Influence of Land Tenure on Global Environmental Change: effects on deforestation and biodiversity in Brazil

Dissertation

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Summary

Land tenure, i.e. the rules, norms, and regulations that govern who owns what land and what can be done with it, is considered a key indirect driver of global environmental change as it is intrinsically connected to the use of land resources. The association between land tenure and the use of natural resources has been studied often throughout the past decades, from a variety of scientific disciplines, and is considered an effective governance tool for mitigating global change. However, different configurations of tenure regimes and property rights are seen to both provide incentives for accelerating environmental change, as well as preventing it altogether. Thus, environmental effects of land tenure are still scientifically inconclusive or ambiguous, with expectations and evidence often predicting effects in contrasting directions. In this dissertation, I aim to gain a better understanding on the effects of different land tenure regimes on environmental change at large spatiotemporal scales, specifically focusing on agricultural-driven deforestation and biodiversity change as two aspects of environmental change. In my approach, I use newly available data and interdisciplinary methods, including econometric techniques for the estimation of causal-effects as well as macroecological approaches to test for patterns of effects at large scales.

A prerequisite for implementing this approach was finding observational data that were measured at a fine-grain, yet were spatially-explicit. Thus, after identifying key data gaps and limitations, I chose to focus specifically on Brazil, as its uniquely comprehensive data compilation on land tenure made it possible to implement this approach. I grouped several Brazil-specific tenure categories/sources in order to more closely correspond to classical types of land-tenure regimes that are also generally present in the forested tropics, including private lands, untitled rural settlements, undesignated public lands, strictly-protected and sustainable-use protected areas, as well as indigenous, quilombola, and communal lands (IPLC lands). Subsequently, to quantify environmental change, I used two main variables, forest conversion to agriculture and potential species diversity within a parcel (i.e. a property). To estimate causal effects from observational data, I used a quasi-experimental approach that combined matching with regression analysis. I matched similar properties on their agricultural suitability, accessibility to markets, human-population-density, and other climatic variables – all of which were identified as potentially confounding variables that typically bias the estimation of causal effects. Additionally, I tested for the effects of different land tenure regimes on forest-conversion-to-agriculture across 49 different spatiotemporal scales in Brazil by creating “quasi-repetitions” across Brazil’s biomes during distinct periods of time, and synthesized the effect direction and magnitude at broad and narrow scales. Finally, I tested the sensitivity of these results to the presence of omitted variable bias by calculating Rosenbaum bounds.

I found strong evidence that across vastly different contexts, the lack of well-defined tenure rights on public lands causes increased agriculture-driven deforestation, though on the other hand, effects were surprisingly less clear for potential biodiversity changes. Moreover, I found private tenure increased likelihood of deforestation, and decreased biodiversity as compared to most other most other tenure regimes, suggesting that among all alternative tenure interventions that might reduce the deforestation, interventions leading to private tenure would be the least reliable and typically among the least effective options. I also found both indigenous and quilombola tenure regimes had ambiguous effects on deforestation and biodiversity in Brazil. While both indigenous and

quilombola regimes often have deforestation-decreasing/biodiversity-increasing effects, these are not consistent or reliable across all contexts. On the one hand, these results evidence that effects of IPLC regimes can at times be similar, but rarely worse than undesignated/untitled counterfactuals. However, they also indicate that interventions aiming to guarantee environmental conservation outcomes in IPLC regimes must likely engage in in-depth contextual studies to guarantee these outcomes, and view IPLC tenure regimes as strategic partners for conservation, rather than a mechanism for a desired outcome. Additionally, I found that both conservation-focused regimes – strictly protected areas and sustainable use areas – decreased deforestation consistently and reliably across spatiotemporal scales, and that strictly protected areas significantly increased potential biodiversity compared to private regimes. This provides evidence that – despite doubts about their effectiveness – such conservation-focused tenure regimes are essential instruments for environmental conservation outcomes.

Overall, findings from this research strongly evidence how the lack of property rights and/or poorly defined property rights are drivers of deforestation. Interventions on these undesignated or untitled lands provide an opportunity to decrease deforestation rates in Brazil (e.g. through the creation of more conservation regimes, the recognition of IPLC land claims, or regularizing and providing broader legal options for informal land settlers). This is particularly relevant for the vast amount of undesignated lands in the Amazon. Results also suggest that environmental policies targeting either undesignated/untitled and private lands must consider forest and non-forest ecosystems alike in order to wholly ensure biodiversity conservation outcomes.

In sum, findings from this research provide a better understanding of land tenure as a driver of environmental change at large spatial and temporal scales. Future research could delve into the factors that drive differences of effects between deforestation and potential biodiversity change, as well as better understanding how differences in the specific property rights of different tenure regimes drive conservation effectiveness. However, this research will likely be contingent upon major investments in findable, accessible, interoperable and reusable (FAIR) data infrastructure in order to provide key insight into socioecological challenges related to land tenure around the world.

Zusammenfassung

Landbesitz, d.h. die Regeln, Normen und Vorschriften, die bestimmen, wer welches Land besitzt und was damit gemacht werden darf, gilt als ein wichtiger indirekter Treiber globaler Umweltveränderungen, da er untrennbar mit der Nutzung von Landressourcen verbunden ist. Der Zusammenhang zwischen Landbesitz und der Nutzung natürlicher Ressourcen wurde in den letzten Jahrzehnten von einer Vielzahl wissenschaftlicher Disziplinen untersucht und gilt als wirksamer Ansatzpunkt zur Eindämmung des globalen Wandels. Es wird jedoch davon ausgegangen, dass unterschiedliche Konfigurationen von Besitzverhältnissen und Eigentumsrechten sowohl Anreize für die Beschleunigung von Umweltveränderungen bieten als auch diese gänzlich verhindern können. So sind die Auswirkungen von Landbesitz auf die Umwelt wissenschaftlich noch immer nicht eindeutig oder mehrdeutig, wobei Vorhersagen und Beobachtungen oft in entgegengesetzte Richtungen weisen. In dieser Dissertation möchte ich ein besseres Verständnis für die Auswirkungen unterschiedlicher Landbesitzverhältnisse auf Umweltveränderungen in großen räumlichen und zeitlichen Maßstäben gewinnen, wobei ich mich insbesondere auf die landwirtschaftlich bedingte Entwaldung und die Veränderung der biologischen Vielfalt als zwei Aspekte der Umweltveränderungen konzentriere. Dabei verwende ich neu verfügbare Daten und interdisziplinäre Methoden, einschließlich ökonometrischer Verfahren zur Schätzung kausaler Effekte und makroökologischer Ansätze, um Wirkungsmuster auf großen Skalen zu testen.

Eine Voraussetzung für die Umsetzung dieses Ansatzes war es, Beobachtungsdaten zu finden, die feinkörnig gemessen wurden und dennoch räumlich explizit waren. Nachdem ich die wichtigsten Datenlücken und -beschränkungen identifiziert hatte, entschied ich mich daher, mich speziell auf Brasilien zu konzentrieren, da die dortige umfassende Datensammlung zu Landbesitz die Umsetzung dieses Ansatzes ermöglichte. Ich habe mehrere brasilienspezifische Landbesitzkategorien zussamgefasst, um eine bessere Übereinstimmung mit den klassischen Arten von Landbesitzsystemen zu erreichen, die auch in den bewaldeten Tropen im Allgemeinen anzutreffen sind, darunter Privatland, unbenannte ländliche Siedlungen, nicht ausgewiesenes öffentliches Land, streng geschützte und nachhaltig genutzte Schutzgebiete sowie indigenes, Quilombola- und kommunales Land (IPLC-Land). Zur Quantifizierung der Umweltveränderungen habe ich zwei Hauptvariablen verwendet: die Umwandlung von Wald in landwirtschaftlich genutzte Flächen und die potenzielle Artenvielfalt innerhalb einer Parzelle (d. h. eines Grundstücks). Um die kausalen Auswirkungen von Beobachtungsdaten zu schätzen, habe ich dabei einen quasi-experimentellen Ansat benutzt, der Matching mit Regressionsanalyse kombiniert. Dabei habe ich Grundstücke verglichen, die sich hinsichtlich ihrer landwirtschaftlichen Eignung, der Zugänglichkeit zu Märkten, der Bevölkerungsdichte und anderer klimatischer Variablen ähnelten - allesamt Variablen, die als potenzielle Störfaktoren identifiziert wurden und die Schätzung der kausalen Auswirkungen in der Regel verzerren. Darüber hinaus habe ich die Auswirkungen verschiedener Landbesitzsysteme auf die Umwandlung von Wald in Landwirtschaft auf 49 verschiedenen räumlichen und zeitlichen Skalen in Brasilien getestet. Dazu habe ich "Quasi-Wiederholungen" in den brasilianischen Biomen während verschiedener Zeiträume erstellt und die Richtung und das Ausmaß der Auswirkungen im großen und kleinen Maßstab zusammengefasst. Schließlich habe ich mithilfe von Rosenbaumgrenzen getestet, ob diese Ergebnisse durch ausgelassene Variablen verzerrt waren.

Ich habe deutliche Belege dafür gefunden, dass das Fehlen klar definierter Eigentumsrechte auf öffentlichem Land in sehr unterschiedlichen Kontexten zu einer verstärkten landwirtschaftlich bedingten Entwaldung führt, wohingegen die Auswirkungen auf potenzielle Veränderungen der biologischen Vielfalt überraschenderweise weniger deutlich waren. Darüber hinaus habe ich festgestellt, dass privater Landbesitz die Wahrscheinlichkeit der Entwaldung erhöht und die biologische Vielfalt im Vergleich zu den meisten anderen Besitzverhältnissen verringert, was darauf hindeutet, dass unter allen alternativen Besitzverhältnissen, die die Entwaldung verringern könnten, Interventionen, die zu privatem Landbesitz führen, die am wenigsten verlässlichen und typischerweise auch die am wenigsten wirksamen Optionen sind. Ich habe auch herausgefunden, dass sowohl indigene als auch Quilombola-Besitzverhältnisse ambivalente Auswirkungen auf die Entwaldung und die biologische Vielfalt in Brasilien haben. Zwar haben sowohl indigene als auch Quilombola-Regelungen häufig entwaldungsmindernde und biodiversitätssteigernde Auswirkungen, doch sind diese nicht in allen Kontexten konsistent oder zuverlässig. Einerseits belegen diese Ergebnisse, dass die Auswirkungen von IPLC-Regelungen manchmal ähnlich, aber selten schlechter sein können als die von nicht ausgewiesenen/unbenannten Flächen. Sie deuten aber auch darauf hin, dass Interventionen, die darauf abzielen, Umweltschutzergebnisse in IPLC-Regelungen zu garantieren, wahrscheinlich von eingehenden kontextbezogenen Studien begleitet werden sollten, um diese Ergebnisse zu garantieren. Dementsprechend sind IPLC-Besitzregelungen eher als strategische Partner für den Naturschutz zu betrachten und nicht als einen Mechanismus für ein gewünschtes Naturschutzergebnis. Darüber hinaus habe ich festgestellt, dass beide auf den Schutz ausgerichteten Systeme - streng geschützte Gebiete und Gebiete mit nachhaltiger Nutzung - die Entwaldung konsistent und zuverlässig über räumliche und zeitliche Skalen hinweg verringert haben und dass streng geschützte Gebiete die potenzielle biologische Vielfalt im Vergleich zu privaten Systemen deutlich erhöht haben. Dies ist ein Beleg dafür, dass solche auf den Schutz ausgerichteten Besitzverhältnisse - trotz Zweifeln an ihrer Wirksamkeit - wesentliche Instrumente für den Umweltschutz sind.

Insgesamt zeigen die Ergebnisse dieser Untersuchung deutlich, dass fehlende Eigentumsrechte und/oder schlecht definierte Eigentumsrechte die Entwaldung vorantreiben. Interventionen auf diesen nicht ausgewiesenen oder nicht benannten Flächen bieten die Möglichkeit, die Entwaldungsraten in Brasilien zu senken (z. B. durch die Schaffung von mehr Schutzregelungen, die Anerkennung von IPLC-Landansprüchen oder die Legalisierung und Bereitstellung breiterer rechtlicher Optionen für informelle Landbesiedler). Dies ist besonders wichtig für die große Menge an nicht ausgewiesenem Land im Amazonasgebiet. Die Ergebnisse deuten auch darauf hin, dass umweltpolitische Maßnahmen, die auf nicht ausgewiesenes und privates Land abzielen, Wald- und Nicht-Wald-Ökosysteme gleichermaßen berücksichtigen müssen, um die Erhaltung der biologischen Vielfalt vollständig zu gewährleisten.

Zusammenfassend lässt sich sagen, dass die Ergebnisse dieser Forschung zu einem besseren Verständnis von Landbesitz als Treiber von Umweltveränderungen auf großen räumlichen und zeitlichen Skalen führen. Künftige Forschungsarbeiten könnten sich mit den Faktoren befassen, die für die unterschiedlichen Auswirkungen von Entwaldung und potenziellen Veränderungen der biologischen Vielfalt verantwortlich sind, sowie mit einem besseren Verständnis dafür, wie die Unterschiede in den spezifischen Eigentumsrechten der Landbesitzkategorien die Wirksamkeit des Naturschutzes beeinflussen. Diese Forschung wird jedoch wahrscheinlich von größeren Investitionen in eine auffindbare, zugängliche, interoperable und wiederverwendbare (FAIR)

Dateninfrastruktur abhängen, um wichtige Einblicke in sozioökologische Herausforderungen im Zusammenhang mit Landbesitz auf der ganzen Welt zu erhalten.

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AME	Average Marginal Effects
AoH	Area of Habitat
ATM	Average Treatment effects in the remaining Matched sample
CAR	Catastro Ambiental Rural (Rural Environmental Cadaster)
CARE	Collective benefit, Authority to control, Responsibility, and Ethics
CBD	Convention on Biological Diversity
CEM	Coarsened Exact Matching
DETER	Sistema de Detecção do Desmatamento em Tempo Real na Amazônia (System for Real-time Deforestation Detection System in the Amazon)
FAIR	Findability, Accessibility, Interoperability, and Reusability (refering to scientific data)
FAO	Food and Agriculture Organization
GBIF	Global Biodiversity Information Facility
GBIF	Global Biodiversity Information Facility
GDP	Gross Domestic Product
GEO BON	Group on Earth Observations Biodiversity Observation Network
GHG	Greenhouse gas
GLMs	Generalized Linear Models
GO	Government Organization
IBC	Instituto del Bien Común (Peru)
ICCA	Indigenous and Community Conserved Areas consortium
IGAC	Instituto Geográfico Agustín Codazzi
INCRA	Instituto Nacional de Colonização e Reforma Agrária (National Institute of Colonization and Agrarian Reform)
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
IPLC	Indigenous Peoples and other Local Communities
IUCN	International Union for Conservation of Nature
LGAF	Land Governance Assessment Framework
LU	Land-use
LUC	Land-use Change
MBI	Market-based Instruments

MCC	Millenium Challenge Corporation
NRGF	Natural Resource Governance Framework
PA	Protected Area
PDSI	Palmer's Drought Severity Index
PES	Payments for Ecosystem Services
PPBio	Programa de Pesquisa em Biodiversidade (Program for Biodiversity Research)
PPCDAm	Plano de Ação para Prevenção e Controle do Desmatamento na Amazônia Legal (Action Plan for the Prevention and Control of Deforestation in the Legal Amazon)
PPCerrado	Plano de Ação para Prevenção e Controle do Desmatamento e das Queimadas no Cerrado (Action Plan for the Prevention and Control of Deforestation and Burning in the Cerrado)
PRODES	Projeto de Monitoramento do Desmatamento na Amazônia Legal por Satélite (Project for Monitoring Deforestation in the Legal Amazon by Satelite)
RAN	Registro Agrario Nacional (Mexico)
REDD+	Reducing Emissions from Deforestation and Forest Degradation
RIC	Registro de Infromación Catastral (Guatemala)
RRI	Rights and Resources Institute
SDGs	Sustainable Development Goals
SEAD	Secretaria Especial da Agricultura Familiar e do Desenvolvimento Agrário (Special Secretariat for Family Agriculture and Agrarian Development)
SiBBr	Brazilian System of Information on Biodiversity
SIGEF	Sistema de Gestão Fundiária (Land Management System)
SIGTIERRAS	Sistema Nacional de Información Territorial: Infraestructura Nacional de Datos Espaciales, Dirección General del Registro Nacional (Ecuador)
UN	United Nations
UNEP	United Nations Environment Programme
WB	World Bank
WCMC	World Conservation Monitoring Centre
WRI	World Resources Institute

1. Introduction

1.1 Background

Better understanding the anthropogenic causes of global environmental change, and how these causes may be leveraged for both socioeconomic development gains and environmental conservation, is currently one of the central challenges for the global scientific community (IPCC, 2019). However, understanding these causes of change (i.e. direct and indirect drivers of environmental change) is a complex challenge given the combination of multi-scale policies, processes and institutions that may determine how natural resources are used and changed by their users (IPCC, 2019; Lambin et al., 2001). Land tenure, i.e. the rules, norms, and regulations that govern who owns what land and what can be done with it, is considered a key driver of land use – and thereby a key, albeit indirect, driver of global environmental change (Cox & FAO, 2002; Lambin et al., 2001). Thus, as addressed in many current international agreements and conventions (e.g. Sustainable Development Goals (SDGs), the Convention on Biological Diversity (CBD), or the Post-2020 Biodiversity Framework), land tenure may have far reaching socio-ecological implications.

Major shifts in global agricultural production and consumption patterns of the past century have highly impacted land-use (Dasgupta, 2021; Kastner, 2021; Müller et al., 2021). These shifts but have been possible through changes in land allocation, distribution, and ownership in many parts of the world. For instance, it is estimated that approximately 90 million ha of global agricultural land have been acquired by foreign investors since the early 2000's (Müller et al., 2021), with uncertain implications for local, family, and traditional landholders and their livelihoods, particularly in the global south (Ceddia, 2019; Müller et al., 2021; Rudel & Hernandez, 2017). Thus, the association between land tenure and land-use/land-use change (LU/LUC) is well-known, as land ownership may have direct ramifications for economic development, wealth distribution, food security and sovereignty, as well as gender inequalities (FAO, 2015; IPCC, 2019). At the same time, the SDGs outline in their target to eradicate poverty (SDG target 1.4), that it is essential to ensure that "*all men and women, in particular the poor and the vulnerable, have equal rights to economic resources, as well as access to basic services, ownership and control over land and other forms of property, inheritance,* [and] *natural resources…*" (UN General Assembly, 2015, p. 15). Here, it is clear that the global development agenda considers issues of land ownership and control of natural resources as necessary conditions to alleviate poverty. It is moreover increasingly evident that, as global LUC continues intensifying or accelerating, the impacts of land tenure on socioeconomic outcomes may be manifold.

Intrinsically connected to the use of land resources, land tenure may also strongly influence the natural environment, and thus the sustainable management of natural resources over long periods of time and from local to global scales. The importance of this association – and its far-reaching consequences – has also become increasingly evident over the past years as it is now estimated that at least 75% of the global land area has been substantially impacted by humans, with the vast majority of this change driven by agricultural and food systems (IPBES, 2018; IPCC, 2019). LUC and consequent habitat loss is the leading cause for biodiversity loss (IPBES, 2019), and is a significant contributor of greenhouse gas (GHG) emissions, with LUC for food and agricultural uses driving approximately 25% of total anthropogenic emissions (IPBES, 2019; IPCC, 2019). Land tenure, particularly the formalization of property rights and the improvement of tenure security, has been widely considered to be an effective governance tool for mitigating these global changes, especially in the global south. These ongoing climate and environmental crises underscore the importance of better understanding the influence of land systems on environmental change.

Undoubtedly, land tenure systems and their impacts on socioeconomic and environmental outcomes are connected in complex ways, and the need to further understand these connections is not new. The association between land tenure and the use of natural resources has been often studied throughout the past decades, from a variety of scientific disciplines. Yet, scientific consensus is still inconclusive or ambiguous, with often contrasting theoretical expectations and empirical evidence, depending on the field or the scope of a given study (see **1.2** State-of-the-art in research). In this dissertation, I investigate the influence of land tenure as a driver of global environmental change using an interdisciplinary lens, newly available data and methods, and focusing on the effects of different land tenure regimes on tropical forests and biodiversity. Thereby, the aim is to contribute robust evidence on this land-tenure-environment relationship for a better understanding of the complex human-environment interactions centered around land and its sustainable future.

1.2 State-of-the-art in research

Theoretical expectations regarding the relationship between land tenure and its environmental impacts have a long academic history, with modern understandings of this concept that can be dated to the 1800-1900's (Baland & Platteau, 1996). These theoretical ideas regarding the effects of land tenure on environmental change are constantly being developed, revised, contextualized, and challenged. This subsection (**1.2**) covers theoretical ideas and related empirical evidence stemming from various disciplines including development economics, political economy and property rights theory, as well as the economics of environmental conservation, and conservation and sustainability sciences that identify land tenure as an indirect driver of global environmental change. Expectations of these different scientific fields on the effects of land tenure on environmental change may differ in direction and magnitude, as illustrated in **Fig. 1**.

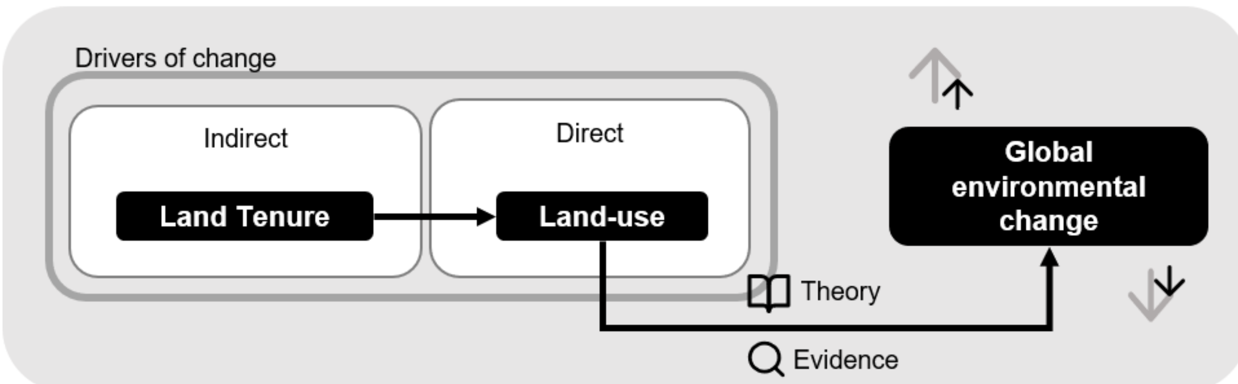

Figure 1. Conceptual model of land tenure as an indirect driver of global environmental change. Throughout this section (**1.2**), I overview how theories and evidence from different fields and perspectives explain this main concept of land tenure as a driver of global environmental change. Note that, global environmental change may be expected to increase or decrease at different magnitudes (or also, remain the same), depending on different theories and supporting empirical evidence. Throughout **Figs 2-4**, the book icon will indicate theories, and the magnifying glass icon will indicate empirical evidence.

Note that here, I use the term "global environmental change" to indicate this as my main outcome of interest. However, as I overview and summarize a large body of literature that aims to understand and explain how land tenure impacts environmental change, I use different terms for different outcomes, which can include: environmental collapse, sustainability of natural resources, environmental conservation, decrease in deforestation, and nature solutions. While I use these terms in this section to best synthesize expectations from the literature about different aspects of the influence of land tenure on the environment, these different terms should be interpreted as different aspects of global environmental change. The remainder of this subsection is organized in three parts that relate land tenure to environmental change 1) The "commons" and the communal, 2) The formalization and privatization of land, and 3) Solutions for regulating negative environmental externalities.

1.2.1. "The commons" and the communal

A group of theories initially developed as a part of classic and neoclassical economic schools of thought includes the "tragedy of the commons". Here, the idea was that open-access, commonly held resources inevitably collapse due to over exploitation (Gordon, 1954; Hardin, 1968). The "tragedy" explained that finite, non-excludable resources (e.g. fisheries or pastures), are likely to be overly exploited by "free-riders", that is, individuals maximizing their gains of access to a resource without paying for it. As free-rider resource-users increase, the resource is inevitably depleted, and thus the best possible solution to this problem is to privatize the common resource because it will only be successfully sustained by excluding resource-users. Therefore, under this paradigm, all resources that were not held privately would be inevitably destined to collapse. These ideas regarding the inevitability of collapse of natural resources widely influenced scientific thought regarding the use of natural resources (Frischmann et al., 2019; Sandler, 2015). Broadly applied to land, these classical expectations would predict land-use change and environmental degradation in all land tenure regimes except for private lands, i.e. those where the land owner has the right to exclude other users from a resource.

However, while the idea of an inevitable "tragedy" has received a lot of attention (Frischmann et al., 2019), there is limited empirical evidence that supports the idea that open-access resources inevitably collapse – especially when examining ecological systems that are more complex than Hardin's initial metaphor of a pasture. Instead of empirically observing this kind of collapse of natural resources, researchers observed that there were many instances where common-pool resources were effectively and sustainably managed, rather than exploited. Furthermore, this emerging body of research noted that classical expectations on the effects of property relied on many unmet theoretical assumptions, for instance, perfectly functioning markets, or zero transaction costs (Frischmann et al., 2019).

Challenging previous theories on the collapse of common-pool resources, new institutional economics emerged as a body of theoretical ideas on the broader governance and regulation of natural resource-use (Naidu, 2009; Sandler, 2015). Here, the study of cooperative, self-regulatory approaches gained further traction and property rights systems were identified as a key institutional element of the market (Boudreaux, 2015) because they defined the combination of rights, privileges and limitations of the owner over a particular resource. Theory began questioning and studying who should have property rights (Baland & Platteau, 1996), how property rights may incentivize the long-term sustainability of natural resources (Baland & Platteau, 1996; Ostrom, 2009), and the conditions for the success or failure of this long-term sustainability (Libecap, 1994; Ostrom, 2009).

Most notably within institutional/political economics, Elinor Ostrom's "Rational Choice Theory on Collective Action", argued that while the "tragedy of the commons" framework could predict outcomes in auctions and competitive market situations well, it failed to predict outcomes under alternative conditions. Ostrom identified a number of design principles that determine the conditions under which collective action is more likely to occur and persist. These design principles include factors that are particularly relevant to property and land-resources, such as: 1) clearly defined boundaries, 2) proportional equivalence between benefits and costs, 3) collective-choice agreements, and 4) monitoring, among others (Ostrom, 2009).

Ostrom proposes that the interaction of resources, governance systems, users, and these design principles determine socioenvironmental outcomes. Instead of theoretical expectations predicting a "tragic" collapse of all commonly-held resources, Ostrom's influential body of work opens up the paradigm to the study of more complex socioecological systems, varying from traditional fishing communities to indigenous-forest peoples. Property rights are defined as the rights that determine a user's capacity to exercise effective authority over a resource. Commonly recognized rights are characterized as a “bundle of rights” which includes the right to use a resource (e.g. access and withdrawal), the right to earn income from a resource, the right to transfer that resource to others (i.e. alienation), and the right to enforce property rights (e.g. exclusion or management) (Robinson, Masuda, et al., 2017).

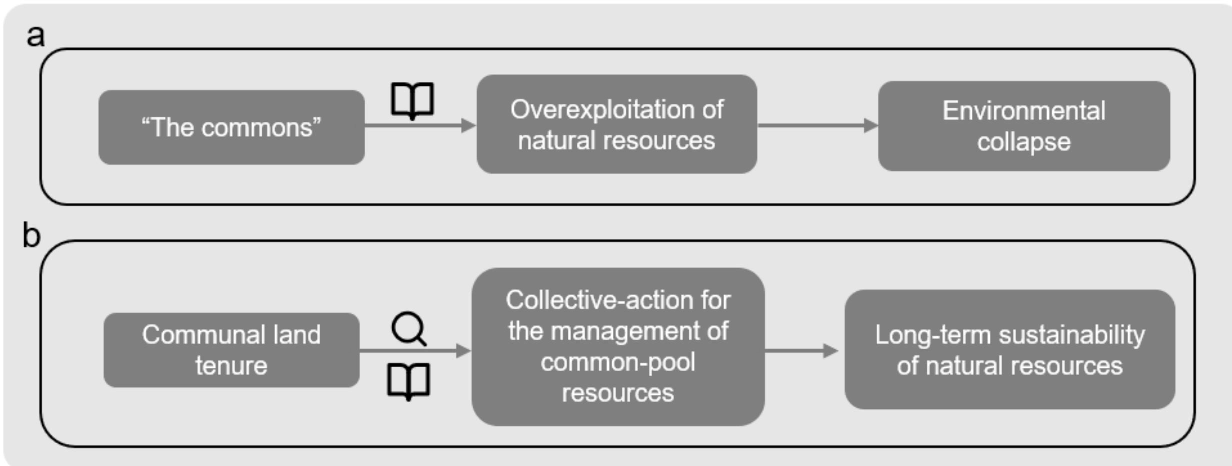

Figure 2. Conceptual model summarizing how two different groups of theories expect different environmental outcomes from (a) “The commons” and (b) communal land tenure. “The commons” is theorized to lead to environmental collapse due to the overexploitation of natural resources, whereas communal land tenure regimes are observed and theorized to lead to the long-term sustainable management of common-pool resources due to their effective management through collective action.

In sum, these schools of thought describe two scenarios that cause adverse environmental change when may be costly to prevent otherwise. The first, the “tragedy”, is one where the lack of exclusion rights over a resource results in its overexploitation and eventual depletion, and where the “costs” of preventing this overexploitation are presumably excessive. The second scenario is one where collective organization around the management of a resource ensures its long-term sustainability, and the cost of organizing this management is borne among the resource-users.

1.2.2 The privatization and formalization of land

In parallel to the work on cooperative and self-regulatory socioecological systems, other research fields in economics also further developed ideas on the economic advantages of the formalization of property rights. A predominant idea has been that the formalization of land rights provides landholders/owners various economic benefits, for many reasons (Feder & Feeny, 1991). First, the formalization of property rights (e.g. through a land title) would increase assurance, that is, the guarantee that any returns on investment could be claimed. An increase in assurance would increase investment incentives in a particular parcel of land, and long-term capital benefits. Second, the formalization of property rights would allow for land to be alienated, that is, given, sold, or rented out. Alienation also implies collateralizability, that is, land can be used as a collateral for loans or other financing. Both of these mechanisms would increase access to capital (de Soto, 2000). Third, the formalization of property rights presumably decreases land conflicts, as it clarifies legal ownership. In turn, this formalization is thought to decrease the need to spend capital on a property’s monitoring or in-person defense and thus the resources spent on managing conflicts would be freed (Deininger & Castagnini, 2006). Relatedly, when the need for being present to defend the land decreases, opportunities for doing off-farm work increase, thereby increasing access to capital (Fenske, 2011).

Overall, these purported economic benefits focus on the formalization of *private* property rights. The privatization of land assumes, for efficiency reasons, that land will be put to its most productive purpose (Birdyshaw & Ellis, 2007; Deininger et al., 2003a), which has historically been agriculture-related land-uses (Ceddia, 2019; Kastner, 2021; Rudel & Hernandez, 2017). This implies a high likelihood of environmental change and the use of natural resources. Environmental change can be further exacerbated when land privatization is promoted alongside policy incentives to specifically carry out agricultural land transformations (Binswanger, 1991). This has often been the case across many tropical regions in the past several decades, e.g. in Brazil, the "colonization" of northern regions was specifically incentivized via quotas of minimum agricultural production, and through tax exemptions for income from agricultural land (Binswanger, 1991; Damasceno et al., 2017). Thus, notwithstanding the positive economic benefits expected from the privatization of land rights, the widespread promotion of these rights likely to have an impact on land-use and other land-related resources (Deininger, 2003).

Despite these strong expectations that the formalization of land rights can have many economic benefits along with associated land-use change and environmental impact, empirical studies conducted during the past few decades do not reliably find evidence for these expectations. Studies find that the formalization of property rights does not necessarily lead to increases in economic efficiency for many reasons (i.e. increases in agricultural investments, productivity, or intensification) (Place, 2009). Many studies find that land titling, specifically, does not necessarily guarantee expected benefits, such as an increase in land tenure security (Brasselle et al., 2002; Deininger & Jin, 2006), increasing the likelihood of a property being sold (Payne et al., 2009), or increasing collateralizability (Field & Torero, 2006; Galiani & Schargrodsky, 2010; Kerekes & Williamson, 2010). There are several reasons these expected benefits may not hold empirically; for instance, tenure security may be increased via other mechanisms besides land titling, such as informal agreements (Brasselle et al., 2002; Deininger & Jin, 2006). Also, collateralizability effects may be low because financial institutions remain reluctant to provide loans to the poor even if they can deliver land as collateral (Field & Torero, 2006; Galiani & Schargrodsky, 2010). Contextual factors can also influence these effects, as recently-titled properties may not necessarily integrate into local markets due to policies and regulations that prevent rapid integration (e.g. Brazil and India have implemented gentrification-management regulations after titling interventions) (S. T. Holden & Otsuka, 2014).

Due to data and methodological limitations, it is often difficult for empirical studies to account for issues of endogeneity, which are issues related to unobserved and unaccounted factors driving – and confounding – an outcome (i.e. in other words, endogeneity occurs when an explanatory variable is correlated with the error term). In the case of land tenure, endogeneity may arise from reverse causality, as land-use change may actually be a mechanism for securing property rights (Binswanger, 1991; Fenske, 2011), but, it may also occur when unobserved, confounding factors are unaccounted for (e.g. informal tenure agreements between family members). For these reasons, economic effects of land formalization and privatization often appear empirically ambiguous, and highly context-dependent, despite the policies in place incentivizing it for its benefits. This implies environmental effects may be similarly ambiguous and context-dependent, as private property rights appear to both increase and decrease opportunities and incentives for land-use change.

The exclusive emphasis on the formalization of private property rights, has resulted in more ambiguous theory on the expected economic and environmental outcomes of the formalization of other tenure regimes. These tenure regimes commonly include indigenous peoples and other local communities (IPLCs), whose formal recognition of property rights in the territories they claim can be contentious (Larson et al., 2015). However, support for this recognition has recently grown in strength, particularly within scientific communities interested in finding synergies between these tenure regimes and environmental conservation due to the substantial spatial overlap between IPLC land claims and forests/natural landscapes (Blackman & Veit, 2018; Fa et al., 2020; Garnett et al., 2018; O'Bryan et al., 2021). The recognition of IPLC land rights is expected to yield positive conservation outcomes because IPLC traditional knowledge and practices are expected to best manage natural resources for long-term environmental sustainability (Arnold & Stewart, 1991; Begotti & Peres, 2020; Charnley & Poe, 2007; Chazdon, 2008), albeit specific economic outcomes of this formalization are less clear.

Inspired by ideas in institutional and political economics, many studies have aimed to empirically measure the environmental benefits of recognizing and formalizing communally held lands (i.e. IPLC property rights) (Baragwanath & Bayi, 2020; Blackman et al., 2017; Blackman & Veit, 2018; Probst et al., 2020; Vélez et al., 2020). To this end, studies will often compare the effects of IPLCs in promoting environmental conservation to different counterfactuals. These include, for instance, comparisons to protected areas (PAs) (Ellis & Porter-Bolland, 2008; Porter-Bolland et al., 2012; Sze et al., 2022), to private lands (Gabay & Alam, 2017), or even other IPLCs across different contexts (Baynes et al., 2015), often finding they perform similarly, if not better, than other tenure regimes in protecting natural resources. (Ceddia et al., 2015; Fa et al., 2020; O'Bryan et al., 2021). This body of evidence clearly challenges previous notions that only privately-held land (or land where access is otherwise restricted) can be successful in ensuring the long-term sustainability of natural resources (Hayes & Ostrom, 2005).

There is growing empirical consensus that a strong association exists between IPLC lands and the conservation of global natural landscapes (Fa et al., 2020; Garnett et al., 2018), and furthermore, there is growing recognition that formalizing IPLC lands may have positive environmental outcomes, particularly in reducing tropical deforestation (Sze et al., 2022). However, it is often difficult to answer whether this relationship is 1) indeed a causal effect or only a strong spatial association, and 2) consistently and generally found across contexts, or highly specific to every unique case. Aside from publication and selection biases of many studies that commonly hinder syntheses on this topic, causal approaches are often limited to considering smaller spatial and temporal extents, i.e. typically measuring deforestation impacts <5 years after an intervention, and usually at sub-regional scales (Robinson, Masuda, et al., 2017). Indeed, recent causally-rigorous studies conducted at various spatiotemporal scales in Ecuador, Colombia and Indonesia highlight these scale-related limitations in understanding long-term effects of tenure interventions in IPLC regimes (Buntaine et al., 2015; Kraus et al., 2021; Vélez et al., 2020), and in fact, some find that the direction and magnitude of effects on deforestation depend on specific regional contexts (Vélez et al., 2020). While measuring the short-term, local effects of tenure interventions in regimes is key, clarifying the causality of these interventions, as well as the direction and magnitude of these effects at larger spatiotemporal scales remains crucial in understanding long-term effects of IPLC regimes on environmental change.

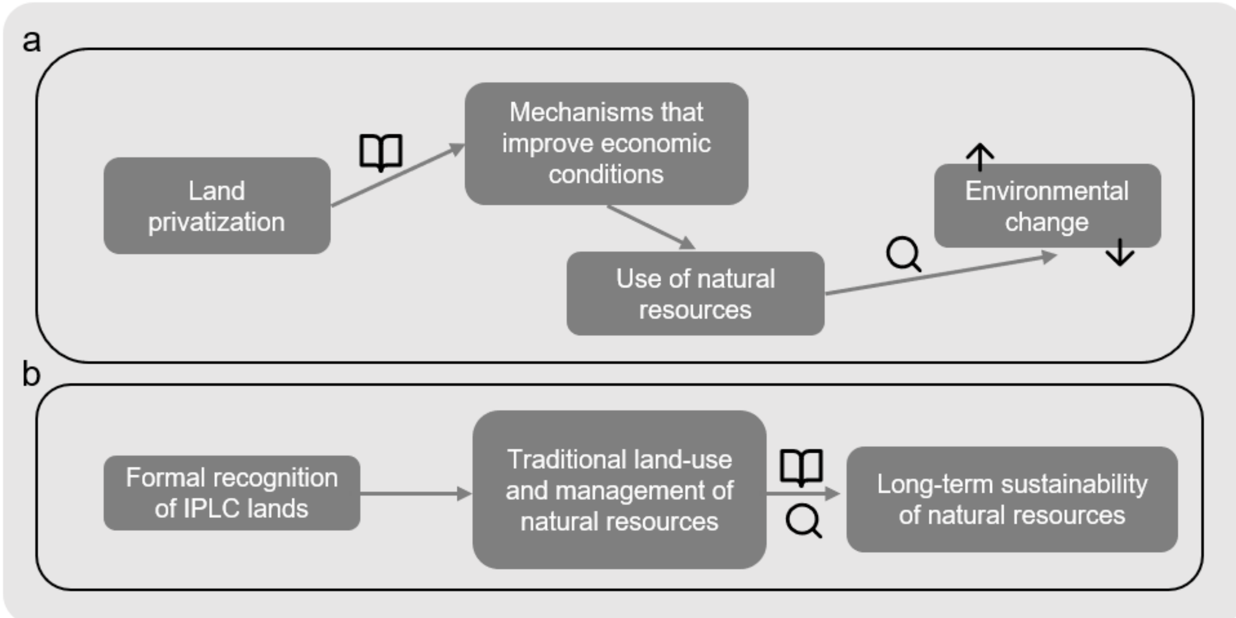

Figure 3. Conceptual model summarizing how (a) the privatization of land is expected to cause environmental change through mechanisms that improve economic conditions and imply the use of natural resources (albeit with mixed empirical effects) and (b) the formal recognition of IPLC land claims is expected to lead to the long-term sustainability of natural resources (albeit empirical observations are often of association not necessarily cause and effect).

In sum, promising theoretical expectations have led to the wide promotion of land privatization and formalization policies around the world, with an estimated US$2.5 billion spent on land titling efforts over the past couple of decades (Tseng et al., 2020). Yet, there is little clarity on how land privatization and formalization affect economic and associated environmental outcomes, both in private as well as alternative tenure regimes. While there seems to be much more consensus on the environmental benefits of formally recognizing IPLC lands, there is little discussion in the literature on the specific property-rights-mechanisms that drive this. Vice versa, while there is detailed discussion on the mechanisms that can drive environmental change in private tenure regimes, but little consensus on the general direction effects. Thus, it is still unclear how land formalization and privatization policy incentives affect environmental change in different tenure regimes.

1.2.3 Solutions for regulating negative environmental externalities

In recent years, a better understanding of anthropogenic-driven environmental change and degradation (IPCC, 2019; Kehoe et al., 2015; Lambin et al., 2001). Aiming to gain better insight into the socioeconomic drivers of environmental change, scholars from these fields often study processes that have caused adverse environmental change as an externality, i.e. an unaccounted indirect impact of a given process (e.g. pollution is a negative externality of manufacturing processes). In turn, many theories relating land tenure to these processes have been in association to policy and regulatory instruments thought to mitigate these negative environmental externalities, particularly those that change structures and incentives of property rights to influence land use. These include, for instance, command-and-control instruments (e.g. a law setting a limit to the amount of timber to be harvested from a forest), market-based instruments (e.g. a tax for purchasing timber, payments for ecosystem services), and more.

How do conservation interventions implicate land tenure? One of the most classic regulatory process aimed at decreasing environmental degradation and loss has been the creation of protected areas (PAs) as conservation regimes. PAs can be considered a standard command-and-control instrument, as they generally restrict access and use of resources in a top-down way (Barrett et al., 2013). Despite economic theory apprehension on whether this kind of instrument works – command-and-control instruments are considered inefficient because they are unlikely to fulfill their objectives at the lowest possible cost (Barrett et al., 2013) – the creation of PAs in the past decades has been prolific. By 2020, almost 17% of global land was under protection (UNEP-WCMC & IUCN, 2021), representing a major shift in land tenure systems around the world.

Empirical studies on the effectiveness of PAs in decreasing deforestation as well as in generally mitigating environmental change, show mixed results. Firstly, it can be complicated to define and causally measure "conservation effectiveness" (Chape et al., 2005) because research shows PA placement is often biased to "high and far" places, i.e. places that are not of agricultural interest where anthropogenic pressure is low to begin with (Joppa & Pfaff, 2009). Second, effectiveness of PAs can also be highly subject to governance regimes (Nolte et al., 2013), and hindered by underfunding which can result in "paper parks" that legally exist, yet, do not achieve any conservation objectives (Blackman et al., 2015; Watson et al., 2014). Thus, global assessments of PA effectiveness during the past couple decades show mixed findings, with some studies finding PAs mitigate forest loss (Leverington et al., 2010), albeit not much (Yang et al., 2021), while others finding no difference between PAs and unprotected counterfactuals in mitigating human pressure on the environment (Geldmann et al., 2019).

Here, it is important to note that the establishment of PAs coupled with implicit changes in land tenure can directly affect the property rights of people living in and depending on the resources in these areas, which are commonly indigenous, traditional and poor populations. The socioeconomic impacts of these interventions fueled heated arguments of "people vs. parks" (Agrawal & Redford, 2009), which often posited conservation and socioeconomic outcomes in opposition to each other. Consequently, the effectiveness and efficiency of creating conservation regimes as a command-and-control instrument that prevents environmental degradation, is unclear at best, and socially contentious at worst. Gaining better clarity on the effectiveness of this instrument – as well on its socioeconomic impacts – is key, given how recent studies show that threats to the integrity of PAs are increasing globally (Kroner et al., 2019; Mascia & Pailler, 2011).

How does land tenure mediate effects of conservation interventions? Market-based instruments (MBI) have been embraced by many as a tool to address pressing environmental conservation needs, as they are considered more flexible and economically efficient than command-and-control instruments (Gómez-Baggethun & Muradian, 2015; Greiner & Stanley, 2013). Specifically, payments for ecosystem services (PES) schemes have been designed for global market actors to compensate those in regions providing positive environmental externalities, thereby incentivizing environmental conservation (Gómez-Baggethun & Muradian, 2015; Wunder, 2005, 2015). Most notably within PES schemes are those designed to decrease tropical deforestation rates, for instance, the United Nations (UN) framework for Reducing Emissions from Deforestation and forest Degradation (REDD+), is designed to compensate carbon sequestration services and has been established in over 30 countries (UN-REDD Programme, 2021). The expectation is that direct, monetary payments from countries in the global north to countries in the global south should decrease global deforestation. While these market-based approaches could, in theory, widely apply from communally-held to privately-held forests, expectations regarding their effectiveness differ depending on local land tenure systems because these determine the practical boundaries of ecosystem services to be provided. In other words, land tenure determines the excludability of the supply of ecosystem services to be compensated, and if there is no excludability there is no guarantee there will be any demand (Gómez-Baggethun & Muradian, 2015). Thus, theoretical expectations concerning such MBI imply that conservation outcomes are only likely under conditions of formalized tenure regimes (e.g. private properties, well-defined PAs, well-defined and legally recognized indigenous lands).

Indeed, empirical studies examining the implementation of such interventions find that contextual and institutional aspects, including land tenure, are major factors in their implementation (Börner et al., 2017; Duchelle et al., 2014; Fletcher et al., 2016; Larson et al., 2013). In fact, while some studies argue that such PES schemes increase tenure security, (Jones et al., 2020; Larson et al., 2013), other studies also find that participants may hesitate to participate due to fears and concerns their property rights will be affected through such programs (Jayachandran et al., 2017). Thus, broader, national-level implementation of such MBI/PES may be hindered by unresolved land tenure issues common across many tropical contexts (e.g. overlaps, conflicts, and informal land claims where property rights are not clearly defined) (Börner et al., 2010, 2017; Duchelle et al., 2014; Larson et al., 2013; Naughton-Treves & Wendland, 2014). Hence, despite theoretical expectations on the effectiveness of MBIs, formalized, or, better-defined land tenure systems are found to be necessary in order to guarantee desired conservation outcomes.

Land sparing and land sharing. Beyond the instruments that prevent or regulate environmental change, emerging theories from conservation and sustainability sciences also debate what are the best solutions for meeting global agricultural demands while conserving the natural environment. The “land sharing versus land sparing” debate broadly posits whether agricultural production should be intensified on a smaller expanse of land in order to be spared for conservation purposes, or whether low-intensity, small-scale agriculture over larger expanses of land can best provide solutions both for humans and nature (Fischer et al., 2008; Phelps et al., 2013). In this debate, land tenure systems play a key role, as they represent the actual physical limits of land allocation in either of the scenarios. Pre-existing land tenure configurations may limit or expand opportunities for either land sparing or sharing. Certain land tenure regimes may also have more agility for land intensification than others, depending on their specific property rights (e.g. private regimes with easier access to capital through alienation or collateralizability). Other tenure regimes may play key roles in ensuring a place for land-sparing exists (e.g. protected areas or indigenous lands (Ceddia et al., 2015)). In contrast, small-scale agriculture is more likely to happen in tenure regimes without the agility or ability to engage in high-input land-uses, where commercial withdrawal rights as well as due process rights are not granted or less secure (e.g. informally held lands part of land tenure systems that are poorly governed). While it is likely both land sparing and sharing play important roles in a given landscape (Mertz & Mertens, 2017), both views must consider the existing tenure-regime configurations currently governing natural resource use on these lands.

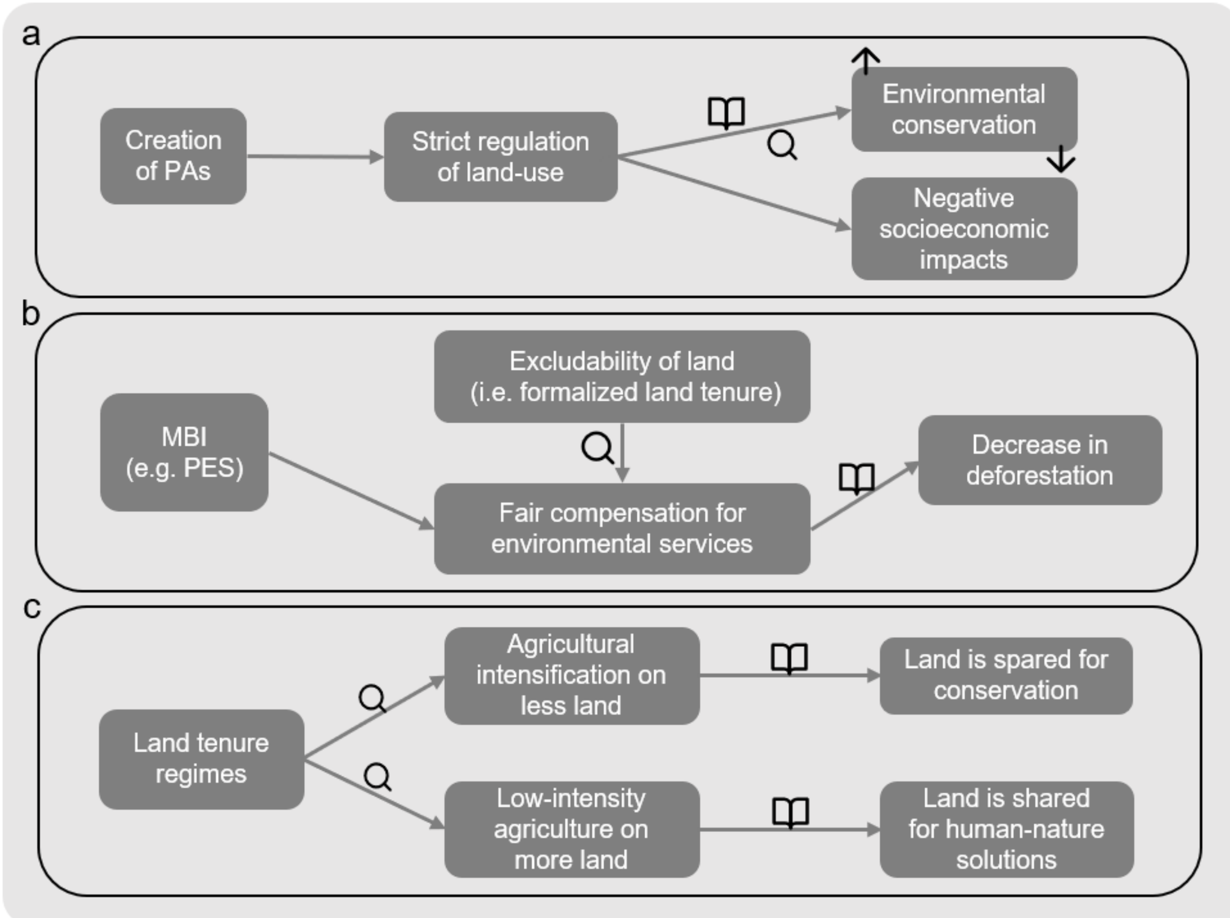

Figure 4. Conceptual model summarizing: (a) how creating protected areas (PAs) is expected to lead to environmental conservation through the strict regulation of land-use, however, empirical evidence on these effects are mixed, and many raise concerns about the negative socioeconomic impacts caused by these interventions; (b) how market-based instruments (MBIs), such as payments for ecosystem services (PES) are expected to decrease deforestation by fairly compensating environmental services that normally go unaccounted for. Empirically, however, improving land tenure is observed to be a prerequisite for the success of these interventions, as only formalized tenure regimes are effectively able to participate in such MBIs; (c) how both land sparing and land sharing theories rely on pre-existing land tenure allocations to implement either conservation or human-nature solutions in a given landscape.

In summary, theoretical expectations on the influence of land tenure on environmental change come from various scientific fields. When examining this change from a variety of lenses, it is evident that land tenure can play different roles in determining the direction and magnitude of expected environmental impact. Different configurations of tenure regimes and property rights are seen to both provide incentives for accelerating environmental change, as well as preventing it altogether. Thus, how different tenure regimes can generally influence environmental change, to what extent, in what magnitude, and under what conditions remains theoretically ambiguous.

1.3 Research gaps

To date, empirical studies on the effects of land tenure on environmental change have not added consensus to many theoretical expectations. Instead, many studies further evidence the complex effects of land tenure interventions, adding further ambiguity to scientific understanding of this relationship. In turn, conservation and land-governance policies alike could greatly benefit from more conclusive evidence on the effects of land tenure on environmental change, especially in countries underrepresented in research, where environmental conservation is under most pressure and in-depth contextual studies are often not feasible due to resource and time constraints. The following, remaining research gaps hinder building clearer consensus, and thus, robust generalizations on the most likely long-term effects of land tenure policies and interventions:

1) *The findability and accessibility of spatially explicit data on land tenure across large spatiotemporal scales*

One of the most important elements that often hinders further research on the effects of land tenure on environmental change is the feasibility of collecting appropriate data for conducting causally rigorous tests, or, the availability of such data for conducting syntheses studies at larger spatiotemporal scales. For instance, as overviewed in **1.2.2**, causally rigorous studies on the effects of land privatization often find mixed results, with the direction of effects seemingly contingent on characteristics of each specific context. Yet, in parallel, there is a high level of scientific consensus that the formalization of IPLC land rights has positive environmental outcomes, regardless of any specific context (notwithstanding identified conditions required for collective action (Ostrom, 2009)). Recent meta-analytical and review-type approaches aim to synthesize and resolve common questions on the effects land tenure on environmental outcomes (Ojanen et al., 2017; Robinson et al., 2014; Tseng et al., 2020). However, these studies highlight inherent challenges and limitations of empirical evidence to date.

First, studies that use a credible counterfactual to estimate causal effects of land tenure interventions are much scarcer than literature that merely controls for confounding effects, which is considered less rigorous (albeit not invalid) (Robinson et al., 2014; Tseng et al., 2020). Second, identifying lasting environmental impacts requires measurement over long temporal periods and spatial extents due to inherently dynamic ecological and land-use patterns. Collecting and randomizing this kind information on the impacts of land tenure at large spatial and temporal extents is largely unfeasible, and thus, empirical evidence at these larger scales is commonly lacking from many studies. For instance, out of 79 sites considered in one meta-analysis, the median study period was 10 years and the median study area was of 548 km^2 (Robinson et al., 2014).

