The Differential Responses of Farmers on Private and Public Lands to Droughts in the Brazilian Amazon *

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Abstract

Climate change has increased the frequency of drought events in the Amazon. Deforestation worsens local climate dynamics, and weather shocks can, in turn, influence land-use decisions. In the Brazilian Amazon, where land tenure is a mix of public and private holdings, landholders on public lands may lack incentives to manage forests sustainably. This study examines how droughts differentially affect pasture expansion on public versus private lands. Using spatially matched comparisons within 0.5° grid cells, we find that during drought years, ranchers on public lands expand pasture area by 30% more than in baseline years, while private landholders show no significant response. We explore mechanisms behind this difference and find that drought-induced pasture degradation leads to expansion in both tenure types. However, ranchers on public lands expand pasture by 20% more than those on private lands in response to similar degradation. Moreover, only public landholders continue to expand even after controlling for degradation, suggesting additional drivers, such as lower deforestation costs in drier, more flammable forests. These findings indicate that climate-induced droughts increase deforestation pressure on public forests, which store substantial carbon stocks and cover over two-thirds of the Amazon. This vulnerability complicates efforts to mitigate climate change. Policymakers should enhance monitoring of public lands during drought years and improve enforcement of public property rights.

1 Introduction

Despite decades of regulation and improved monitoring, deforestation in the Amazon remains one of the most pressing global environmental challenges (Franco et al. 2025). Much of the ongoing

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deforestation is occurring on public land, where land grabbers and squatters regularly occupy and clear the land for productive use (e.g. cattle grazing). Previous research highlights weak institutions can contribute to the growth of deforestation on these public lands (Mendelsohn 1994; Balboni et al. 2023). When land tenure is insecure, as it is for much of the publicly occupied land, the risk of losing ownership causes land holders to discount the returns from preserving the value of the land, creating weaker incentives to invest in costly conservation or restoration practices and stronger incentives to expand into forested territory when opportunities arise (Farzin 1984; Bohn and Deacon 2000). In this paper, we demonstrate that droughts, which have become more frequent and severe in the Amazon (Ottoni et al. 2023; Bottino et al. 2024), have the potential to amplify these incentives, leading to more rapid deforestation of public lands.

Understanding how land tenure may alter incentives to deforest is particularly relevant in the Brazilian Amazon, where the remaining forest is spread across a mosaic of public and private lands. Between 2019 and 2021, roughly half of the deforestation occurred on indigenous lands, units of conservation, and undesignated public forests (Moutinho, Alencar, et al. 2022). Commercial agriculture is the main driver of deforestation throughout the region, with pasture-fed cattle ranching alone accounting for 90% of deforestation (Tyukavina et al. 2017; Mapbiomas 2024). Cattle ranching in the Amazon is characterized by extensive pasture management with little investment in irrigation and commercial inputs like fertilizer and lime (Garrett et al. 2021; Feltran-Barbieri and Féres 2021). As a result, ranchers depend heavily on rainfall, making them particularly vulnerable to climate shocks.

In the Amazon, climate shocks, and in particular droughts, have become far more frequent and intense. Several "once-in-a-century droughts" occurred in the last few decades and the 2023-2024 drought recorded the lowest levels of rainfall in 120 years of measurement (Clarke et al. 2024). These droughts not only degrade the productivity of land being used for agriculture or grazing, they also reduce the cost of clearing new land through burning, a common practice in the Amazon (Boucher, Roquemore, and Fitzhugh 2013). Both of these channels can put additional pressure on ranchers to deforest and expand into more productive land (Staal et al. 2020). Yet the extent to which land tenure mediates such responses has not been clarified theoretically or verified empirically in the literature.

To explore how ranchers' responses to droughts can differ based on land tenure, we first introduce a simple model of a rancher's dynamic decision to expand their pasture holdings into new, previously forested land. We first examine how pasture expansion rates differ across land tenure regimes in the absence of climate shocks. Our model incorporates two key differences between the incentives faced by ranchers on public versus private land. First, ranchers on public lands face a threat of eviction, weakening their incentive to expand their pasture holdings. Second, and working in the opposite direction, the fact that there is no opportunity cost to using more public land strengthens the incentive for ranchers on public lands to expand their pasture holdings. Thus, in the absence of drought shocks, the difference in the baseline rate of pasture expansion on private versus public lands is ambiguous.

Next, to understand how the rate of pasture expansion responds to climate shocks, we introduce two drought impacts in the model. First, droughts degrade the quality of existing pastureland, increasing the demand for new, non-degraded pastureland. Second, to capture the fact that it is easier to clear forested land when it is drier, we model drought shocks as reducing the cost of clearing new land for pasture. While our model illustrates that the drought-induced degradation of pasture will drive up pasture expansion rates equally across public and private lands, we find that the reduction in clearing costs drives a larger increase in pasture expansion on public lands. In sum, the findings from our theoretical model suggest that droughts systematically amplify pasture expansion on public relative to private lands, even when baseline expansion rates differ, thereby highlighting how climate shocks and tenure insecurity interact to shape land-use dynamics.

To empirically examine whether droughts have a more pronounced impact on pasture expansion within public versus private lands, we use observed land-use behavior across public and private land in the Brazilian Amazon. We test whether droughts induce greater pasture expansion on public than on private lands by comparing their responses within the same spatial and climate context. Our approach exploits plausibly exogenous variation in the timing and severity of droughts across 0.5° grid cells in the Brazilian Amazon and contrasts land-use changes between public and private areas inside each cell. This within-cell, differences-in-differences framework controls for all time-invariant local characteristics and common shocks, isolating how land tenure mediates farmers' responses to droughts. In line with the conceptual model, we interpret stronger drought responses on public lands as evidence that droughts systematically amplify pressure for pasture expansion on public relative to private land.

