Land-tenure regimes determine tropical deforestation rates across socio-environmental contexts

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Abstract

Many tropical forestlands are experiencing changes in land-tenure regimes, but how these changes may affect deforestation rates remains theoretically and empirically ambiguous. Using Brazil's uniquely comprehensive land-tenure and deforestation data and quasi-experimental methods, we analyzed causal effects of six alternative tenure regimes on deforestation across 49 spatiotemporal scales corresponding to distinct regional-historical contexts. We find that poorly defined public tenure regimes caused increased deforestation relative to any alternative regime in most contexts. Private tenure often reduced this deforestation, but did so less effectively and less reliably than alternative well-defined regimes, except for remote regions where on-the-ground government control is limited and where there are extensive private-actor-targeting environmental policies. Directly privatizing public conservation or indigenous lands, in turn, would almost always increase long-term deforestation. The results of our cross-scale synthesis inform how conservation, titling, and other tenure-intervention policies may align with climate-change mitigation, biodiversity-protection, and broader environmental sustainability goals. They are directly relevant to ongoing political debates about privatizing Amazonian conservation and indigenous lands and, given their high generality across diverse socio-environmental settings, potentially applicable in other tropical forest regions that model their forest policies after those in Brazil.

Main Text

Tropical deforestation, mostly via conversions of forestlands to agriculture or other human-dominated systems, causes widespread degradation of biodiversity (1) and carbon stocks (2). Land tenure rights regulate how and by whom tropical forestlands can be used, and are thus central to deforestation-related sustainability challenges (3). Land tenure rights are also fiercely contested, leading to shifts in land-tenure regimes in many tropical forest nations. On the one hand, governments place public lands under protection or respond to land claims of indigenous groups, local communities, or landless settlers (4, 5). On the other hand, private tenure rights are promoted by liberalizing state control and opening various land-based sectors to privatization (6), or restricted through land reforms or environmental policies (3).

The shifts in land-tenure regimes* resulting from these interventions may have long-run impacts on deforestation rates. Diverse interest groups use claims of improved forest conservation to promote different – often mutually conflicting – tenure interventions ranging from privatization to recognition of communal rights. Policy-makers deciding on these politically charged processes require robust information on the most likely, long-term effects of different interventions on forests. In particular, government programs and NGOs need generalizable knowledge to design robust overall strategies with respect to different land-tenure forms or interventions, especially in many tropical regions where capacity for context-specific assessments is often limited. However, scientific insights remain ambiguous. Firstly, theoretical predictions on the effects of different land-tenure regimes often contradict one another (**Table 1**, **Table S1**). Secondly, partly due to data limitations (7), empirical synthesis has been constrained to meta-studies across case studies of limited comparability (8–10), and to large-*n* but single-scale studies focused on one or few tenure regimes (11, 12). To date, systematic large-*n* assessments of the effects of alternative tenure regimes on deforestation across different scales or regional and temporal contexts are lacking, hampering robust generalizations on the most likely long-term effects of land-tenure policies.

Here, we provide such systematic testing and synthesis of land-tenure effects on tropical deforestation across different spatiotemporal contexts (see **Methods**; details in ***SI Appendix***). We analyzed 33 years of agriculture-driven deforestation across Brazilian forestlands, which harbor the world's largest biodiversity and living carbon stores, but are under pressure from ambitious agroeconomic development (13, 14). We capitalize on Brazil's uniquely comprehensive data on both land tenure (15) and land-use changes (16), and use quasi-experimental approaches to quantify deforestation effects (**Methods**). To explore likely long-term deforestation effects of land-tenure shifts in tropical regions resulting from major intervention trends such as (re)designation of public lands, communal or private titling, registration, or privatization, we compare six alternative tenure regimes against two counterfactuals, *i)* undesignated and untitled public lands with poorly defined

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* Here, we define 'land-tenure regime' as the combination of tenure-related governance factors that exist over a given parcel of land and are stable over a certain period of time. This includes the 'bundle of rights' associated with the respective tenure category (Table S2), but also the implications that these rights may have for tenure security, as well as the tenure category's predisposition for being subject to particular types of policies or regulations.

tenure rights (hereafter 'undesignated/untitled') and *ii)* individually held private lands (hereafter 'private').

Table 1. Exemplary hypothesized deforestation effects of different tenure regimes and regime changes. For a given tenure regime or regime change, both deforestation-promoting and deforestation-inhibiting effects may be expected via different, often non-mutually exclusive, causal mechanisms. A broader overview of hypotheses with reference to the bundles of rights associated with tenure regimes that mediate these mechanisms is provided in Tables S1-2.

Tenure regime / regime changes	Predicted long-term effect	Hypothesized mechanisms
Leaving public lands undesignated to any use, and untitled (if occupied)	Deforestation-inhibiting	Undesignated/untitled status inhibits forest-displacing land-use activities, both because untitled settlers cannot easily access credit and because the uncertainty regarding applicable regulations discourages outside investments, making these lands *de facto* reserves (37, 38).
	Deforestation-promoting	Undesignated/untitled lands lack both clear supervision by any designated agency (39) and effective exclusion rights. As a result, they often become *de facto* open-access environments and as such, are prone to unsustainable exploitation by rational-strategic agents (40–42). Governments rarely place restrictions on deforesting undesignated/untitled public lands – or even incentivize it by granting claims based on prior clearance (43), or by allowing settlement conditionally on putting the land to productive use (44). Due to relatively higher land prices for existing private lands on formal markets, poor small-holders or landless individuals searching for land may see themselves forced to clear undesignated/untitled lands at the development 'frontier' (45).
Replacing undesignated/untitled with private tenure through registration, regularization, or titling	Deforestation-inhibiting	Being granted private tenure rights incentivizes settlers to make longer-term investments in forest-conserving land uses because the extensive exclusion and due-process rights of private landholders reduce their risk of financial default through outside invasion or government seizure (40), thus providing assurance that they will be the sole beneficiaries of their investments. Private titles enable improved enforcement of environmental policies as they facilitate holding specific individuals accountable for complying with environmental obligations (20), such as the obligation to retain certain amounts of forest under Brazil's Forest Code.
	Deforestation-promoting	The lower default risk combined with comprehensive withdrawal and alienation rights of private tenure regimes sparks investments in forest-displacing activities (46). For example, private landholders can more easily access credit to expand their agricultural fields by using land as collateral (38). Similarly, sell and lease rights will under functioning land markets result in an eventual transfer of land to whoever can use it most profitably, which will most typically be through an agricultural use (47).
Recognizing claimed land rights of indigenous or local communities	Deforestation-inhibiting	Communities collectively holding land typically create societal rules about resource use. Community members tend to follow these to avoid social exclusion, leading to reduced degradation of communally regulated forest resources (48).
	Deforestation-promoting	Communities will often fail at effectively managing common forest resources, due to different impediments to collective action, such as free-riding and conflicting interests (49).
Privatizing any lands under statutory public ownership, including those under indigenous or conservation regimes	Deforestation-inhibiting	Public institutions often provide ineffective forest governance, e.g., due to limited monitoring and enforcement capacity, high corruption (39), or liberal granting of use concessions for short-term state revenues (48). Even those publicly owned forests that are under private or community-based management will not be used sustainably in countries with a history of short-lived government institutions, as government proposals for sustaining these resources for long-term benefits will lack credibility (50). Privatization of public lands promotes the more sustainable, productive use of natural resources by enabling more agile, innovative, and thus effective use at the production margin (39) and internalizing long-term costs of degradation into decisions (51).
	Deforestation-promoting	Individual tenure regimes fail to fully internalize non-monetary (e.g., biodiversity, cultural) or future values of forest resources that accrue mainly to society, rather than the individual. Thus, state-controlled forest governance is necessary for maintaining forest where this is not the most profitable land-use form (39).

Results and Discussion

We found that 17.4% of Brazil's originally forested 30-m pixels lost forest to agriculture between 1985 and 2018 (**Fig. 1a**). The vast majority of this deforestation occurred on private (78%) and undesignated/untitled lands (19%; **Fig 1c**). The latter 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. Such undesignated/untitled tenure regimes cover vast areas across the tropics, and in Brazil alone account for almost one hundred million hectares (963,357 km²; (17), an area larger than Tanzania (**Fig. 1b**). Different hypothesized mechanisms may drive deforestation under such undesignated/untitled tenure regimes up or down (**Table 1**, **Table S1**). Here, we aimed to test the predominant prediction that such regimes cause increased agriculture-driven deforestation.

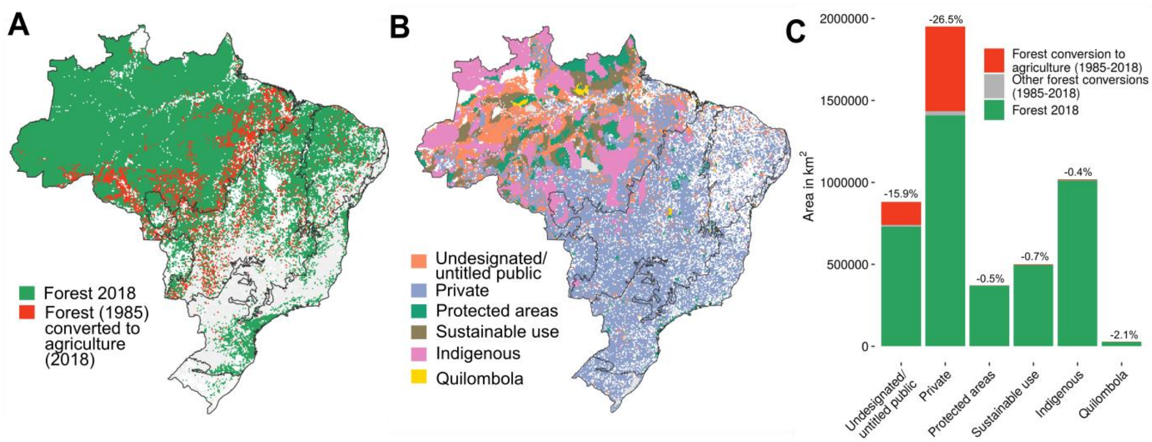

Figure 1. Forest conversion to agriculture (1985-2018) and spatial distribution of different land-tenure regimes in Brazil. A) shows all forest cover (including plantation, savanna, and mangrove tree cover) converted to farming (pasture, agriculture, annual perennial, and semi-perennial crops, including mosaic of agriculture and pasture) (16). B) shows the spatial distribution of six different land-tenure regimes, collated from Imaflora's Atlas of Brazilian Agriculture (15). C) 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.

