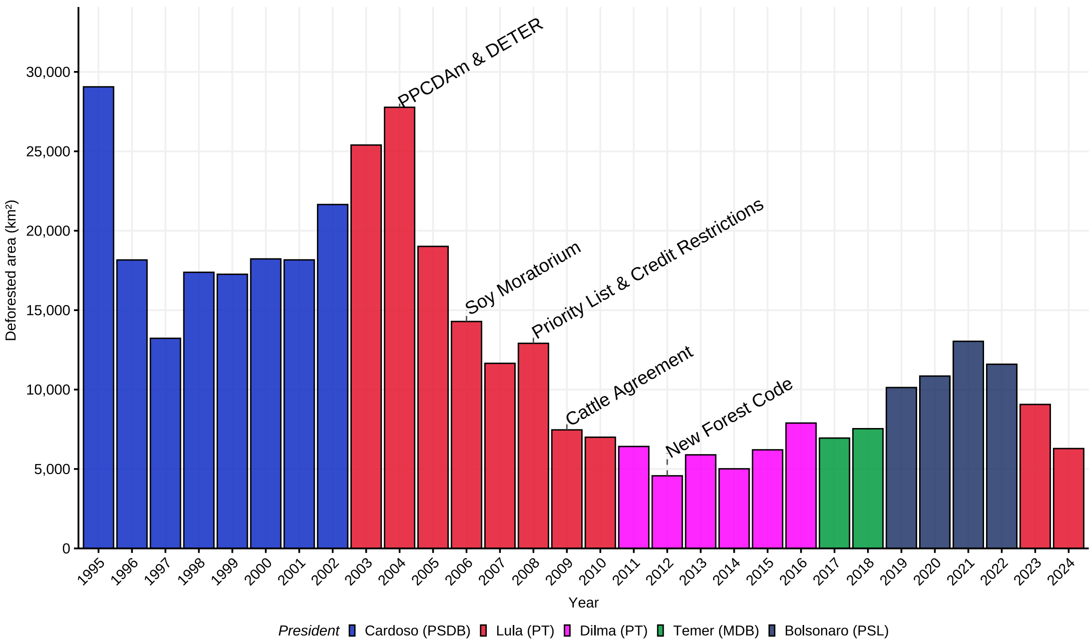
of Deforestation in the Legal Amazon (PPCDAm) (Brazilian Ministry of the Environment, MMA, 2023).

Deforestation and conservation policies in the Brazilian Amazon

The term "Brazilian Amazon" can refer to two distinct spatial definitions that are relevant for conservation policy. The BLA is an administrative region established by federal law, covering nine states. It extends beyond the ecological Amazon rainforest and includes parts of the neighboring Cerrado and Pantanal biomes, as well as transitional ecotone areas. In contrast, the Amazon biome corresponds to the ecological boundaries of the tropical rainforest and is therefore smaller and more homogeneous in its environmental characteristics. Many federal conservation policies apply to the legal boundaries of the BLA, whereas others target only the biome.

Deforestation dynamics in the Brazilian Amazon have fluctuated significantly over the past three decades. Figure 1.1 illustrates annual deforestation rates (in km^2) from 1995 to 2024, alongside changes in presidential administrations and the most important environmental policies in the context of this thesis.

Figure 1.1. Deforestation in the Brazilian Legal Amazon and presidency by year



Note: Deforestation is shown as bar charts. Different colors indicate different federal presidencies. Text labels above bars indicate most relevant Amazon conservation policies discussed in the thesis. *Source*: Adapted from Schielein (2020). Deforestation data based on PRODES, from TerraBrasilis, (Assis et al., 2019).

From 1995 to 2002, under President Cardoso, deforestation rates fluctuated between approximately 13,000 and over 29,000 km^2 annually. The highest value in the series—over 29,000 km^2—was recorded in 1995. During this period, the Brazilian Forest Code (FC) was transformed to more strictly regulate land use and management on private properties. Starting in 2001, the FC required landowners to keep a legal reserve of natural vegetation[6] and areas of permanent preservation in environmentally sensitive areas[7] (Soares-Filho, Rajão, et al., 2014). However, enforcement capacity remained limited and deforestation continued to rise from 2001 to 2004.

The following administration of President Lula, beginning in 2003, initially saw a sharp increase in deforestation, peaking at more than 27,000 km^2 in 2004, despite the strict FC regulation. However, this period also marked a decisive shift in environmental policy. Especially the launch of the PPCDAm in 2004 was a key turning point. PPCDAm is a concerted effort by several ministries, state agencies, and national institutes to prevent and control deforestation in the Amazon (West and Fearnside, 2021). It has undergone several distinct phases and the current 5th phase, from 2023 to 2027, is supposed to prepare to end deforestation by 2030 (MMA, 2023).

The federal agency Brazilian Institute of Environment and Renewable Natural Resources (IBAMA) plays a central role in enforcing environmental regulations. It is the federal environmental enforcement authority responsible for issuing fines, conducting inspections, executing embargoes on non-compliant properties, and coordinating field operations to suppress illegal deforestation and environmental crimes. Under PPCDAm, enforcement by IBAMA improved significantly in 2004 supported by a satellite-based Real-Time Deforestation Detection System (DETER) (Assunção, Gandour, and Rocha, 2023).

Furthermore, protected areas and indigenous territories expanded considerably by more than 709 thousand km^2 between 2002–2009 (Soares-Filho, Moutinho, et al., 2010; West and Fearnside, 2021), and a Rural Environmental Registry (CAR) and land tenure regularization helped to clarify and strengthen land rights. Later, starting in 2008, more targeted measures were introduced, such as the Priority List (PL) of high-deforestation municipalities (Arima, Barreto, et al., 2014; Cisneros, Zhou, and Börner, 2015; Assunção and Rocha, 2019) and rural credit restrictions to farmers in violation of the FC (Assunção, Gandour, Rocha, and Rocha, 2019), in order to target specific drivers and hotspots of deforestation.

Additionally, following a successful civil society campaign by Greenpeace (2006), a Soy Moratorium was signed by major soy traders, in which they pledged to not buy soy from farms in the Amazon biome that had deforested after the July 2006[8] cutoff date (Gibbs, Rausch, et al., 2015; Heilmayr et al., 2020). Greenpeace

[6] In the Amazon biome, landowners are required to keep 80% of their property as natural vegetation.

[7] For example riversides or hilltop areas.

[8] This was later changed to July 2008 to comply with the 2012 FC revision.

repeated this process later for the cattle sector, which led to the Cattle Agreement of 2009 (Greenpeace, 2009; Gibbs, Munger, et al., 2016). By 2012, annual deforestation had fallen to under 5,000 km^2, the lowest level on record. While it is acknowledged that public policies, civil society campaigns and supply-chain interventions significantly contributed to this success, macroeconomic factors, such as the 2008 financial crisis, fluctuations in commodity prices and the Brazilian Real–US-Dollar exchange rate also played an important role (Richards, R. Myers, et al., 2012; Assunção, Gandour, and Rocha, 2015; Richards, 2021; West and Fearnside, 2021).

The year 2012 marked a turning point, not only in deforestation trends but also politically. Under the presidency of Rousseff, a revision of the FC was approved, lobbied for by the agricultural sector (Soares-Filho, Rajão, et al., 2014). The sudden increase in enforcement efforts under President Lula unfolded the magnitude of illegal deforestation and landowners in violation of the FC. Under the old FC, landowners would have been required to restore the native vegetation at their own expense to clear their environmental debt. With the FC revision, an amnesty was granted for all illegal deforestation on "small" properties, up to 440 hectares,[9] before 2008 (Soares-Filho, Rajão, et al., 2014). The change in political spirit also manifested itself in the impeachment of President Rousseff in 2016 and the election of President Bolsonaro in 2018, a decidedly anti forest-conservation candidate. The recovering worldwide economy, especially demand from China, and increasing commodity prices, again contributed to the growing deforestation trend from 2012–2021.

In 2022, former President Lula was re-elected into office, promising a recovery of institutional control and renewed political commitment to forest protection. A decline in deforestation was once again observed, dropping to around 9,000 km^2 in 2023, with a further reduction in 2024 to just above 6,000 km^2.

1.2 Conceptual framework: Market failure in forest frontiers

Forests are defined as a group of trees with a certain height and canopy cover over some land surface. For example, the Food and Agricultural Organization (FAO) defines forests as "[l]and spanning more than 0.5 hectares with trees higher than 5 meters and a canopy cover of more than 10 percent, or trees able to reach these thresholds in situ" (FAO, 2023b, p. 7). By contrast, deforestation is defined as the permanent reduction of the tree canopy cover below the minimum 10 percent threshold (FAO, 2023b). Land cover neutrally describes the physical surface of the

[9] Which are 90% of all rural properties in Brazil (Soares-Filho, Rajão, et al., 2014).

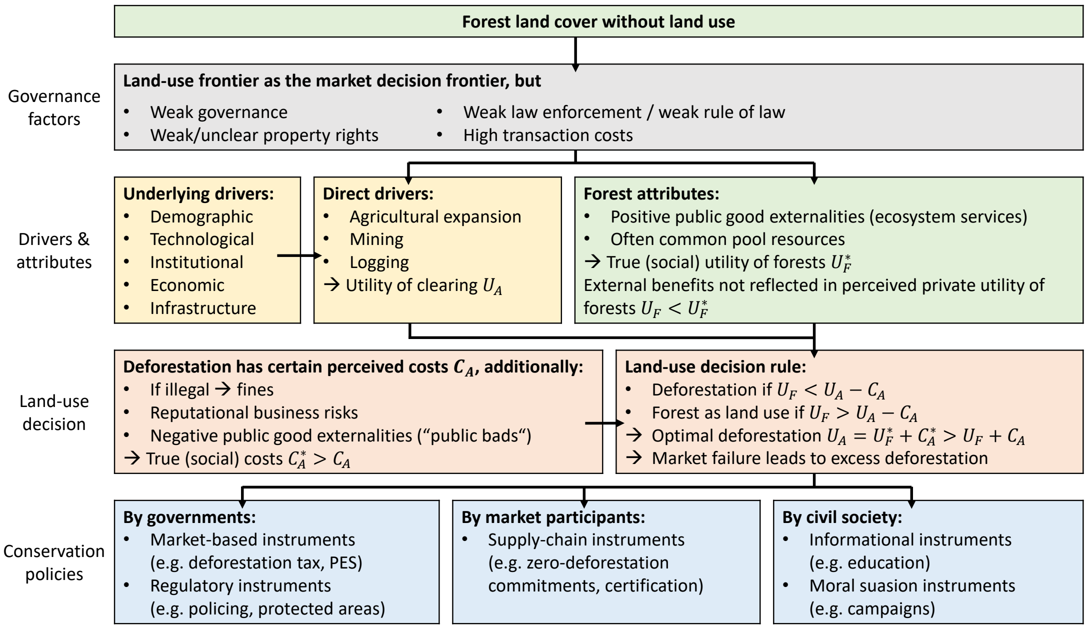
land, which could be, for example, forest, agriculture, or urban areas. This is distinct from land use, which describes how humans use or manage the land, often in a socio-economic context (Comber, 2008). Forest can therefore also be a land use, for example in a publicly managed protected area or if a private landowner decides to keep her land forested. However, in certain contexts it is possible to have forest land cover without a formally defined or officially recognized land use. This situation can occur, for instance, in remote parts of the Amazon rainforest, where land is covered in forest but remains outside the reach of formal administrative or economic systems. Such areas may still be inhabited or used by indigenous communities, whose livelihoods and territorial governance often stand in contrast, and at times in resistance, to external state or market pressures. From an official perspective, however, if these lands are not classified as protected areas or indigenous territories, their land use is undefined.

As human activities expand into forested areas, people make decisions about how to use the land—whether to maintain the forest or to convert it to other land uses. The land-use frontier represents the geographical edge where new land uses expand into previously unmanaged or less intensively used landscapes. It can be understood as the spatial manifestation of the market frontier, where economic incentives influence land-use decisions (J. H. V. Thünen, 1990; Angelsen, 2007). At this frontier, following a utilitarian framework, people evaluate alternative uses of land based on expected utility, deciding, for example, whether to engage in smallholder subsistence farming or to grow soy crops for the international export market. The decision to convert forests to other land uses thus depends on whether the expected utility from the alternative land use exceeds that from maintaining forest cover. This is the point at which land-cover changes such as deforestation or agricultural expansion are most likely to occur.

Figure 1.2 depicts this decision process in a conceptual framework. The utility from alternative land use (U_A) is influenced by many underlying factors (cf. Drivers & attributes in Figure 1.2). These can be demographic, for example increasing demand from growing populations; technological, if innovations raise crop yields; institutional, such as political factors supporting or constraining forest conversion; economic, including high agricultural prices; or infrastructural, such as roads that improve accessibility (Angelsen and Kaimowitz, 1999; Lambin, Turner, et al., 2001; Geist and Lambin, 2002; Busch and Ferretti-Gallon, 2023; Haddad et al., 2024). Depending on these drivers and land characteristics, expected returns to land use vary across activities. For instance, in some regions mining may yield higher expected private returns than agriculture, making it the dominant direct driver of deforestation there.

Thus, economic incentives drive land-use decisions, however, many governance factors—which are crucial for functioning markets—do not function properly at the land-use frontier. It is characterized by overall weak governance, i.e. unclear and difficult-to-enforce property rights, low levels of law enforcement and rule of

Figure 1.2. Conceptual framework of market failure in forest frontiers



Note: The figure illustrates how land-use decisions at forest frontiers are influenced by underlying and direct drivers, governance factors, and the relative utilities and costs of forest vs. alternative land uses. Because the private utility of forests ($\boldsymbol{U_F}$) is typically lower than their true social utility and the private costs of deforestation ($\boldsymbol{C_A}$) lower than their true social costs, land-use choices result in excess deforestation. Policies seek to correct this market failure by realigning private incentives with social welfare.

law, and also generally high transaction costs (cf. Governance factors in Figure 1.2). Most forest land at this frontier remains undesignated, i.e. publicly owned without a clearly defined management regime. As such, they are perceived as CPR, which means it is difficult to restrict people's access to these lands (non-excludable) but once deforested, the forest cannot be used by others anymore (rival). This can lead to over-exploitation and competition for claims to forest land (C. Araujo et al., 2009; Azevedo-Ramos et al., 2020; Moutinho and Azevedo-Ramos, 2023). The result could be a land rush, where there is a race to clear and claim forest land to prevent others from doing it (Dell'Angelo et al., 2017), ending in a potential "tragedy of the commons" (Hardin, 1968).

All of these market and non-market factors influence the land-use decision made by people at the land-use frontier. In essence, they evaluate the utility they receive from forest land use (U_F) vs. the alternative (U_A). However, forests also provide ecosystem services (Costanza, d'Arge, et al., 1997). Because these positive public-good externalities are not reflected in the perceived private utility of forests, the true social utility of forests is higher than the private utility ($U_F^* > U_F$) (Taye et al., 2021). Before a forest area is deforested, however, the costs of conversion and alternative land management (C_A) also need to be evaluated (cf. Land-use decision in Figure 1.2). These could, for example, be labor and capital costs to actually deforest and turn the land into productive use. Depending on where and how deforestation occurs, there could exist additional deforestation costs that could go unperceived. Deforestation could be illegal, and even if law enforcement is weak, this could change in the future and fines or penalties could be issued retroactively, driving up the costs of deforestation. Furthermore, deforestation or unsustainable behavior could be punished by market actors along the supply chain. For example, consumers or exporters could choose to not buy deforestation-contaminated goods, leading to considerable reputational business risks associated with deforestation. If deforestation leads to air or water pollution, or carbon emissions, for example from burnings, then it could also create negative public good externalities[10] (Balboni et al., 2023). The additional consequences and costs of pollution from deforestation would have to be paid by someone else, for example in form of higher societal health care costs. The true (social) costs of deforestation could therefore be considerably higher than the perceived private costs ($C_A^* > C_A$).

The land-use decision rule by people at the forest frontier will thus depend on these three variables: the perceived utility from the forest (U_F), the utility of alternative land use (U_A), and the perceived costs of conversion (C_A). Deforestation occurs if $U_F < U_A - C_A$ and forest is kept as land use when $U_F > U_A - C_A$. However, this leads to more deforestation than if the true social utilities and costs were used. According to this framework, optimal deforestation is reached when the utility of alternative land use is equal to the social utility of forests (U_F^*) plus the true social costs of deforestation (C_A^*), i.e. $U_A = U_F^* + C_A^*$. Notably, $U_F^* + C_A^* \geq U_F + C_A$, which means that markets fail to internalize the full social costs and utilities of land-use decisions. Consequently, this market failure results in excess deforestation beyond socially optimal levels (Kaimowitz and Angelsen, 1998; Balboni et al., 2023). Indeed, the CPR properties of forest, the positive public good externalities in form of ecosystem services, and the negative public good externalities from deforestation are all elements that contribute to market failure, which results in mutually reinforcing excess deforestation.

One way to remedy these market failures is through policy instruments that alter private incentives and try to improve market outcomes, i.e., to conserve forests and to reduce deforestation. Within the conceptual framework, conservation policies can act through five potential levers: governance factors, underlying drivers, direct drivers, forest attributes, and the costs of deforestation. Each of these levers ultimately shifts one or more of the variables that determine land-use choices—U_F, U_A, or C_A—although the precise mechanism is often policy- and context-dependent and may involve several channels simultaneously, for example in coordinated or hybrid policies. Depending on their origin, policy instruments can be classified by

[10] Also referred to as "public bads".

governance or organizing actor: the state, market participants, or the civil society (M. C. Lemos and Agrawal, 2006; Lambin, Meyfroidt, et al., 2014; Lambin and Thorlakson, 2018; Abbott and Snidal, 2021).

Governments have the power to change regulations, for example by prohibiting deforestation beyond a certain level. However, these rules also need to be enforced, which can be a challenge due to the remoteness of land-use frontiers. Alternatively, governments can use market-based instruments, for example by changing the private costs of deforestation $C_A \uparrow$ through a deforestation tax, or by increasing the private utility of keeping forest land $U_F \uparrow$ through subsidies.[11] These government-organized policies refer to the governments of the countries in which the forests are located.

Governments of countries importing deforestation-related goods represent their populations, who are the final consumers in international supply chains. They can also use regulatory or market-based instruments, such as due-diligence obligations, grievance procedures, or tariffs, to influence demand for deforestation-linked commodities in importing countries. By reducing market access or increasing compliance requirements for products associated with illegal or unsustainable clearing, such measures alter the underlying drivers of deforestation and diffuse back into producers' land-use decisions through lower private utility from clearing ($U_A \downarrow$) or higher perceived costs of non-compliance ($C_A \uparrow$) (Bager, Persson, and dos Reis, 2021).[12]

Market-based instruments are preferred by many economists as they are theoretically cost-efficient (Perman et al., 2011). However, cost efficiency presumes effectiveness. Since land-use frontiers can also be interpreted as the market frontier, the foundational assumptions of well-functioning markets often do not hold. Land-use frontiers are typically characterized by many weak governance factors. In such contexts, market-based instruments like PES or deforestation taxes struggle to deliver the expected outcomes, as the basic institutional infrastructure needed for effective implementation is underdeveloped. For instance, PES schemes require clearly defined land tenure and monitoring capacity to verify conditionality and additionality, yet land tenure insecurity is widespread in frontier regions (Börner, Wunder, Wertz-Kanounnikoff, Tito, et al., 2010; Wunder, Börner, et al., 2020).

[11] Generally referred to as Payment for Ecosystem Services (or also Payment for Environmental Services, PES). These can be arranged by the government to private landowners, e.g. in the form of subsidies, but there are various possible institutional arrangements. REDD+ is a form of PES between countries, while it is also possible that PES schemes are organized entirely by the private sector (Wunder, 2006; Engel, Pagiola, and Wunder, 2008; Wunder, Schulz, et al., 2024).

[12] An example is the EU Deforestation Regulation (EUDR) by the European Union (EU). It regulates imports of high-deforestation-risk commodities, such as soy or palm oil, which could change economic or institutional deforestation pressures (cf. Drivers & attributes in Figure 1.2), although direct deforestation impacts remain unclear (G. M. d. Oliveira, Ziegert, et al., 2024).

Similarly, deforestation taxes rely on the ability to observe and attribute land-use changes to individuals, which is often infeasible in practice. As a result, these instruments may be ineffective or lead to adverse selection. In contrast, command-and-control policies—such as protected area designations or environmental law enforcement—have demonstrated relatively greater effectiveness in frontier contexts (Pfaff et al., 2015; Abman, 2018; Börner, Schulz, et al., 2020; Wunder, Schulz, et al., 2024). These instruments do not depend on the existence of functioning markets. Nevertheless, there is a growing recognition that the effectiveness of any instrument depends on the quality of institutions and the governance environment (Börner, Schulz, et al., 2020). Many policies, therefore, seek to improve market conditions at the frontier, for instance by formalizing land tenure and enhancing legal enforcement capacity. In this sense, strengthening the institutional foundation can be seen as a precondition for market-based instruments to function as intended, rather than a substitute for more direct regulatory action (Börner, Wunder, Wertz-Kanounnikoff, Tito, et al., 2010; Robinson, Holland, and Naughton-Treves, 2014; Börner, Marinho, and Wunder, 2015; Robinson, Masuda, et al., 2018).

Most policies organized by the private-sector, i.e. market participants, affect the supply-chain. Traders (i.e. exporters and importers of goods) and processors can agree to deforestation moratoria or zero-deforestation commitments (ZDC) (Gibbs, Rausch, et al., 2015; Gibbs, Munger, et al., 2016; Lambin, Gibbs, et al., 2018; Garrett, Levy, et al., 2019; Grabs et al., 2021; zu Ermgassen, Bastos Lima, et al., 2022; Lambin and Furumo, 2023). Voluntary labeling with third-party certification, or other forms of supply-chain transparency are also ways to signal a sustainable origin of production to consumers in return for a price premium (DeFries, Fanzo, et al., 2017; Gardner et al., 2019). Retailers or brands could implement traceability standards or supplier screening that block non-compliant or embargoed producers, while financiers could make access to credit or insurance conditional on environmental compliance (Kedward et al., 2023). In the terms of the conceptual framework, these instruments change the private utility of land use U_A by decreasing demand or the threat of potential market exclusion. Perceived private land-conversion costs C_A could also increase through loss of finance or higher transaction costs due to transparency demands. However, the effectiveness of supply-chain instruments depends on the market share across firms and jurisdictional coverage. Supply-chain policies also need to be monitored and enforced, which can be costly if they are self-organized and not in alignment with public governance goals (Ziegert and Sotirov, 2024).

Beyond state and supply-chain instruments, a further class of informational and moral suasion instruments operates through education, information provision, persuasion, and capability building rather than through contracting power (Oates and Baumol, 1988; Konar and Cohen, 1997; McCormick, 1998; S. Gupta et al., 2007; Bowen, Tang, and Panagiotopoulos, 2020). Such instruments often originate from civil-society actors, but can also come from private initiatives or government agencies, and include public disclosure, third-party audits, non-governmental assessments, media reporting, or deforestation alerts (Tietenberg, 1998; Moffette, Alix-Garcia, et al., 2021). They can also comprise training, education, or technical assistance that aim to build the capabilities needed to adopt forest-compatible production systems (e.g., agroforestry, improved pasture and soil management, fire prevention) and to meet environmental and land-tenure requirements (Piñeiro et al., 2020). In the conceptual framework, these instruments primarily raise perceived private conversion costs C_A by increasing reputational and business risks, may lower the private utility of alternative land uses U_A if buyers refuse such products, and can increase the perceived utility of keeping forest U_F by reducing knowledge and adoption barriers. However, because informational and moral suasion interventions generally rely on voluntary behavior and lack direct enforcement, they tend to have limited effectiveness when used in isolation and are typically considered complements to regulatory and market-based policies (S. Gupta et al., 2007; Bryant et al., 2024).

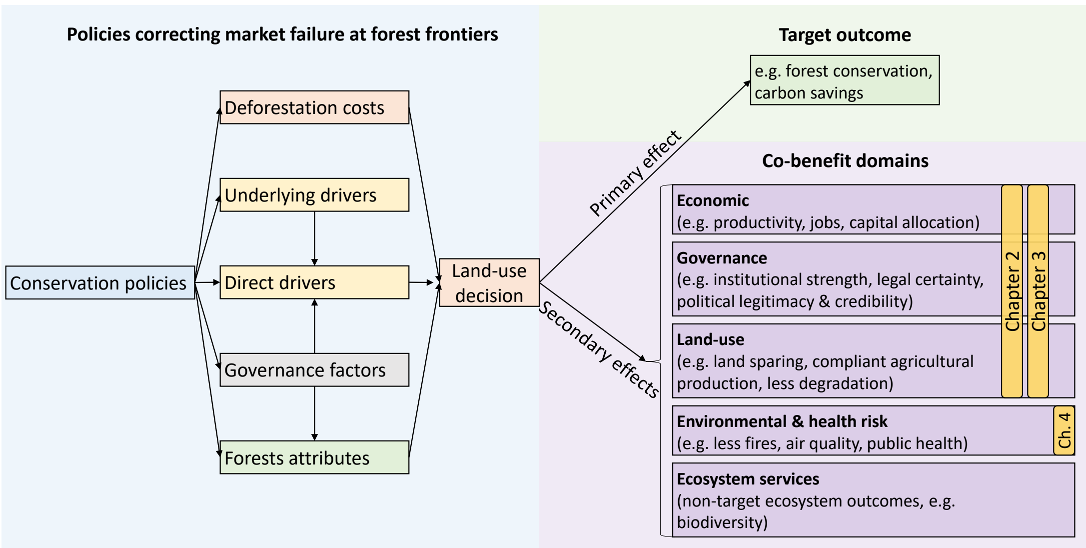
A potential critique to the conceptual framework presented in Figure 1.2 is that alternative land uses are an important income source to many poor rural households. Decreasing excess deforestation may be beneficial from a societal perspective, but may generate less income for households at the frontier region. Local economic and equity effects of conservation policies therefore need to be well understood and must already be considered in the policy design stage. Importantly, conservation policies and the economy does not have be at odds. Emerging theoretical and empirical evidence suggests that conservation policies can trigger economic development, especially in the context of underdevelopment and extensive agriculture as in the Amazon (Jayachandran, 2022). For example, Koch et al. (2019) and Moffette, Skidmore, and Gibbs (2021) show how conservation policies in the Brazilian Amazon limit the expansion of cattle ranchers previously relying on extensive ranching by increasing deforestation costs C_A. Farmers then invest their capital into their existing farmland, increasing agricultural productivity while reducing deforestation.

1.3 Research objective: Co-benefits of conservation policies

Following the conceptual framework presented in Figure 1.2, conservations policies can reduce deforestation by influencing the variables determining the land-use decision. Crucially, depending on the exact mechanism of change, all policies can potentially have impacts beyond deforestation reductions. Positive side-effects are the so-called co-benefits of conservation, while unintended negative side-effects are referred to as trade-offs. Co-benefits can come, for example, in form of positive public good externalities from ecosystem services of intact forests, or from reducing public bads from deforestation, both of which would increase social welfare compared to a scenario without conservation. Furthermore, realigning private land-use choices with social welfare can also have social and economic consequences downstream. For example, Moffette, Skidmore, et al. (2021) develop a theoretical model in which deforestation from extensive ranching is constrained due to regulatory policies. As a consequence, ranchers intensify their land use, which leads to on-farm investments, productivity gains, and less pasture degradation. In this scenario the conservation policy would not only have prevented deforestation by raising deforestation costs, but also triggered development in the agricultural sector, potentially leading to economic growth or job creation.

Figure 1.3 extends the conceptual framework by illustrating when and how co-benefits materialize across different domains. Starting at the policy stage, there are five potential levers to influence the land-use decision: the deforestation costs, the underlying and direct drivers, the governance factors, and the forest attributes. Depending on the levers chosen, the exact mechanism of change through the land-use decision variables (U_A, U_F, C_A) follows from the conceptual framework in Figure 1.2.

Figure 1.3. From policy-induced changes in land-use decisions to co-benefit domains



Note: Building on the conceptual structure in Figure 1.2, this figure illustrates how conservation policies generate a range of outcomes once they affect land-use decisions. Policies can intervene through five levers—governance factors, underlying and direct drivers, deforestation costs, and forest attributes—which influence the land-use choice by altering the decision variables ($\boldsymbol{U_A}, \boldsymbol{U_F}, \boldsymbol{C_A}$). While the primary objective is forest conservation, interventions can also produce secondary effects across several co-benefit domains, including land use, economic, reductions in environmental & health risks, governance-related, and ecosystem-service co-benefits. Chapters 2–4 examine selected domains empirically.

In contrast, Figure 1.3 shows how conservation policies can achieve their primary objective by altering land-use decisions at the forest frontier, but this may generate secondary effects that can be beneficial across various domains. For example, protecting forests will also preserve other non-target ecosystem outcomes as co-benefits, such as biodiversity or water cycles. Environmental & health risk co-benefits are associated with reductions in public bads due to deforestation, for example less air pollution and improved public health, but potentially also avoided carbon emissions, if they are not a policy target. Economic co-benefits can arise when conservation policies incentivize productivity and efficiency, for example by reallocating labor or capital towards higher-value agricultural sectors. Land-use co-benefits include more efficient use of already-cleared land, such as through land-use intensification. Governance co-benefits may materialize through improved enforcement capacity, more regulatory compliance, or improved land-tenure administration, but can also include increased legitimacy, credibility, transparency, and accountability in environmental governance.

Importantly, the relationship between policy levers, incentive channels (U_F, U_A, C_A), and co-benefits is not one-to-one. A single policy may activate several mechanisms simultaneously and a single mechanism can also generate co-benefits across multiple domains. For instance, raising deforestation costs may primarily act through the C_A channel, but it can also trigger agricultural intensification and productivity gains (economic and land-use co-benefits), reduce fire activity and air pollution (environmental & health risk co-benefits), and preserve ecosystems (ecosystem services co-benefits). Similarly, governance-focused interventions can both lower transaction costs and produce governance co-benefits such as stronger legal certainty. The two figures are therefore complementary: the conceptual framework illustrates the behavioral channels of forest loss and protection, while the co-benefits categorization highlights the potential range of secondary benefits.

It is important to note that the categorization in Figure 1.3 focuses exclusively on positive side-effects of conservation policies. This follows directly from the definition of co-benefits as win–win outcomes alongside the primary objective of reducing deforestation. However, conservation policies may also generate trade-offs or unintended negative side-effects, for example compliance costs, distributional impacts, or leakage. These aspects do not appear in the figure because they fall outside the conceptual scope of co-benefits, but they remain relevant for a complete policy assessment (Ferraro, 2003). Potential negative effects are therefore discussed where appropriate in the discussion Section 1.5 and the empirical chapters.

1.3.1 Research questions

Despite extensive work on the effectiveness of conservation policies, much less is known about how policies in the Brazilian Amazon have impacted broader outcomes. Understanding these impacts is essential, both for evaluating the full welfare implications of conservation policies and for informing policy design. This motivates the central research question of this dissertation: What are the co-benefits of conservation policies in the Brazilian Amazon? Addressing this question requires moving beyond the narrow focus on avoided forest loss to examine how policies affect land-use dynamics, supply-chains, productivity, or public health. Each chapter therefore investigates specific mechanisms and outcomes of conservation policies and their potential co-benefits.