Clearly, limitations in spatially explicit data across large spatiotemporal scales hinders research aiming to better measure, synthesize, resolve, and build consensus around common questions on the effects of land tenure. Therefore, investigating current availability and access of spatially explicit land tenure data at larger spatiotemporal scales is a crucial research gap, and a prerequisite for future improvements in research on the effects of land tenure on environmental change.

2) *The cross-comparison of different land tenure regimes to understand relative effectiveness*

Theoretical expectations on the effects of land tenure regimes on environmental change are rarely compared in the literature to more than two-three tenure regimes at a time. For instance, the “commons” is most often compared to communal lands (**1.2.1**), private lands are typically compared to untitled lands (**1.2.2**), and the effectiveness of IPLC lands is often compared to PAs (**1.2.3**). Similarly, empirical studies of the past decades rarely consider more than one (or few) tenure regimes/categories at a time, typically limited to establishing one or two counterfactuals when assessing effects. Furthermore, the failure to distinguish between open-access lands (lands that lack excludability) and communally held lands (lands held and managed communally by a group and not a single entity) (as overviewed in **1.2.1**) has meant open-access tenure regimes have been rarely evaluated. Yet, it is estimated that over 70% of global forests are publicly-owned and managed by governments with only a fraction being assigned to particular uses (White & Martin, 2002), meaning that assessments of publicly owned, open-access forest-lands remain sorely lacking. Overall, the lack of cross-comparisons of different regimes limits the understanding the relative environmental effects of different tenure regimes. This relative understanding of effectiveness may help answer practical policy questions such as, “how do different kinds of tenure interventions rank in yielding environmental conservation outcomes?”

3) *Estimating the effects of different land tenure regimes on environmental change across large scales*

To date, there is a lack of systematic, large-*n* assessments on the effects of different tenure regimes on environmental change across different spatiotemporal scales, i.e. across different regions and time periods. As meta-analyses and review-type studies identify (Ojanen et al., 2017; Robinson et al., 2014; Tseng et al., 2020), assessments on the effects of land tenure regimes are commonly constrained to case-studies at smaller spatiotemporal scales, e.g. ≤10 years, and ≤550 km^2. Currently evaluated spatiotemporal scales limit practical understanding on the effects of land tenure interventions, which are typically implemented across entire regions or countries, and which may have lasting environmental effects over decades.

4) *Effects of land tenure on environmental outcomes related to biodiversity and ecosystems*

Finally, environmental effects of tenure interventions specific to biodiversity and ecosystems are still sorely lacking, as these are most often measured in terms of forest cover or agricultural practices (Tseng et al., 2020). Though these forest, agricultural, and biodiversity and ecosystems outcomes are closely linked, better understanding the effects of underlying socioeconomic drivers such as land tenure specifically on changes in biodiversity and ecosystem is a pressing global concern (IPBES, 2019).

1.4 Objectives and research questions

Given the diverging theoretical expectations and inconclusive empirical evidence regarding the association between land tenure and environmental change, the main objective of this dissertation is to gain a better understanding on the effects of different land tenure regimes on environmental change at large spatiotemporal scales. In order to empirically test said effects, I specifically focus on agricultural-driven deforestation and biodiversity change as two important, yet complementary aspects of environmental change. To carry out this objective, I ask following specific research questions:

1. What is the state of global land tenure data findability and accessibility?
2. According to predominant theories and evidence to date, how are different land tenure regimes expected to affect environmental change in comparison to each other?
3. What are the effects of different land tenure regimes on deforestation?
4. What are the effects of different land tenure regimes on biodiversity change?

1.5 Overview of remaining chapters

In Chapter **2. Materials and Methods**, I begin by explaining overarching definitions used in this thesis, followed by a brief introduction to the causal and macroecological approaches I used in answering my research questions (**2.1 Approach**). In subsection **2.2 Data scoping and processing,** I overview the extensive data aspects and processes that I used in this thesis, beginning by describing the protocol used in searching for global land tenure data and explaining the methodological choice to focus the empirical analyses specifically on Brazil. In **2.2.1**, I briefly describe Brazil's land tenure data. In **2.2.2** I define the variables used to quantify environmental change in Brazil, including forest conversion to agriculture, the quantification of potential species diversity, as well as additional covariates and resources used to conduct the empirical analyses. Then, in subsection **2.3 Study context**¸ I proceed to provide detailed background information on Brazil's geography, governance, and historical background to their land tenure system. In **2.4 Study design and methods for empirical analyses**, I bring together all of the different data and detail the quasi-experimental setup that I used (using matching to establish counterfactuals), the regression analyses used for estimating effects of land tenure on deforestation and biodiversity change in Brazil, the subsequent sensitivity analyses, and finally, and the approach used for synthesizing found effects across large scales.

In Chapter **3. Results**, I first report on the availability and state of land tenure data in Latin America (**3.1**). Then, based on reviewed literature, I conduct a qualitative analysis on the expected effects of different land tenure regimes in Brazil on deforestation (**3.2**), which crucially links theoretical expectations reviewed in the Introduction (**1.2**) to subsequent empirical tests conducted across Brazil (**3.3-4**). Subsection **3.3** presents results on the effects of land tenure regimes on deforestation, and **3.4** on the effects of land tenure regimes on potential biodiversity in Brazil (**3.4**).

Chapters **4-5, Discussion** and **Conclusions,** discuss findings, including data gaps, theoretical and empirical implications, and conclude with policy implications as well as future outlook for research on the effects of land tenure on global environmental change.

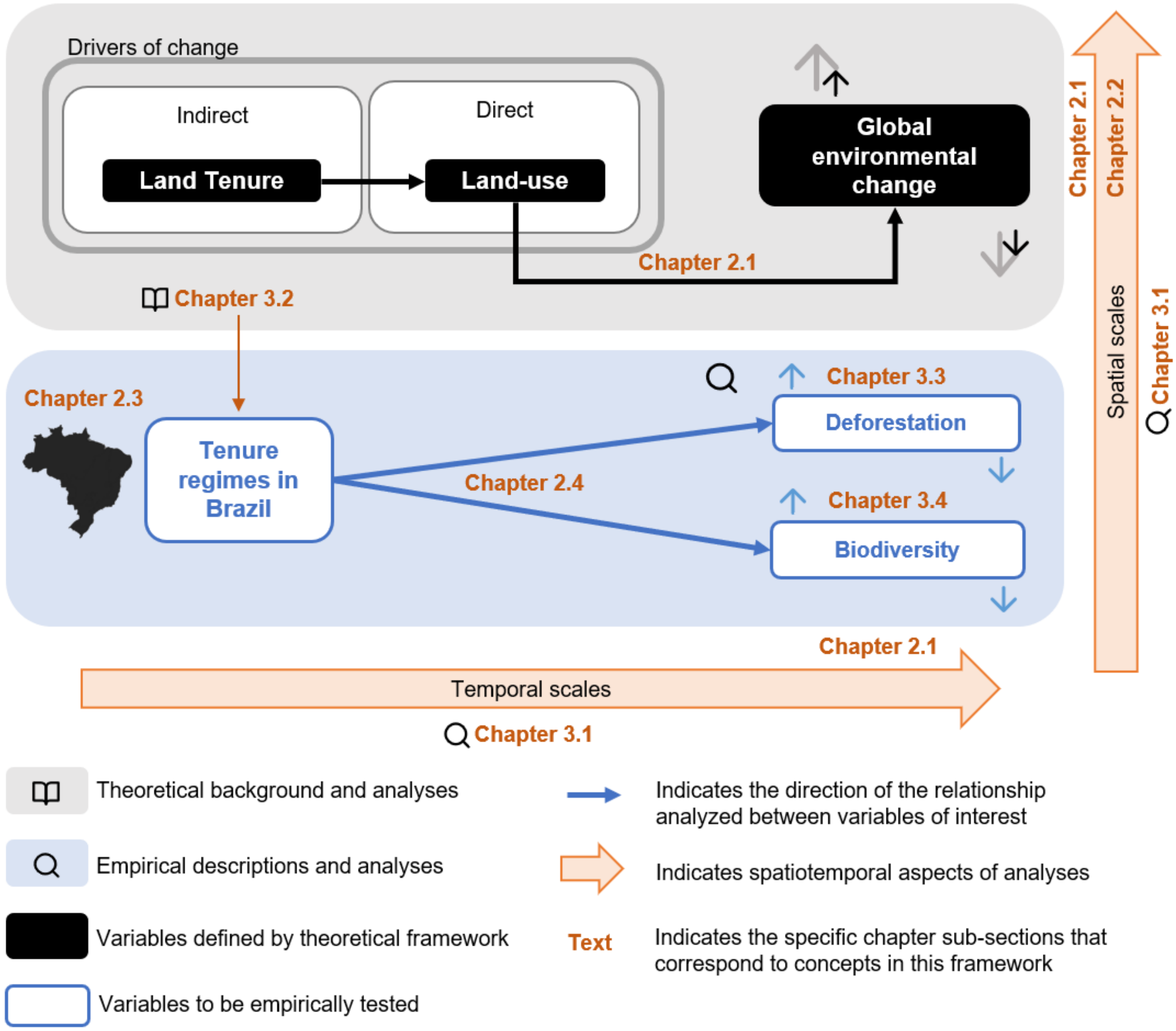

Figure 5. Conceptual framework illustrating the approaches used in this dissertation to investigate the influence of land tenure on global environmental change. The top panel (in grey, and as used in **Fig. 1**) illustrates the theoretical background and analyses used in the thesis, whereas the bottom (in blue) illustrates empirical descriptions and analyses.

First, to illustrate where materials and methods fit in this conceptual model, I place **Chapter 2.1**, near the main black arrow linking land tenure as an indirect driver of environmental change, as well as near the upper limits of the arrows along the axes (in orange), to illustrate how I use causal and macroecological approaches to establish this relationship. **Chapter 2.2** is located alongside these orange arrows, as it indicates the challenge of scoping and processing data at large scales. Located in the bottom panel (in blue), **Chapter 2.3** describes the study context in Brazil, and **Chapter 2.4**, describes the quasi-experimental methods and regression analyses used to estimate the effects of different tenure regimes in Brazil on both deforestation and biodiversity.

Second, to illustrate where results fit in this model, I place **Chapter 3.1** alongside spatial and temporal arrows to illustrate how it reports on the state of land tenure data at different spatiotemporal scales. As **Chapter 3.2** analyzes theoretical expectations of different land tenure regimes in the context of Brazil, it is located in the theoretical panel (grey) with an arrow pointing to the empirical (blue) panel, illustrating its crucial role as a bridge between theory and empirical evidence. Finally, **Chapters 3.3** and **3.4** report findings on the effects of different land tenure regimes on deforestation and biodiversity in Brazil as measures of global environmental change.

2. Materials and Methods

2.1 Approach

To carry out the objective and address the research questions I established, I use an interdisciplinary approach that includes analyzing theories on land tenure from a variety of fields, but also adopting specific methodologies from both economics and ecology. Seeing as different fields can often use different vocabulary and terms for similar concepts (or vice versa, similar terms with different meanings), in this section I define some of the main concepts pertaining to land tenure that I use throughout this dissertation (**1.5.1**). I also provide a brief background and rationale on the statistical/econometric (**1.5.2**) and macroecological approaches that I use in designing the empirical analyses I carried out (**1.5.3**).

2.1.1 Land tenure definitions

Land tenure is the overall concept describing the governance relationship between people and land; i.e. the combination of rules, norms and regulations that determine who owns what land, and in what ways they can use the resources of that land (Cox & FAO, 2002)

Property rights are the specific rights that determine in what ways the land and associated resources can be used. These specific rights are often described in the literature as "the bundle of rights" because they are commonly grouped together. Not all categories of land tenure are always granted all property rights, however, commonly granted property rights may include:

- *Access*: the right to enter land and access its resources
- *Withdrawal*: the right to use land and its resources. Withdrawal rights can sometimes be further restricted to commercial or non-commercial rights.
- *Management*: the right to plan, regulate, and manage the use of land and its resources
- *Exclusion*: the right to determine who has access rights to a particular land property and its resources
- *Alienation*: the right to transfer land and its resources to others
- *Due process and compensation*: the right to receive fair compensation in cases of land expropriation

(Definitions of property rights are based on (Robinson, Masuda, et al., 2017; Schlager & Ostrom, 1992). See **Table 1** for a characterization of commonly associated bundles of rights with specific tenure regimes.)

Land tenure security is the level of perceived assurance that the property rights in question will be respected or upheld (Robinson, Masuda, et al., 2017).

Land tenure regimes is an overarching concept that is used throughout this dissertation to describe specific categories of land tenure. For the purposes of the analyses conducted here, each category, or "land tenure regime" incorporates the combination of 1) property rights legally assigned and felt on the ground, 2) the implications that these property rights may have for tenure security, and 3) the policies and regulations that may thus apply to certain tenure categories under a given governance system. For example, in a private land tenure regime, specific property rights are commonly legally assigned by granting a land title. The land title in combination with trust in the broader governance system assures the private property owner that their land belongs to them and will not be illegally expropriated by anyone else without reason or compensation. Additionally, a policy that is enforced by the government may prohibit private tenure regimes from certain land uses (e.g. clearing land next to water sources is prohibited in many tropical contexts). Here, the policy and governance context effectively regulate and restrict the property rights of this particular private land tenure regime.

Table 1 Bundles of rights usually associated with common tenure regimes. For each regime, I define the typical *number of resource users involved in land decision-making* (second column), as well as the main types of *rights holders* (who hold this particular bundle of rights) and main *duty holders* (who are responsible for upholding the associated bundles of rights through, e.g., monitoring of properties), where GO indicates government organization. The bundles of rights associated with the tenure regimes are characterized according to general patterns, loosely based on (Robinson, Masuda, et al., 2017) with color shading from red to green indicating the extensiveness and/or level of guarantee of rights granted along six different rights dimensions (access, withdrawal, management, exclusion, alienation, due process).

Tenure regime	**Number of users**	**Bundles of rights usually included**						**Main right holder**	**Main duty holder**
		Access	Withdrawal	Management	Exclusion	Alienation	Due Process		
Private lands	1	++	+	+	++	++	++	Individual(s)/ Entity	Individual(s)/ Entity
Public lands	1 or few	++	++	+	-	-	--	Citizenry	GO
Protected Areas	1 or few	-	--	+	++	--	+	Citizenry	GO
IPLC lands	Usually many	++	+	+/-	+/-	+/-	-	Community	GO

++ indicates full guarantee of rights
+ indicates some guaranteed rights that are usually subject to specific restrictions (e.g., environmental/agricultural policies)
+/- indicates some rights, guaranteed under certain legal conditions, circumstances, or clauses
- indicates little guarantee of, or severely limited, rights
-- indicates no guarantee of any rights

2.1.2 Estimation of causal effects

The aim of this research is to estimate the causal effects of different land tenure regimes on environmental change. While there is an expected association – due to theoretical expectations and empirical observation – robust evidence on the causal link over broad spatiotemporal scales is lacking. Conducting a classical experiment to estimate causal effects is not feasible or desirable, therefore, a quasi-experimental setup is used in order to estimate effects using observational data. In this case specifically, this requires using parcel-level, spatially explicit data on different land tenure regimes paired with high-resolution environmental variables, and taking measures to account for the non-random allocation of those tenure regimes using econometric techniques to approximate a randomized experiment as much as possible.

One of the central challenges in using a quasi-experimental approach is to define an appropriate counterfactual, that is, a group that is plausibly similar enough to those who receive a treatment (e.g. a specific tenure regime) to compare outcomes from said treatment. This is a central challenge of the analysis because it hinges on observing what was not observed – and justifying why what is observed is sufficient to draw credible causal inferences. The other central challenge in using this approach is to ensure that treatment assignment is independent of the potential outcome. In the case of this research question, it is well known tenure regimes are not randomly distributed across landscapes. In fact, certain landscapes (e.g. forests or savannas) are more prone to certain land uses (e.g. agriculture), which can influence the assignment of particular ownership types. Thus, conducting statistical analyses on the raw data would not only be highly biased, but also highly dependent upon model selection. For these reasons, it is essential to establish a credible counterfactual and account for all the factors that influence the assignment of tenure regimes in order to estimate effects on environmental change. The methodological choices taken to address both challenges are detailed in section **2.5.1**.

2.1.3 Macro-ecological and syntheses approaches

Macroecological approaches can be characterized as 1) studying large-scale systems, and 2) explaining patterns that emerge at these large-scales (McGill, 2019). These approaches often emphasize the use of empirical observations and statistical techniques, that necessarily imply excluding experimental approaches because conducting experiments at this scale is largely unfeasible. Macroecological approaches are commonly driven by the goal of pursuing generality and finding patterns that "[escape] from contingency", i.e. finding effects that are only relevant to one specific context (McGill, 2019). These approaches that use large amounts of data over vast spatiotemporal scales – often at very fine spatial grains – are becoming more common throughout the past couple of decades, due to the increasing availability of satellite-based (as well as other environmental) data.

On the one hand, given the dynamic nature of environmental change, observing broad patterns enables a better understanding of the global processes driving this change, as well as the circumstances or contexts under which certain theoretical expectations may hold. This prioritization reflects how environmental fields are currently shifting focus towards more broadly defining global priorities for conservation (IPBES, 2018; IPCC, 2019). On the other hand, human systems are complex – and controlling for all the elements in a given socio-ecological system is an arduous challenge. While macro-economic approaches do study large-scale systems, they commonly do this at very coarse grains (e.g. at the country or multi-country level). However, land-related policy and decision-making have implications at multiple governance scales; i.e. policies often rely on compliance at an individual level (e.g. household) but are normally implemented across regions or countries.

Thus, instead of choosing a specific case or site to study said effects in detail, or only describing patterns at coarse grains, the aim of this dissertation is to test for effects at larger extents (following a macro-ecological approach) while including the rigor of causal inference (following econometric methods). Through this approach, the aim is to provide empirical insight into whether land tenure broadly has an effect on environmental change, and whether these effects have generally consistent patterns that confirm/contradict predominant theoretical expectations. Alternatively, effects found only under particular conditions or scales, can inform emerging middle-range theories on *how* land systems change, that is, enabling “contextual generalizations that describe chains of causal mechanisms explaining a well-bounded range of phenomena” (Meyfroidt et al., 2018).

2.2 Data scoping and processing

In order to estimate the effects of land tenure on global environmental change at broad spatiotemporal scales, a prerequisite is finding usable observational data that are measured at a fine-grain, yet are spatially-explicit. Therefore, in order to implement my proposed quasi-experimental, macroecological approach, the initial methodological step in this dissertation involved searching for spatially explicit data on land tenure across large spatiotemporal extents.

Finding and gaining access to spatially explicit data on land tenure regimes at a parcel-level is, however, particularly challenging. Findable and accessible mean that data should be identifiable via clear and explicit metadata and retrievable via an open protocol, as defined by the FAIR guiding principles for scientific data (Wilkinson et al., 2016). In contrast to other openly available datasets (e.g. global climatic or land-cover datasets that are openly available at high-resolutions), parcel-level data on land tenure requires the physical demarcation of social relations, which can be costly and time-consuming to produce. This demarcation can often be contentious because of possible overlaps or otherwise informal and/or intangible agreements on specific property rights (Robinson et al., 2014). Parcel-level data on land tenure is also commonly protected by data privacy laws, as it involves personal, sensitive information. As a result of the contentious nature of the data, alongside high costs and data protection considerations, land tenure data holders may not always prioritize findability and accessibility aspects of these data.

Initially, with the aim of gathering as many spatially explicit data on land tenure as possible, an explorative, data gathering effort was carried out from October 2017 to September 2019. This effort included searching for both openly available as well as privately held data, and was carried out in three main ways: 1) searching the web for openly available published spatial data on land tenure 2) contacting and networking with agencies or stakeholders that privately-collected or held data on land ownership, and 3) identifying and contacting statistical/geographical/spatial agencies of target countries (see **Table 4** in **3.1** for results of this data gathering effort).

The web search for openly available spatially-explicit land tenure data was informed by existing data compilations (e.g. Landmark Map, Global Forest Watch), and was otherwise carried out online using key search terms in various languages (e.g. property, land ownership, registry, and cadaster). Found sources were systematically documented, and initial findings informed subsequent steps in contacting data stakeholders and sources, as well as targeted countries.

Stakeholders that either collected, had access, or otherwise had expertise on land tenure data were first approached via email. Where possible, further engagement took place via online and in-person meetings, as well as at international conferences. Although this process was informative (see section **3.1**), no significant gains in access to databases/datasets or further data sources resulted from this effort of engaging with stakeholders. Given the learned challenges in acquiring spatially explicit land tenure datasets from global organizations or stakeholders, a third approach was subsequently prioritized, wherein countries in the Latin-American region were specifically targeted.

Data agencies in Latin America (i.e. statistical, spatial, or geographic institutes, environmental or agricultural ministries, etc.) were specifically targeted in the data search for two reasons. First, it is a region with a variety of socio-environmental contexts, where accelerating environmental change threatens existing ecosystems as well as the livelihoods of those that depend on them

(Fedele et al., 2021). Second, it is a region where there has already been a history of investment in land tenure regularization (Blackman et al., 2017; de Soto, 2000; Field & Torero, 2006; Holland et al., 2017). This record of investment indicated the possibility that data on land tenure or land ownership may have been documented, maintained, and likely available over broad spatial extents.

To engage with these agencies, a contact protocol was established which involved identifying agencies, emailing their contact addresses, sending repeated email reminders, and telephoning if no answer was received or if this was the only contact provided. Contact with agencies in 16 countries was successfully established. In a few rare cases, in-country research partners held meetings with these government agencies, and helped process formal requests for access to data. However, there was only limited data access that resulted from this process (see **Table 4** in Results chapter). Thus, to the end of prioritizing scientific analysis over data collection and harmonization, the remainder of this research focuses specifically on Brazil, due to the existing, openly-available dataset on land tenure compiled (Imaflora et al., 2018). Therefore, the following subsection (**2.2.1**) describes Brazil's land tenure data and the definition of land tenure categories used in this dissertation.

2.2.1 Brazil's land tenure data

Although data limitations in many countries meant that access to parcel-level data on land tenure was scarce, Brazil's uniquely comprehensive data on land tenure (Imaflora et al., 2018) made it possible to study the effects of land tenure on environmental change at broad spatial and temporal scales. Brazil harbors the world's largest biodiversity and living carbon stores (Global Forest Watch, 2021; Mittermeier et al., 2005), yet these are under pressure from ambitious agroeconomic development (Abessa et al., 2019; Rajão et al., 2020). Thus, Brazil has extensive experience in linking tenure reform with environmental/agricultural policies across a variety of socioeconomic and ecological conditions, which allows for empirical testing of predominant theories across large spatiotemporal extents and is an important feature required for finding patterns of effects that transcend contingency (McGill, 2019). Moreover, insights of the environmental implications of these policies are often transferred to inform policy strategies in other tropical regions (Duchelle et al., 2014; Shankland & Gonçalves, 2016; Tollefson, 2015), indicating the relevance of findings in Brazil to tenure-related policies in other similar regions.

I used the publicly available Atlas of Brazilian Agriculture (Imaflora et al., 2018) for the empirical analyses conducted in this dissertation (**3.3-3.4**). This dataset includes spatially explicit parcel-level data mapping land-tenure for 83.4% of the Brazilian territory, and is based on 18 official, most up-to-date data sources, which were integrated using an expert-vetted system to systematically resolve data conflicts resulting from, e.g., overlapping land claims due to due illegally fabricated land titles and/or mapping errors (Sparovek et al., 2019).

For most tenure categories, publicly available data do not include information on the date of each parcel's formalization (i.e., titling or demarcation). Despite likely changes in official ownership status, it can be assumed that for the majority of parcels, the basic type of tenancy (e.g., public institutions vs. indigenous communities vs. private individuals) did not change over the course of the study period. However, several steps were taken to minimize possible bias in statistical analyses and further conclusions.

Firstly, tenure sub-categories defined via programs that only came into existence after the study period began were excluded (i.e. both titled and untitled parcels from the Terra Legal program).

Secondly, robustness tests for selected tenure categories were performed, with documented "treatment" dates, where parcels for which today's tenure category was non-existent or unclear at the beginning of the respective study period were filtered out (see **2.5.3**). Thirdly, possible biases in the quasi-experimental setup due to remaining statistical imbalance were also considered when presenting results, as well as the sensitivity of results to the possibility of any omitted variables, and systematic differences in initial forest cover between "treatment" and "control" units. Specific steps taken are outlined in section **2.4**.

Categorization of Brazil's land tenure data

The Imaflora dataset distinguishes 14 different tenure categories, including several different subcategories of private and public lands that are products of Brazil's specific land-administration history. I grouped several Brazil-specific tenure categories/sources in order to more closely correspond to classical types of land-tenure regimes that are also generally present in the forested tropics. These categories remain sufficiently specific to the context of Brazil, which enables country-specific conclusions. These tenure regimes are characterized by specific bundles of rights, as well as the regulations that norm how the tenants can interact with their land resources (e.g. **Table 1**,**Table 2**).

Table 2. Tenure regimes in Brazil and associated bundles of rights (adaptation of **Table 1** to tenure system in Brazil). 14 land-tenure categories distinguished in Brazil (first column) are re-categorized into eight tenure regimes (second column). The bundles of rights associated with the tenure regimes are characterized according to past and current legislation in Brazil (see references in last column), with color shading from red to green indicating the extensiveness and/or level of guarantee of rights granted along seven different rights dimensions (access, subsistence withdrawal, commercial withdrawal, management, exclusion, alienation, due process).

Brazil tenure categories	**Tenure regime**	**Tenants**	**Bundles of rights** (usually included)							**Main right holder**	**Main duty holder**	**References**
			Access	Withdrawal (subsistence)	Withdrawal (commercial)	Management	Exclusion	Alienation	Due Process			
CAR poor (properties with more than 5% of overlapping areas with neighbors) CAR premium (properties with less than 5% of overlapping areas with neighbors) SIGEF (Private properties registered in INCRA systems) Private properties from Terra Legal program	Private lands	1	++	+	+	+	++	++	++	Individual(s), firm, or other entity	Individual(s), firm, or other entity	Lei 4.947 art. 22 1966, (FAO/SEAD, 2017)
Communitary lands	Communal lands	Many	++	+	+/-	+/-	+/-	+/-	-	Community	GO	Decreto N. 6.040, 2007, Lei N. 11.284, 2006, (Leuzinger & Lingard, n.d.; Soares-Pinheiro, 2018).
Quilombola lands	Quilombola lands	Usually many	++	+	+	+	+	--	+	Community	Community	Consitucao Federal art. 68, Decreto N. 6.040, 2007, (Carvalho & Carvalho, 2016; Sociedade Brasileira de Direito Público, 2002).
Homologated Indigenous land	Indigenous lands	Usually many	++	+	--	+	+	--	+	Community	GO	Consitucao Federal art. 231.

Non-homologated indigenous land												1996, Decreto N. 6.040, 2007, (Paixao et al., 2015).
Full protection conservation unit	Protected Areas	1 or few	-	--	--	+	++	--	+	Citizenry	GO	Lei nº 6.938, de 31 de agosto de 1981, Lei Complementar n° 140, de 8 de dezembro de 2011
Sustainable use conservation unit	Sustainable use Protected Areas	1 or few	+/-	+/-	+/-	+	+	--	+	Citizenry/ Community	GO	Lei nº 6.938, de 31 de agosto de 1981, Lei Complementar n° 140, de 8 de dezembro de 2011
Undesignated public forests	Undesignated public lands	1 or few	++	++	++	++	-	-	--	Citizenry	GO	(Sparovek et al., 2019)
Rural settlements	Untitled rural settlements on public lands	1 or few	++	++	++	+	-	-	--	Individual	GO	Lei Nº 12.465, de 11 de julho de 2017
Lands from the Terra Legal program (both titled and untitled), Military areas, Water, and Urban lands were omitted from subsequent analyses.												

++ indicates full guarantee of extensive rights
+ indicates some guaranteed rights that are usually subject to specific (e.g., environmental) restrictions
+/- indicates some rights, guaranteed under certain legal conditions, circumstances, or clauses
- indicates little guarantee of, or severely limited, rights
-- indicates no guarantee of any rights

Private lands. This category includes lands that are privately owned by individual persons, companies, or other entities (but not communities; see below). Of all tenure regimes, private tenure guarantees tenants the most extensive set of rights (**Table 2**), although some resource-withdrawal rights are regulated through existing agricultural and environmental policies. Private properties from different sources (CAR, SIGEF) are combined under this category. While a small percentage of these private lands may have shifted tenure categories during the study period, most had already been settled and formally recognized as private lands before the mid-1980s (e.g. (Duchelle et al., 2014)); note that subsequent changes in the specific property owners are not relevant to the definition of a tenure regime). By contrast, all private properties titled under the Terra Legal program were excluded from the analyses, as this program only started in 2009 and, accordingly, these properties experienced shifts in tenure categories during the study period. Note that deforestation effects of property titling under the Terra Legal program were recently the focus of different study (Probst et al., 2020).

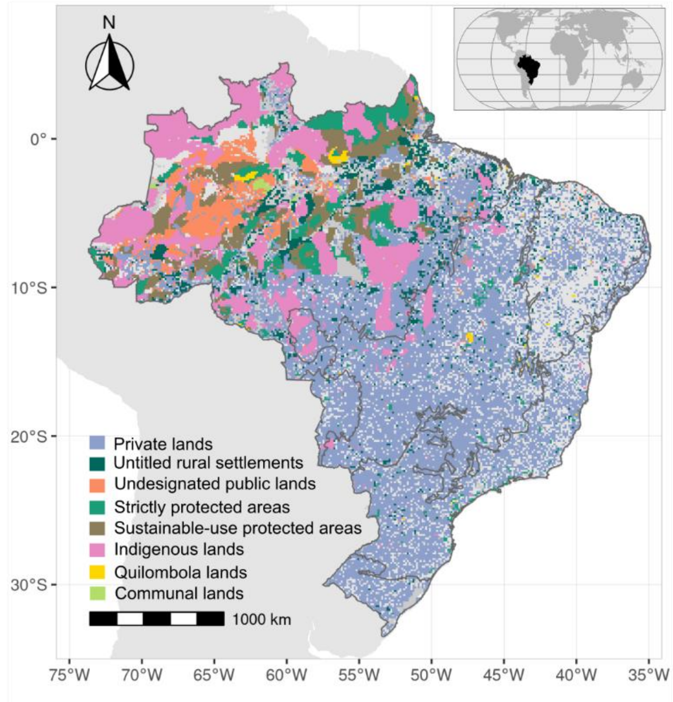

Figure 6. Geographic distribution of land tenure regimes in Brazil, with grey outlines indicating borders of Brazil's biomes. Note that, although difficult to see at a coarse resolution, most tenure regimes are indeed found across all biomes (albeit, fewer and smaller indigenous, quilombola, and communal regimes exist in the Pampa and Pantanal biomes).

Untitled rural settlements on public lands (hereafter "untitled"). This category refers to publicly owned lands that may be occupied by settlers from the agrarian reform, yet have no formally recognized tenure rights (e.g. via registration or titling). Withdrawal use rights are usually not regulated, however, where rural settlements were historically permitted, settlers were required to put at least 80% of the occupied land area to "productive use". Unlike private landowners, however, these settlers never had any exclusion rights, alienation rights, or rights to due process (neither formally nor otherwise guaranteed; **Table 2**). All parcels that are part of the Terra Legal program were excluded altogether because the specific design of this program may have incentivized some settlers to clear forestland in anticipation of the later titling process (Probst et al., 2020), which could bias the perception of effects of these regimes on forests. Currently untitled lands have had this status throughout the 1985-2018 period.

Undesignated public lands (hereafter "undesignated"). Common to all lands included in this category is that while they are publicly owned, the state has not formally assigned them to any purpose. Withdrawal use rights on undesignated lands are usually not regulated, and *de jure* existing regulations are typically not enforced in these lands.

Conservation-focused tenure regimes. The classification of conservation-focused tenure regimes used by the Ministry of Environment of Brazil is followed here, corresponding to the commonly distinguished categories of fully protected areas (*Unidades de Conservação de Protecão Integral*) and sustainable-use areas (*Unidades de Conservação de Uso Sustentavel*). These two categories mainly differ in their access and withdrawal-use rights, with strict-protection regimes strictly restricting access and prohibiting all extraction or withdrawal, whereas sustainable-use regimes afford certain access and withdrawal rights, as long the long-term sustainability of natural resources is ensured (**Table 2**). Neither category affords alienation rights to the citizenry or communities that are technically the main rights holders. Unlike the private and undesignated/untitled lands included in this study, substantial percentages of the parcels under either conservation-focused tenure regime have only come under the respective regime during the course of the study period. Beyond qualitatively assessing consistency of results between more and less "tenure-stable" regions and periods, additional robustness tests for these categories were thus performed. Specifically, statistical analyses were repeated on time-filtered datasets that excluded parcels that either were not under today's tenure category for at least the latter 80% of the respective study period or for which the formal designation date was unknown, using establishment dates from (Ministerio do Meio Ambiente, 2020a).

Indigenous peoples and local community (IPLC) based tenure regimes. Brazil distinguishes three main categories of community-based tenure – indigenous, quilombola, and communal. To analyze hypothesized effects of IPLC tenure, the distinction between these three tenure regimes was maintained due to their very different histories, legal statuses, and granted bundles of rights (**Table 2**). Specifically, indigenous lands are statutorily publicly owned, but managed by indigenous communities with ancestral claims, who are granted strictly non-commercial withdrawal (i.e., subsistence-use) rights. Both homologated (formally recognized) and non-homologated indigenous lands were combined into a single category, as this distinction mostly reflects differences in *de jure* formalization, rather than in the tenure rights *de facto* assumed on the ground. Quilombola lands, by contrast, are communally managed yet privately owned by self-defined communities of descendants from escaped African slaves. Many quilombos have been granted official titles, which legally guarantee commercial as well as non-commercial withdrawal rights. However, quilombola as well as indigenous communities do not have alienation rights (i.e., their lands cannot be sold, leased, used as business collateral, or dismembered).

The third type of IPLC lands, communal lands (*Territórios Comunitários*), are publicly owned but grant certain rights to different groups of self-defined communities traditionally managing forest resources (e.g., *Castanheiros*, *Seringueiros*). Communal tenure regimes are relatively heterogeneous in their rights regulations (**Table 2**). They typically afford the tenants non-commercial withdrawal rights, but the afforded commercial-withdrawal, management, exclusion, and alienation rights vary and are not always clearly defined. Communal tenure is generally the least formalized tenure regime in Brazil, which is also reflected in limited due-process rights (**Table 2**). Thus, due to the of the ambiguity of communal lands' bundles of rights and because there were insufficient registered communal land parcels to support the quasi-experimental design in all biomes except Amazonia, the main analyses of IPLC tenure regimes were limited to indigenous and quilombola tenure. However, additional deforestation results for communal tenure are provided in the Appendix (**Figure 19**), and biodiversity results are described in **Figs. 16-17**).

Many lands claimed by IPLCs are still unmapped or are mapped but not yet officially registered (Damasceno et al., 2017; World Bank, 2014) and were thus not included in these analyses. Brazil's indigenous and quilombola communities have long been tenants of their lands, and the recognition of their tenure rights through the 1988 constitution was a result of ongoing political and legal processes that precede the study period here (1985-2018). Later formalization steps such as demarcation and registration thus constituted changes from informal to formalized versions of the same *de facto* tenure regimes. In the case of indigenous lands, Law 6.001 of 1973 uses the reference to forest populations in the constitution of 1967 to define indigenous lands as reserved areas occupied by forest populations or indigenous peoples. The law prohibited any activity that would displace occupants of these lands (including buying/selling or renting), and non-indigenous peoples were prohibited from hunting, fishing, or conducting other extractive or agricultural activities on these reserved areas. Furthermore, FUNAI has been part of all demarcation procedures of indigenous lands since 1976, despite constant changes in the specific legal procedures in place. Similarly, quilombola lands have in most cases *de facto* existed throughout the past 100 years, despite varying levels of social conflict and legal recognition. Quilombo activists had formed strong political movements to demand land rights after the colonization process of the 1970s brought many settlers to the central, north, and northeastern regions of Brazil, where most quilombola lands are located (Bowen, 2010). Recognition of their specific bundle of rights began in the mid-1980s, coinciding with the first period of analysis, and culminated in their legal recognition through the 1988 constitution and the establishment of a dedicated institution to demarcate quilombola lands (*Fundacão Cultural Palmares*). Since then, several legislative documents have further outlined demarcation processes, which INCRA took over in 2009.

Military lands, urban and transport-related lands, and water categories were omitted from the analyses, as these are less relevant to the hypothesized mechanisms relating land-tenure regimes to environmental change.

2.2.2 Defining variables of environmental change

For the quantification of environmental change as related to land resources, I used two variables: I first used deforestation as an accessible LU/LUC metric which is often openly accessible at high-resolutions and commonly used in many other land tenure studies, which facilitates comparing results to existing literature. In order to expand traditional LU/LUC metrics and encompass potential changes in biodiversity as well, I next decided to use an assessment of potential species diversity based on a global map of ecosystem types (see **2.3.2**). In using this assessment, I was able to conduct spatially explicit statistical analyses at broad spatiotemporal scales using all land parcels in Brazil as my observational units, which would not have been possible if I were limited to using site-specific sampling records of local species diversity.

Conversion of forest-to-agriculture

Calculations of forest-to-agriculture conversions over different time periods between 1985 and 2018 were carried out using 30-m-resolution annual land-cover/use dataset provided by Mapbiomas (*Project MapBiomas - Collection 4.0 of Brazilian Land Cover & Use Map Series*, n.d.). Forest-to-agriculture conversions were defined as any case where forest cover (either natural or plantation), savanna, or mangrove cover changed to any category of farming (pasture, agriculture, annual, perennial, and semi-perennial crops, and mosaic of agriculture and pasture) over the respective time period considered.

Quantification of potential species diversity

For the assessment of potential species diversity within a parcel, I rely on a global dataset of ecosystem types (GlobES), that was compiled by Remelgado and Meyer (currently in revision, example data available at (*GEO BON Data Portal: GlobES*, n.d.), also used in (Arlé et al., 2021)). To infer potential species occurrences, we matched this dataset with species-specific habitat requirements of 6,042 amphibians, birds, mammals, and reptiles (as documented in the Birdlife/IUCN Red List of Threatened Species). For each species, we calculated the Area of Habitat (AoH) (Brooks et al., 2019) within a given 1km^2 cell and year as the proportion of ecosystem types that correspond to the species' respective preferred habitats. We only considered cells that fell into the extant distributional range and altitudinal limits of a given species. The quality, accuracy, and uncertainty of these data were ensured by conducting validations multiple scales and spatial resolutions.

We calculated potential species richness at the pixel scale, as the number of species with any portion of habitat that was greater than 0. This metric is most sensitive to changes in AoH of rare species, as even small changes in different ecosystems' areas could potentially remove all AoH for those species (or conversely, add species). Furthermore, we calculated the inverse of Simpson's diversity index at the pixel scale, using the following equation:

$$\frac{1}{D} = \frac{1}{\sum_{i=1}^{s} p_i^2}$$

where p_i is the percent of a given pixel's area covered by habitat for species *i*, and *s* is the number of species with any portion (>0) of habitat in that pixel. $\frac{1}{D}$ is a measure of diversity that accounts for the evenness in habitat occupancy among the different co-occurring species in a pixel. Thus, the inverse Simpson's index is most sensitive to changes in AoH of common species, as these have the greatest impacts on the community's evenness.

To obtain parcel level estimates of potential local diversity (at the scale of 1km^2) for both metrics, I averaged the values across all pixels that fell within a given land parcel.

Covariates and other resources used

Similar sets of covariates commonly known to influence deforestation and environmental change were used in all statistical analyses. These covariates included those used for matching, as well as those used in regression models. Matching covariates used in both analyses were: market accessibility (represented by travel time to nearest city; (Nelson, 2008)), agricultural suitability (represented by slope and elevation; (Yamazaki et al., 2017)), parcel area in ha (Imaflora et al., 2018), and population density (Freire et al., 2016). Given the non-random assignment of tenure regimes and their biodiversity, and the subsequent statistical models required to estimate this change, percent cover of natural area in 1992 (*Project MapBiomas - Collection 4.0 of Brazilian Land Cover & Use Map Series*, n.d.) and Palmer's Drought Severity Index (PDSI) were additionally as matching covariates for biodiversity analyses in order to capture climatic and environmental differences between treatment and control units.

2.3 Study Context

This dissertation takes advantage of the uniquely rich environmental context in Brazil for answering questions specifically related to tenure and environmental change. In order to adequately interpret how theoretical expectations regarding the effects of land tenure on environmental change apply to the Brazilian context, in this section, I overview some basic geographic and governance aspects of the Brazil.

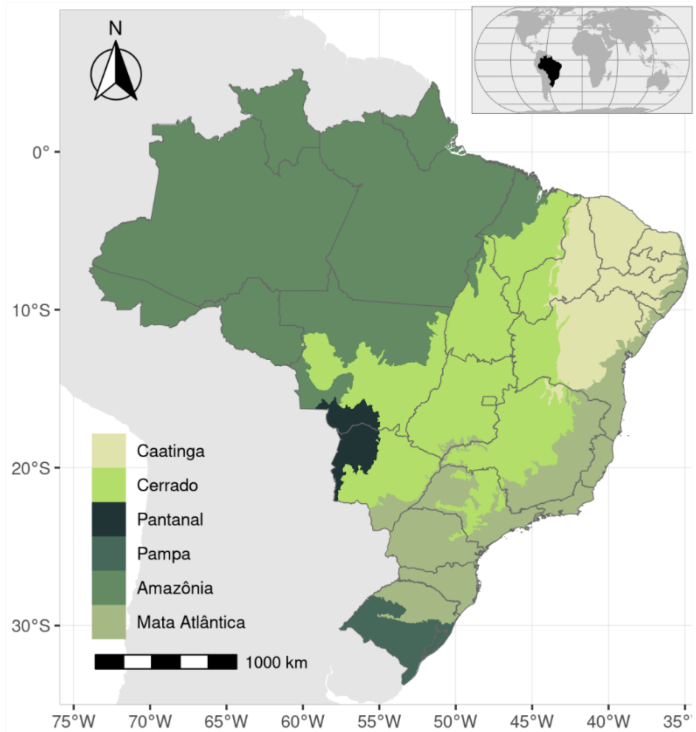

Figure 7. Federal states (outlines) and six terrestrial biomes (colors) in Brazil, including tropical rainforest in Amazonia, Atlantic forest in Mata Atlantica, savanna in the Cerrado, semi-arid dry tropical forest in the Caatinga, pastures in Pampa, and wetlands in Pantanal.

Brazil is the fifth largest nation in the world, and is a federal republic divided into 26 states and one federal district. It is the largest country in South America, sharing borders with almost every other country or territory (with the exception of Chile and Ecuador). Brazil is officially grouped into six different biomes, the largest which is the Amazon biome or Amazônia, which roughly follows the Amazon river basin and is mostly characterized as tropical rainforest. The Atlantic forest, or Mata Atlântica, is considered to be highly degraded as it was the first environment to be exploited during Portuguese colonization (Mittermeier et al., 2005), however, it is still characterized by high levels of biodiversity and endemism. The Cerrado is mostly characterized as savanna, grassland, and dry forest ecosystems, and is also a global biodiversity hotspot which is critically threatened by agricultural expansion (Klink & Machado, 2005). The Caatinga is a semi-arid region characterized as a seasonal dry tropical forest, the Pampas are flat, agropastoral grasslands, and the Pantanal is one of the world's largest wetlands (de Albuquerque et al., 2012; Harris et al., 2005; Zarbá et al., 2022)

As of 2000, Brazil had approximately 225Mha of intact forest landscapes, representing 24% of the global total (Global Forest Watch, 2021). As large carbon sinks, Brazil's forests are well regarded to play important roles in water cycling and climate regulation. According to the Convention on Biological Diversity (CBD), Brazil is estimated to hold between 10-15% of the world's biological diversity – the single most biodiverse country in the world. Globally, it is the most species-rich country (Mittermeier & Mittermeier, 1997), second only to Indonesia in species endemism (Ministerio do Meio Ambiente, 2020b). Although Brazil is considered a global leader in the environmental conservation due to its efforts to create conservation-focused protected areas throughout the past several decades (Mittermeier et al., 2005), both forests and biodiversity in Brazil remain critically threatened. During 2002-2020, the total area of humid primary forest decreased by 7.7% (Global Forest Watch, 2021).

Brazil is also global agricultural-production giant, producing roughly 30% of the world's soy and 15% of its beef, and with the agricultural sector responsible for ~20% of the country's GDP (Stabile et al., 2019). Historically, most of Brazil's agricultural growth has been through agricultural expansion, rather than intensification, and the vast majority of this expansion has occurred in the Amazon and Cerrado biomes (Lapola et al., 2014). Particularly after the 1970's, with the construction of the Transamazon highway, this expansion is regarded to have increased income, access to schools and healthcare. Yet, these gains also increased economic-and-land inequality and produced widespread environmental and degradation and loss of natural land-cover (Fearnside, 2005; Lapola et al., 2014; Stabile et al., 2019). While deforestation reduced drastically after the mid-2000's, pressures on the natural environment remain and deforestation rates have increased in recent years(Escobar, 2020; Rajão et al., 2020).

Given the global importance of environmental conservation in Brazil, alongside important policy measures that have been in place for decades (Nepstad et al., 2009), a large part of the response to these threats has been an investment in sophisticated monitoring systems. For instance, annual deforestation is recorded via satellite imagery by the *Monitoring Deforestation in the Brazilian Amazon by Satellite Project* (PRODES) alongside schemes like DETER and DETER-B which detect real-time deforestation at fine-scales (25ha). These monitoring schemes have been shown to reduce deforestation through enforcement activities (Assunção et al., 2013; Diniz et al., 2015). On the other hand, although monitoring biodiversity change is a much more challenging task, the Brazilian Program for Biodiversity Research (PPBio) launched in 2004 and plot-level information is being collected and integrated into international databases (e.g. Global Biodiversity Information Facility (GBIF)) through the Brazilian System of Information on Biodiversity (SiBBr).

As Brazil remains a key global agricultural player, as well as a critical leader in environmental conservation monitoring and policies, ongoing forest and biodiversity change monitoring and enforcement efforts will be key in order to enable scientific insight on their patterns and threats.

2.3.1 Land tenure in Brazil

Modern land-tenure regimes as they exist in Brazil today, with all the rights and regulations that apply to them, exemplify the complex historical processes of land distribution common to tropical countries. Deliberate colonization of the central and northern regions was encouraged since the 1930s, but occurred at a large scale during the period of military dictatorship (1964-1985). The Land Statute enacted in 1964 brought forth the concept of land fulfilling a "social function" – creating legal instruments for land expropriation and taxation as official means of land redistribution and regularization. In parallel, the Forest Code created in 1965 (Federal Law No. 4.771) required private landowners to leave 20-80% of the land under native vegetation, depending on the region. Soon thereafter, in the 1970s, The National Institute of Colonization and Agrarian Reform (INCRA) was created with the purpose of reclaiming unproductive land and settling the landless. Settlers were specifically incentivized to replace forest with cattle pastures or croplands. However, the official creation of these settlements was largely ineffective and many were never formalized – oftentimes large "unproductive" farms persisted, and illegal occupation of lands continued to be common. At the same time, in addition to the existing occupants of these regions (e.g., indigenous peoples, rubber *Seringueiros,* and riverine communities), land grabbers staked claims on land by counterfeiting land titles (*Grilheiros)* or creating "ghost" property owners (Damasceno et al., 2017; Fearnside, 2001; World Bank, 2014).

When the dictatorship ended and a new constitution was written in 1988, protected areas were planned on existing public lands, and the law recognized autonomous land rights for indigenous peoples and quilombolas for the first time. Still, the formalization of many these areas took 10 years to even begin, with registration and demarcations processes still ongoing to date. On the other hand, land-use rights and (dis)incentives for deforestation in public and private lands were targeted through a variety of environmental policies and programs. This included efforts specifically focusing on mitigating deforestation in the Amazon and the Cerrado biomes, often incorporating issues relating to land tenure regularization (e.g. PPCDAm (2004), PPCerrado (2010), REDD+, and the soy moratoria (2006)) (Cunha et al., 2016; Soterroni et al., 2019). It further included the regularization of *de facto* public and private lands resulting from the colonization process of the 1970s as part of the new Forest Code – the Native Vegetation Protection Law (Lei 12.651 2012). The new Forest Code provides incentives for the voluntary registration of rural public and private properties in the official Rural Environmental Cadastre (CAR), facilitating GIS-based forest monitoring of tenants' compliance with requirements to maintain certain levels of native vegetation coverage (20-80% depending on the biome (Azevedo et al., 2017; Soares-Filho et al., 2014)). Altogether, these regulations, policies, and programs have roughly defined the *de jure* and *de facto* tenure regimes in Brazil for the past 50 years.

2.4 Study design & methods for empirical analyses

Bringing together key data sources (as overviewed in **2.2**) and contextual understanding (**2.3**), this section now focuses on the quasi-experimental and statistical steps required to estimate causal effects of different land tenure regimes on deforestation and potential species diversity in Brazil.

2.4.1 Quasi-experimental analysis & matching

To estimate causal effects from observational data, I used a quasi-experimental approach that combined matching with regression analysis (Iacus et al., 2011). Matching addresses the bias that would arise if simpler regression designs were applied to the raw tenure dataset due to high levels of imbalance resulting from non-random assignment of parcels into “treatment” and “control”/comparison groups, and the high degree of model dependency from these simpler models. The specific matching method that was used, coarsened exact matching (CEM)(Iacus et al., 2011), addresses this bias by pruning the dataset to matched pairs of parcels that are highly similar with regard to potentially confounding variables in a stratified way. As recommended for large datasets, and for best improvement of balance levels, I conducted one-to-one matching, meaning that for each land parcel coded as “treatment”, I identified exactly one “control” or comparison parcel. Average treatment effects in the remaining matched sample (ATM) are subsequently estimated through regression on the balanced uncoarsened data subset.

Drawing inferences from a study design using matching requires meeting the assumption that there is no omitted-variable bias, i.e. all possible variables that influence non-random assignment are captured using matching variables. For the first analysis, where effects of tenure regimes are estimated for forest-to-agriculture conversion rates, this meant controlling for five standard, commonly used variables that are known to influence deforestation. For the second analysis, where effects of tenure regimes are estimated for two metrics of biodiversity change, two additional variables meant to capture climatic and environmental factors were also included (see previous section on Covariates in **2.2.2**). Risks of omitted-variable bias were also mitigated by 1) including fixed effects for federal states in regression analyses to capture subnational governance differences, 2) spatially clustering standard errors by municipality, and 3) assessing sensitivity of results against potential omitted-variable bias using Rosenbaum bounds (see **2.4.3**). Moreover, possible bias due to systematic differences in initial forest cover was also specifically addressed (**Figure 11**).

Commonly, impact evaluation of a given intervention, i.e. an instantaneous/short-term event would typically control for pre-treatment covariates to avoid the risk that covariates on the causal “pathway from exposure to outcome” might confound part of the investigated effect (VanderWeele, 2019). However, the intent of this dissertation was not to measure the direct impact of a particular intervention (e.g. a titling program), but rather, the longer-term effects of alternative, where stable tenure regimes (i.e. treatments) acted continuously throughout the study period. Correspondingly, for the first analysis on deforestation, the time-varying population-density variable was averaged over the years of the respective period (including linearly interpolated/extrapolated values as necessary). Time-varying population-density and PDSI variables were used accordingly in the panel-analysis for the second analysis (see following section on regression analysis **2.4.2**).

Coarsened-exact matching was implemented using the "*cem"* package (Iacus et al., 2009) in R versions 3.5.1-4.0.2 (R Core Team, 2020). CEM involves temporarily "coarsening" each confounding variable into bins (predetermined strata). Automated coarsening was used for elevation, slope, human-population-density, and, in the case of the biodiversity analyses PDSI. Manual bins were defined for travel time to nearest city, for parcel area, and, in the case of the biodiversity analyses, percent natural cover. Travel time to nearest city was divided into bins of 0-2, >2-6, >6-12, >12-24, and >24 hours, parcel area was divided into 14 bins of 0-2, >2-5, >5-15, > 15-50, >50-100, >100-500, >500-1,000, >5,000-10,000, >10,000-50,000, >50,000-100,000, >100,000-500,000, >500,000-1,000,000 ha. By conducting CEM individually for each defined spatiotemporal extent, I assured exact matching considering the total spatial and temporal variation in the covariates at the respective scale.

In the case of the biodiversity analyses, in order to best align with the resolution at which the biodiversity data were available, only parcels larger than or equal to 100 ha are included in the analysis, meaning this analysis does not apply to smaller farms/parcels. Given these scale-related constraints, bins for the area variable are manually defined to roughly correspond to fiscal modules (100<300, 300-<1000, 1000-<3000, 3000-<10000, 10000-<30000, 30000-<100000, 100000-<3000000, 3000000-<10000000 ha). Bins for percent cover natural area were manually defined as 0-<25%, 25-<50%, 50-<75%, and >75%.

While CEM, in particular, has a range of advantages over other matching approaches (Iacus et al., 2011), identifying exact matches is generally difficult when there is little overlap in parcel-level similarity among covariates. For the deforestation analysis, the large number of parcels in the Imaflora dataset (~4 million) afforded retaining sufficiently large data subsets for unbiased parameter estimation for most tenure-regime comparisons and spatiotemporal scales (44 to 34,218 of unique observations, corresponding to ≥6 observations per parameter; (Vittinghoff & McCulloch, 2007); see **Tables 12-13**). Due to very small numbers of matched parcels (4 to 28), deforestation effects for communal regimes were not estimated in the Caatinga, Cerrado, and Mata Atlântica, nor for any regime other than undesignated/untitled and private in the Pampas and Pantanal biomes.

