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

Andrea Pacheco[1]*, Carsten Meyer[1, 2, 3]*

[1]German Centre for Integrative Biodiversity Research (iDiv) Halle-Jena-Leipzig, Puschstrasse 4, 04103 Leipzig, Germany.

[2]Institute of Geosciences and Geography, Martin Luther University Halle-Wittenberg, Halle (Saale), Germany.

[3]Institute of Biology, Leipzig University, Leipzig, Germany

*Correspondence to: andrea.pacheco@idiv.de, carsten.meyer@idiv.de

Abstract

Many tropical forestlands are experiencing changes in land-tenure regimes, but how these changes may affect deforestation rates remains 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 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 in remote regions where on-the-ground governance is limited and there are extensive environmental policies. Directly privatizing conservation regimes or indigenous lands, in turn, would most likely increase deforestation. Our cross-scale synthesis informs how conservation, titling, and other tenure-intervention policies may align with climate-change, biodiversity, and broader environmental sustainability goals and are directly relevant to ongoing political debates regarding land privatization/protection in Amazonia.

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).

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. The shifts in land-tenure regimes resulting from land-rights 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 transferable 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 (11–13). 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 (14, 15). We capitalize on Brazil's uniquely comprehensive data on both land tenure (16) and land-use changes (17), 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 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 (18, 19).
	Deforestation-promoting	Undesignated/untitled lands lack both clear supervision by any designated agency (20) 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 (21–23). Governments rarely place restrictions on deforesting undesignated/untitled public lands – or even incentivize it by granting claims based on prior clearance (24), or by allowing settlement conditionally on putting the land to productive use (25). 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' (26).
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 (21), thus providing assurance that they will be the sole beneficiaries of their investments.
	Deforestation-promoting	Private titles enable improved enforcement of environmental policies as they facilitate holding specific individuals accountable for complying with environmental obligations (27), such as the obligation to retain certain amounts of forest under Brazil's Forest Code. The lower default risk combined with comprehensive withdrawal and alienation rights of private tenure regimes sparks investments in forest-displacing activities (28). For example, private landholders can more easily access credit to expand their agricultural fields by using land as collateral (19). 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 (29).
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 (30).
	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 (31).
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 (20), or liberal granting of use concessions for short-term state revenues (30). 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 (32). 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 (20) and internalizing long-term costs of degradation into decisions (33).
	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 (20).

Results and Discussion

Poorly defined tenure drives deforestation across spatiotemporal scales

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; (34), 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.

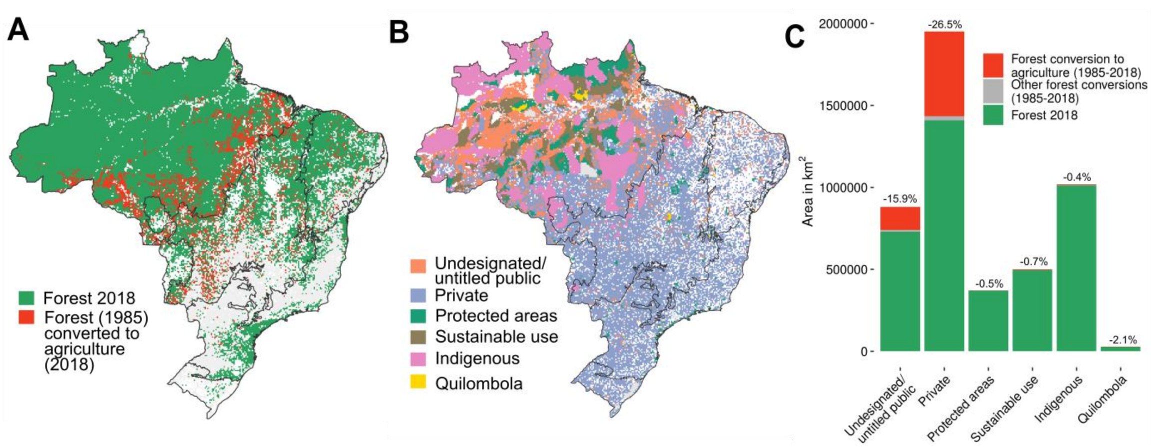

Figure 1. Forest conversion to agriculture (1985-2018) and spatial distribution of different land-tenure regimes in Brazil. A) shows all forest cover (including natural forests, plantations, savannas, and mangrove tree cover) converted to farming (pasture, agriculture, annual perennial, and semi-perennial crops, including mosaic of agriculture and pasture) (17). B) shows the spatial distribution of six different land-tenure regimes, collated from Imaflora's Atlas of Brazilian Agriculture (16). 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 quasi-experimental study design that combines matching with a generalization procedure to estimate average treatment effects (ATE) of undesignated/untitled regimes on deforestation in Brazil. For this, we first matched land parcels under alternative tenure regimes to undesignated/untitled land parcels. We used matching covariates known to influence deforestation to broadly capture factors that are likely to be relevant for policy-makers when deciding on shifts in tenure regimes. We then assessed the generalizability of these matched data subsets compared to the entire population of land parcels, using Tipton's index of generalizability (T-index), a metric which captures similarities across covariates in different populations. In order to broaden generalizability of our results, we generated weights for these subsets to more closely represent the covariate distribution of the entire population of land parcels. Subsequently, we estimated population-wide effects via regression analyses while explicitly incorporating generated weights (details in **Methods** and ***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 ~12.4-23.2% relative to all other tenure regimes (**Fig. 2a**, large circles). To assess the consistency 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 141 out of 196 cases (lower deforestation in 6 cases, non-significant in 49, **Fig. S1**, **Tables S3/S8**). These results were qualitatively robust to weighting all cases by balance levels of their respective datasets post-matching, and to filtering out strictly protected and sustainable-use protected areas that were only officially established after the beginning of the respective time period or had unknown establishment dates. Tipton's generalizability index also indicated that covariate distributions were similar across all alternative tenure-regime comparisons and undesignated/untitled land parcels, meaning these results are highly generalizable to the entire population of land parcels at the respective spatial-temporal scales (see ***SI Appendix***; **Fig. S3/S4**, **Tables S3/S6-7**). 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 (35).

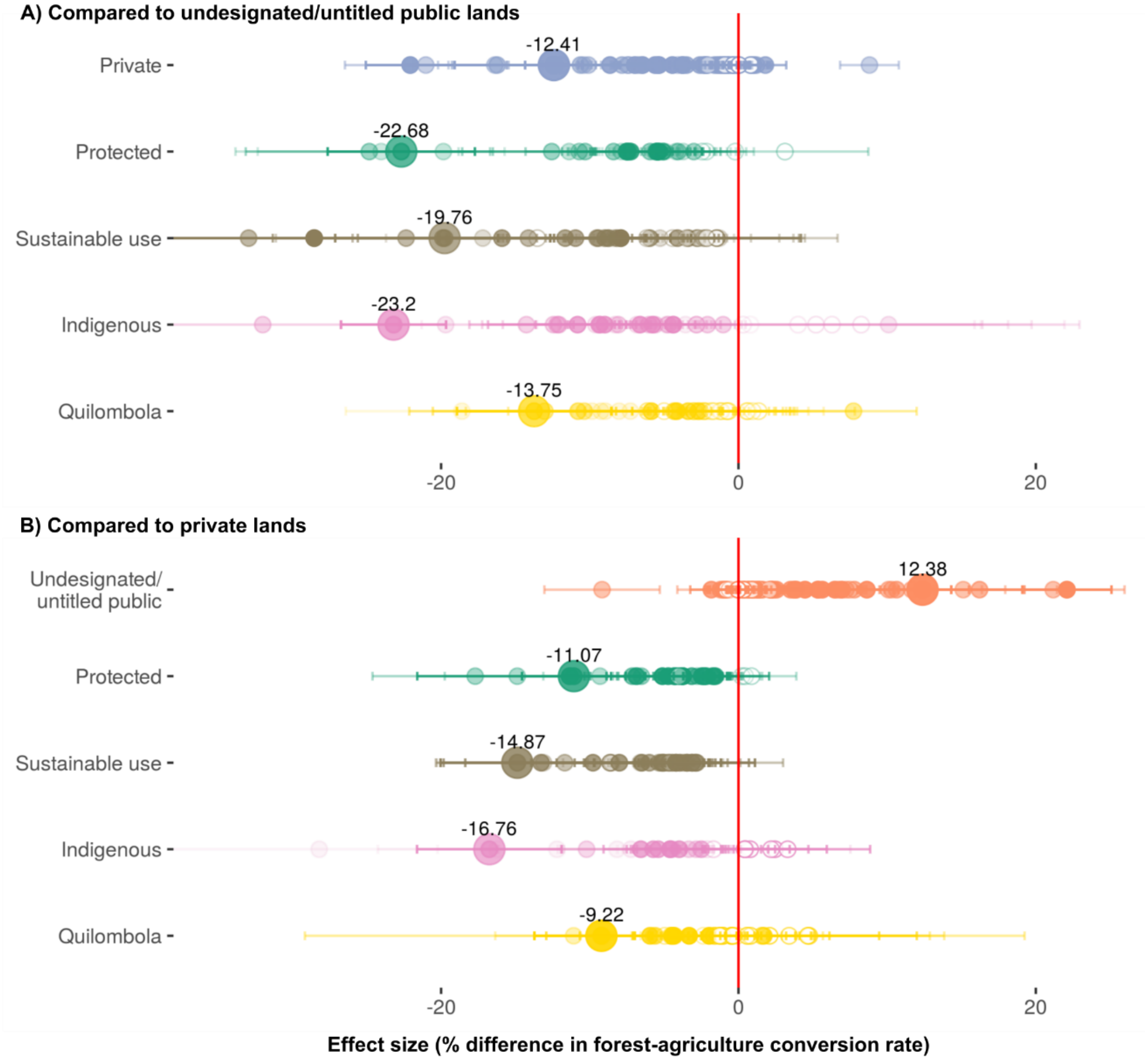

Figure 2. Effects of alternative land-tenure regimes on forest-to-agriculture conversion rates in Brazil. Circles indicate effects sizes estimated at different spatial-temporal scales, compared to two alternative counterfactuals: A) undesignated/untitled public lands with poorly defined tenure rights, 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 for results from time-filtered robustness tests.

Private tenure decreases deforestation vis-à-vis poorly defined tenure, but less so than alternative, well-defined regimes

Over recent decades, global development policies strongly promoted placing undesignated/untitled public lands under private tenure regimes (36) through tenure interventions such as regularization, titling, or registration. Conservation and sustainable-development organizations alike commonly support such interventions (27), 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 consistent these effects are across different contexts.

Similarly to how we analyzed effects of undesignated/untitled tenure, we thus assessed the directionality, magnitude, and consistency 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 12.4% 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. S5**). Yet, these deforestation-reducing effects were not consistent across narrower regional-historical contexts. At these narrower scales, net effects of private tenure were deforestation-decreasing in only 61.7% of cases (63.2% if balance-weighted, deforestation-increasing: 8.5%/8.2% if weighted, non-significant: 29.8%/28.6%; **Fig. S2/S4, Table S8**). These findings indicate that the environmental benefits of tenure interventions promoting private rights over undesignated/untitled lands more often outweigh the risks than vice versa. Yet, they also suggest that private tenure does 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 regimes, including indigenous, community-based, strict-protection, and sustainable-use protection 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 tenure regimes against an undesignated/untitled counterfactual across 34 different scales (***SI Appendix***). To enable indirect comparisons of the performance of the alternative regimes, we weighed the undesignated/untitled counterfactuals to represent the covariate distribution in the entire population of parcels at each respective scale, which effectively standardized the counterfactuals across the tenure-regime comparisons (***Methods***, ***SI Appendix***). These tests revealed that under most regional-historical contexts, private tenure underperformed all alternative regimes in protecting forests with the exception of quilombola regimes (privately owned lands of communities of self-identified descendants of Afro-Brazilian slaves; see also next section). 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%; 2.2% if balance-weighted), and was second-most likely to cause the lowest reductions/highest increases (after quilombola, 26.2%; 28% if balance-weighted; **Table S6**). Overall, these results suggest that among the 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 across vastly different socio-environmental settings.

Protection-oriented tenure regimes reliably decrease deforestation, while effects of community-based regimes are ambiguous

We expected that strict-protection and sustainable-use protection 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 (37) and debate about the extent to which the conserved natural resources should be open to sustainable use (38, 39). Our results support our hypothesis in that strict-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 (88.2% and 76.5% of cases with significant negative effects, respectively, **Table S6**). The above results were robust both to weighting by balance post-matching, and to filtering later-established conservation areas (**Fig. S3 A**, **Tables S3-S6**, see ***SI Appendix***). However, whereas sustainable-use regimes were about five times more likely to outperform than to underperform alternative regimes in protecting forests (largest/smallest deforestation reductions in 41.2/8.8% of cases; 42.6/7.4% if balance-weighted; 47.4/9.2% if time-filtered), this *relative* performance was much less clear for strict-protection regimes (26.5/14%; 26.3/10.6%; 15.7/7.9%; **Tables S6/S7**; note these differences were not confounded by protected-area siting (40), see ***SI Appendix,*** indirect comparisons of relative effects are based on standardized counterfactuals, and T-index scores were all ≥ 0.5, indicating effect estimates are generalizable to the entire population, **Table S3**). This indicates that while any conservation-focused regime may reduce deforestation more reliably than alternative regimes under very different contexts, specifically sustainable-use protection regimes may most reliably achieve *large* reductions across contexts.

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 (4), and are increasingly embraced by environmental policies as critical partners for conserving biodiversity and carbon (41). Provided that IPLC land claims exist, IPLC tenure rights might be recognized over any land. Thus, we assessed effects against both undesignated/untitled and private-tenure counterfactuals. Our results showed that against either counterfactual, both indigenous and quilombola tenure regimes decreased Brazil-wide deforestation during 1985-2018 (**Fig. 2/S1**). Yet, our tenure-regime comparisons across the different spatial-temporal scales yielded inconsistent results. Significant deforestation-reducing effects only emerged in 58.3-59.8% of these comparisons (depending on balance-weighting; **Table S6**). The only specific comparisons that fairly consistently showed deforestation-reducing effects of IPLC tenure were those of indigenous tenure vis-à-vis an undesignated/untitled counterfactual (76.5-82.8% of cases, Table S6). Indigenous tenure reduced deforestation via-à-vis private tenure in only 59.4-70.4% of cases. However, the latter results were only generalizable from matched parcels to larger parcel populations in 17% of cases, mostly in the Cerrado biome (T-index ≥ 0.5 in **Table S4**), reflecting the biased siting of indigenous reserves in areas farther from cities and at higher elevations, relative to the population averages in other biomes (**Table S9**).

Quilombola tenure, in turn, reduced deforestation least reliably and often least effectively among the compared tenure regimes against either counterfactual, notably lacking significant effects during most periods in Caatinga – the biome where most quilombola lands are situated (**Fig. S1/4**). These ambiguous results on the effects of community-based tenure regimes on deforestation rates are in line with diverging theoretical arguments (**Table 1**; **Table S1**). Overall, the limited generalizability and transferability of IPLC-tenure effects on deforestation rates evident in our results suggest that synergies between IPLC tenure and forest conservation objectives may indeed arise in diverse contexts. However, designing policies with these synergies in mind will likely require detailed contextual knowledge to ensure IPLC tenure interventions have positive forest outcomes.

Benefits of private ownership vs. public reserve regimes for protecting forests

While we designed our analysis and cross-contextual synthesis approach to identify consistent (and thus potentially transferable) 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 (strict-protection, sustainable-use, and indigenous) had consistently weaker deforestation-reducing effects vis-à-vis undesignated/untitled regimes than quilombola tenure (communal yet private regimes; **Fig S1)**. Even more surprisingly, private tenure changed from being deforestation-increasing via-à-vis an undesignated/untitled regime 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, not confounded by systematic differences in initial forest cover (**Fig. S7**), and were highly generalizable to the entire Amazonian population of undesignated/untitled and private land parcels (Tipton's index ≥ 0.8; **Tables S3/S4**). 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 (42, 43). 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 (44). This could mean 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 (45) might be effective in reducing deforestation, as it partially transfers 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 (46). 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 likely increased deforestation in most regional-historical contexts, even when solely counting generalizable cases (i.e., 66.7% of country-wide, 77.8% of biome-specific long-term, and 75% of biome-specific short-term tests; mean effects ranging from 1.6% to 28.2% 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 more effectively reduced recent deforestation on Amazonian undesignated/untitled lands than public reserve regimes, directly replacing those alternative public regimes with private tenure would have most likely increased deforestation in Amazonia, particularly after the year 2000 (60.7% of all tested time-periods, 80% 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 (15, 47).

Conclusions

In summary, against a backdrop of oftentimes ambiguous empirical evidence, theories, and interest groups' claims, our study can shed new light on the direction and relative magnitude of the net effects of alternative land- tenure regimes on tropical deforestation. We achieved this through systematic quasi-experimental testing, using weights to generalize our results to the entire population, and synthesizing results across different spatiotemporal scales and contexts. Our results may inform environmental practitioners about likely environmental impacts of different land-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 (48), we could derive several conclusions that were consistent across highly diverse environmental, socio-political, and economic contexts in Brazil. These highly consistent results may be more likely than others to also hold for yet other tropical contexts, and therefore, may be most relevant to other countries that model their forest-governance policies after those in Brazil (49, 50). 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, in turn, appear more likely to cause large reductions. Large reductions are least certain when promoting private land rights, although our more context-specific Amazonian results indicate that this can be highly effective where there are constraints to on-the-ground government control and if 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 most likely increase deforestation. For those tenure regimes for which our assessment does not indicate high generalizability or consistency of effects across scales, 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 context-dependent effects.