Our empirical findings provide significant evidence that ranchers on public lands respond more to droughts: we find that during drought years, ranchers on public lands expand pasture area by 30% more than in baseline years, while private landholders' land use shows no significant difference. These findings are in line with the predictions from our theoretical model.

However, we find that these results also depend on the region's climate context. When we disaggregate the analysis by climatic conditions, a more nuanced pattern emerges. Ranchers on public lands consistently expand their pasture area during droughts, regardless of whether the region is dry or humid. In contrast, ranchers on private lands increase pasture expansion during droughts only in drier regions. This heterogeneity suggests that in areas where rainfed agriculture predominates, even private landholders may lack incentives to invest in sustainable land management, behaving more like their counterparts on public lands.

Investigating the possible mechanisms behind our main results, we find that droughts affect pasture degradation, which in turn affects pasture expansion. Ranchers in public lands are more sensitive to pasture degradation than private land farmers. This finding provides evidence that public land ranchers are less inclined to invest in and restore their degraded pastureland, preferring instead to move to new land.

We also find that farmers in public lands still react to droughts after controlling for pasture degradation, which is evidence that other mechanisms are at play. One potential mechanism is that the environmental deregulation after 2012 incentivizes farmers to claim more public land, and the reduction in clearing costs during droughts makes provides better opportunities to claim new land. While our data do not allow us to identify this potential mechanism unequivocally, we find supporting evidence consistent with land claims expanding during droughts in the post-2012 period.

This study contributes significantly to three branches of literature. A large body of research investigates the relationship between land tenure security and land use. Building on the classic argument by Hardin 1968, who demonstrated in his *Tragedy of the Commons* that open-access resources tend to be overexploited in the absence of well-defined property rights—so that, for open-access forests, land is quickly converted to agricultural use and little or no effort is made to regenerate or maintain it—subsequent works have examined how different forms of tenure influence land-use decisions. Mendelsohn 1994 proposed a model predicting that farmers with low tenure security would discount the future more heavily and adopt less sustainable practices.

The effects of land tenure security on deforestation, however, are theoretically ambiguous and depend on the nature of the extractive process. Better tenure security can provide incentives to invest in land and make it more productive; thus, if deforestation is capital-intensive, greater security may lead to higher investment and more deforestation. Conversely, if deforestation is labor-intensive and farmers in public lands view the land as a public good, they may lack incentives to invest and manage it sustainably, thereby increasing deforestation (Krishna et al. 2017; Walker et al. 2025).

Subsequent empirical works examined this relationship across different contexts, particularly in the Brazilian Amazon. Pacheco and Meyer 2022 show that private tenure outperforms undesignated public lands in reducing deforestation, while protected public lands such as conservation units and Indigenous territories outperform private tenure. Similarly, Baragwanath and Bayi 2020 find that granting property rights to Indigenous communities leads to lower deforestation. Other studies explore specific policy and institutional changes: Probst et al. 2020 find that recipients of land titles under the *Terra Legal* program increased deforestation after titling, while Qin et al. 2023 document substantial reductions in deforestation following the demarcation of protected areas, particularly those under strict protection. Conversely, Keles, Pfaff, and Mascia 2023 show that the erasure of protected areas increases deforestation when these areas had previously constrained agricultural expansion. Overall, the meta-analysis by Robinson, Holland, and Naughton-Treves 2014 synthesizes these findings, confirming that protected public lands tend to yield better conservation outcomes than private lands, while private tenure generally outperforms undesignated public lands, especially in South America.

A related branch of literature investigates the relationship between agricultural efficiency and land use. Many studies investigate how increasing cattle stocking rates and restoring degraded pasture could help preserving and restoring native vegetation (Cohn et al. 2014; Spera 2017; Feltran-Barbieri and Féres 2021). We show that cattle intensification will remain a challenge as long as public land is readily available.

The literature on the impact of climate shocks on land-use decisions remains nascent, and theory suggests that the effects of droughts on deforestation through agricultural land-use change are complex and ambiguous (Desbureaux and Damania 2018). On the one hand, climate shocks reduce agricultural yields, lowering the profitability of agriculture, raising the value of outside options, and thereby reducing deforestation pressure. On the other hand, drought-induced land degradation may lead farmers to seek new cultivating areas, increasing deforestation.

Empirical evidence on this relationship remains limited. A handful of studies examining droughts and deforestation in Africa—where subsistence agriculture dominates—find that droughts tend to increase deforestation (Desbureaux and Damania 2018; Leblois 2021; Vaglietti, Delacote, and Leblois 2022). In the Amazon, Staal et al. 2020 document a feedback mechanism between drought and deforestation, estimating that deforestation contributes to about 4% of the region's observed drying. They also find that while deforestation responses to droughts vary across locations, deforestation increases by an average of 0.13% in drought years.

We connect these branches of literature by investigating how climate shocks affect land-use responses across different land tenures. To the best of our knowledge, this is the first study to explore how the economic incentives generated by the interplay of climate and different land tenures may lead to significantly different land use responses. The findings also inform policymakers to increase monitoring of public lands during drought years and design mechanisms to better enforce public property rights.

2 Background

2.1 Land Tenure and Property Rights

A mosaic of public and private lands, the region has a long history of insecure land tenure in which land grabbers and squatters often occupy public land (Azevedo-Ramos et al. 2020). Public lands are mainly divided into conservation units, indigenous lands, and undesignated public forests. Some conservation units may allow sustainable agricultural production, but others strictly forbid economic activities. Undesignated public forests are not protected and are especially vulnerable to occupation.