The L_1 measure developed by King et al. (Iacus et al., 2009) was used to calculate remaining imbalance post-matching. Moreover, imbalance measures are explicitly incorporated into figure visualizations as well as the cross-scale synthesis of results (see **Tables 6-7**).

2.4.2 Regression analysis

To estimate the effect of different regimes on forest conversion to agriculture, generalized linear models (GLMs) with a binomial error distribution and a logit link are fitted to each respective matched dataset. Uncoarsened variables are used as model covariates and federal state is additionally included as a fixed-effect to control for state-level differences in governance regimes and effectiveness. To control for possibly remaining spatial autocorrelation in model residuals, standard errors are clustered by municipality. I estimate:

$$logit(p) = \beta_0 + \beta_1 tf + \beta_2 l + \beta_3 s + \beta_4 tt + \beta_5 pd + \beta_6 r + \beta_7 st + \varepsilon$$

where *p* is the per-pixel probability of forest conversion, *tf* is the tenure form, *l* is the average elevation in meters, *s* is the average slope in degrees, *tt* is the average travel time to nearest city in minutes, *pd* is the average population density, *r* is the area of the parcel in ha, and *st* the federal state. Note that binomial models of percentage forest loss automatically capture differences in initial forest area, by evaluating the total forest areas (counts of pixels) that were converted to agriculture vs. those that remained.

Average marginal effects (AME) were computed using the "*margins*" package in R (Leeper, 2021), transforming coefficient estimates to average per-forest-pixel probability of conversion to agriculture with respect to the tenure form in question (Leeper, 2017) (**Table 12-13**).

To estimate effects of different tenure regimes on both species richness and Simpson's diversity index, I fitted a time fixed effects regression model using longitudinal data (also known as a panel-data regression model, or a cross-sectional time series) to each matched dataset at the scale of Brazil. This controls for factors that are constant over time, but differ amongst parcels (elevation, slope, area, and travel time to nearest city), as well as for factors that differ over time (population density, and climate indices). I estimate absolute values of species richness using a quasi-Poisson distribution:

$$SpRichness_{it} = \beta_0 + \beta_1 Tenure_i + \alpha_i + \gamma_{it} + \varepsilon_{it}$$

And estimate Simpson's index of diversity is modelled using a log transformation:

$$ln(invSimpson's_{it}) = \beta_0 + \beta_1 Tenure_i + \alpha_i + \gamma_{it} + \varepsilon_{it}$$

With i=1, n and t=1,…T. The α_i are the factors are time invariant, which include the year (1996, 2007, and 2018), area of the parcel in ha, elevation in meters, slope in degrees, and travel time in minutes, and federal state as a dummy variable. The γ_{it} are the time variant factors, which include average population density, and average value of the Palmer's Drought Severity Index (PDSI). Standard errors are clustered by municipality again, because, as stated above this approach is typically used to account for remaining spatial auto-correlation in modelling land-use changes. Average marginal effects of the matched samples are computed using the *margins* package in R, and reported at the scale of the response variable for ease of interpretation (percent change in either biodiversity metric).

2.4.3 Sensitivity analyses

Rosenbaum bounds are calculated as a sensitivity analysis to assess whether model estimates are robust to the possible presence of omitted-variable bias. Rosenbaum bounds quantify the sensitivity of regression results to different magnitudes of hypothetical bias that might be caused by missing important confounders in the matching procedure (Rosenbaum, 2007). Here, the magnitudes of bias (Γ) are expressed as the change in the odds of being selected into treatment or control caused by the addition of a hypothetical unobserved confounder.

2.4.4 Seeking generality: synthesis over broad scales

By empirically estimating effects of different land tenure regimes on environmental change at larger spatial and temporal extents, I aim to synthesize the direction, strength, and consistency of these effects at broader scales. Rather than near-term impacts of specific tenure-intervention events (e.g., titling), this approach captures the differential impacts of alternative regimes over periods of several years to decades, and provides synthesis of how consistent effects of these regime differences are across different regional-historical contexts.

Causal effects in complex human-environment systems are often highly context-specific (Meyfroidt et al., 2018), which can limit the transferability of conclusions from contextually bound studies. Yet, effects shown to be generalizable across very different socio-environmental contexts may also be most likely to be generalizable to yet other contexts. Based on this tenet, for the first analysis, where effects of different tenure regimes on forest conversion to agriculture were estimated, 49 different combinations of spatial and temporal extents were defined for analysis. The full quasi-experimental procedures for each tenure-regime comparison was repeated for each of these spatiotemporal scales.

Thus, capturing the net agriculture-to-forest conversion over the full 1985-2018 period, one “large” spatiotemporal extent covering the entire spatial extent of Brazil and was first defined and analyzed. Then, analyses were repeated over the same temporal extent (1985-2018) but over the six narrower spatial extents defined by Brazil’s biomes (Amazônia, Caatinga, Cerrado, Mata Atlântica, Pampa, and Pantanal). These biomes correspond to highly distinctive environmental and socioeconomic conditions, ranging from early-colonized, economically diversified, and intensively governed regions, to newly emerging agroeconomic frontiers, economically marginalized drylands, and remote rainforest areas.

Additionally, analyses were repeated over both large and narrower spatial extents over six narrower temporal extents, which were defined to coincide with major deforestation periods in Brazil. The first temporal extent (1985-1990), during which several tenure types first received legal recognition, was a time of deep economic crisis, high inflation rates, and high levels of social unrest. The period of 1990-1995 represents a time of economic recovery; elections in 1994 contributed towards increasing access to agricultural credit in several key federal states, agricultural mechanization increased in key regions, and El Niño-related droughts and fires added to a sharp peak in deforestation rates in 1995. During 1996-1999, as well as 2000-2004, there was steady economic growth, with deforestation peaking again in 2004. 2005-2012 marks a period of declining deforestation rates after a drop in global soy prices and renewed environmental legislation and enforcement focused on the private sector (e.g., the soy moratorium of 2006; (Nepstad et al., 2014), the proposal of REDD+; (Moutinho et al., 2011)). Finally, the period of 2013-2018 corresponds to the most recent amendment of the Forest Code, which has been widely criticized for its leniency in granting amnesty for past deforestation and lowering the requirements for restoration (Soares-Filho et al., 2014).

To assess which statements on deforestation effects of tenure-regime differences might be transferable across diverse socio-environmental contexts (e.g., different environmental settings, time periods, or administrative levels), scale-specific effects were synthesized in two ways. First, for each comparison (e.g., private vs. undesignated/untitled), the consistency of the direction of the causal effect was assessed by calculating percentages of scale-specific models with, respectively, significant deforestation-increasing (positive), significant deforestation-decreasing (negative), and no significant effects (see **Tables 6-7**). These analyses address the applied question of how reliably a particular tenure-regime change might decrease long-term deforestation rates under different (e.g., unknown, or unforeseeable) socio-environmental contexts. Second, the consistency of the relative ranking of alternative tenure regimes was assessed by the magnitudes of their effects vis-a-vis a given counterfactual, by calculating percentages of scales at which each tenure regime showed higher/lower effects than all others (**Table 6**). These analyses address the applied question of which of alternative tenure-regime changes might most/least reliably cause *large* reductions in deforestation.

Robustness of the results of this cross-scale synthesis against possible bias in the relative reliability of the tenure-comparison- and scale-specific causal tests was also assessed. To this end, balance-weighted percentages were additionally calculated, effectively downweighing any cases where covariate overlap post-matching remained low. All main conclusions of this analysis are based on qualitatively consistent balance-weighted/unweighted results. Specifically, balance-weighted percentages of cases with significant-negative, significant-positive, and nonsignificant effects were calculated by weighting each tenure-comparison- and scale-specific result contributing to a given percentage value by the inverse of the remaining imbalance (L_1) in the respective dataset (see **Tables 6-7**).

Similarly, weighted percentages of scales at which each tenure category had higher/lower-ranked effects than all others were calculated by weighting the entire set of tenure-regime comparisons contributing to the ranking at a given scale by the inverse imbalance (L_1) of the least-balanced dataset at that scale (**Table 6**). In addition to this balance-weighing, robustness against violations of the assumption of constant treatment of parcels with strict-protection and sustainable-use regimes were calculated by using results based on time-filtered datasets to calculate alternative versions of percentages with significant-negative, significant-positive, and nonsignificant effects (see **Tables 7, 14**).

How often tenure regimes were ranked as most/least effective in reducing deforestation could also be biased by systematic differences in the different regimes' exposures to deforestation pressures. Such bias would in principle be possible, as these assessments of relative effectiveness are based on comparisons among the regimes' effect sizes at each scale, which were all estimated with unique combinations of matched parcels. In particular, the indirect comparison of strict-protection vs. sustainable-use regimes (vis-a-vis an undesignated/untitled counterfactual) could be expected to be potentially affected by differences in geographical siting of the different types of conservation areas relative to deforestation pressures, which has been previously reported for Amazonia (Pfaff, Robalino, Sandoval, et al., 2015). Thus, matched parcels' average covariate values at the specific scales where they were most/least effective were assessed to evaluate whether their differing percentages of most/least effective cases reflected any systematic differences. While some cases where the two tenure regimes do differ with respect to specific covariates, these cases did not indicate any systematic bias. For example, strict-protection regimes were often ranked as less effective in reducing deforestation than sustainable-use areas in the Amazonia and Mata Atlântica biomes, despite occurring in, respectively, more remote, and higher-elevation areas on average (cf. (Joppa & Pfaff, 2009).

Note that, while these quasi-repetitions allowed for systematic cross-scale testing of effects of different tenure regimes on deforestation, testing of effects on biodiversity change, on the other hand, has different challenges that require a different setup for analysis. In this case, given the scale-constraints in the biodiversity metrics used (see previous section on matching), analyses were only conducted at one "large" spatial extent of all Brazil. There were carried out using panel data analysis for biodiversity change in 1996, 2006, and 2018 (see previous section on Regression Analysis), allowing for interpretations of findings over one broad spatiotemporal extent, as well as three particular time periods. Rather than emulating quasi-repetitions as before, this approach was deemed most appropriate for biodiversity change analysis, given the scale-dependency of this change, and for best interpretation of biodiversity change using derived metrics (Brooks et al., 2019). Thus, the quasi-experimental setup for estimating effects on biodiversity complements the deforestation analysis at multiple scales by using panel analysis, and estimating effects at macro-spatiotemporal scale.

3. Results

3.1 State of land tenure data in Latin America: gaps, challenges, opportunities

The initial methodological step in this dissertation involved searching for spatially explicit data on land tenure across large spatiotemporal extents, and this search involved networking and engagement with major organizations and stakeholders with expertise on land tenure data. However, this effort did not lead to any substantial finding or gaining access to spatially-explicit land tenure datasets. Concrete data collected or held by these agencies were rare, unavailable due to data privacy restrictions, or not shared (**Table 3**).

Table 3. Stakeholders and organizations contacted for access to possible land tenure data during October 2017-September 2019.

Stakeholder contacted	Possible findable, accessible data or expertise
Land Governance Assessment Framework (LGAF) from the World Bank (WB)	Have collected nationally representative household survey data with geographic information of land governance indicators. Openly available data aggregated at national scale.
Landmark Map	Host a compilation of indigenous or communal land data vector and/or gridded data, with listed sources.
Prindex	Have conducted nationally representative household surveys in 140 countries on land tenure security perceptions. Openly available data are aggregated at national scale.
Word Resources Institute (WRI)	Generally, provide expertise on the land tenure and are active partners/members of other initiatives (e.g. Landmark, the Access Initiative, Land and Resource Rights Initiative). Particular projects, e.g. the Congo Basin Forest Atlas, contain data on land-use allocation (forest concessions, protected areas, mining permits, etc.), and these data are hosted by Global Forest Watch.
Rights and Resources Initiative (RRI)	Provided expertise and published literature comparing property rights around the world.
Cadasta	Hold spatially-explicit parcel-level data on regularized properties from around the world, as a result of their consulting work in establishing country cadasters.
Millennium Challenge Corporation (MCC)	Publish national-level indicators and scorecards regarding property rights
Indigenous and Community Conserved Areas (ICCA) Consortium	Advocacy group with expertise
International Union for the Conservation of Nature (IUCN)	Leads the Natural Resource Governance Framework (NRGF), an initiative that sets standards and guidance for decision-makers for decisions on the use of natural resources and the distribution of

Stakeholder contacted	Possible findable, accessible data or expertise
	nature's benefits. The implementation of NRGF could potentially include tenure and property-rights-related data and standards
Access Initiative	The Access Initiative is a network with members around the world that focus on promoting transparency and access to data. They have databases on access improvement (2005-2010, and 2008-2010) and on ENR (environment and natural resource relevant) commitments.

As a result of better understanding the lack of findable and accessible data sourced from global/international stakeholders and experts, I subsequently prioritized directly contacting specific agencies in the Latin American region (see **2.2**). Of these targeted agencies, and throughout the region, I found a wide range in data findability, accessibility, interoperability, and reusability (FAIR guiding principles for scientific data) (Wilkinson et al., 2016). Therefore, although this targeted data search effort led to finding and accessing some spatially-explicit datasets on land tenure, challenges related to data accessibility and interoperability still presented barriers in using this data for scientific analysis (**Table 4**).

Table 4. Latin American countries and agencies contacted for access to spatially explicit land tenure data. There was a wide variety of findability (F), accessibility (A), interoperability (I), and Reusability (R) found (as defined by (Wilkinson et al., 2016)). This varied from countries where there is very little known about the existence of this data, to countries where there is a great level of detail of data collected, yet with some interoperability limitations. When found, FAIR principles are indicate using a checkmark. Though non-exhaustive, the different organizations for each country show the plural nature of institutional governance of land tenure in the region, with both centralized and decentralized countries having different data statuses.

Country	Organization	Description of data	F	A	I	R
Argentina	Consejo Federal del Catastro	Data exists, is available, but is not centralized and must be requested from each individual administrative unit in the country	✓	✓		
Belize	Land Information Center form the Ministry of Natural Resources and Immigration	Unable to establish contact after multiple emails and telephone calls to different agencies; Unsure if data exists, or, if data exists, location, storage, and access protocols are unknown				
Bolivia	GeoBolivia, Instituto Nacional de Reforema Agraria	Some data regarding specific tenure categories exists, although with access limitations (not retrievable using an open protocol). Parcel-level data for the entire country may exist, but is not made publicly accessible	✓			
Brazil	Imaflora compiles information from Sistema Nacional de	Data exists, is open access	✓	✓	✓	✓

Country	Organization	Description of data	F	A	I	R
	Informação sobre Meio Ambiente, Cadastro Ambiental Rural					
Chile	Centro de información de Recursos Naturales	Data exists, is accessible but at a prohibitive cost	✓			
Colombia	Instituto Geográfico Agustín Codazzi	Data findable, is openly accessible, but not interoperable as it lacks the metadata required for representing information (i.e. no information on tenure categories, preventing scientific analysis)	✓	✓		
Costa Rica	Sistema Nacional de Información Territorial: Infraestructura Nacional de Datos Espaciales, Dirección General del Registro Nacional	Access to data requires a formal request from a citizen of that country	✓			
Ecuador	Sistema Nacional de Información de Tierras Rurales e Infraestructura Tecnológica (SIGTIERRAS)	Unsure if data exists, or, if data exists, location, storage, and access protocols are unknown				
Guatemala	Registro de Información Catastral (RIC)	Access to data requires a formal request from a citizen of that country	✓			
Honduras	Dirección General de Catastro y Geografía	Access to data requires a formal request from a citizen of that country; Unsure if data exists, or, if data exists, location, storage, and access protocols are unknown				
Mexico	Registro Agrario Nacional, Secretaría de Desarrollo Agrario, Territorial y Urbano	Data is findable, was accessible*, but is not interoperable as the provided metadata does not distinguish between different tenure categories leading to overlaps in definitions that prevent scientific analysis	✓	✓*		

Country	Organization	Description of data	F	A	I	R
Nicaragua	Catastro Nacional y Registro Público de la Propiedad, Dirección General de Catastro Físico, Catastro Fiscal de la Dirección General de Ingresos del Ministerio de Hacienda y Crédito Público	Access to data requires a formal request from a citizen of that country	✓			
Panama	Autoridad Nacional de Administración de Tierras	Unable to establish contact after multiple emails and telephone calls to different agencies				
Paraguay	Servicio Nacional de Catastro, Ministerio de Hacienda	Unsure if data exists, or, if data exists, location, storage, and access protocols are unknown				
Peru	Portal Nacional de Datos Abiertos - Organismo de formalización de la propiedad informal, Ministerio de agricultura y riego, Instituto Bien Común (IBC)	Data on only certain tenure regimes is findable and accessible.	✓	✓		
Uruguay	Dirección Nacional del Catastro	Data findable and accessible, but not interoperable as it does not include metadata which provides information on land tenure categories	✓	✓		

* Though this data was accessible in 2019, it is no longer accessible in 2022

Engaging these data sources and agencies highlighted a few main challenges in acquiring spatially explicit land tenure data. First, for several countries, there was an infrastructure challenge: a land tenure data system did not really exist, was precarious, or was under development. This lack of infrastructure may be related to either the data collection (i.e. demarcating properties in the field and connecting these to a geographic information system), or to central data management and storage. Second, there was a major conceptual/terminology challenge because specific definitions of land tenure and their categories varied widely across countries. For instance, most agencies required repeated explanations of what kind of data I was searching for, and were generally confused by tenure categories that at the same time were anonymized. Although I repeatedly emphasized that this research did not require a property owner's name, but rather a categorization of whether the property was privately, publicly, or otherwise held, many stakeholders I spoke to did not understand this concept, and would often have to explain the terminologies for different categories used in that particular country. These different terminologies and thematic variations in categories made it difficult to compare across countries because they would require an in-depth assessment of what rights each country's categories include.

For most of these agencies, the concept of spatially-explicit land tenure data referred to the demarcation of private lands. Indigenous and other traditional, or communally held lands were usually under the jurisdiction of other agencies, which may be connected, but were often independent. On the other hand, gaining information or access to data on public lands was the most challenging; besides protected areas and areas such as military bases, publicly owned lands tended to not be demarcated or recorded, making it difficult to systematize information on these in many countries. Overall, these institutional gaps – and at times, overlaps – led to many cases where countries had some data openly accessible (e.g. protected areas, indigenous lands), yet the full picture for the entire country (e.g. a wall-to-wall map) remained elusive, inhibiting multi-national or national-scale assessments of tenure regimes.

Evidently, there is a crucial gap between data collection/management and its publication for general public access. While a few countries did maintain publicly available, vector data of demarcated lands, these data rarely include important metadata required for any further analysis; i.e. they lack basic information such as definitions or categories. Further information, such as date of creation/demarcation was rarely found.

For instance, in Mexico, due to the land reform at the beginning of the century and subsequent recent amendments, the land tenure system is well-documented and data were openly available through the *Registro Agrario Nacional* (RAN). However, while parcel-level data on the main categories of land were available (*ejidos*, *tierras de uso comun*, and *tierras comunales*)[1], accompanying materials from the RAN (or other published material found online) did not define distinguishing factors between categories in terms of property rights. Furthermore, overlaps between categories and protected areas further confounded the legal use of natural resources these lands may claim. These limitations indicate these data were non-interoperable, and are currently no longer accessible (see footnote). Another example is the land tenure registration system in Colombia, where almost the entire country was mapped and made openly available through the *Instituto Geográfico Agustín Codazzi* (IGAC), yet the meaning of parcels remained unclear. Although the vector data included columns with alphanumeric codes, translating tables for these codes were not available to the public, meaning these data remain non-interoperable[2]. While the lack of strict categories, and the *de facto* complexity of land ownership are a reality of many tropical countries, the lack of further, clarifying, openly available information regarding land inhibits scientific analysis.

Overall, the concept of publishing openly available land tenure data was sensitive because these data involve collecting and publishing personal information that needs to either be anonymized or otherwise excluded from public versions of data. They also involve explicitly mapping relationships that at times may be contentious, which may open doors for unintended conflict, and can even endangering certain communities/people groups that do not wish to be mapped valid reasons (e.g. land-grabbing or other exploitative activities). For these reasons, data privacy

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[1] Data were available at the time of this exploratory data search (2017-2019). As of June 2022, these data are not openly available, as external access to the RAN server has been blocked.

[2] Note that in both of cases of Mexico and Colombia, the RAN and the IGAC were contacted for further clarification, yet these requests for providing any further information than publicly available were denied.

restrictions remain important to safeguard. Nonetheless, data with respect to indigenous peoples and local communities (IPLCs) should be guided by the CARE principles for indigenous data governance (Carroll et al., 2021), which ensure indigenous people equitably benefit from these data.

To conclude, spatially explicit data on land tenure were found to be rarely findable, accessible, interoperable or reusable – from global stakeholder sources as well as from targeted countries in Latin America. Apart from global efforts that have certain thematic foci (e.g. mapping IPLC's (Garnett et al., 2018)), and national-level indices, substantial gaps and steep challenges for empirically testing the association between land tenure and environmental change remain. Notwithstanding the need to safeguard data privacy, the lack of access to spatially explicit land tenure data hinders scientific progress in this area (Rissman et al., 2017).

While these gaps present an opportunity for future research and the further development of land tenure registration systems, in light of overviewed limitations, the data available in Brazil presented a unique opportunity for scientific analysis. With parcel-level data covering over 80% of its territory, and a full explanation of its categorization and data harmonization process, *The Atlas of Brazilian Agriculture* (Imaflora et al., 2018) was the only FAIR dataset that allowed for cross-comparison of all tenure regimes in a country. Moreover, these data cover a large part of the continent's tropics, where the natural environment is under acute anthropogenic-driven threats. At the same time, the categories represented in these data are often found in neighboring countries, with common governance challenges, under similar environmental pressures. Thus, the openly available parcel-level land tenure data in Brazil allowed for a quasi-experimental analysis to be set up at large spatiotemporal scales in a context where understanding the underlying causes of environmental change is of utmost importance.

3.2 Theoretically expected effects of different land tenure regimes on environmental change

Before conducting any empirical analyses, it was important to contextualize the theoretical and empirical expectations regarding effects of land tenure on deforestation to the specific counterfactual questions set up in subsequent analyses. To this end, a qualitative analysis was employed to examine how predominant hypotheses in the literature would predict deforestation effects of each tenure regime comparison (e.g. private lands relative to undesignated/untitled lands).

First, predominant hypothesized mechanisms were gathered from the literature and categorized by their thematic dimension. Of these thematic dimensions, there were two main groups of hypothesized mechanisms identified, those which involve a particular property right within "the bundle of rights", and those which involve other dimensions of tenure (e.g. tenure security, broader governance settings). Then, a six-point scale was used to classify the predicted deforestation effect a particular hypothesis would expect from a given tenure comparison that exists within the quasi-experimental setup. This six-point scale describes two levels of deforestation increase, two levels of deforestation decrease, and no effect (**Table 5**).

Though this qualitative overview is non-exhaustive, the hypothesized mechanisms analyzed include a variety of disciplines, ranging from neo-classical and political economics, to sustainability sciences. Note that while surveying the literature, articulating specific mechanisms was quite challenging, as these are not often explicitly stated. Thus, while this analysis reflects only an interpretation of these theories and hypotheses, the main intent was to *i*) show the large breadth of expected effects of tenure on deforestation, as well as *ii*) contextualize subsequent empirical findings to the existing theoretical body of work, and discuss how certain theories/hypotheses may be un/supported by these findings.

Table 5 Non-exhaustive overview of hypotheses linking land tenure to deforestation, along with their predictions on the direction and relative strength of effects of different land-tenure regimes on deforestation rates. The top group of hypotheses, *Bundles of Rights*, are classified by the rights dimension that they mainly address, either directly or through a series of mechanisms, and the bottom group, *Cross-cutting themes*, relates to other tenure-related aspects. Arrows indicate predicted increases/decreases of deforestation of a shift from either undesignated/untitled (left) or private lands (right) to each alternative tenure regime. Arrows follow a six-point scale, with the dark green downward-pointing arrow indicating the strongest predicted decreases in deforestation and the dark red upward-pointing arrows indicating the strongest increases. Note that these are *ceteris paribus* predictions, assuming that the specified mechanisms would affect deforestation rates in isolation, rather than in an interplay of multiple mechanisms. Also note that these predictions reflect the specific bundles of rights associated with land-tenure regimes in Brazil. Because not all hypotheses are relevant to all comparisons, some cells are left blank.

Thematic dimension	**Hypothesized mechanism**	**References**	**Predicted effect of tenure regime on deforestation, relative to undesignated/untitled public lands**						**Predicted effect of tenure regime on deforestation, relative to private lands**					
			Private	Protected area	Sustainable use	Indigenous	Quilombola	Communal	Undesignated/ untitled public	Protected area	Sustainable use	Indigenous	Quilombola	Communal
Bundle of Rights														
Exclusion	Open-access, common-pool resources are by definition non-excludable. Low exclusion rights will increase deforestation through unsustainable use by multiple competing resource users (Gordon, 1954; Hardin, 1968; Browder et al., 1997). Undesignated/untitled public lands lack both clear supervision by any designated agency and effective exclusion rights, making them often de-facto open access environments. Traditionally, community-based tenure regimes have been viewed as facing similar challenges in excluding outside users due to different impediments to collective action (Grafton, 2000; Sandler, 2015).	Gordon, 1954; Hardin 1968; Browder & Godfrey, 1997; Grafton 2000; Sandler 2015	↘	↘	↘	↘	↘	↘	↗	→	↗	↗	↗	↗
Alienation	Alienation rights allow tenants to use land as collateral in business transactions and to access credit, thus providing them larger financial means to engage in forest-displacing agricultural	de Soto 2000; Place and Otsuka, 2002	↗	↘	↘	↘	↘	↗	↘	↘	↘	↘	↘	↘

Thematic dimension	Hypothesized mechanism	References	Predicted effect of tenure regime on deforestation, relative to undesignated/untitled public lands						Predicted effect of tenure regime on deforestation, relative to private lands					
			Private	Protected area	Sustainable use	Indigenous	Quilombola	Communal	Undesignated/ untitled public	Protected area	Sustainable use	Indigenous	Quilombola	Communal
	activities. By contrast, land without alienation rights (e.g. untitled public lands, indigenous lands, and quilombola lands) do not provide these options, thus inhibiting investments in deforestation-promoting land uses (de Soto, 2000; Place & Otsuka, 2002).													
Alienation	Under sufficiently functioning land markets, rights to rent out or sell land will eventually result in lands being transferred to those entities who can put them to the financially most productive use, which will often be a non-forest use (Deininger et al., 2003a).	Deininger et al., 2003	↗	↘	↘	↘	↘	↗	↘	↘	↘	↘	↘	↘
Alienation and withdrawal rights	Only land that can be legally be sold or otherwise alienated by the current tenant is potentially available to people searching for land for farming (mainly private, and to a lesser extent communal and undesignated/untitled lands). Because the expected higher agricultural profits enabled by commercial withdrawal rights tend to be factored into land prices for private lands on formal land markets, these are often unaffordable to poor smallholders or land-less settlers searching for land. These will thus instead be forced to settle on undesignated public lands at the “frontier” (Binswanger, 1991).	Binswanger, 1991	↘	↘	↘	↘	↘	↘	↗	↘	↘	↘	↘	↗

Thematic dimension	**Hypothesized mechanism**	**References**	**Predicted effect of tenure regime on deforestation, relative to undesignated/untitled public lands**						**Predicted effect of tenure regime on deforestation, relative to private lands**					
			Private	Protected area	Sustainable use	Indigenous	Quilombola	Communal	Undesignated/ untitled public	Protected area	Sustainable use	Indigenous	Quilombola	Communal
Withdrawal and market integration	Tenure forms that grant commercial withdrawal rights are economically more capable of engaging in high-input land-uses, facilitating deforestation at comparatively larger scales. This effect is stronger if tenants are more capable of commercializing their resources through greater market integration (Anderson et al., 2018).	Anderson, 2018	↗	↘	↗	↘	↗	→	↘	↘	↘	↘	↘	↘
Withdrawal and perceived tenure security (e.g., through private titles)	Tenure forms with commercial withdrawal rights and high perceptions of tenure security provide greater incentives to engage in forest-displacing land-use activities (e.g., cropping or cattle ranching). For example, private tenure, with both commercial withdrawal rights and often higher tenure security, should thus lead to higher deforestation rates compares to undesignated/untitled lands, where commercial withdrawal is unregulated or encouraged, but there is little assurance of future benefits from current investments in land-use (Liscow, 2013).	Liscow, 2013	↗	↘	↘	↘	↘	↘	↘	↘	↘	↘	↘	↘
Withdrawal (non-commercial)	Deforestation through subsistence use is most likely to occur in contexts where land users are dependent on unsustainably exploiting their forest resources for their short-term survival (e.g., during climate-induced resource shortages and in absence of alternative livelihood options)(Perrings, 1989). Where this is the case, tenure regimes with highly	Perrings, 1989	↘	↘	↘	↗	↗	↗	↗	↘	→	↗	↗	↗

Thematic dimension	Hypothesized mechanism	References	Predicted effect of tenure regime on deforestation, relative to undesignated/untitled public lands						Predicted effect of tenure regime on deforestation, relative to private lands					
			Private	Protected area	Sustainable use	Indigenous	Quilombola	Communal	Undesignated/ untitled public	Protected area	Sustainable use	Indigenous	Quilombola	Communal
	restricted or no withdrawal rights for subsistence (mostly fully protected areas) will have lower deforestation rates than all those with withdrawal rights for subsistence. Among those tenure regimes, those that *only* grant withdrawal rights for subsistence (e.g., indigenous), will have higher rates of deforestation compared to those tenure regimes that grant tenants restricted commercial withdrawal rights (e.g., quilombola, communal, sustainable-use areas) and those that do not explicitly prohibit commercial exploitation (e.g., rural settlements on public lands). Those tenure regimes that enable full integration into markets (private properties) will least strongly affect forest resources via subsistence withdrawal, as the latter regimes provide better options for alternative (non-subsistence-withdrawal) ways of sustaining livelihoods.													
Withdrawal (commercial and non-commercial)	Tenure regimes where resource withdrawal is either not restricted or incentivized will see higher deforestation rates (Angelsen, 1999; Fearnside, 2005; Redo et al., 2011). For example, undesignated/untitled public lands will often have higher deforestation, as governments rarely place restrictions of deforesting them, or even incentivize it by granting land claims based on prior clearance of forest, or by allowing settlement conditionally	Angelsen, 1999; Fearnside, 2005; Redo, 2011	↘	↘	↘	↘	↘	↘	↗	↘	↘	↘	→	↘

Thematic dimension	Hypothesized mechanism	References	Predicted effect of tenure regime on deforestation, relative to undesignated/untitled public lands						Predicted effect of tenure regime on deforestation, relative to private lands					
			Private	Protected area	Sustainable use	Indigenous	Quilombola	Communal	Undesignated/ untitled public	Protected area	Sustainable use	Indigenous	Quilombola	Communal
	on putting the land to productive (i.e., agricultural) use.													
Withdrawal (commercial and non-commercial)	Tenure regimes that grant but regulate rights to withdraw forest resources incentivize tenants to manage these resources for long-term sustainability, leading to lower deforestation rates compared to regimes with no or more unregulated withdrawal rights (Bray et al., 2008; Duchelle et al., 2012; Ellis & Porter-Bolland, 2008; Nepstad et al., 2006; Porter-Bolland et al., 2012).	Nepstad et al., 2006; Bray et al., 2008; Ellis and Porter-Bolland, 2008; Duchelle, 2012; Porter-Bolland et al., 2012	↘	→	↘	↘	↘	↘	↗	↗	↘	↘	↘	↘
Exclusion & due process (or other mechanisms increasing tenure security)	Tenure forms with stronger exclusion rights, together with due-process rights or other mechanisms that provide tenure security, create the highest incentives for investments in the resource, by providing assurance that the later benefits from resource withdrawal or other exploitation can be enjoyed exclusively (Deacon, 1994b; Deininger et al., 2003b; Birdyshaw & Ellis, 2007). Thus, tenure forms with greater assured exclusivity of resource rights are expected to lead to the allocation of land to the use form of greatest long-term economic utility to the tenant. This will commonly be agricultural uses in private farms and public rural settlements, and forest uses in protected areas, sustainable use areas, and indigenous reserves, with more ambiguous	Birdyshaw and Ellis, 2007; Ellis, 2007; Deacon_et al., 1994; Deininger et al., 2003	↗	↘	↘	↘	↘	↗ ↘	↘	↘	↘	↘	↘	↘

Thematic dimension	Hypothesized mechanism	References	Predicted effect of tenure regime on deforestation, relative to undesignated/untitled public lands						Predicted effect of tenure regime on deforestation, relative to private lands					
			Private	Protected area	Sustainable use	Indigenous	Quilombola	Communal	Undesignated/ untitled public	Protected area	Sustainable use	Indigenous	Quilombola	Communal
	outcomes expected for other community-based tenure regimes.													
Types of tenants and main rights holders	Traditional communities collectively holding land (e.g. indigenous, quilombola, and other communities with traditionally-rooted land-tenure regimes) typically create societal rules to effectively manage common forest resources and govern their use. Community members tend to follow these rules to avoid social exclusion, leading to reduced degradation of communally regulated forest resources, relative to state-managed resources (Mendelsohn & Balick, 1995; Gibson et al., 2000; Baland & Platteau, 2000). Undesignated/untitled public lands are expected to have higher rates of deforestation than indigenous, quilombola, and communal lands.	Mendelsohn and Balick, 1995; Gibson et al., 2000; Baland and Platteau 2000				↘	↘	↘						
Exclusion	In contexts where the holder of monitoring, enforcement, or other duties has limited capacity to meet these duties, excludability is impaired. In low-governance regions, where public institutions have limited capacities, tenure regimes where the state is the main duty holder should thus have higher deforestation rates than tenure regimes where local tenants are responsible for these duties (Angelsen, 1999; Grafton, 2000; Fearnside, 2005; Nolte et al., 2013). Among the latter regimes, the ability to	Angelsen, 1999; Grafton, 2000; Fearnside, 2005, Nolte et al. 2013	↘	→	→	→	↘	→	↗	↗	↗	↗	↘	↗

Thematic dimension	**Hypothesized mechanism**	**References**	**Predicted effect of tenure regime on deforestation, relative to undesignated/untitled public lands**						**Predicted effect of tenure regime on deforestation, relative to private lands**					
			Private	Protected area	Sustainable use	Indigenous	Quilombola	Communal	Undesignated/ untitled public	Protected area	Sustainable use	Indigenous	Quilombola	Communal
	fulfill these duties and thus effectively exclude intruders should increase with the number of people available for these tasks (e.g., higher for quilombola communities than for individual private tenants).													
Cross-cutting themes														
Number of resource users and/or decision-makers	Decision making regarding the use and conversion of forests have higher transaction costs in community-based tenure forms because it takes more time and resources to reach decisions with larger numbers of people (Naidu, 2009; Ostrom, 2009). Individuals or small groups, in turn, have lower transaction costs involved in this decision-making process, meaning that they are more agile in responding to economic pressures or incentives to allocate the land to its most profitable use (which in many contexts implies converting forest to cropland or cattle ranching). Thus, tenure regimes with higher numbers of resource decision-makers are expected to decrease deforestation compared to those with lower numbers of decision-makers	Naidu 2009; Ostrom, 2009	↗	→	→	↘	↘	↘	↘	↘	↘	↘	↘	↘
Number of resource users and/or	Tenure regimes where ownership is shared among larger numbers of people are better equipped to monitor and protect their land, decreasing the	Sakurai et al., 2004; Ostrom 2009	↗	→	→	↘	↘	↘	↘	↘	↘	↘	↘	↘

Thematic dimension	Hypothesized mechanism	References	Predicted effect of tenure regime on deforestation, relative to undesignated/untitled public lands						Predicted effect of tenure regime on deforestation, relative to private lands					
			Private	Protected area	Sustainable use	Indigenous	Quilombola	Communal	Undesignated/ untitled public	Protected area	Sustainable use	Indigenous	Quilombola	Communal
decision-makers	likelihood of deforestation as compared to properties with fewer people (Ostrom, 2009; Otsuka et al., 2004). Thus, tenure regimes with higher numbers of owners, resources users, or decision-makers are expected to decrease deforestation compared to tenure forms with fewer numbers.													
Number of resource users and/or decision-makers	Tenure regimes with higher numbers of individual users are expected to be more likely to unsustainably exploit forest resources for individual short-term gain and thereby cause the collapse of the resource system than tenure forms with few or one user(s)(Browder et al., 1997; Gordon, 1954; Klingler & Mack, 2020).	Gordon, 1954; Browder et al., 1997; Klingler and Mack, 2020	↘	→	→	↗	↗	↗	↗	↗	↗	↗	↗	↗
Tenure security	Low levels of tenure security are commonly viewed as inhibiting tenants' engagement with their land resources (e.g., investment) due to elevated risk that all or some tenure rights may be cut short before they see the benefits of their investment (S. Holden & Yohannes, 2002). Higher levels of tenure security are thus classically expected to incentivize users to more readily “invest” in increasing the profitability of the land resource. In most tropical forestland contexts, this hypothesis would predict these to be investments into allocating the land to a more profitable use (e.g., through a conversion of	Holden and Yohannes, 2002; Angelsen, 1999; Fearnside, 2005; Deininger and Jin, 2006; Fenske, 2011; Robinson et al., 2004	↗ ↘			↘	↘	↘	↗ ↘			↘	↘	↗ ↘

Thematic dimension	Hypothesized mechanism	References	Predicted effect of tenure regime on deforestation, relative to undesignated/untitled public lands						Predicted effect of tenure regime on deforestation, relative to private lands					
			Private	Protected area	Sustainable use	Indigenous	Quilombola	Communal	Undesignated/ untitled public	Protected area	Sustainable use	Indigenous	Quilombola	Communal
	forest to cropland or cattle ranching), but these may also be investments into, e.g., restoring a degraded land resource. By contrast, lower levels of tenure security may also be expected to increase deforestation-causing activities if land clearing is used to solidify claims on the land (Angelsen, 1999; Fearnside, 2005; Deininger & Jin, 2006; Fenske, 2011). While private land tenure is classically viewed as providing the highest tenure security and thus assurance levels, this view is not universal (Robinson et al., 2014). Assuming that classical views on tenure-form–tenure-security relationships broadly hold and that landholders are mainly economically/personal-survival motivated, this set of hypotheses would predicts a skewed u-shaped relationship between tenure security and deforestation rates, where deforestation is medium-high at very low tenure security levels (e.g., informal settlements on public lands), lowest at intermediate levels of tenure security (i.e. indigenous, quilombola, and communal lands), and highest under highest assurance levels (e.g. private tenure).														
Governance (monitoring and enforcement)	Tenure regimes where the state (i.e., citizenry) is the main or exclusive rights and duty holder, such as protected areas or other lands administered by public institutions, are expected to have lower	Grafton, 2000	↗	→	→	→	↗	→	↘	↘	↘	↘	→	↘

Thematic dimension	**Hypothesized mechanism**	**References**	**Predicted effect of tenure regime on deforestation, relative to undesignated/untitled public lands**						**Predicted effect of tenure regime on deforestation, relative to private lands**					
			Private	Protected area	Sustainable use	Indigenous	Quilombola	Communal	Undesignated/ untitled public	Protected area	Sustainable use	Indigenous	Quilombola	Communal
	deforestation rates than other tenure regimes because the state is more likely to benefit from economies of scale for monitoring, enforcing, processing of information, and other management-related activities that prevent deforestation (Grafton, 2000).													
Governance (monitoring and enforcement)	Tenure forms where a single entity is the main rights holder (i.e., private tenure) provide better opportunities for state or federal agencies to enforce environmental legislation than tenure forms where the main rights holder is a community, unknown, or abstract (e.g., citizenry) because this increases accountability in adhering to targeted environmental legislation meant to decrease deforestation. Thus, tenure forms where single entities are the main rights holders are expected to decrease deforestation in comparison to those with multiple entities as rights holders (Arima et al., 2014; Hargrave & Kis-Katos, 2013).	Hargrave and Kis-Katos, 2013; Arima et al. 2014	↘						↗	↗	↗	↗	↗	↗
Governance (monitoring and enforcement)	In countries with a history by short-lived government institutions or volatile political directions, government programs proposing investments in the long-term sustainability of forest resources will lack credibility. Therefore, publicly owned forests will not be used sustainably, even if	Deacon, 1994	↘	→	→	→	↘	→	↗	↗	↗	↗	→	

Thematic dimension	Hypothesized mechanism	References	Predicted effect of tenure regime on deforestation, relative to undesignated/untitled public lands						Predicted effect of tenure regime on deforestation, relative to private lands					
			Private	Protected area	Sustainable use	Indigenous	Quilombola	Communal	Undesignated/ untitled public	Protected area	Sustainable use	Indigenous	Quilombola	Communal
	these are under partial private or community-based management (Deacon, 1994a).													
Governance	Public institutions in countries with poorly developed governance systems and/or high levels of external debt are more likely to sell or lease rights to exploit national resources (e.g., forestlands) at abnormally low prices. This increases the likelihood of inefficient, resource-intensive land-use forms (e.g. agricultural expansion rather than intensification). In such contexts, resource users are also more likely to overexploit resources (whether sold or leased) beyond the legal limit allowed because the perceived likelihood of enforcement is low (Baland & Platteau, 2000). Thus, under precarious governance contexts, all publicly owned forestland is expected to be more likely to experience deforestation.	Baland and Platteau, 2000	↘	→	→	→	↘	→	↗	↗	↗	↗	→	

With different fields focusing on different aspects of tenure, **Table 5** demonstrates the many hypothesized mechanisms that predict effects of different regimes on deforestation, and their range of expected effect directions and magnitudes. This includes hypotheses that predict opposite effects across tenure regimes even when considering the same mechanism (e.g. see hypotheses on number of resource users), as well as tenure regimes where theories have high levels of consensus or disagreement on the direction of expected effects. Aside from the breadth of literature and the range of expected effects that were assessed, this analysis evidences a few important patterns regarding the link between certain tenure regimes and expected deforestation rates.

To begin, the number of arrows pointing in different directions highlight how theoretical expectations focusing on certain tenure comparisons lack consensus to date. Of evaluated hypotheses, the comparison of private lands relative to undesignated/untitled public lands (and vice versa) had the highest number of arrows pointing in different directions (11 arrows pointing towards a decrease in deforestation, 9 to an increase). While this is not a quantitative evaluation of all possible hypotheses of expected effects from this specific comparison, these arrows demonstrate that the idea of privatizing unprotected public lands is highly contested – either for conservation purposes, or otherwise. These hypotheses do describe certain conditions under which the privatization of unprotected public lands might decrease deforestation rates. For instance, deforestation is predicted to be more likely under conditions where exclusion of resource users is not possible, or under conditions with precarious land governance, specifically when there is a lack of effective monitoring and enforcement. Under these conditions, private lands are predicted to be an effective tool in decreasing deforestation, as otherwise unprotected public lands may suffer from the lack of individual accountability for enforcement. Natural resource use in undesignated/untitled public lands may also be particularly vulnerable to volatile, short-lived governments, where the use of natural resources is more likely to be exploitative and unsustainable given the short time-horizon of investment of these regimes.

Another pattern found in this analysis was the differing predictions regarding communal regimes compared to both private and undesignated/untitled lands. Relative to private regimes, 10 arrows predict a decrease in deforestation, and 5 an increase; relative to undesignated/untitled 11 arrows predict a decrease, and 5 an increase. In this case, although there are fewer arrows that predict an increase in deforestation, these mostly correspond to dominant historical hypotheses regarding the inability of communal regimes to sustainably manage natural resources due to their lack of exclusion rights. However, most of the hypothesized mechanisms that predict a decrease in deforestation from communal regimes challenge this underpinning assumption, focus on other aspects of communal regimes, and qualify the conditions for successful management of natural resources in communal regimes (as overviewed in the Introduction). Here, hypothesized mechanisms that predicted a decrease in deforestation from all communal regimes (i.e. indigenous, quilombola, and communal lands) relative to both private and undesignated/untitled lands pertained to conditions that may be either determined by their governance context or inherently determined by the regimes themselves.

First, all of these communally-managed regimes do not have any alienation rights in Brazil, meaning that many land-market-related mechanisms that imply the land gets put to its most financially-productive use (e.g. using land as collateral, or selling to another user with more agility to transform the land) do not apply to these regimes, and deforestation is expected to decrease. Second, the withdrawal rights granted for these regimes in Brazil are all non-commercial, limiting the extent and scale to which agricultural activities such as cropping or cattle ranching may be conducted, and thereby also decreasing the likelihood of forest conversion to agriculture[3]. For both of these mechanisms the loss of forest/natural cover is not expected given the restrictions and regulations to their property rights relative to both private and undesignated/untitled lands. Third, the societal rules that often constitute the internal governance of these regimes, alongside the higher number of resource users and decision-makers decrease the likelihood of deforestation. These conditions, i.e. the incentive to belong to a group, the need to invest in the long-term sustainability of a communal resource, as well as an increased ability to monitor a resource as a larger group, pertain to characteristics inherent to these tenure regimes, rather than how they are legally regulated.

Notably, hypothesized mechanisms in the literature often do not consider the specific arrangement of rights that may observed in reality. In the case of Brazil, Quilombola lands are communally held, yet are privately owned – indicating a particular set of property rights that is distinct from indigenous and communal regimes (**Table 2**), and may often be overlooked in the literature. It is thus also difficult to qualify the general expectations of lands that are communally-held, yet have legally ambiguous property rights (see section on Communal lands in **2.2.1**). Thus, expected deforestation effects of communal regimes in this study, i.e. indigenous, quilombola, and communal lands, are not clear. This analysis might suggest that empirical effects of these regimes might be best be interpreted in light of those conditions that are both determined by broader governance contexts (i.e. spatial or temporal dynamics), as well as those that are inherent to these regimes (spatiotemporal consistencies).

Finally, it is important to note that there was general consensus found for expected effects protected areas and sustainable-use areas. These regimes are generally expected to decrease deforestation relative to both private and undesignated/untitled lands. In rare cases, a few hypotheses predict that protected regimes will increase deforestation compared to private, specifically under conditions of precarious land governance where there is little monitoring and enforcement. Here, protected regimes are expected to be similarly vulnerable to these conditions as unprotected public regimes are, and thus, deforestation might increase relative to private lands.

In sum, different theories predict effects of tenure regimes on deforestation in different directions, and magnitudes, with both consistent and inconsistent patterns. This analysis not only confirms the range of expected effects, but it also adds to the historical overview of these ideas provided in the **1.2** by examining expectations of different regimes across hypotheses. Additionally, the number of different mechanisms describe the complex link between tenure and forests – as well as broader environmental change. As this link is mediated by many factors and/or conditions, empirically measuring the direct causal impact of each of individual mechanisms would be largely unfeasible. However, measuring effects of different tenure regimes may likely capture key distinguishing factors between hypotheses, thus indicating empirical support for certain mechanisms over others. Altogether, this effort aids in the contextualization of subsequent

[3] Note notwithstanding hypothesis by Perrings 1989 actually predicts opposite deforestation effects from the same mechanism relating to non-commercial (subsistence) withdrawal rights

empirical findings to the broader literature, which contains many contested relationships that have yet to be tested at large spatiotemporal scales.

3.3 Empirical effects of different land tenure regimes on deforestation in Brazil

I found that in Brazil, between 1985-2018, 17.4% of land was converted from forest to agriculture (**Fig. 1A**). Forest cover included plantations, savannas and mangrove tree cover, and agriculture included the pasture, agriculture, annual perennial and semi-perennial crops, as well as mosaic and agriculture and pasture categories (*Project MapBiomas - Collection 4.0 of Brazilian Land Cover & Use Map Series*, n.d.). The vast majority of this deforestation (78%) occurred on private lands, followed by undesignated/untitled lands (19%; **Fig 1B**).

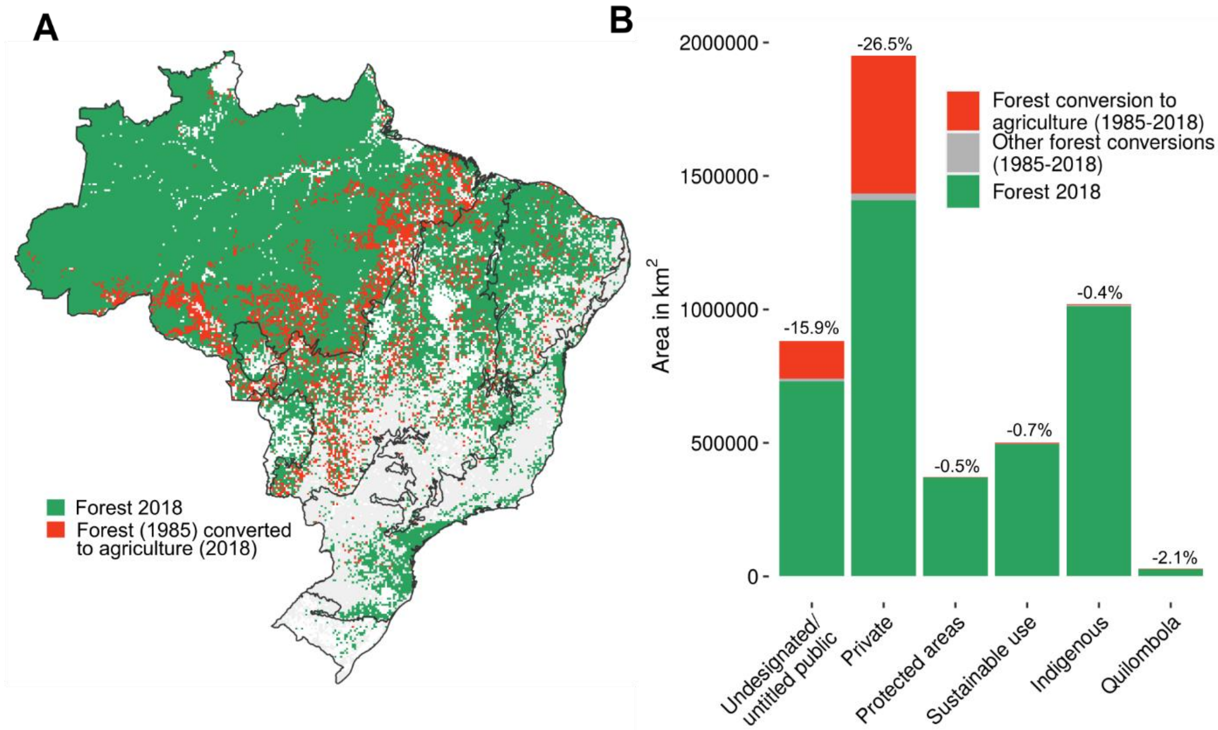

Figure 8. Forest conversion to agriculture (1985-2018). A) shows all forest cover (including plantation, savanna, and mangrove tree cover) converted to farming (pasture, agriculture, annual perennial, and semi-perennial crops, including mosaic of agriculture and pasture) (*Project MapBiomas - Collection 4.0 of Brazilian Land Cover & Use Map Series*, n.d.). B) shows total areas of forest that were converted to agriculture (red) or other land uses (grey) between 1985 and 2018, and remaining forest cover in 2018 (green), across all Brazil-wide parcels under each tenure regime. Percentages of total original (1985) forest-cover per tenure regime that were converted to agriculture by 2018 are indicated above each bar.

In order to empirically estimate the effects of different land tenure regimes on agriculture-driven deforestation in Brazil, I first conducted a matching analysis. Following my quasi-experimental study design, I use coarsened-exact matching (CEM) to pair observations from different tenure regimes (i.e. "treatments") to two counterfactual, or comparison groups (i.e. "quasi-controls"). For the first comparison group, I merged both undesignated public lands and untitled rural settlements into one category, undesignated/untitled public lands (hereafter "undesignated/untitled"). This merged category captures all unprotected public lands with poorly defined property rights, i.e. lands with no official designation, or, if harboring rural settlers, with no formally recognized exclusion, alienation, or due process property rights (see **Table 2**). For the second comparison group, I used private lands. These two comparison groups effectively establish two credible counterfactuals to mimic random treatment assignment.

After matching, matched data subsets should have improved levels of balance, meaning that covariates of both treatment and control groups are more evenly distributed than in the raw data. Overall, across all tenure-regime-specific tests at each spatiotemporal scale, CEM improved balance by 5-79% (0-73% for time-filtered tests). In some cases, matched datasets were too small to conduct further statistical analyses (communal lands outside of Amazonia, any tenure-regime comparison other than undesignated/untitled and private in Pampa and Pantanal). However, in large part due to the vast amount of land parcels in the Imaflora dataset, I was able to obtain sizeable matched datasets for the majority of tenure-regime comparisons at all spatiotemporal scales considered. The number of specific matches (*n)* are reported alongside model outputs (see **Tables 12-13**). Though post-matching datasets achieved between 24% and 90% balance in covariate values, to make cases of high remaining imbalance easily recognizable, imbalance was visualized as transparency gradients in all plots of estimated effects (**Figs. 9-10, 19-21**).

I first tested the predominant hypotheses regarding the effects of undesignated/untitled lands on deforestation, which are mostly expected to increase deforestation relative to most tenure regimes, though there are mixed expectations as they relate to private lands (**Table 5**). Undesignated/untitled lands are publicly owned lands with poorly defined tenure rights that are not yet designated to any use, but may be inhabited by rural settlers without a formally recognized land claim or title. These kinds of regimes cover vast areas across the tropics, and in Brazil alone account for almost one hundred million hectares (963,357 km²;(Sparovek et al., 2019), an area larger than Tanzania, most of which are located in the Amazonian biome (**Fig. 8**). I found that undesignated/untitled regimes increased Brazil-wide deforestation between 1985 and 2018 by ~13.3-23.6%, on average, relative to all other tenure regimes (**Fig. 9**, large circles). These tests were repeated for 49 different combinations of narrower spatiotemporal extents in order to assess the consistency of these findings across different contexts in the Brazil. Altogether, these tests consistently showed higher deforestation under undesignated/untitled compared to the respective other tenure regimes in 140 out of 197 cases (lower deforestation in 5 cases, non-significant in 52, **Table 7**).