Materials and Methods

Tenure Data

We used the comprehensive, publicly available data on land-tenure categories compiled by Imaflora (v.1812; (16)) for 83.4% of the Brazilian territory, which is based on 18 official sources, and was integrated using an expert-vetted system to systematically resolve data conflicts resulting from, e.g., overlapping land claims due to illegally fabricated land titles or mapping errors (51) (***SI Appendix*, sections 1 and 2.1**). 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 ownership status, it can be assumed that for the majority of parcels, the basic type of tenancy (e.g., public institutions vs. indigenous communities vs. private individuals) did not change over the course of our study period. However, as this assumption could be 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 (e.g. all Terra legal parcels were excluded from the analyses). 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. Specific steps are further outlined in our description of the tenure categories analyzed below (also see ***SI Appendix*** section 2.2) and of our study design and statistical approach (also see ***SI Appendix*** sections 3.3-3.6).

We grouped several Brazil-specific categories to correspond to general tenure categories present in most tropical forest nations. Private tenure ('private') was defined as properties with individual ownership, and we included properties from different sources (CAR, SIGEF) but excluded all properties titled by the Terra Legal program, as it only began operating in 2009. Note that deforestation effects of property titling under the Terra Legal program were recently the focus of different study (52). Undesignated and untitled public lands ('undesignated/untitled') were defined as those publicly owned, yet not formally assigned to any purpose or with otherwise poorly defined tenure rights. 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, as the program may bias deforestation behavior in anticipation of a land title (52). We followed the categorization of Brazil's Ministry of Environment for conservation-focused tenure regimes, distinguishing strict-protection ('protected areas') from sustainable-use protected regimes ('sustainable use'). Areas of environmental protection (*Áreas de Proteção Ambiental*) are excluded from the Imaflora dataset, and thus not included in this analysis (16). We maintained three categories for indigenous or local community-based (IPLC) tenure regimes, ('indigenous', 'quilombola', and 'communal') given the differences in their histories, legal statuses, and bundle of rights (***SI Appendix,* section 2.2**). Communal lands were excluded from the main results reported here due to the heterogeneity of their bundle of rights, and because there were insufficient recorded communal land parcels to support our analyses in all biomes except for Amazonia. Results for communal regimes are provided in **SI Appendix Fig. S1-S2**, and **Tables S3-S4**.

Forest cover and covariate data

We used the 30-m-resolution annual land-cover/use data provided by Mapbiomas (17) 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 that are likely to be relevant for policy-makers when deciding on shifts in tenure regimes. These include market accessibility (represented by travel time to nearest city; (53)) and agricultural suitability (represented by slope and elevation; (54)). Both of these variables strongly determine achievable land rents and thus the opportunity costs of 'assigning' parcels to particular tenure regimes, while also capturing the inherent bias of the siting of different tenure regimes (40). We also included human population density (55) as larger populations can more strongly influence policy processes for formalization of property rights (e.g., via titling of private regimes or recognition of IPLC land claims), whereas lower population density implies more liberty to create conservation regimes or leave land undesignated. Finally, we included parcel area in ha (16), because property size influences the prices landholders pay for receiving land titles (52), as well as specific forest/agricultural policy requirements and levels of compliance to these policies (56)(see details in ***SI Appendix,* section 2.4**).

Study design

Our goal was to assess and synthesize the direction, strength, and consistency of the longer-term effects that plausible shifts between alternative land-tenure regimes would have on agriculture-related deforestation rates in Brazil. 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 apply across diverse socio-environmental settings within Brazil, and thus, potentially transfer 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**).

Matching

For each of these scales and tenure-regime comparisons, we tested effects using a quasi-experimental study design (***SI Appendix,* section 3**). We first applied coarsened-exact matching implemented in the '*cem*' package (57) in R (versions 3.5.1-4.0.2) (58), which involves temporarily 'coarsening' each confounding variable into bins (predetermined strata), and dropping unmatched observations from the sample. 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. We use the L_1 measure developed by King et al. (57) to calculate remaining imbalance post-matching. 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 ***SI Appendix*** 3.).

Estimation of population-wide effects

Post-matching, we faced the limitation that although exact-matching using CEM improved the balance in the data and the robustness of estimates, dropping non-matched observations limited the generalizability of effects exclusively to the matched subsample of data. Given our overarching aim to determine the generality of effects, we applied recently developed statistical methods that extend the generalizability of effects from a sample of data to a broader population (59). Specifically, we first conducted a generalizability assessment of each of these tenure-regime comparisons at each scale considered using the *generalize* package in R (59). We calculated Tipton's index of generalizability (T-index), a metric that describes levels of covariate similarity between two groups (i.e., here, between the matched subset of land parcels and the entire population of parcels at a given spatial-temporal scale) (**Tables S3-S4**). To distinguish cases where matched data subsets were sufficiently different than the entire population of land parcels, we also calculated the absolute standardized mean difference (ASMD), of each covariate (***SI Appendix*, Table S10**). Then, we generated weights in order for the matched-data subsample to more closely represent the entire population of land parcels. We calculated parcels' weights as the inverse odds of their probability of being matched, meaning that observations with a greater probability of being in the entire population had greater weights. Weights were calculated using Lasso, and were incorporated into the estimation of effects.

We estimated effects 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, the previously generated weights to resemble the entire population of parcels, 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 estimated:

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

where p is the per-pixel probability of forest conversion, tf is the tenure regime, 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, *w* is the generated weights, 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 (60), transforming coefficient estimates to average per-forest-pixel probabilities of conversion to agriculture with respect to the tenure form in question (61) (**Tables S3-S4**).

Finally, we tested the sensitivity of our results to potential omitted-variable bias by calculating Rosenbaum bounds (***SI Appendix*, sections 3.4**, **Tables S3-S4, S9**). 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**).

Consistency of findings across scales

We formally synthesized the estimated scale-specific effects via two complementary approaches. First, we assessed the consistency of the direction of the effects by calculating percentages of scale-specific models with, respectively, significant deforestation-increasing, significant deforestation-decreasing, and no significant effects (***SI Appendix*, Table S6-8**). Second, we assessed how consistent the relative rankings of alternative tenure regimes were in terms of 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 S6-8**). Note that, although relative ranks were inherently indirect comparisons of alternative tenure regimes to differently-matched counterfactuals, both the undesignated/untitled counterfactual and the private counterfactuals were weighted to represent the covariate distribution in the entire population of parcels at each respective scale. This weighting thus effectively provided a standardized counterfactual for effect estimations across all tenure-regime comparisons at a given scale.

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

*Correspondence to: andrea.pacheco@idiv.de, carsten.meyer@idiv.de

This appendix includes:

Supplementary text (extended materials and methods)
Figures S1 to S5
Tables S1 to S10
Legends for Datasets S1 to S2
SI References

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 ownership status, it can be assumed that for the majority of parcels, the basic type of tenancy (e.g., public institutions vs. indigenous communities vs. private individuals) did not change over the course of 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 transferable 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 titling 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 strict-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). Note, Areas of Environmental Protection (*Áreas de Proteção Ambiental*) are excluded from the Imaflora dataset, and thus not included in this analysis (8).

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 which are have been shown to be relevant for policy-makers when deciding on shifts in tenure regimes under many different contexts. These include market accessibility (represented by travel time to nearest city; (16)), agricultural suitability (represented by slope and elevation; (17)), population density (18), and parcel area in ha (8). See sections 3.3 and 3.6 for details on covariate use. Agricultural suitability and accessibility have been shown to be key determinants of achievable land rents, and are thus good proxies for the opportunity costs associated with ‘assigning’ land to any given tenure regime. These covariates thus also reflect general tendency of potentially more profitable (i.e., private) tenure regimes to be established near markets and in agriculturally suitable lowlands, whereas likelihoods of acknowledging IPLC rights or creating conservation regimes are higher in more remote and/or steeper terrains of relatively lower economic importance (9, 19–23). Higher population density can translate into higher pressure on policy-makers to ‘assign’ or recognize tenure regimes that allow the use of natural resources (e.g. private, IPLC), whereas lower population density implies lower economic, social, and/or political costs of creating conservation regimes or of leaving lands undesignated. Finally, we included parcel area in ha (8) because the price landholders pay for receiving land titles depends on a property’s size (12) and because certain forest/agricultural policies apply differently to certain tenure regimes depending on parcel size (e.g., requirements for private landholders to maintain a certain percentage of forest cover, or requirements for rural settlements to maintain certain levels of agricultural productivity). Furthermore, compliance with these policies has also been shown to depend on property size (24).

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 consistent and thus, how transferable effects of these regime differences are across different regional-historical contexts.

To estimate causal effects from observational data, we used a quasi-experimental study design that combined matching with a generalization assessment and subsequent regression analysis that incorporates weights to estimate population-wide effects (25). To test the extent to which the effects were consistent 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 consistency of the net direction of the effects, and *ii)* the consistency of their relative strength in comparison to the effects of other tenure regimes.

3.1. Plausible changes in land-tenure regimes

We defined '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 (26, 27)).

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, 28, 29). Notwithstanding this practice, causal effects in complex human-environment systems are often highly context-specific (30), which can limit the transferability of conclusions from contextually bound studies. Yet, effects shown to be consistent across very different socio-environmental contexts may also hold in 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 statistical 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; (31), the proposal of REDD+; (32)). 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 estimate causal effects from observational data, we used a quasi-experimental study design, and combined matching with a subsequent regression analysis that included weights to generalize from matched samples to population-wide effects (25, 33). Matching addressed the bias that would arise due to 'treatment' assignment not being independent of the outcome. For instance, landscapes (e.g. savannas) may be more prone to certain land-uses (e.g. agriculture), which may influence 'treatment' assignment (e.g. titling agricultural land to a private land holder, recognizing a forest as part of an indigenous land claim). If simpler regression designs were applied to the tenure dataset due to this non-random assignment of tenure regimes into experimental 'treatment/control' groups, results would be highly biased and model dependent due to high levels of imbalance. Thus, we specifically used coarsened exact matching (CEM; (25)), which addressed this bias by pruning the dataset to matched pairs of parcels that were highly similar with regard to potentially confounding variables in a stratified way. We conducted one-to-one matching, meaning that each pair of parcels contained one parcel coded as 'treatment' under one of two compared alternative tenure regimes, and another (the 'control' or 'counterfactual') under the respective other regime. Effects were subsequently estimated via regression on the balanced-improved 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 was currently the most feasible approach for addressing our question. 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.

We also note that our analysis relies on the non-interference assumption, i.e., that the outcome of an observation is not affected by any other 'treatment'. This would require the deforestation of a land parcel under a particular tenure regime to be unaffected by neighboring (or even distant) tenure regime dynamics. While this would be difficult to prove empirically, recent research on deforestation 'spillover' effects of both conservation and indigenous regimes onto other tenure regimes found non-significant or minimal effects for most of Brazil (34). This study found only one case of spillover effects in the Amazonian state of Pará during 2000-2004, where conservation regimes were shown to cause decreasing deforestation outside their boundaries, whereas indigenous lands caused ‘leakage’, i.e., increasing deforestation elsewhere. This means that, although we cannot rule out there may be some ‘spillover’ effects at play in our study system, these are likely negligible in most cases. In those cases where these effects might not be negligible, our results would likely underestimate deforestation-decreasing effects of conservation regimes, while overestimating deforestation-decreasing effects of indigenous lands in Amazonia.

Estimating causal effects via matching also 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 (35). 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 (36) in R versions 3.5.1-4.0.2 (37). 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 (25), 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; (38); 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. (36) 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 Improving generalizability and estimating Average Treatment Effects (ATE)

In this study we defined our estimand of interest as Average Treatment Effects (ATE), i.e. the average difference between two tenure regimes (a 'treatment' and 'counterfactual') on forest converted to agriculture. While other studies might have different estimands of interest (e.g. Average Treatment Effects on the Treated (ATT), or even on the Untreated (ATU))(39) our aim was to capture population-wide effects in order to broadly measure the influence of different tenure regimes across Brazil.

We faced the limitation that although exact-matching using CEM improved the balance in the data and the robustness of estimates, dropping non-matched observations limited the generalizability of effects exclusively to the matched subsample of data (i.e. meaning effect estimated would be average treatment effects on the matched sample (ATM)). Given our overarching aim to determine the generality of effects, we applied recently developed statistical methods that extend the generalizability of effects from a sample of data to a broader population (40). Thus, using these statistical techniques and ensuring data requirements were met (39, 40), the matched data subsample resulting from the matching procedure was used to estimate effects that were generalizable to the broader, target population of all Brazilian land parcels (ATE).

We specifically used a weighting approach to thus extend effect estimates to the entire population of Brazilian land parcels – within each particular spatiotemporal tenure-regime comparison. For this, we first obtained a stratified representative sample of the entire population of land parcels (of each tenure-regime comparison, at each spatiotemporal scale considered) in order to facilitate subsequent computational processing times. We used the same covariates used for matching (i.e. elevation, slope, travel time to nearest city, human population, and area) to stratify the entire population of parcels and extract a representative sample. Then, using the matched-data subsets and the stratified representative sample of the entire population, we conducted a generalizability assessment of each of these tenure-regime comparisons at each scale considered using the *generalize* package in R (40). We calculated Tipton's index of generalizability (T-index), a metric that describes levels of covariate similarity between two groups (i.e. here, the matched subset of data, and the entire population of land parcels) (**Tables S3-S4**). T-index values range from 0-1, with values closest to 1 describing a population that is highly generalizable, and values under 0.5 are likely not generalizable because the two groups are too dissimilar.

After assessing generalizability, we generated weights in order for the matched subsample to more closely represent the entire population. Weights were calculated as the inverse odds of their probability of being matched, meaning that observations with a greater probability of being in the entire population had greater weights, and were obtained via lasso. Here, it is important to note that we trimmed the population to only include observations that did not exceed bounds of the matched covariates, in order to comply with the coverage assumption as a necessary condition to make further generalizations (40). To best characterize cases where matched data subsets were sufficiently different than the entire population of land parcels, we also calculated the absolute standardized mean difference (ASMD), of each covariate (**Table S10**). Finally, in order to estimate ATE, weights were incorporated into subsequent regression models using the uncoarsened matched-data subset (see section 3.5).

Note, in a few cases (4 undesignated/untitled models in Caatinga (**Table S3**)), T-index calculations failed due to a low sample size of the entire population of land parcels, preventing any statement on the generalizability of these cases to their entire populations. While weights were still generated and included in the final statistical models (see 3.5), effect estimates may not be generalizable to the entire population of land parcels in these cases, but only apply to the (weighted) matched-data subsample.

3.5. Regression analyses

For each scale and tenure-regime comparison, we estimated effects 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, previously generated weights, 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.

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

where p is the per-pixel probability of forest conversion, *tf* is the tenure regime, 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, w is the generated weights, 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 (41), transforming coefficient estimates to average per-forest-pixel probability of conversion to agriculture with respect to the tenure form in question (42) (**Tables S3-S4**).

Note that in rare cases, insufficient observations distributed across federal states prevented the estimation of coefficients for all parameters in those models. We addressed this by consecutively merging geographically-adjacent states until parameters could be estimated, keeping the merging protocol as consistent as possible across models (see **Table S5**). This merging of states allowed for the correct estimation of parameters in 3 models. However, the model still failed to converge in the remaining 3 models, likely due to the insufficient number of observations distributed across federal states causing a pattern in the data commonly known as complete separation (43, 44). While this kind of convergence issue in logistic regression is well known (43, 44), achieving model convergence for these cases would likely require using a different modeling approach, and could involve excluding federal state as a variable in the model (44). We maintained the modeling approach that was most appropriate for the vast majority of data in this analysis, and report models with convergence issues (**Tables S3-5**).

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 (45). 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 showed that both Hodges-Lehman estimates and *p*-values were not highly sensitive to possible small omitted-variable bias (Γ = 1.1), and were still reasonably robust to possible large omitted-variable bias (Γ = 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 Γ = 1.1, to 48.72%, 44.48% and 46.92% at Γ = 1.5 (**Table S8**; relative error calculated as percentage of the magnitude of the respective median effect size at Γ=1). Average sensitivities of significance of effects (p ≤ 0.05) ranged from, respectively, 2.7%, 4.2% and 3.2% of models with a sensitive effect significance at Γ = 1.1, to 17.3%, 15.6% and 18.11% at Γ = 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.

3.6. 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 in two ways. First, for each comparison (e.g., private vs. undesignated/untitled), we assessed the consistency 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 consistency 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. Note that, although these relative rankings indirectly compare alternative tenure regimes to differently-matched counterfactuals, as a part of the analysis that extends the generalizability of effect estimates, both undesignated/untitled and private counterfactuals were weighted to represent the covariate distribution in the entire population of parcels at each respective scale evaluated. This weighting effectively provided a standardized counterfactual for all estimations across tenure-regime comparisons at different scales.

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 (46).

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 downweigh 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 strict-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 strict-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 (47). 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, strict-protection regimes were often ranked as less effective in reducing deforestation than sustainable-use areas in the Amazonia and Mata Atlântica biomes, despite occurring in, respectively, more remote, and higher-elevation areas on average (cf. (19).

3.7 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 addressed 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.