The origins of Brazil's land tenure system are deeply rooted in inequality. During the colonial period under Portuguese rule, large estates were granted to a small elite through a feudal system called *sesmarias*, while most peasants cultivated land informally (Mueller, L. Alston, and Harris 2011). When the *sesmarias* were abolished in 1823, no new rules immediately replaced them, creating a legal vacuum in which both subsistence farmers and powerful landowners invaded unclaimed public land. Conflicts were often resolved by force: as a 19th-century senator observed, "land conflicts were settled by the blunderbuss"[1] (Westin 2020b).

In 1850, the *Lei de Terras* (Land Law) sought to impose order by requiring that land acquisitions occur via purchase, ending free occupation (Westin 2020b). While designed to curb arbitrary appropriation, the law consolidated land concentration by excluding small farmers without capital (Araujo et al. 2009). From then on, Brazil's rural landscape combined a minority of secure large landholders with a large share of informal occupants, patterns that later shaped frontier expansion.

For centuries, the Amazon was sparsely populated and economically peripheral, apart from the rubber boom [2] at the turn of the 20th century. As such, the land dynamics described above remained largely outside the Amazon. This changed under the military dictatorship (1964–1985), which promoted settlement as part of a national integration strategy. The federal government empowered the National Institute for Colonization and Agrarian Reform (INCRA) to manage colonization programs, appropriating large tracts of state lands along new highways such as the Transamazônica

[1] Bacamarte, an old firearm.

[2] The rubber boom, fueled by rising global demand for latex used in industrial goods such as tires, brought a temporary influx of migrants and capital to the region, but collapsed once Southeast Asian plantations outcompeted Amazonian extraction.

(Carrero et al. 2022; Araujo et al. 2009; L. Alston, Gary D. Libecap, and Mueller 2000). Between 1970 and 1985, the population of Pará, a state at the center of Brazil's agricultural frontier, nearly doubled, and farmland expanded rapidly, as both smallholders and large ranchers moved into the region (L. Alston, Gary D. Libecap, and Mueller 2000; Garrett et al. 2021).

With the return to democracy, Brazil enacted a new Constitution in 1988, reshaping the legal framework for land and environment. The charter enshrined the *social function of property*, requiring land ownership to contribute to social welfare and environmental protection; reinforced protections for Indigenous territories; and mandated the creation of conservation units. Public lands were recognized as belonging to the Union or state governments to be allocated for collective purposes such as agrarian reform, conservation, or Indigenous recognition (Mueller 2018).

At the same time, the Constitution institutionalized contradictions that persist. While prohibiting privatization of undesignated public lands, Article 191 allowed squatters to transform occupation into ownership via "productive use" (usufruct after one year and full title after five years), creating incentives to deforest in order to demonstrate cultivation (Araujo et al. 2009).

Although official settlement programs distributed small plots, much occupation occurred informally on "vacant" or undesignated public lands (Araujo et al. 2009). A frontier logic took hold that deforestation itself could confer property rights: clearing forest and establishing pasture signaled "productive use" and strengthened claims—according to a squatter, "the best title is the biggest axe" (L. Alston, Gary D. Libecap, and Mueller 2000). Thus, the 1988 charter was both a democratic breakthrough (expanding Indigenous and environmental rights) and a continuation of frontier dynamics where occupation and clearing could translate into claims.

Subsequent laws deepened these contradictions. The Terra Legal program (2009), one of the world's largest titling initiatives, granted conditional rights to occupants of undesignated public lands who could prove cultivation prior to 2008. Despite safeguards, the program legitimized illegal occupation and spurred speculation (Probst et al. 2020; Carrero et al. 2022).

In 2012, Congress approved a major reform of the Forest Code. While presented as improving compliance and monitoring, it amnestied illegal deforestation prior to 2008 and introduced the self-declaratory Rural Environmental Registry (*Cadastro Ambiental Rural*, CAR) (Garrett et al. 2021; Mueller 2018). Although CAR does not establish ownership, it has been used by land grabbers to bolster claims over invaded areas, especially when paired with deforestation to signal "productive use" (Carrero et al. 2022). Over the following decade, repeated amnesties and expansions in the legalizable size of claims further incentivized invasion and regularization expectations (Abdenur et al. 2020; Carrero et al. 2022). Nonetheless, the Brazilian Federal Police often arrests landgrabbers

(Polícia Federal em Rondônia 2024).

The Amazon today is a patchwork of tenure regimes: titled private farms and agrarian reform settlements, Indigenous lands, conservation units, and vast undesignated public forests—on the order of $\sim$100 million hectares (Pacheco and Meyer 2022; Azevedo-Ramos et al. 2020). Clearing in undesignated lands is always illegal (Assunção, Gandour, and Rocha 2023). Yet these areas are disproportionately targeted by land grabbing and speculative occupation, with many operators anticipating future regularization given the historical pattern of amnesties and titling (Carrero et al. 2022).

In sum, the trajectory of property rights in Brazil reveals a persistent tension between formal protections and informal practices. From colonial concentration to the 1988 Constitution and subsequent reforms, the Amazon faces the consequences of these contradictions: insecure tenure, state-sponsored colonization, and permissive or contradictory legislation converge to make public lands a vast agricultural frontier where land-grabbing is opportunistic, but property rights are insecure (Carrero et al. 2022; Mueller 2018; L. Alston, Gary D. Libecap, and Mueller 2000). As a result, deforestation in public lands is one of the most pressing problems in the Amazon. Between 2019 and 2021, 51% of the total deforestation occurred in indigenous lands, units of conservation, and undesignated public forests (Moutinho, Alencar, et al. 2022).