The first two chapters deal with similar questions as in the Moffette, Skidmore, et al. (2021) example above. Chapter 2 analyzes land-use change and soy supply-chain responses following the public disclosure of priority municipalities with high deforestation in the Brazilian Amazon. This Priority List (PL) is a unique policy instrument in that it did not introduce new conservation rules, but instead informed regulatory agencies—such as IBAMA and rural credit suppliers—where to concentrate their efforts. In addition, the publication of municipality names on the PL operates as a moral suasion instrument, making the policy a hybrid of regulatory and informational approaches. Prioritized enforcement primarily raises deforestation costs ($C_A \uparrow$) through fines or credit constraints, while the public disclosure may influence deforestation through reputational channels or changes in market demand. Importers may avoid sourcing from listed municipalities, which can reduce the perceived private utilities of clearing forest land ($U_A \downarrow$). The main research question for this chapter is: How do producers of agricultural goods and traders adjust land-use and trading patterns in publicly disclosed high-deforestation municipalities? Firstly, it examines whether the PL reduced deforestation and associated land-use conversion in targeted municipalities. Secondly, it investigates how agricultural producers adjusted their land-use decisions following the disclosure, particularly regarding the allocation of pasture and cropland. Thirdly, it assesses how traders responded in sourcing and export-market choices when municipalities were publicly labelled as high-deforestation areas. The co-benefits evaluated in this chapter relate primarily to land-use and economic domains, specifically the soy supply-chain. However, closely related governance co-benefits are also discussed.

Chapter 3 focuses on the enforcement mechanism of regulatory instruments. Following the introduction of PPCDAm and DETER in 2004, the enforcement of environmental regulations increased considerably due to stronger political commitment and improved detection systems. Enforcement actions included fines for illegal deforestation, embargoes of non-compliant farmers, but also confiscation or destruction of machinery used in illegal deforestation. These measures raise the costs of illegal deforestation ($C_A \uparrow$) and of agricultural expansion into forests. In theory, higher deforestation costs induce farmers to use scarce land more productively, which in the Amazon often implies reallocating land from pastures to soy. However, because soy production has different input requirements, adjustments may not only be limited to land-use patterns but could also reshape local economies.

The main research question for this chapter is: How does stricter environmental enforcement impact land use and local economies in forest frontiers? Firstly, it investigates whether enforcement redirects agricultural expansion away from forests towards already-cleared land. Secondly, it examines how productivity, and formal employment and wages adjust across agricultural subsectors. Thirdly, it assesses how broader economic indicators, such as agricultural GDP and rural credit allocation, respond to changes in enforcement intensity. Chapter 3 thereby evaluates the economic and land-use co-benefits, but also discusses the governance co-benefits associated with stricter environmental enforcement .

Chapter 4 turns to the environmental & health risk co-benefits of conservation by studying air quality and human health. Regionally focused conservation efforts, such as the Amazon Soy Moratorium, rural credit restrictions, and the Cattle Agreement, reduced deforestation pressures and associated fires more within the Amazon biome than just outside it. Fewer deforestation-related fires should translate into lower smoke exposure and improved air quality and human health. In the conceptual framework, a combination of supply-chain and regulatory policies that suppress clearing and burning reduce a major public bad by increasing the costs of deforestation ($C_A \uparrow$) and lowering the private utilities of clearing ($U_A \downarrow$). These shifts may create substantial welfare gains for affected populations, including fewer pollution-related hospitalizations and deaths. The research question underlying this chapter is: To what extent did biome-focused conservation policies reduce deforestation-related fires and improve ambient air quality and human health outcomes in exposed populations? Firstly, it examines whether conservation measures reduced deforestation-related fire activity in treated regions. Secondly, it assesses whether these changes in fire activity translated into improvements in ambient air quality, measured, for example, through fine particulate matter ($PM_{2.5}$) concentrations. Thirdly, it quantifies the resulting health effects by estimating the impact of improved air quality on hospitalizations and mortality.

1.3.2 Methodology and data

This dissertation applies counterfactual impact evaluation to estimate causal effects of conservation policies in the Brazilian Amazon. The intuition behind this method is to compare observed outcomes in exposed places with credible estimates of what would have occurred absent exposure—the counterfactual. Identification hinges on constructing these “no-treatment” trajectories in ways that minimize alternative explanations. Research designs are matched to policy timing and geographic scope, for example they accommodate staggered rollout, allow for treatment effect dynamics, or address endogeneity.

Methodologically, the dissertation implements three complementary quasi-experimental designs. Chapter 2 uses the generalized synthetic control method (Xu, 2017) to estimate dynamic treatment effect paths under staggered municipal inclusion in the PL, while absorbing latent, time-varying confounders via interactive fixed effects (Bai, 2009). Chapter 3 estimates a two-stage least squares panel instrumental variable model, in which enforcement intensity is instrumented by satellite-based visibility impairments due to cloud cover that affect real-time deforestation monitoring capacity. Specifications include weather and climatic control variables, municipality and year fixed effects, and two-way clustered inference to identify causal effects on land use and local economies. Chapter 4 exploits a geographic discontinuity in conservation-policy coverage at the Amazon biome boundary in a double-differences design to link reduced deforestation-related fires to declines in air pollution and respiratory morbidity. The quasi-random course of the biome boundary and sample restriction to observations at close distance to the border ensure comparability of municipalities. Two-way fixed effects and time-varying controls further mitigate confounding bias and absorb unobserved heterogeneity.

All analyses rely on data assembled from publicly available sources. Municipality-by-year panel data sets were build for each study, integrating satellite-based land cover, land-cover change, air pollution or weather/climatic data (e.g. MapBiomas, INPE, MODIS, TerraClimate, SISAM), and socioeconomic and agricultural statistics (IBGE). Additionally, each chapter also uses additional datasets to answer specific research questions. Examples are destination-resolved commodity flows (Trase) in Chapter 2, formal labor and wages data (RAIS) in Chapter 3, or health outcomes (DATASUS) in Chapter 4. The exact methodological details and data processing steps are explained in the individual chapters.

1.4 Summary of findings

Across the three empirical chapters, the dissertation shows that conservation policies in the Brazilian Amazon effectively reduced deforestation and generated effects that extend beyond avoided forest loss.

In Chapter 2, the public disclosure of high-deforestation municipalities reduced deforestation and pasture expansion in listed municipalities. This suggests that environmental enforcement and credit restrictions raised perceived private costs of clearing C_A, preventing forest loss. Soy area expanded primarily by replacing pastures and other cropland, intensifying land use on already cleared land—a more sustainable way of increasing production than at the cost of forests and a land-use co-benefit of the PL. Traders exported additional soy exclusively to China and other non-EU markets, likely mitigating reputational risks associated with sourcing from municipalities with high-deforestation history. However, by reducing consumer market options, businesses risk losing out on profits, thereby reducing the utilities of clearing forest land U_A, reinforcing the overall conservation effect. Nevertheless, more compliant agricultural production can be considered an economic co-benefit.

Chapter 3 examines one of the enforcement mechanisms underlying the PL more closely, namely stricter environmental enforcement. Fines for illegal deforestation, which directly raise C_A, reduced forest loss and pasture expansion in the Amazon biome. Municipalities facing stronger monitoring and sanctions also experienced significant increases in employment and wages in the soy sector, leading to production and productivity gains. Soy crops started expanding on already-cleared land rather than forests. Rural credit appears to have shifted towards fewer but larger loans, potentially reflecting more compliance with environmental regulations. All these outcomes, including the land-use and economic co-benefits, were achieved without measurable losses in agricultural GDP, indicating that more enforcement of environmental regulations need not come at the costs of overall economic performance.

Chapter 4 links conservation to environmental & health risk co-benefits. Conservation measures implemented in the Amazon biome between 2006 and 2009 reduced deforestation pressure further inside the biome than in adjacent non-biome BLA regions, leading to lower fire activity and reduced ambient air pollution—mainly fine particulate matter ($PM_{2.5}$). Improvements in air quality translated into fewer hospitalizations and avoided deaths. These results reveal a substantial public-health premium to forest conservation that is often omitted from private land-use decisions, as highlighted by the difference between private and social deforestation costs (C_A and C_A^*, respectively) in the conceptual framework.

1.5 Contributions and discussion

This dissertation makes four main contributions to the literature on conservation science, environmental & land-use economics, and sustainable development. Together these contributions improve our understanding of how conservation policies change the land-use incentive structure and can generate multiple secondary effects.

First, the dissertation contributes to the conceptual understanding of conservation policies as multi-dimensional interventions. It develops a framework that shows how different conservation policies can correct land-use market failures by linking policy levers to the main variables determining land-use decisions (U_F, U_A, C_A). Policies that change land-use decisions, even when designed primarily to curb deforestation, can lead to a variety of secondary effects. The framework therefore provides an integrated structure for analyzing conservation policies by their mechanisms and categorizing co-benefits across different domains (Börner, Baylis, et al., 2016; Raymond et al., 2017; Newell, Dale, and Lister, 2022; Balboni et al., 2023; Newell, 2023; Molina et al., 2024; Helgeson et al., 2025).

Second, the dissertation adds to the literature on land-use and supply-chain responses to conservation. Existing research has documented substantial declines in deforestation following major policy reforms in Brazil, yet much less is known about how producers and traders adjust within the broader land-use system and new incentive structure. It shows that producers use scarce land more efficiently by intensifying agricultural production, while traders follow a dual-strategy of pursuing profitable opportunities but also minimizing business risks. By focusing on these behavioral adjustments, the dissertation complements earlier work on land-use change and supply-chain governance in the Brazilian Amazon (Lambin, Gibbs, et al., 2018; Gardner et al., 2019; Garrett, Levy, et al., 2019; zu Ermgassen, Ayre, et al., 2020; Grabs et al., 2021; zu Ermgassen, Bastos Lima, et al., 2022; Lambin and Furumo, 2023).

Third, the dissertation finds evidence of economic co-benefits of conservation policies. It demonstrates that land-use adjustments following stricter environmental enforcement also influence labor and capital allocations used in agricultural production, thereby transforming local economies. This contribution speaks to the broader development economics literature, which has often treated environmental regulation and rural economic transformation as separate or even competing domains (Tallis et al., 2008; Garrett, Cammelli, et al., 2021; Jayachandran, 2022; Barros and Chimeli, 2025; R. C. Lima et al., 2025; Silveira et al., 2025).

Fourth, the dissertation provides evidence on health-related co-benefits to conservation. By linking conservation policies to changes in fire activity, air quality, and human health, the analysis connects environmental governance to human well-being in a quantifiable manner. This contribution complements and extends emerging research on the social costs and risks of deforestation, which can affect humans in many direct and indirect ways (S. Myers et al., 2013; Bauch et al., 2015; Mastel et al., 2018; Ellwanger et al., 2020; Newell, 2023; Moutet et al., 2025; S. S. Myers et al., 2025).

1.5.1 Limitations

Despite rigorous impact evaluation methods, carefully designed identification strategies, and robustness checks, limitations to causal inference remain. First and foremost, there is context specificity. All findings were derived from the Brazilian Amazon region during the 2000–2018 period, a particular policy era with a dedicated forest conservation focus. The external validity to other biomes, time periods, political and institutional settings, or commodity markets may be limited. On the other hand, considering the internal validity, there is considerable overlap in the timing, mechanisms, and goals and intentions of the various conservation policies post-2004 in the Brazilian Amazon. Even though impact evaluation designs try to identify and isolate individual policy effects, it seems unlikely that results are completely independent of other policy impacts with complementary and mutually reinforcing conservation goals. Furthermore, there are always data and measurement constraints to empirical estimations. All satellite-based data sources, such as land-cover classification, fire detection, air pollution, and weather variables are prone to measurement error, for example due to cloud coverage, coincidental timing of measurement, or misclassifications (Wuepper, Oluoch, and Hadi, 2025). Other datasets, such as administrative or economic records, also face constraints. For example, Brazilian agricultural production data is mainly based on expert estimates, health outcomes mostly capture severe cases and may miss morbidity outside formal care or misclassifications by doctors, formal employment and credit data cannot capture informal relationships often present in rural settings, and trade data may not cover re-exports or informal trade flows.

Other limitations concern the net effects of conservation policies. While some robustness checks in this dissertation account for the possibility of spillover effects, leakage can still occur, not only in the close vicinity, but also to other biomes, regions, or even deforestation displacements to other countries (Bastos Lima, Persson, and Meyfroidt, 2019; Meyfroidt et al., 2020; Moffette and Gibbs, 2021; Villoria et al., 2022). Similarly, general equilibrium effects can differ substantially from estimated effect sizes, leading to over- or underestimated net impacts (R. Araujo, Assunção, and Bragança, 2025). Finally, methodological approaches are carefully designed to address key confounders, but omitted variable bias is always possible if unobserved shocks correlate with both policies and outcomes. Additionally, estimates focus on short- to medium-term responses and may not capture long-run changes in policy impact, for example due to political pressure or institutional weakening (Kuschnig et al., 2023; Nunes et al., 2024), or rebound effects due to increases in productivity or efficiency (Szerman et al., 2022).

1.5.2 Directions for future research

Future research could focus on the bigger and smaller picture, respectively. Considering the smaller picture, distributional and political economy effects of conservation policies warrant closer attention. For example, there is some evidence that the co-benefits in agricultural productivity studied in this dissertation were more likely to benefit larger producers rather than smallholders, who face more binding constraints in adapting to changing incentives (Crepin, 2024). Conservation-induced land scarcity incentivizes more efficient use of land, but can also lead to more land conflicts (Fetzer and Marden, 2017; Furumo et al., 2024). While some evidence suggests that there are governance co-benefits of environmental enforcement in form of less homicides due to reduced land conflicts (G. M. d. Oliveira and B. V. Miranda, 2024), there is also opposing evidence showing that smallholders with unclear land titles are expelled from their lands, often under threats of violence, following a sudden appreciation of land values (Sauer, 2018; Kröger, 2024). The increasing presence of organized crime, which can include public actors, further complicates the complex dynamics of conservation, development, and violence at forest frontiers, which remain insufficiently understood and represent an important direction for future research (Stahlberg, 2022; Ferreira, 2025).

Additionally, future work should examine the impacts of conservation on indigenous communities, traditional populations, and gender-differentiated groups. These groups often rely on forest resources in ways that differ fundamentally from commercial producers and/or are particularly vulnerable. Analyzing these heterogeneous effects will help ensure that conservation policy outcomes are evaluated not only on average, but also in terms of their distribution across social groups (Börner, Baylis, et al., 2016).

Furthermore, there are possibly more co-benefits to conservation policies than analyzed and discussed here. These co-benefits need to be identified and quantified, also along their monetary valuation, if possible (Strand et al., 2018). A deeper knowledge of co-benefits is necessary to assess the synergies and trade-offs of different political objectives. To identify possible co-benefits, the understanding of the exact mechanisms of change of each policy and their context is crucial. Qualitative studies can provide such valuable context, while evaluations of conservation policies in other frontier regions are needed to assess the external validity of the findings.

Such replications and cross-country studies also contribute to the bigger picture of the effectiveness of conservation policies and their co-benefits. While methodological and data collection advances can help to improve the internal validity of studies, comparative studies help to understand the context-specific conditions for the effectiveness of policy instruments. Additionally, cross-country studies are necessary to understand and analyze complex leakage effects, such as cross-commodity or cross-border displacement and indirect land-use change. Finally, analyses of long-run and general equilibrium effects can deepen our understanding of policy success or failure.

1.5.3 Policy implications

The findings of this dissertation suggest several implications for the design and implementation of conservation policies in tropical forest frontiers and possibly elsewhere. A central insight is that policymakers can focus on single-target interventions that correct market failures driving excessive deforestation. By narrowing the gap between private land-use incentives and the social optimum—thereby improving welfare, the initial benefit—such policies create the basis for co-benefits. In forest conservation, ecosystem services and reductions in environmental & health risks are the most likely co-benefits, but this dissertation shows that economic, land use, and governance co-benefits can also materialize. In contrast, policies with multiple objectives simultaneously risk diluting their effectiveness, as success metrics and mechanisms may become unclear or even competing with each other (Bode et al., 2011).

Nevertheless, the discussion also shows the need to consider equity and vulnerability in the implementation of conservation policies. The distribution of co-benefits and trade-offs, such as adjustment costs, is unlikely to be uniform. Large producers and smallholders differ in their ability to adjust to changing incentives, and vulnerable groups may face disproportionate burdens unless conservation measures are accompanied by targeted support. Complementary instruments, such as improved access to credit, technical assistance, or clearer and more secure land tenure, can help ensure more equitable welfare gains (Robinson, Masuda, et al., 2018).

Involving stakeholders already in the policy design stage, such as local communities and supply-chain actors, can help prevent policies from disproportionately benefiting or disadvantaging specific groups. Better alignment of goals across actors can also reduce potential points of failure, such as leakage across commodities or jurisdictions. A related challenge that requires political attention concerns land conflicts arising from increased land scarcity due to conservation, as well as the emergence of organized crime in some forest frontiers. Although governance co-benefits of conservation policies exist, more targeted institutional interventions will likely be needed to address these issues.

Finally, communicating co-benefits may enhance public and political support and encourage long-term investment in conservation (Bain et al., 2016). For example, explaining how forest protection can be compatible with local development may help shift perceptions of conservation being primarily restrictive to possibly being welfare improving.

1.6 Conclusion and structure of the dissertation

This introduction outlined the motivation for studying conservation policies in tropical forest frontiers, the conceptual framework linking market failures to land-use decisions, and the research objective of examining the co-benefits that can arise when policies try to correct these land-use market failures. The following chapters show that interventions targeting deforestation can generate multiple co-benefits across economic, land-use, environmental & health risk, and governance domains. At the same time, these effects depend on initial conditions and contextual factors, and are therefore likely to be heterogeneous. Understanding both, the primary conservation outcomes and the secondary effects they induce is essential for designing policies that are effective, welfare enhancing, equitable, broadly supported, and consistent with broader SDGs (Börner, Schulz, et al., 2020).

The dissertation consists of three empirical chapters, each written as a self-contained paper but connected through the common conceptual framework, the use of counterfactual impact evaluation methods, and the focus on co-benefits of conservation policies in the Brazilian Amazon. Following this introductory chapter, Chapter 2 analyzes how the public disclosure of priority municipalities affected land use and supply-chain dynamics. Chapter 3 examines how intensified environmental enforcement reshaped land use and local economic outcomes. Chapter 4 studies how conservation-induced reductions in deforestation and fires translated into improvements in air quality and human health. Together, these chapters provide complementary evidence on the multiple impacts of conservation policies on deforestation and beyond.

Chapter 2

Beyond Deforestation Reductions: Public Disclosure, Land-Use Change and Commodity Sourcing*

Joint with Elías Cisneros and Jan Börner

Abstract

Global commodity supply chains contribute significantly to environmental degradation and greenhouse gas emissions. Improving supply chain transparency can create public awareness and encourage relevant actors to improve their ecological footprint. We exploit Brazil's Priority List policy for the Amazon region, a public disclosure mechanism introduced in 2008 that effectively reduced deforestation rates, to study how land users and commodity traders respond to the corresponding reputational risk exposure. Specifically, we combine remotely sensed land use data with spatio-temporally disaggregated soy trade statistics covering 15 years and 770 municipalities to measure the effect of priority listing on land-use change, sourcing patterns, and trade destinations. Using the generalized synthetic control method, we find that priority listing led to a sizable drop in deforestation and a corresponding reduction in pasture expansion. At the same time, soy expansion increased significantly, but instead of expanding into natural forests, it mostly replaced existing pastures and other cropland. The additional soy production was exported predominantly to China, whereas exports to the EU stagnated.

JEL Classification: F18, Q17, Q56, Q58

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Keywords: Brazilian Amazon, Deforestation, Land-use change, Soy exports, Supply chain

2.1 Introduction

Policy makers, businesses and individual consumers must collectively take action to protect the environment and adapt to impacts of climate change (cf. Delabre et al., 2021). In the agricultural sector, policy makers often focus on individual land users by promoting sustainable practices and enforcing regulations to reduce deforestation. Command and control measures have been proven to help conserve forests if they are enforced effectively (Hargrave and Kis-Katos, 2012; Börner, Kis-Katos, et al., 2015). At the same time, consumers in some world regions are slowly shifting their consumption behavior towards more sustainable patterns (cf. Sesini, Castiglioni, and Lozza, 2020). Meanwhile, international traders of agricultural commodities react to both governmental enforcement and changing consumer preferences vis-à-vis international markets (Carlson et al., 2017; Lambin, Gibbs, et al., 2018; Gardner et al., 2019).

This paper analyzes how enhanced public scrutiny affects agricultural production and related land use decisions, as well as the corresponding sourcing patterns of international traders (i.e., exporters and importers). Specifically, we investigate the effects of Brazil's public disclosure policy—the so-called "list of priority municipalities"—a publicly disclosed list of municipalities with high deforestation rates in the Brazilian Amazon starting in 2008.

The Priority List (PL) has effectively reduced deforestation rates (Arima, Barreto, et al., 2014; Cisneros et al., 2015), arguably driven by intensified monitoring and stricter law enforcement (Assunção and Rocha, 2019). At the same time, public disclosure may have encouraged additional stakeholder action to mitigate the negative consequences of intensified public scrutiny and related reputational risks (Neves and Whately, 2016; Koch et al., 2019; Sills et al., 2020; dos Santos Massoca and Brondízio, 2022). Since cattle and soy production have been the main drivers of deforestation in the Brazilian Amazon (Rajão et al., 2020), traders of these goods may have entertained an interest in avoiding exposure to negative public attention and civil society campaigns in order to maintain trade relationships with PL municipalities. If international traders were sensitive to such public pressure, they could either invest in leveraging existing business relationships towards making production more sustainable or in reducing exposure by preferentially sourcing from regions subject to lower public attention or deforestation pressure. In any case, the reputational risk of doing business in a PL municipality is most likely to be higher for traders who export to countries with an active civil society and a free press. A bad reputation could lead to a reduction of business opportunities, termination of contracts with business partners, or decreased overall demand (Wolf, 2013; Gibbs, Munger, et al., 2016).

To study the behavior of agricultural producers and traders, we use a novel data set of detailed trade statistics across Brazilian municipalities providing yearly production and export quantities of soy over 15 years (Godar et al., 2015; Lathuillière, Suavet, et al., 2022). The disaggregated information on destination regions and trading groups allow us to investigate heterogeneous shifts in production and sourcing after PL. We connect trade shifts to environmental outcomes using yearly data on remotely sensed forest losses and industrial soy expansion. To analyze the dynamic impacts across time, we use the generalized synthetic control (GSC) method (Xu, 2017). The synthetic control method creates a counterfactual for an individual treated unit by weighting controls such that they have similar pre-treatment outcome paths (Abadie, Diamond, and Hainmueller, 2010). The GSC generalizes the concept to multiple treatment units. It also addresses a potential bias from the staggered treatment patterns (cf. de Chaisemartin and D'Haultfœuille, 2020; Callaway and P. H. Sant'Anna, 2021; Goodman-Bacon, 2021; Sun and Abraham, 2021; Borusyak, Jaravel, and Spiess, 2024), while relaxing the assumption of constant unobserved confounders (Kreif et al., 2016).

Our results show that PL municipalities became relatively more sustainable despite growth in soy production. During our study period, PL municipalities reduced deforestation and pasture expansion by 70.2% and 85.1%, respectively. At the same time, soy area expansion increased by 103.4% in PL municipalities. Both effects are possible because PL municipalities expanded soy production almost exclusively on existing pastures and other cropland—61.1% and 49.7% more, respectively, than in non-PL municipalities. The PL thereby transformed local agricultural production away from extensive cattle ranching to intensive soybean production with much higher per hectare returns. Furthermore, the PL has led to a significant increase in soy exports to China (83.2%) and other non-European Union (EU) countries (71.3%), while exports to the EU stagnated. Despite the more forest-benign agricultural production, exporters thus remained reluctant to source additional soy for the EU from regions with a history of high illegal deforestation rates and corresponding greenhouse gas emissions as well as biodiversity loss.

By showing how farmers and downstream actors in international agricultural commodity value chains react to enhanced public environmental scrutiny, the present study contributes to several strains of the literature. First, it improves our understanding of how global trade flows respond to government regulations (Gardner et al., 2019; le Polain de Waroux et al., 2019) and rising societal sustainability concerns (Shao and Ünal, 2019; Asioli, Aschemann-Witzel, and Nayga, 2020). Second, we provide new knowledge for the design of incentives that encourage traders to develop voluntary sustainability standards (DeFries, Fanzo, et al., 2017) and agree to zero-deforestation commitments (ZDC) (Gibbs, Rausch, et al., 2015; Gibbs, Munger, et al., 2016; Heilmayr et al., 2020; Levy et al., 2023). Third, we expand the literature on public disclosure strategies to reduce public bads, such as corruption (Jacobs and Anechiarico, 1992; Wehner and de Renzio, 2013), air pollution (Tietenberg, 1998; Zhou, Cao, and Feng, 2021), tax fraud (Rusina, 2020) and human right abuses (Hendrix and Wong, 2012; Ausderan, 2014) by focusing on first- and second-order effects of "naming and shaming" in the context of the bioeconomy.

The remainder of this paper is structured as follows: Section 2.2 provides more background on Brazil's forest conservation history and introduces the PL policy. Section 2.3 presents the data, while Section 2.4 outlines the empirical strategy. Section 2.5 presents the results, Section 2.6 discusses, and Section 2.7 concludes.

2.2 Institutional and theoretical background

2.2.1 Brazil's forest conservation efforts and the Priority List

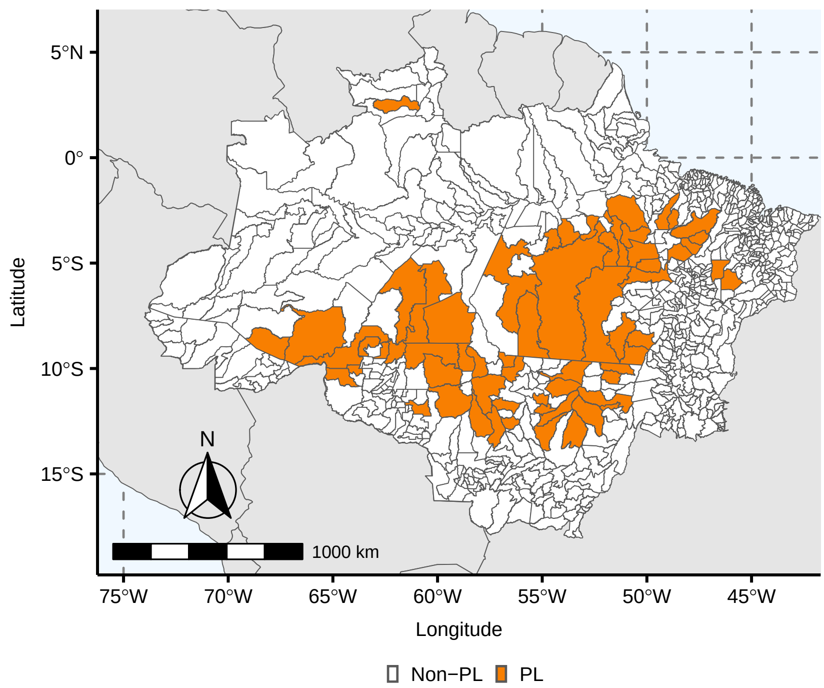
Deforestation rates in the Brazilian Amazon peaked in the late 1990s and early 2000s. Nepstad, McGrath, et al. (2014) referred to this period as the agro-industrial expansion, when cattle and cash-crops, induced by global demand and innovations such as genetic modification, started expanding into the Amazon forest. Correspondingly, soy and cattle production were identified as the main drivers of deforestation in Brazil (Nepstad, McGrath, et al., 2014; Rajão et al., 2020).

Forest conservation is regulated by Brazil's Forest Code (FC), which requires private properties in the Brazilian Legal Amazon (BLA) to reserve a minimum of 80% (50% before 1996) as natural forest. However, the rigorous enforcement of the FC only started in 2004, when an incoming federal government launched the Action Plan for the Prevention and Control of Deforestation in the Legal Amazon, introducing a series of innovative environmental governance measures. An additional 709 thousand km^2 of forests were designated as conservation areas and indigenous territories between 2002 and 2009 (Soares-Filho, Moutinho, et al., 2010). A new satellite-based real-time deforestation detection system enabled the environmental enforcement agency to systematically detect infractions, identify landholders and issue fines. In 2008, the Brazilian Central Bank restricted credit access to farmers in non-compliance with the FC. At the same time a unified geo-referenced Rural Environmental Registry (CAR) was introduced to enable environmental regularization and enforcement.

Also in 2008, Brazil's government introduced the PL—a list of priority municipalities with high deforestation rates—aiming to enhance compliance with the existing conservation policies (Brazilian Ministry of the Environment, MMA, 2021). The three major criteria for being included in the PL were: (i) high deforestation rates during the previous three years, (ii) the total amount of deforested area and (iii) forest loss increases in three out of five recent years. Its first version contained

35 entries. Eight additional municipalities were listed in 2009, another seven in 2011, two in 2012, eight more in 2017, two in 2018, and eight in 2021 (cf. Appendix Figure 2.A.1). PL municipalities faced intensified monitoring and environmental law enforcement, as well as a stricter implementation of environmental regulations (Cisneros et al., 2015; Assunção and Rocha, 2019). Municipalities have the opportunity to be removed from the PL if 80% of the eligible area is registered in CAR and annual forest losses remain below 40 km^2. Collectively, these measures should not only control deforestation, but also promote a transformation towards more sustainable agricultural production at local level (Neves and Whately, 2016). Figure 2.1 shows a map with the location of PL municipalities during our study period.

Figure 2.1. Map of Priority List municipalities



These public policy measures were bolstered by civil society initiatives and private sector actions in the mid-2000s. In 2006, Greenpeace published a report linking large soy traders to deforestation in the Brazilian Amazon. These traders, with a 90% market share, then agreed to a ZDC called the Soy Moratorium, which was the first of its kind in the tropics. While close to 30% of soy expansion occurred at the expense of natural vegetation in 2004, forest to soy conversion dropped to 1% by 2014 in the Amazon biome (Gibbs, Rausch, et al., 2015). In 2009, state prosecutors and non-governmental organizations (NGOs) in the state of Pará pressured beef retailers and slaughterhouses owning meatpacking companies into a ZDC. Facing litigation threats, retailers boycotted slaughterhouses that continued processing beef connected to illegal deforestation in the Amazon. The four largest meatpacking companies in Brazil later signed a ZDC with Greenpeace to stop purchasing cattle from properties in the Amazon biome with deforestation history after the agreement (Gibbs, Munger, et al., 2016).

2.2.2 Forest conservation, commodity sourcing and demand

Brazil's PL has shown to reduce deforestation significantly. Arima, Barreto, et al. (2014) analyzed the PL 's effect on deforestation and avoided carbon emissions for the first three years (2009–2011) after its introduction. Using a difference-in-differences approach, they calculated up to 10,653 km^2 and 1.44×10^{-1} petagrams of carbon of avoided deforestation and emissions, respectively. Cisneros et al. (2015) advanced the understanding of the PL by focusing on three potential mechanisms: (i) administrative disincentives imposed by the list that increase the transaction costs of legal deforestation, (ii) reputational risk for all stakeholders in the municipality due to public "naming and shaming" and (iii) external support or pressure, for example, such as by supportive NGOs or law enforcement agencies, respectively. The administrative disincentives may have played a minor role in promoting additional forest conservation, as most deforestation was illegal and would not have been affected by additional authorization requirements. There is evidence that the other two potential mechanisms—reputational risk and external support/pressure—played a more significant role in reducing deforestation. However, when isolating the mechanisms behind the external support/pressure impact channel, Cisneros et al. (2015) found no relevant contribution to the overall effect (on average 4,022 km^2 avoided deforestation) of the PL (between 2008 and 2012). This suggests that reputational risks remain as a key driver behind the PL-induced conservation effect.