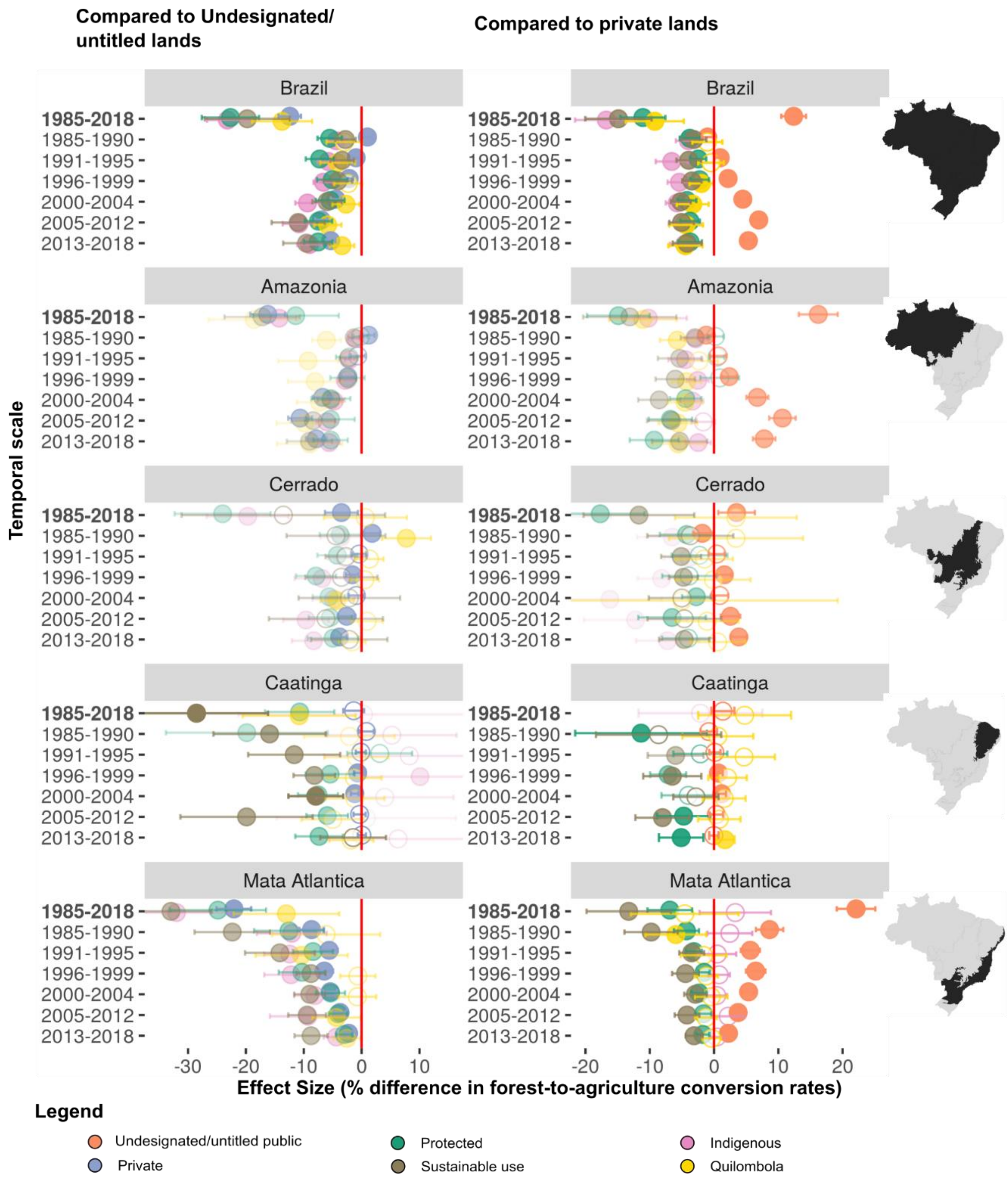

Fig. S1. Effects of alternative land-tenure regimes on forest-to-agriculture conversion rates in Brazil, disaggregated to different spatiotemporal scales. Circles indicate effects sizes estimated at the respective scale vis-a-vis two alternative counterfactuals: a) undesignated/untitled public lands,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

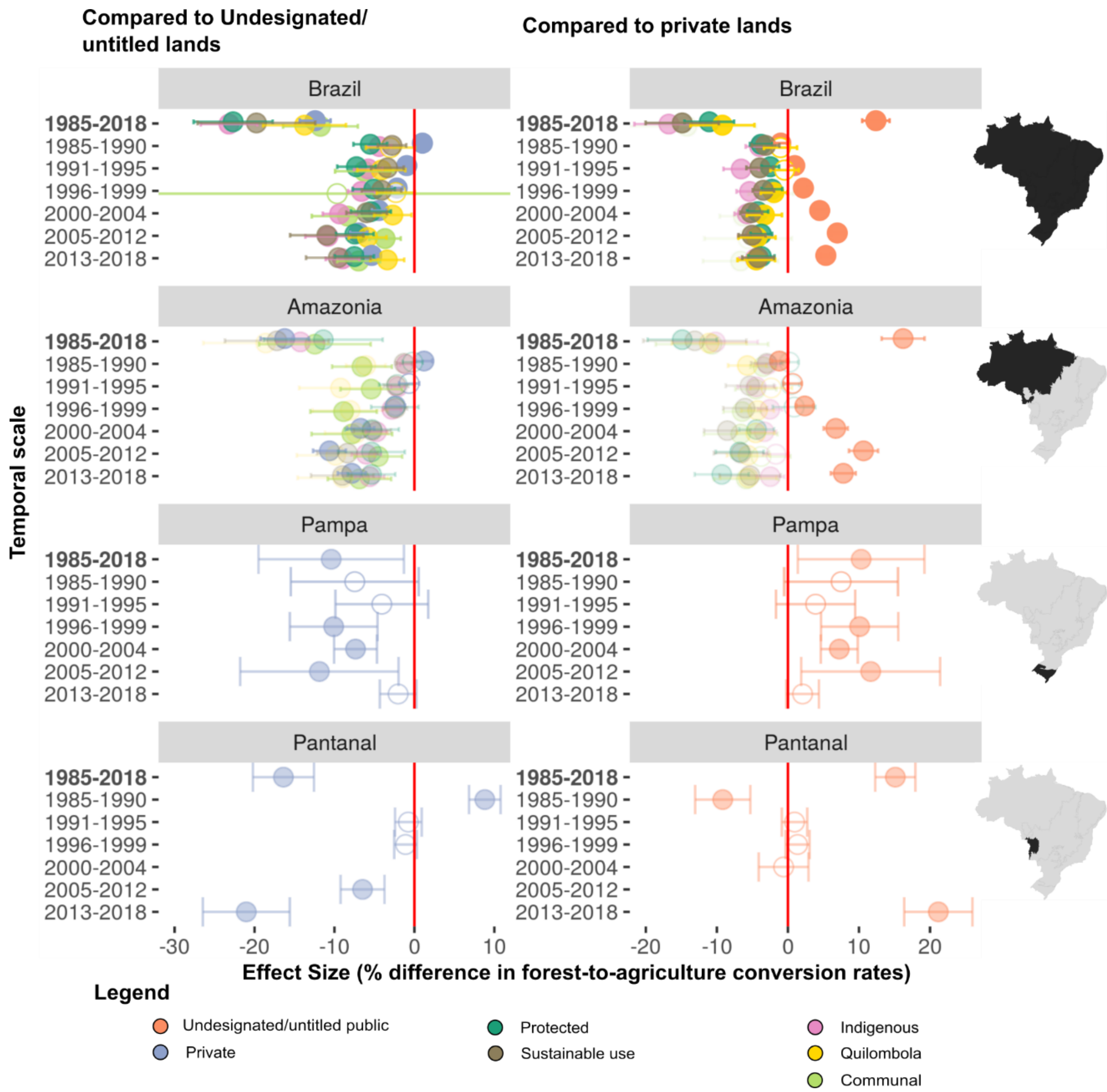

Fig. S2. Effects of alternative tenure regimes on forest-to-agriculture conversion rates at different spatiotemporal scales, complementing Fig. S1 by showing additional results for communal tenure regimes for Brazil and the Amazonia biome, and for private and undesignated/untitled regimes for Pampa and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

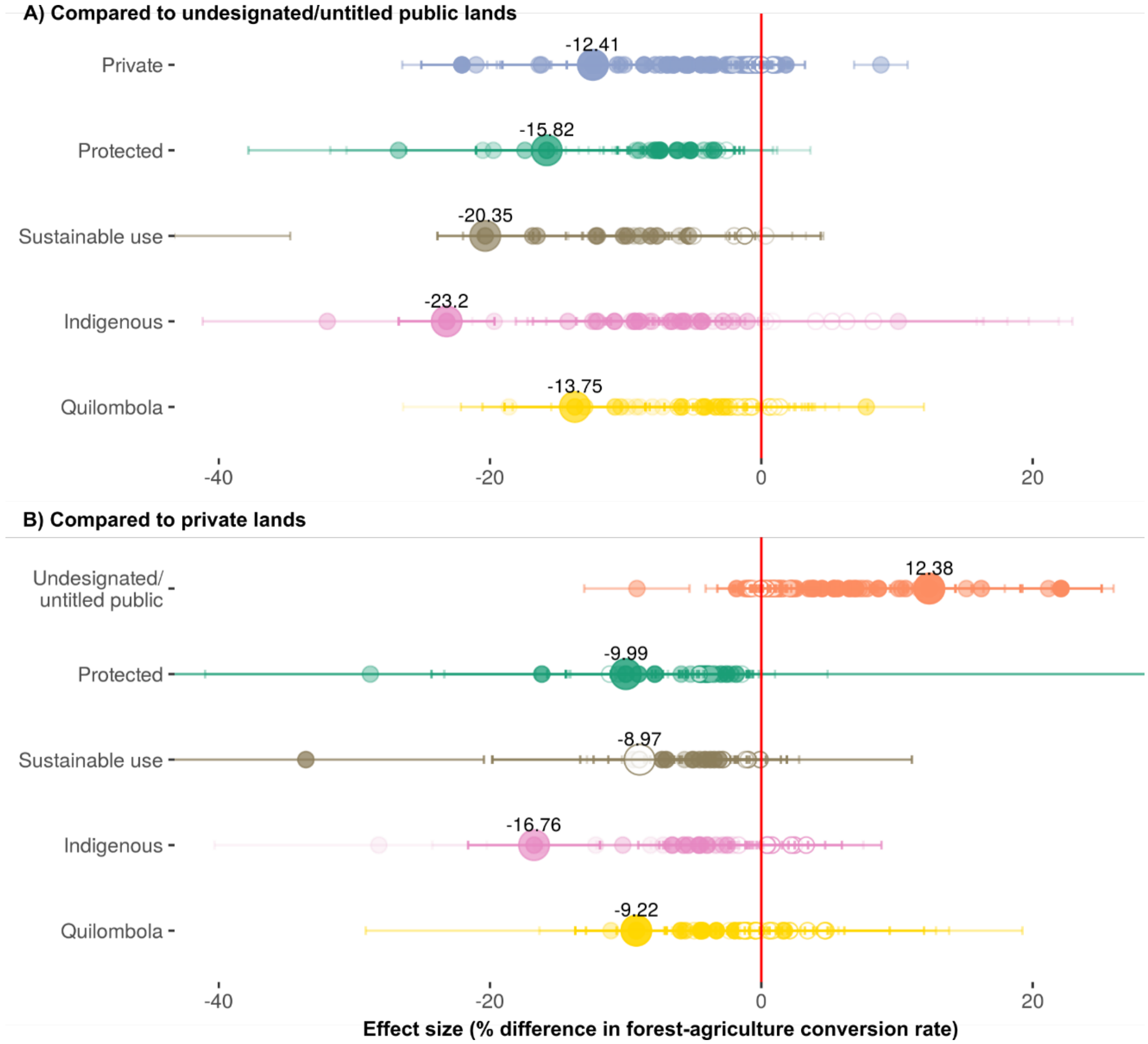

Fig. S3. Robustness test of effects of alternative tenure regimes on forest-to-agriculture conversion rates in Brazil using filtered time-series data for protected and sustainable-use areas (i.e., only areas established before/during beginning of each temporal scale considered; see section 2.2). Circles 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).

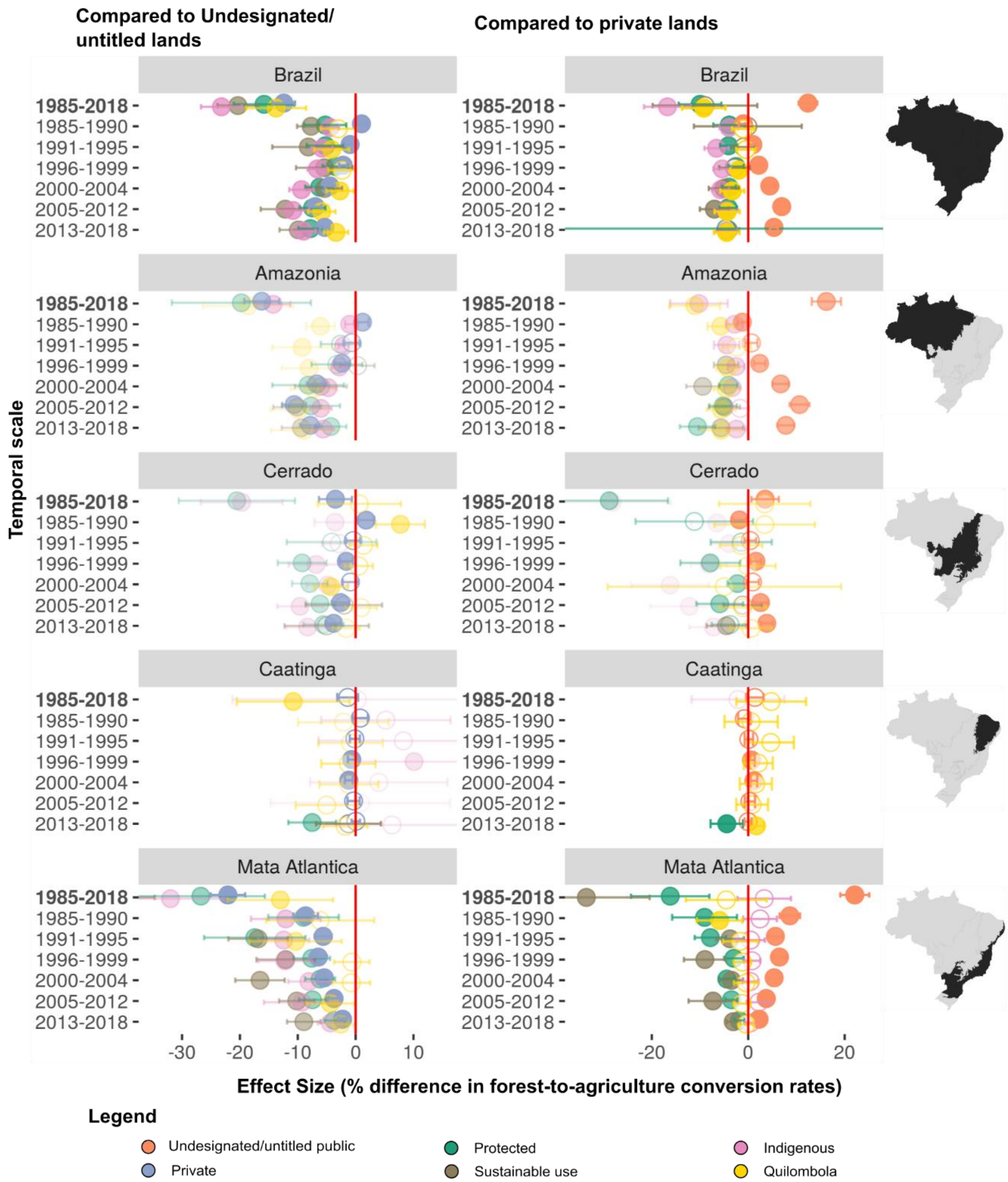

Fig. S4. Spatiotemporal disaggregation of robustness test of effects of alternative tenure regimes on forest-to-agriculture conversion rates in Brazil using filtered time-series data for protected and sustainable-use areas (i.e., only areas established before/during beginning of each temporal scale considered;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

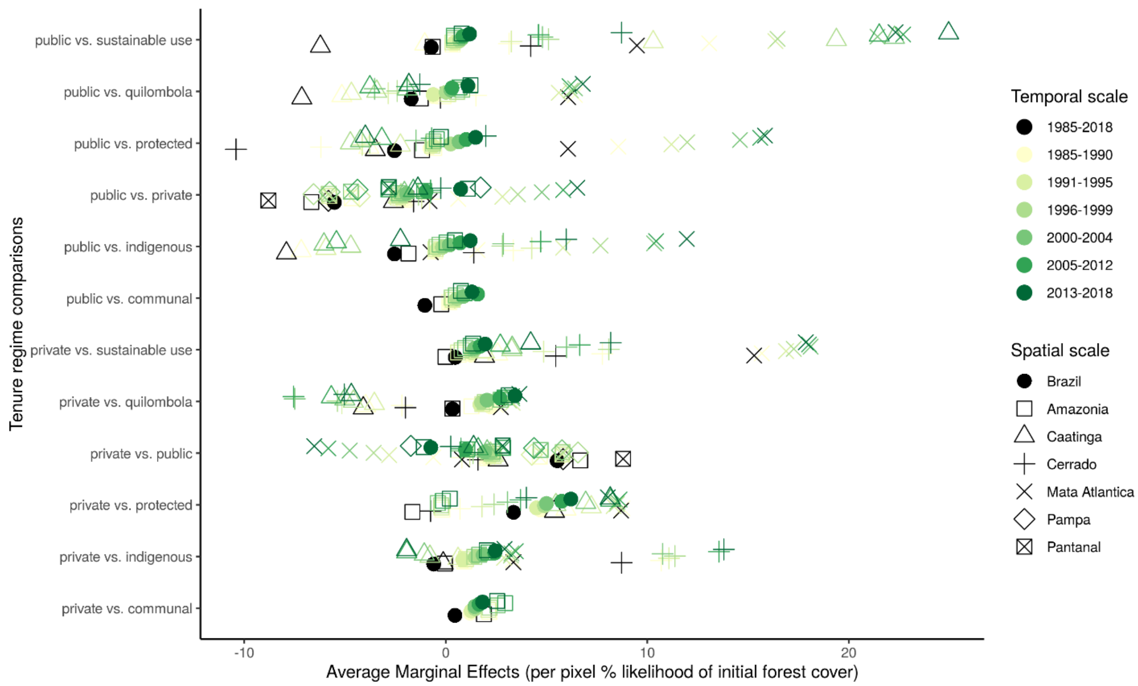

Fig. S5. Differences in initial forest cover between matched treatment and control units for different tenure-regime comparisons at different spatial and temporal scales. Average marginal effects indicate the per-pixel likelihood being forest-covered at the beginning of each time period considered. At 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	Undesignated /untitled c	Protected	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 (48–50). 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 (51, 52).	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 (53, 54).	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 (55).	Deininger et al., 2003	↗	↘	↘	↘	↘	↗	↘	↘	↘	↘	↘	↘
Alienation and withdrawal rights	Only land that can be legally be sold or otherwise alienated by the current tenant is potentially available to people searching for land for farming (mainly private, and to a lesser extent communal and undesignated/untitled lands). Because the expected higher agricultural profits enabled by commercial withdrawal rights tend to be factored into land prices for private lands on formal land markets, these are often unaffordable to poor smallholders or land-less settlers searching for	Binswanger, 1991	↘	↘	↘	↘	↘	↘	↗	↘	↘	↘	↘	↗