2.2 Droughts and Deforestation

The Amazon Basin is an exceptionally humid region that covers more than 40% of South America's land area and has a mean annual precipitation of 2300mm (Fisch, Marengo, and C. Nobre 2006). Rainfall is not equally distributed across the Amazon, however. The Southern portion of the Amazon receives less than 2000mm of rainfall, while the Northwestern portion of the Amazon, in the bordering region of Brazil, Colombia, and Venezuela, receives over 3000mm of rainfall, and there is no dry season in this region (Fisch, Marengo, and C. Nobre 2006; Michot et al. 2019). Nevertheless, most of the Amazon has marked rainy and dry seasons (Marengo 2004; NASA 2021; Skidmore 2023). The rainy season lasts from November to March, and the dry season from May to September, with August being the driest month (Fisch, Marengo, and C. Nobre 2006; NASA 2021). April and October are considered transition months (Fisch, Marengo, and C. Nobre 2006). The rainforest acts as a water recycling system and pumps humidity back into the atmosphere (NASA 2021). As a result, significant rainfall takes place even during the dry months (Bacellar 2022).

Despite the region being naturally humid, climate change has increased the frequency and intensity of drought events in the Amazon. The 2023-2024 drought recorded the lowest level of rainfall in 120 years of measurement, and several "once-in-a-century droughts" took place in the last few decades (Clarke et al. 2024). Different studies document increases in the duration of the dry season (Bottino et al. 2024). Droughts in the Amazon are also related to the El Niño Southern Oscillation (ENSO) phenomenon. The region suffered drier conditions during the ENSO years of 1983, 1995/1995, 1997/1998, 2005, 2010, 2015/2016 and 2023. Moreover, climate change is making extreme ENSO events more frequent (Skidmore 2023). The rainforest is increasingly at risk of savannization as the region's climate shifts to drier conditions (Bottino et al. 2024).

Local disturbances of the region's ecosystem also affect rainfall patterns. Deforestation affects the region's climate by altering the water cycle. Trees pump water from the soil into the air through their leaves, and this transpiration mechanism acts as an essential buffer during droughts (Staal et al. 2020). However, a feedback mechanism between climate and deforestation may exist, as the drivers of deforestation are also affected by climate shocks. Deforestation substantially increased in 2005, 2007, 2010, and 2015, and those years were marked by intense El Niño events, high air temperatures, or severe drought (Qin et al. 2023).

Droughts may affect land use change by lowering deforestation costs. In Brazil, fire is often used in the deforestation process. Fires do not occur naturally in the Amazon. As a humid rainforest, fires only occur during warm and dry conditions (Bottino et al. 2024). Deforestation usually occurs during the dry months, when the forest is easier to cut down and burn (Boucher, Roquemore, and Fitzhugh 2013). Droughts might amplify deforestation by making the vegetation more flammable and making it easier to use traditional slash-and-burn techniques. Furthermore, farmers often use fire to clear pasture of weeds, and dryer vegetation makes it easier for the fire to escape into nearby areas (Staal et al. 2020).

Another channel through which climate shocks may affect land use change is by affecting agricultural yields. Agriculture, a sector particularly vulnerable to climate shocks, relies on precipitation and temperature for production. Farmers may respond to droughts by altering land use, and the farmers' circumstances may alleviate or increase deforestation pressure (Mendelsohn 1994; Amacher, Koskela, and Ollikainen 2009; Robinson, Holland, and Naughton-Treves 2014; Balboni et al. 2023). On the one hand, considering the farmer has access to outside options, climate shocks reduce agricultural yields, thereby increasing the value of alternative options and alleviating deforestation pressure. On the other hand, farmers may attempt to expand their cropland or pasture to new areas to compensate for degradation caused by climate shocks, potentially leading to long-term land-use changes.

Pasture-fed cattle ranching is the agricultural activity that drives the most deforestation in the

Brazilian Amazon. It is responsible for 90% of the total deforestation in the region (Mapbiomas 2024). Cattle ranching in the Amazon is characterized by extensive pasture management. Investments, such as irrigation and commercial inputs, including fertilizers and lime, are uncommon (Garrett et al. 2021). These extensive practices lead to across-the-board pasture degradation due to inadequate land use and management of vegetation or livestock, which often results from overgrazing, lack of fertilization, and pest control (Feltran-Barbieri and Féres 2021). Since reverting this degradation requires considerable investments, ranchers often move to newly deforested land (Garrett et al. 2021). As a result, ranchers depend heavily on rainfall, and this extensive production system leaves farmers particularly vulnerable to climate shocks.

3 Conceptual Model

Understanding how land tenure conditions shape ranchers' responses to climate shocks requires a framework that explicitly incorporates the risk environment faced by farmers in the Amazon. The Brazilian Amazon is characterized by a mosaic of private and public lands, where the latter include conservation units, Indigenous territories, and vast expanses of undesignated public forests (UPFs). In these public lands, ranchers often occupy irregularly and face persistent legal and extralegal threats to their tenure security. Federal law prohibits the privatization of these forests, yet enforcement is uneven, and waves of amnesties and titling initiatives have blurred the boundaries between licit and illicit claims (Carrero et al. 2022).

To formalize these dynamics, we model a farmer's land use as an asset-replacement problem. Farmers are assumed to graze cattle on a single parcel that degrades over time, and this parcel can be "replaced" by clearing a fresh plot of land. Each season, the farmer observes the quality of the land, which is a function of time and drought spells, and then chooses to continue grazing on the current piece of land or to replace the land and clear a new plot.