To this end, we used a robust quasi-experimental study design that compares deforestation in matched land parcels under alternative tenure regimes (***SI Appendix***, full results in **Tables S3/S4**). Our Brazil-wide analyses revealed that, on average, undesignated/untitled regimes increased deforestation between 1985 and 2018 by ~13.3-23.6% relative to all other tenure regimes (**Fig. 2a**, large circles). To assess the generality and potential transferability of these results across different contexts in the tropics, we repeated these quasi-experimental tests for 48 different combinations of narrower spatial and/or temporal extents. These extents correspond to highly distinct socio-environmental contexts, characterized by different bioclimatic regions with distinct agricultural sectors and environmental governance regimes, as well as by different historical time periods since the mid-1980s defined by major macro-economic events, national policies, and deforestation highs or lows (***SI Appendix***). These tests revealed higher deforestation under undesignated/untitled compared to the respective other tenure regime in 140 out of 197 cases (lower deforestation in 5 cases, non-significant in 52, **Fig. S1**, **Tables S3/S7**). These results 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 ***SI Appendix***; **Fig. S3a/S4**, **Tables S3/S6**). 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, substantiating appeals for installing alternative tenure regimes (18).

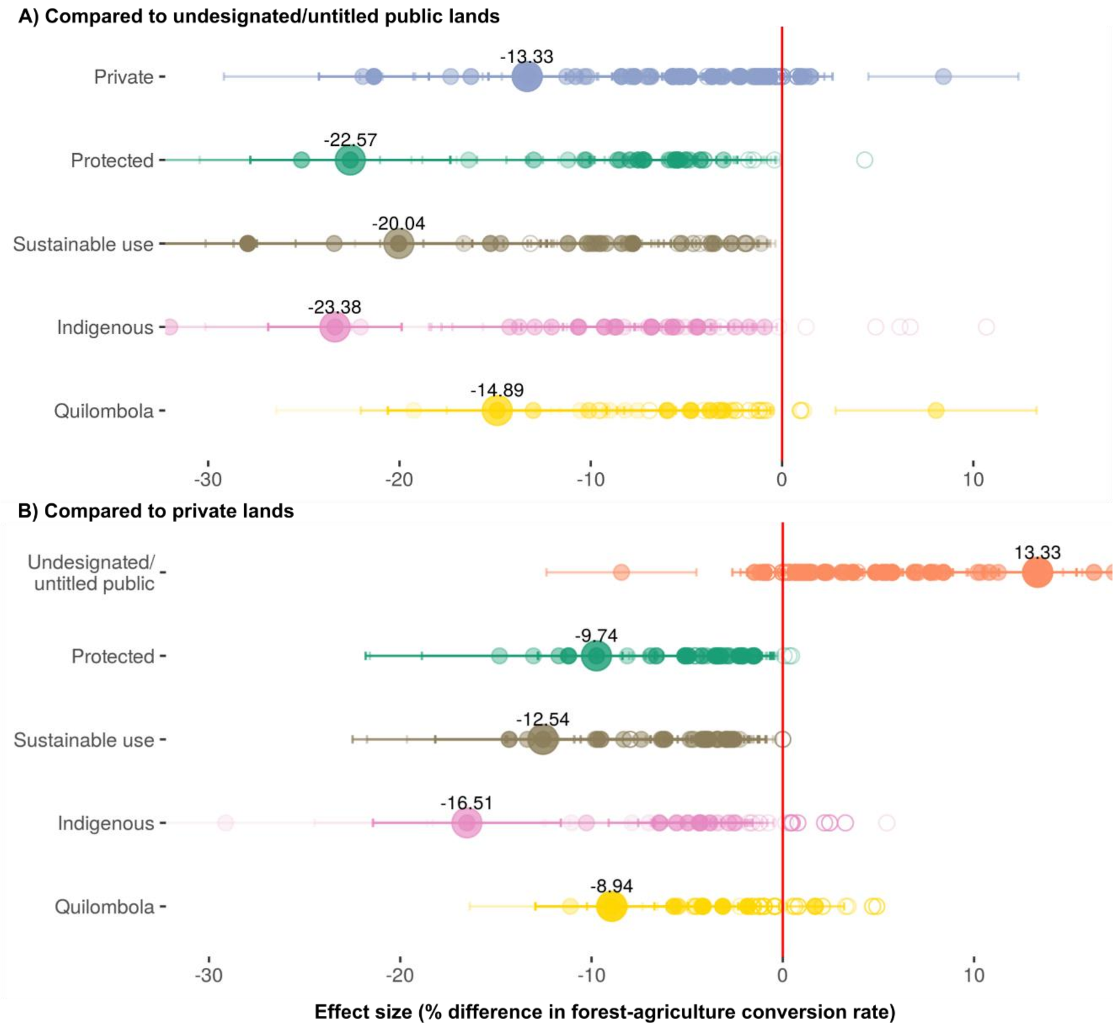

Figure 2. Effects of alternative land-tenure regimes on forest-to-agriculture conversion rates in Brazil. Circles indicate effects sizes estimated at different spatial-temporal scales, compared to two alternative counterfactuals: A) undesignated/untitled public lands with poorly defined tenure rights, 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). See Fig. S1/S2 for detailed presentation of scale-specific results for all tenure regimes and Fig. S3/S4/S5 for results from time-filtered robustness tests.

Over recent decades, global development policies mostly promoted placing undesignated/untitled public lands under private tenure regimes (19) through tenure interventions such as regularization, titling, or registration. Conservation and sustainable-development organizations alike commonly support such interventions (20), hoping that associated improvements in tenure security and clarity will promote more sustainable resource management – although shifts to private regimes may also promote deforestation via other mechanisms (**Table 1**, **Table S1**). The relative importance of these deforestation-promoting and -inhibiting mechanisms is likely context-specific. To guide more general policies, an important first step is thus to quantify their combined net effects and how generalizable these effects are across different contexts.

Similarly to how we analyzed effects of undesignated/untitled tenure, we thus assessed the directionality, magnitude, and generality of net effects of replacing undesignated/untitled tenure with private tenure across the 49 distinct spatiotemporal scales. In our quasi-experimental analysis setup, private tenure would have 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 (**Fig. 2a**; note that these analyses are not confounded by differing initial forest covers; see ***SI Appendix***, **Fig. S6**). Yet, these deforestation-reducing effects were not general across narrower regional-historical contexts. At these 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%; **Fig. S2/S4, Table S7**). This indicates that the environmental benefits of tenure interventions promoting private rights over undesignated/untitled lands more often outweigh the risks than vice versa. However, these interventions do not reliably lead to improved forest outcomes. Indeed, recent titling activities in Brazil's Amazon region have caused deforestation increases in the years immediately following the interventions (12).

Beyond private tenure, different interest groups advocate for various other regimes with different but similarly well-defined tenure rights to replace undesignated/untitled tenure, including indigenous, community-based, full-protection, and sustainable-use regimes (**Table S2**). We assessed which of the alternative regimes could reduce deforestation most effectively and most reliably. To this end, we compared effects of these alternative regimes against an undesignated/untitled counterfactual across 34 different scales (***SI Appendix***). These tests revealed that private tenure underperformed alternative regimes in protecting forests under most regional-historical contexts. Specifically, 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), was least likely to cause high deforestation reductions (2.9%; 11.8% if balance-weighted), and was second-most likely to cause the lowest reductions/highest increases (25.5%; 16.7% if balance-weighted; **Table S5**). 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.

We expected that full-protection and sustainable-use regimes would reduce deforestation most strongly, as the associated bundles of rights are specifically designed for conservation purposes (**Tables S1-2**). Fully protected areas, in particular, remain the mainstay of global conservation strategies, despite concerns about management effectiveness (21) and debate about the extent to which the conserved natural resources should be open to sustainable use (22, 23). Our results support our hypothesis in that full-protection and sustainable-use regimes had, respectively, the second- and third-strongest deforestation-reducing effects at large scales (**Fig. 2/S1**). 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 S5**). The above results were robust both to weighting by balance post-matching and to filtering later-established conservation areas (**Fig. S3 A**, **Tables S5/S6**, see ***SI 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 full-protection regimes (29.4/14.7%; 23.5/20.6%; 10.5/7.9%; **Tables S5/S6**; note these differences are not driven by protected-area siting, see ***SI Appendix***). This indicates that while any conservation-focused regime may reduce deforestation more reliably than alternative regimes under very different contexts, specifically sustainable-use regimes can most reliably achieve *large* reductions.

We also analyzed effects of tenure held by indigenous peoples and local communities (IPLCs). IPLCs have recognized tenure rights over a large and growing portion of the world's forestlands (24), and are increasingly embraced by environmental policies as critical partners for conserving biodiversity and carbon (25). Since IPLC tenure rights might be recognized over any land, we assessed effects against both undesignated/untitled and private-tenure counterfactuals. Our cross-scale analyses yielded no significant effects for nearly half of all cases (64 of 130, **Table S7**). Moreover, they revealed differential effects for different types of IPLCs (which were qualitatively robust to balance-weighting). Against either counterfactual, tenure by specifically indigenous communities reduced deforestation more effectively than all other regimes across Brazil over 1985-2018 (**Fig. 2/S1**). 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; **Fig. S1**/**Table S5**). 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; **Fig. S1/4**; **Table S5**). These ambiguous results on the effects of community-based tenure regimes are in line with diverging theoretical arguments (**Table 1**; **Table S1**). Overall, the evidence provided by our quasi-experimental tests suggests that synergies between IPLC tenure and forest conservation objectives arise often, but not reliably across different contexts.