Thematic dimension	Hypothesized mechanism	References	Predicted effect of tenure regime on deforestation, relative to undesignated/untitled public lands						Predicted effect of tenure regime on deforestation, relative to private lands					
			Private	Protected area	Sustainable use	Indigenous	Quilombola	Communal	Undesignated /untitled c	Protected	Sustainable use	Indigenous	Quilombola	Communal
	land. These will thus instead be forced to settle on undesignated public lands at the 'frontier' (56).													
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 (57).	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 (58).	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)(59). 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 (60–62). 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 /untitled c	Protected	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 (63–67).	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 (68–70). 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 (71–73). 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 (60, 51, 61, 74). 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	↘	→	→	→	↘	→	↗	↗	↗	↗	↘	↗

Thematic dimension	Hypothesized mechanism	References	Predicted effect of tenure regime on deforestation, relative to undesignated/untitled public lands						Predicted effect of tenure regime on deforestation, relative to private lands					
			Private	Protected area	Sustainable use	Indigenous	Quilombola	Communal	Undesignated /untitled c	Protected	Sustainable use	Indigenous	Quilombola	Communal
Cross-cutting themes														
Number of resource users and/or decision-makers	Decision making regarding the use and conversion of forests have higher transaction costs in community-based tenure forms because it takes more time and resources to reach decisions with larger numbers of people (75, 76). 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 (75, 77). 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)(48, 50, 78).	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 (79). 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 (60, 61, 80, 81). While private land tenure is classically viewed as providing the highest tenure security and thus assurance levels, this view is not universal (82).	Holden and Yohannes, 2002; Angelsen, 1999; Fearnside, 2005; Deininger and Jin, 2006; Fenske, 2011; Robinson et al., 2004	↗ ↘			↘	↘	↘	↗ ↘			↘	↘	↗ ↘

Thematic dimension	Hypothesized mechanism	References	Predicted effect of tenure regime on deforestation, relative to undesignated/untitled public lands						Predicted effect of tenure regime on deforestation, relative to private lands					
			Private	Protected area	Sustainable use	Indigenous	Quilombola	Communal	Undesignated /untitled c	Protected	Sustainable use	Indigenous	Quilombola	Communal
	Assuming that classical views on tenure-form–tenure-security relationships broadly hold and that landholders are mainly economically/personal-survival motivated, this set of hypotheses would predicts a skewed u-shaped relationship between tenure security and deforestation rates, where deforestation is medium-high at very low tenure security levels (e.g., informal settlements on public lands), lowest at intermediate levels of tenure security (i.e. indigenous, quilombola, and communal lands), and highest under highest assurance levels (e.g. private tenure).													
Governance (monitoring and enforcement)	Tenure regimes where the state (i.e., citizenry) is the main or exclusive rights and duty holder, such as protected areas or other lands administered by public institutions, are expected to have lower 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 (51).	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 (83, 84).	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 (85).	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 (73). 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).

Brazil tenure categories	**Tenure regime**	**Tenants**	**Bundles of rights** (usually included)							**Main right holder**	**Main duty holder**	**References**
			Access	Withdrawal (subsistence)	Withdrawal (commercial)	Management	Exclusion	Alienation	Due Process			
CAR poor (properties with more than 5% of overlapping areas with neighbors) CAR premium (properties with less than 5% of overlapping areas with neighbors) SIGEF (Private properties registered in INCRA systems) Private properties from Terra Legal program	Private lands	1	++	+	+	+	++	++	++	Individual(s), firm, or other entity	Individual(s), firm, or other entity	Lei 4.947 art. 22 1966, (86)
Communitary lands	Communal lands	Many	++	+	+/-	+/-	+/-	+/-	-	Community	GO	Decreto N. 6.040, 2007, Lei N. 11.284, 2006, (87, 88).
Quilombola lands	Quilombola lands	Usually many	++	+	+	+	+	--	+	Community	Community	Consitucao Federal art. 68, Decreto N. 6.040, 2007, (89, 90).
Homologated Indigenous land Non-homologated indigenous land	Indigenous lands	Usually many	++	+	--	+	+	--	+	Community	GO	Consitucao Federal art. 231. 1996, Decreto N. 6.040, 2007, (91).
Full protection conservation unit	Strictly 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 lands from Terra legal program	Undesignated/un titled public lands	1 or few	++	++	++	+	-	-	--	Citizenry	GO	MP 759/2016, Lei Nº 8.629, de 25 de fevereiro de 1993, (9)
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 (ImbAfter). Scores on Tipton's index of generalizability are reported (T-index), with values closer to 1 indicating high levels of generalizability, and scores ≤ 0.5 preventing generalizability between matched samples and entire population of land parcels at each respective scale and comparison. 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	**Spatial scale**	**Temporal scale**	***n***	**SE**	***p* value**	**lower_ci**	**upper_ci**	**Effect**	**ImbBefore**	**ImbAfter**	**T-index**
Communal	Amazonia	1985-1990	914	0.019	0.001	-0.103	-0.028	-0.066	0.761	0.422	0.835
Communal	Brazil	1985-1990	1,148	0.000	0.000	-2.755	-2.755	-2.755	0.809	0.277	0.697
Communal	Amazonia	1985-2018	912	0.036	0.000	-0.194	-0.055	-0.124	0.761	0.428	0.850
Communal	Brazil	1985-2018	1,146	0.024	0.000	-0.164	-0.071	-0.118	0.810	0.281	0.702
Communal	Amazonia	1991-1995	914	0.020	0.006	-0.092	-0.016	-0.054	0.761	0.414	0.831
Communal	Brazil	1991-1995	1,148	0.019	0.001	-0.099	-0.026	-0.063	0.809	0.247	0.708
Communal	Amazonia	1996-1999	914	0.021	0.000	-0.130	-0.047	-0.089	0.761	0.398	0.857
Communal	Brazil	1996-1999	1,148	2,474,231	1.000	-4,849,403	4,849,403	-0.097	0.810	0.251	0.701
Communal	Amazonia	2000-2004	914	0.026	0.002	-0.129	-0.028	-0.078	0.761	0.403	0.846
Communal	Brazil	2000-2004	1,148	0.023	0.000	-0.128	-0.039	-0.083	0.810	0.256	0.707
Communal	Amazonia	2005-2012	912	0.015	0.003	-0.075	-0.015	-0.045	0.761	0.432	0.833
Communal	Brazil	2005-2012	1,148	0.010	0.000	-0.056	-0.017	-0.037	0.810	0.244	0.695
Communal	Amazonia	2013-2018	908	0.020	0.001	-0.108	-0.029	-0.069	0.763	0.425	0.842
Communal	Brazil	2013-2018	1,146	0.014	0.000	-0.097	-0.043	-0.070	0.811	0.274	0.715
Indigenous	Amazonia	1985-1990	456	0.004	0.011	-0.018	-0.002	-0.010	0.743	0.531	0.889
Indigenous	Brazil	1985-1990	902	0.008	0.000	-0.059	-0.029	-0.044	0.721	0.273	0.920
Indigenous	Caatinga	1985-1990	44	0.057	0.361	-0.060	0.164	0.052	0.892	0.636	0.753
Indigenous	Cerrado	1985-1990	80	0.018	0.049	-0.071	0.000	-0.036	0.871	0.700	0.875
Indigenous	Mata Atlantica	1985-1990	194	0.031	0.000	-0.181	-0.061	-0.121	0.772	0.474	0.862
Indigenous	Amazonia	1985-2018	456	0.015	0.000	-0.172	-0.113	-0.143	0.743	0.535	0.899
Indigenous	Brazil	1985-2018	902	0.018	0.000	-0.267	-0.197	-0.232	0.722	0.286	0.920
Indigenous	Caatinga	1985-2018	44	0.110	0.978	-0.213	0.219	0.003	0.892	0.682	NA
Indigenous	Cerrado	1985-2018	80	0.036	0.000	-0.268	-0.126	-0.197	0.882	0.650	0.738
Indigenous	Mata Atlantica	1985-2018	194	0.047	0.000	-0.412	-0.228	-0.320	0.773	0.536	0.901
Indigenous	Amazonia	1991-1995	456	0.004	0.000	-0.030	-0.012	-0.021	0.743	0.531	0.891
Indigenous	Brazil	1991-1995	902	0.009	0.000	-0.076	-0.040	-0.058	0.721	0.273	0.916
Indigenous	Caatinga	1991-1995	44	0.075	0.271	-0.064	0.229	0.082	0.892	0.636	0.656
Indigenous	Cerrado	1991-1995	82	0.015	0.066	-0.058	0.002	-0.028	0.871	0.659	0.790
Indigenous	Mata Atlantica	1991-1995	194	0.023	0.000	-0.169	-0.080	-0.124	0.772	0.536	0.883
Indigenous	Amazonia	1996-1999	456	0.006	0.000	-0.040	-0.016	-0.028	0.743	0.531	0.897

Treatment	Spatial scale	Temporal scale	*n*	SE	*p* value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	T-index
Indigenous	Brazil	1996-1999	902	0.009	0.000	-0.083	-0.049	-0.066	0.721	0.273	0.895
Indigenous	Caatinga	1996-1999	44	0.049	0.040	0.005	0.197	0.101	0.892	0.636	0.828
Indigenous	Cerrado	1996-1999	80	0.024	0.005	-0.116	-0.021	-0.068	0.882	0.650	0.856
Indigenous	Mata Atlantica	1996-1999	194	0.024	0.000	-0.168	-0.076	-0.122	0.773	0.526	0.913
Indigenous	Amazonia	2000-2004	456	0.007	0.000	-0.061	-0.034	-0.047	0.743	0.535	0.881
Indigenous	Brazil	2000-2004	900	0.011	0.000	-0.115	-0.073	-0.094	0.722	0.282	0.907
Indigenous	Caatinga	2000-2004	44	0.061	0.508	-0.079	0.159	0.040	0.892	0.682	0.763
Indigenous	Cerrado	2000-2004	80	0.019	0.009	-0.085	-0.012	-0.048	0.882	0.675	0.823
Indigenous	Mata Atlantica	2000-2004	194	0.018	0.000	-0.117	-0.045	-0.081	0.773	0.536	0.904
Indigenous	Amazonia	2005-2012	456	0.009	0.000	-0.079	-0.042	-0.060	0.743	0.535	0.906
Indigenous	Brazil	2005-2012	902	0.014	0.000	-0.136	-0.080	-0.108	0.723	0.282	0.902
Indigenous	Caatinga	2005-2012	44	0.079	0.919	-0.147	0.163	0.008	0.892	0.727	0.771
Indigenous	Cerrado	2005-2012	80	0.020	0.000	-0.136	-0.057	-0.096	0.882	0.650	0.832
Indigenous	Mata Atlantica	2005-2012	194	0.034	0.006	-0.159	-0.027	-0.093	0.773	0.526	0.896
Indigenous	Amazonia	2013-2018	454	0.009	0.000	-0.074	-0.038	-0.056	0.743	0.533	0.912
Indigenous	Brazil	2013-2018	896	0.011	0.000	-0.111	-0.069	-0.090	0.724	0.277	0.866
Indigenous	Caatinga	2013-2018	46	0.060	0.298	-0.056	0.181	0.063	0.891	0.652	0.810
Indigenous	Cerrado	2013-2018	80	0.019	0.000	-0.120	-0.045	-0.083	0.883	0.650	0.941
Indigenous	Mata Atlantica	2013-2018	194	0.011	0.000	-0.065	-0.022	-0.044	0.774	0.536	0.864
Private	Amazonia	1985-1990	8,066	0.005	0.024	0.002	0.022	0.012	0.638	0.353	0.829
Private	Brazil	1985-1990	34,212	0.005	0.033	0.001	0.019	0.010	0.663	0.123	0.807
Private	Caatinga	1985-1990	10,020	0.007	0.213	-0.005	0.021	0.008	0.714	0.142	0.756
Private	Cerrado	1985-1990	9,670	0.007	0.012	0.004	0.032	0.018	0.718	0.256	0.896
Private	Mata Atlantica	1985-1990	5,130	0.011	0.000	-0.108	-0.065	-0.086	0.744	0.160	0.794
Private	Pampa	1985-1990	404	0.041	0.068	-0.155	0.005	-0.075	0.843	0.391	0.592
Private	Pantanal	1985-1990	260	0.010	0.000	0.068	0.108	0.088	0.695	0.462	0.951
Private	Amazonia	1985-2018	8,064	0.015	0.000	-0.193	-0.132	-0.162	0.641	0.353	0.820
Private	Brazil	1985-2018	34,216	0.010	0.000	-0.143	-0.105	-0.124	0.663	0.126	0.808
Private	Caatinga	1985-2018	10,020	0.009	0.135	-0.032	0.004	-0.014	0.716	0.137	0.758
Private	Cerrado	1985-2018	9,672	0.015	0.016	-0.063	-0.007	-0.035	0.718	0.261	0.894
Private	Mata Atlantica	1985-2018	5,134	0.015	0.000	-0.251	-0.191	-0.221	0.743	0.113	0.794
Private	Pampa	1985-2018	404	0.046	0.025	-0.195	-0.013	-0.104	0.843	0.465	0.554
Private	Pantanal	1985-2018	262	0.019	0.000	-0.202	-0.126	-0.164	0.696	0.458	0.959
Private	Amazonia	1991-1995	8,062	0.006	0.310	-0.019	0.006	-0.007	0.640	0.357	0.813
Private	Brazil	1991-1995	34,216	0.004	0.027	-0.018	-0.001	-0.010	0.663	0.125	0.809
Private	Caatinga	1991-1995	10,024	0.004	0.766	-0.010	0.007	-0.001	0.715	0.140	0.759
Private	Cerrado	1991-1995	9,670	0.006	0.499	-0.016	0.008	-0.004	0.718	0.258	0.893

Treatment	Spatial scale	Temporal scale	*n*	SE	*p* value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	T-index
Private	Mata Atlantica	1991-1995	5,130	0.008	0.000	-0.071	-0.041	-0.056	0.744	0.161	0.792
Private	Pampa	1991-1995	404	0.030	0.169	-0.099	0.017	-0.041	0.843	0.416	0.548
Private	Pantanal	1991-1995	260	0.009	0.374	-0.024	0.009	-0.008	0.695	0.462	0.961
Private	Amazonia	1996-1999	8,060	0.006	0.000	-0.036	-0.012	-0.024	0.641	0.359	0.819
Private	Brazil	1996-1999	34,216	0.003	0.000	-0.028	-0.015	-0.022	0.663	0.126	0.810
Private	Caatinga	1996-1999	10,024	0.003	0.043	-0.013	0.000	-0.007	0.715	0.138	0.753
Private	Cerrado	1996-1999	9,670	0.006	0.005	-0.027	-0.005	-0.016	0.718	0.258	0.897
Private	Mata Atlantica	1996-1999	5,132	0.008	0.000	-0.080	-0.049	-0.065	0.743	0.141	0.792
Private	Pampa	1996-1999	404	0.028	0.000	-0.156	-0.046	-0.101	0.843	0.436	0.553
Private	Pantanal	1996-1999	262	0.007	0.126	-0.025	0.003	-0.011	0.695	0.458	0.960
Private	Amazonia	2000-2004	8,064	0.009	0.000	-0.084	-0.050	-0.067	0.641	0.354	0.818
Private	Brazil	2000-2004	34,214	0.004	0.000	-0.053	-0.036	-0.044	0.663	0.125	0.808
Private	Caatinga	2000-2004	10,022	0.003	0.001	-0.018	-0.005	-0.012	0.716	0.137	0.758
Private	Cerrado	2000-2004	9,672	0.007	0.179	-0.022	0.004	-0.009	0.718	0.259	0.894
Private	Mata Atlantica	2000-2004	5,132	0.006	0.000	-0.066	-0.042	-0.054	0.743	0.142	0.791
Private	Pampa	2000-2004	404	0.014	0.000	-0.100	-0.047	-0.074	0.843	0.431	0.527
Private	Amazonia	2005-2012	8,062	0.010	0.000	-0.127	-0.086	-0.106	0.641	0.353	0.816
Private	Brazil	2005-2012	34,218	0.005	0.000	-0.080	-0.059	-0.070	0.663	0.128	0.807
Private	Caatinga	2005-2012	10,022	0.005	0.510	-0.014	0.007	-0.004	0.716	0.135	0.760
Private	Cerrado	2005-2012	9,672	0.008	0.001	-0.041	-0.011	-0.026	0.719	0.261	0.896
Private	Mata Atlantica	2005-2012	5,134	0.005	0.000	-0.047	-0.027	-0.037	0.743	0.109	0.795
Private	Pampa	2005-2012	404	0.051	0.019	-0.218	-0.020	-0.119	0.843	0.455	0.549
Private	Pantanal	2005-2012	262	0.014	0.000	-0.092	-0.037	-0.065	0.696	0.466	0.966
Private	Amazonia	2013-2018	8,060	0.009	0.000	-0.096	-0.061	-0.078	0.641	0.355	0.865
Private	Brazil	2013-2018	34,214	0.004	0.000	-0.061	-0.045	-0.053	0.662	0.130	0.805
Private	Caatinga	2013-2018	10,022	0.004	0.928	-0.007	0.007	0.000	0.715	0.135	0.759
Private	Cerrado	2013-2018	9,672	0.006	0.000	-0.050	-0.026	-0.038	0.719	0.261	0.898
Private	Mata Atlantica	2013-2018	5,134	0.004	0.000	-0.030	-0.015	-0.023	0.742	0.113	0.787
Private	Pampa	2013-2018	404	0.012	0.081	-0.043	0.003	-0.020	0.843	0.460	0.555
Private	Pantanal	2013-2018	262	0.028	0.000	-0.265	-0.156	-0.210	0.696	0.450	0.950
Protected	Amazonia	1985-1990	108	0.006	0.737	-0.015	0.010	-0.002	0.896	0.611	0.556
Protected	Brazil	1985-1990	748	0.011	0.000	-0.076	-0.034	-0.055	0.724	0.283	0.914
Protected	Caatinga	1985-1990	52	0.071	0.005	-0.338	-0.059	-0.198	0.855	0.615	0.762
Protected	Cerrado	1985-1990	118	0.018	0.040	-0.073	-0.002	-0.037	0.899	0.644	0.881
Protected	Mata Atlantica	1985-1990	328	0.031	0.000	-0.186	-0.065	-0.126	0.709	0.500	0.943
Protected	Amazonia	1985-2018	108	0.038	0.003	-0.188	-0.040	-0.114	0.896	0.611	0.703
Protected	Brazil	1985-2018	740	0.025	0.000	-0.276	-0.177	-0.227	0.728	0.297	0.913