3.1 Dynamics

Formally, let $d \in \{0, 1\}$ denote a farmer's binary decision to replace the land. Land quality $q \in [0, \bar{q}]$ is an endogenous state variable and is assumed to decrease over time. Land quality is negatively impacted by drought spells $z \in \{0, 1\}$, an exogenous state variable that follows a stationary Markov process. Land quality in period t can be expressed using the state equation:

$$q_t = \begin{cases} g(q_{t-1}, z_t) & \text{if } d_t = 0, \\ \bar{q} & \text{if } d_t = 1. \end{cases}$$

That is, current land quality q_t depends on last period's land quality and whether there is a drought spell in the current period. The condition $0 < g_q < 1$ ensures that land quality is decreasing over time, and $g(q, 1) < g(q, 0)$ implies land quality is negatively impacted by drought. If a farmer replaces the parcel, land quality "resets" to its maximum $\bar{q}$.

Let's consider a farmer's utility $u(q, d)$ as a function of grazing profits $\pi(q)$, which are assumed to be increasing and concave in land quality, and depends on replacement costs:

$$u(q, d) = \begin{cases} \pi(q) & \text{if } d = 0, \\ \pi(\bar{q}) - \rho - f(z) & \text{if } d = 1. \end{cases}$$

Here, ρ represents the rental rate (shadow price of land) and $f(z)$ represents the cost of clearing new land, assumed to decrease in drought spells: $f(1) < f(0)$. Each season, the farmer observes (q_t, z_t) and makes a land-use decision to maximize expected net present utility $V(q, z)$, represented with the Bellman equation:

$$V(q, z) = \max \left\{ u(q, 0) + \beta \mathbb{E}_{z'} V(g(q, z'), z'),\ u(\bar{q}, 1) + \beta \mathbb{E}_{z'} V(g(\bar{q}, z'), z') \right\},$$

where $\beta < 1$ denotes the discount factor. The Bellman equation says that the farmer must compare the benefit of waiting to replace the land with the benefit of replacing the land, where these benefits consist of present-day utility and the expected future benefits of all future land use.

Consider now a farmer's policy function $d^*(q, z)$, which describes their optimal land replacement decision, given a land quality of q and a drought spell of z. Define the "advantage-of-waiting" as:

$$\begin{aligned} \Delta(q, z) &= \underbrace{\left[\pi(q) + \beta \mathbb{E}_{z'} V\left(g\left(q, z'\right), z'\right)\right]}_{\text{Wait}} - \underbrace{\left[\pi(\bar{q}) - \rho - f(z) + \beta \mathbb{E}_{z'} V\left(g\left(\bar{q}, z'\right), z'\right)\right]}_{\text{Replace}} \\ &= \underbrace{\pi(q) - \left[\pi(\bar{q}) - \rho - f(z)\right]}_{\text{Present advantage of waiting}} - \underbrace{\beta \left[\mathbb{E}_{z'} V\left(g\left(\bar{q}, z'\right), z'\right) - \mathbb{E}_{z'} V\left(g\left(q, z'\right), z'\right)\right]}_{\text{Future disadvantage of waiting}} \end{aligned}$$

The advantage-of-waiting function sufficiently describes a farmer's policy function: as long as $\Delta(q, z) > 0$, i.e., the benefits to waiting are larger than the benefits of replacing, a farmer will wait to replace their land. This can be decomposed into a present advantage of waiting and a future disadvantage of waiting. The present advantage compares the benefits of the current degraded land $\pi(q)$ to those the would receive from undegraded land $\pi(\bar{q})$ net of clearing costs $\rho + f(z)$. For high values of land quality, the present advantage will be positive and will decrease over time as the land degrades. The future disadvantage of waiting is the relative future benefit of starting the next period with less degraded land if the farmer replaces the current land. Note that $V(g(\bar{q}, z'), z') - \mathbb{E}_{z'} \geq V(g(q, z'), z')]$ since $\bar{q} \geq q$; thus, the future disadvantage of waiting is weakly positive and becomes greater over time as the land degrades. Thus, the present advantage of waiting must be large enough to offset the future disadvantage of waiting in order to delay replacing the land.

The policy function can be completely described by a threshold rule using the advantage-of-waiting function, $\Delta(q, z)$. Since $\pi(q)$ is assumed to be strictly increasing in land quality q and the cost of clearing $\rho + f(z)$ does not depend on land quality, then the value function $V(q, z)$ is strictly increasing in q, and thus, $V(g(q, z'), z')$ is strictly increasing in q. As a result, $\Delta(q, z)$ is strictly increasing in q and there exists a threshold land quality $q^*(z)$ such that we can define the farmer's policy function as:

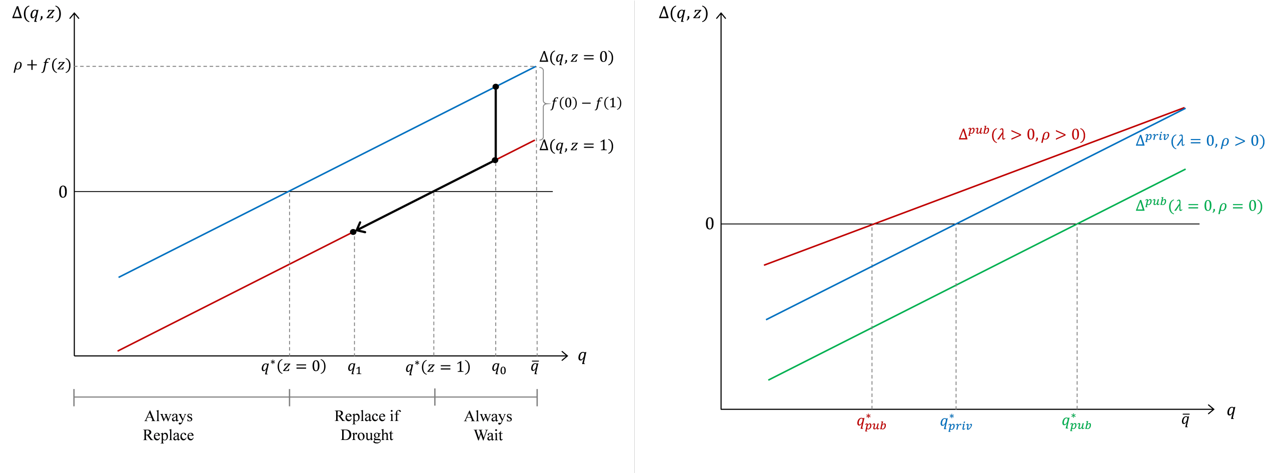
$$d^*(q, z) = \begin{cases} 0 & \text{if } q > q^*(z) \quad \text{(Wait)}, \\ 1 & \text{if } q \leq q^*(z) \quad \text{(Replace)}, \end{cases}$$