In contrast to Cisneros et al. (2015), Assunção and Rocha (2019) argued that external pressure, in the form of command and control measures, is responsible for the deforestation reduction. Using a difference-in-differences approach, they estimated a deforestation reduction of 11,218 km^2 between 2008 and 2011. They found no effects on agricultural gross domestic product (GDP) or crop production, but a significant increase in the number of fines, which could have been an important mechanism behind the external pressure impact channel.

With the introduction of the PL, priority municipalities received intensified regulatory monitoring and enforcement. The increase in the number of fines, along with embargoes and rural credit restrictions, increased the costs of agricultural expansion through illegal land clearing. In fact, these costs were so high that agribusiness successfully lobbied for revisions of the FC and, in 2012, an amnesty was granted for illegal deforestation that occurred before 2008 (Nepstad, McGrath, et al., 2014; Soares-Filho, Rajão, et al., 2014). Still, the PL and similar disclosure measures at the level of individual stakeholders implied that local actors were increasingly exposed to public scrutiny. The sensitivity of producers and traders to such exposure may differ depending on their deforestation history, international trade relationships, and related individual or company-specific levels of risk aversion.

Soy is the most traded agricultural commodity worldwide (Chatham House, 2021). Brazil and the United States (US) are by far the largest producers and exporters (M. Lima et al., 2019), making soy one of Brazil's main export goods (Brazilian Ministry of the Economy, ME, 2021). The PL 's effect on soy production and exports has not been analyzed before. Assunção and Rocha (2019) estimated its effect on agricultural GDP and overall crop production and found no significant effects. Koch et al. (2019) further disaggregated agricultural production into cattle, dairy, and crop productivity. Using matching and a difference-in-differences approach, they found that the number of cattle heads increased significantly in priority municipalities, but they found no effects on pasture area or crop and dairy production. Lastly, Sills et al. (2020) examined a state-level policy response to the PL—the Green Municipality Program (PMV)—in the state of Pará. This program aimed to get municipalities off the PL by supporting efforts to comply with off-listing requirements. Apart from compliance, the PMV encouraged intensifying cattle production as a means of enabling the long-term sustainability of agricultural production in PL municipalities. An econometric analysis showed that the program increased total value added in PMV-enrolled municipalities, but not via agricultural intensification. The authors hypothesized that this could be due to reductions in compliance risks and costs, which led to increased economic investments. Indeed, dos Santos Massoca and Brondízio (2022) explained how stakeholder cooperation in PL municipalities can drastically reduce CAR registration costs. Furthermore, dos Santos Massoca and Brondízio (2022) showed that most soy-dominated PL municipalities were successful in complying with off-listing requirements and attribute this to strong economic incentives.

The PL policy can be understood as a cross-compliance mechanism that enhances the effect of environmental law enforcement by public disclosure. Thus, it may provoke diverse reactions by producers, traders, and, potentially, also consumers. Historically, deforestation in the Amazon has been primarily driven by pasture expansion. However, Bowman et al. (2012) argued that, given the low productivity and profitability of cattle ranching in the Amazon, pasture expansion has been a form of acquiring land tenure for future land speculation. Koch et al. (2019) showed formally how an expected increase in fining activity against illegal land clearing and land regularization through CAR may lead farmers to abandon speculative cattle expansion in favor of agricultural intensification. Therefore, intensification could be driven by the increased costs of illegal expansion, which allocates constrained capital more towards existing pastures. A constrained expansion also implies changes in relative returns to alternative land uses on land already under production. Nepstad, McGrath, et al. (2014, p. 1118) called soy "the most profitable Amazon land use”. Thus, pasture-to-soy conversion would often represent a profitable intensification option. Correspondingly, we expect that an effective PL will divert the expansion of soy on natural forests to previously cleared and less profitable land uses, such as the abundant and extensively used pastures in the region.

Zu Ermgassen, Ayre, et al. (2020) corroborated the notion that many traders and importing countries care about the conditions of production in their international supply chains. They summarized ZDCs by different actors along the soy supply chain and showed that major soy traders had already agreed to a ZDC in the Amazon biome in 2006, while, from 2014 onward, individual companies further committed to eradicate deforestation completely from their supply chain. National governments, especially in the EU, also committed to eliminate deforestation along commodity supply chains by 2020 in the Amsterdam Declaration. This declaration by importing countries’ national governments indicates increased consumer awareness, which is also expressed in the growing popularity of voluntary certification schemes, such as the Roundtable on Sustainable Palm Oil or Fairtrade (DeFries, Fanzo, et al., 2017). Reis et al. (2020) analyzed the sourcing dynamics of soy traders in Brazil between 2003 and 2017 and found that some large traders are geographically “stickier” than others, which may imply high exposure to deforestation and related reputational risk. This is understandable given that many trading companies make substantial investments in processing and transport infrastructure, such as silos, crushing facilities, and ports. Reis et al. (2020) conjectured that stickier traders could use their trusted relationships to move producers towards more sustainable production. Conversely, forest conservation policies and ZDCs might become less effective if sourcing became less sticky and environmentally responsible traders were replaced by opportunistic competitors.

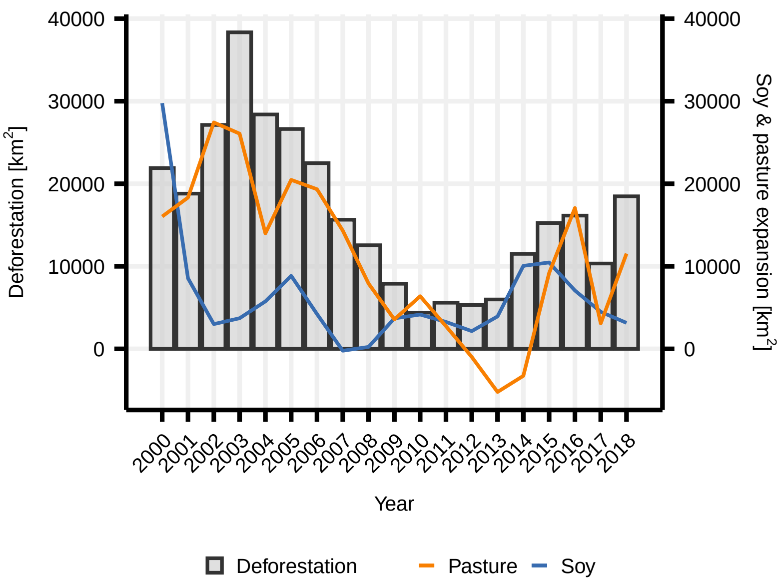
On the demand side, we expect consumers in democratic and industrialized countries with active civil societies, such as in the EU, to be more sensitive to news about agricultural commodities associated with deforestation than consumers under authoritarian regimes, such as in China. Accordingly, the PL could result in a shift of export volumes from the EU destinations to China. On the other hand, consumers in both destinations rarely consume Brazilian soy directly, as it is mostly used as an intermediary good, such as feed for cattle or pigs (Smaling et al., 2008). This likely puts a limit on consumer awareness vis-à-vis directly consumed agricultural commodities, such as coffee or cocoa. Lastly, the response of soy traders to the PL policy is particularly difficult to anticipate. Large international trading companies could be more worried about reputational risks when dealing with soy from PL municipalities than smaller competitors and could start sourcing soy from other municipalities. However, Reis et al. (2020) found that large traders are comparatively sticky and thus potentially instrumental to enact local transformation to more sustainable production practices. In the short term, however, these sticky traders would be linked to soy from high deforestation origins. As the PL policy was only introduced after the Soy Moratorium, many traders could also have already stopped sourcing soy from areas subject to high deforestation pressure.

2.3 Data

We use Brazilian municipalities as unit of analysis because they are the target of the PL as well as the spatial unit to which commodity trade is traced to inform consumers and policy makers about the level of deforestation risk associated with soy production. Out of 5,573 municipalities in Brazil, we focus on 770 BLA municipalities, which had at least 10% forest cover in 2000. For our analysis, we compose a unique panel data set, covering 15 years from 2004 to 2018, based on remotely sensed annual land-cover estimates and soy trade statistics. The resulting spatio-temporal variation in outcomes allows us to identify the differential effect of the PL policy on land conversion and export patterns.

Annual data of remotely sensed land use and land cover change per municipality was obtained from MapBiomas (2021). Figure 2.2 shows annual rates of newly deforested areas, soy and pasture expansion in km^2 for the BLA from 2000–2018. Deforestation rates decreased from the early 2000s until 2010 and gradually increased since then. Pasture expansion, the main direct cause of land cover change in the region, followed deforestation rates very closely, with the exception of 2012–2014 when pastures contracted. Soy expansion also followed deforestation, but less expressively.

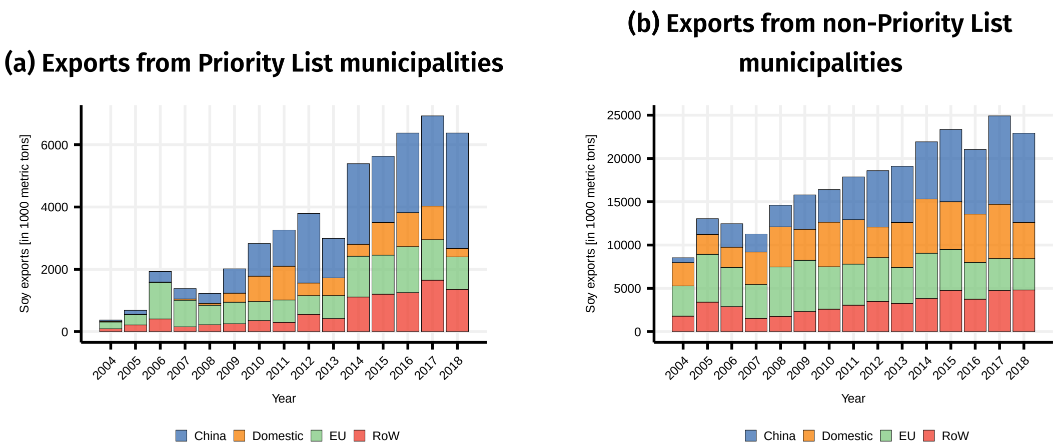
Figure 2.2. Forest loss and soy & pasture expansion in the Brazilian Legal Amazon



Note: Based on authors' own calculations using MapBiomas (2021) 5.0 land cover data.

Supply chain data on soy exports, including information on production origin, quantity exported, importer and exporter group as well as the destination country was retrieved from Trase (Lathuillière, Suavet, et al., 2022). Export statistics are reported for soy beans, soy bean oil, and soy bean cake. Lathuillière, Suavet, et al. (2022) recalculate each quantity to soy-equivalent tons and reports the total exported sum.[1] Figure 2.3 shows the total soy quantity in thousands tons exported from the BLA by destination over time. We differentiate between exports from PL municipalities in Figure 2.3a and exports from BLA municipalities that were never on the PL and serve as the donor pool for the GSC method (Figure 2.3b). The overall development is similar in both figures. Exports to the EU remain constant, while exports to China and the rest of the world (RoW) are generally increasing over time. In 2018, exports to China from the entire BLA amounted to over 14 million tons, whereas only 4.7 and 6.2 million tons went to the EU and the RoW, respectively.

Figure 2.3. Soy exports from the Brazilian Legal Amazon by destination over time



Note: Based on Lathuillière, Suavet, et al. (2022) data using total aggregates by year. RoW stands for 'rest of the world', i.e., all other countries combined.

Control variables include potential confounders, such as context-specific time trends based on location specific soil suitability to grow soy and transportation costs. We use maps of Global Agro-Ecological Zones from the Food and Agricultural Organization (FAO) to calculate a soil suitability index to produce soy for each municipality (FAO/IIASA, 2012). Commodity transport costs at the municipality level are from Victoria et al. (2021).

1. Soy products remaining in Brazil are declared as domestic consumption and are not connected to a trader group. Missing municipality-year entries in the database are treated as zero exports.

2.4 Empirical strategy

There are several challenges in estimating the causal effect of the PL on soy-related outcomes. First, it is difficult to find valid counterfactuals for municipalities that were selected from a list of top deforesters. Second, it is difficult to ensure parallel pre-treatment trends for a variety of outcomes, which is a necessary pre-condition for using panel data estimators. Third, the PL had a staggered introduction, starting with 35 municipalities in 2008 and extending to 62 by 2018. Combining matching with panel data estimators like fixed effects is known to produce good results in the evaluation of public policies using observational data (Ferraro and J. J. Miranda, 2017). However, sequential treatment prevents the identification of a clear pre-treatment period for matching. Traditional two-way fixed effects estimations have been shown to be biased under heterogeneous treatment effects (de Chaisemartin and D'Haultfœuille, 2020; Callaway and P. H. Sant'Anna, 2021; Goodman-Bacon, 2021; Sun and Abraham, 2021; Borusyak et al., 2024).

Therefore, our identification strategy builds on the GSC method developed by Xu (2017). This approach has several advantages. The underlying interactive fixed effects (IFE) model estimates unobserved time-varying confounders called latent factors and unit-specific intercepts, which relaxes the common parallel trends assumption of difference-in-differences models and the additive nature of classical fixed effects. Secondly, it generalizes the common synthetic control method (Abadie et al., 2010) by allowing for several treated units and different instances of treatment inception. The IFE is defined as:

$$Y_{i,t} \;=\; \delta_{i,t} B_{i,t} + t \cdot \boldsymbol{Z}_{i0}\, \theta_{i,t} + \lambda_i' f_t + \alpha_i + \xi_t + \epsilon_{i,t} \tag{2.1}$$

where $Y_{i,t}$ denotes the inverted hyperbolic sine of our outcome variables (deforestation, soy and pasture expansion, and soy exports) for municipality i at time t.[2] The treatment indicator $B_{i,t}$ equals 1 for all years t after being PL in year T_0 and $\delta_{i,t}$ are the heterogeneous municipality-year specific treatment effects. $\boldsymbol{Z}_{i0}$ denotes time-invariant covariates that are interacted with a linear trend, t. Since the PL

2. The inverse hyperbolic sine has the advantage of being defined at zero and yielding near-zero positive values for very small values, while allowing for interpreting small coefficients as percentage changes similarly to a log transformation. However, Bellemare and Wichman (2020) noted that the presence of zero and near-zero values in the data can lead to biased elasticity estimates. Therefore, we followed Bellemare and Wichman's advice and calculated the elasticity using their exact formulation by retransforming predicted outcomes using the hyperbolic sine function and applying the equation:

$$\frac{\hat{\hat{\delta}}_{i,t}}{100} = \frac{\hat{Y}_{i,t}(1)}{\hat{Y}_{i,t}(0)} - 1 \tag{2.2}$$

where $\hat{\hat{\delta}}_{i,t}$ is the estimated exact treatment effect as a percentage change.

municipalities are the ones with most cleared lands, they also bear the largest potential to expand soy area and production. We control for the potential to convert forest for soy production and becoming PL by controlling for initial conditions in $\boldsymbol{Z}_{i0}$. As initial conditions we use the relative amount of deforestation in three years prior to treatment, the share of forest cover in 2000, soy soil suitability and commodity transport costs. f_t' is a $r \times 1$ vector of unobserved common factors and λ_i is a $r \times 1$ vector of unknown factor loadings. The factor components take a linear additive form with r elements: $\lambda_i' f_t = \lambda_{i1} f_{1t} + \ldots + \lambda_{ir} f_{rt}$ which will serve to approximate pre-treatment trends of the treated units. Finally, we include municipality and time fixed effects, α_i and ξ_t, respectively. The idiosyncratic error term is denoted by $\epsilon_{i,t}$. Given the potential outcomes $Y_{i,t}(1)$ and $Y_{i,t}(0)$, the annual average treatment effect on the treated can be defined as the difference between the realized outcomes of the treated units and their counterfactuals (Xu, 2017):

$$ATT_{t,t>T_0} = \frac{1}{N_{treated}} \sum_i [\, Y_{i,t}(1) - Y_{i,t}(0) \,] = \frac{1}{N_{treated}} \sum_i \delta_{i,t} \tag{2.3}$$

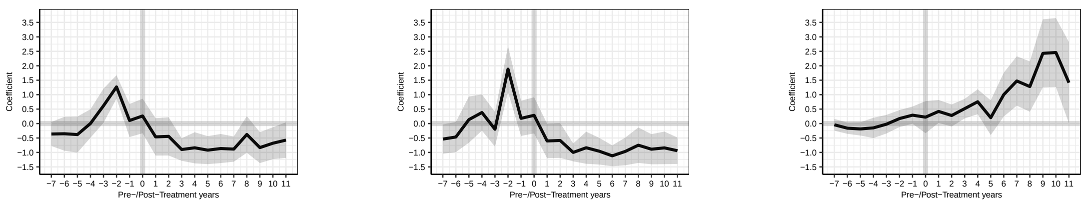
To calculate the yearly average treatment effect on the treated (ATT), the GSC method models the counterfactual outcomes in three steps. First, a reduced form of the IFE model (Equation 2.1) is estimated using only data from the control group. Following Bai's (2009) estimation methodology, this provides estimates of $\hat{\beta}$, $\hat{\theta}$, $\hat{\alpha}$ and $\hat{\xi}_t$, as well as the factor loadings, $\hat{f}_t$, and common factors of the controls, $\hat{\lambda}_i$. Provided these estimates, the second step consists of estimating the common factors and unit fixed effects of the treated units by minimizing the mean squared errors of the predicted treated outcome in pre-treatment years. The third step uses all estimates to forecast the counterfactual outcomes in treated years. The optimal number of common factor components, r, are derived by a leave-one-out-cross-validation procedure and choosing the model that minimizes the mean squared prediction error. Standard errors are bootstrapped and clustered at the municipality level. The GSC model is estimated using Xu's `gsynth` package in `R` (R Core Team, 2021).

2.5 Results

GSC estimates are presented in Figures 2.4–2.6 as gap plots, where the line represents the estimated difference between treated municipalities and the counterfactual synthetic controls, i.e., the yearly ATT. 95% confidence intervals are depicted as shaded gray area. Appendix Figures 2.A.5–2.A.7 show the average outcome paths of treated and synthetic controls used to estimate the gap plots. We first re-estimate the effects of PL on deforestation to establish a baseline to the literature (cf. Arima, Barreto, et al., 2014; Cisneros et al., 2015; Assunção and Rocha, 2019; Koch et al., 2019). Figure 2.4a shows a statistically significant pre-program difference between the synthetic control and the treated units two years prior to treatment. This points to a selection of high deforesting municipalities into treatment and a potential overestimation of the treatment effects (Ashenfelter, 1978; Heckman and J. A. Smith, 1999). In post-treatment years, impact estimates turn negative and significant two years after treatment. On average, deforestation rates decreased by 70.2% after being on the PL.[3] This corresponds to an avoided forest loss of 23,816 km^2 (± 3,691 km^2) between 2008 and 2018. Pasture expansion decelerated on average by 85.1% (Figure 2.4b), which translates to a total of 14,494 km^2 prevented pasture expansion. In contrast, soy expansion (Figure 2.4c) increased further in PL municipalities. On average 103.4%, or a total of 739.8 km^2 of additional soy area in comparison to non-PL municipalities.

Figure 2.4. The effects of the Priority List on deforestation and land use

(a) Forest losses **(b) Pasture expansion** **(c) Soy expansion**



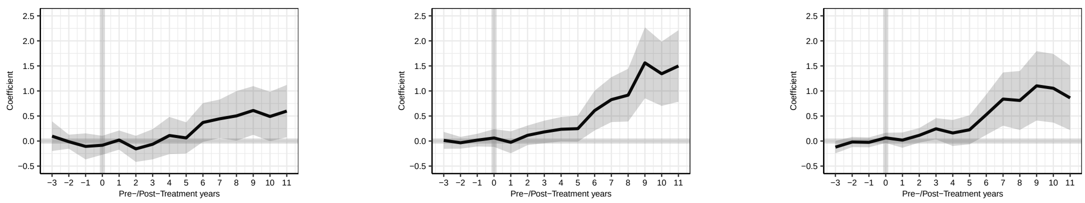
To better understand the source and type of soy area expansion we use annual MapBiomas (2021) land cover data to track land-use changes from year to year. We differentiate between soy expansion located on previous forest, pasture, or other cropland and re-estimate the PL 's effect on these sub-types. Figure 2.5 shows the results of the GSC estimates. While soy expanded significantly on all types of previous land use, effects are largest for soy replacing pastures and other crops. Expansion increased by 61.1% on prior pastures and 49.7% on other cropland (or 237.6 km^2 and 214.5 km^2 between 2008 and 2018, respectively). Expansion on prior forests is statistically significant—a 24.1% increase—but economically negligible with a total of 5.9 km^2. These effects are driven by accelerated soy expansion in PL municipalities, above the general increasing soy expansion trend in the BLA (cf. Appendix Figure 2.A.6 depicting trends over time).

Decreasing deforestation and a simultaneous soy expansion on non-forest areas points towards a forest-benign agricultural trend shift in PL municipalities. This shift could have been due to the public disclosure effect, that is, a strategic response by local producers and their trading partners to secure their share on in-

3. The GSC method estimates a synthetic control for each treated municipality. The average treatment effect is therefore calculated as in Equation 2.3, while considering the total number of years on the PL for each municipality and calculating the exact percentage change following Equation 2.2.

Figure 2.5. Effects on soy expansion by previous land use

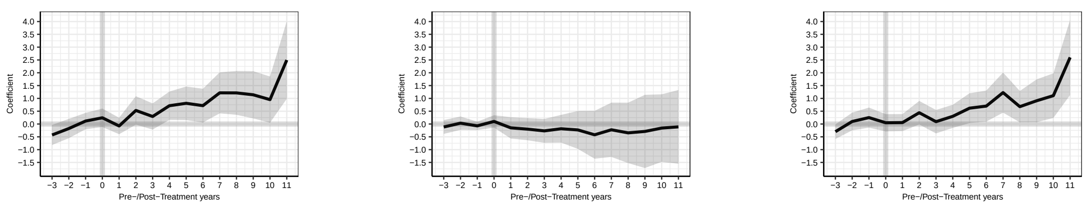
(a) Forest to soy **(b) Pasture to soy** **(c) Other crops to soy**



ternational markets with higher preference for sustainable soy, such as in the EU. Figure 2.6 shows the yearly effects of the PL on soy exports patterns by destination. The results show significant increases in soy exports to China (Figure 2.6a) and to the RoW (Figure 2.6c), by 83.2% (or 237.5 thousand tons for the 2008–2018 period) and 71.3% (115.8 thousand tons), respectively. In contrast, soy exports to the EU remained unaffected (Figure 2.6b).[4] In combination with the findings on land-use change above, these post-disclosure trade patterns suggest that PL may have had tangible impacts at both ends of the soy value chain. First, producers gradually abandoned forest-to-soy conversion, arguably in response to increased public scrutiny at both domestic and international levels. Second, EU markets were largely excluded from the post-disclosure surge in soy exports from PL municipalities, even though expansion has arguably become more sustainable.

Figure 2.6. Effects on soy exports by destination

(a) Exports to China **(b) Exports to the EU** **(c) Exports to the RoW**



2.6 Discussion

We analyze the impact of the Brazilian Amazon deforestation PL on local deforestation, pastures and soy expansion, and global soy export patterns. Our evaluation approach uses a new land-use data source (MapBiomas, 2021) and an innovative

4. Appendix Figure 2.A.4 checks for a potential reorientation from export to domestic markets in PL municipalities, but shows no significant impact.

estimation method, the GSC approach. Our estimation suggests an average annual avoided forest loss of 2,165 km^2 between 2008 and 2018. Cisneros et al. (2015) estimated an average annual reduction in deforestation of 800 km^2 between 2008 and 2012. Arima, Barreto, et al. (2014) calculated an average annual reduction between 580 and 2,660 km^2 for the period 2008–2011, while Assunção and Rocha (2019) estimated a larger effect of 2,850 km^2 avoided deforestation over the same time period. Koch et al. (2019) again estimated smaller quantities of 660 to 740 km^2 between 2008 and 2014. Thus, our estimates are at the higher end of the distribution, but we also cover a longer post-treatment period with additional treated municipalities. Our estimates show that effects materialized only slowly, with the highest impacts after 3–7 years of treatment (cf. Figure 2.4). Nonetheless, our estimates are likely overestimated due to unusually high deforestation rates in pre-treatment years.

The reduction in deforestation was driven chiefly by a reduction in pasture expansion (cf. Figure 2.4). The total prevented pasture expansion due to the PL corresponds to 60.9% of total avoided deforestation. While pastures and cattle ranching boast low profitability per hectare, they require little capital and have historically been used to claim ownership for land speculation (Bowman et al., 2012). The PL arguably increased the expected costs (including reputational risks) of agricultural expansion into natural vegetation, thus raising the opportunity costs of extensively used pastures vis-à-vis pasture-to-soy conversion (Nepstad, McGrath, et al., 2014). Our results show that post-disclosure soy expansion occurred predominantly on pastures and other crops (cf. Figure 2.5). The significant, but small, amount of forest-to-soy expansion consists of only around 1% of the overall expansion effect. Previous results in Assunção and Rocha (2019) and Koch et al. (2019) found no evidence for a PL effect on crop production. However, both papers relied on aggregate crop production estimates by the national statistics institute, which do not allow for a spatially explicit differentiation between different crops and land cover changes. Our analysis is the first to use remotely sensed land cover data and detailed export statistics to show that the PL led to a considerable land-use intensification through pasture-to-soy conversion.

Arguably, the dominant form of intensification via pasture-to-soy conversion could have led to a re-allocation of pastures and thus indirectly cause deforestation elsewhere (Arima, Richards, et al., 2011). Indirect land-use change or leakage has been observed in other area-based conservation policies (cf. Meyfroidt et al., 2020). This would violate the stable unit treatment value assumption, as the treatment effect identification hinges on the assumption of the treatment not having an effect on control units. Appendix Figure 2.A.5b shows that pasture expansion decreased in general, also outside of PL municipalities. Moreover, Cisneros et al. (2015) explicitly tested for leakage effects of the PL and found no effect on deforestation in neighboring municipalities. Hence, we do not expect that the PL has triggered additional forest-to-pasture conversion in the region.

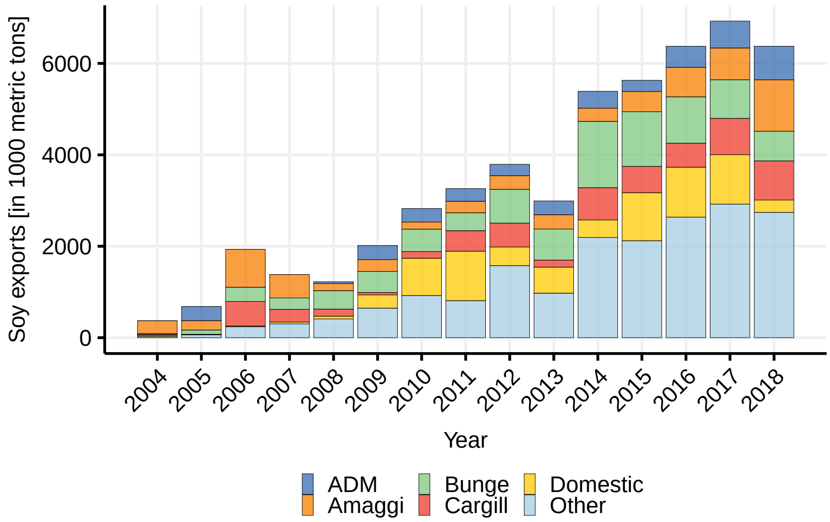
Regarding soy exports, we find an increase in exports to China and non-EU countries, while exports to the EU stagnated. Over the 2008–2018 period, PL municipalities exported an additional 237,500 and 115,800 tons of soy to China and non-EU countries, respectively, compared to similar non-PL municipalities. Despite continued demand for sustainably produced protein and energy crops, the EU 's share in global soy trade has declined in recent years. Instead, China has become the dominant buyer of Brazilian soy, importing 55.5% of total production, while the EU only imports 10.7% (cf. Appendix Figure 2.A.3). Arguably, Chinese consumers and importers are less sensitive to information about deforestation associated with their soy imports than their European counterparts. Hence, even if the Brazilian agribusiness may have duly responded to the additional public pressure imposed by the PL, non-European markets may have been the only viable target to accommodate their response to increasing global commodity demand.

A closer look at the sourcing patterns of the large international trading companies who export to multiple global destinations may help to complete the picture. Figure 2.7 shows total soy exported by the main global traders that bought from PL municipalities during the last 15 years. Although an impact evaluation at this level of data disaggregation is not feasible, there are notable patterns. Over the course of our study period, the share of domestically consumed soy sourced from PL municipalities increased, as did the share sourced by other, mainly Asian traders such as Cofco and Mitsui. This could reflect shifts in sourcing preferences in some of the traditional destinations of soy produced in the Amazon region, such as in the EU (Escobar et al., 2020).

Considering Cisneros, Zhou, and Börner's (2015) three proposed PL impact channels—administrative disincentives, reputational risks and regulatory enforcement—reputational risk likely drives traders' decisions on sourcing locations and export destinations. Sticky traders may continue to source soy for European markets, but without expanding business in municipalities with historically high deforestation records so as to avoid additional reputational risk exposure. Other traders may be less selective with respect to sourcing locations, especially exporting to non-EU markets, such as in China. Land use and deforestation decision, on the other hand, are made by producers. In response to the PL, producers reduced illegal deforestation and legally expanded soy cultivation on pastures and other cropland. This intensification strategy aligns with both the reputational risk and the enforcement mechanism hypotheses (Cisneros et al., 2015).

Figure 2.7 also shows how sparse the data basis for soy exports in the Amazon during the early 2000s is. Most Amazon municipalities, including PL ones, did not yet produce or export much soy before 2008. It is difficult to find good matches when many municipalities have almost no soy production before 2008, and this is also a reason why we selected the GSC method, which constructs counterfactuals by a weighted average of multiple control group municipalities, instead of, for example, a 1:1 nearest-neighbor matching.

Figure 2.7. Total soy exports from Priority List municipalities by trader



Note: Based on aggregation of soy exports from Priority List municipalities including domestic consumption using Lathuillière, Suavet, et al. (2022) data.

Overall, the Brazilian PL added an innovative instrument to the conservation toolbox. Instead of creating new laws or administrative structures, it relied on the existing institutional framework to leverage the local governance system in a spatially targeted fashion. Private sector ZDCs could act in a complementary way, but only if they cover relevant market shares (Levy et al., 2023). Otherwise, flexible commodity markets are likely to adapt, such that commodity flows associated with high deforestation risk are redirected to export destinations with low levels of due diligence. Improved supply-chain transparency in such market segments will then arguably become less likely to leverage more sustainable production patterns.

This caveat of private ZDCs also applies to public due diligence policies, such as the recently adopted regulation to reduce imported deforestation in the EU Deforestation Regulation (EUDR). The EUDR will require importers to invest in additional transparency and reporting obligations and involve the establishment of new public monitoring and enforcement capacities (Halleux, 2022). The current EUDR framework relies on a classification system for regions with low versus high risk of deforestation. This system could, in principle, be operated like Brazil's PL. Clear entry and exit rules, informed by remote-sensing based deforestation monitoring, could incentivize jurisdictions to take targeted action towards reducing the risk of being temporarily excluded from access to EU markets.

However, as long as global commodity trade is only partially under the scrutiny of binding due diligence regulations, national environmental policy remains the only effective means to end illegal deforestation. Cost-effective and spatially targeted policy instruments like the Brazilian PL can contribute towards this goal.