These effects of undesignated/untitled regimes were qualitatively robust both to weighting all cases by balance levels of their respective datasets post-matching, and to filtering out protected and sustainable-use areas that were only officially established after the beginning of the respective time period or had unknown establishment dates (see **Tables 6, 7, 14**). Overall, these results provide strong evidence that across vastly different contexts, the lack of well-defined tenure rights on public lands causes increased agriculture-driven deforestation.

Given hypotheses in the literature have mixed expectations for the effects of undesignated/untitled regimes vis-à-vis private lands in particular (**Table 5**), I also examined the consistency of these effects in further detail. I found that although private lands caused a 13.3% average reduction in deforested area compared to the matched parcels under undesignated/untitled tenure across Brazil over the period 1985-2018, these deforestation-reducing effects were not general across narrower regional-historical contexts. In some regions (e.g. Cerrado and Caatinga biomes), there is often no significant effect found between private and undesignated/untitled regimes, indicating similar levels of deforestation for both. Furthermore, at narrower scales, net effects of private tenure were deforestation-decreasing in only 64.6% of cases (65.4% if balance-weighted, deforestation-increasing: 8.3%/8.1% if weighted, non-significant: 27.1%/26.6%; **Table 7**). This may indicate that the environmental benefits of tenure interventions promoting private rights over undesignated/untitled lands may more often (yet, not reliably) outweigh the risks than vice versa.

Hypotheses on the effects of private regimes often expect these to increase deforestation relative to other tenure regimes (excluding undesignated/untitled), yet private regimes might also be expected to decrease deforestation under certain governance conditions (**Table 5**). In this regard, when synthesizing effects (**Table 6**), I found that private tenure had the highest risk among all alternative regimes of increasing deforestation over the undesignated/untitled counterfactual (8.8% of scales considered; 8.4% if balance-weighted; **Table 6**, **Fig. 9A**). Private regimes were least likely to cause high deforestation reductions (2.9%; 11.8% if balance-weighted), and were second-most likely to cause the lowest reductions/highest increases (25.5%; 16.7% if balance-weighted; **Table 6**, **Fig. 9A**). Moreover, I found private regimes increased likelihood of deforestation as compared to most other most other tenure regimes. Excluding undesignated/untitled regimes, all alternative regimes decrease likelihood of deforestation compared to private lands by 8.9-9.74% for all Brazil during 1985-2018 (**Fig. 9B**). Overall, these results suggest that among all alternative tenure interventions that might reduce the deforestation associated with undesignated/untitled tenure by installing better-defined tenure rights, interventions leading to private tenure would be the least reliable and typically among the least effective options under vastly different socio-environmental settings.

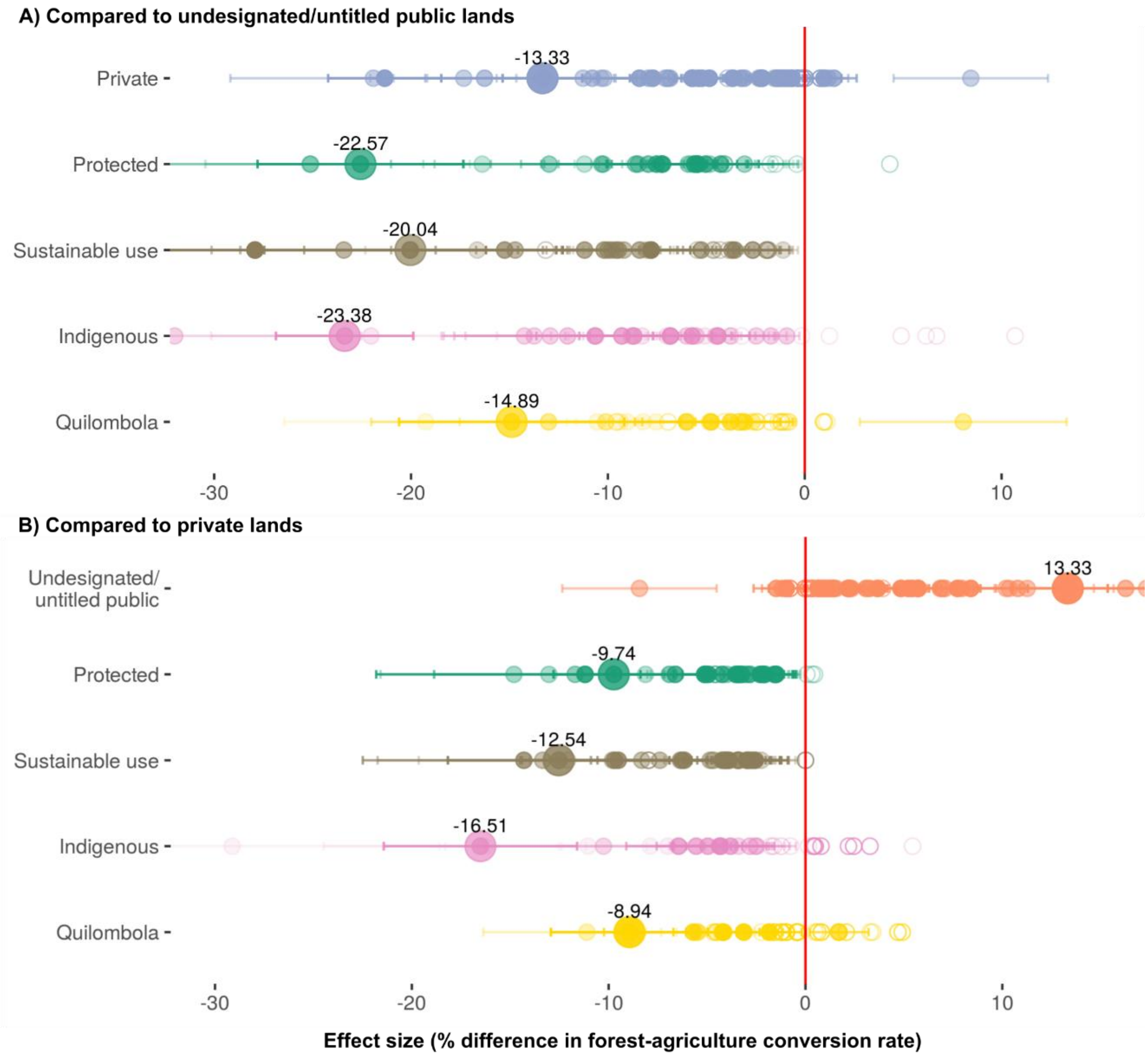

Figure 9. Effects of alternative land-tenure regimes on forest-to-agriculture conversion rates in Brazil. Circles indicate effects sizes estimated at different spatial-temporal scales, compared to two alternative counterfactuals: A) undesignated/untitled public lands with poorly defined tenure rights, and B) private lands. Labelled effect sizes (larger circles) report effects across Brazil over the time period 1985-2018. Effects to the left of the zero line indicate a decrease in average parcel-level deforestation rate (to the right: increase). Filled circles indicate statistically significant effects ($p < 0.05$; non-filled: not significant); upper/lower confidence intervals are plotted to the left/right of each circle centroid. Higher transparency of filled circles indicate high levels of imbalance in the matched dataset (multivariate imbalance measure L_1).

Hypotheses on the effects of both strictly protected areas and sustainable use protected areas have a high level of consensus in the literature, as these conservation-focused regimes are generally regarded to decrease deforestation against either counterfactual (except in contexts of poor governance where private regimes may be more effective in curbing deforestation) (**Table 5**). I found strict-protection and sustainable-use regimes had, respectively, the second- and third-strongest deforestation-reducing effects at large scales (**Figs. 9-10**). The two regimes also most consistently achieved at least some reduction in deforestation across the narrower regional-historical contexts (85.3% and 76.5% of cases with significant negative effects, respectively, **Table 6**). The above results were robust both to weighting by balance post-matching and to filtering later-established conservation areas (**Tables 7, 14**; and **Figs. 20-21** in **Appendix**). However, whereas sustainable-use regimes were three times more likely to outperform than to underperform alternative regimes in protecting forests (largest/smallest deforestation reductions in 32.4/8.8% of cases; 35.3/11.8% if balance-weighted; 47.4/11.8% if time-filtered), this was not the case for strict-protection regimes (29.4/14.7%; 23.5/20.6%; 10.5/7.9%; **Table 6**); note these differences are not driven by protected-area siting, see **Figure 11**). This indicates that while any conservation-focused regime may reduce deforestation more reliably than alternative regimes under very different contexts, specifically sustainable-use regimes may most reliably achieve large reductions.

In contrast to conservation-focused regimes, hypotheses related to communal regimes (i.e. indigenous, quilombola, and communal lands) expect mixed effects with regard to both counterfactuals; though some of these expectations hinge on meeting particular governance conditions (**Table 5**). I found that both indigenous and quilombola tenure regimes decreased deforestation against both counterfactuals at the scale of Brazil 1985-2018, however, these effects were not highly consistent throughout narrower scales. For both regimes together, the cross-scale analyses yielded no significant effects for nearly half of all cases (64 of 136, **Table 7**); yet, effects between these regimes are quite different. Against either counterfactual, tenure by specifically indigenous communities reduced deforestation more effectively than all other regimes across Brazil over 1985-2018 (**Figs. 9, 19**). Indigenous tenure also more often outperformed than underperformed other regimes in protecting forests at narrower scales (largest/smallest decreases in 17.7/10.8% of cases against undesignated/untitled; 39.5/3.5% against private; **Figure 9**, **Table 6**). By contrast, quilombola communities, self-identified descendants of Afro-Brazilian slaves who privately own their communal lands, reduced deforestation least reliably and often least effectively, notably lacking any deforestation-reducing effects in Caatinga – where most quilombola lands are situated (overall 47.1% significant reductions/lowest reductions or highest increases in 40.2% of cases over untitled-undesignated, 30.0/10.5% over private; **Figure 10**, **Table 6**). These ambiguous results on the effects of community-based tenure regimes are in line with diverging theoretical arguments (**Table 5**). Overall, the evidence provided by these tests suggests that synergies between IPLC tenure and forest conservation objectives arise often, but not reliably across different contexts.

In using a cross-contextual synthesis approach, I am able to identify consistent patterns of effects across diverse social-environmental settings; but also, unique patterns at particular scales (**Figure 10**). Notably, I found important divergences from the overall effects in Brazil in Amazonia, where 90.5% of Brazil's remaining undesignated/untitled forest is situated (**Figure 8**). Here, all three public reserve regimes (strict-protection, sustainable-use, and indigenous) had consistently weaker deforestation-reducing effects against undesignated/untitled regimes than quilombola tenure (**Figure 10**). Even more surprisingly, private lands changed from deforestation-increasing relative to undesignated/untitled in 1985-1990 to being the second-most (after quilombola) or most strongly deforestation-decreasing regime from the early 2000s (**Figure 10**). Both results were robust to balance-weighting and not confounded by systematic differences in initial forest cover (**Figure 11**).

These counter-intuitive effects in Amazonia might indicate support for theories that predict that, under certain governance conditions, private tenure regimes may decrease deforestation vis-à-vis alternatives. However, it is important to note that empirical testing of these practical alternatives indicates that directly replacing any public reserve regimes (strict-protection, sustainable-use, and indigenous) with private lands would have likely increased deforestation in Amazonia (87.7% of all tested time-periods, 88.8% after 2000, **Figure 10**). This apparent paradox could indicate that privatization may only effectively address the specific deforestation mechanisms acting on undesignated/untitled public lands in the Amazon – but not those on protected, sustainable-use areas, or indigenous lands.

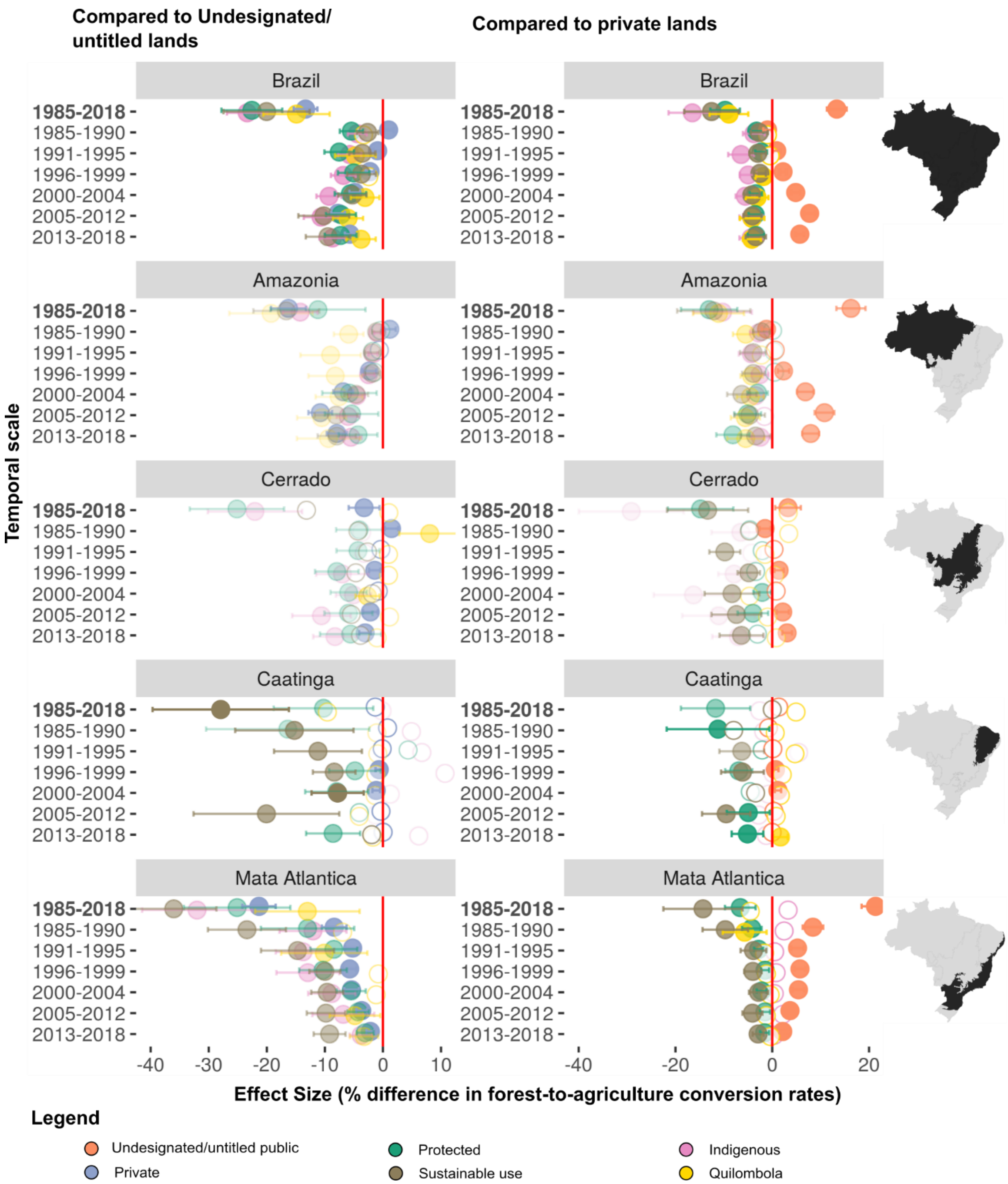

Figure 10. Effects of alternative land-tenure regimes on forest-to-agriculture conversion rates in Brazil, disaggregated to different spatiotemporal scales. Circles indicate effects sizes estimated at the respective scale vis-a-vis two alternative counterfactuals: a) undesignated/untitled public lands, and b) private lands. Effects to the left of the zero line indicate a decrease in average parcel-level deforestation rate (to the right: increase). Filled circles indicate statistically significant effects ($p \leq 0.05$; non-filled: not significant), upper/lower confidence intervals are plotted to the left/right of each circle centroid. Higher transparency of filled circles indicates high levels of imbalance in the matched dataset (multivariate imbalance measure L_1).

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Table 6. Synthesis of the directions and relative magnitudes of effects of different land-tenure regimes across spatiotemporal scales. For this cross-scale synthesis, I consider only scales at which deforestation effects of all five alternative tenure regimes were consistently testable vis-à-vis the respective counterfactual. The left section on Direction of estimated effects on deforestation reports the numbers and percentages of scale-specific model estimates finding likelihood of deforestation of alternative tenure regimes vis-à-vis the counterfactual increases (↗), decreases (↘), or is non-significant (non-sig.). The right section on Ranking by relative magnitude of effect size, reports the percentages of all compared spatiotemporal scales where each regime ranked as more deforestation-decreasing ("best") and less deforestation-decreasing/more increasing ("worst") than all alternatives regimes (based on their respective effect sizes). In this ranking, effects that were statistically indistinguishable from 0 were placed in between deforestation-decreasing and -increasing (e.g. private land tenure reduced deforestation vis-à-vis an undesignated/untitled public regime more effectively (larger negative effect size) than all alternative regimes at 2.94% of the compared spatiotemporal scales, while decreasing deforestation least effectively or most strongly increasing deforestation at 25.49% of scales). All percentages are also reported as weighted by the level of balance (L_1) in the underlying dataset ("w. by balance"), downweighing cases where datasets still had low levels of overlap in covariate values post-matching. Note that in order to keep comparisons consistently comparable across spatiotemporal scales, this table does not include results for Pampa and Pantanal, nor for communal lands. Also note that these percentages synthesize "narrower scales" only. For Brazil-wide results for the full 1985-2018 period, see **Fig. 9**.

	Direction of estimated effects on deforestation													**Ranking by relative magnitude of effect size**						
	↗ (count)	↗ (count w. by balance)	% ↗	% ↗(w. by balance)	↘(count)	↘ (count w. by balance)	% ↘	% ↘(w. by balance)	non-sig. (count)	non-sig. (w. by balance)	% non-sig.	% non-sig. (w. by balance)	Total models	best	best (w. by balance)	worst	worst (w. by balance)	non-sig.	non-sig. (w. by balance)	Total models
Compared to undesignated/untitled lands																				
Private lands	3	2.27	8.8%	8.4%	23	18.34	67.7%	67.8%	8	6.44	23.5%	23.8%	34	2.94%	2.22%	25.49%	27.49%	8	2.66	34
Protected areas	0	0.00	0.0%	0.0%	29	14.12	85.3%	87.1%	5	2.09	14.7%	12.9%	34	29.41%	28.52%	14.71%	11.11%	5	1.59	34
Sustainable use areas	0	0.00	0.0%	0.0%	26	15.09	76.5%	79.1%	8	3.99	23.5%	20.9%	34	32.35%	33.38%	8.82%	7.35%	8	2.71	34
Indigenous lands	0	0.00	0.0%	0.0%	25	12.67	73.5%	80.9%	9	2.99	26.5%	19.1%	34	17.65%	22.54%	10.78%	8.62%	9	2.99	34
Quilombola lands	1	0.49	2.9%	3.0%	16	7.30	47.1%	45.3%	17	8.34	50.0%	51.7%	34	17.65%	13.34%	40.20%	45.43%	17	6.82	34
All of the above vs. undesignated/untitled	4	2.75	2.4%	2.9%	119	67.51	70.0%	71.7%	47	23.85	27.7%	25.3%	170							
Compared to private lands																				
Public lands	22	17.47	71.0%	71.5%	3	2.27	9.7%	9.3%	6	4.71	19.4%	19.3%	31	0.00%	0.00%	77.15%	81.49%	6	1.94	31
Protected areas	0	0.00	0.0%	0.0%	22	13.71	71.0%	77.2%	9	4.06	29.0%	22.9%	31	13.71%	10.90%	6.99%	5.21%	9	2.80	31
Sustainable use areas	0	0.00	0.0%	0.0%	28	16.69	90.3%	89.4%	3	1.98	9.7%	10.6%	31	36.29%	43.82%	1.88%	1.21%	3	0.94	31
Indigenous lands	0	0.00	0.0%	0.0%	17	7.21	54.8%	49.0%	14	7.52	45.2%	51.1%	31	39.52%	36.65%	3.49%	2.53%	14	7.16	31
Quilombola lands	0	0.00	0.0%	0.0%	10	5.54	32.3%	30.0%	21	12.94	67.7%	70.0%	31	10.48%	8.63%	10.48%	9.56%	21	9.26	31
All of the above vs. private	22	17.47	14.2%	18.6%	80	45.41	51.6%	48.3%	53	31.20	34.2%	33.2%	155							

Table 7. Synthesized direction of effects of all assessed land-tenure regimes on deforestation across all assessed scales (see **Table 6** for general description). Unlike results in **Table 6**, which consider only tenure regimes and scales for which consistent comparisons were possible, results here are based on all "narrower" scales where a given land-tenure regime could be compared against the respective counterfactual (i.e., excl. results for Brazil for the 1985-2018 period, but also incl., e.g., private-vs-undesignated/untitled comparisons for Pampa and Pantanal). These results are thus more comprehensive (based on more scales) than those in **Table 6** if single tenure regimes are viewed in isolation. However, unlike **Table 6**, they are not comparable across tenure regimes as they are based on inconsistent combinations of scales. Information that is redundant with that in **Table 6** (based on the same scales) is shaded in grey.

	Direction of estimated effects on deforestation												
	↗ (count)	↗ (count w. by balance)	% ↗	% ↗(w. by balance)	↘(count)	↘ (count w. by balance)	% ↘	% ↘(w. by balance)	non-sig. (count)	non-sig. (w. by balance)	% non-sig.	% non-sig. (w. by balance)	Total models
Compared to undesignated/untitled lands													
Private lands	4	2.81	8.33%	8.07%	31	22.72	64.58%	65.35%	13	9.24	27.08%	26.58%	48
Protected areas	0	0.00	0.00%	0.00%	29	14.12	85.29%	87.11%	5	2.09	14.71%	12.89%	34
Sustainable use areas	0	0.00	0.00%	0.00%	26	15.09	76.47%	79.07%	8	3.99	23.53%	20.93%	34
Indigenous lands	0	0.00	0.00%	0.00%	25	12.67	73.53%	80.91%	9	2.99	26.47%	19.09%	34
Quilombola lands	1	0.49	2.94%	3.02%	16	7.30	47.06%	45.27%	17	8.34	50.00%	51.70%	34
Communal lands	0	0.00	0.00%	0.00%	13	8.53	100.00%	100.00%	0	0.00	0.00%	0.00%	13
All of the above compared to undesignated/untitled	5	3.29	2.54%	2.98%	140	80.42	71.07%	72.87%	52	26.65	26.40%	24.15%	197
Robustness check: protected areas and sustainable use areas filtered by known year of creation													
Protected areas	0	0	0.00%	0.00%	22	0.07	88.00%	100.00%	3	0.00	12.00%	0.00%	25
Sustainable use areas	0	0	0.00%	0.00%	14	0.05	73.68%	27.93%	5	0.13	26.32%	72.07%	19
All of the above compared to undesignated/untitled	5	3.29	2.89%	4.37%	121	51.34	69.94%	68.15%	47	20.70	27.17%	27.47%	173
Compared to private lands													
Public lands	31	22.72	64.58%	65.35%	4	2.81	8.33%	8.07%	13	9.24	27.08%	26.58%	48
Protected areas	0	0.00	0.00%	0.00%	24	15.06	72.73%	78.76%	9	4.06	27.27%	21.24%	33
Sustainable use areas	0	0.00	0.00%	0.00%	29	17.45	90.63%	89.81%	3	1.98	9.38%	10.19%	32
Indigenous lands	0	0.00	0.00%	0.00%	18	7.86	52.94%	48.76%	16	8.26	47.06%	51.24%	34
Quilombola lands	1	0.69	2.94%	3.35%	11	6.39	32.35%	30.86%	22	13.61	64.71%	65.79%	34

Direction of estimated effects on deforestation													
	↗ (count)	↗ (count w. by balance)	% ↗	% ↗ (w. by balance)	↘ (count)	↘ (count w. by balance)	% ↘	% ↘ (w. by balance)	non-sig. (count)	non-sig. (w. by balance)	% non-sig.	% non-sig. (w. by balance)	Total models
Communal lands	0	0.00	0.00%	0.00%	10	3.84	76.92%	77.50%	3	1.11	23.08%	22.50%	13
All of the above compared to private	32	23.41	16.49%	20.35%	96	53.40	49.48%	46.40%	66	38.27	34.02%	33.26%	194
Robustness check: protected areas and sustainable use areas filtered by known year of creation													
Protected areas	0	0.00	0.00%	0.00%	20	0.96	83.33%	95.67%	4	0.04	16.67%	4.33%	24
Sustainable use areas	0	0.00	0.00%	0.00%	14	0.20	77.78%	67.00%	4	0.10	22.22%	33.00%	18
All of the above compared to private	32	23.41	18.71%	30.08%	77	22.04	45.03%	28.32%	62	32.37	36.26%	41.59%	171

3.3.1 Sensitivity analyses

As a sensitivity analysis, I calculated lower and upper bounds for both Hodges-Lehmann point estimates and p-values using the "*rbounds*" package in R. These calculations show that both Hodges-Lehman estimates and p-values were not highly sensitive to possible small omitted-variable bias ($\Gamma = 1.1$), and were still reasonably robust to possible large omitted-variable bias ($\Gamma = 1.5$). Across tenure-regime comparisons, spatial scales, and temporal scales, average sensitivities of estimated effects ranged from, respectively, 11.18%, 10.12% and 10.78% relative error at $\Gamma = 1.1$, to 48.72%, 44.48% and 46.92% at $\Gamma = 1.5$ (**Table 15**; relative error calculated as percentage of the magnitude of the respective median effect size at $\Gamma=1$). Average sensitivities of significance of effects ($p \leq 0.05$) ranged from, respectively, 2.7%, 4.2% and 3.2% of models with a sensitive effect significance at $\Gamma = 1.1$, to 17.3%, 15.6% and 18.11% at $\Gamma = 1.5$ (**Table 15**). No systematic patterns in sensitivity to possible omitted-variable bias across tenure-regime comparisons, regions, or time periods were found, except for results based on lower sample sizes (mainly comparisons involving quilombola tenure and those in the Caatinga biome), which were on average slightly more sensitive. This analysis implies that the magnitude of estimated differences in outcomes between treatment and control units, and their significance, is only slightly sensitive to the possibility of a missing confounder, if present. Note that this sensitivity test cannot indicate whether or not an unobserved-confounder bias is actually present.

The existence of any potential bias in these estimates due to differences in initial forest cover was also assessed because estimated effects of could have been biased by differences in initial forest cover between matched parcels that resulted from forest-to-agriculture conversions prior to the respective treatment periods. In particular, forest conversion rates on private lands might change with decreasing forest cover, as the Forest Code prohibits additional deforestation once forest cover decreases to a certain threshold (e.g. 80% in the Amazonia biome). Similarly, parcels in old deforestation frontiers might have already been past their deforestation peaks before the study period began, whereas those in newly emerging frontiers might not yet experience the magnitude of deforestation that is this yet to come.

To assess possible bias in these conclusions due to systematic differences in initial forest cover, I modelled the initially forest-covered percentages of the matched parcels' areas at each spatiotemporal scale as a function of their treatment (i.e., tenure-regime identity). To this end, I fitted GLMs with a binomial error distribution and a logit link to the respective matched datasets to estimate the per-pixel likelihood of being initially forest-covered. Beyond a dummy variable distinguishing treatment and control, all covariates from the main regression analyses were included to compare the same parcels that were also originally matched (see section **2.4.2** Regression analysis). No systematic unidirectional differences between treatment and control across scales were detected, indicating that the main conclusions of this analysis are not biased by such differences. However, there were differences in either direction of individual cases and thus it is not possible to completely rule out that these might partly explain differential forest trajectories for some tenure regimes and spatiotemporal scales. I address this caveat by basing the main conclusions on results that were consist across spatiotemporal scales and by ruling out this bias when drawing insights from scale-specific results (e.g., the changing relative effectiveness of tenure regimes in curbing Amazonian deforestation).

The motivation to use this approach (e.g., instead of directly matching parcels on initial forest cover) was because the aims of this dissertation are not to assess total forest losses of different tenure regimes over their entire lifetimes (which would require accounting for any prior deforestation already internalized in parcels' initial forest cover). Rather, the aim was to assess whether tenure regimes consistently differed in their ability to retain remaining forest cover over different time periods (defined by their unique historical deforestation trends, policies, etc.). Here, differences in the magnitude of additional percentage losses among the matched parcels are already internalized in the way percentages are modelled by binomial GLMs. Finally, parcel-level differences in initial forest cover do not necessarily reflect prior forest-to-agriculture conversions, but may also reflect natural spatiotemporal heterogeneity in land cover (e.g., due to mosaics of forest and non-forest vegetation, landslides, etc.) as well as earlier agricultural expansion over non-forest vegetation, particularly outside the Amazonia biome.

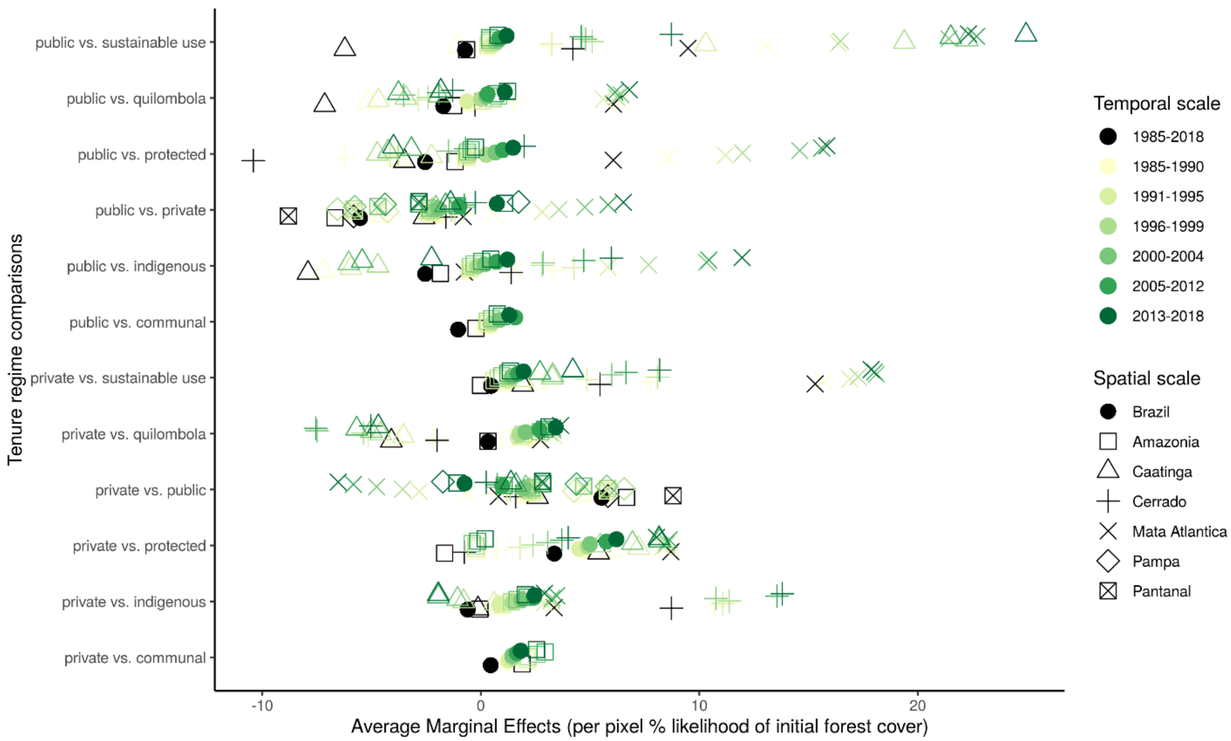

Figure 11. Differences in initial forest cover between matched treatment and control units for different tenure-regime comparisons at different spatial and temporal scales. Average marginal effects indicate the per-pixel likelihood being forest-covered at the beginning of each time period considered. At the parcel level, these can be interpreted as average deviation in initial percentage forest cover of the parcels treated with a given tenure regime relative to their matched counterfactual parcels. Temporal scales and spatial scales are indicated by color and shape, respectively, with broader scales (Brazil, 1985-2018) indicated in black. Symbols clustering closely around 0 and/or deviating from 0 in either direction indicate that the cross-scale synthesis results are unlikely biased by systematic differences in initial forest cover.

3.4 Empirical effects of different land tenure regimes on biodiversity in Brazil

After establishing effects of different tenure regimes on forest to agriculture conversion rates, a similar methodology was applied to answer the question of how different tenure regimes impact changes in biodiversity – specifically in this case changes in species' habitats. While it is clear that species diversity and distributions are affected by land-use changes (LUC) such as forest-to-agriculture conversions, overall impacts of these changes for species' habitats and their diversity remains unclear, as different species may be more/less sensitive to different LUCs.

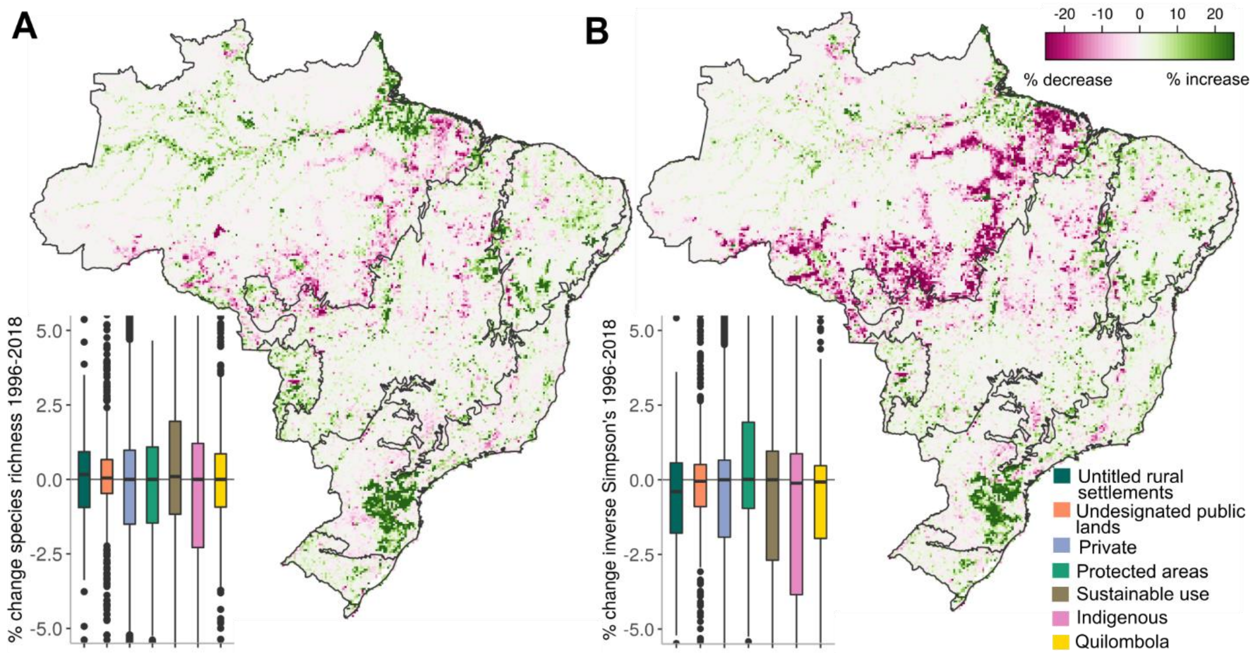

Figure 12. Changes in potential biodiversity (1996-2018) and spatial distribution of different land tenure regimes in Brazil. A) shows percent changes in potential species richness, with percent increases mapped in green, percent decreases in pink, and no change in white. Increases/decreases >= 25% are indicated in darkest shades. Boxplots disaggregate these percent changes per tenure regime. B) shows percent changes in the potential Simpson's diversity index, with increases/decreases and boxplots analogous to A). Potential species richness and Simpson's diversity index are calculated as detailed in **2.2.2**.

As to be expected, I found both potential species richness (count of all species with AoH in a given 1km^2 pixel) and Simpson's diversity index (calculated as detailed in **2.3.2**) had the highest values in the Amazon biome. Hence, unsurprisingly, highest biodiversity losses for both metrics (per-pixel percent decrease higher than 25%) during the period of 1996-2018 were concentrated in this biome **Fig. 12**), mainly following the well-known "deforestation arc" along the frontier of the Amazon and Cerrado biomes. Highest increases in both biodiversity metrics (per-pixel percent change increase higher than 25%) during the period of 1995-2018 were concentrated in the southern part of the Mata Atlantica (in federal states Paraná and Santa Catarina). Potential species richness, in particular, had high increases along the Amazon river basin.

Per-pixel percent increases/decreases of both biodiversity metrics occurred throughout the country, in all tenure regimes. On average, per-pixel percent increases/decreases of both metrics during 1996-2018 was concentrated at -2.5-2.5%, yet, maximum and minimum changes are well above

80% (decreases) and 600-1000% (increases in Simpson's and richness, respectively), demonstrating the drastic changes in some regions during this time period. Boxplots (**Fig. 12**) show the wide variation of the percent change of both richness and Simpson's index per tenure regime, with whiskers outside of the plotted area due to extreme maximum and minimum values.

Upon visual inspection, the distribution of both potential species richness and the inverse of Simpson's index per tenure regime during 1996, 2007, and 2018, did not suggest any obvious temporal trends (**Figs. 13-14**). However, differences between tenure regimes are easily observed; on average, communal, indigenous, and sustainable use regimes tended to have higher values in richness and Simpson's diversity index from 1996-2018 (median richness values ranging from 500-750 species, and median Simpson's index values ranging from 400-600) (**Figs. 13-14)**. Private lands, rural settlements, and undesignated lands tended to have lower values (median richness values ranging from 300-400 species, and median Simpson's index values ranging from 200-300) (**Figs. 13-14**). Yet, extremely high values were found for both metrics in all tenure regimes (richness above 750 species, and Simpson's above 800 index values). A high number of outliers were particularly notable for private lands and rural settlements, indicating that these regimes also had highly biodiverse properties despite having lower biodiversity on average.

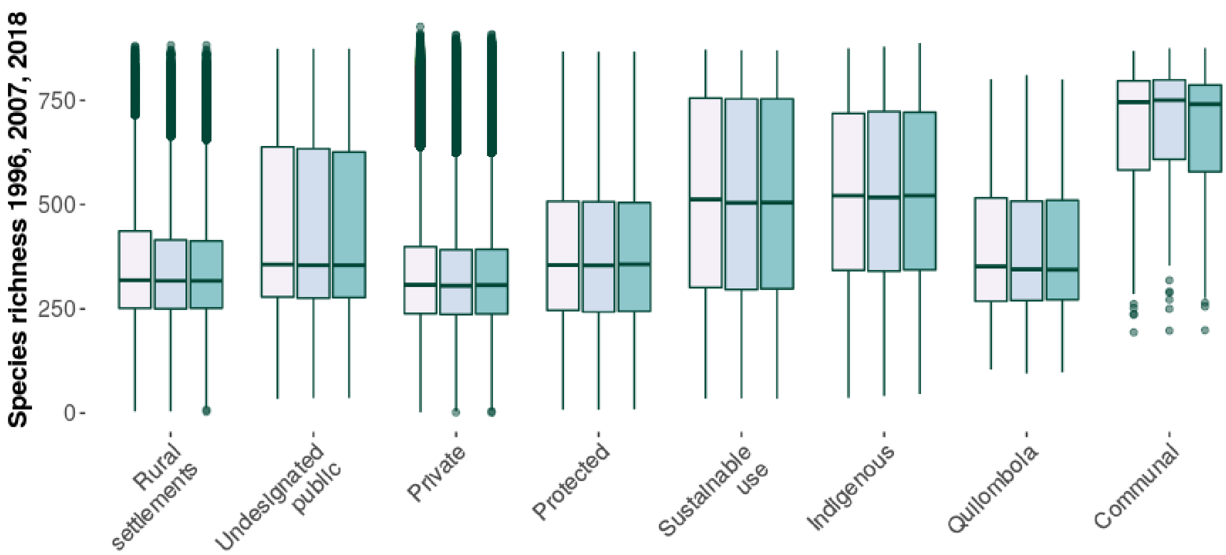

Figure 13. Boxplot of potential species richness values in 1996, 2007, 2018, per tenure regime. Using the per-pixel mean of each parcel, species richness is defined as the count of all species with habitat in a given pixel.

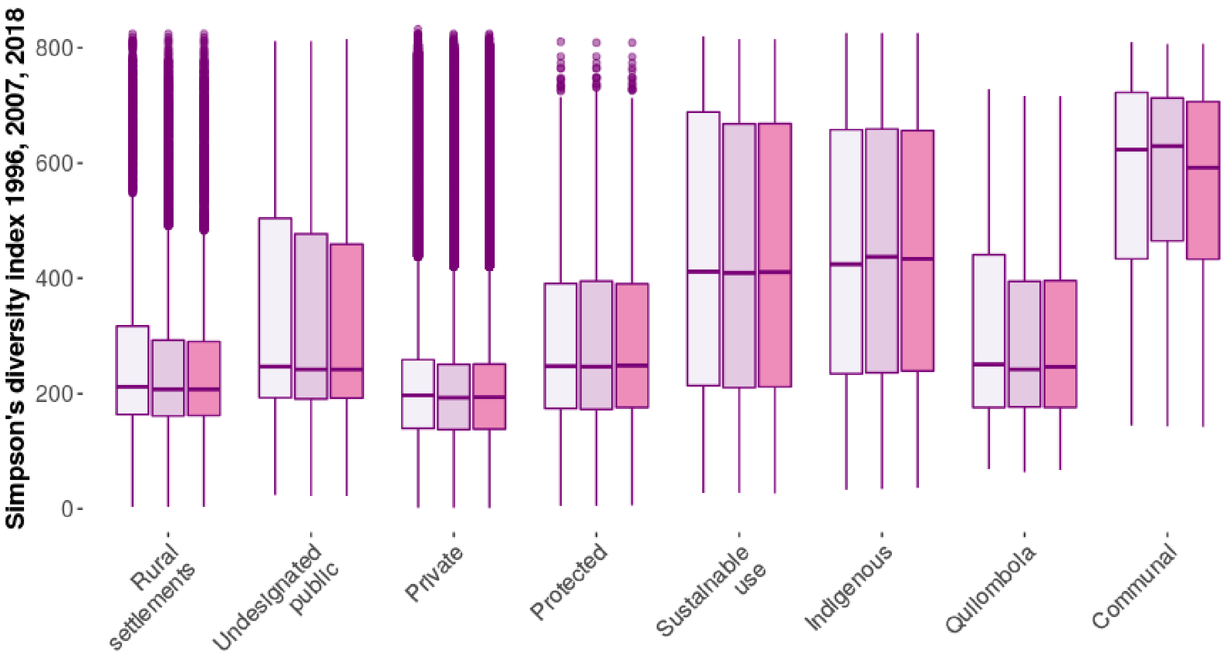

Figure 14. Boxplot of the potential inverse of Simpson's diversity index in 1996, 2007, 2018, per tenure regime. Using the per-pixel mean of each parcel, the inverse of Simpson's index is a measure of diversity that accounts for the evenness in habitat occupancy among co-occurring species in a pixel.

To estimate effects of different tenure regimes on biodiversity change as changes in habitat, I applied a similar quasi-experimental setup as used before on deforestation to potential species richness and the inverse of Simpson's diversity index. I analyzed effects on these biodiversity metrics at one large spatiotemporal scale (1996-2018), using a time fixed effects regression model (also known as a panel data regression model, see section **2.4.2**), and in contrast with the previous analysis on deforestation, I compared six tenure regimes (untitled rural settlements, undesignated lands, strictly protected and sustainable-use protected areas, indigenous, and quilombola lands) to one counterfactual, private tenure regimes.

Using CEM, I successfully matched 11-87% of the original quasi-treatment observations to a comparable private parcel (**Table 8**), where communal lands yielded the smallest subset of matched, trimmed observations, and rural settlements the largest. After matching, levels of balance highly improved, with L_1 measures of matched datasets varying from 40-54%, (compared to 78-97% pre-matching). Communal lands were the only case where balance remained high after matching (L_1 = 85%), and for this reason, they were excluded from subsequent statistical analyses, as these high levels of imbalance would likely produce biased model estimates and should not be further interpreted.

Table 8: Matching results from CEM. The original number of unmatched quasi-treatment observations (*n* unmatched dataset*)*, and unmatched quasi-control (in parentheses) have high levels of imbalance (L_1 before matching), with values ranging from 76-97% (fifth column). Matched datasets (*n* matched dataset) demonstrate improved balance measures (L_1 after matching), with values ranging from 40-85% (last column). Matched datasets are composed of exactly half observations as quasi-treatment, and half quasi-control. Percent of original treatment observations that were matched are reported (% treatment matched, third column), i.e. out of 765 communal parcels, 11.63% are successfully matched to a private parcel using CEM.

Tenure comparison relative to private lands	*n* unmatched dataset	*n* matched dataset	% treatment matched	L_1 before matching	L_1 after matching
Rural settlements	7522 (501606)	13074	86.91%	76.22%	41.18%
Undesignated lands	9861 (495996)	2876	14.58%	78.80%	48.33%
Protected areas	570 (494920)	518	45.44%	96.19%	53.28%
Sustainable use protected areas	698 (494778)	430	30.80%	95.17%	40.93%
Indigenous	599 (495029)	682	56.93%	96.15%	54.05%
Quilombola lands	362 (494749)	530	73.20%	94.55%	47.17%
Communal	765 (494548)	178	11.63%	97.46%	85.39%

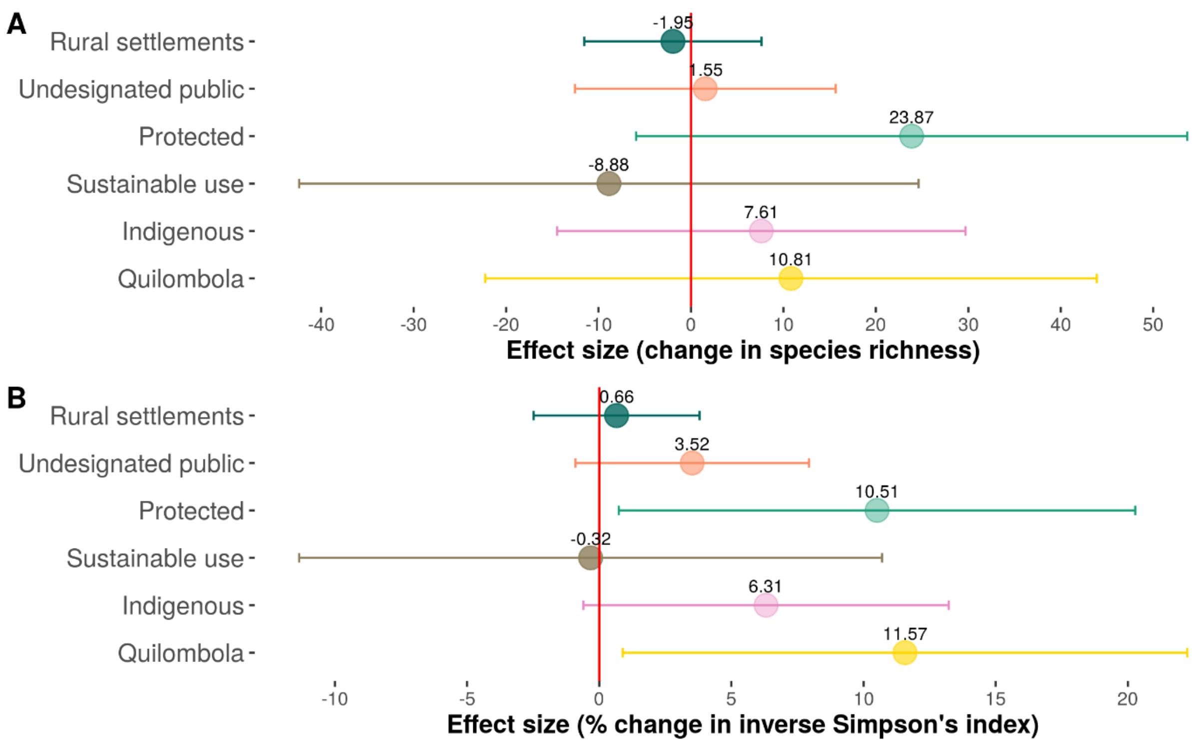

Figure 15. Effects of alternative tenure regimes on two biodiversity metrics in Brazil during 1996-2018, potential species richness (A), and potential Simpson's diversity index (B). Circles indicate effects sizes estimated compared to private lands. Effects to the left of the zero line indicate a decrease in either metric, to the right: increase. Upper/lower confidence intervals are plotted to the left/right of each circle centroid. Higher transparency of filled circles indicate high levels of imbalance in the matched dataset (multivariate imbalance measure L_1).

Similar to my approach used in the deforestation analysis, I first tested the hypotheses on effects of private tenure regimes on LUC and subsequent species habitat loss in comparison to rural settlements and undesignated lands. These hypotheses have mixed expectations; while many predict private lands should prevent habitat loss because well-defined property rights solve problems of exclusion and provide incentives for long-term resource sustainability, many others argue that private regimes provide the incentives for land-use conversions that result in species' habitat degradation and loss.

I found that, for both potential species richness and inverse Simpson's diversity index, effects of private regimes compared to both rural settlements and undesignated lands were not statistically significant in Brazil during 1996-2018, with average effects ranging from -1.95-1.55 species, and 0.66-3.52% (richness and Simpson's index, respectively, **Figure 15**). This indicates private regimes broadly cause very similar effects on potential biodiversity gains/losses as compared to rural settlements and undesignated lands. However, I take advantage of the longitudinal/panel data set of this analysis, and calculate estimated effects specifically at each year; 1996, 2007, and 2018 (**Figure 16**). Here, although estimated effects of private regimes relative to rural settlements remained nonsignificant for species richness during all time periods (**Figure 16A,** 95% CI ranging from –15-9.7 **Table 10**), in 2007 and 2018, undesignated lands increased Simpson's diversity index relative to private lands (**Figure 16B,** 95% CI ranging from 0.8-9.9%). In contrast to findings on deforestation, estimated effects of rural settlements on both biodiversity metrics seem to be very similar to the effects of private regimes. Furthermore, effects of undesignated lands seem to have increased Simpson's index relative to private regimes in later years (2007, 2018). These findings do not indicate support for hypotheses that predict private tenure regimes decrease habitat loss in comparison to undesignated lands or rural settlements. Instead, results suggest privatization of undesignated lands may have different effects on potential biodiversity than they do on forest-to-agriculture conversions.

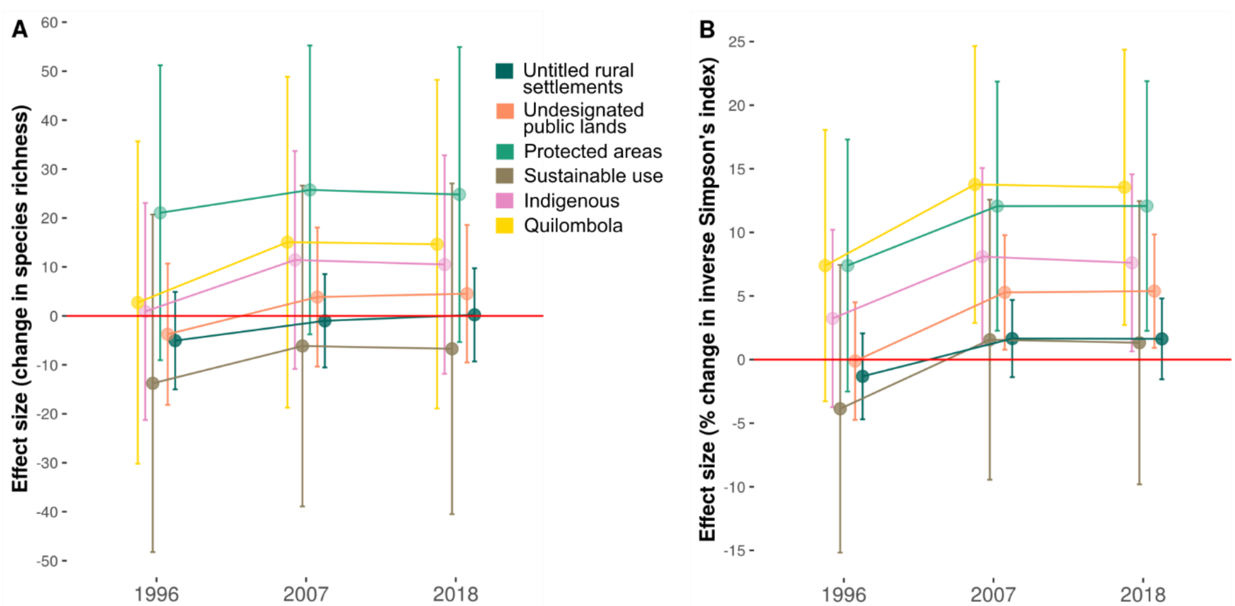

Figure 16. Effects of alternative tenure regimes on two biodiversity metrics in 1996, 2007, and 2018; potential species richness (A), and potential of Simpson's diversity index (B). Circles indicate effects sizes estimated compared to private lands. Effects above the zero line indicate an increase in either metric, below, a decrease. Upper/lower

confidence intervals are plotted above/below each circle centroid. Higher transparency of filled circles indicate high levels of imbalance in the matched dataset (multivariate imbalance measure L_1).

I next tested hypotheses that mostly expect private tenure regimes to decrease species' habitats relative to alternative tenure regimes (i.e. conservation or IPLC regimes), except under poor governance conditions where private regimes may be more effective in preventing loss. I found that most effects of private regimes against other alternatives on both species richness and Simpson's diversity index during 1996-2018 were not statistically significant (although generally positive, with average effects on richness ranging from -8.9-23.9, and on Simpson's index 0.3-11.6% **Figure 15**). Protected areas and quilombola lands were the exception, as they significantly increased Simpson's index (10.5 and 11.6%, respectively **Figure 15**). However, estimated effects calculated at 2007 and 2018 (**Figure 16**) show that most alternative regimes (besides rural settlements and undesignated lands) significantly increased Simpson's diversity index (with the exception of sustainable-use areas **Figure 16B**, **Table 10**). This is in contrast to nonsignificant effects found for species richness across all alternative tenure regimes. These results do not clearly support hypotheses predicting private regimes decrease species' habitats relative to alternatives, as average effects throughout 1996-2018 are nonsignificant. However, year-specific calculations of estimated effects do suggest that the privatization of alternative tenure regimes, in particular, protected areas, indigenous, and quilombola lands, decreased the diversity of species' potential AoH from 2007-2018.