The threshold $q^*(z)$ marks the value of the land quality below which a farmer chooses to replace their land. Higher values of $q^*(z)$ represent land replacement at younger ages and with less degradation, meaning a higher rate of pasture expansion for any given drought conditions z. Larger differences in $q^*(z)$ across drought conditions, $q^*(1) - q^*(0)$, represent a larger set of land quality values that are subject to replacement with a drought, thereby representing a larger pasture expansion in response to a drought.

3.2 Tenure differences and drought effects

We can now use this theoretical framework to depict the effect that droughts and land tenure, individually, have on a farmer's land-use choice, and, finally, use our model to predict how droughts and land tenure interactions affect land-use decisions.

Let us first analyze the effects of droughts on a generic farmer's decisions. Drought affects the farmer's calculus in two ways, as illustrated in Figure 1 (a) (notice that the advantage-of-waiting curve, $\Delta(q, z)$, is upward sloping). First, it induces a "curve shift" by reducing the cost associated with replacing the current parcel of land with a new one. This can be represented by a downward shift in $\Delta(q, z)$ by $f(0) - f(1)$, and results in an increase in the quality threshold q^*. Thus, for land with quality $q(0) < q < q(1)$, a drought induces land replacement (i.e., pasture expansion) by decreasing the cost of replacement.



(a) Impacts of droughts on land replacement

(b) Private versus public land conversion

Fig. 1

Second, drought induces a "state shift" by negatively affecting the quality of land itself, and thus, the profits associated with waiting. This can be seen in the movement from q_0, the quality of land without drought $(z = 0)$, to q_1, the quality of land with drought $(z = 1)$. Thus, any land quality that is pushed over the threshold $q^*(1)$ by droughts will be replaced by the farmer. Both mechanisms, therefore, lead to a reduction in the advantage of waiting and thus increase pasture expansion.

Let us now examine the impact of different land tenure systems on land use. We can adjust two parameters to differentiate public from private land use.

First, the parameter β represents the discount factor associated with a risk-free discount rate. Suppose a farmer on public land faces some probability of being evicted. Assuming that the probability of eviction is memoryless and is independent of land quality, then we can represent a public farmer's discount factor as $\beta(1 - \lambda)$, where $0 < \lambda < 1$ is the constant hazard of being evicted.

The motivation for the eviction hazard constant λ is rooted in Brazil's contemporary land-tenure realities. Farmers on public lands are subject to state enforcement risks such as fines, embargoes, and the confiscation of means of production (Moutinho and Azevedo-Ramos 2023). They also face violent disputes with other claimants, which have historically resulted in threats, forced displacements, and even assassinations (A. Sant'Anna and Young 2010; Carrero et al. 2022). Other documented sources of tenure insecurity on public lands include the risk of arrest by state law enforcement and the reduced marketability of properties with contested or fraudulent titles[3].

[3]The appendix provides news sources illustrating: (1) arrests of land grabbers by federal police, (2) the classification of properties with disputed or fraudulent titles as "distressed assets" in the land market, and (3) violent disputes among competing claimants.

Second, the parameter ρ represents the opportunity cost of converting forested land to pasture. For private land owners with scarce land, we can think of $\rho > 0$, either due to the shadow value of land that is already owned by the farmer, or the rental/purchase price of land on the private market. For public land users, expanding into public forest has no such opportunity cost, and thus, $\rho = 0$.

We are now in a position to conduct some comparative statics to see how land use differs across private and public land. The comparative statics for a given drought spell z are depicted in Figure 1 (b). With a positive probability of being evicted, public farmers have a smaller discount factor. This has the effect of rotating $\Delta(q, z)$ up around the point at $\bar{q}$, illustrating that the advantage of waiting for public farmers is higher than for private farmers for all $q < \bar{q}$ because they discount more the future disadvantage of waiting. Thus, $q^*_{pub}(z) < q^*_{priv}(z)$. In other words, public farmers are less likely to incur a costly investment in land clearing if there is some probability that they will be evicted and unable to reap the rewards. Thus, the probability of eviction incentivizes public farmers to hold onto land longer than private farmers, allowing the land to degrade more. Private farmers, on the other hand, are more likely to replace land and expand pasture, all else equal.

In contrast, with no opportunity cost associated with land $\rho = 0$, public farmers have a constant lower advantage of waiting than private farmers, regardless of the land quality. This has the effect of shifting $\Delta(q, z)$ down in a parallel fashion. As a result, $q^*_{pub}(z) > q^*_{priv}(z)$: the relative ease of converting new land into pasture incentivizes public farmers to hold on to land for shorter periods of time and expand pasture at greater rates than private farmers. Overall, whether public farmers expand pasture at greater rates than private farmers depends on whether the "opportunity-cost-of-land" effect dominates the "probability-of-getting-evicted" effect.