While we designed our cross-contextual synthesis approach to identify likely generalizable effects across diverse social-environmental settings, we found important divergences from overall effects for Amazonia, where 90.5% of Brazil's remaining undesignated/untitled forest is situated (**Fig. 1**). Here, all three public reserve regimes (full-protection, sustainable-use, and indigenous) had consistently weaker deforestation-reducing effects against undesignated/untitled than quilombola tenure (**Fig S1**). 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 (**Fig. S1**). Both results were robust to balance-weighting and not confounded by systematic differences in initial forest cover (**Fig. S6**). These counter-intuitive Amazonian effects might be explained by the region's specific environmental governance setting. Over recent decades, Amazonian private landholders 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 (26, 27). At the same time, understaffing and logistic difficulties due to Amazonia's remoteness may disproportionately hamper the effectiveness of government policing of the region's public reserves (28). This might indicate that for remote public lands with poorly defined tenure rights and limited public capacity for on-the-ground control, a privatization that is strongly coupled to extensive environmental obligations (29) may be effective in reducing deforestation, by partially transferring responsibility and accountability for forest governance from public institutions to specific individuals. Moreover, this suggests 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.

These findings for Amazonia also raise the broader question of how a more general privatization of *any* publicly-owned lands in the tropics might affect deforestation rates. Globally, over 70 percent of forestlands, including most indigenous and conservation lands (as well as undesignated/untitled lands), are statutorily owned and administered by public institutions (24). Different hypotheses predict that replacing public with private tenure would reduce deforestation, fueling arguments for liberalizing state control over these lands (notwithstanding counter-hypotheses; **Table 1**; **Table S1**). Our systematic tests comparing matched parcels under alternative public regimes against private parcels did not find support for a general public-private dichotomy (**Fig. 2b**). Instead, they showed that replacing any public regime other than undesignated/untitled with private tenure would have increased deforestation in most regional-historical contexts (all country-wide, 81.8% of biome-specific long-term, and 65.3% of biome-specific short-term tests; mean effects ranging from 2.1% deforestation reduction to 40.0% deforestation increase; results qualitatively robust to balance-weighting and time-filtering; **Fig. 2b, Fig. S1/S3/S4**; **Table S4**). In fact, despite our earlier findings that private tenure would more effectively reduce recent deforestation on Amazonian undesignated/untitled lands than alternative public regimes, directly replacing those alternative regimes with private tenure would have most likely increased deforestation in Amazonia (81.0% of tested time-periods, all periods after 2000; **Table S4**). This apparent paradox indicates that privatization may only effectively counter the specific deforestation mechanisms acting on Amazonian undesignated/untitled public lands – but not those on state-protected or indigenous lands. These insights may inform current political debates about potentially privatizing protected areas or indigenous reserves in Amazonia or elsewhere (14, 30).

In summary, against a backdrop of oftentimes ambiguous empirical evidence, theories, and interest groups' claims, our study can shed new light on the net direction, magnitude, and generality of the effects of alternative tenure land-regimes on tropical deforestation. We achieved this through systematic quasi-experimental testing and synthesis across different spatiotemporal scales and contexts. Our results may inform environmental practitioners about likely environmental impacts of different tenure regimes. Moreover, they may offer guidance to policy-makers about which of alternative tenure interventions might reduce long-term deforestation rates most effectively and most reliably under different socio-environmental settings. This can help clarify how different tenure policies might align or misalign with forest-dependent sustainable-development goals such as climate-change mitigation and biodiversity conservation.

Despite the context-specificity of human-environment systems (31), we could derive several conclusions that are generalizable across different regional-historical contexts in Brazil, characterized by highly diverse environmental, socio-political, and economic conditions, and that might thus be more likely than others to also hold for other tropical contexts. In particular, placing undesignated/untitled public lands with poorly defined tenure rights under any other tenure regime will likely substantially reduce deforestation. Reducing deforestation appears most probable when implementing conservation-focused regimes, where sustainable-use regimes are most likely to cause large reductions. Large reductions are least certain when promoting private land rights, although this can be highly effective where there are constraints to on-the-ground government control and private rights are coupled to extensive environmental obligations. Finally, privatizing public lands other than undesignated or untitled, such as protected areas or indigenous reserves, will usually increase deforestation. For those tenure regimes for which our results do not indicate very generalizable effects, such as IPLC-based regimes, guidance to sustainability policies should be based on further research into the context-distinguishing factors. Expanding the systematic cross-scale testing shown here to other tropical regions will be contingent on governments making parcel-level land-tenure information more accessible. Greater transparency is particularly crucial with regard to private and IPLC tenure rights, which cover much of the remaining tropical forest estate but showed the most ambiguous effects.

Materials and Methods

Data

We used the comprehensive, publicly available data on land-tenure categories compiled by Imaflora (v.1812; (15)) for 83.4% of the Brazilian territory (***SI Appendix,* sections 1 and 2.1**). We grouped several Brazil-specific categories to correspond to general tenure categories present in most tropical forest nations, including private tenure with individual ownership ('private'), undesignated and untitled public lands with poorly defined tenure rights ('undesignated/ untitled'), conservation-focused tenure regimes (distinguishing full-protection from sustainable-use regimes), and indigenous or local community-based (IPLC) tenure regimes (distinguishing indigenous and quilombola lands in our main analyses; ***SI Appendix,* section 2.2**). We used the 30-m-resolution annual land-cover/use data provided by Mapbiomas (16) for our calculations of forest-to-agriculture conversion rates (***SI Appendix,* section 2.3**). We used a set of covariates known to influence forest-to-agriculture conversion, including market accessibility (represented by travel time to nearest city; (32)), agricultural suitability (represented by slope and elevation; (33)), parcel area in ha (15), and population density (34) (***SI Appendix*, section 2.4**).

Study design

Our goal was to assess and synthesize the direction, strength, and generality of the longer-term effects that plausible shifts between alternative land-tenure regimes would have on agriculture-related deforestation rates. Rather than quantifying near-term impacts of specific tenure-intervention events such as titling, we thus wanted to capture the differential forest-to-agriculture conversion rates under alternative land-tenure regimes over periods of several years to decades (***SI Appendix*, section 3.1**). Moreover, we wanted to evaluate the extent to which the deforestation effects of these tenure-regime differences might be generalizable across diverse socio-environmental settings within Brazil, and thus, potentially transferable to other tropical forest regions. To this end, we systematically tested effects across 49 different combinations of spatial and temporal extents that correspond to highly diverse regional and historical environmental, socioeconomic, and policy contexts (i.e. across Brazil's entire territory and its biomes Amazônia, Caatinga, Cerrado, Mata Atlântica, Pampa, and Pantanal, and across our entire study period 1985-2018 and sub-periods 1985-1990, 1990-1995, 1996-1999, 2000-2004, 2005-2012, and 2013-2018; ***SI Appendix*, section 3.2**).

For each of these scales and tenure-regime comparisons, we tested effects using a quasi-experimental study design (***SI Appendix*, sections 3.3 and 3.4**). We first applied coarsened-exact matching implemented in the '*cem'* package (35) in R (versions 3.5.1-4.0.2) (36), which involves temporarily 'coarsening' each confounding variable into bins (predetermined strata). We then estimated the causal effect by fitting generalized linear models (GLMs) with a binomial error distribution and a logit link to the respective matched dataset. We used the uncoarsened variables as model covariates and additionally included federal state 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, we clustered our standard errors by municipality (***SI Appendix*, section 3.4**). We tested the sensitivity of our results to potential omitted-variable bias by calculating Rosenbaum bounds (***SI Appendix*, sections 3.4**, **Tables S3-S4, S8**). We extensively tested the robustness of our results to violations of our constant-treatment assumption and to possible biases due to remaining imbalance post-matching, differing initial forest cover of treatment and control parcels, and geographical siting of tenure regimes (***SI Appendix*, sections 3.5 and 3.6**).

We formally synthesized the estimated scale-specific effects via two complementary approaches. First, we assessed the generality of the direction of the causal effect by calculating percentages of scale-specific models with, respectively, significant deforestation-increasing, significant deforestation-decreasing, and no significant effects (***SI Appendix*, Table S5**). Second, we assessed the generality of the relative ranking of alternative tenure regimes 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 (***SI Appendix*, Table S5**).

Code Availability

All code used for the empirical analyses is available on GitHub (https://github.com/pacheco-andrea/tenure-defor-br).

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Supplementary information for:

Land-tenure regimes determine tropical deforestation rates across socio-environmental contexts

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This appendix includes:

Supplementary text (extended materials and methods)
Figures S1 to S6
Tables S1 to S8
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Supplementary Information Text

Materials and Methods

1. Study context: land tenure in Brazil

Modern land-tenure regimes as they exist in Brazil today – with all rights and regulations that apply to them – exemplify the complex historical processes of land distribution common to tropical nations. 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 (1–3).

When the dictatorship ended and a new constitution was written in 1988, protected areas (PAs) 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)) (4, 5). 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 (6, 7)). Altogether, these regulations, policies, and programs have roughly defined the *de jure* and *de facto* tenure regimes in Brazil for the past 50 years (**Table S2**).

2. Data

2.1. Land tenure data

We used the publicly available data on land tenure compiled by Imaflora (v. 1812) (8). This spatially explicit parcel-level dataset maps land-tenure for 83.4% of the Brazilian territory. It 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 (9). These data likely represent the most reliable and comprehensive parcel-level land-tenure information available for any large tropical country. Nevertheless, we acknowledge remaining uncertainties in the depicted spatial patterns, particularly in certain regions where overlapping land claims are reportedly higher than elsewhere (*9*). Our analytical approach across multiple regions (see sections 3.2 and 3.5) partly buffers against possible biases introduced from high data uncertainties in any particular region.