2.7 Conclusion

This study provides new insights into the impacts of the deforestation PL, Brazil's public disclosure policy to reduce legal and illegal deforestation in the Amazon region. We confirm the findings of earlier work, attributing 23,816 km^2 of avoided deforestation between 2008–2018 to the PL. Importantly, this is the first study using remotely sensed land cover data to disentangle the land-use dynamics behind the effects of the PL. Deforestation reductions are mainly driven by 14,494 km^2 of prevented pasture expansion. At the same time, the expansion of intensive soy production was on average twice as high in PL municipalities than in non-PL municipalities. However, this expansion happened almost exclusively on previously cleared lands, such as in the form of pasture-to-soy conversion (237.6 km^2). Increased soy production in PL municipalities was shipped to non-EU destinations such as China and other countries (237,500 and 115,800 tons, respectively), while the market share of established exporters diminished.

Our results suggest that agribusiness is unlikely to forego business opportunities in the face of the reputational risks imposed by Brazil's PL or similar types of value chain transparency initiatives. Legal opportunities to expand soy production in PL municipalities were duly exploited and exposure to business risk was minimized by diversifying into markets that place less importance on imported deforestation than many EU destinations. As long as such markets continue to grow, it is unlikely that voluntary ZDCs or due diligence legislation imposed unilaterally by the EU will help to curb Amazon deforestation, which has been constantly rising since 2012.

Appendix 2.A Appendix to: Beyond Deforestation Reductions: Public Disclosure, Land-Use Change and Commodity Sourcing

2.A.1 Appendix figures

Figure 2.A.1. Number of Priority List municipalities over time

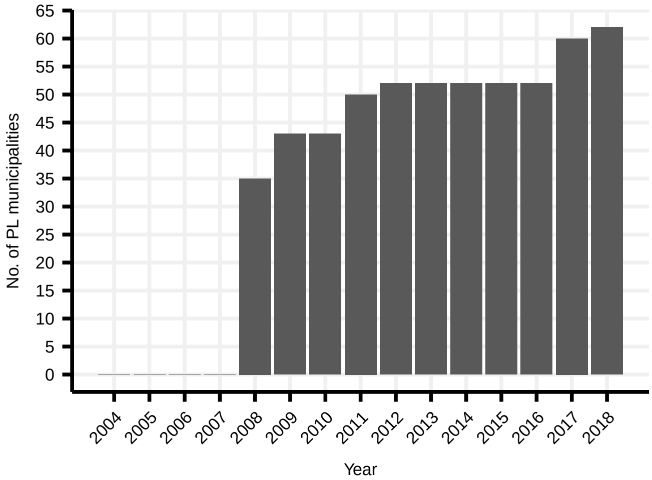
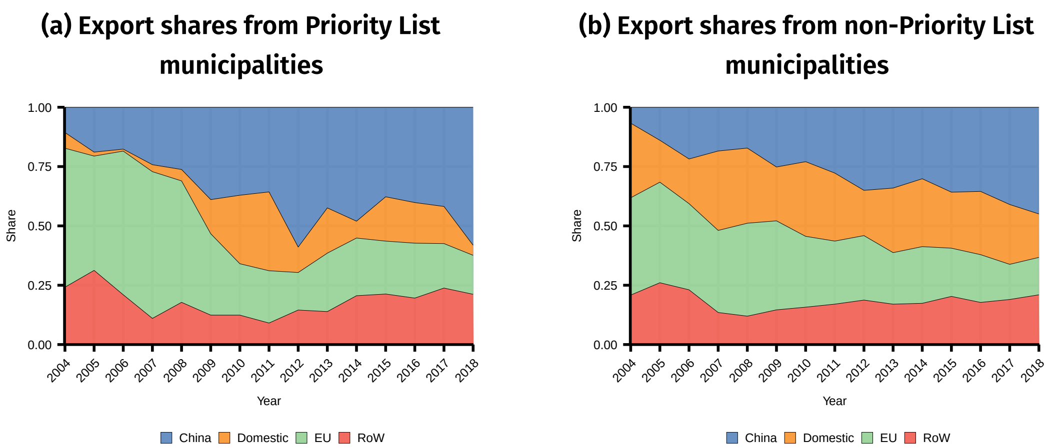


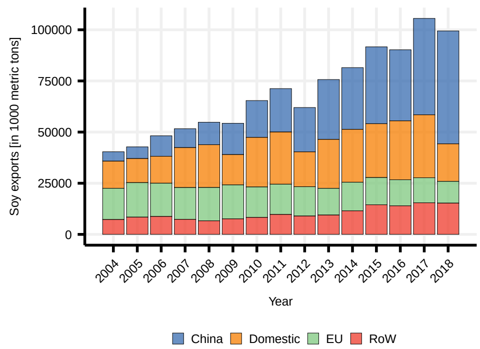
Figure 2.A.2. Soy export shares from the Brazilian Legal Amazon by destination over time



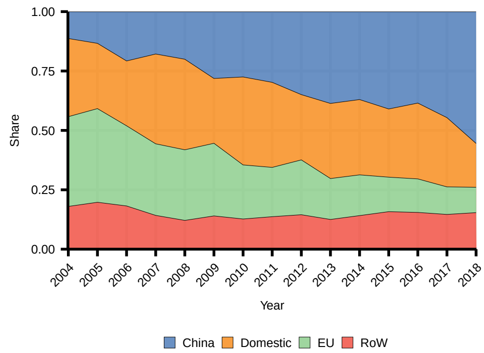
Note: Based on aggregation of soy exports from municipalities in the Brazilian Legal Amazon including domestic consumption using Lathuillière, Suavet, et al. (2022) data. RoW stands for 'rest of the world', i.e., all other countries combined.

Figure 2.A.3. Soy export from entire Brazil by destination over time

(a) Total soy exports from Brazil



(b) Soy export shares from Brazil



Note: Based on aggregation of soy exports from Brazilian municipalities including domestic consumption using Lathuillière, Suavet, et al. (2022) data. RoW stands for 'rest of the world', i.e., all other countries combined.

Figure 2.A.4. Effects on soy production for domestic consumption

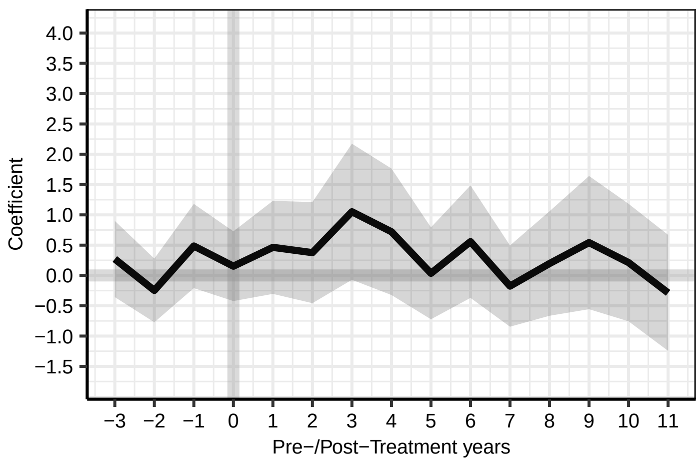
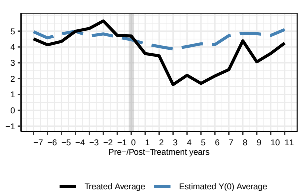
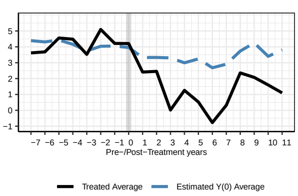


Figure 2.A.5. Deforestation and land use paths of Priority List municipalities and their estimated counterfactual

(a) Forest losses



(b) Pasture expansion



(c) Soy expansion

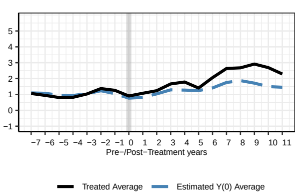
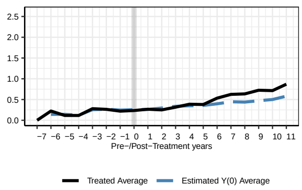
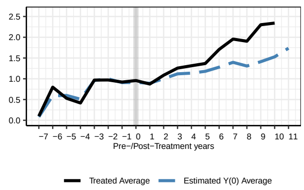


Figure 2.A.6. Soy expansion by previous land use paths of Priority List municipalities and their estimated counterfactual

(a) Forest to soy



(b) Pasture to soy



(c) Other crops to soy

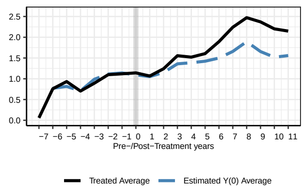
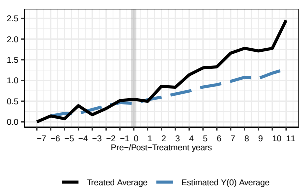
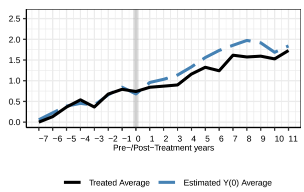


Figure 2.A.7. Soy exports by destination paths of Priority List municipalities and their estimated counterfactual

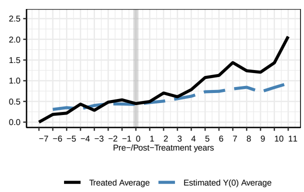
(a) Exports to China



(b) Exports to the EU



(c) Exports to the RoW



Chapter 3

Balancing Growth and Conservation: The Economic Impact of Environmental Enforcement in the Brazilian Amazon*

Joint with Gustavo Magalhães de Oliveira and Jan Börner

Abstract

This paper examines the causal economic impacts of environmental enforcement in the Brazilian Amazon between 2002 and 2016. Using an instrumental variable strategy that exploits exogenous variation in satellite monitoring capacity from cloud cover, we show that stricter enforcement by the federal environmental agency IBAMA significantly reduces deforestation and pasture expansion while shifting agriculture towards soy cultivation. These land-use changes are accompanied by higher soy sector employment, wages, and productivity, without measurable losses in agricultural GDP. However, enforcement reduces the number of rural credit contracts and leads to a notable contraction in municipal public administration jobs. The results indicate that well-targeted enforcement can redirect economic activity towards more intensive and sustainable production without sacrificing output, but complementary measures, such as tailored credit access and technical support are important to address adjustment costs and ensure inclusive gains.

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JEL Classification: F18, Q17, Q56, Q58
Keywords: Brazilian Amazon, Deforestation, Environmental enforcement, Land-use change, Labor markets

3.1 Introduction

Economic growth and development in resource-rich regions such as the Brazilian Amazon often relies on natural resource exploitation, including agriculture, mining, or logging (Bebbington et al., 2018). While these activities generate income and profits for local populations, they have also fueled widespread deforestation and biodiversity loss, raising concerns regarding long-term sustainability (Laurance, Sayer, and Cassman, 2014; C. A. Nobre et al., 2016; Haddad et al., 2024). The Amazon, with its great ecological significance, exemplifies this challenge of balancing environmental conservation and economic development (Garrett, Gardner, et al., 2017; Garrett, Cammelli, et al., 2021).

Historically, government policies have incentivized economic expansion in the region, often at the expense of the natural vegetation (Fearnside, 2005). Infrastructure projects, such as the construction of the Trans-Amazonian Highway, and financial incentives for agricultural settlers have led to rapid land conversion for cattle ranching and soybean cultivation, key drivers of deforestation in the Amazon (Fearnside and de Alencastro Graça, 2006; R. Walker et al., 2011; Rajão et al., 2020; R. Araujo et al., 2025). In an attempt to control the environmental consequences of uncontrolled agricultural expansion, Brazil implemented a series of environmental policies in the 2000s, subsumed under the Action Plan for the Prevention and Control of Deforestation in the Legal Amazon (PPCDAm) in 2004. This policy contains a variety of measures to combat illegal deforestation through satellite-based monitoring and enforcement mechanisms (Assunção, Gandour, and Rocha, 2015; West and Fearnside, 2021). Although these new enforcement measures have contributed to reducing deforestation in the region (Assunção, Gandour, and Rocha, 2023), little is known about their collateral effects on local economies, and whether there are trade-offs between environmental conservation and economic development.

Environmental enforcement may influence agriculture-dependent local economies mainly by restricting the expansion of production into forested areas. At first, higher enforcement increases the costs of agricultural expansion, potentially constraining economic activity. However, land users may also respond to stricter enforcement by intensifying production on existing farmland (Koch et al., 2019; Moffette, Skidmore, et al., 2021). Hence, environmental enforcement at the forest frontier can also induce systematic shifts in land-use decisions that may ultimately generate positive economic impacts. With this in mind, this study explores whether and how environmental enforcement affects land-use change and the local economies in the Brazilian Amazon.

A major challenge in assessing the impact of environmental enforcement is the potential endogeneity of enforcement intensity. The allocation of enforcement efforts is often non-random, with more enforcement resources directed to municipalities experiencing high deforestation risks or strong political pressures. As a result, estimating the causal impact of enforcement on deforestation, land use, and economic outcomes requires addressing this selection bias. To overcome this challenge, we follow Assunção, Gandour, and Rocha (2023) and employ a two-stage least squares (2SLS) panel instrumental variable (IV) approach, using a cloud cover derivative as an exogenous instrument for enforcement intensity, which is proxied by deforestation fines issued by Brazil's environmental enforcement agency, the Brazilian Institute of Environment and Renewable Natural Resources (IBAMA). Since mid-2004, IBAMA uses a satellite-based Real-Time Deforestation Detection System (DETER) to allocate enforcement efforts. Cloud cover could obstruct satellite visibility and therefore reduce IBAMA's enforcement ability. However, cloud cover could also be related to environmental or climatic conditions that favor or disfavor deforestation and agricultural production. Our instrument is therefore based on the change in the cloud cover effect post-2004, after the introduction of DETER. Simultaneously, we also control for additional confounding factors to isolate the variation of cloud cover that is plausibly exogenous to economic and institutional factors driving deforestation.

Using a municipality-by-year panel dataset covering 537 municipalities in the Brazilian Amazon biome between 2002 and 2016, we find that increased environmental enforcement by IBAMA significantly reduces deforestation and pasture expansion, while shifting agriculture towards soy cultivation and boosting soy sector employment, wages, and productivity. These changes occur without measurable losses in agricultural gross domestic product (GDP), but they are accompanied by a reduction in the number of rural credit contracts—suggesting consolidation towards larger and more compliant borrowers—and a notable contraction in municipal public administration employment. The results indicate that stricter enforcement can redirect economic activity towards more intensive, sustainable land use without sacrificing output, although it introduces equity and capacity-related challenges that may warrant complementary policy measures.

This study contributes to the growing literature on the role of environmental governance in reconciling conservation and economic development. While previous studies have examined the effects of enforcement on deforestation reduction (Hargrave and Kis-Katos, 2012; Börner, Wunder, Wertz-Kanounnikoff, Hyman, et al., 2014; Börner, Kis-Katos, et al., 2015; Börner, Marinho, et al., 2015; Assunção, Gandour, and Rocha, 2023; Kuschnig et al., 2023; Nunes et al., 2024), few have explored their broader economic and social consequences. As deforestation remains a concern in many developing countries, understanding how enforcement efforts influence land-use and socioeconomic dynamics can inform strategies in other resource-rich regions facing similar challenges in balancing economic growth and development with environmental protection (Laurance et al., 2014; Seymour and Harris, 2019). While strict enforcement may be effective in curbing deforestation, its long-term success depends on political support and institutional factors (Kuschnig et al., 2023; Nunes et al., 2024). Understanding the economic impacts of environmental enforcement can therefore enhance local political support for forest conservation initiatives.

The remainder of this paper is structured as follows. Section 3.2 provides historical and theoretical background on deforestation and enforcement policies in the Brazilian Amazon. Section 3.3 describes the dataset and empirical approach. Section 3.4 presents the results and robustness checks. Section 3.5 discusses and contextualizes the results, and concludes with recommendations for policymakers.

3.2 Institutional and theoretical background

3.2.1 History of deforestation and economic development in the Brazilian Amazon

Economic development in the Brazilian Amazon has followed different historical phases, depending on the various current political and economic priorities. During the colonial period and the 19th-century rubber boom, the region's economy was based on resource extraction, leading to cycles of economic growth and decline (Barbanti, 2015). However, modern large-scale deforestation was primarily triggered by state-led development policies in the mid-20th century (Fearnside, 2005).

Beginning in the 1960s and 1970s, Brazil's military regime sought to integrate the Amazon into the national economy through large infrastructure projects and agricultural incentives (Pokorny et al., 2021). Policies such as the construction of the Trans-Amazonian Highway and fiscal incentives for agricultural settlers encouraged migration and land conversion, accelerating deforestation (Binswanger, 1991; R. Walker et al., 2011). These policies were driven by the intent of territorial expansion to secure national borders and promote economic growth.

By the 1990s and 2000s, deforestation in the Amazon was increasingly linked to global commodity markets, especially the expansion of soybean cultivation and cattle ranching (Nepstad, Stickler, and Almeida, 2006; Richards, R. Myers, et al., 2012). Rising international demand for Brazilian soy and beef—fueled by China's growing imports and trade liberalization—intensified land conversion pressures (le Polain de Waroux et al., 2019). Additionally, weak land tenure systems and speculative land clearing facilitated continuous deforestation, but often in violation of environmental regulations (Soares-Filho, Rajão, et al., 2014; Furumo et al., 2024).

Since then, pasture expansion has been one of the main drivers of deforestation in the Amazon (Skidmore et al., 2021; Hänggli et al., 2023). However, many deforested areas are used inefficiently, as land clearing is often driven by speculative behavior rather than agricultural productivity. Large areas are cleared not for immediate production but in anticipation of future land value appreciation, leading to unnecessary deforestation and land degradation (Bowman et al., 2012; J. Miranda et al., 2019). Moreover, while soybean cultivation has been associated with more intensive land use (Nepstad, McGrath, et al., 2014), its strong dependence on capital and infrastructure requirements may also have contributed to increased deforestation in the region.

In response to the escalating deforestation crisis, Brazil started introducing various conservation policies. As part of the PPCDAm Action Plan to combat deforestation, regulators, among others, expanded the network of protected areas, improved satellite monitoring combined with enforcement measures, and promoted sustainable development programs (Assunção, Gandour, and Rocha, 2015; West and Fearnside, 2021). A cornerstone of Brazil's conservation efforts is its environmental protection agency IBAMA, which is responsible for conducting field inspections, issuing fines, and embargoing farmers in violation of environmental regulations. The creation of DETER considerably improved IBAMA's ability to monitor and respond to ongoing deforestation (Assunção, Gandour, and Rocha, 2023).

Many studies have shown that IBAMA's increased enforcement efforts played a crucial role in reducing deforestation rates in the Amazon after 2004 (Hargrave and Kis-Katos, 2012; Börner, Wunder, Wertz-Kanounnikoff, Hyman, et al., 2014; Börner, Kis-Katos, et al., 2015; Börner, Marinho, et al., 2015; Assunção, Gandour, and Rocha, 2023). Recent empirical evidence further suggests that strong environmental monitoring and enforcement can yield substantial conservation benefits without necessarily harming the economy (Koch et al., 2019; Moffette, Skidmore, et al., 2021; Crepin, 2024; Damm, Cisneros, et al., 2024; Merkus, 2024). However, despite this growing recognition of enforcement as a key policy tool, the pathways through which it may generate broader socioeconomic impacts remain poorly understood in the literature.

3.2.2 Balancing economic growth and environmental conservation

Forests provide essential services like climate regulation, water purification, and biodiversity conservation (Costanza, d'Arge, et al., 1997; Millennium Ecosystem Assessment, 2005; Strand et al., 2018; IPBES, 2019; Taye et al., 2021; Brouwer et al., 2022), but because these benefits are not traded in markets, they are often invisible to the landowners making land-use decisions (Costanza, d'Arge, et al., 1997; Millennium Ecosystem Assessment, 2005; Costanza, de Groot, et al., 2014;

IPBES, 2019). Instead, activities like farming or mining tend to offer more immediate and tangible returns, driving deforestation beyond what would be socially optimal. As a consequence, societal welfare is lower than in an economy in which the value of forests was recognized (IPBES, 2019). This market failure has motivated policy efforts to internalize private incentives with broader environmental goals. Still, while conservation policies aim to secure long-term societal benefits, they can also limit income opportunities for rural households and communities that rely on natural resource use. Navigating this trade-off is becoming more and more urgent, especially as ecosystems such as the Amazon edge closer to critical tipping points (C. A. Nobre et al., 2016; Banerjee et al., 2022; Boulton, Lenton, and Boers, 2022).

Economic cattle farming models for the Brazilian Amazon by Koch et al. (2019) and Moffette, Skidmore, et al. (2021) propose a potential solution to this dilemma. In a context of unrestricted land-use expansion, land is relatively abundant and cheap, and farmers tend to adopt extensive, low-input practices such as cattle ranching with low stocking rates. Since Amazonian soils are of relatively low quality, pastures are often created through a slash-and-burn process to fertilize the soil (de Souza Braz, Fernandes, and Alleoni, 2013; SPA, 2021). After a few years, the degraded or depleted land is abandoned in favor of clearing new forest, rather than investing in soil fertility, yields, or sustainable practices. This results in low productivity per hectare and short production cycles with rapid land degradation. Furthermore, land grabbing of undesignated lands plays an important role in extensive cattle ranching, as without land speculation, many ranches would not be profitable (Bowman et al., 2012; J. Miranda et al., 2019; Azevedo-Ramos et al., 2020).

When deforestation is restricted, for example through stricter monitoring and enforcement of environmental regulations, the option to move into new forestland, illegally or not, becomes much more costly (Koch et al., 2019). As land becomes scarcer and its price increases, rational producers reoptimize land uses inside the existing cleared area, replacing marginal low-rent uses, such as pastures or staple crops, with high-rent soy crops, consistent with a von-Thünen-style bid-rent ordering (J. H. v. Thünen, 1826). Genetically modified soybeans are input-sensitive, scalable, and highly profitable, making farmers pivot toward soy under land scarcity. Although this increases costs, investments can also raise yields per hectare and overall productivity, generating higher value added per unit of land (and potentially per unit of labor) vis-à-vis extensive, low-input expansion (Moffette, Skidmore, et al., 2021). Restrictions on deforestation can also incentivize technology adoption, knowledge transfer, or innovation, which could ultimately lead to more stable and diversified local economies that are less dependent on the boom-and-bust cycles of land clearing. Over time, these shifts can offset or even exceed the short-term economic losses from restricting deforestation—especially if complementary policies such as subsidized credits (Assunção, Gandour, Rocha, and Rocha, 2019) or stakeholder cooperation help farmers decrease (transaction) costs (Neves and Whately, 2016; dos Santos Massoca and Brondízio, 2022), or if supply chains evolve to reward sustainable production, for example through certification (DeFries, Fanzo, et al., 2017) or improved market access, as was the case for deforestation-free soy following the Amazon Soy Moratorium (Gibbs, Rausch, et al., 2015; Gibbs, Munger, et al., 2016; Heilmayr et al., 2020).

Merkus (2024) provides a detailed analysis of economic impacts of the Priority List (PL)—a list of high-deforestation municipalities with increased monitoring and enforcement efforts. He finds that PL municipalities have 6% higher GDP and a 2 percentage point lower share of households receiving income support. When disentangling the GDP effect, it appears to be entirely driven by a higher agricultural GDP, starting about three years after being placed on the PL. Koch et al. (2019) and Moffette, Skidmore, et al. (2021) showed that constrained ranchers in PL municipalities improve pasture productivity via increased stocking rates, while Damm, Cisneros, et al. (2024) provide evidence of pasture-to-soy intensification and increased exports of highly profitable soy to China and non-EU regions. Farms in PL municipalities on average also increased in size, profitability, and fertilizer use (Merkus, 2024). Constraining land expansion could therefore have triggered considerable on-farm investments, increasing agricultural yields and economic outcomes without further large-scale deforestation.

Importantly, PL municipalities did not receive any financial benefits or infrastructure investments to improve their economic situation. Furthermore, being exposed on a public blacklist of high-deforestation municipalities arguably poses more of a business risk than an opportunity (Cisneros et al., 2015; Damm, Cisneros, et al., 2024). Nevertheless, deforestation decreased drastically in these municipalities without detrimental effects on the economy. Increased monitoring and enforcement likely also improved governance through more land tenure security and reduced land-grabbing-related violence (G. M. d. Oliveira and B. V. Miranda, 2024), which could further benefit local economies (Thaler, Viana, and Toni, 2019; Stabile et al., 2020; dos Santos Massoca and Brondízio, 2022). Theory and evidence therefore point to the possibility of achieving substantial economic gains without environmental harm through stricter monitoring and enforcement in the Amazon. However, these effects may be context-specific. The Amazon's low baseline productivity and weak institutions imply that marginal returns to enforcement could be high. In more developed agricultural frontiers or governance settings, effects may differ.

3.3 Data and methods

3.3.1 Data sources

This study builds on a rich municipality-by-year panel dataset covering 537 municipalities in the Brazilian Amazon during the years 2002–2016 to evaluate the effects of environmental enforcement on local economies and land-use dynamics. We focus on municipalities in the Amazon biome with more than 10% forest cover in the year 2000 that received at least one fine until 2015. The core variable for environmental enforcement was derived from distributed fines for illegal deforestation issued by IBAMA (2024), aggregated by municipality and year. Fines serve as a direct measure of enforcement intensity across municipalities, reflecting the agency's efforts to prevent and sanction environmentally harmful practices. Fines have been established as a reliable proxy for enforcement intensity in the Brazilian Amazon in many studies (Hargrave and Kis-Katos, 2012; Assunção, Gandour, and Rocha, 2023; Kuschnig et al., 2023). However, the non-random allocation of enforcement activities calls for an econometric approach to address potential endogeneity. For this purpose, we use a cloud cover derivative as an instrumental variable for fines, similarly to Assunção, Gandour, and Rocha (2023).

DETER was created in mid-2004 by the Brazilian National Institute for Space Research (INPE) as a near-real-time deforestation alert system to support environmental enforcement by IBAMA. The alert system was based on the almost daily images produced by the Terra satellite's Moderate Resolution Imaging Spectroradiometer (MODIS) sensor at 250 meter spatial resolution (Diniz et al., 2015). Starting in late 2015, INPE switched to images from newer satellites at higher resolution and stopped using MODIS images in 2017 (Diniz et al., 2015). Hence, we define 2016 as the final year in our panel dataset. Clouds considerably impair the visibility of satellite images, potentially influencing IBAMA's decision making and their enforcement efforts. We therefore downloaded monthly municipal average MODIS cloud cover data through Google Earth Engine, accompanied by monthly average weather and climate data from TerraClimate (Abatzoglou et al., 2018) as potential control variables.

Data on land use and land-use change were obtained from MapBiomas (2023) Collection 8, a high-resolution dataset providing annual classifications of land cover and transitions. This allows for a detailed examination of land-use changes, such as deforestation or the expansion of agricultural areas. Tracking these changes is critical to understand how enforcement of environmental regulations or conservation policies influence land-use decisions.

Labor and wage data were derived from the Annual Social Information Report (RAIS) published by the Ministry of the Economy. It provides insights into formal employment and income across economic sectors, classified under National Classification of Economic Activities Version 1 (CNAE 1) industry codes. This facilitates an evaluation of shifts in labor demand within and between sectors, such as agriculture, manufacturing, or services. We calculate full-time equivalents (FTEs)[1] of the workforce and hourly wages to make employment statistics comparable across municipalities and time. Rural credit data from the Brazilian Central Bank (BCB) further supplements our socioeconomic variables with information on financial flows. Credit access is particularly relevant in this context, as it supports both agricultural expansion and intensification.

Soy production and data on the number of cattle heads were sourced from the Brazilian Institute of Geography and Statistics's (IBGE, 2024) municipal agricultural and livestock production survey (PAM and PPM, respectively). GDP data from IBGE (2024) further complements the agricultural and economic dataset.

Additionally, we control for the influence of changes in the global soy price. The impact of deviations from a three year moving price average is differentiated by the average soy suitability of each municipality. These data were downloaded from the Food and Agricultural Organization (FAO/IIASA, 2012; FAO, 2023a).

We also follow Assunção, Gandour, and Rocha (2023) and aggregate all independent and endogenous variables from August of the previous year until July of the current year.[2] This is not possible for outcome variables though, which are published at the calendar year (January to December). To avoid overlap and potential reverse causality between cloud cover and deforestation, we lag all independent and endogenous variables by at least one year. This means that, for example, we examine the influence of cloud cover and enforcement from August in $t-2$ until July $t-1$ (henceforth the $t-1$ period) on an outcome in year t (January to December).

Summary statistics of all outcome and control variables pre- and post-2004 are presented in the Appendix Tables 3.A.1–3.A.2.

1. FTEs are based on the total hours worked in a specific (sub-)sector divided by 44 working hours per week for 52 weeks per year.

2. This time period is also referred to as the Satellite-Based Monitoring of Deforestation in the Brazilian Amazon (PRODES) year. The PRODES project is the official satellited-based deforestation tracking used by the Brazilian government. Deforestation maps and statistics are published annually at the end of a PRODES year. IBAMA uses PRODES deforestation maps in multiple ways, e.g. to mask deforested areas from DETER and to concentrate observation and enforcement on the remaining forest areas. The PRODES year definition therefore actually influences IBAMA's enforcement allocation efforts.

3.3.2 Econometric framework: Two-stage least squares instrumental variable approach

The main concern in estimating the causal impact of enforcement on local economies is endogeneity. Enforcement intensity may be endogenously determined, for example, by deforestation risks or local conditions, such as infrastructure. If enforcement efforts are strategically allocated to high-deforestation areas, standard ordinary least squares estimates would be biased.

To address this issue, we implement a panel 2SLS IV approach, where cloud cover serves as an instrument for enforcement intensity, following Assunção, Gandour, and Rocha (2023). Cloud cover obstructs satellite-based monitoring of deforestation by diminishing visibility, thereby reducing the likelihood of IBAMA issuing fines. However, cloud cover or weather conditions associated with denser cloud cover could also affect deforestation rates directly. For example, cloud cover is greater during the wet season, when most slashing as part of the slash-and-burn land-clearing process occurs. Cloud cover could therefore be associated with more fining by IBAMA, especially before the introduction of DETER.

We thus exploit the differential post-2004 sensitivity of satellite-based monitoring to cloudiness by interacting cloud cover with an indicator for the DETER era. The identifying variation comes from the period when DETER was operational, which should isolate the visibility component of clouds as an exogenous instrument. Cloud cover itself is kept as a control variable to absorb any direct effect on land use.

First stage regression: Relevance of the instrument

We estimate the relationship between the interacted enforcement regressor and cloud cover as:

$$\begin{aligned}[\,\mathrm{IHS}(\mathrm{Enforcement}_{i,t-n})\cdot \mathrm{Post}_{t-n}] \;=\; &\alpha + \beta\left(\mathrm{CloudCover}_{i,t-n}\cdot \mathrm{Post}_{t-n}\right)\\ &+ \gamma\,\mathrm{CloudCover}_{i,t-n} + \theta\,\mathrm{IHS}(\boldsymbol{X}_{i,t-n}) + \lambda_i + \tau_t + \varepsilon_{i,t},\end{aligned} \tag{3.1}$$

where:

- IHS stands for the inverse hyperbolic sine transformation,[3]
- $\mathrm{Enforcement}_{i,t-n}$ represents enforcement intensity in municipality i at time $t-n$ with $n \in \{1,2,3\}$, proxied by IBAMA fines for deforestation,

3. The inverse hyperbolic sine transformation is defined at zero and behaves similarly to the natural logarithm for large values, while yielding approximately linear behavior for values near zero. It allows for interpreting small coefficients as semi-elasticities or elasticities, depending on context, similarly to a log transformation (Bellemare and Wichman, 2020).