Most hypotheses in the literature predict both protected and sustainable-use areas prevent habitat loss vis-à-vis private lands (with a few caveats under poor governance conditions). Although this was generally confirmed in previous findings, where these regimes clearly and strongly decreased deforestation, I found surprising results on the effects of conservation regimes for biodiversity. Overall, strictly protected areas and sustainable use protected areas had average effects in contrasting directions during 1996-2018 (**Figure 15**). Albeit nonsignificant, sustainable use areas decreased both species richness and Simpson's index (by -8.87, 95% CI -42.38-24.62 and by -0.32%, 95% CI -11.35-10.70%, respectively) and strictly protected areas increased species richness on average by 23.87 (95% CI -5.93.2-53.68), and significantly increased Simpson's diversity index by 10.51% (95% CI 0.74-20.28%) (**Figure 15**). While these overall, average effects suggest some differences between the effectiveness of strictly protected and sustainable-use protected areas may exist, these differences were not clear when calculating estimated effects specifically at 1996, 2007, and 2018. Here, the effect of sustainable-use areas on Simpson's index changed direction in 2007, and increased diversity by 1.6%-1.3 on average, during 2007-2018 (albeit nonsignificant, 95% CI -9.4-12.6%, and -9.8-12.5%, respectively; **Figure 16**, **Table 10**). Note, strictly-protected areas significantly increased Simpson's index after 2007 by 12%, on average (95% CI 2.2-21.9%; **Figure 16**, **Table 10**), and these results were robust to filtering-out areas that not established before the year 2000 (see sensitivity analyses **Figs. 17B, 18B**). In contrast, robustness tests for sustainable-use areas, showed these estimates were not robust to time-filtering, as average effects drastically increased in magnitude, albeit remaining non-significant (**Figs. 17B, 18B)**. Thus, findings on the effects of conservation regimes indicate differences in effectiveness between strictly-protected and sustainable-use areas in preventing the loss of potential species' habitats may exist. Privatization of strictly-protected areas would most likely cause a decrease in potential biodiversity in these areas, whereas I find no evidence for sustainable-use regimes decreasing/increasing potential biodiversity in Brazil during 1996-2018.

Finally, I test hypotheses on the effects of IPLC regimes, on potential biodiversity change. Expected effects of these tenure regimes are mixed, as some hypotheses predict an increase of habitat loss in these regimes because of problems of exclusion, number of resource-users and decision-makers, and governance conditions, whereas many others expect IPLC regimes to be more effective in mitigating habitat loss than private regimes (**Table 5**). Overall, on average, I found ambiguous evidence for the effects of either indigenous or quilombola regimes on biodiversity, as, although both regimes increased both species richness and Simpson's index during 1996-2018 (increased richness by 7.6, and 10.81, Simpson's by 6.3% and 11.6%, respectively **Figure 15**), effects were mostly non-significant (with the exception of quilombola lands increasing Simpson's index by 11.57% (95% CI 0.89-22.24%; **Figure 15**). However, when calculating estimated effects at each time period, effects of both regimes became much clearer, significantly increasing Simpson's index in 2007 and 2018 (indigenous by 7.6-8.1%, and quilombola by 13.5-13.7%; **Figure 16B**, **Table 10**). Similar to effects of protected areas, this indicates indigenous and quilombola regimes increased potential biodiversity relative to private regimes during 2007-2018. Thus, although overall average effects were often nonsignificant, findings in later years support hypotheses that argue IPLC regimes may be more effective than private regimes in mitigating the loss of diversity in species' habitats in Brazil.

Altogether, although overall mean effect estimates were often nonsignificant for both metrics (**Figure 15**), year-specific estimates (**Figure 16**, **Table 10**) suggest most tenure regimes increased potential Simpson's diversity index over time, relative to private lands. The two exceptions to this finding were rural settlements and sustainable-use regimes, both of which yielded nonsignificant effects, sustainable-use regimes which were not robust to time-filtering tests. This increase in Simpson's index suggests that certain tenure regimes had an influence on potential biodiversity change during 2007-2018, and that the privatization of these regimes would likely cause a decrease in potential biodiversity. These results indicate support for many hypotheses that expect land privatization to increase habitat loss and degradation in comparison to other alternatives such as conservation-focused regimes or IPLC lands.

3.4.1 Sensitivity analysis

As a sensitivity analysis, lower and upper bounds for both Hodges-Lehmann point estimates and p-values were calculated using the “rbounds” package in R (**Table 9**). Analogous to the deforestation analysis, calculations show that both Hodges-Lehman estimates and p-values were not highly sensitive to possible small omitted-variable bias (Γ = 1.1-1.3), but were not as robust to possible large omitted-variable bias (Γ = 1.5) (**Table 9**). Given the nonsignficance of most estimates, it is unsurprising to find that bounds for p-values are often close to 1, indicating little-to-no change in the significance of findings in the presence of an unobserved confounder. However, Hodges-Lehmann estimates for Simpson’s index for the indigenous, quilombola, and undesignated regimes, do indicate slight sensitivity to omitted variable bias, with upper and lower bounds widening at larger values of possible bias (Γ = 1.3-1.5). Thus, similarly to results from the deforestation analysis this analysis indicates that the magnitude of estimated differences in outcomes between treatment and control units, and their significance, is only moderately sensitive to the possibility of a missing confounder, if present for both species richness and Simpson’s index estimates.

Table 9. Rosenbaum bounds for estimates of species richness and Simpson’s index. Upper and lower bounds for both Hodges Lehmann point estimates and p-values are calculated for different Γ levels.

		Rosenbaum bounds for species richness estimates				**Rosenbaum bounds for Simpson's diversity index estimates**			
	Γ	hl_lower	hl_upper	pval_lower	pval_upper	hl_lower	hl_upper	pval_lower	pval_upper
Rural Settlements	1	-41.0480	-41.0480	1.0000	1.0000	-0.1369	-0.1369	1.0000	1.0000
	1.1	-47.3630	-34.7890	1.0000	1.0000	-0.1560	-0.1179	1.0000	1.0000
	1.2	-53.1660	-29.1560	1.0000	1.0000	-0.1735	-0.1006	1.0000	1.0000
	1.3	-58.5420	-24.0630	1.0000	1.0000	-0.1896	-0.0849	1.0000	1.0000
	1.4	-63.5490	-19.4250	1.0000	1.0000	-0.2046	-0.0704	1.0000	1.0000
	1.5	-68.2510	-15.1790	1.0000	1.0000	-0.2186	-0.0569	1.0000	1.0000
Undesignated lands	1	-36.2390	-36.2390	1.0000	1.0000	-0.0689	-0.0689	1.0000	1.0000
	1.1	-43.8610	-28.7220	1.0000	1.0000	-0.0909	-0.0470	1.0000	1.0000
	1.2	-51.1840	-21.5110	1.0000	1.0000	-0.1107	-0.0273	0.9998	1.0000
	1.3	-58.1230	-14.8500	1.0000	1.0000	-0.1286	-0.0096	0.8929	1.0000
	1.4	-64.0890	-8.8647	0.9991	1.0000	-0.1449	0.0064	0.2000	1.0000
	1.5	-68.8380	-3.4185	0.8878	1.0000	-0.1598	0.0211	0.0027	1.0000
Protected areas	1	-67.7320	-67.7320	1.0000	1.0000	-0.2085	-0.2085	1.0000	1.0000
	1.1	-75.5350	-60.1830	1.0000	1.0000	-0.2330	-0.1833	1.0000	1.0000
	1.2	-82.9440	-53.4120	1.0000	1.0000	-0.2554	-0.1605	1.0000	1.0000
	1.3	-90.4920	-47.2180	1.0000	1.0000	-0.2762	-0.1400	1.0000	1.0000
	1.4	-97.1230	-41.3300	1.0000	1.0000	-0.2955	-0.1206	1.0000	1.0000
	1.5	-102.8100	-35.7450	1.0000	1.0000	-0.3130	-0.1031	1.0000	1.0000
Sustainable use protected areas	1	-67.1060	-67.1060	1.0000	1.0000	-0.1578	-0.1578	1.0000	1.0000
	1.1	-76.7550	-57.4590	1.0000	1.0000	-0.1889	-0.1263	1.0000	1.0000

		Rosenbaum bounds for species richness estimates				**Rosenbaum bounds for Simpson's diversity index estimates**			
	Γ	hl_lower	hl_upper	pval_lower	pval_upper	hl_lower	hl_upper	pval_lower	pval_upper
	1.2	-86.3080	-48.5520	1.0000	1.0000	-0.2168	-0.0979	0.9994	1.0000
	1.3	-95.1460	-40.1530	0.9999	1.0000	-0.2423	-0.0719	0.9910	1.0000
	1.4	-102.8300	-32.0770	0.9989	1.0000	-0.2660	-0.0477	0.9421	1.0000
	1.5	-109.8300	-24.3110	0.9900	1.0000	-0.2887	-0.0251	0.7984	1.0000
Indigenous	1	-33.1230	-33.1230	1.0000	1.0000	-0.0602	-0.0602	0.9989	0.9989
	1.1	-41.8970	-24.0620	0.9998	1.0000	-0.0868	-0.0338	0.9589	1.0000
	1.2	-50.1730	-15.5440	0.9893	1.0000	-0.1106	-0.0102	0.7019	1.0000
	1.3	-57.6340	-7.9746	0.8849	1.0000	-0.1320	0.0110	0.2808	1.0000
	1.4	-64.3750	-1.2099	0.5718	1.0000	-0.1514	0.0301	0.0538	1.0000
	1.5	-70.8620	4.9790	0.2214	1.0000	-0.1696	0.0475	0.0051	1.0000
Quilombola	1	-44.3070	-44.3070	1.0000	1.0000	-0.0644	-0.0644	0.9995	0.9995
	1.1	-52.4800	-36.8800	1.0000	1.0000	-0.0875	-0.0413	0.9832	1.0000
	1.2	-60.3830	-30.4040	1.0000	1.0000	-0.1089	-0.0209	0.8578	1.0000
	1.3	-67.9320	-24.9030	1.0000	1.0000	-0.1293	-0.0018	0.5405	1.0000
	1.4	-74.9800	-19.9620	0.9998	1.0000	-0.1476	0.0155	0.2132	1.0000
	1.5	-81.5750	-15.3580	0.9968	1.0000	-0.1648	0.0315	0.0514	1.0000

In addition to calculating Rosenbaum bounds, robustness against the violation of the assumption of constant treatment of protected and sustainable-use regimes were also conducted. Protected areas and sustainable use areas that were not established by the year 2000 were filtered out of original unmatched datasets, i.e. excluding parcels that did not exist as their current category for at least 80% of the study period. Matching and statistical analyses were repeated on these time-filtered datasets (**Figs. 17, 18**). Here, it is clear the overall mean results for 1996-2018 were not robust to this test (**Figure 17**, **Table 10**). Effects of both regimes on species richness were drastically reduced to close to zero, yet remained statistically nonsignificant (95% CI -5.51E-17-5E-17, and -4E-16-4E-16, respectively), and effects of both regimes for Simpson’s index were nonsignificant and widened confidence intervals further than for non-filtered estimates. Furthermore, as described in **3.4**, mean effects of sustainable use areas on Simpson’s index changed directions from decreasing to increasing (from 0.32%, 95% CI -11.35-10.70%; to 10.18%, 95% CI -6.64-26.99%; **Figs. 15, 17**), albeit both effects are nonsignificant. However, results for Simpson’s index calculated at each specific year (1996, 2007, 2018) remained qualitatively robust to filtering out areas created after 2000. Notably, protected areas still increased Simpson’s index in 2007 and 2018, indicating post-2007 biodiversity increases were robust to the assumption that protected areas exerted constant treatment effect throughout this period (**Figure 18**, **Table 10**).

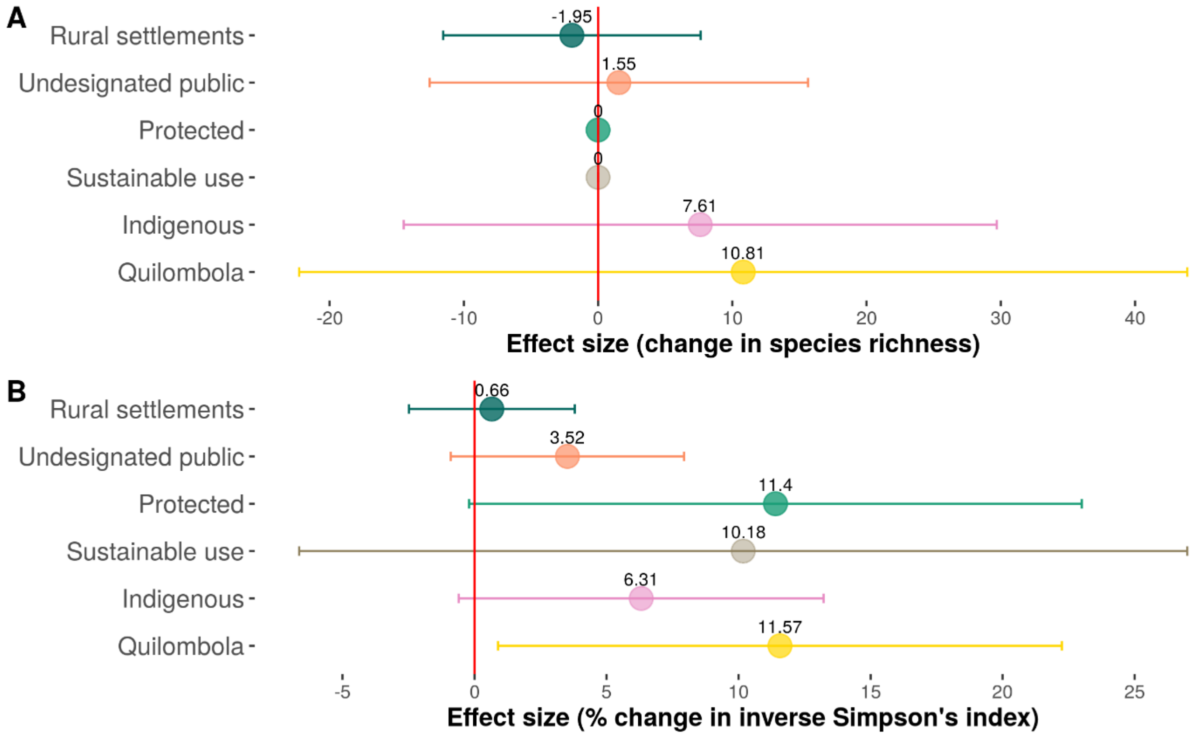

Figure 17. Robustness test of effects of alternative tenure regimes on two biodiversity metrics in Brazil during 1996-2018, potential species richness (A), and potential Simpson's diversity index (B) using filtered time-series data for protected and sustainable-use areas (i.e., only areas established before the year 2000). Circles indicate effects sizes estimated compared to private lands. Effects to the left of the zero line indicate a decrease in either metric, to the right: increase. Upper/lower confidence intervals are plotted to the left/right of each circle centroid. Higher transparency of filled circles indicate high levels of imbalance in the matched dataset (multivariate imbalance measure L_1). Note results of all regimes except protected and sustainable use areas are redundant with **Figure 15**, and are presented here for ease of comparison.

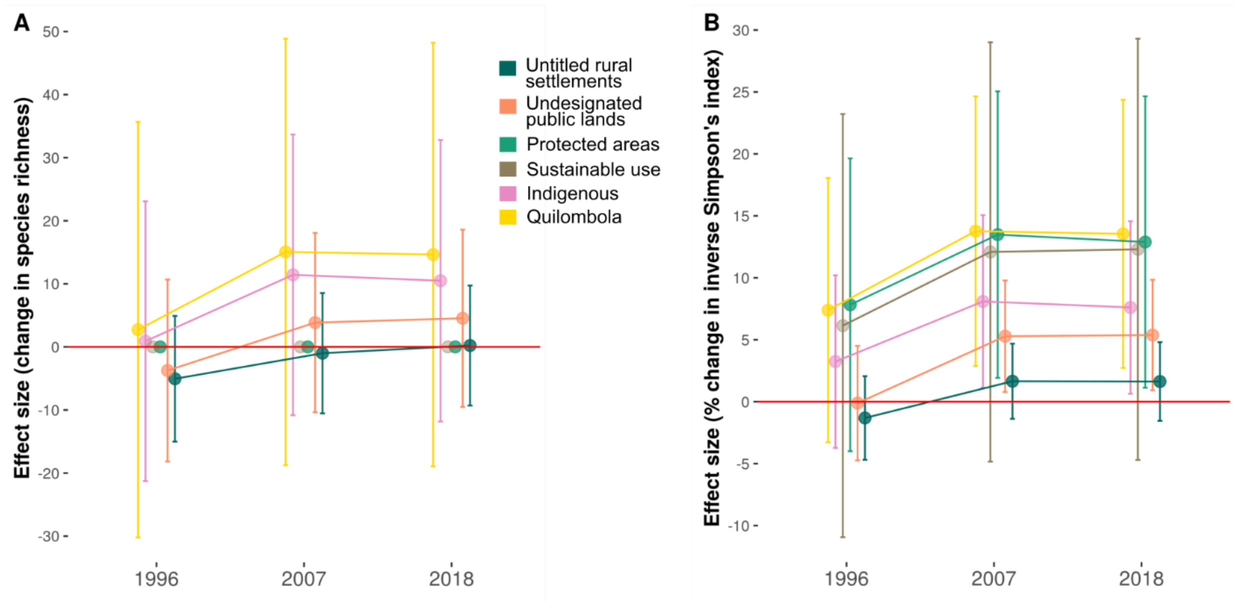

Figure 18. Robustness test of effects of alternative tenure regimes on two biodiversity metrics in Brazil during 1996-2018, potential species richness (A), and potential Simpson's diversity index (B) using filtered time-series data for protected and sustainable-use areas (i.e., only areas established before the year 2000). Circles indicate effects sizes estimated compared to private lands. Effects to the left of the zero line indicate a decrease in either metric, to the right: increase. Upper/lower confidence intervals are plotted to the left/right of each circle centroid. Higher transparency of filled circles indicate high levels of imbalance in the matched dataset (multivariate imbalance measure L_1). Note results of all regimes except protected and sustainable use areas are redundant with **Figs. 15-16**, and are presented here for ease of comparison.

In sum, overall findings of the effects of alternative tenure regimes on two biodiversity measures remained reasonably robust to some possible remaining bias. While time-filtered results calculated at 1996,2007, and 2018 showed that findings for species richness were not robust, they also showed that results for Simpson's index remained qualitatively robust, substantiating the finding that protected areas, alongside other regimes, have increased potential biodiversity post-2007.

Table 10. Model outputs for estimating effects of different tenure regimes on two biodiversity metrics, potential species richness (left), and the inverse of Simpson's index (right). Average Marginal Effects (AME) are reported for each tenure regime (first column) compared to private lands, calculated at each year, 1996, 2007, and 2018, and reporting remaining imbalance levels (L_1), standard error (SE), p-value, and lower and upper confidence intervals.

			Species richness model estimates						**Simpson's diversity index model estimates**					
	Year	**L_1**	**AME**	**SE**	**z**	**p-value**	**lower**	**upper**	**AME**	**SE**	**z**	**p-value**	**lower**	**upper**
Rural settlements	1996		-5.0507	5.0839	-0.9935	0.3205	-15.0148	4.9135	-1.32%	0.0172	-0.7633	0.4453	-4.69%	2.06%
	2007	0.4120	-1.0011	4.8672	-0.2057	0.8370	-10.5407	8.5385	1.65%	0.0155	1.0666	0.2862	-1.38%	4.69%
	2018		0.2189	4.8554	0.0451	0.9640	-9.2976	9.7353	1.63%	0.0162	1.0042	0.3153	-1.55%	4.81%
Undesignated lands	1996		-3.7378	7.3633	-0.5076	0.6117	-18.1696	10.6940	-0.12%	0.0236	-0.0486	0.9612	-4.74%	4.51%
	2007	0.4840	3.8575	7.2506	0.5320	0.5947	-10.3533	18.0684	5.28%	0.0230	2.3020	0.0213	0.79%	9.78%
	2018		4.5401	7.1719	0.6330	0.5267	-9.5165	18.5968	5.38%	0.0228	2.3647	0.0180	0.92%	9.84%
Protected areas	1996		21.0692	15.3700	1.3708	0.1704	-9.0555	51.1938	7.39%	0.0505	1.4624	0.1436	-2.52%	17.30%
	2007	0.5328	25.7387	15.0544	1.7097	0.0873	-3.7674	55.2447	12.06%	0.0500	2.4144	0.0158	2.27%	21.86%
	2018		24.8001	15.3760	1.6129	0.1068	-5.3363	54.9364	12.08%	0.0501	2.4115	0.0159	2.26%	21.89%
Sustainable use protected areas	2007		-13.7637	17.5968	-0.7822	0.4341	-48.2527	20.7253	-3.87%	0.0577	-0.6698	0.5030	-15.18%	7.45%
	2018	0.4840	-6.1535	16.7203	-0.3680	0.7129	-38.9247	26.6177	1.56%	0.0561	0.2785	0.7806	-9.44%	12.57%
	2018		-6.7219	17.2361	-0.3900	0.6965	-40.5039	27.0601	1.33%	0.0568	0.2337	0.8153	-9.81%	12.46%
Indigenous lands	1996		0.9063	11.3170	0.0801	0.9362	-21.2745	23.0872	3.24%	0.0356	0.9107	0.3624	-3.73%	10.21%
	2007	0.5405	11.4277	11.3534	1.0065	0.3142	-10.8246	33.6800	8.09%	0.0356	2.2765	0.0228	1.13%	15.06%
	2018		10.4913	11.3890	0.9212	0.3570	-11.8307	32.8134	7.61%	0.0355	2.1401	0.0324	0.64%	14.57%
Quilombola lands	1996		2.7292	16.8055	0.1624	0.8710	-30.2089	35.6673	7.39%	0.0544	1.3573	0.1747	-3.28%	18.06%
	2007	0.4717	15.0592	17.2434	0.8733	0.3825	-18.7372	48.8556	13.77%	0.0555	2.4790	0.0132	2.88%	24.65%
	2018		14.6492	17.1304	0.8552	0.3925	-18.9258	48.2241	13.55%	0.0552	2.4529	0.0142	2.72%	24.37%
Robustness check: conservation regimes filtered by known year of creation														
Protected areas	1996		0.0000	0.0000	4.9173	0.0000	0.0000	0.0000	7.82%	0.0603	1.2971	0.1946	-4.00%	19.64%
	2007	0.4088	0.0000	0.0000	0.0000	1.0000	0.0000	0.0000	13.49%	0.0590	2.2854	0.0223	1.92%	25.05%
	2018		0.0000	0.0000	0.0000	1.0000	0.0000	0.0000	12.89%	0.0600	2.1470	0.0318	1.12%	24.65%
Sustainable use protected areas	1996		0.0000	0.0000	0.0000	1.0000	0.0000	0.0000	6.14%	0.0871	0.7045	0.4811	-10.94%	23.22%
	2007	0.6506	0.0000	0.0000	0.0000	1.0000	0.0000	0.0000	12.09%	0.0864	1.3999	0.1616	-4.84%	29.01%
	2018		0.0000	0.0000	0.0000	1.0000	0.0000	0.0000	12.30%	0.0868	1.4179	0.1562	-4.70%	29.30%

4. Discussion

The overarching aim of this dissertation was to better understand the influence of land tenure on global environmental change. In contrast to existing evidence which is commonly limited in thematic and geographical scope, the approach I used involved pairing quasi-experimental methods for estimating causal effects at broad spatial and temporal scales. To carry out this approach, I asked four specific research questions that were each addressed throughout Chapter 3:

1. What is the state of global land tenure data findability and accessibility?
2. According to predominant theories and evidence to date, how are different land tenure regimes expected to affect environmental change in comparison to each other?
3. What are the effects of different land tenure regimes on deforestation?
4. What are the effects of different land tenure regimes on biodiversity change?

This chapter will review and discuss the main findings of this dissertation as they relate to the above questions. In section **4.1,** I discuss how the state of land tenure data currently limits further research on this topic globally, and specifically in Latin America (section **4.1**), and discuss the potential for future research and progress in this field. In section **4.2** I review the main empirical findings on the effects of different land tenure regimes on deforestation and biodiversity, and discuss how these findings compare to theoretical and empirical expectations. In section **4.3**, I discuss how my findings can provide specific insight to land-reform and environmental/agricultural policies in Brazil, and how these might influence policies regionally or across other similar contexts. I conclude with a discussion on the limitations of the analyses and outlook for future research in this topic.

4.1 Data gaps

Results from the data search effort overall showed that spatially explicit data on land tenure was extremely difficult to find and access, from both global land tenure stakeholders, as well as local organizations and data institutes. While data were not accessible for a variety of reasons, there were several common prohibitive factors across data holders. The first factor was related to the overall unwillingness and prohibitive sociopolitical aspects of acquiring such data. For instance, the high cost of data collection for some organizations resulted in a highly protective attitude and an overall hesitancy to share datasets outside of their own organization. Other organizations required full, in-person proposals to be presented for the use of the data, as well as personal connections and legal institutional arrangements in order to establish a formal partnership and be able to share data. Second, there were factors related to privacy and institutional barriers that also often prevented data-sharing. For example, there were organizations unable to share data due to inherent privacy constraints, i.e. when the data was collected it was made clear they would not be shared with anyone else, or state organizations that restricted data access to citizens of that country. Third, prohibitively high costs for accessing data remained an important factor, particularly in contexts where spatially explicit tenure data may be used for-profit. Given the effort and resources required to organize such arrangements, it is clear that any future endeavor to acquire similar land tenure data will require local knowledge and partnerships, building personal connections and a continual relationship with a given organization, as well as communicating planned uses for acquired data.

Despite results showing that findable and accessible spatially-explicit land tenure exist, I found there were common barriers for making these data truly accessible, interoperable, and reusable for scientific analysis. These barriers were often related to countries' institutional infrastructure, i.e. the technical and administrative arrangements and structures that govern a land tenure system in a given country. First, there was often an overall lack of nationally-defined land tenure categories that described the legal set of rights attributed to each category. This led to confusion, overlaps and gaps between organizations that did not know who had what information, and whether that information was adequate for scientific purposes. Second, there was a lack of anonymization, or lack of ability to easily anonymize data. This was a particularly confusing factor for data-holders, as they would assume that parcel-level data would inherently include private information regarding a given property owner, which in turn, resulted in overall reluctance to share this data. Third, there was a lack of published date of parcel creation or titling. While it was not uncommon to find publicly accessible parcel-level data, this was often not accompanied with the required information about that parcel (e.g. only a name or number, no translating table or legend). These three elements are essential pieces of information required by many fundamental questions in research on the environmental and economic effects of land tenure and property rights, and this data search effort showed that these elements remain important barriers for future research.

Potentially, these barriers could be partially addressed by the adoption of data standards, e.g. the Land Administration Domain Model (LADM), a data standard for land administration which has been under development over the past several years (Lemmen et al., 2015)). Results from the data search effort show that the pieces of information most often missing from countries' infrastructure, identified in the LADM are 1) a standardized global vocabulary, 2) the documentation of rights, responsibilities, and restrictions (ownership rights), and 3) spatial units (i.e. parcels). The effort also shows that the LADM "package" related to parties (i.e. people and organizations) needs to be easily anonymize-able for stakeholders to be willing to publish the data. Full implementation of such a standard, however, may be further challenging because communal/traditional tenure regimes may have internal definitions, as well as overlapping rights over multiple resource types. Identifying these kinds of differences with nationally-determined legal rights of communal/traditional lands does not bar the implementation of these standards, however differences will need to eventually be addressed and incorporated into the standard, e.g. (Paixao et al., 2015). Thus, while the adoption of LADM or alike standards may be promising for land tenure research in enabling data availability, it still lacks wide implementation, and may likely remain minimally used due to a lack of resources, particularly in the global south (Kalantari et al., 2015; Kalogianni et al., 2021).

4.1.1 Towards better causal inference through potential data improvements

Given these data limitations and barriers, robust analyses on the environmental effects of different land tenure regimes that implement credible causal or counterfactuals study designs remain scarce (Robinson et al., 2014; Tseng et al., 2020), which can be problematic in building consensus on the understanding of these effects. Though the association of some of these relationships may be quite clear (e.g. IPLC regimes claim or hold most of the remaining intact natural landscapes in the world), depending on the counterfactual, these associations do not necessarily indicate causal relationships (e.g. the formalization of IPLC land rights might not necessarily increase environmental conservation outcomes). Here, interpreting correlation as causation is problematic because it may lead to costly interventions and policies where expectations fail to yield outcomes – or may yield outcomes in opposite direction than those expected (e.g. large-scale titling of private properties in the Amazon resulted in deforestation effects before and after titles are awarded (Probst et al., 2020)).

Improvements in data quality and availability could considerably improve causal understanding of the environmental effects of different land tenure regimes. While conducting randomized control trails (RCTs) – that is, randomized field experiments – would be considered the "gold standard" of causal inference, they can be extremely expensive, difficult to implement at vast geographical scales, and are criticized for their low external validity. Using observational data and quasi-experimental design is thus a common solution to answering causal questions in a cost-effective way, and, provided the most appropriate study design is used and required assumptions are met, these methods may also have broader external validity. However, successfully using observational data for improved causal inference in future research on these effects will likely require at least: a date of intervention (e.g. land title granted, specific property rights granted, land demarcated, etc.), a clear categorization of the land tenure system in order to be able to define comparison groups, and a spatial entity that can be related to environmental data. Given these requirements are met, there are several quasi-experimental methods available that could potentially improve inferences in this field (**Table 11**), notwithstanding ongoing progress in this field already (see column *Example references*, **Table 11)**.

Table 11. Non-exhaustive overview of potential methods that could be used for improved causal inference in land tenure research in the future. Final row lists some key overarching references for appropriate implementation of causal study designs and methods in socio-environmental research.

Method	Description	Example references
Panel-data/time-series analysis	Makes repeated observations of one unit of analysis over time, overcoming omitted variable bias of all factors that are time-invariant	(Hargrave & Kis-Katos, 2013; Probst et al., 2020)
Differences-in-differences (DiD) (also known as Before-and-After Control-Intervention Contrast)	Estimates effects by measuring before and after differences between treatment and control groups, meaning that the initial differences between groups are accounted for, and thus does not rely on the randomization of observations.	(Anderson et al., 2018; Heilmayr & Lambin, 2016; Kraus et al., 2021)
Synthetic control design	An artificial control group is created and corroborated through a long pre-treatment period. Effects are subsequently estimated very similarly to DiD. This method is less vulnerable to omitted variable bias under the assumption that if the long pre-treatment period does not reveal any bias from unobserved variables in the synthetic control group, there is likely no bias post-treatment as well.	(Sills et al., 2015; West et al., 2020)
Regression discontinuity design (RDD)	Accounts for omitted variable bias by taking advantage of a "discontinuity" or threshold in the application of a treatment in either space or time (e.g. a test score, or a political border can be used as discontinuities). This threshold will naturally separate treatment and control groups, and they can be compared under the assumption that observations closest to the threshold all have the same unobservable variables between, mimicking a natural experiment.	(Baragwanath & Bayi, 2020; Crespo Cuaresma & Heger, 2019; Wuepper et al., 2020)
Other key references for implementing causal analysis:	(Angrist, 2008; Butsic et al., 2017; Cunningham, 2021; Hernán & Robins, 2020; Larsen et al., 2019; Wauchope et al., 2021)	

Results from the data search effort show there are already countries in the Latin American region where this research could potentially be implemented with a few additional resources and effort. Colombia, for instance, has served as a LADM model (Jenni et al., 2017); and certain tenure regimes are well mapped and have already been rigorously studied (Vélez et al., 2020), indicating a cross-country comparison with other tenure regimes is possible. Mexico has rich parcel-level datasets which could be more rigorously studied, especially with a clearer understanding of rights granted to each tenure category, and their differences and overlaps (Bray et al., 2008; Miteva et al., 2019). Some countries have data mapping potential that could be particularly fruitful through partnerships with local researchers and/or organizations, e.g. Guatemala, Bolivia, Costa Rica, or Nicaragua. Other countries could be approached through a large data compilation effort, e.g. Argentina or Uruguay. Nevertheless, reaching a fruitful level of potential will require addressing the barriers and limitations discussed above.

In conclusion, while there is a variety of data on global land tenure currently being collected, gathered, and maintained, research on land tenure could improve causal rigor and scope of analyses by finding and accessing more interoperable data (i.e. data that is better collected, maintained, that contains metadata, and uses standardized, broadly applicable vocabularies). Provided data privacy concerns are respected, data on public and privately-held lands should be especially prioritized given the large amount of global land that fits in either of these categories. Data on public lands, in particular, should be made available to the public to promote transparency and accountability of state-held lands. The importance of studying both public and private tenure regimes in particular is underscored by the empirical findings of my analyses, which would not be possible without openly available access to these data. Progress in this line of research will improve when important institutional and political barriers that prevent using this data are improved.

Using these data to establish effects rather than associations is not only important for improving our understanding of theoretical relationships, but also for implementing sustainable development and environmental conservation policies relating to land tenure around the world. Identifying the tenure interventions that can be widely beneficial for economic and environmental outcomes remains an important research frontier that is contingent upon the improvement of current data and methods.

4.2 Theoretical and empirical implications

The empirical analyses conducted in this dissertation (**3.3-3.4**) overall provide robust evidence on the effects of different land tenure regimes on environmental change over broad spatial and temporal scales in Brazil. These complementary analyses reveal how tenure regimes may affect deforestation and biodiversity differently, which is a key and novel contribution of this research with implications that are relevant to several scientific fields and policy-making.

In my overview of hypothesized effects of different tenure regimes (**Table 5**), I focused specifically on deforestation effects, expecting that these theories would apply very similarly to biodiversity as they do to forests, especially given the ecological context in Brazil, where forest and biodiversity conservation – particularly habitat loss – go hand in hand (Mittermeier et al., 2005). Moreover, in the literature, these theories have historically rarely been framed as biodiversity issues per se. Instead, they are more commonly referenced as natural resources in general, and when the literature does focus on a particular resource system, these tend to be forests or fisheries. Additionally, when referring to effects on biodiversity, I specifically address changes in species' potential diversity and distributions based on their habitat changes and losses. Therefore, most of the theories that are framed for deforestation in **Table 5** are closely related to those on biodiversity change as changes in potential species' habitats. Nonetheless, it is worth noting that articulating these expectations is a novel contribution of this thesis, as specific expectations for expected effects of land tenure on biodiversity change are rarely addressed in the literature.

The following three subsections discuss how these empirical findings on forests and potential biodiversity compare to existing theory and empirical evidence. Notwithstanding the large spatial overlap of forest cover and high biodiversity in Brazil (**Figs. 8, 12**), results indicate that effects on potential biodiversity may be different than those on forests. I found that undesignated/untitled lands widely increase agriculture-driven deforestation in Brazil, compared to all alternative tenure regimes, and that, out of all other alternatives, private regimes are usually the least-guaranteed to yield positive outcomes for forests. In contrast, I found effects of tenure on both potential species richness and Simpson's index of diversity were overall much less clear, as undesignated lands had similar effects on these metrics as private regimes, and other alternative regimes only significantly increased potential biodiversity vis-à-vis private from 2007-2018. These complementary findings substantiate recent studies on biodiversity change in the region, which suggest there may be trade-offs between protecting forests and endemic species (Green et al., 2019).

Thus, I next discuss key similarities and differences in findings as they relate to the overall theory on the drivers of global environmental change first reviewed in the Introduction. In **4.2.1** I discuss effects of both undesignated/untitled tenure regimes as open-access lands in contrast to effects of communally-held regimes. In **4.2.2**, I discuss the nuanced effects of privatization and formalization – and how these compare to theoretical/empirical expectations, and in **4.2.3**, I discuss findings on conservation-focused regimes, as well as broader implications for theoretical debates in land-system science.

4.2.1 Open-access resources, the "commons", and communal lands

Hypotheses on the expected effects of open-access tenure regimes initially argued that any communally-held resource would likely lead to environmental degradation and collapse (Gordon, 1954; Hardin, 1968). More recent theories, however, argued that communally-held, “common-pool” resources can be successfully sustained in many instances (Sandler, 2015), and that property rights play a key role in the management of said resource (Baland & Platteau, 1996; Boudreaux, 2015; Ostrom, 2009). Yet, when some theories confuse “the commons” with communally-held, and other theories focus exclusively on common-pool resources, there is little understanding on the specific effects of “open-access” land tenure regimes (i.e. those lacking exclusion rights) or other tenure regimes with poorly-defined property rights (e.g. settlements lacking any formally defined rights). Thus, theoretical expectations and current empirical evidence on the effects of undesignated lands and rural settlements on deforestation and biodiversity are ambiguous.

On the one hand, in line with many classical theories, I found consistent and clear results on the effects of undesignated/untitled regimes on deforestation, though on the other hand, effects were surprisingly less clear for potential biodiversity changes. In analyzing effects of different tenure regimes on deforestation, one of the strongest results was on the deforestation-increasing effects of undesignated/untitled public lands compared to all other regimes. Results showed that these lands with poorly defined rights on unprotected public lands increased deforestation across vastly different contexts. This clear and consistent result across scales indicates strong support for all the hypotheses that predict higher deforestation in undesignated/untitled lands, including classic hypotheses on effects of exclusion and alienation rights (Binswanger, 1991; Browder et al., 1997; Gordon, 1954; Grafton, 2000; Hardin, 1968; Sandler, 2015), as well as development economics theories on withdrawal rights (Angelsen, 1999; Bray et al., 2008; Duchelle et al., 2012; Ellis & Porter-Bolland, 2008; Fearnside, 2005; Nepstad et al., 2006; Porter-Bolland et al., 2012; Redo et al., 2011)(**Table 5**). However, in analyzing effects of these regimes on biodiversity change, results were not as clear or consistent. Here, I found no statistically-significant difference in effects between rural settlements and private tenure regimes on either metric of potential biodiversity, and additionally, found an average 5% increase in Simpson’s diversity index in undesignated lands compared to private regimes during 2007-2018. These findings could be explained by hypotheses describing how alienation, withdrawal rights, and market integration mechanisms (Anderson et al., 2018; de Soto, 2000; Place & Otsuka, 2002), and tenure security can increase the likelihood of private regimes of increasing habitat loss in comparison to undesignated lands or untitled rural settlements (Angelsen, 1999; Birdyshaw & Ellis, 2007; Deacon, 1994a; Deininger et al., 2003b; Deininger & Jin, 2006; Fearnside, 2005; Fenske, 2011; S. Holden & Yohannes, 2002; Liscow, 2013; Robinson et al., 2014).

At first these two findings on the effects of undesignated/untitled lands appear contradictory. However, it is important to note that although undesignated/untitled had clear deforestation increasing effects compared broadly against all other alternatives, these effects were more mixed across narrower scales vis-a-vis private regimes, i.e. effects were non-significant in almost 30% of cases comparing undesignated/untitled lands to private. Nonetheless, findings on the effects of these regimes on potential species diversity do suggest there may be further complexity to existing theory and evidence on the influence of land tenure on environmental outcomes, as effects of these regimes on forests might not be the same as those on species’ habitats. Speculatively, this could be due to key differences in the management of different resource-systems and implied land-use conversions (Ostrom, 2009), or even gaps in policies addressing non-forest environmental conservation (see also **4.2.2**). In fact, evidence on the effects of these regimes on potential biodiversity change suggests that effects of open-access regimes (i.e. undesignated lands) may be different than effects of poorly-defined tenure regimes (e.g. rural settlements), calling for careful differentiation between tenure regimes and resource systems alike.

In parallel, in line with many expectations predicting the sustainable management of common-pool resources, I found both indigenous and quilombola regimes on average decreased deforestation in Brazil during 1985-2018, and that both regimes increase potential Simpson's diversity index vis-a-vis private lands during 2007-2018. However, I also found that deforestation-effects of these regimes were often ambiguous, with nonsignificant effects in 47% of narrower scales, which substantiates mixed expectations for these regimes, and suggests that any given community may not always be willing to bear the organizational costs of preventing environmental change (Gordon, 1954; Hardin, 1968; Sandler, 2015). Additionally, I found differential effects between the two regimes, as indigenous lands often reduced deforestation effectively, but quilombola lands were less effective or reliable in reducing deforestation. These differences in effects may be explained by the fact that quilombola lands have commercial withdrawal rights (indigenous lands only have subsistence rights), but could also be explained by hypotheses that expect regimes administered by the state (i.e. indigenous) to have lower deforestation than self-administered regimes (in this case, quilombola lands) because the state can benefit from economies of scale for monitoring, enforcement, and management activities that prevent deforestation (Grafton, 2000).

Altogether, these results on the probable, yet unreliable benefits of IPLC regimes for forests and potential species diversity, strongly evidence the need to distinguish communally-held regimes from open access regimes (or "commons"), corroborating many appeals on this topic (Berkes et al., 1989; Schlager & Ostrom, 1992). More specifically, results evidence that effects of IPLC regimes can at times be similar, but rarely worse than undesignated/untitled counterfactuals (non-significant effects on deforestation found in 38% of cases, deforestation-increasing effects in 1%; **Table 6**), substantiating hypotheses that expect IPLC regimes to be just as effective as other tenure regimes in promoting environmental conservation – if not more (Ceddia et al., 2015; Fa et al., 2020; O'Bryan et al., 2021).

4.2.2 Land privatization and formalization

There is little consensus in the literature on the expected effects of private tenure regimes on environmental/land-use change, as the privatization of land is expected to provide incentives to sustain a resource for a long time, but also the means and incentives to exploit those resources. At the same time, the formalization of IPLC tenure regimes is widely promoted, as there is a high level of consensus that their traditional management of resources is expected to guarantee environmental conservation outcomes.

In line with mixed expectations on the environmental effects of land privatization, I found nuanced effects of private tenure regimes on deforestation and biodiversity. On average, private regimes decreased deforestation vis-à-vis undesignated/untitled lands, implying that promoting private land rights over undesignated/untitled public lands may more often outweigh the deforestation risks than vice versa. However, as previously mentioned (**4.2.1)**, results across narrower scales do not indicate such clear dichotomy between these regimes, as effects were inconsistent across narrower spatiotemporal scales. Furthermore, private tenure regimes increased deforestation compared to most other alternatives (conservation-focused and IPLC regimes).

Deforestation-increasing effects of private tenure regimes support various hypothesized mechanisms in the literature. Private regimes are predicted to increase LUC, as through alienation rights and access to collateral, they have greater access to capital to engage in forest-displacing agricultural activities (de Soto, 2000; Place & Otsuka, 2002), and with functioning markets, land will be transferred to those that will put them to their most financially productive use (Deininger et al., 2003a). These results could also support hypothesized mechanisms related to withdrawal rights, where tenure regimes with commercial withdrawing rights are thought to be more economically capable of high-input land uses, facilitating deforestation at larger scales (Anderson et al., 2018), as well as more incentivized to make further investments, with the assurance of receiving future benefits (Liscow, 2013). Deforestation-increasing results could also support hypotheses on mechanisms related to exclusion and due process, where if there is perceived security that benefits from investments will be enjoyed exclusively, private lands will have the highest incentive to invest in the land use that will be of greatest long-term economic utility (i.e. agricultural uses). (Birdyshaw & Ellis, 2007; Deacon, 1994a; Deininger et al., 2003b).

Moreover, private tenure regimes significantly decreased Simpson's index of diversity compared to most other alternatives during 2007-2017, which surprisingly included undesignated lands (as discussed in **4.2.1**). These results imply that while promoting the privatization of undesignated/untitled lands might often decrease deforestation (albeit unreliably), it likely decreases potential biodiversity. This could suggest that despite maintaining forest-cover, private regimes may still drive non-forest habitat loss, to the detriment of species' habitat diversity. This result could also be supported by previously discussed mechanisms related to deforestation-increasing effects of private tenure regimes, as land in these regimes may still be allocated to its most productive purpose – while maintaining the required amount of forest cover (notwithstanding the legal requirement to maintain a certain percentage of land applies to all native-vegetation-cover, not merely forests (Soares-Filho et al., 2014)).

I found one contrasting result to this overall pattern in the Amazon, where private lands went from having the highest deforestation increasing effects in earlier years (1985-1995), to the second-highest deforestation decreasing effects (after quilombola lands) by the latest time periods (2005-2018) (**Figure 10**, left panel). This particular finding could be explained by hypotheses that expect contexts with weak land governance to have impaired excludability, leading towards less deforestation in regimes where local tenants are responsible for monitoring and enforcement (e.g. private lands and quilombola lands) (Angelsen, 1999; Fearnside, 2005; Grafton, 2000; Nolte et al., 2013). They could also be explained by hypotheses that predict that in regimes where a single entity is the main rights holder (i.e. private tenure), state agencies have better opportunities for environmental legislation enforcement (e.g. Forest Code) on these single-entity regimes than in other regimes (undesignated/untitled, IPLCs), (Arima et al., 2014; Hargrave & Kis-Katos, 2013). Finally, volatile, short-lived governments could also make publicly-administered lands less effective than private regimes in conserving forests (Baland & Platteau, 2000; Deacon, 1994a). It is important to note that results also indicate directly privatizing any of these publicly-administered regimes (PAs, indigenous lands) would most likely lead to increases deforestation (**Figure 10**, right panel).

This finding in the Amazon may appear to contradict other empirical studies, where titling interventions seem to have increased deforestation (Probst et al., 2020), however, my results complement Probst et al.'s, by exclusively considering property owners who have not enrolled in the Terra Legal program, and assessing deforestation effects relative to undesignated/untitled counterfactuals (rather than a before-and-after intervention).Thus, the effects in Amazonia indicate that privatization may only effectively counters the specific deforestation mechanisms acting on Amazonian undesignated/untitled lands (e.g. enforcement strategies (Assunção et al., 2013)), but may not address those on publicly administered lands, which, for instance, could include mechanisms related to relaxing measures for agribusiness, mining, oil, and gas extraction (Ferrante & Fearnside, 2020, 2021) (see **4.3**). Overall, in line with mixed theoretical and empirical expectations, findings on the privatization of land indicate effects of private tenure regimes can be nuanced depending on the counterfactual and the specific context.

In contrast to many strong expectations that the formalization of IPLC lands unambiguously mitigate environmental loss, as discussed in **4.2.1**, I found both indigenous and quilombola tenure regimes had ambiguous effects on deforestation and biodiversity in Brazil. While both regimes often have deforestation-decreasing/biodiversity-increasing effects, these are not consistent or reliable across all contexts (see **4.2.1**, **Table 6**). Thus, despite substantial spatial overlap between indigenous and quilombola lands and high levels of forest cover and potential biodiversity (**Figs. 6, 8, 12**), empirical effects evidence environmental conservation outcomes are not necessarily guaranteed in these regimes. Notably, this indicates deforestation-decreasing/biodiversity-increasing effects of IPLC regimes may often be context-specific, which is a novel qualification on the extent and reliability of expected effects (notwithstanding classical expectations of resource management in communal lands). This finding does not suggest that IPLC regimes have detrimental effects on environmental conservation (rarely do they increase deforestation/decrease potential biodiversity, see **4.2.1**, **Table 6**). Rather, results indicate interventions aiming to guarantee environmental conservation outcomes in these regimes must likely engage in in-depth contextual studies to guarantee these outcomes, and view IPLC tenure regimes as strategic partners for conservation, rather than a mechanism for a desired outcome. This substantiates appeals that the formalization of IPLC lands (e.g. through land titles) is not enough to guarantee environmental conservation outcomes (Robinson et al., 2014; Robinson, Holland, et al., 2017; Vélez et al., 2020).

4.2.3 Regulating negative externalities

As reviewed in the introduction, one of the most common instruments used to regulate and restrict the negative environmental externalities of other socioeconomic processes, is the creation of conservation-focused regimes such as protected areas. While theoretical expectations predict these regimes may broadly – yet, inefficiently – yield conservation outcomes, empirical studies often show protected areas have mixed effects, due to the bias in their location, governance issues which render them ineffective, as well as the implicated disruptions in existing land tenure systems which may have unintended socioecological consequences. Nonetheless, hypotheses mostly predict that conservation regimes decrease deforestation compared to both private and undesignated/untitled tenure regimes, with the exception that in contexts of weak land governance with little-to-no monitoring and enforcement private regimes are better equipped to monitor their land, and thus are likely to decrease deforestation by comparison (**Table 5**).

In line with these hypotheses, I found that both conservation-focused regimes – strictly protected areas and sustainable use areas – decreased deforestation consistently and reliably across spatiotemporal scales., and that strictly protected areas significantly increased potential Simpson's diversity index during 2007-2018. Despite doubts about their effectiveness, these strong results evidence that such conservation-focused tenure regimes are essential instruments for environmental conservation outcomes in vastly different socioecological contexts, for both forests and biodiversity change (Ferreira et al., 2020; Yang et al., 2021). However, while I found sustainable-use areas often had larger effects that strictly-protected areas in preventing deforestation, I also found sustainable-use areas had inconclusive effects on potential biodiversity (overall nonsignificant and highly sensitive to robustness tests). These differential effects substantiate other studies that find differences in strictly protected and multiple-use protected areas (Amin et al., 2019; Blackman, 2015), and studies that find earliest-established protected areas in the Amazon are the most effective in protecting natural vegetation (Gonçalves-Souza et al., 2021). However, future research could focus on better understanding the differences in conservation effectiveness between these regimes that are driven by differences in their respective property rights.

Aside from the effectiveness of conservation-focused regimes, results on the difference of effects on forests and potential biodiversity may be relevant to the land sparing vs. sharing debate. Previous studies in the region suggest agricultural intensification is having land sparing effects, i.e. agricultural intensification has been linked with decreases in deforestation (Garrett et al., 2018; Koch et al., 2019). However, my findings on the effects of private regimes in decreasing deforestation vis-à-vis undesignated/untitled lands, yet, decreasing potential diversity of species' habitats (5% decrease in Simpson's index 2007-2018, see **4.2.1**), indicate a decrease in deforestation may not necessarily imply net biodiversity conservation. Speculatively, instead of private farms contracting agricultural land via intensification (i.e. land sparing), avoided deforestation may be due to agricultural extension leaking onto other ecosystems (possibly, non-forest ecosystems such as wetlands, grasslands, or savannas). The latter possibility would be in line with other studies quantifying agricultural expansion in the region (Ceddia et al., 2014; Graesser et al., 2015). Findings on the difference of tenure effects on forests and potential biodiversity suggest that conservation efforts and policies need to be integrative across ecosystems and resource systems in order to ensure outcomes for biodiversity conservation, as is often argued in the literature (Leclère et al., 2020).

In summary, findings from this research help improve theoretical and empirical understanding on the effects of different land tenure regimes on deforestation and potential biodiversity, and contribute towards building middle-range theory in land-system science by rigorously testing the bounds of these effects. Results specifically quantify the reliability and consistency of effects under different contexts, providing strong support for how different land tenure regimes do in fact determine environmental changes, often times with clear and consistent effect direction and magnitude; e.g. IPLC regimes generally have consistent effect direction, but not magnitude, whereas private regimes have inconsistent effect direction. Synthesis of these results also identify unique contexts and circumstances where effects are shown to deviate from overall patterns, qualifying the context-dependency of some general patterns of effects.

4.3 Policy implications

In Brazil, there have been a myriad of policy instruments that have influenced how land tenure affects environmental change, including agriculture/conservation policies (e.g. soy and beef moratoria, REDD+, the Forest Code), land regularization interventions (e.g. PPCDAm (2004), PPCerrado (2010), CAR), as well as monitoring and enforcement schemes (e.g. DETER). While the work conducted in this dissertation does not evaluate each of these policies specifically, analyses examining the broad effects of different tenure regimes can indicate how well the tenure-related mechanisms of these policies may work. This is key in informing ongoing policies in Brazil, especially given uncertainty regarding changing environmental priorities (Ferrante & Fearnside, 2021; Reydon et al., 2020), as well as other ongoing processes in similar tropical contexts that often model their forest-governance policies after those in Brazil (Shankland & Gonçalves, 2016; Tollefson, 2015).

Overall, findings from this research strongly evidence how the lack of property rights and/or poorly defined property rights are drivers of deforestation. Interventions on these undesignated or untitled lands provide an opportunity to decrease deforestation rates in Brazil (e.g. through the creation of more conservation regimes, the recognition of IPLC land claims, or regularizing and providing broader legal options for informal land settlers). This is particularly relevant for the vast amount of undesignated lands in Amazonia, as a growing number of studies also advise (Azevedo-Ramos et al., 2020; Azevedo-Ramos & Moutinho, 2018). While effects of poorly defined property rights are less clear for potential species richness and diversity, findings do not necessarily indicate property-rights interventions on these lands would be detrimental for species diversity. Instead, results indicate there is little clarity on whether undesignated/untitled decrease biodiversity in comparison to private lands. Furthermore, results suggest that environmental policies targeting either undesignated/untitled and private lands must consider forest and non-forest ecosystems alike in order to wholly ensure biodiversity conservation outcomes (e.g. monitoring and enforcement schemes could incorporate biodiversity indicators besides forest cover loss). This also underscores recommendations from other studies in Brazil, which indicate the success of policies like the Forest Code are contingent upon measures to protect the Caatinga and Cerrado biomes as well as the Amazon (Brock et al., 2021; Klink & Machado, 2005).

Evidence on the effects of private tenure regimes on Brazilian forests and potential biodiversity is mixed. Deforestation-decreasing effects of private regimes against undesignated/untitled regimes was unreliable, while at the same time, private regimes were highly likely to increase deforestation/decrease potential biodiversity against alternative tenure regimes. These findings highlight how the privatization of land alone is unlikely to guarantee environmental conservation outcomes, as land titles can provide different incentives to effectuate environmental conservation and degradation alike.