For most tenure categories, the available data lack, or have incomplete information on the date of each parcel's formalization (i.e., titling or demarcation). Despite possible changes in official tenure 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 our study period. However, as we deemed this assumption problematic for certain tenure categories, we took several steps to minimize possible bias in our statistical analyses and conclusions. Firstly, we performed all analyses over multiple spatial and temporal extents and assessed whether results for Brazilian subregions and time periods with known changes in tenure patterns were qualitatively consistent with those for 'tenure-stable' regions/periods. Secondly, we excluded tenure sub-categories defined via programs that only came into existence after our study periods began. Thirdly, we performed robustness tests for selected tenure categories with documented 'treatment' dates, where we filtered out parcels for which today's tenure category was non-existent or unclear at the beginning of the respective study period. Fourthly, we assessed possible biases in our quasi-experimental setup due to remaining statistical imbalance, omitted variables, and systematic differences in initial forest cover between 'treatment' and 'control' units. We outline the specific steps taken in our description of the tenure categories analyzed (see section 2.2) and of our study design and statistical approach (see 3.3-3.6).

2.2. Categorization of land-tenure regimes

Many countries employ unique categories or subdivisions of land-tenure forms, which makes international comparisons difficult. For instance, 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. However, a central aim of this study was to identify land tenure effects that might be generalizable across different contexts (i.e., potentially including non-Brazilian regions). Therefore, we lumped several Brazil-specific tenure categories to more closely correspond to classical types of land-tenure regimes that are also present in other tropical forest nations, while still sufficiently specific to the context of Brazil to also enable country-specific conclusions. The distinguished tenure regimes are characterized by specific 'bundles of rights' (10) and responsibilities that regulate how the tenants can interact with their land resources (see **Table S2**).

Private lands (hereafter 'private'). 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 S2**), although some resource-withdrawal rights are regulated through existing agricultural and environmental policies. We combined private properties from different sources (CAR, SIGEF) under this category. While a small percentage of these private lands may have shifted tenure categories during our study period, most had already been settled and formally recognized as private lands before the mid-1980s (e.g. (11); note that subsequent changes in the specific property owners are not relevant to our study). By contrast, we excluded all private properties titled under the Terra Legal program from our analyses, as this program only started in 2009 and, accordingly, these properties experienced shifts in tenure categories during our study period. Note that deforestation effects of property tilting under the Terra Legal program were recently the focus of different study (12).

Undesignated and untitled public lands with poorly defined tenure rights (hereafter 'undesignated/ untitled'). 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, or, if they are occupied by settlers, has not recognized any tenure rights of them (e.g., via registration or titling). Withdrawal use rights on undesignated/untitled lands are usually not regulated, and *de jure* existing regulations are typically not enforced. 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 S2**). We merged public properties listed in the Imaflora dataset as either 'undesignated lands' or 'rural settlements' into this category, but excluded all rural-settlement parcels that are part of the Terra Legal program. Our reason for this exclusion was that the specific design of this program may have incentivized some settlers to clear forestland in anticipation of the later titling process (12), which could have biased our perception of the normal effects of untitled/undesignated regimes on forests. Untitled/undesignated lands today have had this status throughout the 1985-2018 period.

Conservation-focused tenure regimes. We followed the classification of conservation-focused tenure regimes used by the Ministry of Environment of Brazil, 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 full-protection regimes severely 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 S2**). 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 our study, substantial percentages of the parcels under either conservation-focused tenure regime have only come under the respective regime during the course of our study period. Beyond qualitatively assessing consistency of results between more and less 'tenure-stable' regions and periods, we thus performed additional robust tests for these categories. Specifically, we repeated our statistical analyses 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 (13).

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, we decided to maintain the distinction between these three tenure regimes, due to their very different histories, legal statuses, and granted bundles of rights (**Table S2**). 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. We combined both homologated (formally recognized) and non-homologated indigenous lands 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 S2**). 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 S2**). We decided to restrict our main analyses of IPLC tenure regimes to indigenous and quilombola tenure, both because of the ambiguity of communal lands' bundles of rights and because there were insufficient registered communal land parcels to support our quasi-experimental design in all biomes except Amazonia. However, we additionally provide results for communal tenure in **Tables S3-S4** and **Fig S1-S2**.

Many lands claimed by IPLCs are still unmapped or are mapped but not yet officially registered (1, 2) and were thus excluded from our 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 our period of analysis (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 (14) 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. Recognition of their specific bundle of rights began in the mid-1980s, coinciding with the first period of our 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.

We omitted military lands, urban and transport-related lands, and water from our analyses, as these are less relevant to the hypothesized mechanisms relating land-tenure regimes to forest-to-agriculture conversion. We thus focused the main analyses on six categories of land-tenure regimes: undesignated/untitled public, private, fully protected, sustainable use, indigenous, and quilombola.

2.3. Land-use change data

We used the 30-m-resolution annual land-cover/use dataset provided by Mapbiomas (15) for our calculations of forest-to-agriculture conversions over different time periods between 1985 and 2018. We defined forest-to-agriculture conversions as any case where either natural or plantation forest cover, 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.

2.4. Covariate data

We used a set of covariates known to influence forest-to-agriculture conversion, including market accessibility (represented by travel time to nearest city; (16)), agricultural suitability (represented by slope and elevation; (17)), parcel area in ha (8), and population density (18). See sections 3.3 and 3.6 for details on covariate use.

3. Study design and analysis

3.1. Overview

Our goal was to assess and synthesize the direction, strength, and generality of the longer-term effects that shifts between alternative land-tenure regimes typically have on forest-to-agriculture conversion rates. Rather than near-term impacts of specific tenure-intervention events (e.g., titling), we thus wanted to capture the differential impacts of alternative regimes over periods of several years to decades, and assess how general effects of these regime differences are across different regional-historical contexts.

To estimate causal effects from observational data, we used a quasi-experimental approach that combines matching with regression analysis (19). To test the extent to which the effects are generalizable across regions and periods within Brazil with diverse socio-environmental settings, and thus potentially transferable to other tropical forest regions, we systematically repeated all analyses over 49 different combinations of spatial and temporal extents. We formally synthesized these scale-specific effects via two complementary approaches, designed to assess *i)* the generality of the net direction of the effects, and *ii)* the generality of their relative strength in comparison to the effects of other tenure regimes.

3.1. Plausible changes in land-tenure regimes

We here define ‘land-tenure regime’ as the combination of tenure-related governance factors that exist over a given parcel of land and are stable over a certain period of time. This includes the bundle of rights associated with the respective tenure category (**Table**

S2), but also the implications that these rights may have for tenure security, as well as the tenure categories' predispositions for being subject to particular types of policies or regulations. Correspondingly, we define 'tenure-regime change' as a stable shift from one such regime to another. Tenure-regime changes are thus not instantaneous events, but gradual processes that may involve different legal and administrative acts (e.g., titling, registration, or other steps) and will only be completed after the resulting changes in rights, regulations, and perceptions have come into effect.

We focused on major types of tenure-regime changes corresponding to tenure-intervention processes that are commonly observed across the tropics and are related to different sustainability questions. Firstly, we focused on shifts from undesignated/untitled public regimes with poorly defined tenure rights to private tenure regimes, which over the past decades have been the most common outcomes of tenure interventions (e.g., through formal titling, registration, or other tenure-regularization processes). Secondly, we considered shifts of such undesignated/untitled to conservation-focused tenure regimes, corresponding to designation of public lands as fully protected or sustainable-use areas. Thirdly, we considered shifts from either undesignated/untitled or from private regimes to community-based tenure regimes, corresponding to processes of recognizing tenure rights claimed by IPLCs (which might involve anything from simple registration to multi-year court battles). Finally, we considered changes from different public regimes to private tenure regimes, corresponding to privatization of state-owned lands (which may affect undesignated/untitled, but also conservation, indigenous, or other public lands (20, 21)).

We note that the specific analysis methods we used (see 3.3) do not *per se* restrict the direction in which the estimated effects may be interpreted. As such, an estimated deforestation-increasing effect of replacing a public with a private tenure regime (e.g., through privatization of formerly protected areas) might equally be interpreted as a deforestation-decreasing effect of the same magnitude of a regime change in the opposite direction (e.g., via government seizure and subsequent protection of private lands). Similarly, we could not claim, in these particular tests, that any specific characteristics of either the private nor the public tenure regime would cause the observed difference in deforestation. Instead, we interpret the observed deforestation differences more neutrally as being due to the combination of relevant differences between the two tenure regimes.

3.2. Analyses at different spatial and temporal scales

Insights on the environmental implications of land-tenure policies from Brazil are commonly transferred to inform policy strategies in other tropical regions, reflecting the data limitations in most other countries, Brazil's extensive experience in linking tenure reform with environmental policies, and Brazil's own active role in South-South development cooperation (11, 22, 23). Notwithstanding this practice, causal effects in complex human-environment systems are often highly context-specific (24), 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, we defined 49 different combinations of spatial and temporal extents of analysis, corresponding to distinct socio-environmental contexts characterized by different bioclimatic regions with distinct agricultural sectors and environmental governance regimes, as well as by different historical time periods that saw different policies, macro-economic events, and trends in deforestation. We repeated the full causal-analysis procedures for each tenure-regime comparison for each of these spatiotemporal scales (see below).

We defined a 'large' spatiotemporal extent covering the entire spatial extent of Brazil and capturing the net agriculture-to-forest conversion over the full 1985-2018 period. In addition, we ran all analyses over the same temporal extent 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, we ran all analyses over both large and narrower spatial extents over six narrower temporal extents, which we 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; (25), the proposal of REDD+; (26)). 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 (6).

3.3. Creating quasi-experiments on shifts in land-tenure regimes

To be able to draw causal inference from observational data, we constructed 'quasi-experiments'. As our quasi-experimental method, we used matching with subsequent regression analysis (19, 27). Matching addresses the bias that would arise if simpler regression designs were applied to our unbalanced tenure dataset due to non-random assignment of forested parcels into 'experimental treatment' by different tenure categories (e.g., most parcels being under private tenure). The specific matching method that we used, coarsened exact matching (CEM; (19)), 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. We conducted one-to-one matching, meaning that each pair of parcels contains one parcel coded as 'treatment' under one of two compared alternative tenure regimes, and another (the 'control' or 'counterfactual') under the respective other regime. The size of the effect is subsequently estimated through regression on the balanced uncoarsened data subset.