Based on these results, we can finally analyze the differential drought effects across public and private farmers. Figure 2 depicts the comparative statics predictions, assuming that private and public farmers have the same baseline pasture conversion rates, $q^*_{pub}(0) = q^*_{priv}(0)$, for simplicity.

First, the drought-induced "curve-shift" effect from reducing the clearing cost of land shifts the advantage-of-waiting curve down. This "curve-shift" effect is represented in 2 by the difference between $q^*(1)_{priv}$ and $q^*(0)_{priv}$ for private lands, and the difference between $q^*(1)_{pub}$ and $q^*(0)_{pub}$ for public lands. Since public and private farmers experience the same reduction in land-clearing costs, $\Delta(q, z)$ shifts down vertically by the same amount for both types of farmers. However, $\Delta(q, z)$ is flatter for public farmers since they discount the future disadvantage of waiting more than private farmers, resulting in a larger horizontal shift in $\Delta(q, z)$ for public farmers. Thus, $q^*(1)_{pub} - q^*(0_{pub}) > q^*(1)_{priv} - q^*(0)_{priv}$: more land will be replaced by public farmers than private farmers in response to the drought, regardless of the baseline pasture conversion rates.

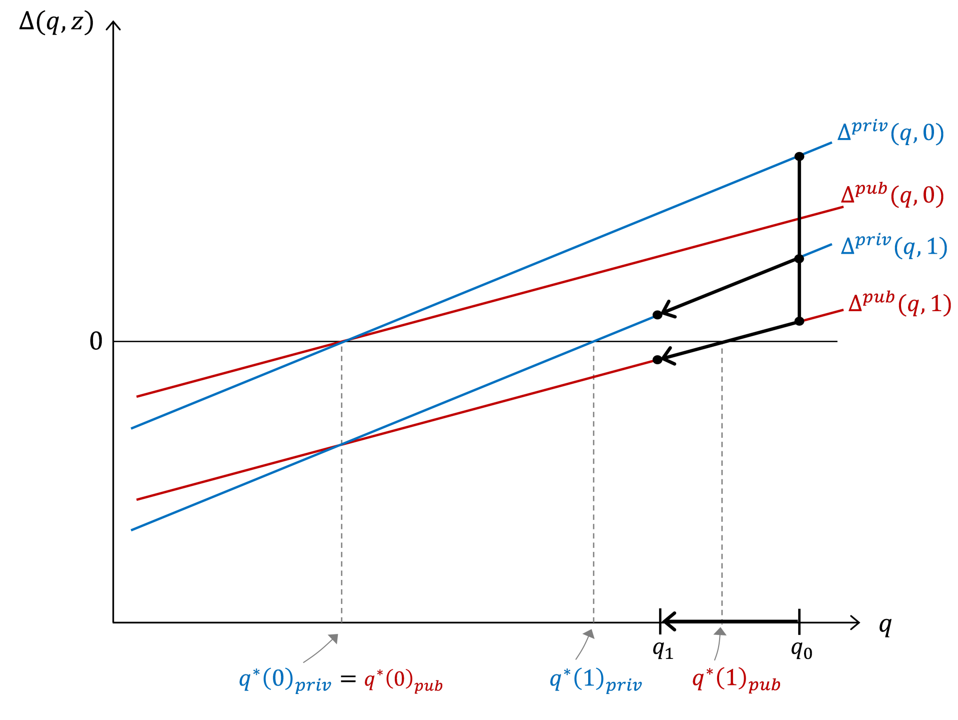


Fig. 2: Drought effects: Public vs. Private

The reduction in deforestation costs during droughts matters relatively more for farmers on public lands, because the risk of eviction makes them reluctant to invest in costly clearing methods that might not yield returns. By reducing the investment cost, droughts shift the balance between the "opportunity-cost-of-land" effect and the "probability-of-getting-evicted" effect towards the former, and therefore, public land farmers react more to droughts.

Second, the "state-shift" effect reduces the quality of land itself. Land quality that is pushed over the threshold $q^*(1)$ by drought will be replaced by the farmer. This is depicted in Figure 2 by the movement from $q^*(0)$, the quality of land without the drought, to $q^*(1)$, the quality of land with the drought. As depicted, the drought induces enough degradation in land quality to justify replacing the land for the public farmer, but not the private farmer. Note that while the curve shift effect is determined by the slope of $\Delta(q, z)$, the size of the state-shift effect will be the same for both public and private farmers: any land quality q that lies within $q(0) - q(1)$ of the threshold $q^*(1)$ will be induced to being replaced. Thus, as long as droughts are assumed to affect land quality equally across land types, the "state-shift" effect will be the same.

Overall, the model underscores that differences in land tenure regimes critically mediate farmers' responses to drought. In the absence of climatic shocks, relative rates of pasture expansion on private versus public lands depend on whether the higher opportunity cost of waiting (which encourages more expansion by private farmers) outweighs the higher opportunity cost of conversion (which restrains it). Droughts, however, introduce additional forces that shift this balance.

The "curve-shift" effect, operating through reductions in clearing costs, induces a larger response on public lands, where tenure insecurity typically discourages costly long-term investments. By contrast, the "state-shift" effect, operating through reductions in land quality, generates symmetric responses across tenure types as long as drought-induced degradation is uniform. Taken together, these mechanisms imply that droughts systematically amplify pasture expansion on public relative to private lands, even when baseline expansion rates differ, thereby highlighting how climate shocks and tenure insecurity interact to shape land-use dynamics.