- $\text{Post}_{t-n} \equiv \mathbf{1}\{t \geq 2005\}$ is an indicator equal to one in the DETER era,
- $\text{CloudCover}_{i,t-n}$ is the average annual cloud cover as a share of the municipal area in municipality i in year $t-n$,
- $\boldsymbol{X}_{i,t-n}$ is a vector of time-varying control variables (e.g., temperature, rainfall, soy price impacts),
- λ_i and τ_t are municipality and year fixed effects, respectively, and
- $\varepsilon_{i,t}$ is the error term, with standard errors two-way clustered by municipality and microregion-year following Assunção, Gandour, and Rocha (2023), addressing serial correlation within municipalities and spatial correlation in cloud cover across municipalities in the same microregion.

The coefficient β captures the differential effect of cloud cover on enforcement intensity in the post-2004 period, when satellite-based monitoring through DETER was effective. This interaction isolates variation in cloud cover that impairs deforestation monitoring specifically via its effect on satellite visibility, while γ absorbs direct effects of cloud cover that may influence deforestation behavior or enforcement indirectly.

Second stage regression: Causal impact of enforcement

The second stage estimates the causal impact of enforcement on land-use change, and economic variables using the predicted values of enforcement from the first stage:

$$\begin{aligned} \text{IHS}(Y_{i,t}) &= \alpha + \phi\, \widehat{\text{Enforcement}}_{i,t-n} + \gamma\, \text{CloudCover}_{i,t-n} \\ &\quad + \theta\, \text{IHS}(\boldsymbol{X}_{i,t-n}) + \lambda_i + \tau_t + \eta_{i,t}, \end{aligned} \tag{3.2}$$

where $Y_{i,t}$ represents the outcome variable of interest, such as land-use change, employment, wages, or GDP, and $\widehat{\text{Enforcement}}_{i,t-n}$ is the predicted enforcement intensity from the first stage. The coefficient ϕ captures the causal impact of enforcement on $Y_{i,t}$.

The validity of the IV approach relies on two core assumptions: relevance and the exclusion restriction. Relevance is established if cloud cover interacted with the post indicator significantly predicts enforcement intensity ($\beta \neq 0$), which we test in the first stage. The exclusion restriction cannot be tested directly. One threat to the exclusion restriction is simultaneity. Deforestation could influence same-year cloud formation and enforcement, for example via changes in local energy balances, potentially increasing or decreasing the probability of cloud formation. Furthermore, IBAMA could also increase enforcement efforts in response to an escalating deforestation crisis. It is thus important to mitigate this source of simultaneity or reverse causality bias by separating the timing of cloud cover and enforcement observations from the deforestation observations. Following Assunção, Gandour, and Rocha (2023), we use cloud cover and enforcement in year $t-1$ to predict deforestation in year t, as land-use decisions in one year are likely to still be influenced by enforcement from prior years. Thus, we additionally also use $t-2$ and $t-3$ year lags to explore possible delays in the response between enforcement and outcome. A second threat to instrument validity is spatial autocorrelation or correlation of cloudiness with agricultural productivity (e.g., via soil conditions or rainfall). We address this by including a rich set of time-varying weather controls, municipality and year fixed effects, and two-way clustering of standard errors by municipality and microregion-year.

Control variables and fixed effects play a critical role in the model validity. The vector of time-varying controls ($\boldsymbol{X}_{i,t-n}$) includes weather and climate variables derived from TerraClimate (Abatzoglou et al., 2018), which control for potential climate influences associated with denser cloud cover. Moreover, we control for exogenous global soy price changes—a major driver of deforestation in the Brazilian Amazon—differentiated by the soy suitability of each municipality, to further mitigate potential omitted variable bias. Municipality fixed effects (λ_i) control for unobservable, time-invariant characteristics such as geography or long-term institutional factors, while year fixed effects (τ_t) account for temporal shocks common to all municipalities, such as macroeconomic trends.

3.4 Results

3.4.1 First-stage results: Instrument validity and strength

Table 3.1 reports the first-stage estimates used to estimate second-stage results presented in Figures 3.1–3.3. The coefficient for cloud cover post-2004 in the one year lag model is negative and statistically significant at the 1% level, indicating that greater cloudiness after the introduction of DETER is associated with a reduction in observed enforcement activity. Particularly, a one percentage point increase in cloud cover in the one-year lag model reduces IBAMA fining by approximately 1.87%. The result for the two- and three-year lag model are still quite similar in effect size and significance. This pattern is consistent with the interpretation that clouds hinder satellite-based monitoring, weakening on-the-ground enforcement capacity precisely when remote sensing is introduced. By contrast, the coefficient for cloud cover itself is consistently positive but statistically indistinguishable from zero, in line with the absence of a pre-existing relationship between cloudiness and enforcement prior to the satellite-monitoring regime.

Instrument strength is satisfactory. The Montiel–Olea–Pflueger effective first-stage F-statistic equals 15.08 in the $t-1$ model, which is above the weak-IV threshold for a single endogenous regressor (2.86, cf. Montiel Olea and Pflueger, 2013). The conventional first-stage F-statistic is larger than 10 for all three models. The effective F-statistic and weak-IV threshold for the other two (i.e. $t-2$ and $t-3$) models are $12.97 > 3.03$, and $10.98 > 3.28$, respectively. Taken together, these metrics indicate that cloud cover post-2004 is a relevant and sufficiently strong instrument for enforcement intensity, with the effective F providing a heteroskedasticity-robust assessment of instrument strength (Montiel Olea and Pflueger, 2013). These results support using cloud-induced variation in observability to identify the causal effects of enforcement in the second stage.

Table 3.1. First-stage: Cloud cover and environmental enforcement

	Enforcement		
	t-1	t-2	t-3
Cloud cover	1.327	0.942	0.658
	(1.186)	(1.187)	(1.195)
Cloud cover post-2004	-1.868***	-1.767***	-1.653**
	(0.640)	(0.641)	(0.652)
Weather controls	✓	✓	✓
Soy price impact control	✓	✓	✓
Municipality fixed effects	537	537	537
Year fixed effects	15	14	13
Observations	8,055	7,518	6,981
R^2	0.646	0.648	0.649
F-test (1st stage)	70.953	61.475	50.454
Effective F-test (1st stage)	15.077	12.967	10.975

Note: All regressions use year and municipality fixed effects. Time-variant controls include weather variables and soy price impacts. Standard errors (in parentheses) are two-way clustered at the municipality and microregion-year level.
*p<0.1; **p<0.05; ***p<0.01.

3.4.2 Second-stage results: Impact of environmental enforcement on local economies

Figures 3.1–3.3 present the estimated second-stage effects of environmental enforcement on land use, agricultural labor markets, production, and broader economic outcomes.

Land-use change

Figures 3.1a and 3.1b show the impact of environmental enforcement on land-use expansion and specific land-use transitions, respectively. We find that environmental enforcement substantially reduces deforestation and pasture expansion. A 1% increase in IBAMA fines would one year later decrease deforestation, and pasture and other crop (i.e. non-soy) expansion by approximately 3.5%, 5.23%, and 1.28%, respectively. These effects are significant at the 1% level for deforestation and pasture expansion, and at the 5% level for other-crops expansion. The estimated fine-deforestation elasticity is substantially larger than the 0.53% in Assunção, Gandour, and Rocha (2023), however, we have to interpret it carefully as a local average treatment effect (LATE) for municipalities whose fines respond to changes in cloud cover. Increasing enforcement has a large deterrent effect in those municipalities where illegal deforestation is ongoing and where satellite detection is impaired by visibility due to cloud cover. It does not necessarily generalize to all municipalities. A one standard deviation decrease in cloud cover of 12 percentage points (cf. Appendix Table 3.A.2) would translate to a 54.5% reduction in deforestation, or around 9.17 square kilometers (km^2) on average per municipality in absolute terms.[4]

The reduction in deforestation is primarily driven by reduced pasture expansion, which is also confirmed by a reduced forest-to-pasture transition in Figure 3.1b. Interestingly, we also find a significant and negative forest-to-soy transition after enforcement, while other-crops-to-soy display a positive significant effect. These two contradicting effects could explain why there is no significant effect on general soy expansion initially in Figure 3.1a. Only after pasture-to-soy transition turns significantly positive does the general soy expansion later also become significant in the $t-3$ model. Taken together, these patterns indicate that enforcement curbs frontier expansion and possibly shifts agricultural activity towards more intensive use of already-converted land with some delay.

Agricultural production

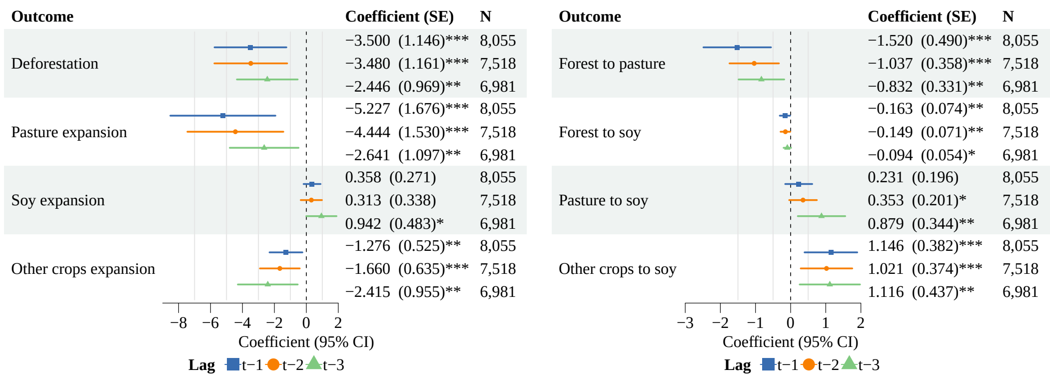
Figures 3.2a and 3.2b show the effects of environmental enforcement on soy and cattle production. Using a different data source for soy expansion,[5] we find a simi-

4. We calculate the new deforestation level after a one standard deviation decrease in cloud cover as $Y' = sinh\left(asinh\left(Y_0\right) + (-3.5 \cdot -1.868 \cdot -0.12)\right)$, where Y_0 is the average 2005–2016 deforestation rate of 16.82 km^2 (cf. Appendix Table 3.A.2). This yields a new deforestation rate $Y' = 7.65$, so the absolute change is $7.65 - 16.82 = -9.17$ and the percentage change $7.65/16.82 - 1 = -54.5\%$.

5. Soy expansion in Figure 3.2a is based on expert data from IBGE (2024) PAM, while the MapBiomas (2023) Collection 8 data used in Figure 3.1a is based on remote sensing and machine learning land cover classification.

Figure 3.1. Second-stage: Environmental enforcement and land-use change

(a) Land-use change **(b) Land-use transitions**



Note: Based on MapBiomas (2023) Collection 8 data. All regressions use year and municipality fixed effects. Time-variant controls include weather variables and soy price impacts. Standard errors (in parentheses) are two-way clustered at the municipality and microregion-year level.
*p<0.1; **p<0.05; ***p<0.01.

lar result as in Figure 3.1a—soy expansion only seems to increase significantly in the $t-3$ model.

Soy production responds positively to enforcement in general: soy production in tons, the value of produced soy in thousand Brazilian Reais, soy value per km^2, and soy value per FTE worker all increase significantly and persistently. Soy yields in tons/km^2 is the only soy production outcome which increases significantly only in the one-year lag model but not in the other models. Particularly interesting is that soy production seems to respond to enforcement immediately compared to the delayed response of soy expansion. Results suggest that 1% increase in fining would increase harvested soy quantity and value by approximately 3% each. Without substantial territorial expansion, this would indicate increased productivity on existing soy fields, potentially through more production inputs.

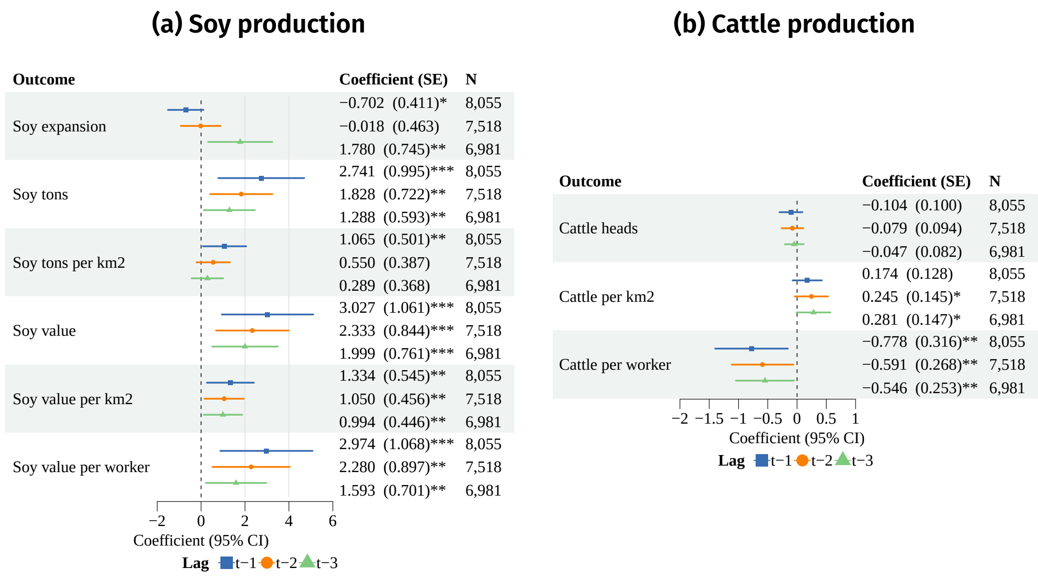
For cattle production, the number of cattle heads does not change. Cattle density indicated by cattle heads per km^2 increases significantly over time, while the number of cattle heads per worker falls significantly, implying more labor input per animal, respectively.

Economic outcomes

Figures 3.3a and 3.3b show the effects of environmental enforcement on workforce and wages in the agricultural sector, as well as broader economic outcomes associated with the agriculture in the Amazon area.

Most notably is the strong and persistent effect on the number of workers and wages in the soy sector shown in Figure 3.3a. A 1% increase in IBAMA fines would

Figure 3.2. Second-stage: Environmental enforcement and agricultural production



Note: Based on IBGE (2024) PAM and PPM data. All regressions use year and municipality fixed effects. Time-variant controls include weather variables and soy price impacts. Standard errors (in parentheses) are two-way clustered at the municipality and microregion-year level.
*p<0.1; **p<0.05; ***p<0.01.

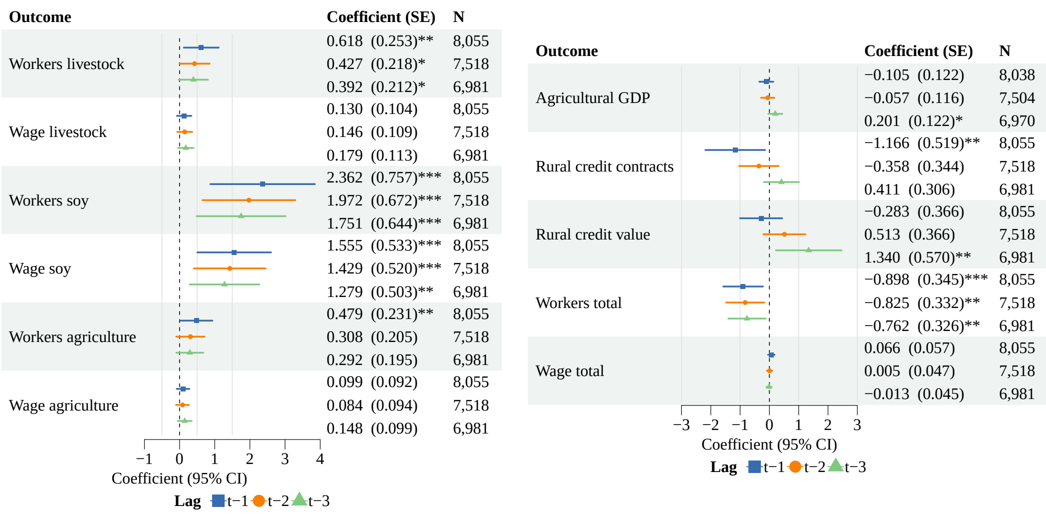
lead to an increase of 2.36% in the soy workforce and 1.56% in their wages one year later. All coefficients for soy workforce and wages are positive and significant for all three lags. While hirings in the soy sector could potentially increase soy production as labor inputs increase, higher wages could either be explained by a general wage increase or the hirings specifically targeting more high-skilled and better paid employees.

The livestock sector also seems to be expanding the workforce significantly, but neither wages nor production results in Figure 3.2b suggest any production or productivity increase in the cattle sector. In fact, a significantly lower number of cattle heads per worker would even suggest a labor productivity decrease, if newly hired workers are really additional and not formalized former clandestine employees.

Aggregate agricultural GDP initially seems unaffected, but becomes significantly positive later in the $t-3$ model, as shown in Figure 3.3b. On the finance side, rural credit contracts decline, but only in the first year. Total credit value is unchanged, hinting at a possibility of consolidation into fewer and larger loans, which could also be more compliant with environmental regulations. An increase in the value of rural credits in the $t-3$ model emphasizes the trend towards larger and more capital-intensive agriculture. Beyond agriculture, total employment falls significantly, whereas average wages remain unchanged. Disentangling the fall in total workers effect, we find that it is driven exclusively by reductions in municipal administrative jobs, particularly city hall administrative staff (CNAE 1, class 75.11-6).

Figure 3.3. Second-stage: Environmental enforcement and economic outcomes

(a) Agricultural workforce **(b) GDP and credits**



Note: Data on workforce and wages are based on RAIS classified by CNAE 1. Agricultural GDP data was sourced from IBGE (2024) and rural credit data from the BCB. All regressions use year and municipality fixed effects. Time-variant controls include weather variables and soy price impacts. Standard errors (in parentheses) are two-way clustered at the municipality and microregion-year level.
*p<0.1; **p<0.05; ***p<0.01.

3.4.3 Robustness checks

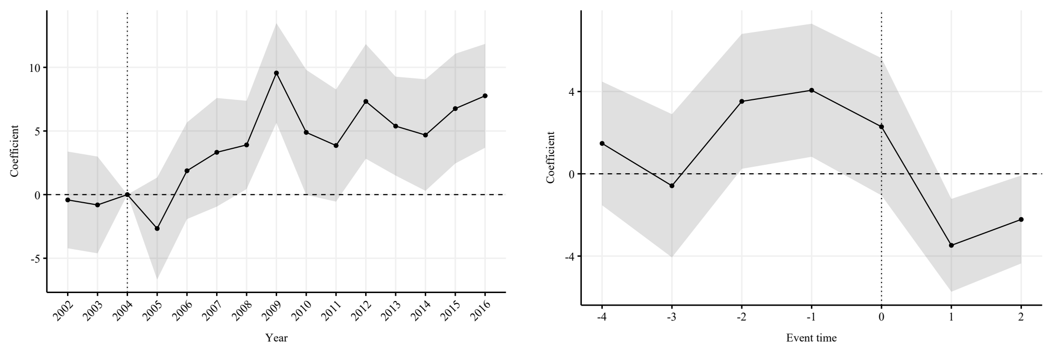
Dynamics and timing

Since the instrument and the endogenous variable—post-2004 cloud cover and post-2004 IBAMA fines, respectively—have a difference-in-differences flavor in this two-way fixed effects panel-IV setting, we want to check for parallel trends in the pre-policy period. Figure 3.4a shows yearly treatment effects of a reduced form regression of post-2004 cloud cover on deforestation—the main outcome targeted by DETER—including control variables. Parallel trends is the main assumption in difference-in-differences regressions. For Figure 3.4a this implies that effect sizes should be close to zero in the 2002–2004 period, which they are. Starting in 2006 the effect of cloud cover on deforestation becomes positive. This means that denser cloud cover in 2005 was associated with more deforestation in 2006, the expected sign, since denser cloud cover impedes satellite-based enforcement strategies to prevent deforestation. The effect becomes significant from 2008 onward, hitting a peak in 2009.

Additionally, since we use various model variations with different lags, we want to more closely examine whether lagged cloud cover is an appropriate predictor of deforestation. We use various leads and lags of our post-2004 cloud cover instrument and include them all in the same reduced form regression to test which ones are significant. Results are shown in Figure 3.4b. Contemporaneous cloud cover, i.e. same year cloud cover and deforestation ($t = 0$), are positively correlated but not statistically significant. The one-year and two-year lags ($t-1$ and $t-2$) have a significant positive association with deforestation. This means that more cloud cover in the two previous years led to more deforestation down the line. In the third year ($t-3$) the relationship between cloud cover and deforestation becomes insignificant. However, we are still able to detect an effect in our main models likely due to strong serial correlation of clouds. Interestingly, the one-year lead ($t+1$) is significantly associated with less deforestation, implying that more cloud cover next year leads to less deforestation this year. Deforesters precisely predicting cloud cover seems unlikely, but this could be due to cloud cover generally being associated with less deforestation in our models, as well as with more IBAMA fines (cf. Table 3.1).

Figure 3.4. Dynamic treatment effects

(a) Yearly effects **(b) Leads/lags**



Note: Dynamic treatment effects of reduced-form regression of the cloud cover instrument on deforestation. (a) shows yearly treatment effects examining pre-policy parallel trends; (b) shows leads/lags of the instrument, examining delays and anticipation effects. 95% confidence intervals are shown as gray ribbons.

Exclusion restriction

The main threat to the validity of the IV approach is a violation of the exclusion restriction. The post-2004 cloud cover instrument must affect deforestation only by changing enforcement intensity. No other pathway from clouds to deforestation is allowed after including controls and fixed effects. In the past, two policy tools have been strategically used by various federal governments to develop the

Brazilian Amazon: infrastructure development and population growth, for example in form of construction of the Trans-Amazonian Highway or financial incentives for settlers, respectively. Both have been identified as direct or indirect drivers of deforestation in the past (Busch and Ferretti-Gallon, 2023). Therefore, the exclusion restriction could be violated if cloud cover systematically predicts population dynamics or infrastructure development, which could then be associated to more or less deforestation. We test this hypothesis in a reduced form regression, where past-year cloud cover is used to predict population dynamics or commodity transportation costs. Population data was sourced from IBGE (2024). Census data was used for the years 2000, 2007, and 2010. Data for all other years are based on estimates created by IBGE (2024). Commodity transport costs were sourced from Victoria et al. (2021) and are based on road network data for the years 2000, 2005, 2010, and 2017. Missing years were linearly interpolated. The results are presented in Table 3.2.

Column (1) shows cloud cover is not systematically related to population growth or decline. A post-2004 cloud cover increase by 1 percentage point is related to 0.11% increase in population, but this effect is not statistically different from zero. On the other hand, column (2) shows that cloud cover and post-2004 cloud cover are significantly related to less transportation costs. This could mean, for example, that roads are systematically being built in the cloudiest municipalities. Since cloud cover predicts infrastructure development, which could then in turn drive deforestation, this would violate the exclusion restriction. We perform a sensitivity analysis in which we drop the top 10% municipalities with the largest transportation cost reductions, i.e. the most road network expansion. Results are presented in Appendix Table 3.A.3 (first-stage) and Appendix Figures 3.A.1–3.A.3 (second-stage).

Results in Appendix Table 3.A.3 show that the post-2004 cloud cover instrument is still a significant predictor of IBAMA fines in all first-stage models. The conventional first-stage *F*-statistic is larger than 10 in all scenarios and even stronger than in our main results (cf. Table 3.1), despite a lower number of observations. This indicates that identification does not hinge on high-infrastructure-growth municipalities. Main results in this more exclusion-restriction-robust subsample also remain mostly unchanged albeit slightly smaller effect sizes for land-use change outcomes.

3.5 Discussion and conclusion

The results presented in this study show a clear picture of how effective environmental enforcement in the Brazilian Amazon has been in reducing deforestation and potentially pushing agriculture from extensive cattle ranching towards more

Table 3.2. Robustness check: Reduced form regression

	Population	Transportation costs
	(1)	(2)
Cloud cover	0.009	-0.769**
	(0.120)	(0.334)
Cloud cover post-2004	0.105	-0.715***
	(0.081)	(0.155)
Weather controls	✓	✓
Soy price impact control	✓	✓
Municipality fixed effects	534	537
Year fixed effects	15	15
Observations	7,997	8,055
R^2	0.985	0.948

Note: All regressions use year and municipality fixed effects. Time-variant controls include weather variables and soy price impacts. Standard errors (in parentheses) are two-way clustered at the municipality and microregion-year level.
*p<0.1; **p<0.05; ***p<0.01.

intensive soybean farming. However, it is important to contextualize these findings in the broader economic and political landscape of the region post-2004.

First and foremost, we find a significant reduction in deforestation following stricter enforcement by the environmental agency IBAMA, an effect that is well documented in the literature (Hargrave and Kis-Katos, 2012; Börner, Wunder, Wertz-Kanounnikoff, Hyman, et al., 2014; Börner, Kis-Katos, et al., 2015; Börner, Marinho, et al., 2015; Assunção, Gandour, and Rocha, 2023; Kuschnig et al., 2023; Nunes et al., 2024). Overall, deforestation in the Amazon decreased quite substantially during the studied period, which is attributed to various political and economic factors (Richards, R. Myers, et al., 2012; Assunção, Gandour, and Rocha, 2015; West and Fearnside, 2021). Nevertheless, IBAMA and DETER undoubtedly contributed significantly to the post-2004 deforestation reduction in the Brazilian Amazon. While Assunção, Gandour, and Rocha (2023) estimated a deforestation-fine elasticity of -0.53% per 1% fine increase, our estimate is considerably larger. Interpreting the result as an elasticity, a 1% increase in fines would lead to a 3.5% decrease in deforestation. However, since this result is a LATE, we can only interpret it in the context of cloud cover variability. If annual cloud coverage was one standard deviation lower than average, DETER could prevent around 4,924 km^2

of deforestation in the Brazilian Amazon biome. Furthermore, this effect seems to be mostly driven by changes at the extensive enforcement margin rather than the intensive margin. This means that the strongest effects are found in municipalities that go from no enforcement to some level of enforcement, rather than increasing fining in an already high-enforcement municipality. In absolute terms, prevented deforestation matches almost exactly prevented pasture expansion.[6] This means that environmental enforcement was effective primarily by reducing extensive and speculative forest-to-pasture transitions.

Secondly, the soy sector seems to benefit from enforcement. While forest-to-soy transition decreases, soy simultaneously expands over other crops. Only after soy also starts to substantially displace pastures, does general soy expansion become significant. This transition of less forest-to-soy, and more soy over other cropland and later pastures, shows a sustainable transition of soy expansion away from forest towards previously cleared lands. This transitions reflects also into labor markets, where more workers in the soy sector get hired and on average also receive higher wages to increase soy production. Additionally, soy production statistics show a larger quantity produced, in total and per km^2, and a higher value of production, in total, per km^2, and per worker. This impact of enforcement on the soy sector is quite robust across different outcomes and metrics (satellite-monitored expansion, reported labor relations, and production statistics). Such land-use intensification from low-rent agriculture and ranching towards high-rent soy production has been found previously in the case of priority municipalities in the Brazilian Amazon (Crepin, 2024; Damm, Cisneros, et al., 2024). Especially financially constrained smallholder farms seem to be converted to larger, better capitalized, and better equipped soy plantations (Crepin, 2024; Merkus, 2024). Interestingly, we find a negative effect on the number of rural credit contracts issued by the BCB. Since 2008, Brazilian banks have to fulfill a due diligence process to not finance deforestation-driven agriculture (Assunção, Gandour, Rocha, and Rocha, 2019). Financial and environmental regulations are likely to constrain smallholder farmers more than large export-oriented soy farmers, which could also be supported by multinational export corporations.

Thirdly, the results show that stricter environmental enforcement can reduce deforestation without harming agricultural GDP. At the same time, there are two notable challenges: access to rural credit becomes more restricted, and municipal administrative capacity declines, contributing to job losses in the overall economy. A practical way forward could be to pair enforcement with targeted and subsidized finance compliant with environmental regulations. For example, credits loans could

6. 9.17 km^2 vs. 7.25 km^2 per municipality, respectively, following a one standard deviation decrease in annual cloud cover.

be tied to productivity investments combined with technical assistance, to help smaller and mid-sized producers adapt.

To conclude, we can say that environmental enforcement seems to be very effective in influencing land-use decisions in the Brazilian Amazon. A cost-benefit analysis by Assunção, Gandour, and Rocha (2023) showed that higher administrative costs already paid off multiple times by just considering the price of carbon of prevented deforestation, not even taking any economic or societal effects into account. There is an emerging literature examining the question of potential spillover effects and positive externalities of effective biodiversity conservation on crime and social stability (G. M. d. Oliveira and B. V. Miranda, 2024). A reduction in homicide rates suggests that stronger enforcement not only regulates land use but also mitigates illegal land speculation, land grabbing, and associated violence (Bowman et al., 2012; J. Miranda et al., 2019; Azevedo-Ramos et al., 2020). In combination with land tenure security reforms that clarify land rights, environmental enforcement could be an effective governance tool to enhance legal compliance and to strengthen formal economic relationships in historically weakly regulated frontier regions, such as the Brazilian Amazon (Thaler et al., 2019; Furumo et al., 2024). In this context, effective enforcement of (environmental) regulations would not only be crucial to prevent illegal deforestation but could also be seen as a prerequisite for other policy approaches (Börner, Marinho, et al., 2015; Stabile et al., 2020). Furthermore, enforcement played a crucial role in strategically redirecting economic efforts from deforestation towards more sustainable and economically advantageous practices. However, rural credit availability could be the major limiting factor in this sustainable transition. Policymakers should therefore focus on ensuring robust credit support systems to facilitate an equitable and sustainable agricultural transition. Nevertheless, since credit can also be used to finance deforestation, banks need to be diligent and verify financial flows. Still, enforcement is not a silver bullet. Deforestation could potentially increase again if enforcement stops and there is no additional oversight. For example, researchers have noted a recent decline in IBAMA's effectiveness to reduce deforestation, driven by cuts in funding and staff, and political interference (Kuschnig et al., 2023; Nunes et al., 2024). The success of environmental enforcement therefore also depends on the broader political and economic support in which it is embedded.