We note that other quasi-experimental designs such as difference-in-difference (or before-after-control-impact) are more suitable than matching in certain situations, and are commonly used for estimating near-term effects of specific tenure interventions such as titling (12). However, such designs are difficult to apply to processes such as tenure-regime shifts that may only manifest gradually over time through combinations of different events. Moreover, they generally cannot be used where longitudinal datasets of sufficient spatiotemporal scope are not available for all experimental treatment types (as is the case for most land-tenure types across the tropics). Therefore, we believe that cross-sectional comparisons using matched data are the most sensible approach for addressing our question that is currently feasible. However, we caution that our data do not capture any actual long-term tenure-regime shifts, but merely differences in tenure-regimes among otherwise highly similar parcels. Thus, our estimated effects should be interpreted accordingly, i.e., as the hypothetical effects of fully completing a tenure-regime shift under the assumption that everything else be kept constant.

Estimating causal effects via matching requires the assumption that there is no ‘unobservable-variable’ bias due to omitting important confounders. We controlled for five commonly used confounders that are known to influence forest-to-agriculture conversion (see 2.4). We additionally minimized risks of unobservable-variable bias by *i)* including fixed effects for federal states to capture subnational governance differences, *ii)* clustering our standard errors by municipality, and *iii)* assessing sensitivity of our results against potential omitted-variable bias using Rosenbaum bounds (see 3.4 for further details). Moreover, we specifically assessed possible bias due to systematic differences in initial forest cover (see 3.6). We note that causal analyses of instantaneous/short-term events would typically control only for pre-treatment covariates, to avoid the risk that covariates on the causal ‘pathway from exposure to outcome’ might block part of the investigated effect (28). However, as we analyzed longer-term effects of alternative stable tenure regimes, our treatments acted continuously throughout the respective study period. Corresponding to such continuous treatment, we averaged the time-variant population-density variable over the years of the respective period (including linearly interpolated/extrapolated values as necessary).

We applied the coarsened-exact matching algorithm implemented in the ‘*cem’* package (29) in R versions 3.5.1-4.0.2 (30). CEM involves temporarily ‘coarsening’ each confounding variable into bins (predetermined strata). We used automated coarsening for elevation, slope, and human-population change, but manually defined bins for travel time to nearest city and for parcel area. We divided travel time to nearest city into bins of 0-2, >2-6, >6-12, >12-24, and >24 hours, and parcel area 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 of our defined spatiotemporal extents, we assured exact matching considering the total spatial and temporal variation in the covariates at the respective scale.

While CEM, in particular, has a range of advantages over other matching approaches (19), identifying exact matches is generally difficult when there is little overlap in parcel-level similarity among covariates. However, the large number of parcels (~4 million) in the Imaflora dataset allowed us to retain 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; (31); see **Tables S3-S4**). Due to very small numbers of matched parcels (4 to 28), we did not estimate effects for communal tenure regimes in the Caatinga, Cerrado, and Mata Atlântica, nor for any regime other than undesignated/untitled and private in the Pampas and Pantanal biomes.

We use the L_1 measure developed by King et al. (29) to calculate remaining imbalance post-matching. Across all datasets that we used for our scale- and tenue-regime-specific tests, CEM improved balance by 5-79% (0-73% for time-filtered tests) (**Tables S3-S4**). Imbalance post-matching ranged from 0.10-0.76, meaning that our datasets achieved between 24% and 90% balance in covariate values. To make cases of high remaining imbalance post-matching easily recognizable, we visualize imbalance as transparency gradients in all plots of estimated effects (**Fig. 2**, **Fig. S1-S4**). Moreover, we explicitly incorporate imbalance into our cross-scale synthesis of results (see 3.5).

3.4. Regression analyses

For each scale and tenure-regime comparison, we estimated the causal effect by fitting generalized linear models (GLMs) with a binomial error distribution and a logit link to the respective matched dataset. We used the uncoarsened variables as model covariates and additionally included federal state 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, we cluster our standard errors by municipality.

$$logit(p) = \beta_0 + \beta_1 tf_1 + \beta_2 l_2 + \beta_3 s_3 + \beta_4 tt_4 + \beta_5 pd_5 + \beta_6 r_6 + \beta_7 st_7$$

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.

We calculated average marginal effects (AME) using the '*margins'* package in R (32), transforming coefficient estimates to average per-forest-pixel probability of conversion to agriculture with respect to the tenure form in question (33) (**Tables S3-S4**).

Lastly, we calculated Rosenbaum bounds as a sensitivity analysis to assess whether our model estimates are robust to the possible presence of omitted-variable bias. Rosenbaum bounds quantify the sensitivity of our regressions results to different magnitudes of hypothetical bias that might be caused by missing important confounders in the matching procedure (34). 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. We calculated lower and upper bounds for both Hodges-Lehmann point estimates and *p*-values (see supplementary files) using the '*rbounds'* package in R. Our 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 S8**; 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 S8**). We did not find any systematic patterns in sensitivity to possible omitted-variable bias across tenure-regime comparisons, regions, or time periods, except that results based on lower sample sizes (mainly comparisons involving quilombola tenure and those in the Caatinga biome) were on average slightly more sensitive. Our 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. We note that this sensitivity test cannot indicate whether or not an unobserved-confounder bias is actually present.

3.5. Cross-scale synthesis of effects

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), we synthesized the scale-specific effects (AMEs) in two ways. First, for each comparison (e.g., private vs. undesignated/untitled), we assessed the generality of the direction of the causal effect by calculating percentages of scale-specific models with, respectively, significant deforestation-increasing (positive), significant deforestation-decreasing (negative), and no significant effects (**Table S5**). 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, we assessed the generality of the relative ranking of alternative tenure regimes 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 S5**). These analyses address the applied question of which of alternative tenure-regime changes might most/least reliably cause *large* reductions in deforestation. We had initially considered using formal meta-analyses as a third way of synthesizing the scale-specific effects, which would have indicated the direction and magnitude of 'average' effects. However, testing indicated high heterogeneity which, in combination with our small sample sizes (i.e., numbers of scale-specific models) precluded us from deriving reliable estimates using meta-analyses (35).

We assessed the robustness of the results of our cross-scale synthesis against possible bias in the relative reliability of the tenure-comparison- and scale-specific causal tests. To this end, we additionally calculated balance-weighted percentages that effectively downweight any cases where covariate overlap post-matching remained low, and based all our main conclusions on qualitatively consistent balance-weighted/unweighted results. Specifically, we calculated balance-weighted percentages of cases with significant-negative, significant-positive, and nonsignificant effects 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 (**Tables S5-S7**). Similarly, we calculated weighted percentages of scales at which each tenure category had higher/lower-ranked effects than all others 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 S5**). In addition to this balance-weighting, we also assessed the robustness against violations of the assumption of constant treatment of parcels with full-protection and sustainable-use regimes (see section 2.2), by using results based on time-filtered datasets to calculate alternative versions of percentages with significant-negative, significant-positive, and nonsignificant effects (**Table S6**; see **Fig. S3-S4** and **Tables S3/S4** for the full time-filtered results; see section 2.2 for explanation of time-filtering).

We also assessed whether differences in how often tenure regimes were ranked as most/least effective in reducing deforestation might 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, we expected the indirect comparison of full-protection vs. sustainable-use regimes (vis-a-vis an undesignated/untitled counterfactual) 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 (36). We thus assessed whether their differing percentages of most/least effective cases reflected systematic differences in their matched parcels' average covariate values at the specific scales where they were most/least effective. While we did find some cases where the two tenure regimes differed with respect to specific covariates, these cases did not indicate any systematic bias. For example, full-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. (37).

3.6 Assessment of potential bias due to differences in initial forest cover

We note that the estimated effects of tenure-regime differences could have been affected by differences in initial forest cover between our 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 our study periods 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 our conclusions due to systematic differences in initial forest cover, we 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, we 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, we included all covariates from our main regression analyses to compare the same parcels that were also originally matched (see 3.3). We detected no systematic unidirectional differences between treatment and control across scales, indicating that our main conclusions are not biased by such differences (see **Fig S6**). However, we found differences in either direction in individual cases and thus cannot rule out that these might partly explain differential forest trajectories for some tenure regimes and spatiotemporal scales. We address this caveat by basing our main conclusions on results that showed consistency 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).

We chose this indirect approach over directly matching parcels on initial forest cover. This was motivated, firstly, by our aim to evaluate all tenure regimes via a consistent modelling protocol. Here, retaining sufficient degrees of freedom for each tenure regime and spatiotemporal scale required us to constrain the total number of matching covariates, as that number affects both the matched dataset sizes and the number of modelling covariates included in the binomial GLMs. Secondly, our specific aim was not to assess total forest losses of different tenure regimes over their entire lifetimes (which would necessitate accounting for any prior deforestation already internalized in parcels' initial forest cover), but 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.