However, I also found that private regimes in the Amazon went from having the highest deforestation-increasing effects to having high deforestation-decreasing effects (**4.2.2**, **Figure 10**, left panel). Over recent decades, private landholders in Amazonia have been subject to stricter forest-protection policies than those in other biomes, including four times higher requirements on retaining forest cover and earlier-implemented commodity moratoria (Gibbs et al., 2015; Soares-Filho et al., 2014). At the same time, understaffing and logistic difficulties due to Amazonia's remoteness may disproportionately limit the effectiveness of government policing of the region's public reserves (Nolte et al., 2013). This might indicate that for remote public lands with poorly defined tenure rights and limited public capacity for on-the-ground control, privatization that is strongly coupled to extensive environmental obligations (Karp, 1993) may be effective in reducing deforestation. Thus, policies that function via partially transferring responsibility and accountability for forest governance from public institutions to specific individuals (e.g. the CAR/Forest code) have likely played a role in decreasing deforestation in private regimes. They also suggest that the stringency of private-actor-focused environmental policies in Brazil's other remote biomes, where remaining forestland is mostly private (Cerrado: 80.4%; Pantanal: 92.8%; **Fig. 1b-c**), may be a key factor determining future Brazil-wide deforestation rates (Gibbs et al., 2015; Soterroni et al., 2019)

Notwithstanding these results in Amazonia, directly privatizing any of the alternative tenure regimes – which is a possibility under current political administration (Ferrante & Fearnside, 2019, 2020, 2021) – would likely lead to increased deforestation and biodiversity loss. Results clearly evidenced the effectiveness of conservation regimes, with largest and most reliable deforestation decreases in both strictly protected areas or sustainable use areas, as well as potential biodiversity-increasing effects of strictly protected areas. These conservation-focused regimes, and the protection of their integrity, remain crucial for environmental conservation, as advised by many other studies (Ferreira et al., 2020; Pfaff, Robalino, Herrera, et al., 2015; Soares-Filho et al., 2010). Although indigenous, quilombola, and other communal tenure regimes in Brazil often had ambiguous effects on both deforestation and potential biodiversity, the recognition of IPLC property rights also remains crucial in ensuring synergies between IPLCs and forests/biodiversity conservation do exist, especially in the Amazon and Cerrado biomes (Baragwanath & Bayi, 2020). Given IPLC property rights are currently under threat (Begotti & Peres, 2020; Ferrante & Fearnside, 2020), future policy geared towards these regimes should note that guaranteeing environmental conservation outcomes will likely require in-depth contextual understanding of local governance conditions.

Aside from policies specific to land tenure interventions, broader options could also be developed for agricultural and environmental policies to leverage land-use incentives and decision-making in different tenure regimes for improving conservation outcomes. Most well-known policies that have been implemented in Brazil (e.g. the Forest Code, soy and beef moratoria, payments for ecosystem services (PES) schemes) have primarily targeted decreasing deforestation in private farms, meaning these policies are already well aligned with the tenure regime where most of Brazil's deforestation occurs and where most forestlands remain (**Figure 8**). As previously argued, my results suggest that stringent environmental obligations in the Amazon have likely played an important role in decreasing deforestation in private farms, corroborating reports that interventions such as the soy moratorium in the Amazon have been broadly successful (Heilmayr et al., 2020). Yet, inconclusive results on the effects of private lands on the potential diversity of species highlights broader challenges in delivering more holistic environmental conservation outcomes, and could suggest that the kinds of policies implemented in the Amazon might be successful in other biomes as well.

Research on the potential implementation of such policies in other biomes estimates that from 2021-2050, 3.6 million ha of native vegetation loss could be prevented in the Cerrado if the soy moratorium were to be extended to this biome (Soterroni et al., 2019). However, actual implementation of the forest code and the soy/beef moratoria have proven to be sluggish and steeply challenging in the Cerrado. This is in part due to inherent difficulties in monitoring fragmented savannas (Brannstrom et al., 2008). However, this is also due to difficulties to monitor and enforce regulations in the cattle supply chain (e.g. where cattle can be raised illegally in one property, and sold in another property that is deforestation “free”). In this way, the Cerrado remains vastly underrepresented in the commitments made by companies in the soy industry sector pledging to eliminate deforestation from their supply chain (Ermgassen et al., 2020). Additionally, implementation of such policies in the Cerrado is also likely at a disadvantage due to agricultural expansion from the legal Amazon already spilling over into this less regulated ecosystem (Moffette & Gibbs, 2021). Alternatively, PES schemes have been found to be successful in reducing vegetation loss and increasing regeneration in both the Amazon and Mata Atlantica (albeit the impact is often small) (Oliveira Fiorini et al., 2020; Ruggiero et al., 2019; Simonet et al., 2019). However, recent analyses suggest that in the Cerrado, implementing PES alone would likely negatively impact poor farmers. Hence, PES should only be implemented as an addition to other market exclusion mechanisms (MEM) which exclude commodity suppliers that produce in properties associated with deforestation (e.g. soy and beef moratoria) (Garrett et al., 2022). Thus, though the implementation of such regulatory and market-based policy instruments in biomes outside of Amazonia remains challenging, finding mechanisms to implement these tools may pave the road to more holistic conservation outcomes that safeguard forest and non-forest animal species alike, as well as the wellbeing of land users with titled and untitled properties alike.

Clearly, leveraging land-use incentives and decision-making for conservation is a complex task requiring heterogenous implementation across regions in Brazil. Nonetheless, findings from this dissertation emphasize the importance of the regularization of land tenure regimes for the successful implementation of various policies. In light of existing challenges in monitoring policy compliance, it is also clear these efforts must address the lack of well-defined property rights in undesignated and untitled lands across Brazil, and in particular in the Amazon and Cerrado. Likewise, publicly available property registries such as the CAR remain essential tools for the success of these monitoring efforts (Garrett et al., 2022; Heilmayr et al., 2020). Additional monitoring metrics related to biodiversity could consider taking advantage of these existing data infrastructures. Further research into the effects of specific policies on the bundles of rights of specific tenure regimes could help characterize and clarify whether any one policy is most effective in delivering conservation outcomes. Notably, privatization of land is not the only policy opportunity with possible synergies for conservation – nor most effective one. Both conservation-focused and IPLC tenure regimes provide opportunities to implement “win-win” socioenvironmental policies in Brazil. Nonetheless, it is likely that effective long-term conservation requires a combination of complementary market-based, supply-chain, and regulatory policies with robust monitoring and enforcement systems – in addition to persistent political will (Soterroni et al., 2019).

4.4 Limitations

One of the aims of this research was to follows a macroecological approach and test for effects of land tenure on environmental change at broad spatiotemporal scales in order to detect generally consistent patterns of effects at large scales. However, despite recent advances in data availability and methods, data limitations constrained the scope of the analyses conducted in this dissertation to a single-country (section **4.1**). While the lack of accessibility to spatially explicit land tenure data remains a challenge for future research, an additional challenge will involve systematizing the specific bundles of rights that are recognized/granted for different land tenure categories across countries. Only then will land tenure data become interoperable, that is, broader patterns of effects of land tenure be extended outside of the context of a single country or system, opening potential for further generalizability of these effects at regional or global scales.

Despite taking considerable measures to decrease bias in the analyses conducted based on observational data (e.g. robustness checks for creation date of both conservation regimes, as well as initial forest cover, see **2.2.1**), there remains a possibility that due to the non-random assignment of comparison groups, results might be confounded by omitted variable bias. I calculated Rosenbaum bounds to test the sensitivity of results to this possibility (sections **3.3.1, 3.4.1**), and some findings could indeed slightly change in the presence of an unobserved variable – albeit this level of sensitivity common in social science research (Keele, 2009). Besides vulnerability to omitted variable bias, a key requirement for using matching methods is finding a region of "common support" or "overlap", i.e. observations in both treatment and control groups with covariates found in the same bins used with coarsened-exact matching (CEM). The large number of land parcels in the Imaflora data compilation permitted finding sufficient observations within regions of overlap which were large enough to be able to conduct subsequent statistical analyses at most spatiotemporal scales. However, imbalance remained high in a few cases (**Tables 8,12,13**), meaning differences between treatment and control groups do not closely approximate random assignment. I accounted for these cases with high remaining imbalance by either not considering them in subsequent analysis, weighing them by balance in further syntheses tables (**Tables 6, 7**), and visually downweighing them in all figures used.

Although exact-matching using CEM overall improved the balance in the data and the robustness of estimated effects, dropping non-matched observations limits the generalizability of effects exclusively to the matched subsample of data (i.e. meaning effects estimated would be average treatment effects on the matched sample (ATM)). Thus, despite the overarching aims of this dissertation to improve the generalizability of effects of land tenure on environmental change, it is possible results may not be generalizable to discarded units of analysis that are outside of the matched covariate bounds (i.e. at different elevation, slope, travel time to nearest city, human population, and parcel areas). Future research could improve upon the generalizability of these estimates by adapting recently developed methods that increase generalizability of a given sample to a target population (Ackerman et al., 2019).

Representing species diversity by modelling species' area of habitat (AoH) enables novel assessments of biodiversity change over large spatial and temporal scales. However, it is important to note these data do not replace in-situ sampling efforts and can only represent potential biodiversity found in an area. Moreover, using these data limited the sample size of matched observations that were possible, as, in order to account for the quantification of habitat requirements as a proportion of a 1km^2 cell, I discarded all land parcels smaller than 1km^2. Thus, findings on potential biodiversity change could be improved by addressing sample size limitations, or increasing the resolution of modelled biodiversity data. Additionally, calculating other biodiversity metrics (e.g. species turnover) could also provide a more comprehensive understanding of the effects of land tenure on biodiversity change at large spatiotemporal scales.

Finally, though this dissertation uses the concept "land tenure regimes" to capture many aspects of land tenure (e.g. bundle of rights, policies that apply to certain regimes, etc.), this necessarily coarsens specific tenure aspects that are currently not possible to measure/observe. For example, this concept was used to speculate *de facto* perceptions of land tenure security, although it is known that titles do not necessarily increase tenure security, and vice versa (Place, 2009). Given the important role of land tenure security, specifically, on hypothesized mechanisms that predict how land tenure affects environmental change, future research may further disentangle these effects by taking advantage of cutting-edge data on global perceptions of land tenure security (*Prindex: Measuring Global Perceptions of Land and Property Rights*, n.d.).

In conclusion, results from this research provide novel evidence on the effects of different land tenure on environmental change at broad-spatiotemporal scales. However, current limitations in data and methods also indicate potential opportunities for future research. Such data-driven causal approaches investigating the socioeconomic drivers influencing global environmental change have increasing potential and relevance in designing sustainability solutions in the digital age.

4.5 Outlook for future research

As discussed above, further research on the effects of land tenure on global environmental change at large scales will likely require major investments in FAIR data infrastructure. Improving the accessibility and interoperability of observational data could bolster methodologies and techniques for investigating the effects of tenure on both socioeconomic and environmental outcomes, and may still improve the generalizability of effects across vast spatial and temporal scales.

Research aiming to further understand environmental change specifically in Brazil could delve into the factors that drive differences between deforestation and potential biodiversity change. Effects on potential biodiversity could be disaggregated to specific biomes or regions, or also by different species groups (e.g. effects on specialists vs. non-specialist animal species, or, effects in forest-dominant vs. non-forest dominant regions). Investigating these differences by incorporating other biodiversity metrics (e.g. species turnover) could also provide a clearer picture of anthropogenic effects on biodiversity change in Brazil.

Moreover, future research could focus on better understanding how differences in the respective property rights of different tenure regimes drive conservation effectiveness. For instance, it is not the same to compare the effects of IPLC regimes that have been granted the full bundle of rights against IPLCs that have not been granted withdrawal or alienation rights. Systematizing granted property rights across different countries' legal frameworks could enable further disentangling the effects of specific property rights in the "bundle" to better understand the mechanisms that drive the effects of different land tenure regimes on environmental change.

Furthermore, better understanding these driving mechanisms could potentially enable future research to also better understand the trade-offs and synergies between environmental and socioeconomic outcomes of different tenure interventions. While the focus of this dissertation has been on environmental outcomes, the goal of many tenure interventions across the tropics has typically been associated with improved livelihoods and wellbeing (e.g. relating to SDG target 1.4, which identifies land tenure as an essential factor in eradicating poverty). By better understanding the causal environmental and socioeconomic effects of different land tenure regimes and their property rights at both narrow and broad scales, international agreements and conventions (e.g. SDGs, CBD, Post-2020 Biodiversity Framework) as well as agricultural and environmental policies can more effectively deliver lasting, "win-win" socioenvironmental outcomes.

5. Conclusions

In this dissertation, I used interdisciplinary approaches and methods to study the influence of land tenure as a key socioeconomic driver of global environmental change. Findings from this dissertation contribute towards better scientific understanding of land tenure as a part of many complex human-environment interactions centered around land and its sustainability. Moreover, these findings can provide concrete recommendations for future research aiming to apply similarly data-driven, causal approaches, in particular in the Latin-American region. How specific tenure regimes affect deforestation and potential biodiversity is directly relevant to current environmental, agricultural, and land-ownership policies in Brazil. Overall, these findings broaden the potential of future research on land tenure across diverse governance landscapes.

On the one hand, findings from this dissertation make an important empirical contribution to the literature by substantiating and/or challenging many fundamental theories on the strong influence of different land tenure regimes on environmental change. In contrast to previous studies which are commonly geographically or thematically constrained, these findings also contribute towards building middle-range theory in this field by testing the bounds of these different effects, and identifying potentially generalizable, and context-dependent effects alike.

Specifically, I found empirical support for theories that predict that lands with poorly defined property rights – such as open-access lands or, in Brazil, undesignated/untitled public lands – increase deforestation in comparison to alternative tenure regimes. This finding was consistent across many different regional scales and temporal contexts in Brazil, indicating robust empirical support for these theories. By contrast, findings on the effects of private property rights on environmental outcomes were not consistent across all regions or periods. Rather than finding private tenure regimes had effects in one single direction, which could contribute towards building empirical consensus, my findings indicate effects of private tenure regimes depend on the spatiotemporal scale evaluated, and on the tenure alternative they are compared against (i.e. the direction of effects depends on the counterfactual). For instance, private tenure regimes were typically more effective than undesignated/untitled regimes in preventing deforestation, yet, broadly less effective than conservation or IPLC tenure regimes in preventing either deforestation or potential biodiversity decreases. However, analyses at narrower scales surprisingly indicated support for private land tenure as an effective governance tool in curbing deforestation in remote contexts such as the Amazon – albeit these effects are likely contingent upon strict environmental policies that externalize accountability to individual landowners. Thus, the effects of private tenure regimes on environmental outcomes vary across contexts, coinciding with many other mixed results in the literature. Therefore, while findings on effects of tenure regimes with poorly defined property rights may be potentially generalizable given their consistency across scales, effects of private tenure regimes may likely remain context-dependent given the variation of effects founds across scales.

Simultaneously, despite theoretical and empirical debate, I found consistent support for the effectiveness of conservation-focused tenure regimes in preventing deforestation and potential biodiversity loss across scales. These findings underscore the importance of maintaining the integrity of protected areas (both strict-protection sustainable-use areas) in guaranteeing conservation outcomes for forests and potential species diversity alike. In contrast, despite broad empirical consensus, I found mixed effects of indigenous and quilombola regimes on deforestation and potential biodiversity in Brazil, indicating tenure interventions *alone* are unlikely to ensure positive environmental conservation outcomes in these regimes. While synergies between IPLC regimes and environmental conservation may still exist, these regimes likely require engaging in in-depth contextual studies to guarantee these outcomes.

Findings also indicated the importance of differentiating effects of different tenure regimes on different natural resource systems, in this case, forests and potential diversity of animal habitats. Differentiating between effects on forests and biodiversity is also an important empirical contribution to the literature, as the estimation of effects on deforestation contextualizes these findings to a classically used proxy for environmental change, while the estimation of changes in potential species' habitats prioritizes better understanding other non-forest ecological impacts. Further understanding human impacts on changes in animal diversity is a research frontier, and estimating the effects of different land tenure regimes on potential biodiversity change represents a first step in improving this understanding at large spatiotemporal scales.

On the other hand, findings from this dissertation are relevant for environmental, agricultural, and land-centered policies and decision-making in Brazil, as well as in broader tropical regions with similar land governance challenges. Current international agreements and conventions (e.g. SDGs, CBD, and the Post-2020 Biodiversity Framework) may also benefit from concrete recommendations on the un/likely environmental impacts of tenure interventions.

Specifically, the regularization of poorly-defined property rights is likely to have positive outcomes for forests. Findings also suggest that although investments in land titling and the regularization of land tenure systems are not misguided, there are important nuances in implementing these types of interventions. For instance, land privatization may provide a better alternative than leaving lands with poorly defined property rights as they currently are. However, private tenure regimes may not necessarily be the best alternative for ensuring forest or potential biodiversity conservation outcomes. If the opportunity exists for IPLC land claims to be recognized, or for the creation of conservation-focused regimes, either of these tenure interventions are likely to have more positive conservation outcomes than private tenure regimes. Still, it should be noted that IPLCs do not claim lands everywhere, and moreover, that tenure interventions involving IPLC regimes likely require investing time and resources into understanding specific contexts to guarantee environmental conservation outcomes. Notwithstanding the above recommendations, my findings also suggest that for particularly remote contexts, policies that externalize the accountability of monitoring and enforcement to specific individuals may deem private tenure regimes as a highly effective land governance mechanism. This is especially relevant for regions such as the Cerrado and Pantanal that are as similarly remote as Amazonia, where most of the remaining forest landscapes are privately owned.

The analyses conducted here were made possible by accessing publicly available, spatially explicit data compilation on different land tenure categories in Brazil. However, research on global land tenure data findability, accessibility, and interoperability also demonstrated that crucial data gaps and needs still hinder future research in this field. Future cross-country comparisons will require the clear definition of the bundles of rights particular to each category of land tenure within a country by using shared, standardized vocabularies and metadata. Research could furthermore progress with the anonymization of properties (while maintaining key metadata on e.g., bundles of rights) and the inclusion of temporal data (e.g. date of titling/creation). Filling these gaps will require local knowledge and partnerships, building personal connections and continuing relationships with data partners, as well as open and clear communication with local stakeholders about how data is used for research. At the same time, it is crucial to acknowledge that better access and availability of spatially explicit data on the socioeconomic drivers of environmental change at large scales will also require investing in data infrastructure and management in many countries where these data aspects may not necessarily be a first priority.

Notwithstanding these data challenges, there is substantial potential for improving our understanding on the effects of socioeconomic drivers on environmental change at broad spatial and temporal scales. Future research could take advantage of rapidly growing communities using observational data for causal inference (e.g. communities in land-system, conservation, and sustainability sciences). This research could consider economic alongside environmental outcomes, and better investigate potential trade-offs involved in property-rights interventions around the world. Future research also interested in understanding the effects of tenure regimes on biodiversity could disaggregate to specific biomes in Brazil to explore what may be driving differences in effects of different tenure regimes on forests and biodiversity (e.g. differences in effects of tenure regimes could be explored in specialist and non-specialist species, or in forest-dominated or savanna-dominated regions).

In conclusion, these findings provide a better understanding of land tenure as a driver of environmental change at large spatial and temporal scales. Testing the effects of different tenure regimes in Brazil provides evidence for this relationship across many different sociopolitical and environmental contexts where the conservation of nature is under increasing pressure. Future research on the causal effects of land tenure on environmental change is contingent upon increased land-tenure data availability in order to provide key insight into socioecological challenges related to land tenure around the world.

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Appendix

Supplementary figures and tables

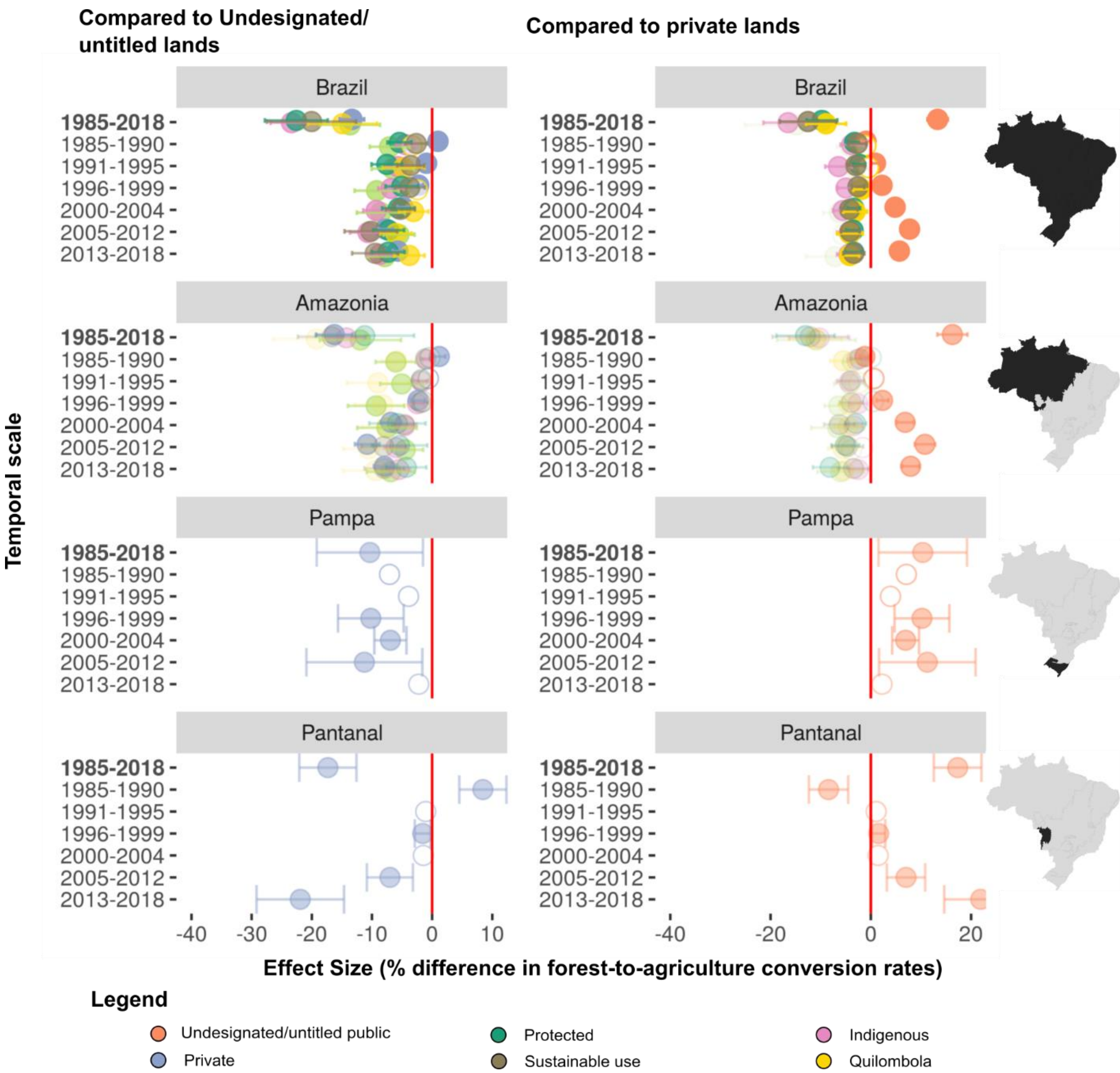

Figure 19. Effects of alternative tenure regimes on forest-to-agriculture conversion rates at different spatiotemporal scales, complementing **Fig. 10** by showing additional results for communal tenure regimes for Brazil and the Amazonia biome, and for private and undesignated/untitled regimes for Pampa and Pantanal. Circles indicate effects sizes estimated at different spatial-temporal scales, where each tenure regime was compared vis-a-vis two alternative counterfactuals: a) undesignated/untitled public lands, and b) private lands. Effects to the left of the zero line indicate a decrease in average parcel-level deforestation rate (to the right: increase). Filled circles indicate statistically significant effects ($p \leq 0.05$; non-filled: not significant); upper/lower confidence intervals are plotted to the left/right of each circle centroid. Higher transparency of filled circles indicate high levels of imbalance in the matched dataset (multivariate imbalance measure L_1). Note that tests for communal tenure had to be based on substantially fewer parcels than those for other tenure regimes, with sufficient parcels post-matching for reliable parameter estimation

only available at the Brazil-wide and Amazonia-wide scales. Similarly, the only reliable comparison possible in the Pampa and Pantanal biomes was undesignated/untitled vs. private, due to a lack of data for other regimes (and/or lack of certain tenure regimes) in these biomes.

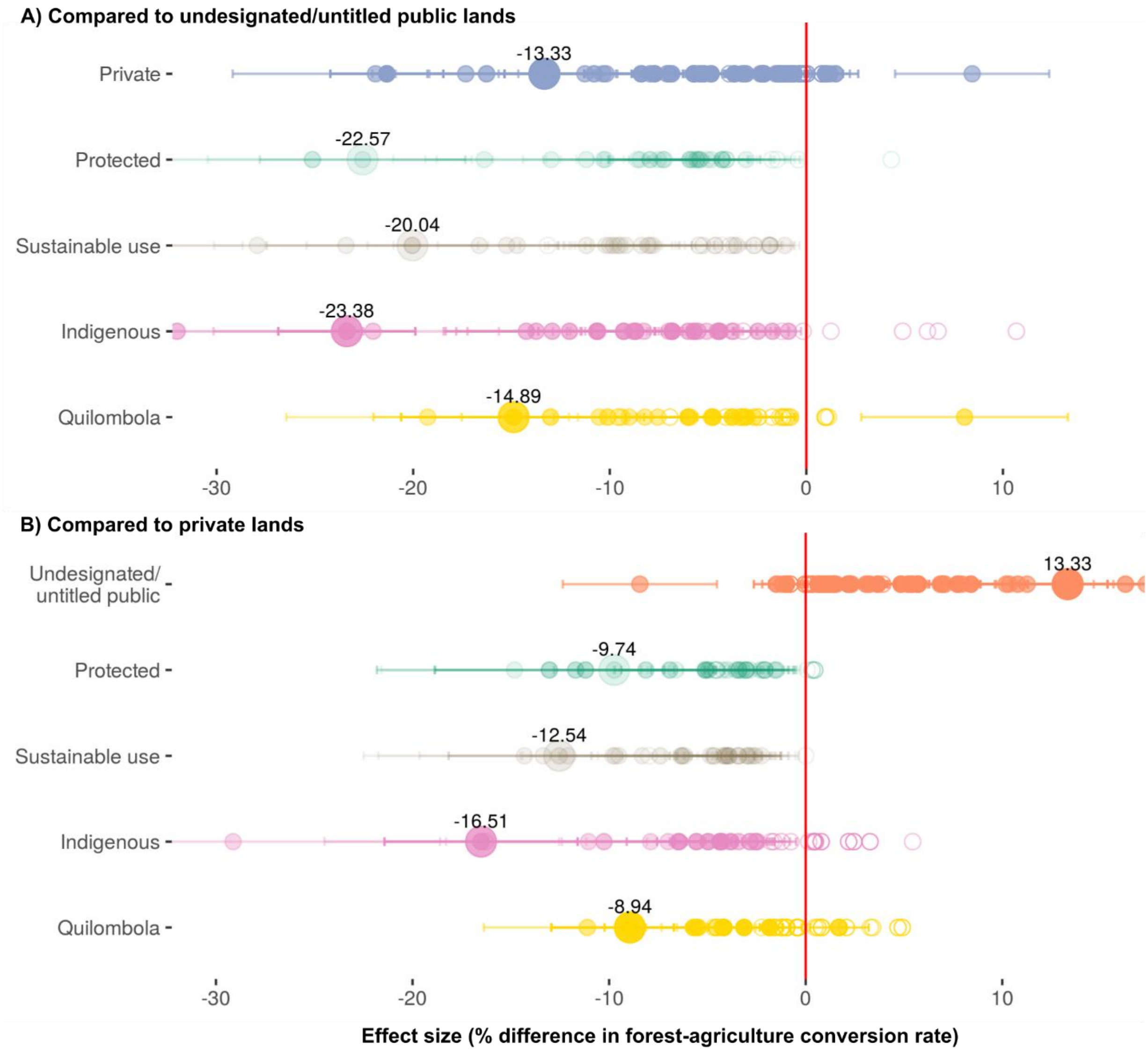

Figure 20. Robustness test of effects of alternative tenure regimes on forest-to-agriculture conversion rates in Brazil using filtered time-series data for protected and sustainable-use areas (i.e., only areas established before/during beginning of each temporal scale considered; see section 2.4.3 Sensitivity analyses). Circles indicate effects sizes estimated at different spatial-temporal scales vis-a-vis two alternative counterfactuals: A) undesignated/untitled public lands, and B) private lands. Labelled effect sizes (larger circles) report effects across Brazil over the time period 1985-2018, Effects to the left of the zero line indicate a decrease in average parcel-level deforestation rate (to the right: increase). Filled circles indicate statistically significant effects ($p \leq 0.05$; non-filled: not significant); upper/lower confidence intervals are plotted to the left/right of each circle centroid. Higher transparency of filled circles indicate high levels of imbalance in the matched dataset (multivariate imbalance measure L_1).

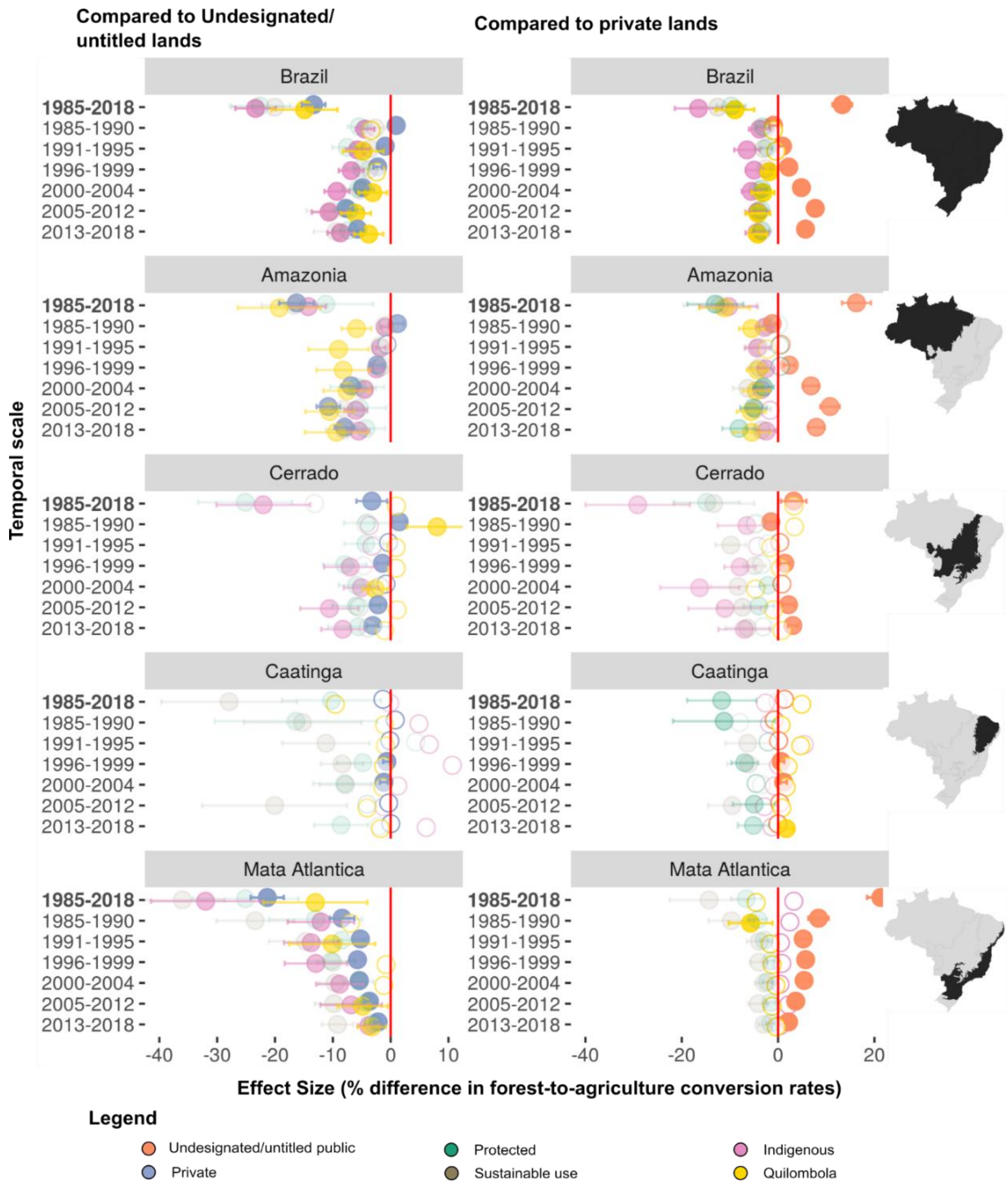

Figure 21. Spatiotemporal disaggregation of robustness test of effects of alternative tenure regimes on forest-to-agriculture conversion rates in Brazil using filtered time-series data for protected and sustainable-use areas (i.e., only areas established before/during beginning of each temporal scale considered; see section 2.2). Circles indicate effects sizes estimated at different spatial-temporal scales vis-a-vis two alternative counterfactuals: a) undesignated/untitled public lands, and b) private lands. Effects to the left of the zero line indicate a decrease in average parcel-level deforestation rate (to the right: increase). Filled circles indicate statistically significant effects ($p \leq 0.05$; non-filled: not significant), upper/lower confidence intervals are plotted to the left/right of each circle centroid. Higher transparency of filled circles indicate high levels of imbalance in the matched dataset (multivariate imbalance measure L_1).

Table 12. Model outputs for estimating effects all tenure regimes on forest conversion to agriculture rates compared to an undesignated/untitled public counterfactual. Average Marginal Effects (Effect) are reported for each specific compared tenure regime (treatment column) at different spatial and temporal scales, with recorded number of observations in matched sample (n), the standard error (SE), p-value, and lower and upper confidence intervals. Imbalance (L1) reported before (ImbBefore) and after matching (ImbAfer), and resulting improvement (ImbImprov). Note that very small numbers (4 to 19) of matched parcel data prevented reliable modelling of effects of communal tenure regimes in the Caatinga, Cerrado, and Mata Atlântica biomes, and for all tenure regimes except undesignated/untitled and private in the Pampas and Pantanal biomes.

Treatment	spatialScale	temporalScale	*n*	SE	*p*_value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	ImbImprov
Communal	Amazonia	1985-1990	914	0.017	0.000	-0.094	-0.027	-0.060	0.761	0.422	0.339
Communal	Amazonia	1985-2018	912	0.034	0.001	-0.187	-0.052	-0.119	0.761	0.428	0.334
Communal	Amazonia	1991-1995	914	0.018	0.006	-0.087	-0.015	-0.051	0.761	0.414	0.348
Communal	Amazonia	1996-1999	914	0.024	0.000	-0.140	-0.046	-0.093	0.761	0.398	0.363
Communal	Amazonia	2000-2004	914	0.025	0.003	-0.125	-0.026	-0.075	0.761	0.403	0.359
Communal	Amazonia	2005-2012	912	0.015	0.003	-0.075	-0.015	-0.045	0.761	0.432	0.329
Communal	Amazonia	2013-2018	908	0.020	0.001	-0.109	-0.029	-0.069	0.763	0.425	0.338
Communal	Brazil	1985-1990	1,148	0.017	0.000	-0.104	-0.038	-0.071	0.809	0.277	0.532
Communal	Brazil	1985-2018	1,146	0.025	0.000	-0.186	-0.086	-0.136	0.810	0.281	0.529
Communal	Brazil	1991-1995	1,148	0.018	0.000	-0.101	-0.031	-0.066	0.809	0.247	0.562
Communal	Brazil	1996-1999	1,148	0.019	0.000	-0.129	-0.056	-0.092	0.810	0.251	0.559
Communal	Brazil	2000-2004	1,148	0.020	0.000	-0.125	-0.045	-0.085	0.810	0.256	0.553
Communal	Brazil	2005-2012	1,148	0.013	0.000	-0.081	-0.029	-0.055	0.810	0.244	0.566
Communal	Brazil	2013-2018	1,146	0.015	0.000	-0.109	-0.049	-0.079	0.811	0.274	0.537
Indigenous	Amazonia	1985-1990	456	0.003	0.005	-0.016	-0.003	-0.009	0.743	0.531	0.212
Indigenous	Amazonia	1985-2018	456	0.015	0.000	-0.172	-0.112	-0.142	0.743	0.535	0.208
Indigenous	Amazonia	1991-1995	456	0.004	0.000	-0.025	-0.009	-0.017	0.743	0.531	0.212
Indigenous	Amazonia	1996-1999	456	0.005	0.000	-0.035	-0.014	-0.025	0.743	0.531	0.212
Indigenous	Amazonia	2000-2004	456	0.007	0.000	-0.058	-0.032	-0.045	0.743	0.535	0.208
Indigenous	Amazonia	2005-2012	456	0.010	0.000	-0.080	-0.040	-0.060	0.743	0.535	0.208
Indigenous	Amazonia	2013-2018	454	0.009	0.000	-0.074	-0.037	-0.055	0.743	0.533	0.210
Indigenous	Brazil	1985-1990	902	0.008	0.000	-0.060	-0.028	-0.044	0.721	0.273	0.448
Indigenous	Brazil	1985-2018	902	0.018	0.000	-0.269	-0.199	-0.234	0.722	0.286	0.436
Indigenous	Brazil	1991-1995	902	0.010	0.000	-0.077	-0.037	-0.057	0.721	0.273	0.448
Indigenous	Brazil	1996-1999	902	0.011	0.000	-0.090	-0.047	-0.068	0.721	0.273	0.448
Indigenous	Brazil	2000-2004	900	0.011	0.000	-0.114	-0.071	-0.093	0.722	0.282	0.440
Indigenous	Brazil	2005-2012	902	0.015	0.000	-0.136	-0.077	-0.107	0.723	0.282	0.441
Indigenous	Brazil	2013-2018	896	0.011	0.000	-0.109	-0.064	-0.087	0.724	0.277	0.447
Indigenous	Caatinga	1985-1990	44	0.038	0.193	-0.025	0.123	0.049	0.892	0.636	0.256
Indigenous	Caatinga	1985-2018	44	0.115	0.989	-0.227	0.224	-0.002	0.892	0.682	0.210

Treatment	**spatialScale**	**temporalScale**	***n***	**SE**	***p*_value**	**lower_ci**	**upper_ci**	**Effect**	**ImbBefore**	**ImbAfter**	**ImbImprov**
Indigenous	Caatinga	1991-1995	44	0.080	0.400	-0.089	0.223	0.067	0.892	0.636	0.256
Indigenous	Caatinga	1996-1999	44	0.056	0.058	-0.004	0.218	0.107	0.892	0.636	0.256
Indigenous	Caatinga	2000-2004	44	0.070	0.858	-0.124	0.149	0.013	0.892	0.682	0.210
Indigenous	Caatinga	2005-2012	44	0.110	0.911	-0.228	0.203	-0.012	0.892	0.727	0.164
Indigenous	Caatinga	2013-2018	46	0.065	0.346	-0.066	0.190	0.062	0.891	0.652	0.239
Indigenous	Cerrado	1985-1990	80	0.019	0.056	-0.075	0.001	-0.037	0.871	0.700	0.171
Indigenous	Cerrado	1985-2018	80	0.041	0.000	-0.302	-0.139	-0.220	0.882	0.650	0.232
Indigenous	Cerrado	1991-1995	82	0.018	0.080	-0.068	0.004	-0.032	0.871	0.659	0.212
Indigenous	Cerrado	1996-1999	80	0.023	0.002	-0.115	-0.025	-0.070	0.882	0.650	0.232
Indigenous	Cerrado	2000-2004	80	0.015	0.001	-0.081	-0.021	-0.051	0.882	0.675	0.207
Indigenous	Cerrado	2005-2012	80	0.026	0.000	-0.156	-0.055	-0.106	0.882	0.650	0.232
Indigenous	Cerrado	2013-2018	80	0.019	0.000	-0.120	-0.045	-0.083	0.883	0.650	0.233
Indigenous	Mata Atlântica	1985-1990	194	0.029	0.000	-0.178	-0.063	-0.120	0.772	0.474	0.298
Indigenous	Mata Atlântica	1985-2018	194	0.048	0.000	-0.415	-0.225	-0.320	0.773	0.536	0.237
Indigenous	Mata Atlântica	1991-1995	194	0.024	0.000	-0.185	-0.090	-0.137	0.772	0.536	0.236
Indigenous	Mata Atlântica	1996-1999	194	0.028	0.000	-0.183	-0.075	-0.129	0.773	0.526	0.247
Indigenous	Mata Atlântica	2000-2004	194	0.021	0.000	-0.129	-0.047	-0.088	0.773	0.536	0.237
Indigenous	Mata Atlântica	2005-2012	194	0.027	0.012	-0.122	-0.015	-0.068	0.773	0.526	0.247
Indigenous	Mata Atlântica	2013-2018	194	0.010	0.000	-0.057	-0.018	-0.038	0.774	0.536	0.238
Private	Amazonia	1985-1990	8,066	0.005	0.024	0.002	0.022	0.012	0.638	0.353	0.285
Private	Amazonia	1985-2018	8,064	0.015	0.000	-0.193	-0.133	-0.163	0.641	0.353	0.288
Private	Amazonia	1991-1995	8,062	0.006	0.323	-0.019	0.006	-0.006	0.640	0.357	0.283
Private	Amazonia	1996-1999	8,060	0.006	0.000	-0.035	-0.012	-0.023	0.641	0.359	0.282
Private	Amazonia	2000-2004	8,064	0.009	0.000	-0.086	-0.051	-0.068	0.641	0.354	0.287
Private	Amazonia	2005-2012	8,062	0.010	0.000	-0.128	-0.087	-0.108	0.641	0.353	0.288
Private	Amazonia	2013-2018	8,060	0.009	0.000	-0.097	-0.062	-0.079	0.641	0.355	0.286
Private	Brazil	1985-1990	34,212	0.005	0.032	0.001	0.019	0.010	0.663	0.123	0.540
Private	Brazil	1985-2018	34,216	0.010	0.000	-0.154	-0.113	-0.133	0.663	0.126	0.537
Private	Brazil	1991-1995	34,216	0.004	0.029	-0.017	-0.001	-0.009	0.663	0.125	0.538
Private	Brazil	1996-1999	34,216	0.004	0.000	-0.030	-0.015	-0.022	0.663	0.126	0.537
Private	Brazil	2000-2004	34,214	0.005	0.000	-0.058	-0.039	-0.048	0.663	0.125	0.538
Private	Brazil	2005-2012	34,218	0.006	0.000	-0.089	-0.066	-0.077	0.663	0.128	0.535
Private	Brazil	2013-2018	34,214	0.004	0.000	-0.066	-0.048	-0.057	0.662	0.130	0.533
Private	Caatinga	1985-1990	10,020	0.006	0.214	-0.005	0.020	0.008	0.714	0.142	0.572
Private	Caatinga	1985-2018	10,020	0.009	0.134	-0.031	0.004	-0.013	0.716	0.137	0.579
Private	Caatinga	1991-1995	10,024	0.004	0.765	-0.010	0.007	-0.001	0.715	0.140	0.575

Treatment	**spatialScale**	**temporalScale**	***n***	**SE**	***p*_value**	**lower_ci**	**upper_ci**	**Effect**	**ImbBefore**	**ImbAfter**	**ImbImprov**
Private	Caatinga	1996-1999	10,024	0.003	0.043	-0.013	0.000	-0.007	0.715	0.138	0.578
Private	Caatinga	2000-2004	10,022	0.003	0.001	-0.018	-0.005	-0.012	0.716	0.137	0.579
Private	Caatinga	2005-2012	10,022	0.005	0.510	-0.014	0.007	-0.003	0.716	0.135	0.580
Private	Caatinga	2013-2018	10,022	0.003	0.932	-0.007	0.007	0.000	0.715	0.135	0.580
Private	Cerrado	1985-1990	9,670	0.006	0.012	0.003	0.026	0.015	0.718	0.256	0.462
Private	Cerrado	1985-2018	9,672	0.014	0.017	-0.059	-0.006	-0.032	0.718	0.261	0.457
Private	Cerrado	1991-1995	9,670	0.005	0.510	-0.014	0.007	-0.004	0.718	0.258	0.460
Private	Cerrado	1996-1999	9,670	0.005	0.005	-0.024	-0.004	-0.014	0.718	0.258	0.460
Private	Cerrado	2000-2004	9,672	0.006	0.179	-0.020	0.004	-0.008	0.718	0.259	0.460
Private	Cerrado	2005-2012	9,672	0.006	0.001	-0.034	-0.009	-0.022	0.719	0.261	0.458
Private	Cerrado	2013-2018	9,672	0.005	0.000	-0.041	-0.021	-0.031	0.719	0.261	0.458
Private	Mata Atlântica	1985-1990	5,130	0.011	0.000	-0.105	-0.063	-0.084	0.744	0.160	0.584
Private	Mata Atlântica	1985-2018	5,134	0.015	0.000	-0.242	-0.185	-0.213	0.743	0.113	0.630
Private	Mata Atlântica	1991-1995	5,130	0.007	0.000	-0.066	-0.039	-0.052	0.744	0.161	0.583
Private	Mata Atlântica	1996-1999	5,132	0.006	0.000	-0.069	-0.046	-0.057	0.743	0.141	0.602
Private	Mata Atlântica	2000-2004	5,132	0.006	0.000	-0.067	-0.042	-0.054	0.743	0.142	0.601
Private	Mata Atlântica	2005-2012	5,134	0.005	0.000	-0.046	-0.027	-0.037	0.743	0.109	0.634
Private	Mata Atlântica	2013-2018	5,134	0.003	0.000	-0.028	-0.015	-0.022	0.742	0.113	0.630
Private	Pampa	1985-1990	404	0.041	0.082	-0.151	0.009	-0.071	0.843	0.391	0.452
Private	Pampa	1985-2018	404	0.045	0.022	-0.192	-0.015	-0.104	0.843	0.465	0.378
Private	Pampa	1991-1995	404	0.029	0.175	-0.096	0.017	-0.039	0.843	0.416	0.427
Private	Pampa	1996-1999	404	0.028	0.000	-0.157	-0.047	-0.102	0.843	0.436	0.407
Private	Pampa	2000-2004	404	0.014	0.000	-0.096	-0.043	-0.069	0.843	0.431	0.412
Private	Pampa	2005-2012	404	0.049	0.022	-0.209	-0.016	-0.113	0.843	0.455	0.387
Private	Pampa	2013-2018	404	0.012	0.074	-0.047	0.002	-0.022	0.843	0.460	0.382
Private	Pantanal	1985-1990	260	0.020	0.000	0.045	0.124	0.084	0.695	0.462	0.233
Private	Pantanal	1985-2018	262	0.024	0.000	-0.221	-0.126	-0.173	0.696	0.458	0.238
Private	Pantanal	1991-1995	260	0.010	0.282	-0.030	0.009	-0.011	0.695	0.462	0.233
Private	Pantanal	1996-1999	262	0.007	0.020	-0.029	-0.003	-0.016	0.695	0.458	0.237
Private	Pantanal	2000-2004	262	0.012	0.220	-0.038	0.009	-0.015	0.695	0.466	0.230
Private	Pantanal	2005-2012	262	0.020	0.000	-0.108	-0.032	-0.070	0.696	0.466	0.230
Private	Pantanal	2013-2018	262	0.037	0.000	-0.292	-0.147	-0.219	0.696	0.450	0.245
Protected	Amazonia	1985-1990	108	0.005	0.438	-0.014	0.006	-0.004	0.896	0.611	0.285
Protected	Amazonia	1985-2018	108	0.042	0.007	-0.194	-0.030	-0.112	0.896	0.611	0.285
Protected	Amazonia	1991-1995	108	0.010	0.071	-0.037	0.002	-0.018	0.896	0.611	0.285
Protected	Amazonia	1996-1999	108	0.011	0.173	-0.037	0.007	-0.015	0.896	0.611	0.285

Treatment	**spatialScale**	**temporalScale**	***n***	**SE**	***p*_value**	**lower_ci**	**upper_ci**	**Effect**	**ImbBefore**	**ImbAfter**	**ImbImprov**
Protected	Amazonia	2000-2004	108	0.024	0.016	-0.105	-0.011	-0.058	0.896	0.611	0.285
Protected	Amazonia	2005-2012	108	0.024	0.022	-0.100	-0.008	-0.054	0.896	0.611	0.285
Protected	Amazonia	2013-2018	110	0.017	0.012	-0.076	-0.009	-0.043	0.896	0.618	0.278
Protected	Brazil	1985-1990	748	0.010	0.000	-0.074	-0.034	-0.054	0.724	0.283	0.440
Protected	Brazil	1985-2018	740	0.027	0.000	-0.278	-0.173	-0.226	0.728	0.297	0.431
Protected	Brazil	1991-1995	742	0.013	0.000	-0.101	-0.050	-0.075	0.726	0.280	0.446
Protected	Brazil	1996-1999	740	0.014	0.000	-0.078	-0.023	-0.050	0.728	0.292	0.436
Protected	Brazil	2000-2004	740	0.014	0.000	-0.083	-0.029	-0.056	0.728	0.297	0.431
Protected	Brazil	2005-2012	738	0.013	0.000	-0.098	-0.046	-0.072	0.729	0.309	0.420
Protected	Brazil	2013-2018	736	0.014	0.000	-0.100	-0.046	-0.073	0.730	0.318	0.412
Protected	Caatinga	1985-1990	52	0.072	0.022	-0.305	-0.023	-0.164	0.855	0.615	0.240
Protected	Caatinga	1985-2018	52	0.044	0.019	-0.188	-0.017	-0.102	0.856	0.538	0.318
Protected	Caatinga	1991-1995	52	0.034	0.197	-0.022	0.109	0.043	0.855	0.577	0.278
Protected	Caatinga	1996-1999	52	0.023	0.034	-0.093	-0.004	-0.048	0.856	0.538	0.317
Protected	Caatinga	2000-2004	52	0.028	0.004	-0.134	-0.025	-0.080	0.856	0.500	0.356
Protected	Caatinga	2005-2012	52	0.027	0.127	-0.093	0.012	-0.041	0.856	0.500	0.356
Protected	Caatinga	2013-2018	52	0.024	0.000	-0.132	-0.039	-0.086	0.856	0.462	0.395
Protected	Cerrado	1985-1990	118	0.020	0.044	-0.081	-0.001	-0.041	0.899	0.644	0.254
Protected	Cerrado	1985-2018	116	0.041	0.000	-0.333	-0.170	-0.251	0.900	0.638	0.262
Protected	Cerrado	1991-1995	118	0.019	0.023	-0.080	-0.006	-0.043	0.899	0.644	0.254
Protected	Cerrado	1996-1999	118	0.019	0.000	-0.117	-0.042	-0.079	0.899	0.627	0.272
Protected	Cerrado	2000-2004	116	0.016	0.000	-0.090	-0.028	-0.059	0.900	0.655	0.245
Protected	Cerrado	2005-2012	116	0.021	0.005	-0.101	-0.018	-0.059	0.900	0.638	0.262
Protected	Cerrado	2013-2018	112	0.027	0.039	-0.108	-0.003	-0.056	0.900	0.625	0.275
Protected	Mata Atlântica	1985-1990	328	0.041	0.002	-0.210	-0.050	-0.130	0.709	0.500	0.209
Protected	Mata Atlântica	1985-2018	326	0.047	0.000	-0.343	-0.159	-0.251	0.709	0.503	0.206
Protected	Mata Atlântica	1991-1995	328	0.021	0.000	-0.125	-0.044	-0.085	0.712	0.494	0.218
Protected	Mata Atlântica	1996-1999	328	0.021	0.000	-0.144	-0.062	-0.103	0.709	0.500	0.209
Protected	Mata Atlântica	2000-2004	328	0.013	0.000	-0.080	-0.030	-0.055	0.709	0.494	0.215
Protected	Mata Atlântica	2005-2012	326	0.009	0.000	-0.061	-0.024	-0.043	0.710	0.509	0.200
Protected	Mata Atlântica	2013-2018	326	0.007	0.000	-0.045	-0.016	-0.031	0.710	0.521	0.189
Quilombola	Amazonia	1985-1990	230	0.013	0.000	-0.085	-0.033	-0.059	0.755	0.687	0.068
Quilombola	Amazonia	1985-2018	230	0.037	0.000	-0.264	-0.121	-0.193	0.755	0.696	0.060
Quilombola	Amazonia	1991-1995	230	0.027	0.001	-0.142	-0.038	-0.090	0.755	0.678	0.077
Quilombola	Amazonia	1996-1999	230	0.024	0.001	-0.128	-0.036	-0.082	0.755	0.687	0.068
Quilombola	Amazonia	2000-2004	230	0.021	0.000	-0.116	-0.035	-0.075	0.755	0.687	0.068