Treatment	Spatial scale	Temporal scale	*n*	SE	*p* value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	T-index
Protected	Caatinga	1985-2018	52	0.031	0.000	-0.167	-0.047	-0.107	0.856	0.538	0.772
Protected	Cerrado	1985-2018	116	0.042	0.000	-0.323	-0.157	-0.240	0.900	0.638	0.909
Protected	Mata Atlantica	1985-2018	326	0.042	0.000	-0.331	-0.165	-0.248	0.709	0.503	0.949
Protected	Amazonia	1991-1995	108	0.012	0.061	-0.044	0.001	-0.022	0.896	0.611	0.724
Protected	Brazil	1991-1995	742	0.012	0.000	-0.097	-0.048	-0.072	0.726	0.280	0.923
Protected	Caatinga	1991-1995	52	0.029	0.275	-0.025	0.087	0.031	0.855	0.577	0.763
Protected	Cerrado	1991-1995	118	0.018	0.017	-0.076	-0.008	-0.042	0.899	0.644	0.827
Protected	Mata Atlantica	1991-1995	328	0.018	0.000	-0.119	-0.049	-0.084	0.712	0.494	0.945
Protected	Amazonia	1996-1999	108	0.015	0.106	-0.054	0.005	-0.024	0.896	0.611	0.696
Protected	Brazil	1996-1999	740	0.013	0.000	-0.077	-0.024	-0.051	0.728	0.292	0.923
Protected	Caatinga	1996-1999	52	0.022	0.012	-0.097	-0.012	-0.054	0.856	0.538	0.782
Protected	Cerrado	1996-1999	118	0.018	0.000	-0.113	-0.044	-0.079	0.899	0.627	0.937
Protected	Mata Atlantica	1996-1999	328	0.020	0.000	-0.143	-0.063	-0.103	0.709	0.500	0.948
Protected	Amazonia	2000-2004	108	0.017	0.002	-0.087	-0.019	-0.053	0.896	0.611	0.591
Protected	Brazil	2000-2004	740	0.013	0.000	-0.079	-0.029	-0.054	0.728	0.297	0.910
Protected	Caatinga	2000-2004	52	0.018	0.000	-0.110	-0.041	-0.075	0.856	0.500	0.826
Protected	Cerrado	2000-2004	116	0.014	0.000	-0.085	-0.028	-0.056	0.900	0.655	0.827
Protected	Mata Atlantica	2000-2004	328	0.013	0.000	-0.079	-0.029	-0.054	0.709	0.494	0.959
Protected	Amazonia	2005-2012	108	0.021	0.012	-0.096	-0.012	-0.054	0.896	0.611	0.599
Protected	Brazil	2005-2012	738	0.012	0.000	-0.098	-0.051	-0.074	0.729	0.309	0.929
Protected	Caatinga	2005-2012	52	0.018	0.001	-0.095	-0.024	-0.059	0.856	0.500	0.775
Protected	Cerrado	2005-2012	116	0.019	0.004	-0.092	-0.017	-0.054	0.900	0.638	0.870
Protected	Mata Atlantica	2005-2012	326	0.008	0.000	-0.057	-0.024	-0.041	0.710	0.509	0.939
Protected	Amazonia	2013-2018	110	0.015	0.000	-0.083	-0.024	-0.054	0.896	0.618	0.588
Protected	Brazil	2013-2018	736	0.013	0.000	-0.100	-0.051	-0.075	0.730	0.318	0.916
Protected	Caatinga	2013-2018	52	0.021	0.000	-0.115	-0.032	-0.074	0.856	0.462	NA
Protected	Cerrado	2013-2018	112	0.023	0.028	-0.094	-0.005	-0.049	0.900	0.625	0.932
Protected	Mata Atlantica	2013-2018	326	0.007	0.000	-0.044	-0.016	-0.030	0.710	0.521	0.940
Quilombola	Amazonia	1985-1990	230	0.013	0.000	-0.086	-0.036	-0.061	0.755	0.687	0.708
Quilombola	Brazil	1985-1990	636	0.016	0.056	-0.061	0.001	-0.030	0.688	0.321	0.950
Quilombola	Caatinga	1985-1990	98	0.040	0.598	-0.099	0.057	-0.021	0.778	0.612	0.853
Quilombola	Cerrado	1985-1990	82	0.022	0.000	0.035	0.120	0.077	0.834	0.512	0.910
Quilombola	Mata Atlantica	1985-1990	148	0.048	0.198	-0.155	0.032	-0.061	0.730	0.527	0.922
Quilombola	Amazonia	1985-2018	230	0.040	0.000	-0.264	-0.108	-0.186	0.755	0.696	0.744
Quilombola	Brazil	1985-2018	634	0.026	0.000	-0.189	-0.086	-0.137	0.695	0.322	0.944
Quilombola	Caatinga	1985-2018	98	0.050	0.030	-0.206	-0.010	-0.108	0.778	0.449	0.825
Quilombola	Cerrado	1985-2018	82	0.037	0.859	-0.065	0.078	0.007	0.835	0.537	0.815

Treatment	Spatial scale	Temporal scale	*n*	SE	*p* value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	T-index
Quilombola	Mata Atlantica	1985-2018	142	0.047	0.005	-0.221	-0.039	-0.130	0.732	0.493	0.901
Quilombola	Amazonia	1991-1995	230	0.026	0.000	-0.144	-0.041	-0.092	0.755	0.678	0.705
Quilombola	Brazil	1991-1995	630	0.015	0.007	-0.072	-0.011	-0.041	0.688	0.330	0.942
Quilombola	Caatinga	1991-1995	98	0.028	0.762	-0.064	0.047	-0.009	0.778	0.592	0.892
Quilombola	Cerrado	1991-1995	82	0.012	0.270	-0.011	0.038	0.014	0.834	0.512	0.908
Quilombola	Mata Atlantica	1991-1995	146	0.041	0.011	-0.183	-0.024	-0.104	0.730	0.521	0.943
Quilombola	Amazonia	1996-1999	230	0.024	0.001	-0.127	-0.033	-0.080	0.755	0.687	0.713
Quilombola	Brazil	1996-1999	632	0.012	0.058	-0.047	0.001	-0.023	0.688	0.335	0.946
Quilombola	Caatinga	1996-1999	98	0.024	0.602	-0.059	0.034	-0.012	0.778	0.490	0.837
Quilombola	Cerrado	1996-1999	82	0.012	0.652	-0.019	0.030	0.006	0.834	0.512	0.846
Quilombola	Mata Atlantica	1996-1999	144	0.016	0.676	-0.037	0.024	-0.007	0.731	0.514	0.901
Quilombola	Amazonia	2000-2004	230	0.020	0.000	-0.112	-0.033	-0.073	0.755	0.687	0.680
Quilombola	Brazil	2000-2004	632	0.012	0.024	-0.051	-0.004	-0.027	0.692	0.323	0.944
Quilombola	Caatinga	2000-2004	96	0.026	0.658	-0.063	0.039	-0.012	0.778	0.563	0.905
Quilombola	Cerrado	2000-2004	82	0.015	0.004	-0.071	-0.014	-0.043	0.835	0.512	0.821
Quilombola	Mata Atlantica	2000-2004	144	0.017	0.654	-0.040	0.025	-0.007	0.732	0.486	0.942
Quilombola	Amazonia	2005-2012	230	0.025	0.000	-0.147	-0.050	-0.098	0.755	0.687	0.710
Quilombola	Brazil	2005-2012	632	0.012	0.000	-0.082	-0.035	-0.059	0.695	0.313	0.950
Quilombola	Caatinga	2005-2012	96	0.027	0.067	-0.104	0.003	-0.050	0.778	0.542	0.899
Quilombola	Cerrado	2005-2012	82	0.014	0.494	-0.018	0.037	0.010	0.835	0.585	0.887
Quilombola	Mata Atlantica	2005-2012	138	0.021	0.041	-0.085	-0.002	-0.043	0.736	0.493	0.935
Quilombola	Amazonia	2013-2018	230	0.029	0.002	-0.146	-0.033	-0.090	0.756	0.687	0.727
Quilombola	Brazil	2013-2018	632	0.011	0.002	-0.055	-0.013	-0.034	0.698	0.323	0.938
Quilombola	Caatinga	2013-2018	96	0.020	0.361	-0.056	0.020	-0.018	0.778	0.500	NA
Quilombola	Cerrado	2013-2018	82	0.012	0.168	-0.040	0.007	-0.016	0.835	0.585	0.840
Quilombola	Mata Atlantica	2013-2018	134	0.012	0.033	-0.049	-0.002	-0.026	0.737	0.582	0.938
Sustainable use	Amazonia	1985-1990	246	0.005	0.011	-0.023	-0.003	-0.013	0.798	0.618	0.800
Sustainable use	Brazil	1985-1990	958	0.009	0.002	-0.046	-0.010	-0.028	0.673	0.347	0.910
Sustainable use	Caatinga	1985-1990	78	0.049	0.001	-0.256	-0.062	-0.159	0.818	0.308	0.911
Sustainable use	Cerrado	1985-1990	88	0.044	0.310	-0.130	0.041	-0.044	0.868	0.545	0.937
Sustainable use	Mata Atlantica	1985-1990	406	0.033	0.000	-0.289	-0.158	-0.224	0.711	0.424	0.967
Sustainable use	Amazonia	1985-2018	246	0.033	0.000	-0.237	-0.107	-0.172	0.798	0.618	0.782
Sustainable use	Brazil	1985-2018	960	0.038	0.000	-0.271	-0.124	-0.198	0.673	0.331	0.937
Sustainable use	Caatinga	1985-2018	80	0.063	0.000	-0.409	-0.162	-0.285	0.818	0.100	0.912
Sustainable use	Cerrado	1985-2018	88	0.090	0.132	-0.311	0.041	-0.135	0.868	0.523	0.912
Sustainable use	Mata Atlantica	1985-2018	406	0.036	0.000	-0.399	-0.260	-0.329	0.710	0.414	0.970
Sustainable use	Amazonia	1991-1995	246	0.008	0.003	-0.038	-0.008	-0.023	0.798	0.618	0.818

Treatment	Spatial scale	Temporal scale	*n*	SE	*p* value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	T-index
Sustainable use	Brazil	1991-1995	958	0.010	0.001	-0.054	-0.013	-0.034	0.673	0.336	0.919
Sustainable use	Caatinga	1991-1995	78	0.041	0.004	-0.196	-0.037	-0.117	0.818	0.333	0.939
Sustainable use	Cerrado	1991-1995	88	0.018	0.135	-0.062	0.008	-0.027	0.868	0.545	0.954
Sustainable use	Mata Atlantica	1991-1995	406	0.031	0.000	-0.202	-0.081	-0.141	0.711	0.424	0.958
Sustainable use	Amazonia	1996-1999	246	0.007	0.001	-0.038	-0.009	-0.024	0.798	0.618	0.754
Sustainable use	Brazil	1996-1999	960	0.013	0.001	-0.066	-0.016	-0.041	0.673	0.329	0.913
Sustainable use	Caatinga	1996-1999	78	0.018	0.000	-0.118	-0.046	-0.082	0.818	0.359	0.914
Sustainable use	Cerrado	1996-1999	90	0.032	0.273	-0.097	0.027	-0.035	0.868	0.533	0.911
Sustainable use	Mata Atlantica	1996-1999	406	0.013	0.000	-0.113	-0.062	-0.088	0.711	0.414	0.965
Sustainable use	Amazonia	2000-2004	246	0.012	0.000	-0.076	-0.029	-0.053	0.798	0.618	0.752
Sustainable use	Brazil	2000-2004	960	0.013	0.000	-0.085	-0.034	-0.060	0.673	0.331	0.916
Sustainable use	Caatinga	2000-2004	80	0.024	0.001	-0.127	-0.032	-0.079	0.818	0.100	0.948
Sustainable use	Cerrado	2000-2004	90	0.045	0.637	-0.109	0.067	-0.021	0.868	0.533	0.874
Sustainable use	Mata Atlantica	2000-2004	406	0.014	0.000	-0.116	-0.063	-0.090	0.710	0.414	0.961
Sustainable use	Amazonia	2005-2012	246	0.019	0.000	-0.122	-0.046	-0.084	0.798	0.618	0.832
Sustainable use	Brazil	2005-2012	958	0.023	0.000	-0.155	-0.064	-0.110	0.673	0.336	0.926
Sustainable use	Caatinga	2005-2012	78	0.058	0.001	-0.313	-0.084	-0.199	0.818	0.333	0.878
Sustainable use	Cerrado	2005-2012	86	0.050	0.221	-0.160	0.037	-0.062	0.869	0.535	0.923
Sustainable use	Mata Atlantica	2005-2012	404	0.017	0.000	-0.127	-0.061	-0.094	0.711	0.441	0.972
Sustainable use	Amazonia	2013-2018	246	0.020	0.000	-0.129	-0.051	-0.090	0.798	0.626	0.805
Sustainable use	Brazil	2013-2018	956	0.021	0.000	-0.136	-0.055	-0.095	0.673	0.379	0.913
Sustainable use	Caatinga	2013-2018	78	0.029	0.611	-0.072	0.042	-0.015	0.818	0.256	0.924
Sustainable use	Cerrado	2013-2018	86	0.034	0.525	-0.088	0.045	-0.021	0.870	0.535	0.967
Sustainable use	Mata Atlantica	2013-2018	404	0.015	0.000	-0.116	-0.059	-0.087	0.711	0.460	0.961
Robustness check: protected areas and sustainable-use areas filtered by known year of creation											
Protected	Brazil	1985-1990	219	0.019	0.005	-0.089	-0.016	-0.052	0.724	0.283	0.914
Protected	Mata Atlantica	1985-1990	80	0.031	0.004	-0.151	-0.029	-0.090	0.709	0.500	0.943
Protected	Amazonia	1985-2018	58	0.061	0.001	-0.318	-0.077	-0.198	0.896	0.611	0.703
Protected	Brazil	1985-2018	351	0.027	0.000	-0.210	-0.106	-0.158	0.728	0.297	0.913
Protected	Cerrado	1985-2018	48	0.051	0.000	-0.306	-0.105	-0.205	0.900	0.638	0.909
Protected	Mata Atlantica	1985-2018	210	0.056	0.000	-0.378	-0.157	-0.267	0.709	0.503	0.949
Protected	Amazonia	1991-1995	56	0.018	0.141	-0.060	0.009	-0.026	0.896	0.611	0.724
Protected	Brazil	1991-1995	357	0.017	0.002	-0.085	-0.020	-0.052	0.726	0.280	0.923
Protected	Cerrado	1991-1995	47	0.040	0.295	-0.119	0.036	-0.041	0.899	0.644	0.827
Protected	Mata Atlantica	1991-1995	201	0.045	0.000	-0.262	-0.087	-0.174	0.712	0.494	0.945
Protected	Amazonia	1996-1999	69	0.022	0.149	-0.076	0.012	-0.032	0.896	0.611	0.696
Protected	Brazil	1996-1999	408	0.011	0.002	-0.057	-0.013	-0.035	0.728	0.292	0.923