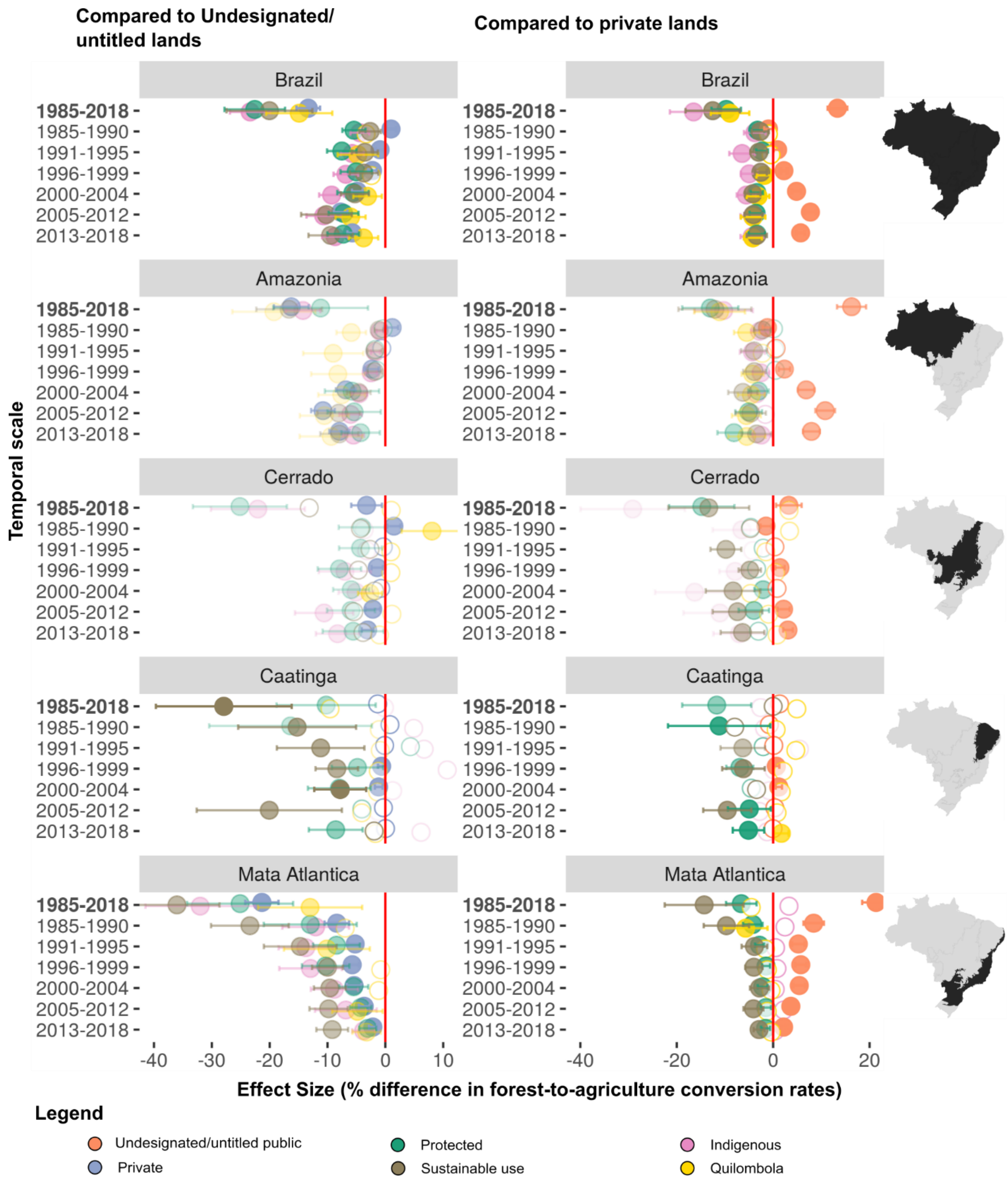

Fig. S1. Effects of alternative land-tenure regimes on forest-to-agriculture conversion rates in Brazil, disaggregated to different spatiotemporal scales. Circles indicate effects sizes estimated at the respective scale vis-a-vis two alternative counterfactuals: a) undesignated/untitled public lands, 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).

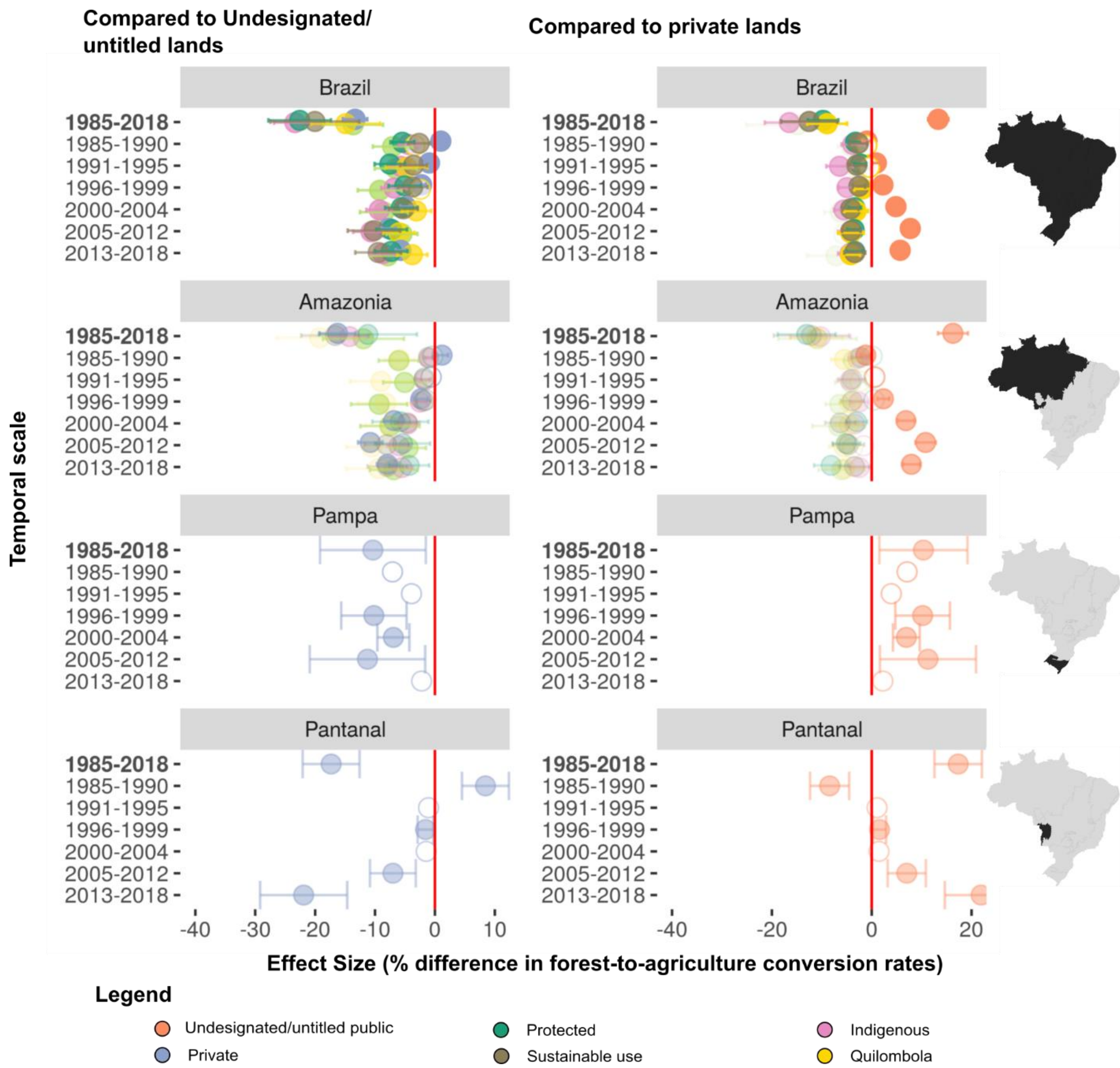

Fig. S2. Effects of alternative tenure regimes on forest-to-agriculture conversion rates at different spatiotemporal scales, complementing Fig. S1 by showing additional results for communal tenure regimes for Brazil and the Amazonia biome, and for private and undesignated/untitled regimes for Pampa and 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.