Appendix 3.A Appendix to: Balancing Growth and Conservation: The Economic Impact of Environmental Enforcement in the Brazilian Amazon

3.A.1 Appendix tables

Table 3.A.1. Summary statistics: Pre-2004

	2002–2004				
	Mean	Median	SD	Min	Max
IBAMA fines (count)	10.37	2.00	31.65	0.00	798.00
Land-use change					
Deforestation (km^2)	50.18	13.50	111.26	-141.19	1628.71
Pasture expansion (km^2)	42.75	11.31	103.30	-174.20	1619.84
Soy expansion (km^2)	5.24	0.00	28.10	-4.71	427.03
Other crops expansion (km^2)	2.11	0.00	21.76	-247.93	367.43
Land-use transitions					
Forest to pasture (km^2)	51.81	15.02	100.49	0.00	1552.23
Forest to soy (km^2)	0.09	0.00	0.87	0.00	19.16
Pasture to soy (km^2)	1.98	0.00	9.65	0.00	149.42
Other crops to soy (km^2)	4.78	0.00	24.89	0.00	365.98
Agricultural workforce					
Workers livestock (FTEs)	59.28	9.58	128.05	0.00	1259.69
Wage livestock (R$/hour)	1.52	1.68	1.23	0.00	9.66
Workers soy (FTEs)	11.96	0.00	94.93	0.00	1826.24
Wage soy (R$/hour)	0.30	0.00	1.05	0.00	16.88
Workers agriculture (FTEs)	119.40	16.17	296.08	0.00	3174.29
Wage agriculture (R$/hour)	1.67	1.79	1.32	0.00	22.14
Soy production					
Soy expansion (km^2)	9.74	0.00	54.78	-144.71	850.00
Soy tons (tons)	16240.46	0.00	108726.20	0.00	1688120.00
Soy tons per km2 (tons/km^2)	48.82	0.00	106.21	0.00	380.00
Soy value (thousand R$)	8358.73	0.00	57593.84	0.00	1060139.00
Soy value per km2 (thousand R$/km^2)	24.46	0.00	55.48	0.00	246.00
Soy value per worker (thousand R$/FTEs)	27.81	0.00	217.07	0.00	6295.62
GDP and credits					
Agricultural GDP (thousand R$)	37995.85	19822.83	70764.16	504.33	968632.84
Rural credit contracts (count)	338.53	137.00	566.92	0.00	5390.00
Rural credit value (thousand R$)	6119.32	1321.04	18848.16	0.00	335411.47
Workers total (FTEs)	2746.05	393.94	16676.55	0.00	276004.01
Wage total (R$/hour)	3.24	2.88	1.68	0.00	20.53
Cattle production					
Cattle heads (thousand heads)	93.59	39.33	137.32	0.00	1527.02
Cattle per km2 (heads/km^2)	566.14	102.95	5014.51	0.00	106658.75
Cattle per worker (heads/FTEs)	7679.05	2686.02	13312.79	0.00	165094.00
Controls					
MODIS cloud cover (share)	0.72	0.76	0.11	0.46	0.92
TerraClimate rain (mm)	166.67	162.87	33.37	85.55	305.32
TerraClimate surface radiation (W/m^2)	1833.47	1886.51	185.34	1334.05	2193.03
TerraClimate temperature (°C)	26.22	26.34	0.86	23.57	28.51
TerraClimate vapor pressure deficit (kPa)	81.88	80.63	15.93	53.19	137.84
TerraClimate wind speed (m/s)	148.03	149.03	33.77	53.08	239.44
Soy price impact (index)	10.24	9.92	4.53	2.16	27.93

Table 3.A.2. Summary statistics: Post-2004

	2005–2016				
	Mean	Median	SD	Min	Max
IBAMA fines (count)	10.53	1.00	30.59	0.00	809.00
Land-use change					
Deforestation (km^2)	16.82	4.56	50.34	-752.55	988.53
Pasture expansion (km^2)	10.41	2.60	53.04	-455.61	984.72
Soy expansion (km^2)	6.51	0.00	35.00	-370.55	685.17
Other crops expansion (km^2)	1.35	0.00	25.31	-454.43	411.35
Land-use transitions					
Forest to pasture (km^2)	21.90	7.67	44.00	0.00	841.81
Forest to soy (km^2)	0.02	0.00	0.20	0.00	6.44
Pasture to soy (km^2)	1.62	0.00	8.75	0.00	212.17
Other crops to soy (km^2)	6.96	0.00	29.02	0.00	559.62
Agricultural workforce					
Workers livestock (FTEs)	126.23	35.61	232.71	0.00	2065.05
Wage livestock (R$/hour)	3.54	3.59	2.47	0.00	36.93
Workers soy (FTEs)	43.51	0.00	254.37	0.00	4739.84
Wage soy (R$/hour)	1.35	0.00	2.88	0.00	33.74
Workers agriculture (FTEs)	228.70	52.99	483.17	0.00	5058.97
Wage agriculture (R$/hour)	3.78	3.73	2.63	0.00	39.33
Soy production					
Soy expansion (km^2)	6.61	0.00	53.38	-1520.94	1402.64
Soy tons (tons)	30373.68	0.00	146593.98	0.00	2088540.00
Soy tons per km2 (tons/km^2)	70.79	0.00	126.46	0.00	450.00
Soy value (thousand R$)	20543.73	0.00	103700.61	0.00	1915553.00
Soy value per km2 (thousand R$/$km^2$)	49.00	0.00	95.34	0.00	509.77
Soy value per worker (thousand R$/FTEs)	87.23	0.00	387.24	0.00	12719.30
GDP and credits					
Agricultural GDP (thousand R$)	46064.74	24773.35	73024.58	526.01	889585.41
Rural credit contracts (count)	351.25	217.00	477.80	0.00	7868.00
Rural credit value (thousand R$)	12977.59	3318.78	33425.45	0.00	630554.91
Workers total (FTEs)	4767.39	875.55	26934.14	0.00	501576.69
Wage total (R$/hour)	6.68	5.98	4.09	0.00	112.44
Cattle production					
Cattle heads (thousand heads)	116.56	52.59	173.32	0.00	2282.45
Cattle per km2 (heads/km^2)	432.17	106.97	3152.62	0.00	67573.56
Cattle per worker (heads/FTEs)	4070.06	1392.12	8313.04	0.00	146200.00
Controls					
MODIS cloud cover (share)	0.71	0.74	0.12	0.38	0.92
TerraClimate rain (mm)	172.73	165.85	45.74	57.51	336.16
TerraClimate surface radiation (W/m^2)	1821.34	1833.29	186.99	1319.94	2363.20
TerraClimate temperature (°C)	26.41	26.47	1.02	23.34	29.25
TerraClimate vapor pressure deficit (kPa)	82.88	81.09	17.90	44.91	155.58
TerraClimate wind speed (m/s)	158.62	157.40	36.62	64.48	308.04
Soy price impact (index)	6.34	7.02	23.06	-81.22	92.53

Table 3.A.3. Robustness check: First-stage regression

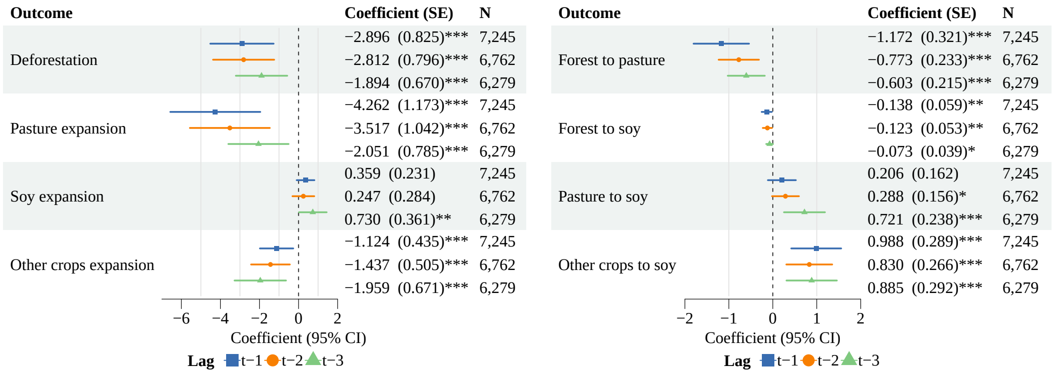
	Enforcement		
	t-1	t-2	t-3
Cloud cover	1.564	1.288	1.048
	(1.230)	(1.232)	(1.248)
Cloud cover post-2004	-2.334***	-2.272***	-2.182***
	(0.680)	(0.684)	(0.699)
Weather controls	✓	✓	✓
Soy price impact control	✓	✓	✓
Municipality fixed effects	483	483	483
Year fixed effects	15	14	13
Observations	7,245	6,762	6,279
R^2	0.650	0.651	0.652
F-test (1st stage)	95.737	86.738	74.679

Note: Subsample analysis where top 10% of municipalities with most infrastructure development were dropped. All regressions use year and municipality fixed effects. Time-variant controls include weather variables and soy price impacts. Standard errors (in parentheses) are two-way clustered at the municipality and microregion-year level.
*p<0.1; **p<0.05; ***p<0.01.

3.A.2 Appendix figures

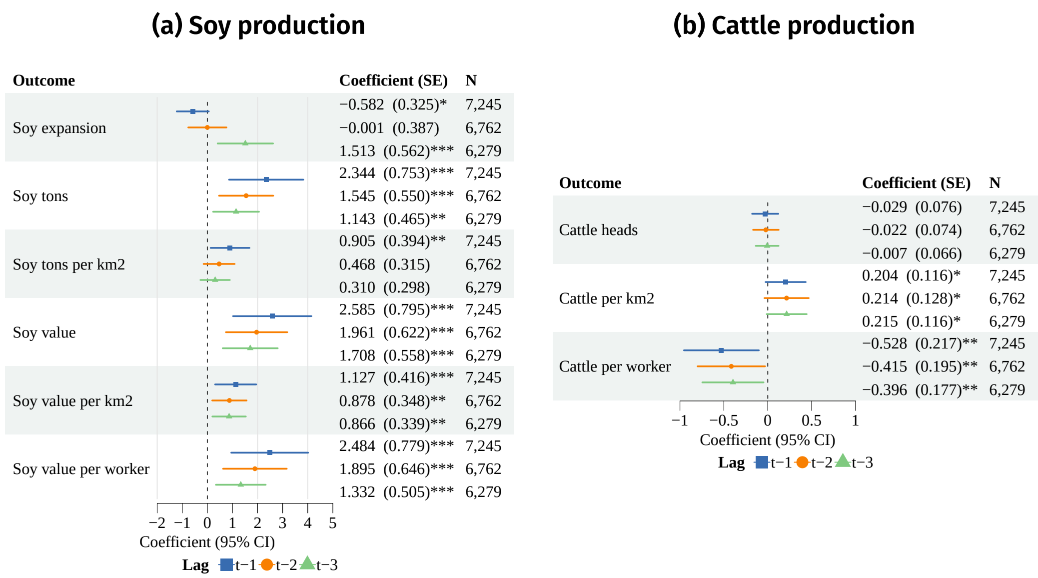
Figure 3.A.1. Robustness check: Second-stage results on land-use change

(a) Land-use change **(b) Land-use transitions**



Note: Subsample analysis where top 10% of municipalities with most infrastructure development were dropped. Based on MapBiomas (2023) Collection 8 data. All regressions use year and municipality fixed effects. Time-variant controls include weather variables and soy price impacts. Standard errors (in parentheses) are two-way clustered at the municipality and microregion-year level.
*p<0.1; **p<0.05; ***p<0.01.

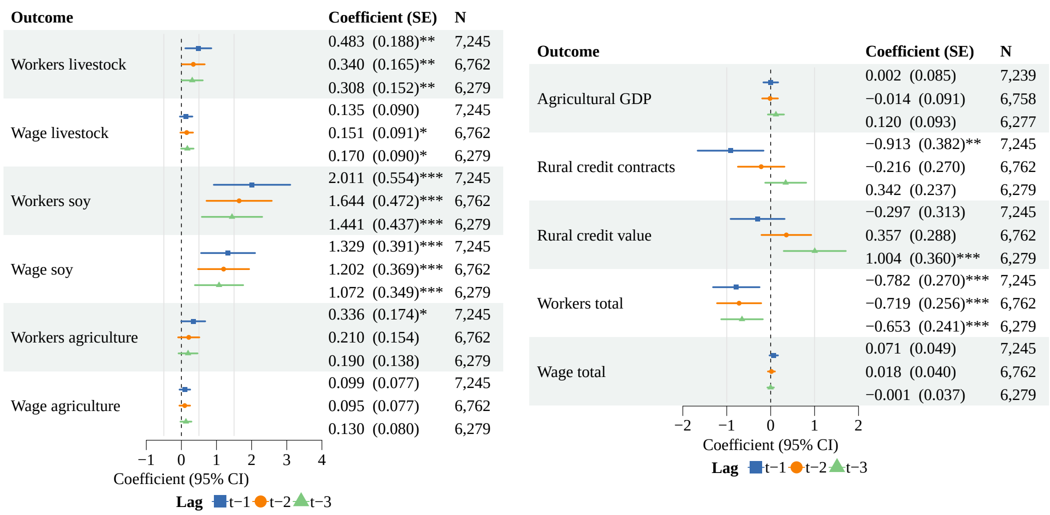
Figure 3.A.2. Robustness check: Second-stage results on agricultural production



Note: Subsample analysis where top 10% of municipalities with most infrastructure development were dropped. Based on IBGE (2024) PAM and PPM data. All regressions use year and municipality fixed effects. Time-variant controls include weather variables and soy price impacts. Standard errors (in parentheses) are two-way clustered at the municipality and microregion-year level.
*p<0.1; **p<0.05; ***p<0.01.

Figure 3.A.3. Robustness check: Second-stage results on economic outcomes

(a) Agricultural workforce **(b) GDP and credits**



Note: Subsample analysis where top 10% of municipalities with most infrastructure development were dropped. Data on workforce and wages are based on RAIS classified by CNAE 1. Agricultural GDP data was sourced from IBGE (2024) and rural credit data from the BCB. All regressions use year and municipality fixed effects. Time-variant controls include weather variables and soy price impacts. Standard errors (in parentheses) are two-way clustered at the municipality and microregion-year level.
*p<0.1; **p<0.05; ***p<0.01.

Chapter 4

Health Benefits of Reduced Deforestation in the Brazilian Amazon⋆

Joint with Jan Börner, Nicolas Gerber, and Britaldo Soares-Filho

Abstract

The conversion of tropical forests in the Amazon region for agriculture and other land uses is associated with health risks linked, for example, to air and water pollution from forest fires and agrochemical use. Several conservation policies introduced in the 2000s aimed at reducing deforestation in the Brazilian Amazon. We exploit variations in the regional targeting of these policies to measure human health externalities of conservation policy enforcement using a double-difference approach at close distance to the Amazon biome border. We find that the change in deforestation pressure reduces forest fire incidence. As a consequence, fine particulate matter concentrations in the air—a main vector for adverse health effects of fire smoke—also decrease. This leads to a reduction in the hospitalization and death prevalence rate due to respiratory health problems and other health benefits for the local population.

JEL Classification: I18, Q23, Q53, Q58
Keywords: Air pollution, Brazilian Amazon, Fire smoke, Human health

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4.1 Introduction

The Amazon forest is one of the most diverse ecosystems in the world and the largest remaining connected tropical rainforest (Foley et al., 2007). Historically, the Amazon basin has served as a large carbon sink, but deforestation and forest degradation have made parts of the region a carbon source (Davidson et al., 2012; Aragão, Poulter, et al., 2014; Brienen et al., 2015; Aragão, Anderson, et al., 2018; Gatti et al., 2021; Mills et al., 2023; Rosan et al., 2024). Still, the region stores a huge amount of carbon as biomass (Malhi, Wood, et al., 2006; Gibbs, Brown, et al., 2007), plays an important role in regulating water cycles (Avissar and Werth, 2005; Lathuillière, Coe, and Johnson, 2016; Staal et al., 2018) and thus remains vital for global and regional climate regulation (Foley et al., 2007; Malhi, Roberts, et al., 2008; Phillips et al., 2009; Soares-Filho, Moutinho, et al., 2010; Davidson et al., 2012; Strand et al., 2018; W. S. Walker et al., 2020). However, the regional climate regulating functions of the forest in the Brazilian Amazon have already been compromised to the extent that agricultural productivity losses linked to further deforestation are likely larger than the economic gains from agricultural expansion (Rodrigues et al., 2012; Leite-Filho et al., 2021; Maeda et al., 2021; Drüke et al., 2023).

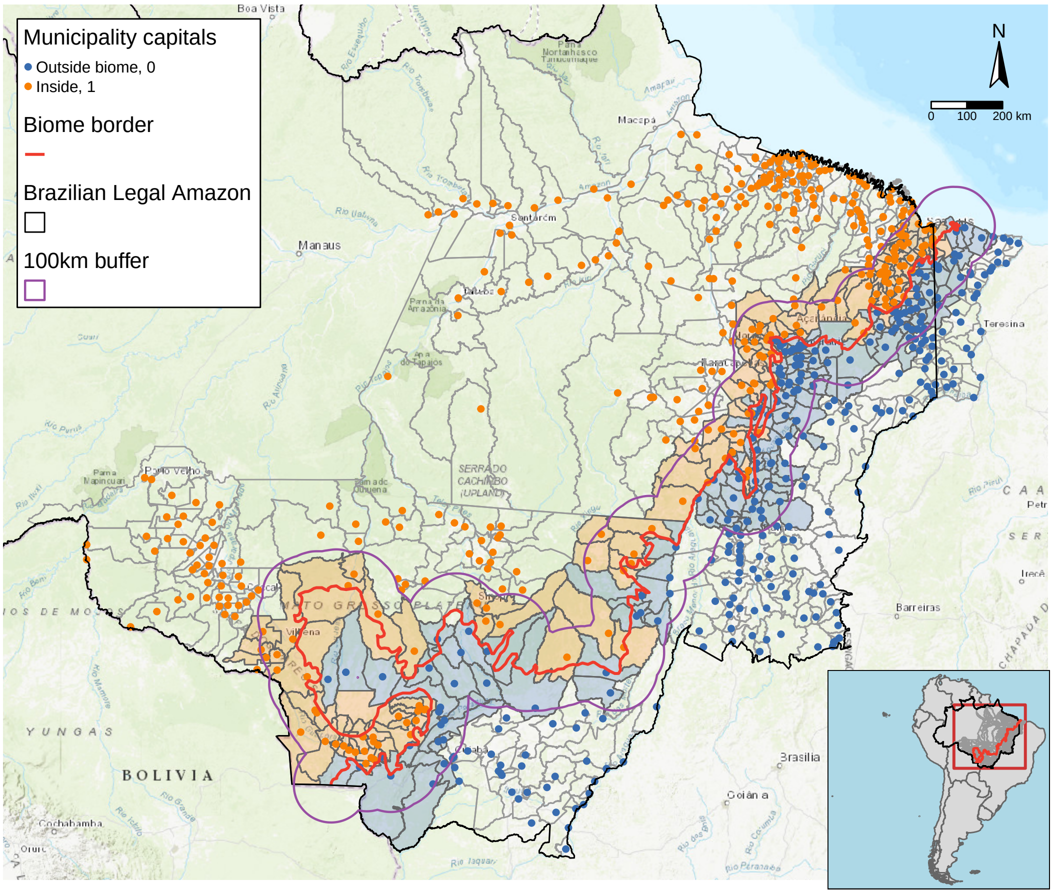
Intact ecosystems, especially tropical rainforests, provide multiple ecosystem services and health benefits beyond climate regulation (Strand et al., 2018; Brouwer et al., 2022). Land-use and ecosystem change, such as forest conversion for agricultural production, not only causes losses in these benefits, but may also pose health risks for the increasing population in many rural areas and urban centers (DeFries, Rudel, et al., 2010; Richards and VanWey, 2015; Mastel et al., 2018). In the Amazon, environmental degradation has been linked to, for example, air pollution from human-induced forest fires (Reddington et al., 2015; de Oliveira et al., 2020; Machado-Silva et al., 2020; Alves de Oliveira et al., 2021; Butt et al., 2021; Campanharo et al., 2021; Rocha and A. A. Sant'Anna, 2022; Fonseca Morello, 2023; Prist et al., 2023; Pullabhotla et al., 2023; Sheehan et al., 2023), contamination of freshwater with (heavy) metals due to mining (Moulatlet et al., 2023; Zanin et al., 2024), and increased incidences of vector borne diseases such as malaria (Bauch et al., 2015; MacDonald and Mordecai, 2019; Ellwanger et al., 2020). Additional human health risks also exist from increasing risks of zoonosis outbreaks (Jones et al., 2013; Dobson et al., 2020; Morand and Lajaunie, 2021) or unsustainable agricultural practices on cleared forestlands (Waichman et al., 2002; Schiesari and Grillitsch, 2010; Pignati et al., 2017; Rekow, 2019; L. N. Lemos et al., 2021; Dias, Rocha, and Soares, 2023).

Deforestation in Brazil is regulated by the Forest Code (FC) from 1996, which was revised in 2012 (Soares-Filho, Rajão, et al., 2014). Its enforcement started in 2004 with the Action Plan to Prevent Deforestation in the Legal Amazon, which applied to the Brazilian Legal Amazon (BLA). The BLA extends the Amazon borders mostly to the clear administrative boundaries of the Amazon states (cf. Figure 4.1), which implies that part of the neighboring Cerrado biome falls under the more rigorous BLA jurisdiction. The FC allows for 20% deforestation on private properties in the Amazon biome, 65% deforestation in the BLA part of the Cerrado biome, and 80% deforestation in the rest of the Cerrado. Deforestation rates started to decrease overall after 2004 (cf. Appendix Figure 4.A.1) but remained elevated in some regions. Therefore, starting in 2008, enforcement was prioritized in high deforestation municipalities in the BLA (Cisneros et al., 2015; Damm, Cisneros, et al., 2024) and the Brazilian Central Bank (BCB) restricted credit access to landowners that violated the FC in the Amazon biome (Assunção, Gandour, Rocha, and Rocha, 2019). Private-sector initiatives, such as the Soy Moratorium and the Cattle Agreement introduced in 2006 and 2009, respectively, emphasized a wider stakeholder approach towards deforestation reduction. In both agreements, traders of soy and cattle committed to not buy from farms that deforested after their respective specific cutoff date—July 2006 for the Soy Moratorium and October 2009 for the Cattle Agreement—but only in the Amazon biome (Gibbs, Rausch, et al., 2015; Gibbs, Munger, et al., 2016; Heilmayr et al., 2020; Moffette and Gibbs, 2021).

Naturally occurring wildfires are rare in moist rainforests, but repeated droughts and forest degradation have increased the occurrence of wildfires in the Amazon region (Davidson et al., 2012; Brando et al., 2014; Aragão, Anderson, et al., 2018; U. Oliveira et al., 2022). The more intense a fire (i.e. the more heat it releases), the higher the injection height, where horizontal winds can transport smoke hundreds and thousands of kilometers (Williamson et al., 2016). Smoke that travels such long distances is usually caused by uncontrolled fires that burn large amounts of biomass. In the Amazon biome, most fires are ignited by humans during the conversion of primary or secondary forests for agricultural use (van der Werf et al., 2009; Aragão and Shimabukuro, 2010; Barlow et al., 2020). When under control, these fires emit smoke at a local scale, posing health risks for the nearby rural and urban populations (Price, Horsey, and Jiang, 2016; Williamson et al., 2016; Fonseca Morello, 2023). For example, in 2019, 69 thousand km^2 of the Amazon basin were affected by fires, of which 85% occurred in areas deforested in 2018 (Cardil et al., 2020).

Lelieveld et al. (2015) estimated that more than 3.3 million premature deaths globally were attributable to outdoor air pollution in 2010, and around 340,000 deaths could be due to fire smoke (Johnston et al., 2012). The main fire smoke air pollutant studied in epidemiology is fine particulate matter ($PM_{2.5}$) with a size of or less than 2.5 micrometers (Williamson et al., 2016). Due to its small size, $PM_{2.5}$ penetrates deep into the lungs, where it can enter the blood system and cause several health problems, including respiratory and cardiovascular diseases, lung cancer, and increased all-cause mortality (Liu et al., 2015; Reid et al., 2016; Williamson et al., 2016; Chen and Hoek, 2020; Dijkhoff et al., 2020; WHO, 2021).

Figure 4.1. Geographic area studied



Note: Boundaries of the Brazilian Legal Amazon (solid black outline) and the Brazilian Amazon biome (solid red line). The buffer outline (in violet) shows a 100 km buffer in each direction of the biome border. Dots show the location of municipality capitals. Orange dots are located within the Amazon biome and blue dots outside. Municipalities whose capitals are located inside the 100 km buffer constitute the selected subsample for the empirical analysis (selected municipalities are also highlighted in orange and blue, respectively). Background map source: World Topographic Map. Esri, TomTom, Garmin, FAO, NOAA, USGS, ©OpenStreetMap contributors, and the GIS User Community.

Adverse health effects have been observed from both, short- and long-term exposure and there seems to be no safe threshold for $PM_{2.5}$ concentrations, although many regulatory agencies have set one. For example, the World Health Organization (WHO) sets a standard of 5 micrograms per cubic meter ($\mu g/m^3$) annual average and 15 $\mu g/m^3$ 24h average for $PM_{2.5}$ concentrations (WHO, 2021). Even if the size of the effect of fire smoke air pollution on health is small, it may affect (very) large populations and the risks are higher for vulnerable people (e.g., infants, elderly and pregnant women) and people with pre-existing conditions.

In this study, we explicitly test the conjecture that controlling deforestation improves local health outcomes via reduced fire smoke emissions by using a unique set of municipal-level panel data covering the years 2003–2017. Our identification strategy explores the Brazilian Amazon biome border as a spatial discontinuity (Assunção, Gandour, Rocha, and Rocha, 2019; Heilmayr et al., 2020; Moffette and Gibbs, 2021). We use a double-difference approach on areas close to the biome border, to control for unobserved time-invariant confounders and to maximize similarities in time-variant confounders between municipalities, respectively. The cutoff year is set to 2007, the first full year after the introduction of the Soy Moratorium. Later policies exclusively focusing on the Amazon biome, such as the BCB credit restrictions and the Cattle Agreement, further decreased deforestation pressure in the Amazon biome and work in favor of our underlying hypothesis. Our results show that the number of fires and associated air pollution decreased significantly. This leads to health benefits for the local population, such as lower hospitalization and mortality rates due to respiratory diseases.

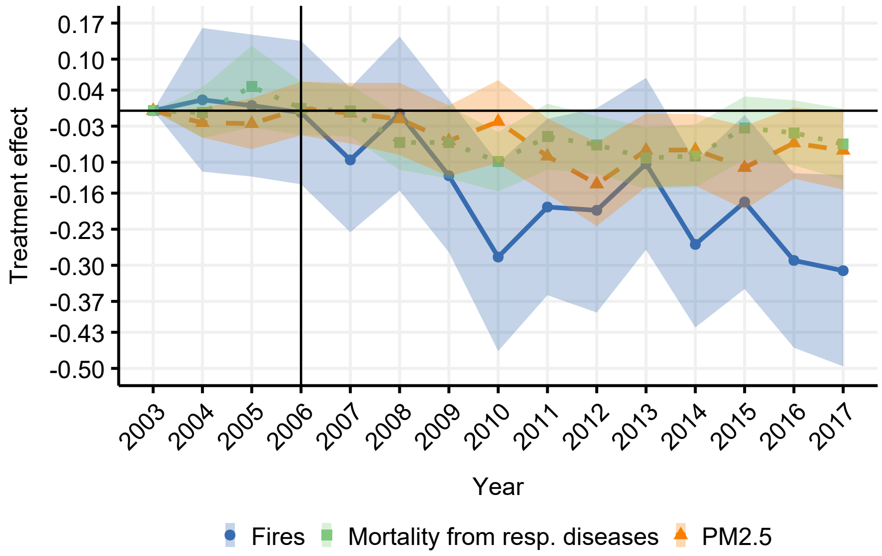
4.2 Results

We start by exploring whether the number of fires and related air pollution in our sample in fact decreased. Figure 4.2 depicts the yearly treatment effects and the 95% confidence interval of the number of fires, $PM_{2.5}$ concentrations in the air—both in decimal percentage changes—and the mortality rate per 1,000 inhabitants due to diseases of the respiratory system. On average, the number of fires decreased by 19.2% (± 8.4), $PM_{2.5}$ concentration in the air decreased by 6.6% (± 4.8), and mortality due to respiratory health problems decreased by 7.2 (± 2.9) cases per 100,000 inhabitants. All of these effects are significant at the 1% level. Numbers in brackets represent the amount to add or subtract to calculate 95% confidence intervals.

4.2.1 Fire and air pollution

The summary statistics in Appendix Table 4.A.1 reveal that the absolute number of fires and air pollution concentrations decreased on both sides of the biome border,

Figure 4.2. Yearly treatment effects and 95% confidence intervals



Note: The treatment effect for the number of fires is represented by the blue solid line and round points, for fine particulate matter air concentration by the orange dashed line and triangular points (both in decimal percentages), and for mortality due to respiratory health problems by the green dotted line and quadratic points (in cases per 1,000). The shaded area represents 95% confidence intervals.

which is probably due to policies addressing deforestation in the entire BLA. While the pre-2007 annual $PM_{2.5}$ concentrations were well above 20 $\mu g/m^3$ on both sides of the biome border and thus more than four times above the recommended long-term exposure limit of 5 $\mu g/m^3$ by the WHO (2021), $PM_{2.5}$ air pollution decreased to less than 20 $\mu g/m^3$ on both sides of the biome border after 2006.

We estimate the average and yearly treatment effects for various other air pollutants. While the average effects indicate whether the overall changes in differences between the two areas are significant, the yearly treatment effects allow us to examine the exact period that these significant differences first occur and how large the differences were prior to treatment. The full results are presented in Appendix Tables 4.A.3–4.A.6 and Appendix Figures 4.A.2–4.A.4. Appendix Figure 4.A.2a shows that the first significant reduction in the number of fires in the Amazon biome occurred in 2010 and their number remained low in most years afterwards. The graphs for all other air pollution concentrations decreased accordingly, indicating a close connection to fires. We pay particular attention to the $PM_{2.5}$ concentration, as it is an indicator of the overall air pollution associated with many health problems and thus the most relevant from an epidemiological perspective. We find an average reduction of 1.97 $\mu g/m^3$ ($\pm$ 0.95) in absolute terms for the period after 2006. As the WHO (2021) recommends an annual average of 5 $\mu g/m^3$ as a safe threshold, this is a sizable effect. All other measured pollutants also decreased significantly with more or less similar effect sizes, although their absolute annual averages were not as high relative to the WHO (2021) limit pre-2007 (cf. Appendix Table 4.A.1). Concentrations for carbon monoxide decreased by 4.9% (± 3.9), nitrogen dioxide by 7% (± 5.6), and sulfur dioxide by 6% (± 4.6). All these results are significant at the 5% level. The only unaffected air pollutant is ozone.

Within each year, average air pollution concentrations vary quite a bit. Appendix Figure 4.A.5 shows the development of monthly average number of fires and $PM_{2.5}$ concentrations over the period 2003–2017. The figure shows a strong correlation between the number of fires and $PM_{2.5}$ concentrations. In the off-fire-season, number of fires are essentially zero and the background $PM_{2.5}$ concentration seems to vary around 10 $\mu g/m^3$, which is twice as large as the WHO annual average guideline of 5 $\mu g/m^3$. During the fire season, number of fires and $PM_{2.5}$ concentrations spike simultaneously and air pollution levels can be more than 10 times their background levels averaged over an entire month, but they also drop off quickly when the fires stop. As a way to measure changes in acute exposure to short-term elevated levels of air pollution, we calculate the relative number of times the WHO short-term air quality exposure guidelines were exceeded. For example, the guideline for daily $PM_{2.5}$ exposure is 15 $\mu g/m^3$, so we calculate the percentage of days within a year at which guideline levels were exceeded and summarize the results in Appendix Table 4.A.1. Additionally, we estimate whether acute exposure to air pollution decreased significantly or not. Results in Appendix Table 4.A.4 show that acute exposure to $PM_{2.5}$ post-2006 just inside the Amazon biome indeed decreased significantly by, on average, 1.5 (± 0.98) percentage points or 5.5 days per year. The only other significant acute air pollution exposure decrease was for sulfur dioxide, which decreased by 0.065 (± 0.05) percentage points or 0.24 days per year.