Treatment	Spatial scale	Temporal scale	*n*	SE	*p* value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	T-index
Protected	Cerrado	1996-1999	46	0.021	0.000	-0.135	-0.050	-0.092	0.899	0.627	0.937
Protected	Mata Atlantica	1996-1999	246	0.016	0.000	-0.107	-0.044	-0.076	0.709	0.500	0.948
Protected	Amazonia	2000-2004	90	0.032	0.009	-0.144	-0.020	-0.082	0.896	0.611	0.591
Protected	Brazil	2000-2004	514	0.013	0.000	-0.087	-0.037	-0.062	0.728	0.297	0.910
Protected	Cerrado	2000-2004	74	0.016	0.000	-0.110	-0.049	-0.079	0.900	0.655	0.827
Protected	Mata Atlantica	2000-2004	268	0.013	0.000	-0.087	-0.036	-0.061	0.709	0.494	0.959
Protected	Amazonia	2005-2012	101	0.026	0.003	-0.127	-0.027	-0.077	0.896	0.611	0.599
Protected	Brazil	2005-2012	704	0.011	0.000	-0.097	-0.052	-0.074	0.729	0.309	0.929
Protected	Cerrado	2005-2012	110	0.013	0.000	-0.087	-0.036	-0.062	0.900	0.638	0.870
Protected	Mata Atlantica	2005-2012	401	0.012	0.000	-0.098	-0.051	-0.075	0.710	0.509	0.939
Protected	Amazonia	2013-2018	121	0.014	0.002	-0.070	-0.016	-0.043	0.896	0.618	0.588
Protected	Brazil	2013-2018	989	0.010	0.000	-0.099	-0.058	-0.078	0.730	0.318	0.916
Protected	Caatinga	2013-2018	59	0.021	0.000	-0.116	-0.034	-0.075	0.856	0.462	NA
Protected	Cerrado	2013-2018	142	0.018	0.002	-0.090	-0.021	-0.055	0.900	0.625	0.932
Protected	Mata Atlantica	2013-2018	376	0.009	0.000	-0.054	-0.020	-0.037	0.710	0.521	0.940
Sustainable use	Brazil	1985-1990	58	0.012	0.000	-0.101	-0.053	-0.077	0.673	0.347	0.910
Sustainable use	Brazil	1985-2018	118	0.018	0.000	-0.239	-0.168	-0.204	0.673	0.331	0.937
Sustainable use	Mata Atlantica	1985-2018	56	0.047	0.000	-0.530	-0.347	-0.438	0.710	0.414	0.970
Sustainable use	Brazil	1991-1995	118	0.032	0.010	-0.144	-0.019	-0.082	0.673	0.336	0.919
Sustainable use	Mata Atlantica	1991-1995	57	0.026	0.000	-0.220	-0.117	-0.169	0.711	0.424	0.958
Sustainable use	Amazonia	1996-1999	110	0.015	0.830	-0.026	0.033	0.003	0.798	0.618	0.754
Sustainable use	Brazil	1996-1999	207	0.025	0.032	-0.103	-0.005	-0.054	0.673	0.329	0.913
Sustainable use	Mata Atlantica	1996-1999	69	0.026	0.000	-0.171	-0.070	-0.121	0.711	0.414	0.965
Sustainable use	Amazonia	2000-2004	145	0.023	0.009	-0.105	-0.015	-0.060	0.798	0.618	0.752
Sustainable use	Brazil	2000-2004	330	0.015	0.000	-0.084	-0.024	-0.054	0.673	0.331	0.916
Sustainable use	Mata Atlantica	2000-2004	101	0.022	0.000	-0.208	-0.122	-0.165	0.710	0.414	0.961
Sustainable use	Amazonia	2005-2012	292	0.021	0.000	-0.142	-0.060	-0.101	0.798	0.618	0.832
Sustainable use	Brazil	2005-2012	566	0.022	0.000	-0.164	-0.079	-0.122	0.673	0.336	0.926
Sustainable use	Cerrado	2005-2012	57	0.034	0.551	-0.086	0.046	-0.020	0.869	0.535	0.923
Sustainable use	Mata Atlantica	2005-2012	164	0.016	0.000	-0.132	-0.071	-0.102	0.711	0.441	0.972
Sustainable use	Amazonia	2013-2018	349	0.017	0.000	-0.128	-0.060	-0.094	0.798	0.626	0.805
Sustainable use	Brazil	2013-2018	1366	0.017	0.000	-0.132	-0.066	-0.099	0.673	0.379	0.913
Sustainable use	Caatinga	2013-2018	79	0.029	0.667	-0.068	0.044	-0.012	0.818	0.256	0.924
Sustainable use	Cerrado	2013-2018	99	0.037	0.178	-0.123	0.023	-0.050	0.870	0.535	0.967
Sustainable use	Mata Atlantica	2013-2018	482	0.015	0.000	-0.119	-0.061	-0.090	0.711	0.460	0.961

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. Scores on Tipton's index of generalizability are reported (T-index), with values closer to 1 indicating high levels of generalizability, and scores ≤0.5 preventing generalizability between matched samples and entire population of land parcels at each respective scale and comparison. 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, for all tenure regimes except undesignated/untitled and private in the Pampas and Pantanal biomes, and for sustainable use areas in Caatinga during the robustness check filtering areas with known dates of creation.

Treatment	Spatial scale	Temporal scale	*n*	SE	*p* value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	T-index
Communal	Amazonia	1985-1990	1,462	0.009	0.000	-0.053	-0.017	-0.035	0.730	0.599	0.714
Communal	Brazil	1985-1990	1,522	0.000	0.000	3.117	3.117	3.117	0.882	0.645	0.442
Communal	Amazonia	1985-2018	1,462	0.040	0.008	-0.186	-0.028	-0.107	0.732	0.595	0.738
Communal	Brazil	1985-2018	1,522	0.050	0.004	-0.240	-0.045	-0.142	0.882	0.645	0.448
Communal	Amazonia	1991-1995	1,462	0.015	0.002	-0.074	-0.016	-0.045	0.731	0.599	0.718
Communal	Brazil	1991-1995	1,522	0.000	0.000	-1.039	-1.039	-1.039	0.882	0.644	0.460
Communal	Amazonia	1996-1999	1,462	0.014	0.000	-0.091	-0.038	-0.064	0.732	0.596	0.724
Communal	Brazil	1996-1999	1,522	0.016	0.138	-0.056	0.008	-0.024	0.882	0.644	0.458
Communal	Amazonia	2000-2004	1,462	0.026	0.012	-0.118	-0.015	-0.066	0.732	0.595	0.713
Communal	Brazil	2000-2004	1,522	0.017	0.000	-0.096	-0.030	-0.063	0.882	0.643	0.467
Communal	Amazonia	2005-2012	1,462	0.021	0.075	-0.079	0.004	-0.038	0.732	0.596	0.723
Communal	Brazil	2005-2012	1,522	0.031	0.075	-0.117	0.006	-0.056	0.882	0.645	0.467
Communal	Amazonia	2013-2018	1,462	0.023	0.012	-0.104	-0.013	-0.058	0.732	0.598	0.718
Communal	Brazil	2013-2018	1,522	0.027	0.014	-0.120	-0.014	-0.067	0.882	0.647	0.461
Indigenous	Amazonia	1985-1990	402	0.009	0.001	-0.046	-0.011	-0.028	0.937	0.587	0.401
Indigenous	Brazil	1985-1990	906	0.010	0.000	-0.059	-0.020	-0.040	0.925	0.329	0.453
Indigenous	Cerrado	1985-1990	100	0.028	0.020	-0.120	-0.010	-0.065	0.950	0.760	0.503
Indigenous	Mata Atlantica	1985-1990	256	0.018	0.170	-0.010	0.059	0.024	0.966	0.234	0.396
Indigenous	Amazonia	1985-2018	402	0.030	0.001	-0.162	-0.042	-0.102	0.937	0.592	0.413
Indigenous	Brazil	1985-2018	906	0.025	0.000	-0.216	-0.119	-0.168	0.923	0.353	0.440
Indigenous	Caatinga	1985-2018	54	0.049	0.667	-0.118	0.075	-0.021	0.992	0.667	0.527
Indigenous	Cerrado	1985-2018	100	0.062	0.000	-0.403	-0.161	-0.282	0.950	0.760	0.475
Indigenous	Mata Atlantica	1985-2018	256	0.028	0.243	-0.022	0.089	0.033	0.959	0.273	0.386
Indigenous	Amazonia	1991-1995	402	0.014	0.001	-0.071	-0.018	-0.044	0.937	0.587	0.419
Indigenous	Brazil	1991-1995	906	0.013	0.000	-0.091	-0.041	-0.066	0.925	0.327	0.471
Indigenous	Cerrado	1991-1995	100	0.020	0.046	-0.078	-0.001	-0.039	0.950	0.760	0.613
Indigenous	Mata Atlantica	1991-1995	256	0.015	0.739	-0.024	0.034	0.005	0.966	0.227	0.442
Indigenous	Amazonia	1996-1999	402	0.009	0.004	-0.042	-0.008	-0.025	0.940	0.587	0.425
Indigenous	Brazil	1996-1999	906	0.009	0.000	-0.072	-0.036	-0.054	0.923	0.349	0.440
Indigenous	Cerrado	1996-1999	100	0.019	0.000	-0.119	-0.044	-0.082	0.950	0.760	0.534

Treatment	Spatial scale	Temporal scale	*n*	SE	*p* value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	T-index
Indigenous	Mata Atlantica	1996-1999	256	0.009	0.353	-0.009	0.025	0.008	0.959	0.266	0.454
Indigenous	Amazonia	2000-2004	402	0.009	0.000	-0.050	-0.017	-0.033	0.940	0.592	0.812
Indigenous	Brazil	2000-2004	906	0.009	0.000	-0.075	-0.039	-0.057	0.923	0.349	0.416
Indigenous	Cerrado	2000-2004	100	0.041	0.000	-0.243	-0.081	-0.162	0.950	0.760	0.404
Indigenous	Mata Atlantica	2000-2004	256	0.008	0.608	-0.012	0.020	0.004	0.959	0.273	0.414
Indigenous	Amazonia	2005-2012	402	0.009	0.071	-0.035	0.001	-0.017	0.937	0.587	0.456
Indigenous	Brazil	2005-2012	906	0.012	0.000	-0.070	-0.022	-0.046	0.923	0.355	0.440
Indigenous	Cerrado	2005-2012	100	0.041	0.003	-0.202	-0.042	-0.122	0.950	0.760	0.451
Indigenous	Mata Atlantica	2005-2012	256	0.013	0.087	-0.003	0.047	0.022	0.959	0.297	0.446
Indigenous	Amazonia	2013-2018	402	0.010	0.014	-0.044	-0.005	-0.025	0.937	0.587	0.409
Indigenous	Brazil	2013-2018	906	0.012	0.000	-0.068	-0.023	-0.046	0.923	0.360	0.446
Indigenous	Cerrado	2013-2018	100	0.025	0.004	-0.121	-0.023	-0.072	0.951	0.740	0.382
Indigenous	Mata Atlantica	2013-2018	256	0.006	0.444	-0.007	0.015	0.004	0.959	0.305	0.430
Protected	Amazonia	1985-1990	72	0.006	0.727	-0.010	0.015	0.002	0.969	0.611	0.663
Protected	Brazil	1985-1990	904	0.008	0.000	-0.053	-0.022	-0.038	0.843	0.237	0.655
Protected	Caatinga	1985-1990	60	0.052	0.030	-0.216	-0.011	-0.114	0.962	0.200	0.579
Protected	Cerrado	1985-1990	172	0.021	0.044	-0.084	-0.001	-0.043	0.901	0.570	0.649
Protected	Mata Atlantica	1985-1990	516	0.010	0.000	-0.062	-0.023	-0.043	0.875	0.283	0.615
Protected	Amazonia	1985-2018	70	0.025	0.000	-0.198	-0.100	-0.149	0.971	0.571	0.374
Protected	Brazil	1985-2018	908	0.018	0.000	-0.146	-0.076	-0.111	0.841	0.244	0.642
Protected	Cerrado	1985-2018	172	0.035	0.000	-0.246	-0.108	-0.177	0.910	0.558	0.640
Protected	Mata Atlantica	1985-2018	516	0.018	0.000	-0.104	-0.034	-0.069	0.872	0.291	0.566
Protected	Amazonia	1991-1995	72	0.006	0.437	-0.007	0.016	0.005	0.969	0.611	0.396
Protected	Brazil	1991-1995	906	0.006	0.000	-0.037	-0.012	-0.025	0.843	0.241	0.638
Protected	Caatinga	1991-1995	58	0.021	0.319	-0.063	0.021	-0.021	0.962	0.414	0.353
Protected	Cerrado	1991-1995	172	0.022	0.305	-0.066	0.021	-0.023	0.901	0.570	0.724
Protected	Mata Atlantica	1991-1995	514	0.006	0.000	-0.044	-0.019	-0.031	0.875	0.304	0.635
Protected	Amazonia	1996-1999	70	0.015	0.567	-0.021	0.039	0.009	0.971	0.600	0.345
Protected	Brazil	1996-1999	904	0.008	0.003	-0.037	-0.008	-0.023	0.843	0.281	0.681
Protected	Caatinga	1996-1999	58	0.014	0.000	-0.099	-0.043	-0.071	0.962	0.448	0.444
Protected	Cerrado	1996-1999	172	0.021	0.068	-0.081	0.003	-0.039	0.901	0.558	0.697
Protected	Mata Atlantica	1996-1999	514	0.004	0.001	-0.024	-0.007	-0.016	0.872	0.300	0.621
Protected	Amazonia	2000-2004	70	0.013	0.000	-0.069	-0.019	-0.044	0.971	0.600	0.273
Protected	Brazil	2000-2004	906	0.008	0.000	-0.057	-0.028	-0.042	0.841	0.280	0.649
Protected	Caatinga	2000-2004	58	0.021	0.061	-0.082	0.002	-0.040	0.962	0.448	0.472
Protected	Cerrado	2000-2004	172	0.011	0.015	-0.050	-0.005	-0.028	0.901	0.547	0.685
Protected	Mata Atlantica	2000-2004	516	0.006	0.000	-0.035	-0.012	-0.023	0.872	0.298	0.619

Treatment	Spatial scale	Temporal scale	*n*	SE	*p* value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	T-index
Protected	Amazonia	2005-2012	70	0.017	0.000	-0.102	-0.034	-0.068	0.971	0.571	0.460
Protected	Brazil	2005-2012	906	0.008	0.000	-0.053	-0.022	-0.037	0.843	0.243	0.645
Protected	Caatinga	2005-2012	60	0.021	0.026	-0.089	-0.006	-0.047	0.962	0.200	0.555
Protected	Cerrado	2005-2012	172	0.027	0.015	-0.118	-0.013	-0.065	0.910	0.535	0.736
Protected	Mata Atlantica	2005-2012	514	0.006	0.008	-0.029	-0.004	-0.016	0.872	0.370	0.637
Protected	Amazonia	2013-2018	70	0.019	0.000	-0.131	-0.055	-0.093	0.971	0.600	0.571
Protected	Brazil	2013-2018	904	0.009	0.000	-0.056	-0.019	-0.037	0.843	0.288	0.648
Protected	Caatinga	2013-2018	60	0.018	0.004	-0.086	-0.017	-0.051	0.962	0.167	0.557
Protected	Cerrado	2013-2018	172	0.021	0.054	-0.081	0.001	-0.040	0.910	0.558	0.742
Protected	Mata Atlantica	2013-2018	510	0.005	0.001	-0.028	-0.007	-0.017	0.872	0.329	0.638
Quilombola	Amazonia	1985-1990	226	0.014	0.000	-0.084	-0.030	-0.057	0.910	0.602	0.382
Quilombola	Brazil	1985-1990	702	0.012	0.381	-0.034	0.013	-0.010	0.867	0.148	0.645
Quilombola	Caatinga	1985-1990	124	0.028	0.832	-0.049	0.061	0.006	0.974	0.323	0.320
Quilombola	Cerrado	1985-1990	92	0.053	0.521	-0.070	0.138	0.034	0.936	0.543	0.460
Quilombola	Mata Atlantica	1985-1990	218	0.024	0.014	-0.107	-0.012	-0.059	0.891	0.266	0.609
Quilombola	Amazonia	1985-2018	226	0.027	0.000	-0.164	-0.058	-0.111	0.910	0.611	0.606
Quilombola	Brazil	1985-2018	702	0.023	0.000	-0.137	-0.047	-0.092	0.867	0.165	0.608
Quilombola	Caatinga	1985-2018	124	0.037	0.196	-0.025	0.120	0.048	0.974	0.323	0.462
Quilombola	Cerrado	1985-2018	92	0.048	0.482	-0.061	0.129	0.034	0.936	0.543	0.587
Quilombola	Mata Atlantica	1985-2018	218	0.043	0.287	-0.129	0.038	-0.045	0.890	0.303	0.656
Quilombola	Amazonia	1991-1995	226	0.015	0.119	-0.053	0.006	-0.024	0.910	0.611	0.439
Quilombola	Brazil	1991-1995	702	0.010	0.641	-0.025	0.015	-0.005	0.867	0.151	0.595
Quilombola	Caatinga	1991-1995	124	0.024	0.056	-0.001	0.095	0.047	0.974	0.306	0.391
Quilombola	Cerrado	1991-1995	92	0.023	0.508	-0.059	0.029	-0.015	0.936	0.543	0.583
Quilombola	Mata Atlantica	1991-1995	218	0.018	0.368	-0.052	0.019	-0.016	0.891	0.275	0.663
Quilombola	Amazonia	1996-1999	226	0.012	0.000	-0.066	-0.020	-0.043	0.910	0.619	0.543
Quilombola	Brazil	1996-1999	702	0.009	0.038	-0.038	-0.001	-0.020	0.867	0.154	0.630
Quilombola	Caatinga	1996-1999	124	0.015	0.171	-0.009	0.051	0.021	0.974	0.323	0.369
Quilombola	Cerrado	1996-1999	92	0.029	0.975	-0.059	0.057	-0.001	0.936	0.543	0.552
Quilombola	Mata Atlantica	1996-1999	218	0.009	0.191	-0.030	0.006	-0.012	0.891	0.275	0.630
Quilombola	Amazonia	2000-2004	226	0.014	0.001	-0.072	-0.019	-0.045	0.910	0.611	0.534
Quilombola	Brazil	2000-2004	702	0.013	0.009	-0.058	-0.008	-0.033	0.867	0.157	0.602
Quilombola	Caatinga	2000-2004	124	0.017	0.354	-0.018	0.049	0.016	0.974	0.323	0.375
Quilombola	Cerrado	2000-2004	92	0.123	0.688	-0.292	0.192	-0.050	0.936	0.543	0.591
Quilombola	Mata Atlantica	2000-2004	218	0.013	0.735	-0.030	0.021	-0.004	0.891	0.303	0.727
Quilombola	Amazonia	2005-2012	226	0.015	0.000	-0.086	-0.026	-0.056	0.910	0.628	0.449
Quilombola	Brazil	2005-2012	704	0.013	0.001	-0.070	-0.017	-0.044	0.867	0.168	0.609