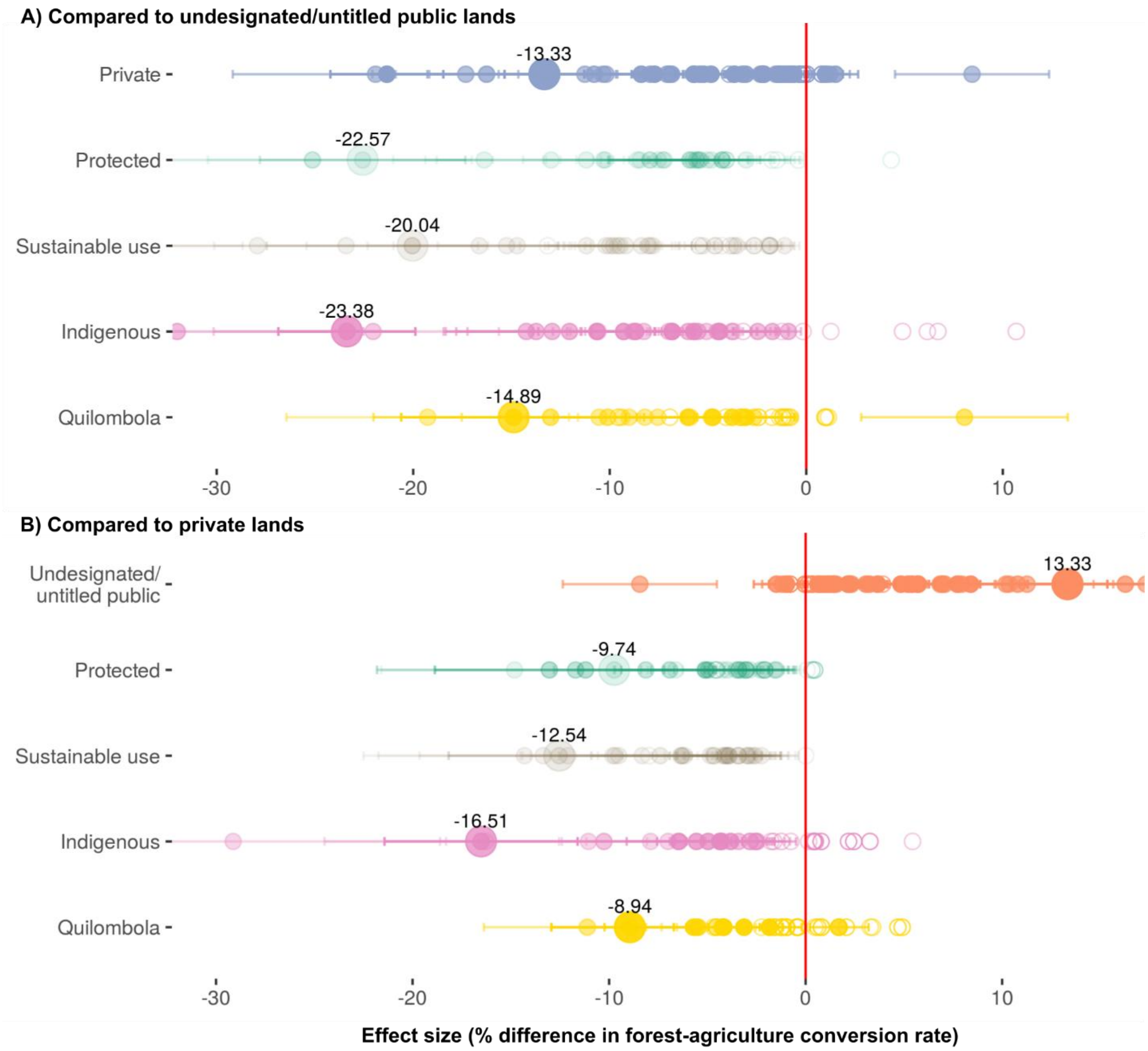

Fig. S3. Robustness test of effects of alternative tenure regimes on forest-to-agriculture conversion rates in Brazil using filtered time-series data for protected and sustainable-use areas (i.e., only areas established before/during beginning of each temporal scale considered; see section 2.2). Circles indicate effects sizes estimated at different spatial-temporal scales vis-a-vis two alternative counterfactuals: A) undesignated/untitled public lands, and B) private lands. (see Fig. S4 for detailed presentation). 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).

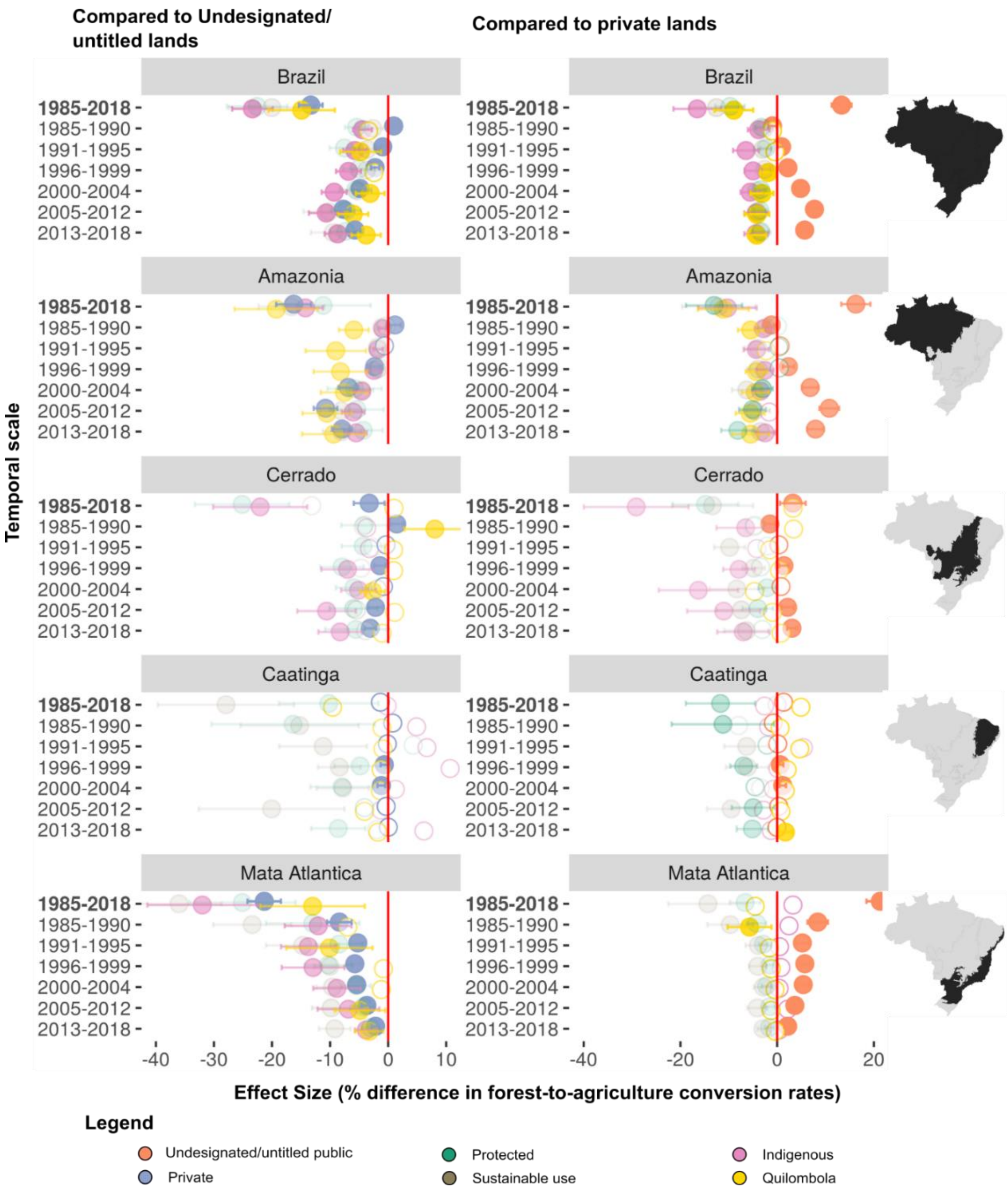

Fig. S4. Spatiotemporal disaggregation of robustness test of effects of alternative tenure regimes on forest-to-agriculture conversion rates in Brazil using filtered time-series data for protected and sustainable-use areas (i.e., only areas established before/during beginning of each temporal scale considered; 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).

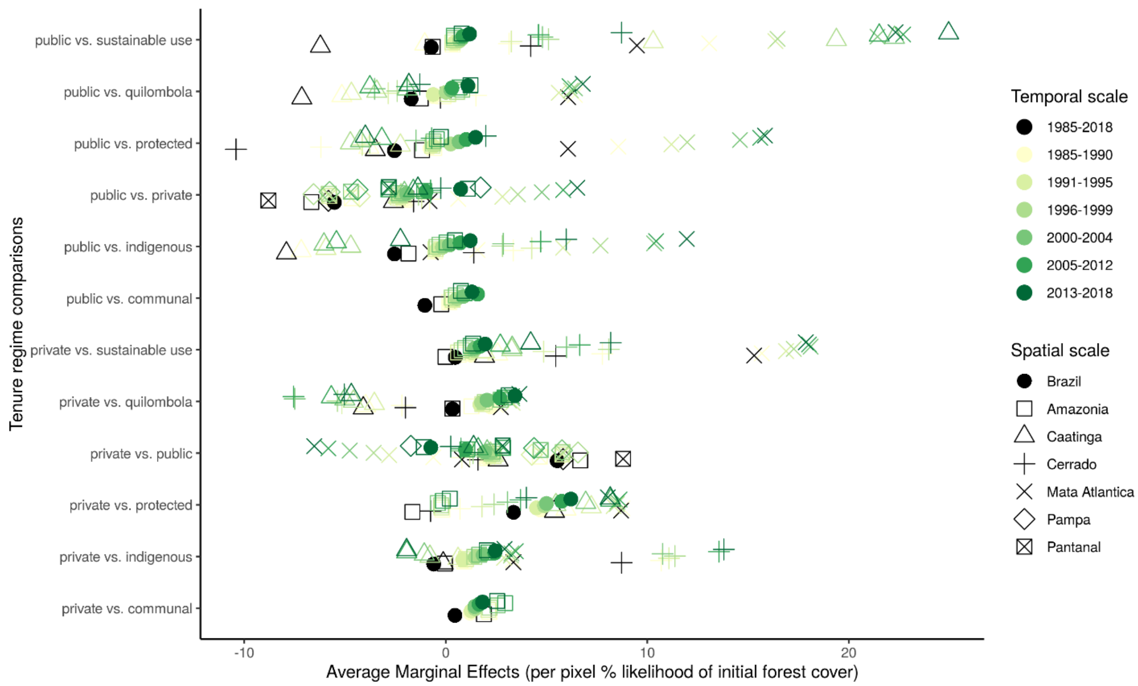

Fig. S4. Differences in initial forest cover between matched treatment and control units for different tenure-regime comparisons at different spatial and temporal scales. Average marginal effects indicate the per-pixel likelihood being forest-covered at the beginning of each time period considered. At 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.

Table S1. 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 (see **Table S2**). 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	Undesignate d/	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 (38–40). 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 (41, 42).	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 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 (43, 44).	de Soto 2000; Place and 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 (45).	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	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	Undesignate d/	Protected area	Sustainable use	Indigenous	Quilombola	Communal
	smallholders or land-less settlers searching for land. These will thus instead be forced to settle on undesignated public lands at the 'frontier' (46).													
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 (47).	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 (48).	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)(49). Where this is the case, tenure regimes with highly 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.	Perrings, 1989	↘	↘	↘	↗	↗	↗	↗	↘	→	↗	↗	↗
Withdrawal (commercial and non-commercial)	Tenure regimes where resource withdrawal is either not restricted or incentivized will see higher deforestation rates (50–52). 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 on putting the land to productive (i.e., agricultural) use.	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/	Protected area	Sustainable use	Indigenous	Quilombola	Communal
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 (53–57).	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 (58–60). 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 outcomes expected for other community-based tenure regimes.	Birdyshaw and Ellis, 2007; Deacon_et al., 1994; Deininger et al., 2003	↗	↘	↘	↘	↘	↗ ↘	↘	↘	↘	↘	↘	↘
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 (61–63). 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 (50, 41, 51, 64). Among the latter regimes, the ability to 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).	Angelsen, 1999; Grafton, 2000; Fearnside, 2005, Nolte et al. 2013	↘	→	→	→	↘	→	↗	↗	↗	↗	↘	↗

Cross-cutting themes

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/	Protected area	Sustainable use	Indigenous	Quilombola	Communal
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 (65, 66). 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 decision-makers	Tenure regimes where ownership is shared among larger numbers of people are better equipped to monitor and protect their land, decreasing the likelihood of deforestation as compared to properties with fewer people (65, 67). 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.	Sakurai et al., 2004; Ostrom 2009	↗	→	→	↘	↘	↘	↘	↘	↘	↘	↘	↘
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)(38, 40, 68).	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 (69). 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 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 (50, 51, 70, 71). While private land tenure is classically viewed as providing the highest tenure security and thus assurance levels, this view is not universal (72). Assuming that classical views on tenure-form–tenure-security relationships broadly hold and that landholders are mainly	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/	Protected area	Sustainable use	Indigenous	Quilombola	Communal
	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 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 (41).	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 (73, 74).	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 these are under partial private or community-based management (75).	Deacon, 1994	↘	→	→	→	↘	→	↗	↗	↗	↗	→	
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 (63). Thus, under precarious governance contexts, all publicly owned forestland is expected to be more likely to experience deforestation.	Baland and Platteau, 2000	↘	→	→	→	↘	→	↗	↗	↗	↗	→	

Table S2. Tenure regimes in Brazil and associated bundles of rights. We re-categorized 14 land-tenure categories distinguished in Brazil (first column) into seven tenure regimes (second column). For each regime, we defined the typical number of tenants involved in land decision-making (third 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 past and current legislation in Brazil, 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).