Treatment	**spatialScale**	**temporalScale**	***n***	**SE**	***p*_value**	**lower_ci**	**upper_ci**	**Effect**	**ImbBefore**	**ImbAfter**	**ImbImprov**
Quilombola	Amazonia	2005-2012	230	0.022	0.000	-0.148	-0.063	-0.105	0.755	0.687	0.068
Quilombola	Amazonia	2013-2018	230	0.028	0.001	-0.148	-0.039	-0.094	0.756	0.687	0.069
Quilombola	Brazil	1985-1990	636	0.017	0.054	-0.068	0.001	-0.033	0.688	0.321	0.367
Quilombola	Brazil	1985-2018	634	0.029	0.000	-0.206	-0.092	-0.149	0.695	0.322	0.373
Quilombola	Brazil	1991-1995	630	0.018	0.008	-0.083	-0.012	-0.047	0.688	0.330	0.358
Quilombola	Brazil	1996-1999	632	0.013	0.059	-0.049	0.001	-0.024	0.688	0.335	0.353
Quilombola	Brazil	2000-2004	632	0.013	0.015	-0.056	-0.006	-0.031	0.692	0.323	0.370
Quilombola	Brazil	2005-2012	632	0.013	0.000	-0.086	-0.034	-0.060	0.695	0.313	0.382
Quilombola	Brazil	2013-2018	632	0.013	0.003	-0.063	-0.013	-0.038	0.698	0.323	0.375
Quilombola	Caatinga	1985-1990	98	0.035	0.765	-0.080	0.059	-0.011	0.778	0.612	0.166
Quilombola	Caatinga	1985-2018	98	0.049	0.051	-0.192	0.001	-0.096	0.778	0.449	0.329
Quilombola	Caatinga	1991-1995	98	0.021	0.678	-0.051	0.033	-0.009	0.778	0.592	0.187
Quilombola	Caatinga	1996-1999	98	0.019	0.528	-0.050	0.026	-0.012	0.778	0.490	0.288
Quilombola	Caatinga	2000-2004	96	0.029	0.642	-0.071	0.043	-0.014	0.778	0.563	0.216
Quilombola	Caatinga	2005-2012	96	0.028	0.139	-0.095	0.013	-0.041	0.778	0.542	0.236
Quilombola	Caatinga	2013-2018	96	0.020	0.378	-0.056	0.021	-0.017	0.778	0.500	0.278
Quilombola	Cerrado	1985-1990	82	0.027	0.003	0.028	0.133	0.081	0.834	0.512	0.322
Quilombola	Cerrado	1985-2018	82	0.035	0.775	-0.059	0.080	0.010	0.835	0.537	0.298
Quilombola	Cerrado	1991-1995	82	0.012	0.435	-0.014	0.033	0.010	0.834	0.512	0.322
Quilombola	Cerrado	1996-1999	82	0.009	0.306	-0.009	0.028	0.010	0.834	0.512	0.322
Quilombola	Cerrado	2000-2004	82	0.011	0.012	-0.047	-0.006	-0.027	0.835	0.512	0.322
Quilombola	Cerrado	2005-2012	82	0.014	0.404	-0.015	0.038	0.011	0.835	0.585	0.250
Quilombola	Cerrado	2013-2018	82	0.011	0.351	-0.031	0.011	-0.010	0.835	0.585	0.250
Quilombola	Mata Atlântica	1985-1990	148	0.047	0.135	-0.161	0.022	-0.069	0.730	0.527	0.203
Quilombola	Mata Atlântica	1985-2018	142	0.046	0.005	-0.220	-0.040	-0.130	0.732	0.493	0.239
Quilombola	Mata Atlântica	1991-1995	146	0.038	0.008	-0.175	-0.027	-0.101	0.730	0.521	0.210
Quilombola	Mata Atlântica	1996-1999	144	0.016	0.616	-0.039	0.023	-0.008	0.731	0.514	0.217
Quilombola	Mata Atlântica	2000-2004	144	0.016	0.470	-0.043	0.020	-0.012	0.732	0.486	0.246
Quilombola	Mata Atlântica	2005-2012	138	0.022	0.030	-0.092	-0.005	-0.048	0.736	0.493	0.243
Quilombola	Mata Atlântica	2013-2018	134	0.013	0.010	-0.057	-0.008	-0.032	0.737	0.582	0.155
Sustainable use	Amazonia	1985-1990	246	0.004	0.004	-0.019	-0.003	-0.011	0.798	0.618	0.180
Sustainable use	Amazonia	1985-2018	246	0.029	0.000	-0.223	-0.110	-0.166	0.798	0.618	0.180
Sustainable use	Amazonia	1991-1995	246	0.006	0.004	-0.031	-0.006	-0.019	0.798	0.618	0.180
Sustainable use	Amazonia	1996-1999	246	0.006	0.003	-0.031	-0.006	-0.019	0.798	0.618	0.180
Sustainable use	Amazonia	2000-2004	246	0.010	0.000	-0.067	-0.026	-0.046	0.798	0.618	0.180
Sustainable use	Amazonia	2005-2012	246	0.019	0.000	-0.118	-0.043	-0.081	0.798	0.618	0.180

Treatment	spatialScale	temporalScale	*n*	SE	*p*_value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	ImbImprov
Sustainable use	Amazonia	2013-2018	246	0.017	0.000	-0.113	-0.046	-0.079	0.798	0.626	0.172
Sustainable use	Brazil	1985-1990	958	0.009	0.005	-0.045	-0.008	-0.026	0.673	0.347	0.326
Sustainable use	Brazil	1985-2018	960	0.038	0.000	-0.275	-0.126	-0.200	0.673	0.331	0.342
Sustainable use	Brazil	1991-1995	958	0.011	0.002	-0.058	-0.013	-0.036	0.673	0.336	0.337
Sustainable use	Brazil	1996-1999	960	0.013	0.003	-0.062	-0.012	-0.037	0.673	0.329	0.344
Sustainable use	Brazil	2000-2004	960	0.012	0.000	-0.077	-0.028	-0.053	0.673	0.331	0.342
Sustainable use	Brazil	2005-2012	958	0.022	0.000	-0.146	-0.058	-0.102	0.673	0.336	0.337
Sustainable use	Brazil	2013-2018	956	0.020	0.000	-0.133	-0.056	-0.095	0.673	0.379	0.294
Sustainable use	Caatinga	1985-1990	78	0.052	0.003	-0.254	-0.051	-0.152	0.818	0.308	0.511
Sustainable use	Caatinga	1985-2018	80	0.060	0.000	-0.397	-0.162	-0.279	0.818	0.100	0.718
Sustainable use	Caatinga	1991-1995	78	0.039	0.004	-0.188	-0.036	-0.112	0.818	0.333	0.485
Sustainable use	Caatinga	1996-1999	78	0.019	0.000	-0.121	-0.047	-0.084	0.818	0.359	0.459
Sustainable use	Caatinga	2000-2004	80	0.023	0.001	-0.123	-0.033	-0.078	0.818	0.100	0.718
Sustainable use	Caatinga	2005-2012	78	0.064	0.002	-0.326	-0.075	-0.201	0.818	0.333	0.485
Sustainable use	Caatinga	2013-2018	78	0.025	0.445	-0.069	0.030	-0.019	0.818	0.256	0.562
Sustainable use	Cerrado	1985-1990	88	0.050	0.387	-0.141	0.055	-0.043	0.868	0.545	0.323
Sustainable use	Cerrado	1985-2018	88	0.097	0.173	-0.321	0.058	-0.132	0.868	0.523	0.346
Sustainable use	Cerrado	1991-1995	88	0.018	0.137	-0.062	0.008	-0.027	0.868	0.545	0.323
Sustainable use	Cerrado	1996-1999	90	0.037	0.204	-0.119	0.025	-0.047	0.868	0.533	0.335
Sustainable use	Cerrado	2000-2004	90	0.048	0.701	-0.113	0.076	-0.018	0.868	0.533	0.335
Sustainable use	Cerrado	2005-2012	86	0.054	0.312	-0.160	0.051	-0.055	0.869	0.535	0.334
Sustainable use	Cerrado	2013-2018	86	0.041	0.351	-0.120	0.043	-0.039	0.870	0.535	0.335
Sustainable use	Mata Atlântica	1985-1990	406	0.034	0.000	-0.301	-0.167	-0.234	0.711	0.424	0.287
Sustainable use	Mata Atlântica	1985-2018	406	0.037	0.000	-0.434	-0.287	-0.360	0.710	0.414	0.297
Sustainable use	Mata Atlântica	1991-1995	406	0.032	0.000	-0.210	-0.084	-0.147	0.711	0.424	0.287
Sustainable use	Mata Atlântica	1996-1999	406	0.014	0.000	-0.127	-0.073	-0.100	0.711	0.414	0.297
Sustainable use	Mata Atlântica	2000-2004	406	0.014	0.000	-0.124	-0.068	-0.096	0.710	0.414	0.297
Sustainable use	Mata Atlântica	2005-2012	404	0.017	0.000	-0.131	-0.065	-0.098	0.711	0.441	0.270
Sustainable use	Mata Atlântica	2013-2018	404	0.014	0.000	-0.119	-0.064	-0.092	0.711	0.460	0.250
Robustness check: protected areas and sustainable-use areas filtered by known year of creation											
Protected	Amazonia	1991-1995	52	0.020	0.186	-0.064	0.012	-0.026	1.000	1.000	0.000
Protected	Amazonia	1996-1999	62	0.030	0.191	-0.097	0.019	-0.039	1.000	1.000	0.000
Protected	Amazonia	2000-2004	76	0.029	0.002	-0.147	-0.034	-0.090	1.000	1.000	0.000
Protected	Amazonia	2005-2012	86	0.032	0.008	-0.146	-0.022	-0.084	1.000	1.000	0.000
Protected	Amazonia	2013-2018	100	0.018	0.011	-0.080	-0.010	-0.045	1.000	1.000	0.000
Protected	Brazil	1985-1990	196	0.021	0.016	-0.093	-0.009	-0.051	1.000	1.000	0.000

Treatment	spatialScale	temporalScale	*n*	SE	*p*_value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	ImbImprov
Protected	Brazil	1985-2018	302	0.033	0.000	-0.252	-0.121	-0.187	1.000	0.993	0.006
Protected	Brazil	1991-1995	302	0.017	0.004	-0.082	-0.016	-0.049	1.000	0.993	0.006
Protected	Brazil	1996-1999	338	0.012	0.004	-0.059	-0.011	-0.035	1.000	0.994	0.006
Protected	Brazil	2000-2004	416	0.015	0.000	-0.095	-0.035	-0.065	1.000	0.995	0.005
Protected	Brazil	2005-2012	540	0.015	0.000	-0.102	-0.045	-0.074	0.999	0.993	0.007
Protected	Brazil	2013-2018	704	0.014	0.000	-0.104	-0.050	-0.077	0.999	0.991	0.008
Protected	Caatinga	2013-2018	52	0.024	0.000	-0.132	-0.039	-0.086	1.000	1.000	0.000
Protected	Cerrado	1985-2018	42	0.049	0.000	-0.335	-0.142	-0.238	1.000	1.000	0.000
Protected	Cerrado	1991-1995	42	0.037	0.137	-0.128	0.018	-0.055	1.000	1.000	0.000
Protected	Cerrado	1996-1999	46	0.023	0.000	-0.132	-0.043	-0.088	1.000	1.000	0.000
Protected	Cerrado	2000-2004	68	0.020	0.001	-0.108	-0.029	-0.069	1.000	1.000	0.000
Protected	Cerrado	2005-2012	96	0.022	0.010	-0.102	-0.014	-0.058	1.000	0.958	0.042
Protected	Cerrado	2013-2018	112	0.027	0.039	-0.108	-0.003	-0.056	1.000	1.000	0.000
Protected	Mata Atlântica	1985-1990	66	0.030	0.006	-0.144	-0.025	-0.084	1.000	1.000	0.000
Protected	Mata Atlântica	1985-2018	136	0.062	0.000	-0.468	-0.227	-0.348	1.000	1.000	0.000
Protected	Mata Atlântica	1991-1995	136	0.045	0.000	-0.263	-0.087	-0.175	1.000	1.000	0.000
Protected	Mata Atlântica	1996-1999	150	0.027	0.000	-0.165	-0.058	-0.111	1.000	1.000	0.000
Protected	Mata Atlântica	2000-2004	170	0.015	0.000	-0.101	-0.043	-0.072	1.000	1.000	0.000
Protected	Mata Atlântica	2005-2012	222	0.014	0.000	-0.093	-0.036	-0.064	1.000	1.000	0.000
Protected	Mata Atlântica	2013-2018	312	0.008	0.000	-0.046	-0.016	-0.031	1.000	1.000	0.000
Sustainable use	Amazonia	1996-1999	90	0.018	0.415	-0.050	0.021	-0.015	1.000	1.000	0.000
Sustainable use	Amazonia	2000-2004	112	0.021	0.001	-0.111	-0.028	-0.070	1.000	1.000	0.000
Sustainable use	Amazonia	2005-2012	200	0.023	0.000	-0.139	-0.050	-0.094	1.000	1.000	0.000
Sustainable use	Amazonia	2013-2018	238	0.019	0.000	-0.123	-0.050	-0.086	1.000	1.000	0.000
Sustainable use	Brazil	1985-1990	54	0.009	0.000	-0.118	-0.083	-0.101	1.000	1.000	0.000
Sustainable use	Brazil	1985-2018	112	0.016	0.000	-0.209	-0.145	-0.177	1.000	0.982	0.018
Sustainable use	Brazil	1991-1995	112	0.035	0.016	-0.155	-0.016	-0.086	1.000	0.982	0.018
Sustainable use	Brazil	1996-1999	190	0.026	0.071	-0.097	0.004	-0.046	1.000	1.000	0.000
Sustainable use	Brazil	2000-2004	276	0.018	0.003	-0.089	-0.018	-0.054	1.000	1.000	0.000
Sustainable use	Brazil	2005-2012	454	0.023	0.000	-0.163	-0.074	-0.119	0.999	0.996	0.004
Sustainable use	Brazil	2013-2018	916	0.020	0.000	-0.134	-0.057	-0.095	0.999	0.996	0.003
Sustainable use	Caatinga	2013-2018	72	0.027	0.433	-0.073	0.031	-0.021	1.000	1.000	0.000
Sustainable use	Cerrado	2005-2012	50	0.036	0.745	-0.082	0.058	-0.012	1.000	0.920	0.080
Sustainable use	Cerrado	2013-2018	86	0.041	0.351	-0.120	0.043	-0.039	1.000	0.953	0.046
Sustainable use	Mata Atlântica	1985-2018	46	0.049	0.000	-0.516	-0.324	-0.420	1.000	1.000	0.000
Sustainable use	Mata Atlântica	1991-1995	46	0.030	0.000	-0.236	-0.116	-0.176	1.000	1.000	0.000

Treatment	spatialScale	temporalScale	*n*	SE	*p*_value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	ImbImprov
Sustainable use	Mata Atlântica	1996-1999	58	0.028	0.000	-0.181	-0.073	-0.127	1.000	1.000	0.000
Sustainable use	Mata Atlântica	2000-2004	78	0.018	0.000	-0.190	-0.118	-0.154	1.000	1.000	0.000
Sustainable use	Mata Atlântica	2005-2012	116	0.018	0.000	-0.138	-0.066	-0.102	1.000	0.983	0.017
Sustainable use	Mata Atlântica	2013-2018	388	0.014	0.000	-0.121	-0.065	-0.093	1.000	0.995	0.005

Table 13. Model outputs for estimating effects all tenure regimes on forest conversion to agriculture rates, compared to a private-lands counterfactual. Average Marginal Effects (Effect) are reported for each specific compared tenure regime (treatment column) at different spatial and temporal scales, with recorded number of observations in matched sample (n), the standard error (SE), p-value, and lower and upper confidence intervals. Imbalance (L1) reported before (ImbBefore) and after matching (ImbAfer), and resulting improvement (ImbImprov). Note that very small numbers (4 to 28) of matched parcel data prevented reliable modelling of effects of communal tenure regimes in the Caatinga, Cerrado, and Mata Atlântica biomes, and for all tenure regimes except undesignated/untitled and private in the Pampas and Pantanal biomes.

Treatment	**spatialScale**	**temporalScale**	***n***	**SE**	***p*_value**	**lower_ci**	**upper_ci**	**Effect**	**ImbBefore**	**ImbAfter**	**ImbImprov**
Communal	Amazonia	1985-1990	1,462	0.010	0.000	-0.057	-0.016	-0.036	0.730	0.599	0.131
Communal	Amazonia	1985-2018	1,462	0.040	0.007	-0.187	-0.030	-0.109	0.732	0.595	0.137
Communal	Amazonia	1991-1995	1,462	0.016	0.010	-0.074	-0.010	-0.042	0.731	0.599	0.131
Communal	Amazonia	1996-1999	1,462	0.014	0.000	-0.092	-0.036	-0.064	0.732	0.596	0.135
Communal	Amazonia	2000-2004	1,462	0.027	0.014	-0.119	-0.013	-0.066	0.732	0.595	0.137
Communal	Amazonia	2005-2012	1,462	0.022	0.083	-0.082	0.005	-0.039	0.732	0.596	0.135
Communal	Amazonia	2013-2018	1,462	0.024	0.012	-0.107	-0.013	-0.060	0.732	0.598	0.134
Communal	Brazil	1985-1990	1,522	0.014	0.004	-0.068	-0.013	-0.041	0.882	0.645	0.237
Communal	Brazil	1985-2018	1,522	0.052	0.004	-0.251	-0.047	-0.149	0.882	0.645	0.237
Communal	Brazil	1991-1995	1,522	0.014	0.024	-0.058	-0.004	-0.031	0.882	0.644	0.239
Communal	Brazil	1996-1999	1,522	0.016	0.142	-0.055	0.008	-0.024	0.882	0.644	0.239
Communal	Brazil	2000-2004	1,522	0.017	0.000	-0.095	-0.030	-0.062	0.882	0.643	0.240
Communal	Brazil	2005-2012	1,522	0.031	0.076	-0.117	0.006	-0.055	0.882	0.645	0.237
Communal	Brazil	2013-2018	1,522	0.030	0.016	-0.129	-0.013	-0.071	0.882	0.647	0.236
Indigenous	Amazonia	1985-1990	402	0.009	0.002	-0.046	-0.011	-0.028	0.937	0.587	0.350
Indigenous	Amazonia	1985-2018	402	0.031	0.001	-0.163	-0.042	-0.103	0.937	0.592	0.345
Indigenous	Amazonia	1991-1995	402	0.013	0.001	-0.069	-0.017	-0.043	0.937	0.587	0.350
Indigenous	Amazonia	1996-1999	402	0.009	0.004	-0.043	-0.008	-0.025	0.940	0.587	0.353
Indigenous	Amazonia	2000-2004	402	0.009	0.000	-0.051	-0.017	-0.034	0.940	0.592	0.348
Indigenous	Amazonia	2005-2012	402	0.009	0.073	-0.034	0.001	-0.016	0.937	0.587	0.350
Indigenous	Amazonia	2013-2018	402	0.010	0.014	-0.044	-0.005	-0.025	0.937	0.587	0.350
Indigenous	Brazil	1985-1990	906	0.011	0.001	-0.061	-0.016	-0.038	0.925	0.329	0.596
Indigenous	Brazil	1985-2018	906	0.025	0.000	-0.214	-0.116	-0.165	0.923	0.353	0.570
Indigenous	Brazil	1991-1995	906	0.014	0.000	-0.091	-0.038	-0.064	0.925	0.327	0.598
Indigenous	Brazil	1996-1999	906	0.009	0.000	-0.067	-0.032	-0.050	0.923	0.349	0.574
Indigenous	Brazil	2000-2004	906	0.010	0.000	-0.076	-0.035	-0.056	0.923	0.349	0.574
Indigenous	Brazil	2005-2012	906	0.013	0.001	-0.068	-0.019	-0.043	0.923	0.355	0.568
Indigenous	Brazil	2013-2018	906	0.012	0.000	-0.068	-0.020	-0.044	0.923	0.360	0.563
Indigenous	Caatinga	1985-1990	54	0.041	0.667	-0.098	0.063	-0.018	0.992	0.630	0.362
Indigenous	Caatinga	1985-2018	54	0.047	0.580	-0.117	0.066	-0.026	0.992	0.667	0.325

Treatment	spatialScale	temporalScale	*n*	SE	*p*_value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	ImbImprov
Indigenous	Caatinga	1991-1995	54	0.049	0.264	-0.041	0.150	0.054	0.992	0.667	0.325
Indigenous	Caatinga	1996-1999	54	0.020	0.937	-0.037	0.040	0.002	0.992	0.667	0.325
Indigenous	Caatinga	2000-2004	54	0.031	0.810	-0.069	0.054	-0.008	0.992	0.667	0.325
Indigenous	Caatinga	2005-2012	54	0.043	0.505	-0.112	0.055	-0.028	0.992	0.630	0.362
Indigenous	Caatinga	2013-2018	54	0.017	0.458	-0.045	0.020	-0.012	0.992	0.593	0.399
Indigenous	Cerrado	1985-1990	100	0.031	0.035	-0.125	-0.005	-0.065	0.950	0.760	0.190
Indigenous	Cerrado	1985-2018	100	0.055	0.000	-0.400	-0.183	-0.291	0.950	0.760	0.190
Indigenous	Cerrado	1991-1995	100	0.023	0.072	-0.088	0.004	-0.042	0.950	0.760	0.190
Indigenous	Cerrado	1996-1999	100	0.017	0.000	-0.112	-0.046	-0.079	0.950	0.760	0.190
Indigenous	Cerrado	2000-2004	100	0.042	0.000	-0.245	-0.081	-0.163	0.950	0.760	0.190
Indigenous	Cerrado	2005-2012	100	0.039	0.004	-0.186	-0.035	-0.111	0.950	0.760	0.190
Indigenous	Cerrado	2013-2018	100	0.028	0.011	-0.124	-0.016	-0.070	0.951	0.740	0.211
Indigenous	Mata Atlântica	1985-1990	256	0.018	0.183	-0.012	0.061	0.024	0.966	0.234	0.732
Indigenous	Mata Atlântica	1985-2018	256	0.030	0.268	-0.025	0.091	0.033	0.959	0.273	0.686
Indigenous	Mata Atlântica	1991-1995	256	0.015	0.746	-0.025	0.035	0.005	0.966	0.227	0.740
Indigenous	Mata Atlântica	1996-1999	256	0.009	0.355	-0.009	0.025	0.008	0.959	0.266	0.694
Indigenous	Mata Atlântica	2000-2004	256	0.008	0.613	-0.012	0.020	0.004	0.959	0.273	0.686
Indigenous	Mata Atlântica	2005-2012	256	0.013	0.092	-0.004	0.047	0.022	0.959	0.297	0.662
Indigenous	Mata Atlântica	2013-2018	256	0.006	0.493	-0.007	0.015	0.004	0.959	0.305	0.655
Protected	Amazonia	1985-1990	72	0.004	0.853	-0.007	0.008	0.001	0.969	0.611	0.358
Protected	Amazonia	1985-2018	70	0.030	0.000	-0.189	-0.072	-0.130	0.971	0.571	0.400
Protected	Amazonia	1991-1995	72	0.004	0.459	-0.005	0.012	0.003	0.969	0.611	0.358
Protected	Amazonia	1996-1999	70	0.008	0.579	-0.012	0.021	0.005	0.971	0.600	0.371
Protected	Amazonia	2000-2004	70	0.011	0.005	-0.052	-0.009	-0.030	0.971	0.600	0.371
Protected	Amazonia	2005-2012	70	0.014	0.000	-0.079	-0.023	-0.051	0.971	0.571	0.400
Protected	Amazonia	2013-2018	70	0.018	0.000	-0.116	-0.047	-0.081	0.971	0.600	0.371
Protected	Brazil	1985-1990	904	0.007	0.000	-0.046	-0.018	-0.032	0.843	0.237	0.606
Protected	Brazil	1985-2018	908	0.016	0.000	-0.128	-0.067	-0.097	0.841	0.244	0.597
Protected	Brazil	1991-1995	906	0.006	0.000	-0.035	-0.011	-0.023	0.843	0.241	0.602
Protected	Brazil	2000-2004	906	0.007	0.000	-0.047	-0.021	-0.034	0.841	0.280	0.561
Protected	Brazil	2005-2012	906	0.007	0.000	-0.049	-0.022	-0.035	0.843	0.243	0.600
Protected	Brazil	2013-2018	904	0.009	0.000	-0.051	-0.017	-0.034	0.843	0.288	0.555
Protected	Caatinga	1985-1990	60	0.054	0.039	-0.218	-0.006	-0.112	0.962	0.200	0.762
Protected	Caatinga	1985-2018	58	0.037	0.001	-0.189	-0.045	-0.117	0.962	0.483	0.479
Protected	Caatinga	1991-1995	58	0.021	0.322	-0.062	0.020	-0.021	0.962	0.414	0.548
Protected	Caatinga	1996-1999	58	0.014	0.000	-0.097	-0.041	-0.069	0.962	0.448	0.513

Treatment	spatialScale	temporalScale	*n*	SE	*p*_value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	ImbImprov
Protected	Caatinga	2000-2004	58	0.026	0.079	-0.096	0.005	-0.045	0.962	0.448	0.514
Protected	Caatinga	2005-2012	60	0.023	0.029	-0.094	-0.005	-0.049	0.962	0.200	0.762
Protected	Caatinga	2013-2018	60	0.017	0.002	-0.084	-0.019	-0.051	0.962	0.167	0.795
Protected	Cerrado	1985-1990	172	0.027	0.082	-0.099	0.006	-0.046	0.901	0.570	0.331
Protected	Cerrado	1985-2018	172	0.035	0.000	-0.216	-0.080	-0.148	0.910	0.558	0.352
Protected	Cerrado	1991-1995	172	0.020	0.288	-0.059	0.017	-0.021	0.901	0.570	0.331
Protected	Cerrado	1996-1999	172	0.019	0.072	-0.072	0.003	-0.034	0.901	0.558	0.343
Protected	Cerrado	2000-2004	172	0.009	0.026	-0.039	-0.002	-0.021	0.901	0.547	0.355
Protected	Cerrado	2005-2012	172	0.016	0.013	-0.072	-0.008	-0.040	0.910	0.535	0.375
Protected	Cerrado	2013-2018	172	0.017	0.073	-0.064	0.003	-0.030	0.910	0.558	0.352
Protected	Mata Atlântica	1985-1990	516	0.010	0.000	-0.063	-0.022	-0.042	0.875	0.283	0.592
Protected	Mata Atlântica	1985-2018	516	0.016	0.000	-0.098	-0.035	-0.066	0.872	0.291	0.581
Protected	Mata Atlântica	1991-1995	514	0.005	0.000	-0.038	-0.019	-0.028	0.875	0.304	0.572
Protected	Mata Atlântica	1996-1999	514	0.004	0.000	-0.023	-0.007	-0.015	0.872	0.300	0.573
Protected	Mata Atlântica	2000-2004	516	0.005	0.000	-0.032	-0.012	-0.022	0.872	0.298	0.574
Protected	Mata Atlântica	2005-2012	514	0.005	0.006	-0.026	-0.004	-0.015	0.872	0.370	0.503
Protected	Mata Atlântica	2013-2018	510	0.005	0.001	-0.025	-0.007	-0.016	0.872	0.329	0.542
Quilombola	Amazonia	1985-1990	226	0.014	0.000	-0.081	-0.028	-0.055	0.910	0.602	0.308
Quilombola	Amazonia	1985-2018	226	0.027	0.000	-0.164	-0.058	-0.111	0.910	0.611	0.299
Quilombola	Amazonia	1991-1995	226	0.015	0.128	-0.051	0.006	-0.022	0.910	0.611	0.299
Quilombola	Amazonia	1996-1999	226	0.012	0.000	-0.066	-0.020	-0.043	0.910	0.619	0.290
Quilombola	Amazonia	2000-2004	226	0.014	0.001	-0.073	-0.017	-0.045	0.910	0.611	0.299
Quilombola	Amazonia	2005-2012	226	0.015	0.000	-0.086	-0.025	-0.056	0.910	0.628	0.281
Quilombola	Amazonia	2013-2018	226	0.020	0.006	-0.093	-0.016	-0.055	0.910	0.628	0.281
Quilombola	Brazil	1985-1990	702	0.011	0.378	-0.032	0.012	-0.010	0.867	0.148	0.719
Quilombola	Brazil	1985-2018	702	0.020	0.000	-0.129	-0.050	-0.089	0.867	0.165	0.701
Quilombola	Brazil	1991-1995	702	0.010	0.666	-0.025	0.016	-0.004	0.867	0.151	0.716
Quilombola	Brazil	1996-1999	702	0.008	0.029	-0.035	-0.002	-0.018	0.867	0.154	0.713
Quilombola	Brazil	2000-2004	702	0.012	0.009	-0.055	-0.008	-0.031	0.867	0.157	0.710
Quilombola	Brazil	2005-2012	704	0.013	0.001	-0.067	-0.016	-0.042	0.867	0.168	0.699
Quilombola	Brazil	2013-2018	704	0.010	0.000	-0.061	-0.023	-0.042	0.867	0.170	0.696
Quilombola	Caatinga	1985-1990	124	0.028	0.822	-0.048	0.060	0.006	0.974	0.323	0.652
Quilombola	Caatinga	1985-2018	124	0.044	0.267	-0.038	0.136	0.049	0.974	0.323	0.652
Quilombola	Caatinga	1991-1995	124	0.026	0.069	-0.004	0.098	0.047	0.974	0.306	0.668
Quilombola	Caatinga	1996-1999	124	0.017	0.218	-0.012	0.054	0.021	0.974	0.323	0.652
Quilombola	Caatinga	2000-2004	124	0.021	0.416	-0.024	0.057	0.017	0.974	0.323	0.652

Treatment	spatialScale	temporalScale	*n*	SE	*p*_value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	ImbImprov
Quilombola	Caatinga	2005-2012	124	0.019	0.675	-0.029	0.046	0.008	0.974	0.323	0.652
Quilombola	Caatinga	2013-2018	124	0.008	0.025	0.002	0.032	0.017	0.974	0.306	0.668
Quilombola	Cerrado	1985-1990	92	0.053	0.524	-0.071	0.139	0.034	0.936	0.543	0.392
Quilombola	Cerrado	1985-2018	92	0.048	0.494	-0.061	0.127	0.033	0.936	0.543	0.392
Quilombola	Cerrado	1991-1995	92	0.023	0.509	-0.060	0.030	-0.015	0.936	0.543	0.392
Quilombola	Cerrado	1996-1999	92	0.026	0.856	-0.047	0.057	0.005	0.936	0.543	0.392
Quilombola	Cerrado	2000-2004	92	0.038	0.223	-0.122	0.028	-0.047	0.936	0.543	0.392
Quilombola	Cerrado	2005-2012	92	0.024	0.695	-0.057	0.038	-0.009	0.936	0.543	0.392
Quilombola	Cerrado	2013-2018	92	0.019	0.670	-0.029	0.045	0.008	0.936	0.543	0.392
Quilombola	Mata Atlântica	1985-1990	218	0.023	0.014	-0.102	-0.012	-0.057	0.891	0.266	0.625
Quilombola	Mata Atlântica	1985-2018	218	0.043	0.291	-0.130	0.039	-0.045	0.890	0.303	0.588
Quilombola	Mata Atlântica	1991-1995	218	0.018	0.359	-0.050	0.018	-0.016	0.891	0.275	0.616
Quilombola	Mata Atlântica	1996-1999	218	0.009	0.179	-0.030	0.006	-0.012	0.891	0.275	0.615
Quilombola	Mata Atlântica	2000-2004	218	0.013	0.740	-0.030	0.022	-0.004	0.891	0.303	0.588
Quilombola	Mata Atlântica	2005-2012	218	0.008	0.129	-0.028	0.004	-0.012	0.890	0.294	0.597
Quilombola	Mata Atlântica	2013-2018	218	0.008	0.628	-0.019	0.011	-0.004	0.890	0.303	0.587
Sustainable use	Amazonia	1985-1990	178	0.009	0.011	-0.040	-0.005	-0.022	0.963	0.607	0.356
Sustainable use	Amazonia	1985-2018	178	0.039	0.002	-0.197	-0.045	-0.121	0.963	0.607	0.356
Sustainable use	Amazonia	1991-1995	178	0.014	0.004	-0.067	-0.013	-0.040	0.963	0.618	0.345
Sustainable use	Amazonia	1996-1999	178	0.012	0.001	-0.062	-0.016	-0.039	0.963	0.607	0.356
Sustainable use	Amazonia	2000-2004	178	0.016	0.000	-0.093	-0.032	-0.063	0.963	0.607	0.356
Sustainable use	Amazonia	2005-2012	178	0.016	0.004	-0.078	-0.015	-0.047	0.963	0.596	0.367
Sustainable use	Amazonia	2013-2018	178	0.016	0.030	-0.065	-0.003	-0.034	0.963	0.596	0.367
Sustainable use	Brazil	1985-1990	1,234	0.009	0.003	-0.045	-0.009	-0.027	0.716	0.245	0.471
Sustainable use	Brazil	1985-2018	1,232	0.029	0.000	-0.182	-0.069	-0.125	0.716	0.237	0.479
Sustainable use	Brazil	1991-1995	1,234	0.009	0.001	-0.047	-0.012	-0.030	0.716	0.238	0.477
Sustainable use	Brazil	1996-1999	1,232	0.006	0.000	-0.038	-0.013	-0.026	0.716	0.239	0.477
Sustainable use	Brazil	2000-2004	1,232	0.008	0.000	-0.058	-0.026	-0.042	0.716	0.240	0.475
Sustainable use	Brazil	2005-2012	1,232	0.009	0.000	-0.060	-0.022	-0.041	0.716	0.235	0.480
Sustainable use	Brazil	2013-2018	1,228	0.011	0.002	-0.055	-0.013	-0.034	0.714	0.233	0.481
Sustainable use	Caatinga	1985-1990	100	0.051	0.120	-0.180	0.021	-0.080	0.895	0.260	0.635
Sustainable use	Caatinga	1985-2018	100	0.000	NA	0.000	0.000	0.000	0.895	0.260	0.635
Sustainable use	Caatinga	1991-1995	98	0.024	0.008	-0.109	-0.017	-0.063	0.895	0.490	0.405
Sustainable use	Caatinga	1996-1999	100	0.022	0.006	-0.106	-0.018	-0.062	0.895	0.260	0.635
Sustainable use	Caatinga	2000-2004	100	0.023	0.137	-0.080	0.011	-0.034	0.895	0.260	0.635
Sustainable use	Caatinga	2005-2012	100	0.025	0.000	-0.145	-0.045	-0.095	0.895	0.260	0.635

Treatment	**spatialScale**	**temporalScale**	***n***	**SE**	***p*_value**	**lower_ci**	**upper_ci**	**Effect**	**ImbBefore**	**ImbAfter**	**ImbImprov**
Sustainable use	Cerrado	1985-1990	156	0.029	0.102	-0.104	0.009	-0.047	0.849	0.500	0.349
Sustainable use	Cerrado	1985-2018	156	0.043	0.002	-0.217	-0.050	-0.134	0.850	0.487	0.362
Sustainable use	Cerrado	1991-1995	156	0.016	0.000	-0.130	-0.066	-0.098	0.850	0.500	0.350
Sustainable use	Cerrado	1996-1999	156	0.012	0.000	-0.072	-0.025	-0.049	0.850	0.487	0.362
Sustainable use	Cerrado	2000-2004	156	0.029	0.004	-0.140	-0.026	-0.083	0.850	0.487	0.362
Sustainable use	Cerrado	2005-2012	158	0.026	0.005	-0.125	-0.023	-0.074	0.850	0.481	0.369
Sustainable use	Cerrado	2013-2018	158	0.023	0.006	-0.109	-0.018	-0.064	0.850	0.494	0.356
Sustainable use	Mata Atlântica	1985-1990	756	0.024	0.000	-0.144	-0.050	-0.097	0.732	0.275	0.457
Sustainable use	Mata Atlântica	1985-2018	754	0.042	0.001	-0.225	-0.061	-0.143	0.732	0.284	0.448
Sustainable use	Mata Atlântica	1991-1995	756	0.014	0.004	-0.065	-0.012	-0.039	0.732	0.286	0.447
Sustainable use	Mata Atlântica	1996-1999	756	0.009	0.000	-0.058	-0.022	-0.040	0.732	0.283	0.449
Sustainable use	Mata Atlântica	2000-2004	754	0.010	0.005	-0.048	-0.009	-0.028	0.730	0.281	0.449
Sustainable use	Mata Atlântica	2005-2012	754	0.011	0.000	-0.062	-0.020	-0.041	0.730	0.268	0.462
Sustainable use	Mata Atlântica	2013-2018	754	0.006	0.000	-0.041	-0.018	-0.029	0.729	0.263	0.467
Undesignated/ untitled public	Amazonia	1985-1990	8,066	0.005	0.024	-0.022	-0.002	-0.012	0.638	0.353	0.285
Undesignated/ untitled public	Amazonia	1985-2018	8,064	0.015	0.000	0.133	0.193	0.163	0.641	0.353	0.288
Undesignated/ untitled public	Amazonia	1991-1995	8,062	0.006	0.323	-0.006	0.019	0.006	0.640	0.357	0.283
Undesignated/ untitled public	Amazonia	1996-1999	8,060	0.006	0.000	0.012	0.035	0.023	0.641	0.359	0.282
Undesignated/ untitled public	Amazonia	2000-2004	8,064	0.009	0.000	0.051	0.086	0.068	0.641	0.354	0.287
Undesignated/ untitled public	Amazonia	2005-2012	8,062	0.010	0.000	0.087	0.128	0.108	0.641	0.353	0.288
Undesignated/ untitled public	Amazonia	2013-2018	8,060	0.009	0.000	0.062	0.097	0.079	0.641	0.355	0.286
Undesignated/ untitled public	Brazil	1985-1990	34,212	0.005	0.032	-0.019	-0.001	-0.010	0.663	0.123	0.540
Undesignated/ untitled public	Brazil	1985-2018	34,216	0.010	0.000	0.113	0.154	0.133	0.663	0.126	0.537
Undesignated/ untitled public	Brazil	1991-1995	34,216	0.004	0.029	0.001	0.017	0.009	0.663	0.125	0.538
Undesignated/ untitled public	Brazil	1996-1999	34,216	0.004	0.000	0.015	0.030	0.022	0.663	0.126	0.537
Undesignated/ untitled public	Brazil	2000-2004	34,214	0.005	0.000	0.039	0.058	0.048	0.663	0.125	0.538
Undesignated/ untitled public	Brazil	2005-2012	34,218	0.006	0.000	0.066	0.089	0.077	0.663	0.128	0.535
Undesignated/ untitled public	Brazil	2013-2018	34,214	0.004	0.000	0.048	0.066	0.057	0.662	0.130	0.533
Undesignated/ untitled public	Caatinga	1985-1990	10,020	0.006	0.214	-0.020	0.005	-0.008	0.714	0.142	0.572
Undesignated/ untitled public	Caatinga	1985-2018	10,020	0.009	0.134	-0.004	0.031	0.013	0.716	0.137	0.579
Undesignated/ untitled public	Caatinga	1991-1995	10,024	0.004	0.765	-0.007	0.010	0.001	0.715	0.140	0.575
Undesignated/ untitled public	Caatinga	1996-1999	10,024	0.003	0.043	0.000	0.013	0.007	0.715	0.138	0.578
Undesignated/ untitled public	Caatinga	2000-2004	10,022	0.003	0.001	0.005	0.018	0.012	0.716	0.137	0.579
Undesignated/ untitled public	Caatinga	2005-2012	10,022	0.005	0.510	-0.007	0.014	0.003	0.716	0.135	0.580
Undesignated/ untitled public	Caatinga	2013-2018	10,022	0.003	0.932	-0.007	0.007	0.000	0.715	0.135	0.580
Undesignated/ untitled public	Cerrado	1985-1990	9,670	0.006	0.012	-0.026	-0.003	-0.015	0.718	0.256	0.462

Treatment	**spatialScale**	**temporalScale**	***n***	**SE**	***p*_value**	**lower_ci**	**upper_ci**	**Effect**	**ImbBefore**	**ImbAfter**	**ImbImprov**
Undesignated/ untitled public	Cerrado	1985-2018	9,672	0.014	0.017	0.006	0.059	0.032	0.718	0.261	0.457
Undesignated/ untitled public	Cerrado	1991-1995	9,670	0.005	0.510	-0.007	0.014	0.004	0.718	0.258	0.460
Undesignated/ untitled public	Cerrado	1996-1999	9,670	0.005	0.005	0.004	0.024	0.014	0.718	0.258	0.460
Undesignated/ untitled public	Cerrado	2000-2004	9,672	0.006	0.179	-0.004	0.020	0.008	0.718	0.259	0.460
Undesignated/ untitled public	Cerrado	2005-2012	9,672	0.006	0.001	0.009	0.034	0.022	0.719	0.261	0.458
Undesignated/ untitled public	Cerrado	2013-2018	9,672	0.005	0.000	0.021	0.041	0.031	0.719	0.261	0.458
Undesignated/ untitled public	Mata Atlântica	1985-1990	5,130	0.011	0.000	0.063	0.105	0.084	0.744	0.160	0.584
Undesignated/ untitled public	Mata Atlântica	1985-2018	5,134	0.015	0.000	0.185	0.242	0.213	0.743	0.113	0.630
Undesignated/ untitled public	Mata Atlântica	1991-1995	5,130	0.007	0.000	0.039	0.066	0.052	0.744	0.161	0.583
Undesignated/ untitled public	Mata Atlântica	1996-1999	5,132	0.006	0.000	0.046	0.069	0.057	0.743	0.141	0.602
Undesignated/ untitled public	Mata Atlântica	2000-2004	5,132	0.006	0.000	0.042	0.067	0.054	0.743	0.142	0.601
Undesignated/ untitled public	Mata Atlântica	2005-2012	5,134	0.005	0.000	0.027	0.046	0.037	0.743	0.109	0.634
Undesignated/ untitled public	Mata Atlântica	2013-2018	5,134	0.003	0.000	0.015	0.028	0.022	0.742	0.113	0.630
Undesignated/ untitled public	Pampa	1985-1990	404	0.041	0.082	-0.009	0.151	0.071	0.843	0.391	0.452
Undesignated/ untitled public	Pampa	1985-2018	404	0.045	0.022	0.015	0.192	0.104	0.843	0.465	0.378
Undesignated/ untitled public	Pampa	1991-1995	404	0.029	0.175	-0.017	0.096	0.039	0.843	0.416	0.427
Undesignated/ untitled public	Pampa	1996-1999	404	0.028	0.000	0.047	0.157	0.102	0.843	0.436	0.407
Undesignated/ untitled public	Pampa	2000-2004	404	0.014	0.000	0.043	0.096	0.069	0.843	0.431	0.412
Undesignated/ untitled public	Pampa	2005-2012	404	0.049	0.022	0.016	0.209	0.113	0.843	0.455	0.387
Undesignated/ untitled public	Pampa	2013-2018	404	0.012	0.074	-0.002	0.047	0.022	0.843	0.460	0.382
Undesignated/ untitled public	Pantanal	1985-1990	260	0.020	0.000	-0.124	-0.045	-0.084	0.695	0.462	0.233
Undesignated/ untitled public	Pantanal	1985-2018	262	0.024	0.000	0.126	0.221	0.173	0.696	0.458	0.238
Undesignated/ untitled public	Pantanal	1991-1995	260	0.010	0.282	-0.009	0.030	0.011	0.695	0.462	0.233
Undesignated/ untitled public	Pantanal	1996-1999	262	0.007	0.020	0.003	0.029	0.016	0.695	0.458	0.237
Undesignated/ untitled public	Pantanal	2000-2004	262	0.012	0.220	-0.009	0.038	0.015	0.695	0.466	0.230
Undesignated/ untitled public	Pantanal	2005-2012	262	0.020	0.000	0.032	0.108	0.070	0.696	0.466	0.230
Undesignated/ untitled public	Pantanal	2013-2018	262	0.037	0.000	0.147	0.292	0.219	0.696	0.450	0.245
Robustness check: protected areas and sustainable use areas filtered by known year of creation											
Protected	Amazonia	2000-2004	50	0.013	0.006	-0.059	-0.010	-0.034	1.000	0.760	0.240
Protected	Amazonia	2005-2012	58	0.013	0.001	-0.067	-0.018	-0.042	1.000	0.828	0.172
Protected	Amazonia	2013-2018	64	0.016	0.000	-0.114	-0.051	-0.083	1.000	0.875	0.125
Protected	Brazil	1985-1990	224	0.016	0.017	-0.070	-0.007	-0.039	1.000	0.955	0.045
Protected	Brazil	1985-2018	350	0.022	0.000	-0.138	-0.050	-0.094	1.000	0.966	0.034
Protected	Brazil	1991-1995	350	0.010	0.000	-0.059	-0.021	-0.040	1.000	0.966	0.034
Protected	Brazil	1996-1999	398	0.010	0.009	-0.045	-0.006	-0.026	1.000	0.965	0.035
Protected	Brazil	2000-2004	490	0.007	0.000	-0.056	-0.027	-0.041	1.000	0.939	0.061

Treatment	spatialScale	temporalScale	*n*	SE	*p*_value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	ImbImprov
Protected	Brazil	2005-2012	634	0.007	0.000	-0.052	-0.023	-0.037	1.000	0.968	0.032
Protected	Brazil	2013-2018	870	0.009	0.000	-0.052	-0.017	-0.034	1.000	0.966	0.034
Protected	Caatinga	2013-2018	58	0.016	0.006	-0.077	-0.013	-0.045	1.000	0.897	0.103
Protected	Cerrado	1985-1990	46	0.071	0.100	-0.255	0.022	-0.116	1.000	1.000	0.000
Protected	Cerrado	1985-2018	66	0.051	0.000	-0.348	-0.148	-0.248	1.000	1.000	0.000
Protected	Cerrado	1991-1995	66	0.029	0.600	-0.071	0.041	-0.015	1.000	1.000	0.000
Protected	Cerrado	1996-1999	72	0.025	0.006	-0.117	-0.020	-0.068	1.000	0.972	0.028
Protected	Cerrado	2000-2004	100	0.012	0.066	-0.045	0.001	-0.022	1.000	0.980	0.020
Protected	Cerrado	2005-2012	140	0.017	0.016	-0.073	-0.007	-0.040	1.000	0.971	0.029
Protected	Cerrado	2013-2018	172	0.017	0.073	-0.064	0.003	-0.030	1.000	0.977	0.023
Protected	Mata Atlântica	1985-1990	108	0.038	0.013	-0.168	-0.020	-0.094	1.000	1.000	0.000
Protected	Mata Atlântica	1985-2018	202	0.040	0.000	-0.232	-0.073	-0.153	1.000	1.000	0.000
Protected	Mata Atlântica	1991-1995	200	0.018	0.000	-0.114	-0.044	-0.079	1.000	1.000	0.000
Protected	Mata Atlântica	1996-1999	226	0.010	0.004	-0.048	-0.009	-0.029	1.000	1.000	0.000
Protected	Mata Atlântica	2000-2004	262	0.007	0.000	-0.054	-0.025	-0.040	1.000	0.992	0.008
Protected	Mata Atlântica	2005-2012	326	0.006	0.000	-0.043	-0.021	-0.032	1.000	0.994	0.006
Protected	Mata Atlântica	2013-2018	486	0.005	0.001	-0.026	-0.007	-0.016	1.000	0.996	0.004
Sustainable use	Amazonia	1996-1999	84	0.012	0.001	-0.064	-0.015	-0.040	1.000	0.976	0.024
Sustainable use	Amazonia	2000-2004	98	0.014	0.000	-0.100	-0.045	-0.072	1.000	0.980	0.020
Sustainable use	Amazonia	2005-2012	150	0.015	0.007	-0.072	-0.011	-0.042	1.000	0.947	0.053
Sustainable use	Amazonia	2013-2018	174	0.016	0.036	-0.064	-0.002	-0.033	1.000	0.966	0.034
Sustainable use	Brazil	1985-1990	56	0.055	0.864	-0.117	0.098	-0.009	1.000	1.000	0.000
Sustainable use	Brazil	1985-2018	120	0.041	0.071	-0.155	0.006	-0.074	1.000	0.950	0.050
Sustainable use	Brazil	1991-1995	120	0.010	0.444	-0.028	0.012	-0.008	1.000	0.950	0.050
Sustainable use	Brazil	1996-1999	198	0.013	0.088	-0.047	0.003	-0.022	1.000	0.980	0.020
Sustainable use	Brazil	2000-2004	274	0.014	0.003	-0.070	-0.014	-0.042	1.000	0.985	0.015
Sustainable use	Brazil	2005-2012	444	0.013	0.000	-0.076	-0.026	-0.051	1.000	0.968	0.032
Sustainable use	Brazil	2013-2018	1170	0.011	0.002	-0.057	-0.013	-0.035	1.000	0.995	0.005
Sustainable use	Cerrado	2005-2012	76	0.021	0.313	-0.064	0.020	-0.022	1.000	0.974	0.026
Sustainable use	Cerrado	2013-2018	158	0.023	0.006	-0.109	-0.018	-0.064	1.000	0.987	0.013
Sustainable use	Mata Atlântica	1985-2018	58	0.073	0.000	-0.406	-0.118	-0.262	1.000	1.000	0.000
Sustainable use	Mata Atlântica	1991-1995	58	0.015	0.005	-0.072	-0.013	-0.042	1.000	1.000	0.000
Sustainable use	Mata Atlântica	1996-1999	74	0.017	0.000	-0.115	-0.050	-0.083	1.000	1.000	0.000
Sustainable use	Mata Atlântica	2000-2004	100	0.014	0.003	-0.072	-0.015	-0.044	1.000	1.000	0.000
Sustainable use	Mata Atlântica	2005-2012	158	0.022	0.002	-0.114	-0.026	-0.070	1.000	1.000	0.000
Sustainable use	Mata Atlântica	2013-2018	708	0.006	0.000	-0.041	-0.018	-0.030	1.000	1.000	0.000

Table 14. Synthesized direction of cross-scale effects of strict-protection and sustainable-use regimes, with percentages based on alternative results that were time-filtered for greater robustness of temporal stability assumptions (see **Tables 6-7** for detailed description). These time-filtered datasets exclude any parcels for which these respective conservation-focused tenure regimes were either not yet established at the beginning of the considered time period or for which the creation date was unknown. Note that in left first table section ("Direction of estimated effects on deforestation"), only the results for strict-protection and sustainable-use regimes (in black) are based on different models. Those for other tenure regimes are as in **Table 6**, but restricted to the scales where all regimes could be consistently compared. Note that due to smaller initial parcel numbers of the time-filtered datasets, the matched time-filtered datasets showed substantially lower balance levels post-matching compared to the non-filtered datasets (see **Tables 12-13**). Therefore, ranking results are not considered based on the time-filtered data reliable, are ignored for the conclusions, and reported here (in grey) for transparency only.

	Direction of estimated effects on deforestation													Ranking by relative magnitude of effect size						
	↗ (count)	↗ (count w. by balance)	% ↗	% ↗(w. by balance)	↘(count)	↘ (count w. by balance)	% ↘	% ↘(w. by balance)	non-sig. (count)	non-sig. (w. by balance)	% non-sig.	% non-sig. (w. by balance)	Total models	best	best (w. by balance)	worst	worst (w. by balance)	non-sig.	non-sig. (w. by	Total models
Compared to undesignated/untitled lands																				
Private lands	1	0.88	5.26%	5.70%	17	13.65	89.47%	88.68%	1	0.87	5.26%	5.62%	19	5.26%	0.00%	27.63%	12.28%	1	0	19
Protected areas	0	0.00	0.00%	0.00%	18	0.07	94.74%	100.00%	1	0.00	5.26%	0.00%	19	10.53%	0.00%	7.89%	0.00%	1	0	19
Sustainable use areas	0	0.00	0.00%	0.00%	14	0.05	73.68%	27.93%	5	0.13	26.32%	72.07%	19	47.37%	26.32%	11.84%	36.84%	5	0	19
Indigenous lands	0	0.00	0.00%	0.00%	18	9.71	94.74%	96.54%	1	0.35	5.26%	3.46%	19	26.32%	73.68%	11.84%	0.00%	1	0	19
Quilombola lands	0	0.00	0.00%	0.00%	12	5.87	63.16%	61.53%	7	3.67	36.84%	38.47%	19	10.53%	0.00%	40.79%	50.88%	7	0	19
All of the above compared to undesignated/untitled	5	0.88	5.26%	2.49%	79	29.36	83.16%	83.29%	15	5.01	15.79%	14.22%	95							
Compared to private lands																				
Public lands	16	13.00	94.12%	93.68%	1	0.88	5.88%	6.32%	0	0.00	0.00%	0.00%	17	0.00%	0.00%	94.12%	100.00%	0	0	17
Protected areas	0	0.00	0.00%	0.00%	16	0.83	94.12%	97.26%	1	0.02	5.88%	2.74%	17	17.65%	13.49%	0.00%	0.00%	1	0	17
Sustainable use areas	0	0.00	0.00%	0.00%	13	0.17	76.47%	64.07%	4	0.10	23.53%	35.93%	17	41.18%	20.63%	2.94%	0.00%	4	0	17
Indigenous lands	0	0.00	0.00%	0.00%	10	5.25	58.82%	52.39%	7	4.77	41.18%	47.61%	17	35.29%	44.84%	0.00%	0.00%	7	0	17
Quilombola lands	0	0.00	0.00%	0.00%	7	4.48	41.18%	39.52%	10	6.86	58.82%	60.48%	17	5.88%	21.03%	2.94%	0.00%	10	0	17
All of the above compared to private	16	13.00	18.82%	35.76%	47	11.61	55.29%	31.92%	22	11.75	25.88%	32.32%	85							

Table 15. Summary of sensitivity analysis using Rosenbaum bounds. Upper and lower bounds for both Hodges Lehmann point estimates and p-values are calculated for different Γ levels. For each tenure-regime comparison, spatial scale, and temporal scale considered, two summaries are provided: 1) the geometric mean deviation of upper/lower bounds of Hodges Lehmann estimates from Γ=1, with deviations expressed as relative error in percent (i.e., relative to the magnitude of the respective median effect size at Γ=1), and 2) the percent of models that changed in statistical significance (p≤0.05).