Treatment	Spatial scale	Temporal scale	*n*	SE	*p* value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	T-index
Quilombola	Caatinga	2005-2012	124	0.017	0.629	-0.025	0.041	0.008	0.974	0.323	0.346
Quilombola	Cerrado	2005-2012	92	0.025	0.667	-0.060	0.038	-0.011	0.936	0.543	0.478
Quilombola	Mata Atlantica	2005-2012	218	0.009	0.145	-0.030	0.004	-0.013	0.890	0.294	0.654
Quilombola	Amazonia	2013-2018	226	0.020	0.006	-0.095	-0.016	-0.056	0.910	0.628	0.400
Quilombola	Brazil	2013-2018	704	0.014	0.001	-0.071	-0.018	-0.045	0.867	0.170	0.600
Quilombola	Caatinga	2013-2018	124	0.008	0.026	0.002	0.032	0.017	0.974	0.306	0.344
Quilombola	Cerrado	2013-2018	92	0.022	0.793	-0.037	0.049	0.006	0.936	0.543	0.650
Quilombola	Mata Atlantica	2013-2018	218	0.007	0.610	-0.018	0.011	-0.004	0.890	0.303	0.637
Sustainable use	Amazonia	1985-1990	178	0.011	0.010	-0.052	-0.007	-0.030	0.963	0.607	0.516
Sustainable use	Brazil	1985-1990	1,234	0.010	0.002	-0.051	-0.012	-0.031	0.716	0.245	0.878
Sustainable use	Caatinga	1985-1990	100	0.050	0.083	-0.184	0.011	-0.086	0.895	0.260	0.751
Sustainable use	Cerrado	1985-1990	156	0.034	0.280	-0.104	0.030	-0.037	0.849	0.500	0.916
Sustainable use	Mata Atlantica	1985-1990	756	0.021	0.000	-0.139	-0.056	-0.098	0.732	0.275	0.908
Sustainable use	Amazonia	1985-2018	178	0.037	0.000	-0.204	-0.058	-0.131	0.963	0.607	0.481
Sustainable use	Brazil	1985-2018	1,232	0.026	0.000	-0.200	-0.097	-0.149	0.716	0.237	0.871
Sustainable use	Cerrado	1985-2018	156	0.044	0.008	-0.203	-0.031	-0.117	0.850	0.487	0.882
Sustainable use	Mata Atlantica	1985-2018	754	0.033	0.000	-0.198	-0.067	-0.133	0.732	0.284	0.918
Sustainable use	Amazonia	1991-1995	178	0.017	0.002	-0.088	-0.020	-0.054	0.963	0.618	0.475
Sustainable use	Brazil	1991-1995	1,234	0.010	0.000	-0.058	-0.020	-0.039	0.716	0.238	0.857
Sustainable use	Caatinga	1991-1995	98	0.022	0.007	-0.103	-0.016	-0.060	0.895	0.490	0.716
Sustainable use	Cerrado	1991-1995	156	0.016	0.001	-0.082	-0.020	-0.051	0.850	0.500	0.874
Sustainable use	Mata Atlantica	1991-1995	756	0.010	0.000	-0.054	-0.015	-0.035	0.732	0.286	0.906
Sustainable use	Amazonia	1996-1999	178	0.016	0.000	-0.091	-0.029	-0.060	0.963	0.607	0.538
Sustainable use	Brazil	1996-1999	1,232	0.008	0.000	-0.050	-0.019	-0.034	0.716	0.239	0.878
Sustainable use	Caatinga	1996-1999	100	0.023	0.005	-0.110	-0.020	-0.065	0.895	0.260	0.732
Sustainable use	Cerrado	1996-1999	156	0.011	0.000	-0.070	-0.025	-0.048	0.850	0.487	0.846
Sustainable use	Mata Atlantica	1996-1999	756	0.011	0.000	-0.065	-0.024	-0.044	0.732	0.283	0.913
Sustainable use	Amazonia	2000-2004	178	0.017	0.000	-0.118	-0.053	-0.085	0.963	0.607	0.557
Sustainable use	Brazil	2000-2004	1,232	0.008	0.000	-0.067	-0.036	-0.052	0.716	0.240	0.868
Sustainable use	Caatinga	2000-2004	100	0.018	0.117	-0.063	0.007	-0.028	0.895	0.260	0.855
Sustainable use	Cerrado	2000-2004	156	0.026	0.056	-0.102	0.001	-0.050	0.850	0.487	0.905
Sustainable use	Mata Atlantica	2000-2004	754	0.009	0.001	-0.046	-0.012	-0.029	0.730	0.281	0.899
Sustainable use	Amazonia	2005-2012	178	0.019	0.001	-0.104	-0.028	-0.066	0.963	0.596	0.489
Sustainable use	Brazil	2005-2012	1,232	0.010	0.000	-0.070	-0.031	-0.050	0.716	0.235	0.868
Sustainable use	Caatinga	2005-2012	100	0.022	0.000	-0.122	-0.038	-0.080	0.895	0.260	0.833
Sustainable use	Cerrado	2005-2012	158	0.029	0.114	-0.104	0.011	-0.046	0.850	0.481	0.854
Sustainable use	Mata Atlantica	2005-2012	754	0.010	0.000	-0.061	-0.023	-0.042	0.730	0.268	0.902

Treatment	Spatial scale	Temporal scale	*n*	SE	*p* value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	T-index
Sustainable use	Amazonia	2013-2018	178	0.022	0.015	-0.096	-0.010	-0.053	0.963	0.596	0.520
Sustainable use	Brazil	2013-2018	1,228	0.011	0.000	-0.064	-0.020	-0.042	0.714	0.233	0.870
Sustainable use	Cerrado	2013-2018	158	0.020	0.020	-0.085	-0.007	-0.046	0.850	0.494	0.880
Sustainable use	Mata Atlantica	2013-2018	754	0.007	0.000	-0.044	-0.018	-0.031	0.729	0.263	0.925
Undesignated/ untitled public	Amazonia	1985-1990	8,066	0.005	0.024	-0.023	-0.002	-0.012	0.638	0.353	0.823
Undesignated/ untitled public	Brazil	1985-1990	34,212	0.005	0.033	-0.019	-0.001	-0.010	0.663	0.123	0.810
Undesignated/ untitled public	Caatinga	1985-1990	10,020	0.007	0.213	-0.021	0.005	-0.008	0.714	0.142	0.759
Undesignated/ untitled public	Cerrado	1985-1990	9,670	0.007	0.012	-0.032	-0.004	-0.018	0.718	0.256	0.893
Undesignated/ untitled public	Mata Atlantica	1985-1990	5,130	0.011	0.000	0.065	0.108	0.086	0.744	0.160	0.793
Undesignated/ untitled public	Pampa	1985-1990	404	0.041	0.067	-0.005	0.155	0.075	0.843	0.391	0.558
Undesignated/ untitled public	Pantanal	1985-1990	260	0.020	0.000	-0.131	-0.053	-0.092	0.695	0.462	0.943
Undesignated/ untitled public	Amazonia	1985-2018	8,064	0.015	0.000	0.132	0.192	0.162	0.641	0.353	0.815
Undesignated/ untitled public	Brazil	1985-2018	34,216	0.010	0.000	0.105	0.143	0.124	0.663	0.126	0.807
Undesignated/ untitled public	Caatinga	1985-2018	10,020	0.009	0.135	-0.004	0.032	0.014	0.716	0.137	0.755
Undesignated/ untitled public	Cerrado	1985-2018	9,672	0.015	0.016	0.007	0.064	0.035	0.718	0.261	0.899
Undesignated/ untitled public	Mata Atlantica	1985-2018	5,134	0.015	0.000	0.191	0.251	0.221	0.743	0.113	0.792
Undesignated/ untitled public	Pampa	1985-2018	404	0.045	0.023	0.014	0.192	0.103	0.843	0.465	0.619
Undesignated/ untitled public	Pantanal	1985-2018	262	0.014	0.000	0.123	0.179	0.151	0.696	0.458	0.958
Undesignated/ untitled public	Amazonia	1991-1995	8,062	0.006	0.309	-0.006	0.019	0.007	0.640	0.357	0.822
Undesignated/ untitled public	Brazil	1991-1995	34,216	0.004	0.027	0.001	0.018	0.010	0.663	0.125	0.818
Undesignated/ untitled public	Caatinga	1991-1995	10,024	0.004	0.766	-0.007	0.010	0.001	0.715	0.140	0.760
Undesignated/ untitled public	Cerrado	1991-1995	9,670	0.006	0.501	-0.008	0.016	0.004	0.718	0.258	0.897
Undesignated/ untitled public	Mata Atlantica	1991-1995	5,130	0.008	0.000	0.042	0.071	0.056	0.744	0.161	0.793
Undesignated/ untitled public	Pampa	1991-1995	404	0.028	0.171	-0.017	0.095	0.039	0.843	0.416	0.595
Undesignated/ untitled public	Pantanal	1991-1995	260	0.009	0.305	-0.009	0.027	0.009	0.695	0.462	0.960
Undesignated/ untitled public	Amazonia	1996-1999	8,060	0.006	0.000	0.012	0.036	0.024	0.641	0.359	0.826
Undesignated/ untitled public	Brazil	1996-1999	34,216	0.004	0.000	0.015	0.029	0.022	0.663	0.126	0.806
Undesignated/ untitled public	Caatinga	1996-1999	10,024	0.003	0.043	0.000	0.013	0.007	0.715	0.138	0.756
Undesignated/ untitled public	Cerrado	1996-1999	9,670	0.006	0.005	0.005	0.027	0.016	0.718	0.258	0.894
Undesignated/ untitled public	Mata Atlantica	1996-1999	5,132	0.008	0.000	0.050	0.080	0.065	0.743	0.141	0.797
Undesignated/ untitled public	Pampa	1996-1999	404	0.028	0.000	0.047	0.155	0.101	0.843	0.436	0.591
Undesignated/ untitled public	Pantanal	1996-1999	262	0.009	0.111	-0.003	0.030	0.014	0.695	0.458	0.959
Undesignated/ untitled public	Amazonia	2000-2004	8,064	0.009	0.000	0.050	0.084	0.067	0.641	0.354	0.833
Undesignated/ untitled public	Brazil	2000-2004	34,214	0.005	0.000	0.036	0.054	0.045	0.663	0.125	0.808
Undesignated/ untitled public	Caatinga	2000-2004	10,022	0.003	0.001	0.005	0.018	0.012	0.716	0.137	0.756
Undesignated/ untitled public	Cerrado	2000-2004	9,672	0.007	0.181	-0.004	0.022	0.009	0.718	0.259	0.897
Undesignated/ untitled public	Mata Atlantica	2000-2004	5,132	0.006	0.000	0.042	0.066	0.054	0.743	0.142	0.790

Treatment	Spatial scale	Temporal scale	*n*	SE	*p* value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	T-index
Undesignated/ untitled public	Pampa	2000-2004	404	0.013	0.000	0.046	0.098	0.072	0.843	0.431	0.577
Undesignated/ untitled public	Pantanal	2000-2004	262	0.018	0.735	-0.041	0.029	-0.006	0.695	0.466	0.941
Undesignated/ untitled public	Amazonia	2005-2012	8,062	0.010	0.000	0.086	0.127	0.106	0.641	0.353	0.820
Undesignated/ untitled public	Brazil	2005-2012	34,218	0.005	0.000	0.059	0.080	0.070	0.663	0.128	0.807
Undesignated/ untitled public	Caatinga	2005-2012	10,022	0.005	0.510	-0.007	0.014	0.004	0.716	0.135	0.760
Undesignated/ untitled public	Cerrado	2005-2012	9,672	0.008	0.001	0.011	0.041	0.026	0.719	0.261	0.895
Undesignated/ untitled public	Mata Atlantica	2005-2012	5,134	0.005	0.000	0.028	0.047	0.037	0.743	0.109	0.796
Undesignated/ untitled public	Pampa	2005-2012	404	0.050	0.020	0.019	0.214	0.116	0.843	0.455	0.576
Undesignated/ untitled public	Amazonia	2013-2018	8,060	0.009	0.000	0.060	0.096	0.078	0.641	0.355	0.813
Undesignated/ untitled public	Brazil	2013-2018	34,214	0.004	0.000	0.045	0.061	0.053	0.662	0.130	0.803
Undesignated/ untitled public	Caatinga	2013-2018	10,022	0.004	0.927	-0.008	0.007	0.000	0.715	0.135	0.759
Undesignated/ untitled public	Cerrado	2013-2018	9,672	0.006	0.000	0.026	0.051	0.039	0.719	0.261	0.890
Undesignated/ untitled public	Mata Atlantica	2013-2018	5,134	0.004	0.000	0.015	0.030	0.022	0.742	0.113	0.793
Undesignated/ untitled public	Pampa	2013-2018	404	0.012	0.078	-0.002	0.044	0.021	0.843	0.460	0.569
Undesignated/ untitled public	Pantanal	2013-2018	262	0.024	0.000	0.164	0.260	0.212	0.696	0.450	0.953
Robustness check: protected areas and sustainable-use areas filtered by known year of creation											
Protected	Cerrado	1985-1990	46	0.062	0.072	-0.234	0.010	-0.112	0.901	0.570	0.649
Protected	Mata Atlantica	1985-1990	108	0.034	0.008	-0.158	-0.023	-0.091	0.875	0.283	0.615
Protected	Cerrado	1991-1995	66	0.032	0.654	-0.078	0.049	-0.014	0.901	0.570	0.724
Protected	Mata Atlantica	1991-1995	200	0.017	0.000	-0.111	-0.046	-0.078	0.875	0.304	0.635
Protected	Cerrado	1996-1999	72	0.032	0.013	-0.141	-0.017	-0.079	0.901	0.558	0.697
Protected	Mata Atlantica	1996-1999	226	0.010	0.004	-0.050	-0.009	-0.030	0.872	0.300	0.621
Protected	Amazonia	2000-2004	50	0.010	0.000	-0.060	-0.022	-0.041	0.971	0.600	0.273
Protected	Cerrado	2000-2004	101	0.011	0.032	-0.043	-0.002	-0.022	0.901	0.547	0.685
Protected	Mata Atlantica	2000-2004	263	0.008	0.000	-0.061	-0.028	-0.044	0.872	0.298	0.619
Protected	Amazonia	2005-2012	58	0.015	0.000	-0.081	-0.024	-0.052	0.971	0.571	0.460
Protected	Cerrado	2005-2012	142	0.025	0.017	-0.108	-0.011	-0.059	0.910	0.535	0.736
Protected	Mata Atlantica	2005-2012	327	0.006	0.000	-0.047	-0.022	-0.035	0.872	0.370	0.637
Protected	Amazonia	2013-2018	64	0.019	0.000	-0.142	-0.069	-0.105	0.971	0.600	0.571
Protected	Caatinga	2013-2018	58	0.017	0.010	-0.078	-0.011	-0.044	0.962	0.167	0.557
Protected	Cerrado	2013-2018	172	0.019	0.051	-0.076	0.000	-0.038	0.910	0.558	0.742
Protected	Mata Atlantica	2013-2018	487	0.005	0.001	-0.029	-0.008	-0.019	0.872	0.329	0.638
Sustainable use	Mata Atlantica	1991-1995	58	0.015	0.011	-0.066	-0.009	-0.038	0.732	0.286	0.906
Sustainable use	Amazonia	1996-1999	84	0.013	0.000	-0.071	-0.020	-0.046	0.963	0.607	0.538
Sustainable use	Mata Atlantica	1996-1999	74	0.022	0.000	-0.133	-0.046	-0.090	0.732	0.283	0.913
Sustainable use	Amazonia	2000-2004	98	0.017	0.000	-0.129	-0.060	-0.094	0.963	0.607	0.557
Sustainable use	Mata Atlantica	2000-2004	100	0.012	0.004	-0.059	-0.011	-0.035	0.730	0.281	0.899

Treatment	Spatial scale	Temporal scale	*n*	SE	*p* value	lower_ci	upper_ci	Effect	ImbBefore	ImbAfter	T-index
Sustainable use	Amazonia	2005-2012	151	0.017	0.003	-0.082	-0.017	-0.050	0.963	0.596	0.489
Sustainable use	Cerrado	2005-2012	76	0.020	0.563	-0.051	0.028	-0.012	0.850	0.481	0.854
Sustainable use	Mata Atlantica	2005-2012	158	0.026	0.004	-0.124	-0.023	-0.073	0.730	0.268	0.902
Sustainable use	Amazonia	2013-2018	175	0.024	0.016	-0.103	-0.010	-0.057	0.963	0.596	0.520
Sustainable use	Cerrado	2013-2018	158	0.021	0.030	-0.086	-0.004	-0.045	0.850	0.494	0.880
Sustainable use	Mata Atlantica	2013-2018	709	0.007	0.000	-0.045	-0.017	-0.031	0.729	0.263	0.925

Table S5. Record of models for which federal states were merged into groups to facilitate the full estimation of parameter coefficients in GLMs in cases where insufficient observations across states prevented it. Geographically adjacent states were consecutively merged *(States grouped*). Model conversion was not achieved in 3/6 models (*Model convergence*), and are thus not reported in results (Tables S3-4).