			Bundles of rights (usually included)									
Brazil tenure categories	**Tenure regime**	**Tenants**	Access	Withdrawal (subsistence)	Withdrawal (commercial)	Management	Exclusion	Alienation	Due Process	**Main right holder**	**Main duty holder**	**References**
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, (76)
Communitary lands	Communal lands	Many	++	+	+/-	+/-	+/-	+/-	-	Community	GO	Decreto N. 6.040, 2007, Lei N. 11.284, 2006, (77, 78).
Quilombola lands	Quilombola lands	Usually many	++	+	+	+	+	--	+	Community	Community	Consitucao Federal art. 68, Decreto N. 6.040, 2007, (79, 80).
Homologated Indigenous land Non-homologated indigenous land	Indigenous lands	Usually many	++	+	--	+	+	--	+	Community	GO	Consitucao Federal art. 231. 1996, Decreto N. 6.040, 2007, (81).
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
Rural settlements Undesignated public forests	Undesignated/untitled public lands	1 or few	++	++	++	+	-	-	--	Citizenry	GO	MP 759/2016, Lei Nº 8.629, de 25 de fevereiro de 1993, (9)

Undesignated lands from Terra legal program												
Military areas, Water, and Urban	(omitted from analysis)											

++ 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

Table S3. Model outputs for all tenure regimes 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 (L_1) 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
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

Treatment	spatialScale	temporalScale	*n*	SE	*p*_value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	ImbImprov
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
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

Treatment	**spatialScale**	**temporalScale**	***n***	**SE**	***p*_value**	**lower_ci**	**upper_ci**	**Effect**	**ImbBefore**	**ImbAfter**	**ImbImprov**
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
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

Treatment	**spatialScale**	**temporalScale**	***n***	**SE**	***p*_value**	**lower_ci**	**upper_ci**	**Effect**	**ImbBefore**	**ImbAfter**	**ImbImprov**
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
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

Treatment	**spatialScale**	**temporalScale**	***n***	**SE**	***p*_value**	**lower_ci**	**upper_ci**	**Effect**	**ImbBefore**	**ImbAfter**	**ImbImprov**
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
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

Treatment	spatialScale	temporalScale	*n*	SE	*p*_value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	ImbImprov
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
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

Treatment	spatialScale	temporalScale	*n*	SE	*p*_value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	ImbImprov
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
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 S4. Model outputs for all tenure regimes 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 (L_1) 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
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
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
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
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
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
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 S5. Synthesis of the directions and relative magnitudes of effects of different land-tenure regimes across spatiotemporal scales. For this cross-scale synthesis, we considered all scales at which deforestation effects of all five alternative tenure regimes were consistently testable vis-à-vis the respective counterfactual (top part: undesignated/untitled; bottom part: private). The left section of the table ('Direction of estimated effects on deforestation') reports, for each tenure regime, the numbers and percentages of scale-specific model estimates predicting an increase or decrease in the likelihood of deforestation of all alternative tenure regimes vis-à-vis the counterfactual. The right section of the table ('Ranking by relative magnitude of effect size') report 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, we placed effects that were statistically indistinguishable from 0 in between deforestation-decreasing and -increasing. For example, 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. We additionally report all percentages as weighted by the level of balance (L_1) in the underlying dataset, which downweights 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 comparisons against communal lands. Also note that these percentages synthesize 'narrower scales' only. For Brazil-wide results for the full 1985-2018 period, See Fig 2 and Fig. S3.

	Direction of estimated effects on deforestation													**Ranking by relative magnitude of effect size**						
	increases (count)	increases (count) weighted by balance	% increases	% increases weighted by balance	decreases (count)	decreases (count) weighted by balance	% decreases	% decreases weighted by balance	non-significant (count)	non-significant weighted by balance	% non-significant	% non-significant weighted by balance	Total models	best	best weighted by balance	worst	worst weighted by balance	non-significant	non-significant weighted by balance	Total models
Compared to undesignated/untitled lands																				
Private lands	3	2.27	8.82%	8.38%	23	18.34	67.65%	67.81%	8	6.44	23.53%	23.80%	34	2.94%	2.22%	25.49%	27.49%	8	2.66	34
Protected areas	0	0.00	0.00%	0.00%	29	14.12	85.29%	87.11%	5	2.09	14.71%	12.89%	34	29.41%	28.52%	14.71%	11.11%	5	1.59	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	32.35%	33.38%	8.82%	7.35%	8	2.71	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	17.65%	22.54%	10.78%	8.62%	9	2.99	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	17.65%	13.34%	40.20%	45.43%	17	6.82	34
All of the above compared to undesignated/untitled	4	2.75	2.35%	2.93%	119	67.51	70.00%	71.73%	47	23.85	27.65%	25.34%	170							
Compared to private lands																				
Public lands	22	17.47	70.97 %	71.46%	3	2.27	9.68%	9.28%	6	4.71	19.35%	19.27%	31	0.00%	0.00%	77.15%	81.49%	6	1.94	31
Protected areas	0	0.00	0.00%	0.00%	22	13.71	70.97%	77.15%	9	4.06	29.03%	22.85%	31	13.71%	10.90%	6.99%	5.21%	9	2.80	31
Sustainable use areas	0	0.00	0.00%	0.00%	28	16.69	90.32%	89.39%	3	1.98	9.68%	10.61%	31	36.29%	43.82%	1.88%	1.21%	3	0.94	31
Indigenous lands	0	0.00	0.00%	0.00%	17	7.21	54.84%	48.95%	14	7.52	45.16%	51.05%	31	39.52%	36.65%	3.49%	2.53%	14	7.16	31
Quilombola lands	0	0.00	0.00%	0.00%	10	5.54	32.26%	29.98%	21	12.94	67.74%	70.02%	31	10.48%	8.63%	10.48%	9.56%	21	9.26	31
All of the above compared to private	22	17.47	14.19 %	18.56%	80	45.41	51.61%	48.27%	53	31.20	34.19%	33.17%	155							

Table S6. Synthesized direction of cross-scale effects of full-protection and sustainable-use regimes, with percentages based on alternative results that were time-filtered for greater robustness of temporal stability assumptions (see sections 2.2. and 3.5; see Table S5 for detailed description). These time-filtered datasets exclude any parcels for which these respective conservation-focused tenure regime was 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 full-protection and sustainable-use regimes (in black) are based on different models. Those for other tenure regimes are as in Table S5, but restricted to the scales where all regimes could be consistently compared. We 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 S3/S4). Therefore, we do not consider the ranking results (‘Ranking by relative magnitude of effect size’) based on the time-filtered data reliable, and ignored them in our conclusions. They are shown here (in grey) for transparency only.

	Direction of estimated effects on deforestation													**Ranking by relative magnitude of effect size**						
	increases (count)	increases (count) weighted by balance	% increases	% increases weighted by balance	decreases (count)	decreases (count) weighted by balance	% decreases	% decreases weighted by balance	non-significant (count)	non-significant weighted by balance	% non-significant	% non-significant weighted by balance	Total models	best	best weighted by balance	worst	worst weighted by balance	non-significant	non-significant weighted by balance	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 S7. Synthesized direction of effects of all assessed land-tenure regimes on deforestation across all assessed scales (see Tables S5/S6 for general description). Unlike results in Tables S5/S6, 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 Tables S5/S6 if single tenure regimes are viewed in isolation. However, unlike results in Tables S5/S6, they are not comparable across tenure regimes as they are based on inconsistent combinations of scales. Information that is redundant with that in Table S5 (as based on the same scales) is shown in grey.

	Direction of estimated effects on deforestation												
	increases (count)	increases (count) weighted by balance	% increases	% increases weighted by balance	decreases (count)	decreases (count) weighted by	% decreases	% decreases weighted by balance	non-significant (count)	non-significant weighted by balance	% non-significant	% non-significant weighted 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 (using filtered results instead)	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
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 (using filtered results instead)	32	23.41	18.71%	30.08%	77	22.04	45.03%	28.32%	62	32.37	36.26%	41.59%	171

Table S8. Summary of sensitivity analysis using Rosenbaum bounds. We calculate upper and lower bounds for both Hodges Lehmann point estimates and *p*-values (see supplementary file #2 for full results) for different Γ levels. For each tenure-regime comparison, spatial scale, and temporal scale considered, we summarize *i*) 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 *ii*) 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**				
Tenure-regime comparisons	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
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%
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%*

Dataset S1 (separate file). Full regression results reporting average marginal effects (AME) for each spatial-temporal scale considered.

Dataset S2 (separate file). Rosenbaum bounds reporting upper and lower bounds for both Hodges-Lehmann estimates and p-values at each spatial-temporal scale considered.

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