4.2.2 Health

Finally, we estimate the impacts of multiple health problems that are linked to air pollution and classified according to the International Classification of Diseases Version 10 (ICD-10) code on hospitalization. This includes diseases of the respiratory system (ICD-10 code J), the circulatory system (ICD-10 code I) and lung cancer (ICD-10 code C34), which have a strong body of evidence linked to air pollution; health problems related to pregnancy (ICD-10 code O) or originating in the perinatal period (ICD-10 code P), which represent vulnerable groups that a have a higher risk of health complications; and the number of sexually-transmitted diseases STDs (ICD-10 codes A50–A64, B20–B24, N70–N73) as a robustness check, since they should not be affected by changes in the local fire smoke regime. The treatment effect of these health outcome variables presented in Appendix Table 4.A.5 are expressed in terms of the number of hospitalizations per 1,000 inhabitants. Overall, hospitalizations due to respiratory diseases decreased on both sides of the biome border, as shown in the summary statistics in Appendix Table 4.A.2. Still, we estimate that Amazon biome focused conservation policies further reduced hospitalizations due to respiratory health problems significantly by 1.71 (± 1.62) cases per 1,000 inhabitants per municipality per year. This link between air pollution and respiratory health problems is well-documented in the literature (Liu et al., 2015; Reid et al., 2016; Chen and Hoek, 2020). Significant and positive health impacts were also found for the circulatory system, lung cancer, and health problems related to pregnancy. The overall treatment effect of the prevalence rate of STDs is insignificant, as expected. The 2010 population census counted around 4.12 million people living within 100 km of the Amazon biome. In absolute terms, the treatment effects amount to around 7,061 (± 6,676), 4,180 (± 2,659), 6,430 (± 5,539) and 46 (± 53) fewer hospitalizations related to respiratory health problems, the circulatory system, pregnancy and lung cancer per year, respectively. As the average hospitalization in the sample cost around 324 United States (US)-Dollars (USD), this amounts to a direct cost reduction of USD 5.74 million (± 4.84) per year (in 2010 USD rate).

Our health data also allow us to estimate the effect on mortality associated with the same diseases or health problems. Since outdoor air pollution is estimated to cause millions of premature deaths worldwide every year, mortality is an important outcome variable (Johnston et al., 2012; Lelieveld et al., 2015). The results are presented in Appendix Table 4.A.6. We also find significant reductions in deaths associated with respiratory problems. The effect size translates to 7.2 (± 2.9) less deaths per 100,000 municipality-inhabitants, or 296 (± 120) in total per year. Additionally, the treatment effects of the death rate due to the circulatory system, pregnancy-related health problems, and the perinatal period are also significant and negative, amounting to 284 (± 162), 15 (± 17) and 83 (± 61) less deaths per year, respectively. Aggregating all effects leads to a reduction of around 678 deaths per year for our local sample.

4.3 Discussion

Currently, research on tropical forest conservation has mainly focused on the environmental benefits of reducing deforestation and forest degradation. This study demonstrates that conservation initiatives can provide sizable public health benefits if associated with a reduction in human-induced forest fires and related air pollution.

Specifically, we demonstrate that biome-focused conservation initiatives and policies in the Brazilian Amazon have reduced the number of fires by almost 20% and improved air quality in affected municipalities by reducing particulate matter concentrations (1.97 $\mu g/m^3$ $PM_{2.5}$ on average). Previous studies have shown that biome-focused conservation efforts had been successful in reducing deforestation. Heilmayr et al. (2020) show that the Amazon Soy Moratorium reduced deforestation by 35%, while Assunção, Gandour, Rocha, and Rocha (2019) estimate that BCB rural credit restrictions reduced deforestation by 60%. We further demonstrate that air quality improvements resulted in significant health benefits for the local population. Hospitalizations linked to fire and smoke-related health issues dropped by almost 18,000 per year and around 680 annual deaths were avoided in a sub-sample population of approximately 4.12 million. Reductions in respiratory health problems were the overall largest contributor to these health improvements. Results for different sub-samples and the full sample can be found in Appendix Tables 4.A.7–4.A.9. Since not all health problems require hospitalizations, it is possible that the true health benefits of reduced fire smoke are even larger. Regarding cost savings, we only calculate direct cost reductions of reduced hospitalizations, which we estimate at almost USD 6 million per year. True economic benefits might lie much higher, if one considers, for example, the reduction of number of sick workdays and increased productivity due to less health issues—hospitalized or not—or the costs of premature death. Overall, our findings indicate that tropical forest conservation has a significant public health premium that has so far remained unaccounted for (Tallis et al., 2008). Future investments in public conservation programs should be evaluated against benchmarks that consider all relevant One Health dimensions (S. Myers et al., 2013; Strand et al., 2018; Ellwanger et al., 2020).

Some potential limitations of this study are worth noting. First, there are obvious climatic and biological differences between the Amazon and the Cerrado biomes separated by the border that our identification strategy is based on. Apart from controlling for a variety of bio-climatic and socio-economic variables, we try to overcome this problem by using data on areas that are at close distance to the biome border. Environmental and socio-economic conditions do not change abruptly at the formal biome border. Instead, one biome continuously transforms into the other often along a south-north gradient in what is called an ecotone (Marimon et al., 2006). Furthermore, as the biome border is not an administrative boundary, our results are unlikely to be driven by other policy effects. Thus, the double-difference estimator should adequately control for the remaining unobserved differences as long as they remain constant over time or develop in parallel. In Appendix Table 4.A.10 we report the results of a parallel trends test for the pre-treatment period. The test is rejected for hospitalizations due to diseases of the circulatory system and health problems originating during the perinatal period, and the air pollutants nitrogen dioxide and ozone. This means the parallel trends assumption may not hold for these outcomes and we cannot rule out the possibility that other factors are driving and biasing their estimated effects reported in the results section and in the Appendix Tables 4.A.3–4.A.9. For the main outcome variables—number of fires, $PM_{2.5}$ concentrations, and hospitalizations and mortality due to respiratory health problems—we cannot reject the hypothesis of parallel trends, which strengthens the robustness and credibility of the double-difference approach for these outcomes. Second, due to the scale of our analysis, we have to rely on remotely sensed measures of fire occurrences and air pollution. While such indicators are prone to measurement error due to cloud cover, we do not expect systematic sources of measurement error within the relatively small bandwidth used as a study area for our empirical analysis. Third, we use a dichotomous indicator for conservation policies, which enables us to estimate treatment effects but does not allow us to attribute precise increments in air pollution reduction to the number of hospitalizations.

Finally, we do not claim that the health improvements we quantify here can be replicated by private sector initiatives like the Amazon Soy Moratorium or the Cattle Agreement anywhere outside the specific policy context of the Brazilian Amazon. The Soy Moratorium was launched after the Brazilian federal government had and continued to introduce a variety of public policies to reduce deforestation, including rigorous monitoring and enforcement measures (Börner, Kis-Katos, et al., 2015), which partly enabled the Soy Moratorium. In fact, deforestation and forest fires in the Amazon increased again after many of these conservation efforts were abolished by the Bolsonaro government (Abessa, Famá, and Buruaem, 2019; Barlow et al., 2020). This indicates that several conditions must be in place for private sustainability governance to effectively result in tangible environmental and health benefits. Their effectiveness might be compromised by leakage if they operate on small market shares or at limited regional scales (Paim, 2021; Villoria et al., 2022).

4.4 Methods

4.4.1 Data

We limited our data collection process to the five states at the Amazon biome border—Maranhão, Tocantins, Pará, Mato Grosso and Rondônia. The data we used for estimation is a subset of this initial dataset but always contained information from all five states, independent of the bandwidth chosen around the biome border. Figure 4.1 depicts the geographic area covered by the five states and their municipalities, the location of the municipality capitals, and the Amazon biome border as well as the BLA region and a 100 km buffer around the biome border. The unit of observation is the municipality. Almost all municipalities studied lie in the Legal Amazon, except for a small number of municipalities in the North-Eastern region of Maranhão, which are close to the biome border but not part of the Legal Amazon anymore. Since air pollution concentrations were measured in the municipality capital cities, assignment to the treated or untreated group depended on the location of the capital. Municipalities with a capital located inside the Amazon biome were assigned to the treated group even if parts of the municipality lie outside the biome. The creation of municipalities in Brazil closely follows the distribution of cities, to the extent that municipalities are named after their capital city (Law 311/1938). There cannot be a municipality without a city from which it originates. As the sampled area is mostly remote and rural, quite often there is only one city per municipality—the capital. With an urbanization rate of close to 70%, according to the 2010 population census, the majority of people reside in cities. The median (mean) city size within 100 km of the biome border is 11,500 (23,162) inhabitants, respectively. Following this treatment assignment, effects on air pollution and subsequent effects on health should be adequately estimated. However, we can be more precise with the estimation of effects on the number of fires. Each fire is identified by geographic point coordinates of latitude and longitude, which means we can actually determine whether a fire burned inside or outside the biome within the same municipality.

Data on fires was downloaded from Brazil's National Institute for Space Research (INPE, 2021) Queimadas program. Fires are measured by satellites if it finds one or more fires in a given pixel raster (1 km x 1 km). Large fires that spread over multiple kilometers are thus counted as multiple fires. Fires that are smaller than 30 m^2 are not detected. Similarly, burnings shorter than the interval between two consecutive satellite images and fires below clouds or tree canopies are undetected. For the purpose of counting, we did not differentiate between burning intensities, fire size, or area burned, apart from counting a fire multiple times if it spreads over a larger area. Hot pixels also generally underestimate the area burned. The number of fires were aggregated by year and municipality but also whether they burned inside or outside the biome.

Meteorological and air pollution indicators were retrieved from the Health Related Environmental Information System (SISAM, 2020), which is part of INPE. It gathers data like precipitation and fine particulate matter concentrations at the geographic center of each municipal capital in six hour intervals. Meteorological and air pollution data was averaged by year. Fire and air pollution summary statistics for the pre- and post-2007 period by treatment group can be found in Appendix Table 4.A.1, together with WHO (2021) guidelines on safe exposure targets. The summary statistics of all other variables are in Appendix Table 4.A.2.

The health data is from DATASUS, the informatics department of Brazil's public and free health care system (SUS, 2020), which covers virtually all Brazilians. DATASUS allows access to an anonymized micro-level database containing information such as the municipality of residence and the hospital, age, postal code, and ICD-10 code for each individual hospitalization. We aggregated this data by the municipality of residence, year of admission and ICD-10 code. We also added the number of health professionals and health establishments for each municipality from DATASUS as control variables. Additional socio-economic information, such as gross domestic product (GDP) and population size, was downloaded from the Brazilian Institute of Geography and Statistics (IBGE, 2024).

We chose 2003 as the first year in our analysis since the reference satellite for fire measurement changed in 2002, introducing an inconsistency in the time series. The new reference satellite also allows for the measurement of more air pollutants. In addition, several new municipalities that were created from municipality divisions in 2002 were added to our sample as independent observations. Later divisions of municipalities were aggregated into their original administrative borders, which follows the idea of minimum comparable areas proposed by Ehrl (2017). The last year of observation is 2017.

4.4.2 Main empirical analysis

Our identification strategy explores a regulatory discontinuity at the Amazon biome border, which has been used previously by different authors to estimate deforestation reduction impacts of various conservation policies (Assunção, Gandour, Rocha, and Rocha, 2019; Heilmayr et al., 2020; Moffette and Gibbs, 2021). Discontinuities can often be exploited in a regression discontinuity design (RDD). The basic idea in this setting would be to compare municipalities that are very close to the Amazon biome border, assuming that municipalities very close to the border are more similar to each other, for example climatically and economically, and are, in fact, comparable. Indeed, all mentioned previous studies that use the Amazon biome border cutoff restrict their observations to some close distance around the border. In this case, the effect of an intervention could be estimated by the difference in outcomes between municipalities just inside and municipalities just outside the biome. Additionally, our data consists of repeated observations of the same municipalities over time, establishing a panel data structure. Double-difference estimators exploit this panel structure by making it possible to control for unobserved time-invariant factors, e.g. geographic factors, through fixed effects. The main assumption is that there is a parallel trend in outcomes between municipalities which would have continued in the absence of an intervention, such as conservation policies. In Appendix Figure 4.A.6 the parallel trends assumption can be inspected visually by comparing the time path of the average number of fires just inside vs. outside the biome. The intervention effect can then be estimated by calculating the difference of the differences between municipalities inside and outside the biome, and before and after an intervention. Both designs rely on creating a counterfactual scenario to predict what outcomes would have been in the absence of an intervention, e.g. how many fires would have burned had there not been a conservation policy (Frölich and Sperlich, 2019). It is possible to combine both approaches, for example, by estimating the intervention effect as the difference between the discontinuities before and after the intervention (Grembi, Nannicini, and Troiano, 2016; Butts, 2021). The combination of approaches relaxes the underlying assumptions of the individual approaches. The parallel trends assumption in double-difference estimation only needs to hold locally for the subset of selected municipalities at close distance to the biome border, while controlling for unobserved time-invariant confounders makes the RDD assumption more likely to hold, as close-by municipalities only need to be similar in time-variant factors.

However, in this case the discontinuity applies to a geographic context as described by Keele and Titiunik (2015), who extended the RDD framework to spatial discontinuities, i.e., using geographic boundaries as discontinuities. As Keele and Titiunik (2015) pointed out, the position alongside a border matters in a geographic context, as two locations with equal distance to a long border are not comparable if they have very different north and south, east and west, or high and low altitude positions along the border. Therefore, a difference-in-discontinuities comparison as in Appendix Figure 4.A.7, which depicts the aggregate annual number of fires by distance to the biome border before and after 2007, is not adequate, as it only considers a one-dimensional distance to the border running variable but does not account for the two or three dimensional aspects of space. How exactly to control for these spatial characteristics depends on different factors that characterize the model setting, e.g. actual geographic location, length of the border or any specific modeling aspects concerning the outcome variable considered. Following previous studies, we control for border segment fixed effects, i.e. splitting the border into segment subsamples to ensure that only municipalities within the same segment are compared (cf. Appendix Figure 4.A.8), and a linear polynomial of latitude and longitude to control for smoothly distributed spatial confounding factors (Dell, Lane, and Querubin, 2018; Wuepper, Crowther, et al., 2024). Both of these spatial controls are interacted with time dummy variables to capture location specific changes over time.

A common problem in spatial RDD is that legal borders are often used as discontinuities, but when considering, for example, country borders, differences can hardly be attributed to a specific intervention. In our case, state or federal laws concerning the Amazon are applied to the defined administrative boundaries of the BLA and thus basically to our entire sample, with the exception of the three mentioned policies (Soy Moratorium, BCB credit restrictions and the Cattle Agreement). Moreover, double-difference estimators are particularly powerful to control for previously existing differences in policies if they do not change over time (Butts, 2021; Wuepper and Finger, 2022).

One particular challenge in our setup is the mobility of fire smoke. The impact of reducing the number of fires, i.e. the corresponding smoke, does not respect borders. Smoke and its pollutants disperse aerially, which reduces the likelihood of a sharp discontinuous jump in air pollution and health outcomes at the border. Therefore, we do not include a one-dimensional distance to the border running variable in our model, which means that the estimation resembles a traditional double-difference setup, where treatment is assigned according to the border cutoff and only a certain bandwidth subsample around the border is selected.

Summarizing the discussion above, our estimated equation takes the following form

$$Y_{i,t} = \alpha + \beta(\text{Inside}_i \times \text{After}_t) + \gamma X_{i,t} + \lambda_i + \tau_t + \phi(f(g_i) \times \text{After}_t) + \sigma_{j,t} + \epsilon_{i,t} \tag{4.1}$$

where $Y_{i,t}$ is the (number of fires, air pollutant or health) outcome variable in municipality i in year t; Inside_i indicates whether the capital city of municipality i is inside the Amazon biome or not; After_t indicates whether year t is before or after 2007; $X_{i,t}$ is a set of time-variant bio-climatic, socio-economic and health-related control variables, which are shown in Appendix Tables 4.A.2 and 4.A.11; τ_t and λ_i are year and municipality fixed effects, respectively; $f(g_i)$ is a linear polynomial of geographic location that is interacted with the After_t period dummy (Specifically, the polynomial takes the form lon + lat + lon × lat, where lon is longitude and lat is latitude of the capital.); $\sigma_{j,t}$ is the interaction term between border segment dummies j with the After_t period dummy to control for changes in spatial patterns over time; and $\epsilon_{i,t}$ is the error term. The treatment effect is given by β. The model is estimated via ordinary least squares regression using the `fixest` package (Bergé, 2018) in R Statistical Software (R Core Team, 2021). All standard errors are clustered at the municipality level. Two-sided t-tests are used to test the null hypothesis whether the β coefficient is equal to zero (no effect). We set 2007 as the first treatment year.

One important methodological choice is bandwidth selection, i.e., the subsample from the initial dataset used for estimation. For the RDD assumption of more similar units at a closer distance to hold, we cannot set the bandwidth too large; however, if we set it too narrow, we risk a mix-up of smoke and treatment effects. In fact, there is no clear bandwidth selection process when considering multiple outcomes and a multidimensional RDD (Dell et al., 2018).

Smoke exposure from controlled burnings is mainly a function of distance to the fire and wind direction (Pearce et al., 2012). While the highest exposure occurs in close vicinity to the fire, air pollution increases again with the settling of the plume. Empirical evidence about the Amazonian region suggests that air pollution effects from local fires were measurable even up to 100 km away (Fonseca Morello, 2023). Ultimately, we use a bandwidth of 100 km—a subsample that does not include the very large state capitals of Cuiabá, Mato Grosso and Palmas in Tocantins, where effects of urban air pollution could confound our estimates. This seems to offer a good trade-off between enough distance to the border to prevent smoke mixing but close enough to ensure comparability. Similar-sized bandwidths were also used in other studies of the Amazon region (Fonseca Morello, 2023; Sheehan et al., 2023). The results from estimations with other bandwidths can be found in Appendix Tables 4.A.7–4.A.9 and are quite similar to our preferred bandwidth.

4.4.3 Robustness checks

One of the crucial assumptions in the double-difference estimations is the parallel trends assumption. Graphically, this means that the two lines indicating outcomes for the treatment and control groups should be parallel prior to treatment. Appendix Figure 4.A.6 depicts the development of the aggregated number of fires by treatment over time within 100 km of the Amazon biome border. Visually, the number of fires just inside the biome seem to develop mostly in parallel to the number of fires just outside the biome before 2007.

We estimate a placebo double-difference treatment effect on the pre-treatment period from 2003 to 2006 on all our outcome variables and the results are presented in Appendix Table 4.A.10. We find significant pre-treatment differences for the air pollutants nitrogen dioxide and ozone, as well as the hospitalization rate for diseases of the circulatory system and health problems originating during the perinatal period. This is a problem for these outcome variables, as we cannot rule out that other factors are driving the effect results, especially for the significant treatment effects found for nitrogen dioxide and the hospitalization rate due to circulatory system health problems. However, for the primary outcomes of interest, namely the number of fires, $PM_{2.5}$ concentrations and the hospitalization/death rate due to respiratory health problems we cannot reject the assumption of parallel pre-treatment trends. This strengthens the robustness and credibility of the double-difference approach for these outcomes.

Another concern are urban and other sources of air pollution, which could lead to larger air pollution concentrations measured in the capital cities and also to health problems. Considering Appendix Figure 4.A.5, we see a strong correlation between acute elevated $PM_{2.5}$ exposure and fire occurrences, even taking $PM_{2.5}$ background levels into account. For this reason, we are confident that the source of elevated air pollution are the fires. In the off-fire-season $PM_{2.5}$ levels vary around 10 $\mu g/m^3$, so urban air pollution seems to be neither increasing nor decreasing. In any way, our effects are estimated using a double-difference approach, so even if emissions from urban sources decreased or increased over time it would only affect our estimates if this change disproportionally affected only one side. To further check how prevalent disproportional changes are, we also estimate treatment effects on all of our control variables. Appendix Table 4.A.11 presents the results of this estimation exercise. We find no effect for GDP per capita or population density, which should both correlate to urban air pollution. Most treatment effects on socio-economic and health control variables are insignificant, except for the density of doctors per municipality. Regarding the bio-climatic control variables, we find a significant effect for humidity. As conservation policies targeted at deforestation that can influence the local ecosystem, we cannot rule out effects on the bio-climatic variables. However, the effects on precipitation, wind speed, wind direction and temperature are not significant. Overall, results suggest that treatment and control group are indeed comparable and mostly follow parallel trends.

Appendix 4.A Appendix to: Health Benefits of Reduced Deforestation in the Brazilian Amazon

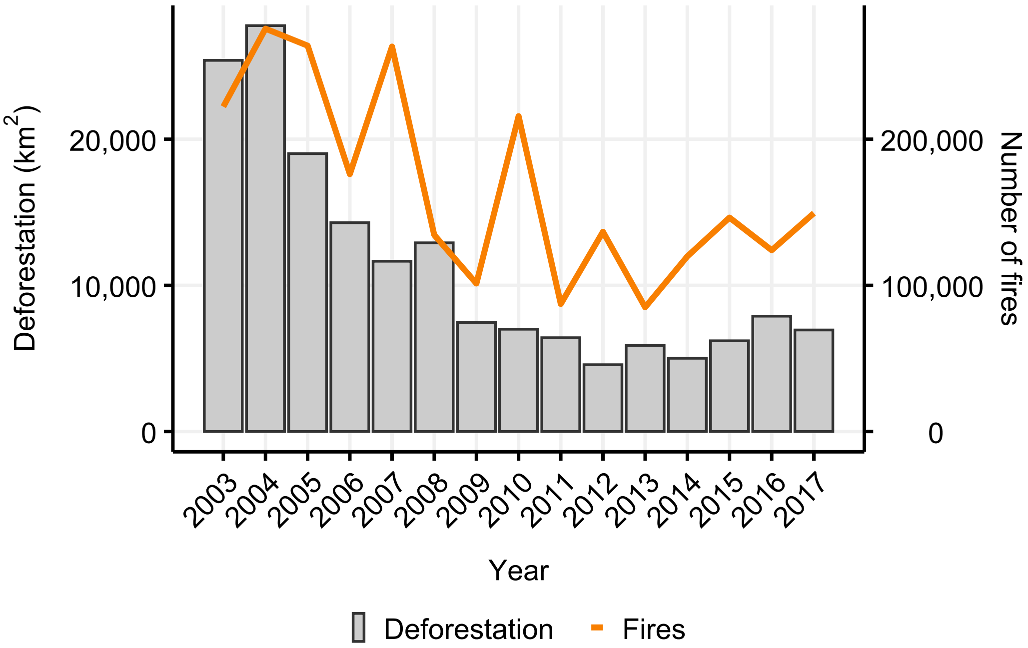
This Appendix includes:

- Appendix figures 4.A.1 to 4.A.8 in Section 4.A.1
- Appendix tables 4.A.1 to 4.A.11 in Section 4.A.2

All the data and code necessary to replicate all results, figures and tables from this study, including from this Appendix, are available at the research data repository bonndata (Damm, 2024) and can be accessed using the following link https://doi.org/10.60507/FK2/OWKNFV.

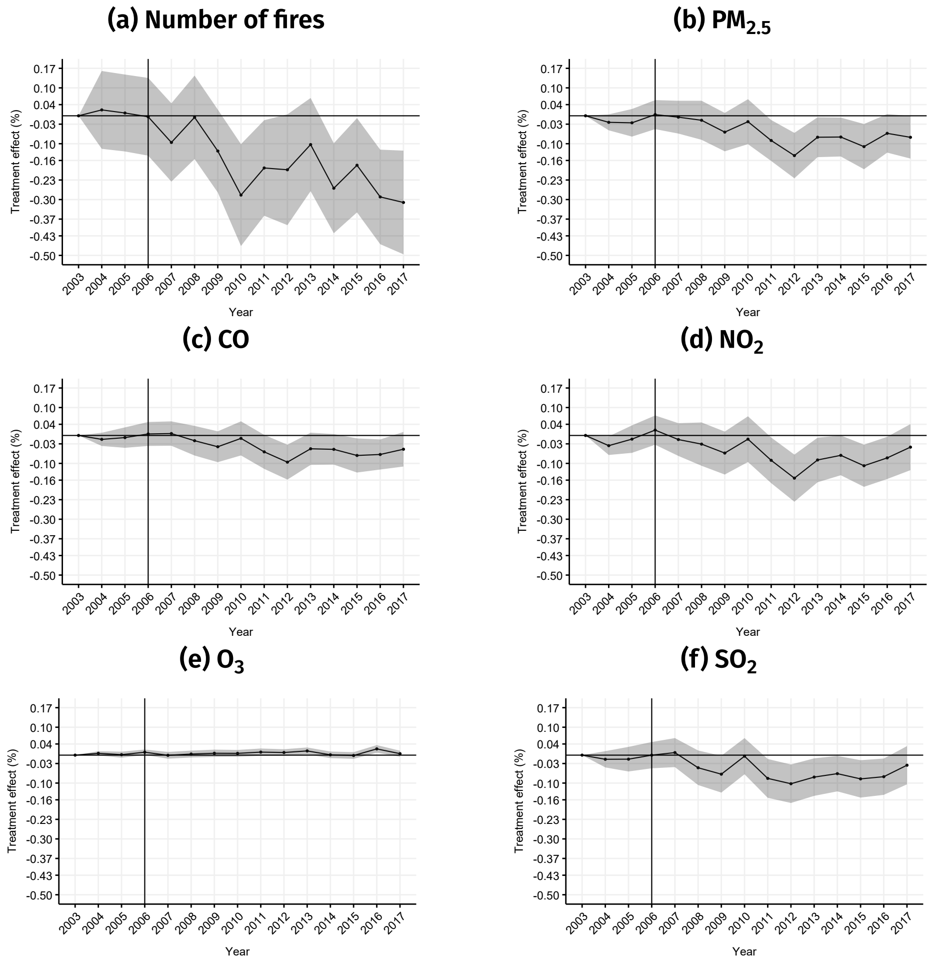
4.A.1 Appendix figures

Figure 4.A.1. Deforestation and number of fires per year in the Brazilian Legal Amazon



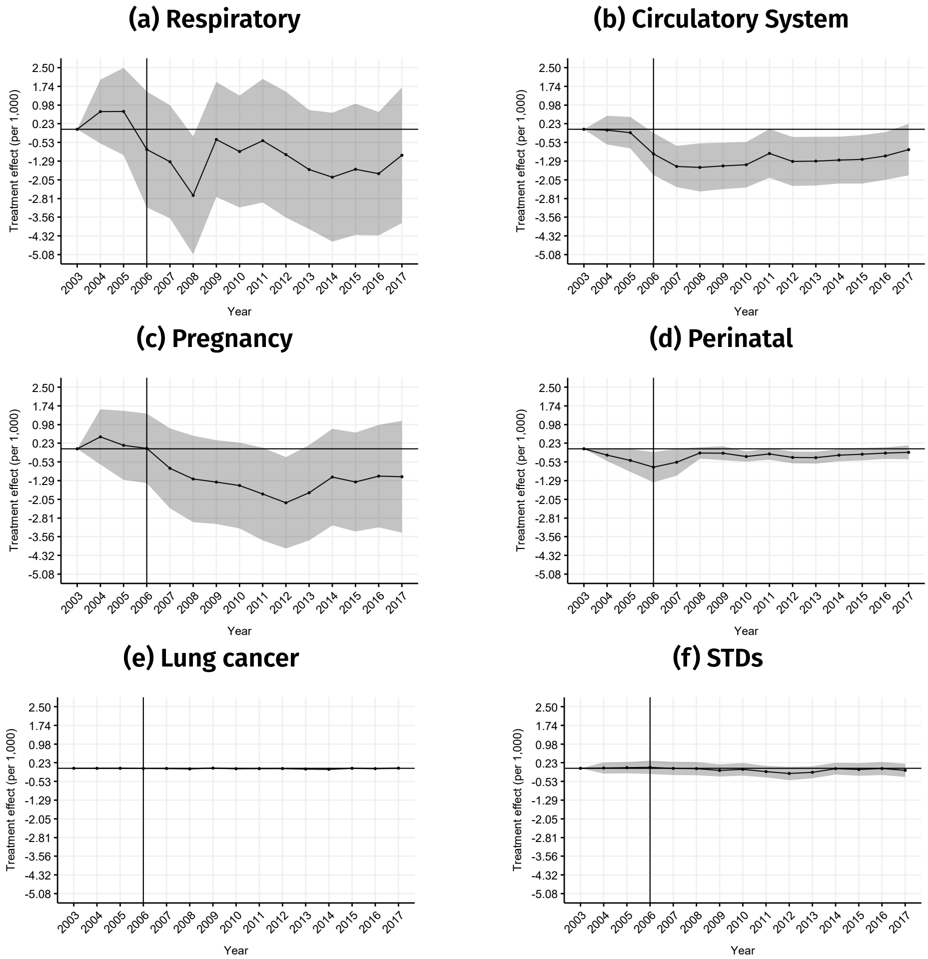
Note: Deforestation is shown by the grey bars and number of fires by the solid orange line. *Sources*: Deforestation: TerraBrasilis, Assis et al. (2019); Fires: INPE (2021)

Figure 4.A.2. Yearly treatment effects for fire and air pollution variables, including a 95% confidence interval



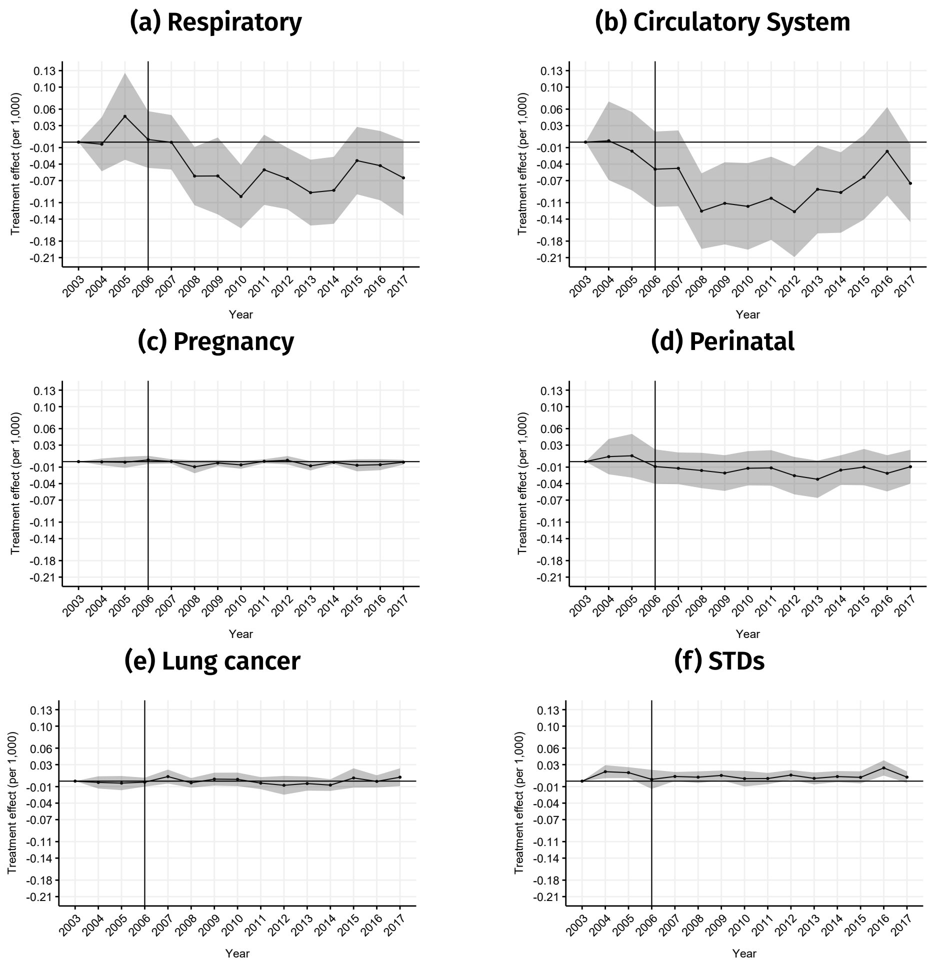
Note: Effect sizes, represented by the solid black line with round points, are in decimal percentages. The shaded grey area shows 95% confidence intervals. **(a)** Is the treatment effect for the number of fires. **(b)** $PM_{2.5}$ is particulate matter with a diameter of less than 2.5 ***μm***. **(c)** CO is carbon monoxide. **(d)** NO_2 is nitrogen dioxide. **(e)** O_3 is ozone. **(f)** SO_2 is sulfur dioxide.