Comparison	Temporal scale	Spatial scale	States grouped	States grouped	States grouped	Model convergence
Private vs. protected	1985-2018	Caatinga	SE+AL	PE+PB		no
Private vs. sustainable use	1985-2018	Caatinga	SE+AL	MG+BA		no
Private vs. communal	1991-1995	Brazil	SE+AL			yes
Private vs. sustainable use	2013-2018	Caatinga	SE+AL	PE+PB		no
Private vs. protected (PA filter)	2013-2018	Brazil	SE+AL	PB+RN	RR+TO	yes
Private vs. sustainable use (PA filter)	1985-2018	Brazil	MS+SP	AP+PA	PB+RN	yes

Table S6. 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') reports the percentages of all compared spatiotemporal scales where each regime ranked as more deforestation-decreasing ('best') and less deforestation-decreasing/more increasing ('worst') than all alternatives regimes (based on their respective effect sizes). In this ranking, 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%	26.23%	28.03%	8	2.66	34
Protected areas	0	0.00	0.00%	0.00%	30	14.62	88.24%	90.19%	4	1.59	11.76%	9.81%	34	26.47%	26.26%	13.97%	10.62%	4	1.31	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	41.18%	42.58%	8.82%	7.35%	8	2.71	34
Indigenous lands	1	0.36	2.94%	2.32%	26	12.97	76.47%	82.82%	7	2.33	20.59%	14.86%	34	14.71%	17.82%	12.99%	10.45%	7	2.33	34
Quilombola lands	1	0.49	2.94%	3.02%	17	7.85	50.00%	48.69%	16	7.79	47.06%	48.28%	34	14.71%	11.12%	37.99%	43.55%	16	6.50	34
All of the above compared to undesignated/untitled	**4**	**3.12**	**2.35%**	**3.31%**	**122**	**68.87**	**71.76%**	**73.17%**	**43**	**22.13**	**25.29%**	**23.52%**	**170**							
Compared to private lands																				
Public lands	21	16.61	77.78%	79.09%	3	2.27	11.11%	10.79%	3	2.13	11.11%	10.12%	27	0.00%	0.00%	81.48%	85.70%	3	0.86	27
Protected areas	0	0.00	0.00%	0.00%	21	12.70	77.78%	83.60%	6	2.49	22.22%	16.40%	27	11.11%	8.64%	6.17%	4.55%	6	1.89	27
Sustainable use areas	0	0.00	0.00%	0.00%	24	14.43	88.89%	90.40%	3	1.53	11.11%	9.60%	27	44.44%	52.20%	3.09%	1.53%	3	0.72	27
Indigenous lands	0	0.00	0.00%	0.00%	19	8.10	70.37%	59.39%	8	5.54	29.63%	40.61%	27	40.74%	36.19%	0.00%	0.00%	8	5.18	27
Quilombola lands	0	0.00	0.00%	0.00%	11	6.39	40.74%	40.11%	16	9.53	59.26%	59.89%	27	10.48%	8.63%	10.48%	9.56%	21	9.26	31
All of the above compared to private	**21**	**16.61**	**15.56%**	**20.33%**	**78**	**43.88**	**57.78%**	**53.70%**	**36**	**21.22**	**26.67%**	**25.97%**	**135**							

Table S7. Synthesized direction of cross-scale effects of different land-tenure regimes, but focusing on scales remaining after time-filtering strict-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 S6 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 strict-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	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%	3.48%	27.63%	30.69%	1	0	19
Protected areas	0	0.00	0.00%	0.00%	18	9.63	94.74%	96.12%	1	0.39	5.26%	3.88%	19	15.79%	12.51%	7.89%	5.22%	1	0	19
Sustainable use areas	0	0.00	0.00%	0.00%	15	8.51	78.95%	80.55%	4	2.06	21.05%	19.45%	19	47.37%	52.65%	9.21%	6.60%	4	1	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%	27.89%	11.84%	7.93%	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	5.26%	3.48%	43.42%	49.57%	7	3	19
All of the above compared to undesignated/untitled	5	0.88	5.26%	1.58%	80	47.38	84.21%	85.24%	14	7.33	14.74%	13.19%	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%	93.05%	0	0	17
Protected areas	0	0.00	0.00%	0.00%	15	9.52	88.24%	89.19%	2	1.15	11.76%	10.81%	17	17.65%	18.28%	0.00%	0.00%	2	1	17
Sustainable use areas	0	0.00	0.00%	0.00%	13	8.34	76.47%	74.87%	4	2.80	23.53%	25.13%	17	35.29%	38.65%	2.94%	3.47%	4	2	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	41.18%	39.21%	0.00%	0.00%	7	4	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%	3.85%	2.94%	3.47%	10	6	17
All of the above compared to private	16	13.00	18.82%	22.79%	46	28.47	54.12%	49.89%	23	15.59	27.06%	27.31%	85							

Table S8. 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 balance	% 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	8.70%	8.33%	30	22.18	65.22%	65.83%	13	9.25	27.66%	27.02%	47
Protected areas	0	0.00	0.00%	0.00%	30	14.62	88.24%	90.19%	4	1.59	11.76%	9.81%	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	1	0.36	2.94%	2.32%	26	12.97	76.47%	82.82%	7	2.33	20.59%	14.86%	34
Quilombola lands	1	0.49	2.94%	3.02%	17	7.85	50.00%	48.69%	16	7.79	47.06%	48.28%	34
Communal lands	0	0.00	0.00%	0.00%	12	7.78	92.31%	91.22%	1	0.75	7.69%	8.78%	13
All of the above compared to undesignated/untitled	6	3.66	3.08%	3.35%	141	80.48	72.31%	73.64%	49	25.70	25.00%	23.39%	196
Robustness check: protected areas and sustainable use areas filtered by known year of creation													
Protected areas	0	0	0.00%	0.00%	23	11.60	88.46%	91.10%	3	1.13	11.54%	8.90%	26
Sustainable use areas	0	0	0.00%	0.00%	15	8.51	78.95%	80.55%	4	2.06	21.05%	19.45%	19
All of the above compared to undesignated/untitled (using filtered results instead)	6	3.66	3.49%	3.76%	123	70.89	71.51%	72.85%	44	23.30	25.43%	23.81%	173
Compared to private lands													
Public lands	29	21.64	61.70%	63.22%	4	2.81	8.51%	8.20%	14	9.79	29.79%	28.58%	47
Protected areas	0	0.00	0.00%	0.00%	25	15.69	75.76%	81.21%	8	3.63	24.24%	18.79%	33
Sustainable use areas	0	0.00	0.00%	0.00%	27	16.42	84.38%	84.50%	5	3.01	15.63%	15.50%	32
Indigenous lands	0	0.00	0.00%	0.00%	19	8.10	67.86%	57.97%	9	5.87	32.14%	42.03%	28
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	1	0.35	7.69%	7.16%	9	3.48	69.23%	70.34%	3	1.11	23.08%	22.50%	13
All of the above compared to private	31	22.69	16.58%	20.15%	95	52.88	50.80%	46.96%	61	37.03	32.62%	32.88%	187
Robustness check: protected areas and sustainable use areas filtered by known year of creation													
Protected areas	0	0.00	0.00%	0.00%	20	12.41	83.33%	86.03%	4	2.01	16.67%	13.97%	24
Sustainable use areas	0	0.00	0.00%	0.00%	14	8.73	77.78%	75.73%	4	2.80	22.22%	24.27%	18
All of the above compared to private (using filtered results instead)	31	22.69	18.90%	22.74%	77	41.91	46.95%	42.00%	56	35.20	34.15%	35.27%	164

Table S9. 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 ($p \leq 0.05$).

	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	12.54%	23.20%	31.57%	41.62%	49.25%	6.25%	10.42%	12.50%	14.58%	20.83%
public vs. protected	8.05%	15.97%	22.73%	28.43%	34.59%	5.71%	5.71%	5.71%	8.57%	8.57%
public vs. sustainable use	6.69%	13.00%	18.38%	23.87%	29.16%	2.86%	5.71%	17.14%	17.14%	28.57%
public vs. indigenous	12.72%	23.31%	32.44%	36.92%	45.80%	0.00%	11.43%	11.43%	11.43%	14.29%
public vs. quilombola	21.39%	37.18%	56.50%	72.26%	83.73%	2.86%	11.43%	14.29%	20.00%	20.00%
public vs. communal	8.14%	15.81%	22.96%	29.59%	35.79%	0.00%	0.00%	0.00%	7.69%	7.69%
private vs. public	11.10%	20.95%	26.96%	37.62%	44.59%	6.25%	10.42%	12.50%	14.58%	18.75%
private vs. protected	8.56%	16.81%	24.82%	32.04%	38.54%	2.94%	8.82%	11.76%	17.65%	17.65%
private vs. sustainable use	7.68%	15.10%	21.92%	28.40%	34.57%	0.00%	3.03%	3.03%	12.12%	21.21%
private vs. indigenous	11.78%	22.68%	31.45%	37.76%	45.25%	3.45%	6.90%	6.90%	20.69%	24.14%
private vs. quilombola	26.46%	47.77%	66.12%	80.85%	103.00%	5.71%	11.43%	14.29%	22.86%	28.57%
private vs. communal	9.11%	17.51%	25.34%	32.62%	39.52%	0.00%	0.00%	0.00%	0.00%	8.33%
Average across tenure-regime comparisons	*12.02%*	*22.44%*	*31.77%*	*40.17%*	*48.65%*	*3.00%*	*7.11%*	*9.13%*	*13.94%*	*18.22%*
Spatial scales										
Brazil	9.91%	18.51%	24.03%	32.31%	40.12%	1.23%	2.47%	2.47%	7.41%	13.58%
Amazonia	12.97%	24.72%	35.44%	44.84%	54.36%	5.95%	10.71%	16.67%	21.43%	28.57%
Caatinga	21.45%	41.81%	60.40%	79.84%	95.11%	4.92%	13.11%	18.03%	22.95%	26.23%
Cerrado	12.14%	22.89%	31.19%	35.93%	42.24%	1.43%	11.43%	11.43%	15.71%	18.57%
Mata Atlantica	7.41%	13.56%	20.28%	26.32%	31.93%	2.86%	4.29%	5.71%	11.43%	12.86%
Pampa	4.93%	9.39%	13.56%	17.37%	20.98%	0.00%	0.00%	0.00%	0.00%	0.00%
Pantanal	9.91%	19.25%	27.97%	35.57%	42.71%	16.67%	16.67%	16.67%	16.67%	25.00%
Average across spatial scales	*11.25%*	*21.45%*	*30.41%*	*38.88%*	*46.78%*	*4.72%*	*8.38%*	*10.14%*	*13.66%*	*17.83%*
Temporal scales										
1985-2018	8.67%	16.52%	23.55%	29.83%	35.58%	1.79%	8.93%	8.93%	12.50%	17.86%
1985-1990	15.47%	28.23%	33.11%	41.12%	52.36%	3.64%	9.09%	14.55%	18.18%	25.45%
1991-1995	16.40%	32.42%	45.49%	56.32%	66.32%	5.36%	8.93%	8.93%	12.50%	19.64%
1996-1999	10.71%	20.62%	29.82%	38.97%	47.27%	1.75%	8.77%	10.53%	14.04%	17.54%
2000-2004	10.41%	18.90%	28.46%	36.88%	44.65%	1.79%	5.36%	7.14%	14.29%	16.07%
2005-2012	10.95%	21.22%	30.91%	39.44%	47.09%	1.79%	5.36%	10.71%	16.07%	19.64%
2013-2018	9.19%	16.90%	24.43%	31.80%	38.10%	8.93%	10.71%	12.50%	17.86%	19.64%
Average across spatial scales	*11.69%*	*22.12%*	*30.82%*	*39.19%*	*47.34%*	*3.58%*	*8.16%*	*10.47%*	*15.06%*	*19.41%*

Table S10. Summary of mean differences in key covariates between matched sample and entire population of Brazilian parcels. For each covariate (elevation (in meters), slope (in degrees), travel time to nearest city (in minutes), human population density , and area (in ha)), we compare the means of both matched sample and the entire population (based on a stratified representative sample of parcels), and report the larger values in bold, for visual aid (e.g. for public vs. private on average, the matched sample had lower elevation than the entire population). We also report the absolute standardized mean difference (ASMD) between the matched sample and the population as a measure of these differences, with values closer to 0 indicating no differences between groups.

Tenure-regime comparison	Mean elevation (matched)	Mean elevation (population)	Elevation ASMD	Mean slope (matched)	Mean slope (matched)	Slope ASMD	Mean travel time (matched)	Mean travel time (population)	Travel time ASMD	Mean population (matched)	Mean population (population)	Human population ASMD	Mean area (matched)	Mean area (population)	Area ASMD
Public vs. private	272.113	**374.451**	0.451	1.056	**1.804**	0.537	**215.834**	170.089	0.260	0.949	**1.460**	0.087	**1594.247**	342.993	0.460
Public vs. protected	**376.100**	313.589	0.349	**1.244**	1.208	0.226	**458.325**	262.197	0.414	**1.611**	1.582	0.098	**68175.168**	4247.486	0.507
Public vs. sustainable_use	**323.718**	312.660	0.296	**1.468**	1.193	0.239	**382.016**	252.794	0.374	**1.483**	1.405	0.106	**48957.248**	4548.720	0.316
Public vs. indigenous	266.058	**311.891**	0.307	0.989	**1.205**	0.246	**519.122**	264.588	0.345	**1.236**	1.202	0.128	**37307.475**	9306.220	0.436
Public vs. quilombola	214.454	**315.179**	0.704	1.064	**1.142**	0.253	199.802	**260.053**	0.294	**1.333**	1.229	0.141	**6424.032**	3900.827	0.284
Public vs. communal	69.436	**220.601**	0.989	**0.130**	0.769	0.784	**555.164**	396.144	0.320	0.649	**0.881**	0.056	**3459.764**	10141.409	0.113
Private vs. public	272.113	**374.650**	0.456	1.056	**1.816**	0.544	**215.834**	169.831	0.264	0.949	**1.493**	0.090	**1594.247**	337.529	0.455
Private vs. protected	423.387	**452.292**	0.278	1.798	**2.039**	0.214	**370.432**	179.991	0.342	**4.044**	1.706	0.190	**27280.312**	106.919	0.656
Private vs. sustainable use	**451.900**	450.391	0.237	**2.773**	2.021	0.383	**290.159**	181.754	0.273	**2.230**	1.749	0.096	**15131.491**	119.635	0.398
Private vs. indigenous	280.292	**456.633**	0.797	1.273	**2.048**	0.572	**463.940**	186.798	0.611	1.520	**1.772**	0.151	**16857.519**	407.279	0.707
Private vs. quilombola	248.856	**456.667**	1.021	1.370	**2.024**	0.501	**207.032**	184.176	0.177	**2.805**	1.741	0.108	**6073.282**	126.352	0.504
Private vs. communal	58.595	**334.289**	1.641	0.293	**1.701**	1.141	**532.994**	255.234	0.673	0.843	**1.458**	0.090	**1529.501**	189.499	0.077
Spatial Scale															
Brazil	269.445	**415.363**	0.652	1.347	**1.859**	0.478	**380.721**	192.179	0.434	**1.826**	1.631	0.094	**18756.604**	2309.234	0.245
Amazonia	95.032	**159.394**	0.698	0.384	**0.776**	0.566	**821.159**	439.857	0.544	0.594	**0.869**	0.082	**67660.044**	7661.397	0.444
Caatinga	331.144	**380.221**	0.394	**1.458**	1.360	0.340	136.230	**137.341**	0.167	**2.064**	1.901	0.140	**3544.037**	338.386	0.557
Cerrado	486.524	**539.708**	0.702	1.189	**1.495**	0.393	**204.386**	188.665	0.295	**1.345**	1.127	0.156	**5553.383**	669.125	0.468
Mata Atlantica	375.530	**494.644**	0.425	2.565	**2.951**	0.259	**111.219**	109.910	0.256	**3.368**	2.414	0.099	**1231.630**	313.810	0.469
Pampa	**177.034**	171.840	0.055	0.856	**1.685**	0.797	**113.198**	94.173	0.322	0.226	**1.001**	0.208	**920.966**	76.675	1.267
Pantanal	182.032	**182.671**	0.054	0.523	**0.855**	0.298	169.578	**182.573**	0.087	0.255	**0.492**	0.074	**1302.871**	1577.547	0.059
Temporal Scale															
1985-2018	292.993	**376.770**	0.532	1.301	**1.641**	0.435	**336.681**	215.714	0.340	**1.774**	1.422	0.120	**19792.971**	2434.078	0.458
1985-1990	292.712	**372.965**	0.553	1.300	**1.614**	0.431	**336.054**	212.098	0.340	**1.370**	1.360	0.114	**19731.498**	1881.821	0.446
1991-1995	292.996	**369.200**	0.533	1.302	**1.603**	0.422	**336.169**	213.175	0.347	**1.439**	1.420	0.103	**19738.298**	2768.898	0.443
1996-1999	293.252	**376.679**	0.545	1.301	**1.621**	0.412	**336.625**	216.628	0.336	**1.577**	1.358	0.124	**19792.447**	2161.936	0.443
2000-2004	293.204	**373.309**	0.532	1.301	**1.673**	0.431	**336.627**	219.472	0.346	**1.776**	1.520	0.122	**19795.835**	2588.518	0.443
2005-2012	293.597	**378.865**	0.560	1.305	**1.652**	0.421	**340.257**	216.207	0.344	**1.873**	1.646	0.115	**20049.582**	2734.127	0.443
2013-2018	291.141	**374.460**	0.566	1.317	**1.646**	0.426	**341.225**	218.240	0.336	**1.999**	1.830	0.100	**20077.450**	1956.230	0.441

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.

SI References

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