	Geometric mean deviation of upper/lower bounds of Hodges Lehmann estimates from Γ=1 (deviation expressed as relative error in percent)					**Percentage of models that change in significance (p≤0.05) from Γ=1**				
	Γ =1.1	Γ =1.2	Γ =1.3	Γ =1.4	Γ =1.5	Γ =1.1	Γ =1.2	Γ =1.3	Γ =1.4	Γ =1.5
Tenure-regime comparisons										
public vs. private	11.03%	20.92%	27.75%	37.95%	45.83%	6.12%	12.24%	12.24%	14.29%	18.37%
public vs. protected	7.22%	14.69%	21.61%	28.11%	34.39%	2.86%	8.57%	8.57%	14.29%	14.29%
public vs. sustainable use	6.28%	12.04%	17.34%	22.53%	27.37%	2.86%	2.86%	2.86%	8.57%	22.86%
public vs. indigenous	9.63%	19.62%	26.83%	36.82%	44.16%	0.00%	5.71%	8.57%	8.57%	11.43%
public vs. quilombola	18.09%	35.18%	53.25%	69.16%	83.49%	0.00%	5.71%	11.43%	17.14%	20.00%
public vs. communal	7.27%	13.91%	20.08%	25.84%	31.20%	0.00%	0.00%	0.00%	0.00%	0.00%
private vs. public	11.03%	20.92%	27.74%	37.95%	45.82%	6.12%	12.24%	12.24%	14.29%	18.37%
private vs. protected	7.69%	15.13%	22.29%	28.89%	35.71%	5.88%	5.88%	8.82%	17.65%	20.59%
private vs. sustainable use	7.54%	14.50%	21.03%	27.56%	33.64%	3.03%	3.03%	6.06%	9.09%	12.12%
private vs. indigenous	12.05%	22.33%	30.68%	40.84%	49.07%	2.86%	2.86%	11.43%	14.29%	20.00%
private vs. quilombola	26.29%	50.38%	74.23%	92.97%	110.13%	2.86%	11.43%	17.14%	28.57%	28.57%
private vs. communal	10.06%	19.36%	28.00%	36.14%	43.82%	0.00%	7.14%	7.14%	21.43%	21.43%
Average across tenure-regime comparisons	*11.18%*	*21.58%*	*30.90%*	*40.40%*	*48.72%*	*2.72%*	*6.47%*	*8.88%*	*14.02%*	*17.34%*
Spatial scales										
Brazil	9.98%	18.72%	24.15%	33.34%	40.25%	0.00%	3.61%	7.23%	9.64%	13.25%
Amazonia	11.23%	22.56%	33.14%	42.81%	52.41%	4.76%	7.14%	9.52%	16.67%	22.62%
Caatinga	17.02%	35.19%	51.14%	66.43%	80.12%	7.35%	17.65%	17.65%	25.00%	25.00%
Cerrado	11.01%	19.96%	28.23%	38.06%	45.83%	0.00%	5.71%	10.00%	15.71%	24.29%
Mata Atlantica	7.24%	13.98%	20.35%	26.33%	31.80%	2.86%	2.86%	5.71%	8.57%	10.00%
Pampa	4.86%	9.25%	13.39%	17.23%	20.74%	0.00%	0.00%	0.00%	0.00%	0.00%
Pantanal	9.50%	17.82%	26.03%	33.40%	40.21%	14.29%	14.29%	14.29%	14.29%	14.29%
Average across spatial scales	*10.12%*	*19.64%*	*28.06%*	*36.80%*	*44.48%*	*4.18%*	*7.32%*	*9.20%*	*12.84%*	*15.64%*
Temporal scales										
1985-2018	7.35%	14.92%	21.55%	27.86%	33.01%	0.00%	7.02%	8.77%	12.28%	14.04%
1985-1990	14.75%	26.86%	31.50%	46.78%	56.66%	5.17%	8.62%	12.07%	17.24%	22.41%
1991-1995	14.59%	28.41%	41.73%	55.70%	67.42%	5.17%	6.90%	6.90%	12.07%	20.69%
1996-1999	10.13%	19.43%	28.15%	36.44%	44.26%	0.00%	7.02%	8.77%	14.04%	17.54%

	Geometric mean deviation of upper/lower bounds of Hodges Lehmann estimates from Γ=1 (deviation expressed as relative error in percent)					**Percentage of models that change in significance (p≤0.05) from Γ=1**				
	Γ =1.1	Γ =1.2	Γ =1.3	Γ =1.4	Γ =1.5	Γ =1.1	Γ =1.2	Γ =1.3	Γ =1.4	Γ =1.5
2000-2004	10.50%	20.16%	29.52%	38.12%	46.30%	5.17%	6.90%	13.79%	17.24%	17.24%
2005-2012	9.41%	18.18%	27.03%	34.86%	42.23%	0.00%	3.45%	3.45%	8.62%	12.07%
2013-2018	8.71%	17.21%	24.91%	31.53%	38.55%	7.02%	10.53%	14.04%	19.30%	22.81%
Average across spatial scales	*10.78%*	*20.74%*	*29.20%*	*38.76%*	*46.92%*	*3.22%*	*7.21%*	*9.68%*	*14.40%*	*18.11%*

Table 16. Full regression results for estimating effects of different tenure regimes on two biodiversity metrics: species richness and Simpson's diversity index.

		Species richness model estimates						**Simpson's diversity index model estimates**					
Model comparison	**variable**	**AME**	**SE**	**z**	**p**	**lower**	**upper**	**AME**	**SE**	**z**	**p**	**lower**	**upper**
private vs. indigenous	area	0.0000	0.0000	1.4347	0.1514	0.0000	0.0000	0.0000	0.0000	1.5462	0.1221	0.0000	0.0000
private vs. indigenous	cattl	0.0000	0.0000	7.0773	0.0000	0.0000	0.0000	0.0000	0.0000	7.9110	0.0000	0.0000	0.0000
private vs. indigenous	crop	0.0000	0.0000	0.7578	0.4486	0.0000	0.0000	0.0000	0.0000	0.7657	0.4439	0.0000	0.0000
private vs. indigenous	elevation	-0.0007	0.0002	-3.6948	0.0002	-0.0011	-0.0003	-0.0010	0.0002	-5.4093	0.0000	-0.0013	-0.0006
private vs. indigenous	pdsi	0.0006	0.0004	1.4285	0.1531	-0.0002	0.0014	0.0009	0.0005	1.7140	0.0865	-0.0001	0.0019
private vs. indigenous	pop	0.0057	0.0080	0.7144	0.4750	-0.0100	0.0214	-0.0031	0.0079	-0.3893	0.6970	-0.0186	0.0124
private vs. indigenous	**private.vs.indigenous**	**0.0277**	**0.0322**	**0.8624**	**0.3885**	**-0.0353**	**0.0908**	**0.0631**	**0.0370**	**1.7072**	**0.0878**	**-0.0093**	**0.1356**
private vs. indigenous	slope	0.0923	0.0186	4.9760	0.0000	0.0560	0.1287	0.1255	0.0193	6.5142	0.0000	0.0877	0.1633
private vs. indigenous	stateAL	-1.0839	0.1458	-7.4344	0.0000	-1.3696	-0.7981	-1.1813	0.1560	-7.5720	0.0000	-1.4870	-0.8755
private vs. indigenous	stateAM	-0.1690	0.0472	-3.5794	0.0003	-0.2616	-0.0765	-0.2116	0.0662	-3.1950	0.0014	-0.3414	-0.0818
private vs. indigenous	stateAP	-0.5898	0.2103	-2.8045	0.0050	-1.0020	-0.1776	-0.6580	0.2604	-2.5269	0.0115	-1.1684	-0.1476
private vs. indigenous	stateBA	-0.6880	0.0630	-10.9145	0.0000	-0.8115	-0.5644	-0.8323	0.0873	-9.5352	0.0000	-1.0033	-0.6612
private vs. indigenous	stateCE	-1.2253	0.1603	-7.6421	0.0000	-1.5395	-0.9110	-1.4102	0.1744	-8.0865	0.0000	-1.7520	-1.0684
private vs. indigenous	stateES	-0.8203	0.0539	-15.2090	0.0000	-0.9260	-0.7146	-0.9876	0.0721	-13.7048	0.0000	-1.1289	-0.8464
private vs. indigenous	stateGO	-0.7951	0.1481	-5.3705	0.0000	-1.0853	-0.5050	-0.9836	0.1671	-5.8855	0.0000	-1.3111	-0.6560
private vs. indigenous	stateMA	-0.7971	0.0976	-8.1637	0.0000	-0.9884	-0.6057	-0.9518	0.1157	-8.2273	0.0000	-1.1786	-0.7251
private vs. indigenous	stateMG	-0.6231	0.1198	-5.2002	0.0000	-0.8579	-0.3882	-0.7833	0.1434	-5.4623	0.0000	-1.0644	-0.5023
private vs. indigenous	stateMS	-0.7256	0.0528	-13.7344	0.0000	-0.8291	-0.6220	-0.9834	0.0717	-13.7094	0.0000	-1.1240	-0.8428
private vs. indigenous	stateMT	-0.4835	0.0753	-6.4172	0.0000	-0.6312	-0.3358	-0.5784	0.0943	-6.1319	0.0000	-0.7632	-0.3935
private vs. indigenous	statePA	-0.3126	0.0520	-6.0072	0.0000	-0.4146	-0.2106	-0.3963	0.0771	-5.1380	0.0000	-0.5475	-0.2451
private vs. indigenous	statePB	-1.5791	0.1719	-9.1866	0.0000	-1.9160	-1.2422	-1.8066	0.1982	-9.1140	0.0000	-2.1951	-1.4181
private vs. indigenous	statePE	-1.0982	0.2126	-5.1648	0.0000	-1.5149	-0.6814	-1.6094	0.2844	-5.6598	0.0000	-2.1668	-1.0521
private vs. indigenous	statePI	-0.8514	0.1615	-5.2720	0.0000	-1.1679	-0.5349	-0.9569	0.2042	-4.6872	0.0000	-1.3571	-0.5568
private vs. indigenous	statePR	-0.7471	0.1591	-4.6955	0.0000	-1.0590	-0.4353	-0.9275	0.1835	-5.0548	0.0000	-1.2871	-0.5679
private vs. indigenous	stateRJ	-0.5791	0.1606	-3.6058	0.0003	-0.8938	-0.2643	-0.6587	0.1837	-3.5851	0.0003	-1.0189	-0.2986
private vs. indigenous	stateRN	-1.8074	0.1386	-13.0387	0.0000	-2.0791	-1.5357	-2.2174	0.1349	-16.4322	0.0000	-2.4819	-1.9529
private vs. indigenous	stateRO	-0.2270	0.0553	-4.1079	0.0000	-0.3353	-0.1187	-0.4319	0.0972	-4.4441	0.0000	-0.6224	-0.2414
private vs. indigenous	stateRR	-0.5264	0.1042	-5.0506	0.0000	-0.7306	-0.3221	-0.6316	0.1348	-4.6848	0.0000	-0.8958	-0.3673
private vs. indigenous	stateRS	-1.0539	0.1267	-8.3198	0.0000	-1.3022	-0.8056	-1.2399	0.1408	-8.8035	0.0000	-1.5159	-0.9638
private vs. indigenous	stateSC	-1.2109	0.1813	-6.6779	0.0000	-1.5664	-0.8555	-1.2659	0.1831	-6.9147	0.0000	-1.6247	-0.9071
private vs. indigenous	stateSE	-1.2055	0.0488	-24.7138	0.0000	-1.3011	-1.1099	-1.7318	0.0683	-25.3428	0.0000	-1.8657	-1.5979
private vs. indigenous	stateSP	-0.6581	0.1125	-5.8499	0.0000	-0.8786	-0.4376	-0.7799	0.1272	-6.1331	0.0000	-1.0291	-0.5307
private vs. indigenous	stateTO	-1.0119	0.1018	-9.9403	0.0000	-1.2115	-0.8124	-1.2022	0.1452	-8.2790	0.0000	-1.4868	-0.9176
private vs. indigenous	travel_time	0.0001	0.0000	4.7135	0.0000	0.0000	0.0001	0.0001	0.0000	5.7191	0.0000	0.0001	0.0002

		Species richness model estimates						Simpson's diversity index model estimates					
Model comparison	**variable**	**AME**	**SE**	**z**	**p**	**lower**	**upper**	**AME**	**SE**	**z**	**p**	**lower**	**upper**
private vs. indigenous	yry2	0.0023	0.0100	0.2266	0.8207	-0.0174	0.0219	-0.0129	0.0128	-1.0069	0.3140	-0.0381	0.0122
private vs. indigenous	yry3	-0.0082	0.0110	-0.7432	0.4574	-0.0297	0.0133	-0.0304	0.0141	-2.1584	0.0309	-0.0581	-0.0028
private vs. protected	area	0.0000	0.0000	0.2997	0.7644	0.0000	0.0000	0.0000	0.0000	-0.1304	0.8962	0.0000	0.0000
private vs. protected	cattl	0.0000	0.0000	0.8472	0.3969	0.0000	0.0000	0.0000	0.0000	2.6184	0.0088	0.0000	0.0000
private vs. protected	crop	0.0000	0.0000	0.2600	0.7949	0.0000	0.0000	0.0000	0.0000	-0.0172	0.9863	0.0000	0.0000
private vs. protected	elevation	-0.0003	0.0001	-3.0506	0.0023	-0.0005	-0.0001	-0.0005	0.0001	-4.1923	0.0000	-0.0007	-0.0002
private vs. protected	pdsi	0.0002	0.0005	0.4296	0.6675	-0.0008	0.0012	0.0000	0.0006	-0.0808	0.9356	-0.0012	0.0011
private vs. protected	pop	-0.0062	0.0035	-1.7456	0.0809	-0.0131	0.0008	-0.0110	0.0043	-2.5773	0.0100	-0.0194	-0.0026
private vs. protected	**private.vs.protected**	**0.0721**	**0.0441**	**1.6331**	**0.1024**	**-0.0144**	**0.1586**	**0.1160**	**0.0521**	**2.2262**	**0.0260**	**0.0139**	**0.2181**
private vs. protected	slope	0.0697	0.0131	5.3101	0.0000	0.0440	0.0955	0.0805	0.0148	5.4300	0.0000	0.0515	0.1096
private vs. protected	stateAL	-0.9624	0.1232	-7.8140	0.0000	-1.2038	-0.7210	-1.1679	0.1934	-6.0403	0.0000	-1.5469	-0.7890
private vs. protected	stateAM	-0.1019	0.0254	-4.0126	0.0001	-0.1516	-0.0521	-0.1399	0.0404	-3.4678	0.0005	-0.2190	-0.0608
private vs. protected	stateAP	-1.2383	0.1906	-6.4955	0.0000	-1.6119	-0.8646	-1.3341	0.1823	-7.3181	0.0000	-1.6914	-0.9768
private vs. protected	stateBA	-0.8879	0.1043	-8.5157	0.0000	-1.0923	-0.6836	-0.9953	0.1278	-7.7871	0.0000	-1.2458	-0.7448
private vs. protected	stateCE	-1.3097	0.1243	-10.5376	0.0000	-1.5533	-1.0661	-1.5024	0.1292	-11.6320	0.0000	-1.7556	-1.2493
private vs. protected	stateDF	-0.8479	0.1077	-7.8714	0.0000	-1.0590	-0.6368	-0.9728	0.1219	-7.9823	0.0000	-1.2117	-0.7340
private vs. protected	stateES	-0.7758	0.1648	-4.7090	0.0000	-1.0987	-0.4529	-1.0190	0.2085	-4.8874	0.0000	-1.4277	-0.6104
private vs. protected	stateGO	-0.9948	0.0925	-10.7521	0.0000	-1.1761	-0.8134	-1.2161	0.1178	-10.3239	0.0000	-1.4470	-0.9853
private vs. protected	stateMA	-1.0059	0.1266	-7.9480	0.0000	-1.2539	-0.7578	-1.2937	0.1096	-11.8037	0.0000	-1.5085	-1.0789
private vs. protected	stateMG	-0.8258	0.0880	-9.3867	0.0000	-0.9982	-0.6534	-1.0216	0.1028	-9.9378	0.0000	-1.2230	-0.8201
private vs. protected	stateMS	-0.8476	0.0698	-12.1400	0.0000	-0.9844	-0.7107	-1.0566	0.0801	-13.1956	0.0000	-1.2135	-0.8996
private vs. protected	stateMT	-0.6339	0.0799	-7.9310	0.0000	-0.7906	-0.4773	-0.7917	0.0989	-8.0058	0.0000	-0.9856	-0.5979
private vs. protected	statePA	-0.3581	0.0450	-7.9602	0.0000	-0.4463	-0.2699	-0.4739	0.0568	-8.3457	0.0000	-0.5852	-0.3626
private vs. protected	statePB	-1.0149	0.0702	-14.4636	0.0000	-1.1524	-0.8774	-1.1067	0.0888	-12.4566	0.0000	-1.2809	-0.9326
private vs. protected	statePE	-1.1132	0.0991	-11.2285	0.0000	-1.3075	-0.9189	-1.3018	0.1052	-12.3726	0.0000	-1.5081	-1.0956
private vs. protected	statePI	-0.8489	0.0776	-10.9346	0.0000	-1.0010	-0.6967	-0.9428	0.0878	-10.7391	0.0000	-1.1148	-0.7707
private vs. protected	statePR	-1.4210	0.1590	-8.9394	0.0000	-1.7325	-1.1094	-1.5812	0.1605	-9.8504	0.0000	-1.8958	-1.2666
private vs. protected	stateRJ	-1.0516	0.1701	-6.1838	0.0000	-1.3849	-0.7183	-1.2419	0.1863	-6.6656	0.0000	-1.6071	-0.8768
private vs. protected	stateRN	-1.3689	0.0963	-14.2129	0.0000	-1.5577	-1.1801	-1.5193	0.0946	-16.0532	0.0000	-1.7048	-1.3338
private vs. protected	stateRO	-0.2944	0.0824	-3.5704	0.0004	-0.4560	-0.1328	-0.3891	0.1069	-3.6401	0.0003	-0.5985	-0.1796
private vs. protected	stateRR	-0.4359	0.0735	-5.9345	0.0000	-0.5799	-0.2919	-0.5464	0.1145	-4.7708	0.0000	-0.7708	-0.3219
private vs. protected	stateRS	-1.3310	0.1214	-10.9669	0.0000	-1.5689	-1.0932	-1.5901	0.1409	-11.2849	0.0000	-1.8663	-1.3140
private vs. protected	stateSC	-1.2815	0.2326	-5.5089	0.0000	-1.7375	-0.8256	-1.4325	0.2300	-6.2275	0.0000	-1.8833	-0.9816
private vs. protected	stateSE	-1.2205	0.2472	-4.9372	0.0000	-1.7050	-0.7360	-1.4008	0.2809	-4.9861	0.0000	-1.9514	-0.8502
private vs. protected	stateSP	-0.6350	0.0800	-7.9350	0.0000	-0.7918	-0.4782	-0.8628	0.0915	-9.4263	0.0000	-1.0422	-0.6834

		Species richness model estimates						Simpson's diversity index model estimates					
Model comparison	**variable**	**AME**	**SE**	**z**	**p**	**lower**	**upper**	**AME**	**SE**	**z**	**p**	**lower**	**upper**
private vs. protected	stateTO	-0.9194	0.1274	-7.2150	0.0000	-1.1692	-0.6697	-1.1499	0.1595	-7.2093	0.0000	-1.4626	-0.8373
private vs. protected	travel_time	0.0001	0.0000	2.5091	0.0121	0.0000	0.0001	0.0001	0.0000	3.7882	0.0002	0.0001	0.0002
private vs. protected	yry2	-0.0008	0.0136	-0.0565	0.9549	-0.0275	0.0260	-0.0112	0.0156	-0.7203	0.4714	-0.0418	0.0193
private vs. protected	yry3	0.0036	0.0160	0.2231	0.8235	-0.0278	0.0349	-0.0137	0.0181	-0.7573	0.4489	-0.0491	0.0217
private vs. quilombola	area	0.0000	0.0000	1.2808	0.2003	0.0000	0.0000	0.0000	0.0000	1.9843	0.0472	0.0000	0.0000
private vs. quilombola	cattl	0.0000	0.0000	-0.4239	0.6716	0.0000	0.0000	0.0000	0.0000	0.1014	0.9192	0.0000	0.0000
private vs. quilombola	crop	0.0000	0.0000	0.1813	0.8561	0.0000	0.0000	0.0000	0.0000	0.4884	0.6252	0.0000	0.0000
private vs. quilombola	elevation	-0.0003	0.0001	-3.1554	0.0016	-0.0005	-0.0001	-0.0005	0.0001	-4.0060	0.0001	-0.0007	-0.0002
private vs. quilombola	pdsi	0.0000	0.0005	-0.0046	0.9963	-0.0011	0.0011	0.0002	0.0007	0.2292	0.8187	-0.0011	0.0015
private vs. quilombola	pop	-0.0121	0.0041	-2.9534	0.0031	-0.0202	-0.0041	-0.0164	0.0049	-3.3398	0.0008	-0.0260	-0.0068
private vs. quilombola	**private.vs.quilombola**	**0.0521**	**0.0511**	**1.0197**	**0.3079**	**-0.0480**	**0.1521**	**0.1193**	**0.0597**	**1.9991**	**0.0456**	**0.0023**	**0.2362**
private vs. quilombola	slope	0.0893	0.0143	6.2277	0.0000	0.0612	0.1174	0.1299	0.0191	6.7873	0.0000	0.0924	0.1674
private vs. quilombola	stateAM	1.1978	0.0865	13.8403	0.0000	1.0282	1.3674	1.3195	0.1173	11.2525	0.0000	1.0897	1.5493
private vs. quilombola	stateAP	0.3745	0.1082	3.4599	0.0005	0.1624	0.5866	0.3697	0.1260	2.9352	0.0033	0.1228	0.6166
private vs. quilombola	stateBA	0.3656	0.0621	5.8887	0.0000	0.2439	0.4872	0.3268	0.0766	4.2647	0.0000	0.1766	0.4770
private vs. quilombola	stateCE	0.2925	0.0667	4.3840	0.0000	0.1617	0.4232	0.2092	0.0592	3.5352	0.0004	0.0932	0.3251
private vs. quilombola	stateDF	0.3425	0.0758	4.5198	0.0000	0.1940	0.4911	0.3274	0.0914	3.5817	0.0003	0.1482	0.5065
private vs. quilombola	stateES	0.7730	0.0855	9.0377	0.0000	0.6054	0.9407	0.7399	0.1261	5.8671	0.0000	0.4928	0.9871
private vs. quilombola	stateGO	0.4648	0.0604	7.6950	0.0000	0.3464	0.5832	0.3970	0.0822	4.8294	0.0000	0.2359	0.5582
private vs. quilombola	stateMA	0.3899	0.0762	5.1156	0.0000	0.2405	0.5392	0.4156	0.0930	4.4667	0.0000	0.2332	0.5979
private vs. quilombola	stateMG	0.5037	0.0731	6.8915	0.0000	0.3605	0.6470	0.3910	0.0774	5.0516	0.0000	0.2393	0.5427
private vs. quilombola	stateMS	0.3600	0.0510	7.0538	0.0000	0.2600	0.4600	0.1453	0.0510	2.8507	0.0044	0.0454	0.2452
private vs. quilombola	stateMT	0.7085	0.1033	6.8558	0.0000	0.5059	0.9110	0.7128	0.1232	5.7856	0.0000	0.4714	0.9543
private vs. quilombola	statePA	0.9302	0.0702	13.2423	0.0000	0.7925	1.0678	0.9572	0.0770	12.4365	0.0000	0.8063	1.1080
private vs. quilombola	statePB	0.0528	0.0622	0.8481	0.3964	-0.0692	0.1747	0.0799	0.0716	1.1165	0.2642	-0.0604	0.2202
private vs. quilombola	statePE	0.2946	0.0813	3.6223	0.0003	0.1352	0.4541	0.3028	0.0816	3.7107	0.0002	0.1429	0.4628
private vs. quilombola	statePI	0.4295	0.0589	7.2925	0.0000	0.3140	0.5449	0.4211	0.0773	5.4498	0.0000	0.2697	0.5726
private vs. quilombola	statePR	0.3994	0.1469	2.7181	0.0066	0.1114	0.6873	0.4136	0.1505	2.7473	0.0060	0.1185	0.7086
private vs. quilombola	stateRJ	0.3779	0.1129	3.3470	0.0008	0.1566	0.5992	0.3684	0.1079	3.4131	0.0006	0.1569	0.5800
private vs. quilombola	stateRN	0.1190	0.0889	1.3379	0.1809	-0.0553	0.2933	-0.0182	0.1037	-0.1757	0.8605	-0.2216	0.1851
private vs. quilombola	stateRO	0.8453	0.1023	8.2650	0.0000	0.6448	1.0457	0.6938	0.1592	4.3572	0.0000	0.3817	1.0059
private vs. quilombola	stateRR	1.1609	0.0897	12.9399	0.0000	0.9850	1.3367	1.3881	0.1145	12.1267	0.0000	1.1637	1.6124
private vs. quilombola	stateRS	-0.0618	0.0754	-0.8203	0.4121	-0.2096	0.0859	-0.2251	0.0652	-3.4533	0.0006	-0.3529	-0.0973
private vs. quilombola	stateSC	-0.9647	0.0948	-10.1769	0.0000	-1.1505	-0.7789	-1.2054	0.1146	-10.5223	0.0000	-1.4300	-0.9809
private vs. quilombola	stateSE	0.1612	0.1165	1.3840	0.1664	-0.0671	0.3894	0.0631	0.1155	0.5463	0.5848	-0.1633	0.2896

Model comparison	variable	Species richness model estimates AME	SE	z	p	lower	upper	Simpson's diversity index model estimates AME	SE	z	p	lower	upper
private vs. quilombola	stateSP	0.5519	0.0864	6.3887	0.0000	0.3826	0.7212	0.5364	0.1076	4.9830	0.0000	0.3254	0.7473
private vs. quilombola	stateTO	0.5383	0.0571	9.4284	0.0000	0.4264	0.6502	0.5458	0.0611	8.9377	0.0000	0.4261	0.6655
private vs. quilombola	travel_time	0.0001	0.0000	2.1935	0.0283	0.0000	0.0001	0.0002	0.0000	3.3974	0.0007	0.0001	0.0003
private vs. quilombola	yry2	-0.0330	0.0155	-2.1271	0.0334	-0.0634	-0.0026	-0.0787	0.0188	-4.1881	0.0000	-0.1156	-0.0419
private vs. quilombola	yry3	-0.0200	0.0192	-1.0399	0.2984	-0.0577	0.0177	-0.0670	0.0240	-2.7928	0.0052	-0.1140	-0.0200
private vs. ruralSettlemt	area	0.0000	0.0000	1.1213	0.2622	0.0000	0.0000	0.0000	0.0000	1.5047	0.1324	0.0000	0.0000
private vs. ruralSettlemt	cattl	0.0000	0.0000	-3.3927	0.0007	0.0000	0.0000	0.0000	0.0000	-5.9843	0.0000	0.0000	0.0000
private vs. ruralSettlemt	crop	0.0000	0.0000	-1.8340	0.0667	0.0000	0.0000	0.0000	0.0000	-0.5912	0.5544	0.0000	0.0000
private vs. ruralSettlemt	elevation	-0.0004	0.0000	-7.7410	0.0000	-0.0005	-0.0003	-0.0006	0.0001	-10.2260	0.0000	-0.0007	-0.0004
private vs. ruralSettlemt	pdsi	0.0014	0.0003	4.8560	0.0000	0.0008	0.0019	0.0013	0.0003	3.9415	0.0001	0.0007	0.0020
private vs. ruralSettlemt	pop	-0.0108	0.0019	-5.6416	0.0000	-0.0146	-0.0071	-0.0127	0.0022	-5.6965	0.0000	-0.0170	-0.0083
private vs. ruralSettlemt	**private.vs.ruralSettlemt**	**0.0017**	**0.0128**	**0.1314**	**0.8954**	**-0.0234**	**0.0268**	**0.0102**	**0.0148**	**0.6916**	**0.4892**	**-0.0188**	**0.0393**
private vs. ruralSettlemt	slope	0.0871	0.0065	13.3283	0.0000	0.0743	0.0999	0.1234	0.0076	16.1818	0.0000	0.1085	0.1384
private vs. ruralSettlemt	stateAL	-1.2370	0.0627	-19.7216	0.0000	-1.3600	-1.1141	-1.3574	0.0547	-24.7984	0.0000	-1.4647	-1.2501
private vs. ruralSettlemt	stateAM	-0.1635	0.0382	-4.2749	0.0000	-0.2384	-0.0885	-0.1620	0.0547	-2.9611	0.0031	-0.2693	-0.0548
private vs. ruralSettlemt	stateAP	-0.6574	0.1519	-4.3271	0.0000	-0.9552	-0.3597	-0.6364	0.1771	-3.5943	0.0003	-0.9835	-0.2894
private vs. ruralSettlemt	stateBA	-0.6806	0.0344	-19.7758	0.0000	-0.7481	-0.6132	-0.7842	0.0505	-15.5230	0.0000	-0.8832	-0.6852
private vs. ruralSettlemt	stateCE	-0.9967	0.0263	-37.9117	0.0000	-1.0483	-0.9452	-1.0796	0.0381	-28.3540	0.0000	-1.1542	-1.0050
private vs. ruralSettlemt	stateDF	-0.5895	0.0464	-12.7078	0.0000	-0.6805	-0.4986	-0.8047	0.0592	-13.6048	0.0000	-0.9207	-0.6888
private vs. ruralSettlemt	stateES	-0.7600	0.0517	-14.6939	0.0000	-0.8614	-0.6586	-0.9710	0.0679	-14.3017	0.0000	-1.1041	-0.8380
private vs. ruralSettlemt	stateGO	-0.7504	0.0336	-22.3083	0.0000	-0.8164	-0.6845	-0.9308	0.0458	-20.3347	0.0000	-1.0205	-0.8411
private vs. ruralSettlemt	stateMA	-0.8021	0.0333	-24.0598	0.0000	-0.8674	-0.7367	-0.9058	0.0457	-19.8348	0.0000	-0.9953	-0.8163
private vs. ruralSettlemt	stateMG	-0.7331	0.0374	-19.6116	0.0000	-0.8063	-0.6598	-0.9018	0.0507	-17.7882	0.0000	-1.0012	-0.8025
private vs. ruralSettlemt	stateMS	-0.7827	0.0349	-22.4278	0.0000	-0.8511	-0.7143	-0.9824	0.0414	-23.7051	0.0000	-1.0637	-0.9012
private vs. ruralSettlemt	stateMT	-0.5073	0.0316	-16.0420	0.0000	-0.5693	-0.4453	-0.6515	0.0443	-14.7048	0.0000	-0.7383	-0.5646
private vs. ruralSettlemt	statePA	-0.4624	0.0415	-11.1354	0.0000	-0.5438	-0.3810	-0.5603	0.0559	-10.0181	0.0000	-0.6699	-0.4507
private vs. ruralSettlemt	statePB	-1.0413	0.0460	-22.6279	0.0000	-1.1315	-0.9511	-1.1980	0.0575	-20.8335	0.0000	-1.3107	-1.0853
private vs. ruralSettlemt	statePE	-0.9861	0.0371	-26.5983	0.0000	-1.0588	-0.9134	-1.0715	0.0459	-23.3258	0.0000	-1.1615	-0.9814
private vs. ruralSettlemt	statePI	-0.7659	0.0277	-27.6537	0.0000	-0.8202	-0.7117	-0.8451	0.0415	-20.3435	0.0000	-0.9265	-0.7637
private vs. ruralSettlemt	statePR	-1.0172	0.0583	-17.4333	0.0000	-1.1316	-0.9029	-1.1883	0.0645	-18.4341	0.0000	-1.3147	-1.0620
private vs. ruralSettlemt	stateRJ	-0.6702	0.0498	-13.4689	0.0000	-0.7678	-0.5727	-1.0023	0.0857	-11.6999	0.0000	-1.1702	-0.8344
private vs. ruralSettlemt	stateRN	-1.0970	0.0382	-28.6828	0.0000	-1.1720	-1.0221	-1.2571	0.0575	-21.8469	0.0000	-1.3699	-1.1443
private vs. ruralSettlemt	stateRO	-0.1739	0.0329	-5.2862	0.0000	-0.2384	-0.1094	-0.2626	0.0624	-4.2047	0.0000	-0.3850	-0.1402
private vs. ruralSettlemt	stateRR	-0.3276	0.0722	-4.5394	0.0000	-0.4690	-0.1861	-0.2895	0.0924	-3.1328	0.0017	-0.4706	-0.1084
private vs. ruralSettlemt	stateRS	-1.2041	0.0415	-28.9863	0.0000	-1.2855	-1.1227	-1.4113	0.0472	-29.8717	0.0000	-1.5039	-1.3187

		Species richness model estimates						Simpson's diversity index model estimates					
Model comparison	**variable**	**AME**	**SE**	**z**	**p**	**lower**	**upper**	**AME**	**SE**	**z**	**p**	**lower**	**upper**
private vs. ruralSettlemt	stateSC	-1.3268	0.0783	-16.9350	0.0000	-1.4803	-1.1732	-1.4836	0.0827	-17.9324	0.0000	-1.6457	-1.3214
private vs. ruralSettlemt	stateSE	-0.8643	0.0303	-28.5638	0.0000	-0.9236	-0.8050	-1.1436	0.0554	-20.6571	0.0000	-1.2521	-1.0351
private vs. ruralSettlemt	stateSP	-0.8732	0.0756	-11.5466	0.0000	-1.0214	-0.7250	-1.0538	0.0705	-14.9563	0.0000	-1.1919	-0.9157
private vs. ruralSettlemt	stateTO	-0.7751	0.0370	-20.9400	0.0000	-0.8476	-0.7025	-0.8596	0.0511	-16.8266	0.0000	-0.9597	-0.7595
private vs. ruralSettlemt	travel_time	0.0002	0.0000	10.6511	0.0000	0.0001	0.0002	0.0003	0.0000	12.8235	0.0000	0.0002	0.0003
private vs. ruralSettlemt	yry2	0.0016	0.0058	0.2723	0.7854	-0.0097	0.0129	-0.0345	0.0082	-4.2286	0.0000	-0.0505	-0.0185
private vs. ruralSettlemt	yry3	0.0008	0.0071	0.1092	0.9130	-0.0132	0.0147	-0.0367	0.0096	-3.8371	0.0001	-0.0555	-0.0180
private vs. sustainableUse	area	0.0000	0.0000	1.2586	0.2082	0.0000	0.0000	0.0000	0.0000	0.9403	0.3471	0.0000	0.0000
private vs. sustainableUse	cattl	0.0000	0.0000	-0.2927	0.7697	0.0000	0.0000	0.0000	0.0000	1.0257	0.3050	0.0000	0.0000
private vs. sustainableUse	crop	0.0000	0.0000	0.7967	0.4256	0.0000	0.0000	0.0000	0.0000	-0.4789	0.6320	0.0000	0.0000
private vs. sustainableUse	elevation	-0.0009	0.0002	-5.3951	0.0000	-0.0013	-0.0006	-0.0012	0.0002	-6.0975	0.0000	-0.0016	-0.0008
private vs. sustainableUse	pdsi	-0.0009	0.0004	-2.1142	0.0345	-0.0016	-0.0001	-0.0011	0.0005	-2.1644	0.0304	-0.0020	-0.0001
private vs. sustainableUse	pop	-0.0033	0.0091	-0.3591	0.7195	-0.0210	0.0145	-0.0084	0.0099	-0.8457	0.3977	-0.0278	0.0110
private vs. sustainableUse	**private.vs.sustainableUse**	**-0.0666**	**0.0585**	**-1.1386**	**0.2549**	**-0.1812**	**0.0480**	**-0.0304**	**0.0628**	**-0.4837**	**0.6286**	**-0.1534**	**0.0927**
private vs. sustainableUse	slope	0.0597	0.0228	2.6159	0.0089	0.0150	0.1045	0.0874	0.0242	3.6090	0.0003	0.0399	0.1349
private vs. sustainableUse	stateAL	-1.2813	0.1429	-8.9664	0.0000	-1.5614	-1.0012	-1.5729	0.1406	-11.1865	0.0000	-1.8485	-1.2973
private vs. sustainableUse	stateAM	-0.1521	0.0453	-3.3549	0.0008	-0.2410	-0.0632	-0.1799	0.0687	-2.6185	0.0088	-0.3145	-0.0452
private vs. sustainableUse	stateAP	-1.1069	0.4100	-2.6996	0.0069	-1.9105	-0.3033	-1.1966	0.4256	-2.8119	0.0049	-2.0307	-0.3626
private vs. sustainableUse	stateBA	-0.6569	0.1098	-5.9805	0.0000	-0.8722	-0.4416	-0.7442	0.1427	-5.2162	0.0000	-1.0238	-0.4646
private vs. sustainableUse	stateCE	-1.1543	0.2046	-5.6419	0.0000	-1.5554	-0.7533	-1.2934	0.2178	-5.9386	0.0000	-1.7202	-0.8665
private vs. sustainableUse	stateDF	-0.0466	0.1743	-0.2674	0.7892	-0.3882	0.2950	0.1116	0.2135	0.5226	0.6012	-0.3069	0.5301
private vs. sustainableUse	stateES	-0.6579	0.1904	-3.4554	0.0005	-1.0311	-0.2847	-0.8247	0.2044	-4.0342	0.0001	-1.2254	-0.4240
private vs. sustainableUse	stateGO	-0.5946	0.1642	-3.6215	0.0003	-0.9164	-0.2728	-0.7074	0.1978	-3.5761	0.0003	-1.0951	-0.3197
private vs. sustainableUse	stateMA	-1.1283	0.1314	-8.5899	0.0000	-1.3857	-0.8708	-1.4082	0.1563	-9.0086	0.0000	-1.7146	-1.1019
private vs. sustainableUse	stateMG	-0.4957	0.2035	-2.4355	0.0149	-0.8946	-0.0968	-0.6212	0.2371	-2.6202	0.0088	-1.0859	-0.1565
private vs. sustainableUse	stateMS	-0.6169	0.1073	-5.7494	0.0000	-0.8272	-0.4066	-0.8197	0.1253	-6.5427	0.0000	-1.0653	-0.5742
private vs. sustainableUse	stateMT	-0.3465	0.0822	-4.2145	0.0000	-0.5077	-0.1854	-0.4842	0.1164	-4.1610	0.0000	-0.7122	-0.2561
private vs. sustainableUse	statePA	-0.5268	0.0697	-7.5605	0.0000	-0.6634	-0.3903	-0.6639	0.0878	-7.5619	0.0000	-0.8360	-0.4918
private vs. sustainableUse	statePB	-1.5730	0.2029	-7.7522	0.0000	-1.9707	-1.1753	-1.6967	0.1993	-8.5144	0.0000	-2.0872	-1.3061
private vs. sustainableUse	statePE	-1.1684	0.1651	-7.0764	0.0000	-1.4920	-0.8448	-1.2443	0.1614	-7.7075	0.0000	-1.5607	-0.9279
private vs. sustainableUse	statePI	-0.9795	0.0806	-12.1559	0.0000	-1.1374	-0.8216	-1.1517	0.1019	-11.3020	0.0000	-1.3515	-0.9520
private vs. sustainableUse	statePR	-0.7244	0.4153	-1.7444	0.0811	-1.5383	0.0895	-0.7467	0.4941	-1.5110	0.1308	-1.7152	0.2218
private vs. sustainableUse	stateRJ	-0.5112	0.2746	-1.8620	0.0626	-1.0493	0.0269	-0.5912	0.2841	-2.0809	0.0374	-1.1481	-0.0344
private vs. sustainableUse	stateRN	-1.1371	0.1020	-11.1433	0.0000	-1.3371	-0.9371	-1.3675	0.1119	-12.2155	0.0000	-1.5869	-1.1481
private vs. sustainableUse	stateRO	-0.0657	0.0711	-0.9233	0.3558	-0.2051	0.0737	-0.0970	0.0946	-1.0247	0.3055	-0.2824	0.0885

		Species richness model estimates						Simpson's diversity index model estimates					
Model comparison	**variable**	**AME**	**SE**	**z**	**p**	**lower**	**upper**	**AME**	**SE**	**z**	**p**	**lower**	**upper**
private vs. sustainableUse	stateRR	-0.0922	0.0431	-2.1405	0.0323	-0.1765	-0.0078	0.0041	0.0618	0.0671	0.9465	-0.1170	0.1253
private vs. sustainableUse	stateRS	-1.0105	0.3134	-3.2240	0.0013	-1.6248	-0.3962	-1.0919	0.3351	-3.2586	0.0011	-1.7487	-0.4352
private vs. sustainableUse	stateSC	-1.1494	0.2451	-4.6893	0.0000	-1.6298	-0.6690	-1.1578	0.2593	-4.4648	0.0000	-1.6660	-0.6495
private vs. sustainableUse	stateSE	-0.8649	0.1287	-6.7195	0.0000	-1.1172	-0.6126	-1.2245	0.1699	-7.2065	0.0000	-1.5575	-0.8915
private vs. sustainableUse	stateSP	-0.4007	0.1679	-2.3874	0.0170	-0.7297	-0.0717	-0.5567	0.1874	-2.9705	0.0030	-0.9240	-0.1894
private vs. sustainableUse	stateTO	-0.5987	0.0795	-7.5352	0.0000	-0.7544	-0.4430	-0.7314	0.1353	-5.4042	0.0000	-0.9966	-0.4661
private vs. sustainableUse	travel_time	0.0001	0.0000	3.8908	0.0001	0.0000	0.0001	0.0002	0.0000	5.5066	0.0000	0.0001	0.0002
private vs. sustainableUse	yry2	-0.0026	0.0164	-0.1558	0.8762	-0.0347	0.0296	-0.0256	0.0207	-1.2395	0.2151	-0.0662	0.0149
private vs. sustainableUse	yry3	-0.0085	0.0192	-0.4404	0.6596	-0.0461	0.0292	-0.0287	0.0233	-1.2302	0.2186	-0.0745	0.0170
private vs. undesignated	area	0.0000	0.0000	-1.1614	0.2455	0.0000	0.0000	0.0000	0.0000	-0.6441	0.5195	0.0000	0.0000
private vs. undesignated	cattl	0.0000	0.0000	0.0614	0.9511	0.0000	0.0000	0.0000	0.0000	-5.4232	0.0000	0.0000	0.0000
private vs. undesignated	crop	0.0000	0.0000	-3.4883	0.0005	0.0000	0.0000	0.0000	0.0000	-3.1078	0.0019	0.0000	0.0000
private vs. undesignated	elevation	0.0000	0.0001	0.1394	0.8891	-0.0001	0.0002	0.0000	0.0001	-0.4330	0.6650	-0.0002	0.0001
private vs. undesignated	pdsi	0.0006	0.0003	1.8509	0.0642	0.0000	0.0012	0.0007	0.0004	1.7633	0.0778	-0.0001	0.0015
private vs. undesignated	pop	-0.0044	0.0041	-1.0740	0.2828	-0.0123	0.0036	-0.0105	0.0046	-2.2632	0.0236	-0.0196	-0.0014
private vs. undesignated	**private.vs.undesignated**	**0.0283**	**0.0193**	**1.4680**	**0.1421**	**-0.0095**	**0.0661**	**0.0486**	**0.0241**	**2.0200**	**0.0434**	**0.0014**	**0.0957**
private vs. undesignated	slope	0.0496	0.0083	5.9972	0.0000	0.0334	0.0658	0.0637	0.0102	6.2530	0.0000	0.0438	0.0837
private vs. undesignated	stateAL	-1.0560	0.1440	-7.3320	0.0000	-1.3383	-0.7737	-1.2225	0.0966	-12.6503	0.0000	-1.4119	-1.0331
private vs. undesignated	stateAM	-0.1154	0.0379	-3.0478	0.0023	-0.1896	-0.0412	-0.1293	0.0631	-2.0490	0.0405	-0.2530	-0.0056
private vs. undesignated	stateAP	-0.6067	0.1877	-3.2323	0.0012	-0.9745	-0.2388	-0.6397	0.2179	-2.9359	0.0033	-1.0667	-0.2126
private vs. undesignated	stateBA	-0.9718	0.0584	-16.6403	0.0000	-1.0862	-0.8573	-1.1305	0.0825	-13.7017	0.0000	-1.2922	-0.9688
private vs. undesignated	stateCE	-1.0275	0.0299	-34.3975	0.0000	-1.0861	-0.9690	-1.1339	0.0533	-21.2544	0.0000	-1.2384	-1.0293
private vs. undesignated	stateDF	-1.7377	0.0692	-25.1061	0.0000	-1.8734	-1.6020	-1.8788	0.0911	-20.6336	0.0000	-2.0573	-1.7004
private vs. undesignated	stateES	-0.5666	0.1792	-3.1621	0.0016	-0.9178	-0.2154	-0.6518	0.2601	-2.5063	0.0122	-1.1615	-0.1421
private vs. undesignated	stateGO	-0.9180	0.0680	-13.4961	0.0000	-1.0513	-0.7847	-1.0042	0.0821	-12.2298	0.0000	-1.1651	-0.8433
private vs. undesignated	stateMA	-0.8233	0.0421	-19.5456	0.0000	-0.9058	-0.7407	-0.9704	0.0624	-15.5419	0.0000	-1.0928	-0.8480
private vs. undesignated	stateMG	-0.8547	0.1024	-8.3481	0.0000	-1.0554	-0.6541	-1.1364	0.1320	-8.6101	0.0000	-1.3951	-0.8777
private vs. undesignated	stateMS	-0.8516	0.0427	-19.9412	0.0000	-0.9353	-0.7679	-1.0778	0.0588	-18.3312	0.0000	-1.1930	-0.9625
private vs. undesignated	stateMT	-0.5670	0.0420	-13.4908	0.0000	-0.6494	-0.4846	-0.7217	0.0640	-11.2801	0.0000	-0.8470	-0.5963
private vs. undesignated	statePA	-0.3196	0.0376	-8.4991	0.0000	-0.3933	-0.2459	-0.4041	0.0639	-6.3209	0.0000	-0.5294	-0.2788
private vs. undesignated	statePB	-1.1450	0.0717	-15.9688	0.0000	-1.2855	-1.0044	-1.2822	0.0828	-15.4872	0.0000	-1.4444	-1.1199
private vs. undesignated	statePE	-1.0600	0.0869	-12.2022	0.0000	-1.2302	-0.8897	-1.2477	0.1071	-11.6516	0.0000	-1.4576	-1.0378
private vs. undesignated	statePI	-0.8988	0.0418	-21.5198	0.0000	-0.9806	-0.8169	-1.0454	0.0687	-15.2064	0.0000	-1.1801	-0.9107
private vs. undesignated	stateRJ	-0.5226	0.0418	-12.5156	0.0000	-0.6045	-0.4408	-0.8997	0.0642	-14.0100	0.0000	-1.0255	-0.7738
private vs. undesignated	stateRN	-1.0602	0.0915	-11.5935	0.0000	-1.2395	-0.8810	-1.2884	0.0979	-13.1640	0.0000	-1.4803	-1.0966

		Species richness model estimates						Simpson's diversity index model estimates					
Model comparison	**variable**	**AME**	**SE**	**z**	**p**	**lower**	**upper**	**AME**	**SE**	**z**	**p**	**lower**	**upper**
private vs. undesignated	stateRO	-0.1617	0.0612	-2.6424	0.0082	-0.2817	-0.0418	-0.2740	0.1185	-2.3134	0.0207	-0.5062	-0.0419
private vs. undesignated	stateRR	-0.2981	0.0542	-5.5028	0.0000	-0.4043	-0.1920	-0.3147	0.0857	-3.6721	0.0002	-0.4826	-0.1467
private vs. undesignated	stateSC	-0.8437	0.0647	-13.0337	0.0000	-0.9706	-0.7168	-0.9387	0.1787	-5.2524	0.0000	-1.2889	-0.5884
private vs. undesignated	stateSE	-0.7452	0.0481	-15.4859	0.0000	-0.8395	-0.6509	-1.0044	0.1623	-6.1880	0.0000	-1.3225	-0.6862
private vs. undesignated	stateSP	-1.0475	0.2866	-3.6542	0.0003	-1.6093	-0.4857	-1.2551	0.2015	-6.2280	0.0000	-1.6501	-0.8601
private vs. undesignated	stateTO	-0.8272	0.0560	-14.7728	0.0000	-0.9370	-0.7175	-0.9214	0.0796	-11.5737	0.0000	-1.0774	-0.7654
private vs. undesignated	travel_time	0.0001	0.0000	6.4368	0.0000	0.0001	0.0002	0.0002	0.0000	7.7707	0.0000	0.0002	0.0003
private vs. undesignated	yry2	-0.0090	0.0093	-0.9682	0.3329	-0.0273	0.0093	-0.0492	0.0135	-3.6409	0.0003	-0.0757	-0.0227
private vs. undesignated	yry3	-0.0073	0.0086	-0.8478	0.3965	-0.0240	0.0095	-0.0503	0.0127	-3.9689	0.0001	-0.0752	-0.0255

Curriculum Vitae

Gracia Andrea Pacheco Figueroa

Education

Ph.D. 2017 – 2022	Environmental economics/Macroecology Thesis title: Influence of Land Tenure on Global Environmental Change:effects on deforestation and biodiversity in Brazil German Centre for Integrative Biodiversity Research (iDiv)/Martin-Luther-Universität Halle-Wittenberg
M.Sc. 2015-2016	Management and Conservation of Tropical Forests and Biodiversity Thesis title: Climate Change Governance: the case of land-use based emissions MRV systems in Central America and the Dominican Republic Tropical Agricultural Research and Higher Education Center (CATIE), Costa Rica. Final grade: 91.48 (out of 100)
B.A. 2009-2013	Major: Intercultural Studies – concentration in International Development Minor: Biology Houghton College (New York, USA) Final grade: 3.648 (out of 4.0) (Cum Laude honors)

Work experience

Feb 2022-2024	Fellow for the upcoming IPBES Nexus Assessment Intergovernmental Science-Policy Platform for Biodiversity and Ecosystem Services (IPBES) (Bonn, Germany) Co-produce parts of the thematic assessment on the interlinkages among biodiversity, water, food, and health, specifically working on Chapter 6: options for delivering sustainable approaches to public and private finance
Oct 2017 – Sep 2021	Scientific employee/Doctoral researcher German Centre for Integrative Biodiversity Research (iDiv) (Leipzig, Germany) Conducted research over large spatiotemporal scales using econometric analyses, spatially explicit models, and synthesizing results across scales to understand the effects of land tenure on environmental conservation outcomes in Brazil.
Aug 2016 – May 2017	Research assistant Cátedra Latinoamericana en Decisiones Ambientales para el Cambio Global (CLADA)/ LEDSenLAC: Low-emissions Development Strategies in Latin America and the Caribbean (Turrialba, Costa Rica) Data cleaning, social-network analysis, interviews, and support for ongoing research on drought and risk governance in Costa Rica as well as climate change mitigation strategies in Latin America.
Aug. 2014 – Dec 2014	Consultant MOPAWI (La Mosquitia, Honduras) Interviews and systematization of the non-profit's work in conservation and indigenous people's rights over the past 30 years.

Sept. 2014 – Dec. 2014 Consultant
Association for a More Just Society (AJS) (Tegucigalpa, Honduras)

Editing and adaptation of the "Impacto Juvenil" program curriculum for at-risk youth in the region.

Sept. 2010 – May 2012 Teaching Assistant
Houghton College (Houghton, New York)

Assisted and taught undergraduate courses in:
Spanish grammar and conversation, Scuba Diving study abroad in Honduras, Food Security and Public Health study abroad in Ecuador

Publications

Andrea Pacheco, Carsten Meyer. Land tenure drives Brazil's deforestation rates across socio-environmental contexts. Nature Communications 13, 5759 (2022). https://doi.org/10.1038/s41467-022-33398-3

Andrea Pacheco, Ruben Remelgado, Eduardo Arlé, Carsten Meyer. Effects of land-tenure regimes on biodiversity change in Brazil. *(in preparation)*

Lucía Zarba, María Piquer-Rodríguez, Sebastien Boillat, Christian Levers, … **Andrea Pacheco…** Mapping and Characterizing of Social Ecological Land Systems of South America. *Ecology and Society* (2022)

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Conferences & invited talks

"Integrating Land System Science and Conservation Science for deeper insights into conservation challenges and opportunities" (Conference symposium). *6th European Congress of Conservation Biology*, Prague, Czech Republic, August 2022

"Land tenure regimes determine tropical deforestation rates across socio-environmental contexts" (Conference talk). *26th Annual Conference of the European Association of Environmental and Resource Economists*, Berlin (online only), Germany, June 2021.

"The Importance of Tenure Form in Avoiding Forest Conversion: 30+ Years of Evidence from Brazil" (Brownbag seminar). USAID. April 2021.

"Effects of land tenure form on deforestation in Brazil: preliminary results from a quasi-experimental setup" (Conference talk). *iDiv Conference*, Leipzig, Germany, August 2019. (awarded 3rd place for best conference talk)

"Effect of land tenure on agriculture-forest dynamics: preliminary results from Latin America" (Conference talk). *Open Science Meeting, Global Land Programme*, Bern, Switzerland, April 2019.

"Towards a global land governance database" (Poster presentation). *Land and Poverty Conference*, World Bank, Washington D.C., March 2018.

List of publications and author contributions

Sections of the methods, results and discussion (**2.2-2.4**, **3.2-3.3**, and **4.2**) appear as a modified version in: Pacheco, A., and Meyer C., (2022). Land tenure drives Brazil's deforestation rates across socio-environmental contexts. As indicated by the Contributor Roles Taxonomy (CRediT), author roles were:

AP: conceptualization, methodology, software, validation, formal analysis, investigation, resources, data curation, writing – original draft preparation, writing – review and editing, visualization, project administration.

CM: conceptualization, methodology, validation, writing – original draft preparation, writing – review and editing, supervision, project administration, funding acquisition.

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Eigenständigkeitserklärung

Hiermit erkläre ich, dass die Arbeit mit dem Titel “*Influence of Land Tenure on Global Environmental Change: effects on deforestation and biodiversity in Brazil*” bisher weder bei der Naturwissenschaftlichen Fakultät III Agrar und Ernährungswissenschaften, Geowissenschaften und Informatik der Martin-Luther-Universität Halle-Wittenberg noch einer anderen wissenschaftlichen Einrichtung zum Zweck der Promotion vorgelegt Wurde.

Ferner erkläre ich, dass ich die vorliegende Arbeit selbstständig und ohne fremde Hilfe verfasst sowie keine anderen als die angegebenen Quellen und Hilfsmittel benutzt habe. Die den Werken wörtlich oder inhaltlich entnommenen Stellen wurden als solche von mir kenntlich gemacht. Ich erkläre weiterhin, dass ich mich bisher noch nie um einen Doktorgrad beworben habe.

Halle (Saale), den . .2022

Andrea Pacheco