Figure 4.A.3. Yearly treatment effects for hospitalizations, including a 95% confidence interval



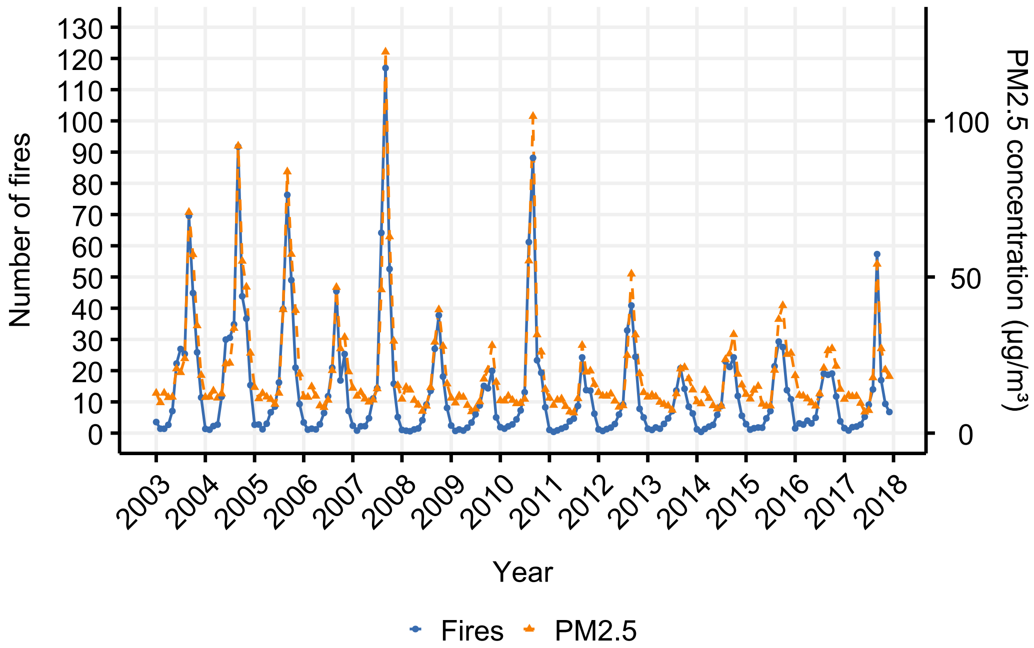
Note: Effect sizes, represented by the solid black line with round points, are per 1,000 inhabitants. The shaded grey area shows 95% confidence intervals. **(a)** Is the treatment effect for hospitalizations due to diseases of the respiratory system. **(b)** is due to diseases of the circulatory system. **(c)** is due to conditions related to pregnancy. **(d)** is due to conditions originating in the perinatal period. **(e)** is due to lung cancer. **(f)** is due to sexually transmitted diseases.

Figure 4.A.4. Yearly treatment effects for mortality, including a 95% confidence interval



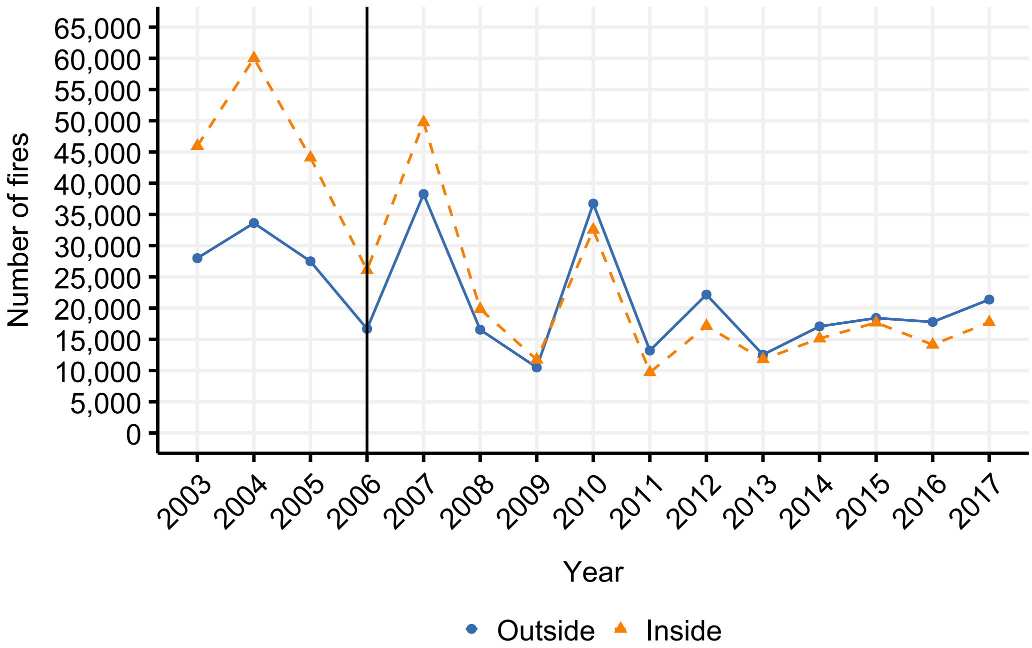
Note: Effect sizes, represented by the solid black line with round points, are per 1,000 inhabitants. The shaded grey area shows 95% confidence intervals. **(a)** Is the treatment effect for mortality due to diseases of the respiratory system. **(b)** is due to diseases of the circulatory system. **(c)** is due to conditions related to pregnancy. **(d)** is due to conditions originating in the perinatal period. **(e)** is due to lung cancer. **(f)** is due to sexually transmitted diseases.

Figure 4.A.5. Average monthly number of fires and $PM_{2.5}$ concentrations



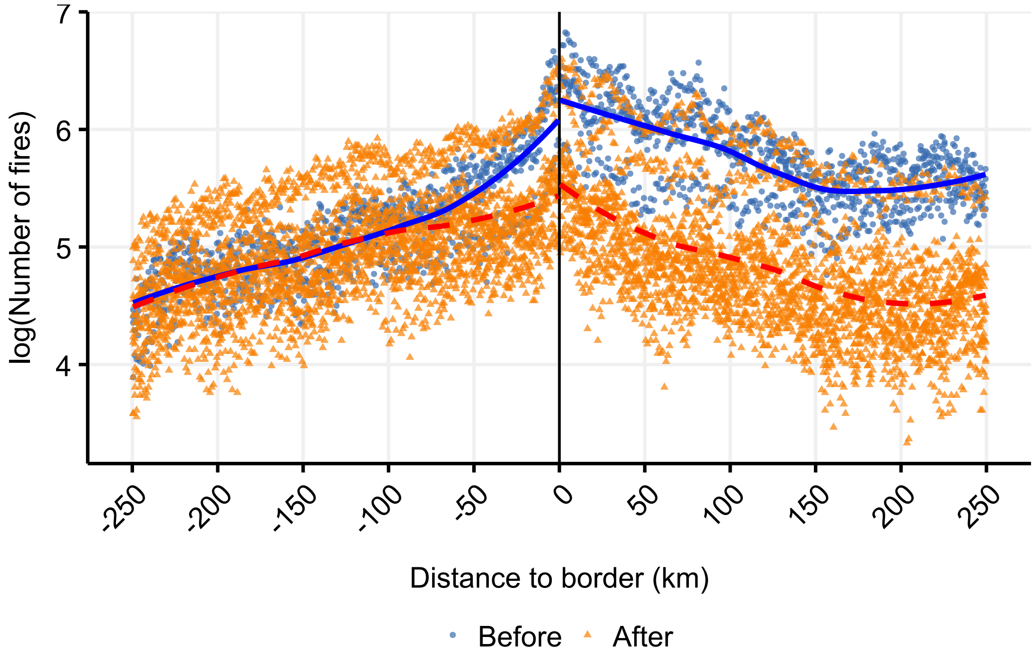
Note: Monthly averages of number of fires (blue solid line with round points) and $PM_{2.5}$ concentrations (orange dashed line with triangular points) of all municipalities within 100 km of the Amazon biome border for the period 2003–2017. *Source*: Author's own calculation based on SISAM (2020) data.

Figure 4.A.6. Aggregated annual number of fires



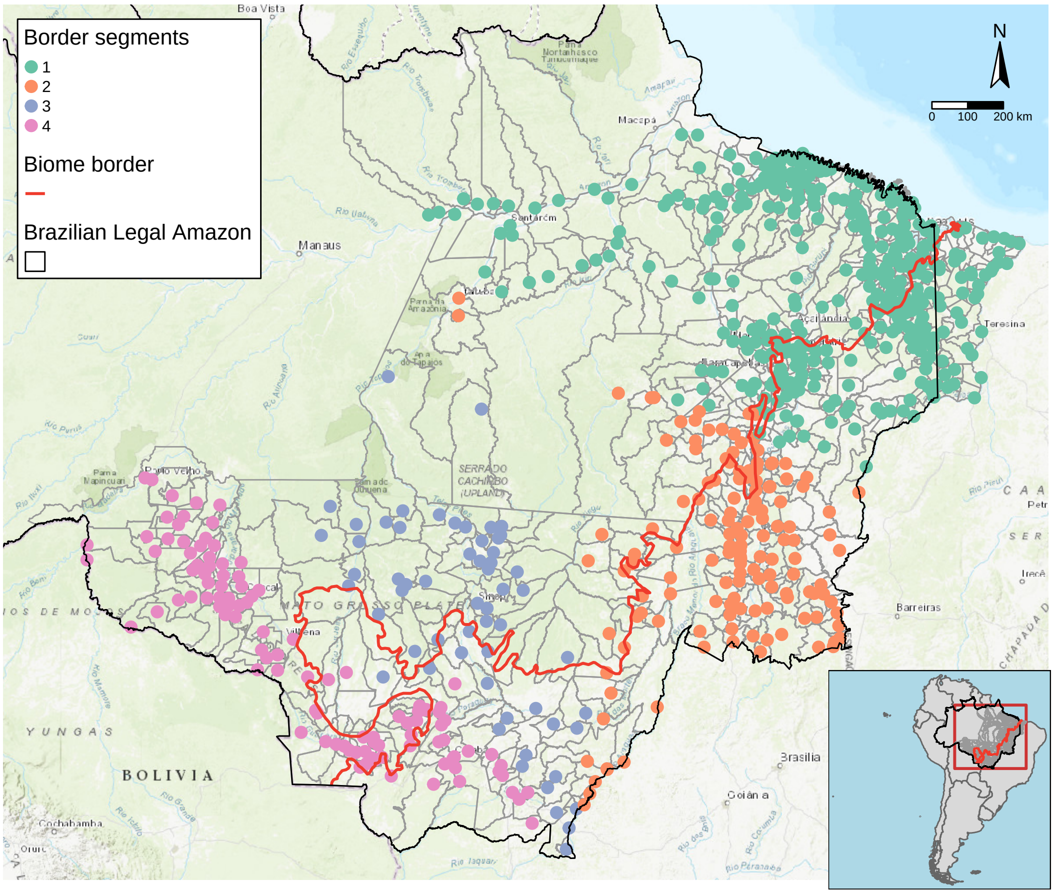
Note: Comparison of annually aggregated number of fires just inside vs. just outside the Amazon biome border and no more than 100 km away from the border. The blue solid line with round points represents the number of fires outside the biome and the orange dashed line with triangular points represents the number of fires inside. *Source*: Author's own calculation based on INPE (2021) data.

Figure 4.A.7. Difference-in-discontinuities in the number of fires



Note: Fires are aggregated annually in 1 km distance-to-the-biome-border bins. Negative distances refer to outside the Amazon biome positive distances are inside the biome. Before refers to the period 2003–2006, after to the period 2007–2017. Each blue round point represents the log of the sum of fires within that 1 km bin in one before-period year. Orange triangular points represent the same for the after-period. Non-parametric LOESS regression lines were added to visualize potential non-linearities in the data. The solid blue lines show LOESS regression lines for the before period (inside and outside the biome) and the dashed red lines show LOESS regression lines for the after period (inside and outside the biome). *Source*: Author's own calculation based on INPE (2021) data.

Figure 4.A.8. Distribution of border segments



Note: Each municipality was assigned a border segment. Interaction terms between border segment and time dummies in the regression control for potential location-specific changes over time. There are four border segments along the Amazon biome border, represented by green, orange, blue and pink dots. Background map source: World Topographic Map. Esri, TomTom, Garmin, FAO, NOAA, USGS, ©OpenStreetMap contributors, and the GIS User Community

4.A.2 Appendix tables

Table 4.A.1. Summary statistics of fire, air pollution, and acute exposure to elevated short-term air pollution levels

	Before		After		WHO (WHO, 2021) air
Variable	Outside	Inside	Outside	Inside	quality guidelines
Number of fires	165.65	252.21	120.3	117	
	(302.3)	(443.91)	(207.44)	(246.33)	
$PM_{2.5}$	23.23	27.52	17.91	18.07	5 (annual average)
	(11.88)	(19.9)	(6.81)	(10.66)	15 (24h average)
$PM_{2.5}$ above guideline (%)	37.12	35.84	31.66	28.54	
	(9.27)	(11.62)	(9.85)	(10.23)	
CO	264.6	309.7	212.2	213.58	4,000 (24h average)
	(120.23)	(216.4)	(63.69)	(108.72)	10,000 (8h average)
CO above guideline (%)	0.03	0.13	0.01	0.04	
	(0.12)	(0.34)	(0.06)	(0.18)	
NO_2	2.48	2.53	2.18	1.75	10 (annual average)
	(1.2)	(1.88)	(0.91)	(1.07)	25 (24h average)
NO_2 above guideline (%)	0.83	1.3	0.52	0.48	
	(1.71)	(2.59)	(1.15)	(1.38)	
O_3	31.85	30.85	32.5	30.48	60 (6 months average)
	(4.97)	(7)	(5.27)	(7.69)	100 (8h average)
O_3 above guideline (%)	2.7	2.26	2.25	1.36	
	(1.69)	(1.98)	(2.01)	(1.77)	
SO_2	0.82	0.96	0.64	0.61	40 (24h average)
	(0.45)	(0.69)	(0.3)	(0.38)	
SO_2 above guideline (%)	0.02	0.11	0.01	0.03	
	(0.1)	(0.38)	(0.07)	(0.2)	

Note: Numbers represent averages of each group in each period. The outside group refers to municipalities whose capitals lie outside the Amazon biome but no further than 100 km away from the border (correspondingly for the inside group). The before period is from 2003 to 2006, and the after period from 2007 to 2017. Number of fires are the sum of fires per year per municipality. Air pollution concentrations (in micrograms per cubic meter) are annual averages from six hour interval measurements. Above guideline refers to the percentage of time intervals per year where the respective air pollution levels exceeded the WHO short-term exposure guidelines. $PM_{2.5}$ is particulate matter with a diameter of less than 2.5 μm; CO is carbon monoxide; NO_2 is nitrogen dioxide; O_3 is ozone; and SO_2 is sulfur dioxide. Standard deviations in parentheses.

Table 4.A.2. Summary statistics of health outcome and control variables

	Before		After	
	Outside	Inside	Outside	Inside
Control variables				
Temperature (°C)	26.97 (0.85)	26.43 (1.08)	27.03 (0.92)	26.46 (1.12)
Precipitation (mm/day)	1.1 (0.21)	1.18 (0.23)	0.93 (0.23)	0.98 (0.24)
Humidity (%)	73.67 (3.24)	77.33 (3.69)	76.4 (5.8)	79.46 (5.34)
Wind speed (m/s)	1.44 (0.76)	1.52 (0.97)	1.42 (0.77)	1.47 (0.98)
GDP per capita (1,000R$/pop)	2.58 (4.32)	2.15 (1.68)	5.77 (7.19)	4.97 (4.02)
Agricultural GDP share	0.27 (0.15)	0.29 (0.16)	0.23 (0.14)	0.24 (0.15)
Population density (pop/km^2)	13.94 (19.46)	36.04 (123.84)	14.75 (22.9)	31.53 (117.99)
Doctors (per 1,000)	4.99 (2.53)	4.79 (2.65)	8.95 (3.28)	8.57 (2.81)
Health establishments (per 1,000)	0.54 (0.31)	0.61 (0.35)	0.81 (0.45)	0.88 (0.49)
Wind direction (in degree)	75.51 (55.49)	93.96 (95.92)	72.26 (44.6)	92.29 (94.83)
Hospitalizations per 1,000				
Respiratory	10.12 (8.21)	11.87 (8.53)	7.85 (6.63)	9.37 (7.97)
Circulatory system	4.26 (3.04)	5.04 (3.85)	4.28 (2.58)	4.47 (2.82)
Pregnancy	16.33 (6.74)	15.93 (5.9)	13.81 (5.08)	12.19 (4.9)
Perinatal	0.83 (1.09)	0.98 (2.16)	0.97 (0.9)	0.8 (1.14)
Lung cancer	0.02 (0.07)	0.03 (0.06)	0.05 (0.12)	0.04 (0.09)
STDs	0.48 (0.7)	0.79 (0.82)	0.37 (0.53)	0.51 (0.64)
Deaths per 1,000				
Respiratory	0.12 (0.16)	0.14 (0.19)	0.21 (0.23)	0.18 (0.18)
Circulatory system	0.22 (0.23)	0.21 (0.25)	0.31 (0.26)	0.23 (0.23)
Pregnancy	0 (0.02)	0.01 (0.03)	0.01 (0.03)	0 (0.02)
Perinatal	0.05 (0.08)	0.05 (0.09)	0.06 (0.1)	0.05 (0.08)
Lung cancer	0.01 (0.03)	0 (0.02)	0.01 (0.05)	0.01 (0.04)
STDs	0.01 (0.04)	0.01 (0.04)	0.01 (0.04)	0.01 (0.03)

Note: Numbers represent averages of each group in each period. The outside group refers to municipalities whose capitals lie outside the Amazon biome but no further than 100 km away from the border (correspondingly for the inside group). The before period is from 2003 to 2006, and the after period from 2007 to 2017. Temperature is in degree Celsius; precipitation is in millimeters per day; humidity is in percent; wind speed in meters per second; GDP per capita is in thousand Brazilian Reais in 2012 constant prices; agricultural GDP share is the GDP share of the agricultural sector; population density is inhabitants per square km; doctors and health establishments are per 1,000 inhabitants; and wind direction is in degrees. Hospitalizations and deaths are per 1,000 inhabitants associated with the named health problem/disease according to the ICD-10 classification. STDs are sexually transmitted diseases. Standard deviations in parentheses.

Table 4.A.3. Estimated treatment effects for fire and air pollution variables

	Number of fires (1)	$PM_{2.5}$ (2)	CO (3)	NO_2 (4)	O_3 (5)	SO_2 (6)
Treatment effect	-0.192***	-0.066***	-0.049**	-0.070**	0.001	-0.060**
	(0.043)	(0.025)	(0.020)	(0.029)	(0.004)	(0.024)
Observations	5,955	4,395	4,395	4,395	4,395	4,395
R^2	0.683	0.828	0.893	0.833	0.963	0.776
Municipality fixed effects (293)	✓	✓	✓	✓	✓	✓
Year fixed effects (15)	✓	✓	✓	✓	✓	✓

Note: Coefficients are given as decimal percentage changes. All regressions use year and municipality fixed effects. Time variant control variables are discussed in the Methods section. $PM_{2.5}$ is particulate matter with a diameter of less than 2.5 μm; *CO* is carbon monoxide; NO_2 is nitrogen dioxide; O_3 is ozone; and SO_2 is sulfur dioxide. Standard errors (in parentheses) are clustered at the municipality level.
*p<0.1; **p<0.05; ***p<0.01.

Table 4.A.4. Estimated treatment effect on acute exposure to short-term elevated air pollution

	$PM_{2.5}$ above guideline (1)	CO above guideline (2)	NO_2 above guideline (3)	O_3 above guideline (4)	SO_2 above guideline (5)
Treatment effect	-0.015***	0.000	-0.002	-0.001	-0.001**
	(0.005)	(0.000)	(0.002)	(0.001)	(0.000)
Observations	4,395	4,395	4,395	4,395	4,395
R^2	0.902	0.442	0.586	0.802	0.388
Municipality fixed effects (293)	✓	✓	✓	✓	✓
Year fixed effects (15)	✓	✓	✓	✓	✓

Note: Coefficients are given as percentage point changes. All regressions use year and municipality fixed effects. Time variant control variables are discussed in the Methods section. $PM_{2.5}$ is particulate matter with a diameter of less than 2.5 μm; *CO* is carbon monoxide; NO_2 is nitrogen dioxide; O_3 is ozone; and SO_2 is sulfur dioxide. Standard errors (in parentheses) are clustered at the municipality level.
*p<0.1; **p<0.05; ***p<0.01.

Table 4.A.5. Estimated treatment effect on hospitalizations

	Hospitalizations per 1,000					
	Respiratory	Circulatory system	Pregnancy	Perinatal	Lung cancer	STDs
	(1)	(2)	(3)	(4)	(5)	(6)
Treatment effect	-1.712**	-1.014***	-1.559**	0.121	-0.011*	-0.093
	(0.826)	(0.329)	(0.685)	(0.142)	(0.007)	(0.069)
Observations	4,395	4,395	4,395	4,395	4,395	4,395
R^2	0.604	0.567	0.633	0.430	0.137	0.403
Municipality fixed effects (293)	✓	✓	✓	✓	✓	✓
Year fixed effects (15)	✓	✓	✓	✓	✓	✓

Note: The dependent variables are hospitalizations per 1,000 inhabitants associated with the named health problems according to the ICD-10 classification. All regressions use year and municipality fixed effects. Time variant control variables are discussed in the Methods section. Standard errors (in parentheses) are clustered at the municipality level.

*p<0.1; **p<0.05; ***p<0.01.

Table 4.A.6. Estimated treatment effect on mortality

	Deaths per 1,000					
	Respiratory	Circulatory system	Pregnancy	Perinatal	Lung cancer	STDs
	(1)	(2)	(3)	(4)	(5)	(6)
Treatment effect	-0.072***	-0.069***	-0.004*	-0.020***	0.002	0.000
	(0.015)	(0.020)	(0.002)	(0.008)	(0.003)	(0.003)
Observations	4,395	4,395	4,395	4,395	4,395	4,395
R^2	0.352	0.381	0.086	0.152	0.099	0.119
Municipality fixed effects (293)	✓	✓	✓	✓	✓	✓
Year fixed effects (15)	✓	✓	✓	✓	✓	✓

Note: The dependent variables are deaths from hospitalizations per 1,000 inhabitants associated with the named health problems according to the ICD-10 classification. All regressions use year and individual fixed effects. Time variant control variables are discussed in the Methods section. Standard errors (in parentheses) are clustered at the municipality level.

*p<0.1; **p<0.05; ***p<0.01.

Table 4.A.7. Estimated treatment effect for fire and air pollution variables at multiple bandwidths

	Number of fires (1)	$PM_{2.5}$ (2)	CO (3)	NO_2 (4)	O_3 (5)	SO_2 (6)
25 km	-0.103**	-0.059**	-0.049**	-0.059**	-0.003	-0.053*
	(0.050)	(0.028)	(0.024)	(0.029)	(0.004)	(0.027)
Observations	3,045	1,815	1,815	1,815	1,815	1,815
75 km	-0.185***	-0.085***	-0.065***	-0.096***	-0.004	-0.080***
	(0.044)	(0.025)	(0.020)	(0.028)	(0.004)	(0.024)
Observations	5,325	3,795	3,795	3,795	3,795	3,795
150 km	-0.187***	-0.044*	-0.030	-0.046*	0.003	-0.042*
	(0.042)	(0.024)	(0.019)	(0.027)	(0.004)	(0.022)
Observations	7,260	5,700	5,700	5,700	5,700	5,700
250 km	-0.156***	-0.014	-0.008	-0.023	0.004	-0.019
	(0.042)	(0.023)	(0.019)	(0.026)	(0.004)	(0.021)
Observations	9,030	7,470	7,470	7,470	7,470	7,470
Full sample	-0.255***	-0.077***	-0.055***	-0.101***	-0.023***	-0.057***
	(0.040)	(0.022)	(0.018)	(0.025)	(0.005)	(0.020)
Observations	11,895	10,335	10,335	10,335	10,335	10,335

Note: Coefficients are given as decimal percentage changes. All regressions use year and municipality fixed effects. Time variant control variables are discussed in the Methods section. $PM_{2.5}$ is particulate matter with a diameter of less than 2.5 μm; *CO* is carbon monoxide; NO_2 is nitrogen dioxide; O_3 is ozone; and SO_2 is sulfur dioxide. Standard errors (in parentheses) are clustered at the municipality level.
*p<0.1; **p<0.05; ***p<0.01.

Table 4.A.8. Estimated treatment effect on hospitalizations at multiple bandwidths

	Hospitalizations per 1,000					
	Respiratory	Circulatory system	Pregnancy	Perinatal	Lung cancer	STDs
	(1)	(2)	(3)	(4)	(5)	(6)
25 km	-0.163	-0.720	-0.159	0.177	-0.028***	-0.115
	(1.023)	(0.483)	(0.875)	(0.136)	(0.009)	(0.087)
Observations	1,815	1,815	1,815	1,815	1,815	1,815
75 km	-1.285	-0.898***	-1.363*	0.240	-0.013*	-0.107
	(0.880)	(0.341)	(0.702)	(0.152)	(0.007)	(0.068)
Observations	3,795	3,795	3,795	3,795	3,795	3,795
150 km	-1.949**	-1.004***	-0.937	0.083	-0.013**	-0.104
	(0.798)	(0.318)	(0.647)	(0.120)	(0.006)	(0.065)
Observations	5,700	5,700	5,700	5,700	5,700	5,700
250 km	-1.674**	-1.125***	-0.556	-0.036	-0.009	-0.119*
	(0.788)	(0.307)	(0.612)	(0.105)	(0.006)	(0.066)
Observations	7,470	7,470	7,470	7,470	7,470	7,470
Full sample	-1.577**	-0.930***	-0.756	-0.184*	-0.009*	-0.059
	(0.759)	(0.289)	(0.583)	(0.100)	(0.005)	(0.065)
Observations	10,335	10,335	10,335	10,335	10,335	10,335

Note: The dependent variables are hospitalizations per 1,000 inhabitants associated with the named health problems according to the ICD-10 classification. All regressions use year and municipality fixed effects. Time variant control variables are discussed in the Methods section. Standard errors (in parentheses) are clustered at the municipality level.
$^{*}p<0.1$; $^{**}p<0.05$; $^{***}p<0.01$.

Table 4.A.9. Estimated treatment effect on mortality at multiple bandwidths

	Deaths per 1,000					
	Respiratory	Circulatory system	Pregnancy	Perinatal	Lung cancer	STDs
	(1)	(2)	(3)	(4)	(5)	(6)
25 km	-0.060***	-0.061**	-0.001	-0.025**	-0.002	-0.001
	(0.018)	(0.026)	(0.003)	(0.010)	(0.004)	(0.004)
Observations	1,815	1,815	1,815	1,815	1,815	1,815
75 km	-0.071***	-0.064***	-0.003	-0.021***	0.000	0.002
	(0.015)	(0.020)	(0.002)	(0.008)	(0.003)	(0.003)
Observations	3,795	3,795	3,795	3,795	3,795	3,795
150 km	-0.066***	-0.064***	-0.003	-0.018***	0.000	-0.001
	(0.014)	(0.019)	(0.002)	(0.007)	(0.003)	(0.003)
Observations	5,700	5,700	5,700	5,700	5,700	5,700
250 km	-0.055***	-0.070***	-0.003	-0.018***	0.001	-0.002
	(0.014)	(0.018)	(0.002)	(0.007)	(0.002)	(0.003)
Observations	7,470	7,470	7,470	7,470	7,470	7,470
Full sample	-0.052***	-0.062***	-0.002	-0.018***	0.001	-0.003
	(0.013)	(0.017)	(0.002)	(0.006)	(0.002)	(0.002)
Observations	10,335	10,335	10,335	10,335	10,335	10,335

Note: The dependent variables are deaths per 1,000 inhabitants associated with the named health problems according to the ICD-10 classification. All regressions use year and municipality fixed effects. Time variant control variables are discussed in the Methods section. Standard errors (in parentheses) are clustered at the municipality level.

*p<0.1; **p<0.05; ***p<0.01.

Table 4.A.10. Estimated treatment effect on outcome variables for the pre-treatment period

	Number of fires (1)	$PM_{2.5}$ (2)	CO (3)	NO_2 (4)	O_3 (5)	SO_2 (6)
Mechanisms	0.016	0.024	0.021	0.045*	0.010**	0.016
	(0.050)	(0.021)	(0.017)	(0.023)	(0.004)	(0.021)
Observations	1,588	1,172	1,172	1,172	1,172	1,172
	Respiratory (1)	Circulatory system (2)	Pregnancy (3)	Perinatal (4)	Lung cancer (5)	STDs (6)
Cases	-0.254	-0.545*	-0.069	-0.439*	-0.005	0.021
	(0.876)	(0.312)	(0.554)	(0.243)	(0.011)	(0.103)
Deaths	0.023	-0.026	0.001	-0.004	-0.001	0.001
	(0.025)	(0.025)	(0.003)	(0.013)	(0.004)	(0.005)
Observations	1,172	1,172	1,172	1,172	1,172	1,172

Notes: Pre-treatment refers to the period from 2003 to 2006. All regressions use year and municipality fixed effects. Dependent variables: $PM_{2.5}$ is particulate matter with a diameter of less than 2.5 μm; *CO* is carbon monoxide; NO_2 is nitrogen dioxide; O_3 is ozone; and SO_2 is sulfur dioxide. Coefficients for fires and air pollution are given as decimal percentage changes. Cases are hospitalizations per 1,000 inhabitants associated with the named health problems according to the ICD-10 classification, and deaths are deaths resulting from these hospitalizations per 1,000 inhabitants. Standard errors (in parentheses) are clustered at the municipality level.
*p<0.1; **p<0.05; ***p<0.01.

Table 4.A.11. Estimated treatment effect on control variables

	Temperature (1)	Precipitation (2)	Humidity (3)	Wind speed (4)	GDP per capita (5)	Agricultural GDP share (6)	Population density (7)	Doctors (8)	Health establishments (9)	Wind direction (10)
Treatment effect	0.008	-0.003	-0.379***	0.008	-0.435	0.006	0.188	-0.464*	-0.014	-3.074
	(0.014)	(0.010)	(0.141)	(0.009)	(0.498)	(0.009)	(1.576)	(0.265)	(0.039)	(4.518)
Observations	4,395	4,395	4,395	4,395	4,395	4,395	4,395	4,395	4,395	4,395
R^2	0.969	0.691	0.925	0.978	0.884	0.867	0.991	0.820	0.795	0.794
Municipality fixed effects (293)	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Year fixed effects (15)	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓

Note: All regressions use year and municipality fixed effects. Dependent variables: GDP per capita is in thousand Brazilian Reais in 2012 prices; agricultural share is the GDP share of the agricultural sector in percentage; population density is inhabitants per square km; doctors and health establishments are per 1,000 inhabitants; temperature is in degree Celsius; precipitation is in millimeters per day; humidity is in percentage; wind speed is in meters per second; and wind direction is in degrees. Standard errors (in parentheses) are clustered at the municipality level.

$^{*}p<0.1$; $^{**}p<0.05$; $^{***}p<0.01$.

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