Our simple model highlights the interaction of land tenure and climate shocks in shaping land-use responses, but it omits other relevant dynamics. One important example is investment in pasture restoration. Because farmers on public lands face weaker incentives to undertake costly, long-term improvements, they are less likely to restore degraded pasture. A higher eviction hazard (λ) further discourages such investments, as the risk of losing access to the land reduces the expected returns from long-term restoration. When drought reduces the cost of clearing new land, these farmers may therefore favor expansion into previously uncleared areas rather than investing in restoration.

4 Empirical Framework

Our empirical strategy links the theoretical predictions of Section 3 to observed land-use behavior across public and private land. We test whether droughts induce greater pasture expansion on public than on private lands by comparing their responses within the same spatial and climate context. Our approach exploits plausibly exogenous variation in the timing and severity of droughts across 0.5° grid cells in the Brazilian Amazon and contrasts land-use changes between public and private areas inside each cell. This within-cell, differences-in-differences framework controls for all time-invariant local characteristics and common shocks, isolating how land tenure mediates farmers' responses to droughts. In line with the conceptual model, we interpret stronger drought responses on public lands as evidence that droughts systematically amplify pressure for pasture expansion on public relative to private land.

4.1 Data

Our analysis combines high-resolution spatial data on land use, land tenure, and climate to examine how droughts affect pasture expansion across public and private land in the Brazilian Amazon. We merge annual land-cover classification from the MapBiomas Project (Souza et al. 2020) with land-tenure information from the Atlas of Brazilian Agriculture and drought indicators from the Standardized Precipitation-Evapotranspiration Index (SPEI). These datasets together provide consistent coverage of forest, pasture, and climatic conditions across both public and private lands from 2003 to 2021. The resulting grid-cell level panel allows us to track changes in pasture area and quality through time and to relate them to exogenous variation in drought conditions within the same local environment.

MapBiomas uses Landsat satellite images and machine learning algorithms to classify images into land uses, including pasture, based on variables such as spectral color bands and indices of vegetation moisture. For pasture predictions, MapBiomas only uses data from the wet season, as pasture is vulnerable to climate conditions. In the Amazon, this class may occur in recently deforested areas, even before farming activities have started. Areas of natural pasture are predominantly classified as grassland or wetland, separated from manmade pasture. MapBiomas also provides data on the quality of the pasture, which is classified as healthy, moderately degraded, or severely degraded based on normalized vegetation moisture information. The data is available at a refined spatial resolution of 30m x 30m.

The SPEI is a relative measure of drought based on the water balance of a reference period, indicating water surpluses (positive values) or deficits (negative values) based on monthly precipitation and potential evapotranspiration. It represents the number of standard deviations from the normally accumulated climatic water balance for the respective location and time of the year, and is available at the 0.5° level. Our analysis considers the average drought conditions from January to December. We also explore seasonal drought conditions when investigating the possible mechanisms of droughts and differential pasture expansion. [4]

The Atlas of Brazilian Agriculture, created using a methodology described by Sparovek et al. 2019, integrates public datasets of land tenure from different sources. Therefore, it can be considered a snapshot of Brazilian land tenure as of 2019. It addresses any overlap between private and public land using a hierarchical model that classifies the land tenures based on a priority system.[5] The Atlas of Brazilian Agriculture also provides information on property registration under the Rural Environmental Registry (CAR), which was instituted in 2012 by the Brazilian government to facilitate the environmental regulation of rural properties since many did not comply with Brazil's forest legislation (Garrett et al. 2021). CAR consists of a self-declaratory registry where landowners input their farms' geolocated boundaries into the system, as well as the location of the farm's forest reserve. The Atlas of Brazilian Agriculture considers two types of land registered in CAR: "CAR

[4]The SPEI index compares the drought conditions in a given number of months to the historical average drought conditions over those months. For instance, SPEI-3 May 2009 compares the drought conditions from March to May 2009 to the historical average drought conditions for those months. The primary drought variable we use in this paper is SPEI-12 December.

[5]For instance, private land under CAR has less priority than indigenous land because indigenous rights are established in the Brazilian Constitution.

premium" consists of properties that are likely to be legitimate private lands and have less than 5% of their area overlapped with other properties; "CAR poor," in turn, are properties registered in CAR with more than 5% of their area overlapped with other properties. CAR-poor properties are likely to represent the illegitimate occupation of public lands, and thus, we removed such lands from our analysis to better distinguish between public and private lands. Robustness tests show that including those lands in our analysis does not change our results.

Table 1 presents descriptive statistics for the years 2003 to 2021. On average, cells experienced droughts 28% of the time, and 65% of the land area is considered public. On average, private lands have a higher percentage of pastureland than public lands, with the former having a 20% pasture saturation rate, compared to 12% for the latter. Figure 3 shows the average occurrence of droughts over our studied area, where we can observe a higher drought rate in the southern and western portions of the map.

Tab. 1: Basic Summary Statistics

	Mean	SD	Min	Max	N
Dry	0.2840	0.4509	0.00	1.00	50620
Share of Public Lands	0.6555	0.3259	0.00	1.00	50620
Pasture Growth (int)	0.0218	0.1100	-1.14	3.50	50600
Private Pasture Growth	0.0276	0.1214	-1.13	3.50	22580
Public Pasture Growth (int)	0.0171	0.0996	-1.14	1.85	28020
Pasture Growth (tot)	0.0349	0.1374	-1.13	4.36	50600
Public Pasture Growth (tot)	0.0408	0.1488	-1.13	4.36	28020
Pasture Saturation	0.1554	0.2379	0.00	1.00	50600
Private Pasture Saturation	0.1985	0.2457	0.00	1.00	22580
Public Pasture Saturation	0.1206	0.2254	0.00	1.00	28020
Pasture Growth From Forest	0.8690	0.2288	0.00	1.00	39885

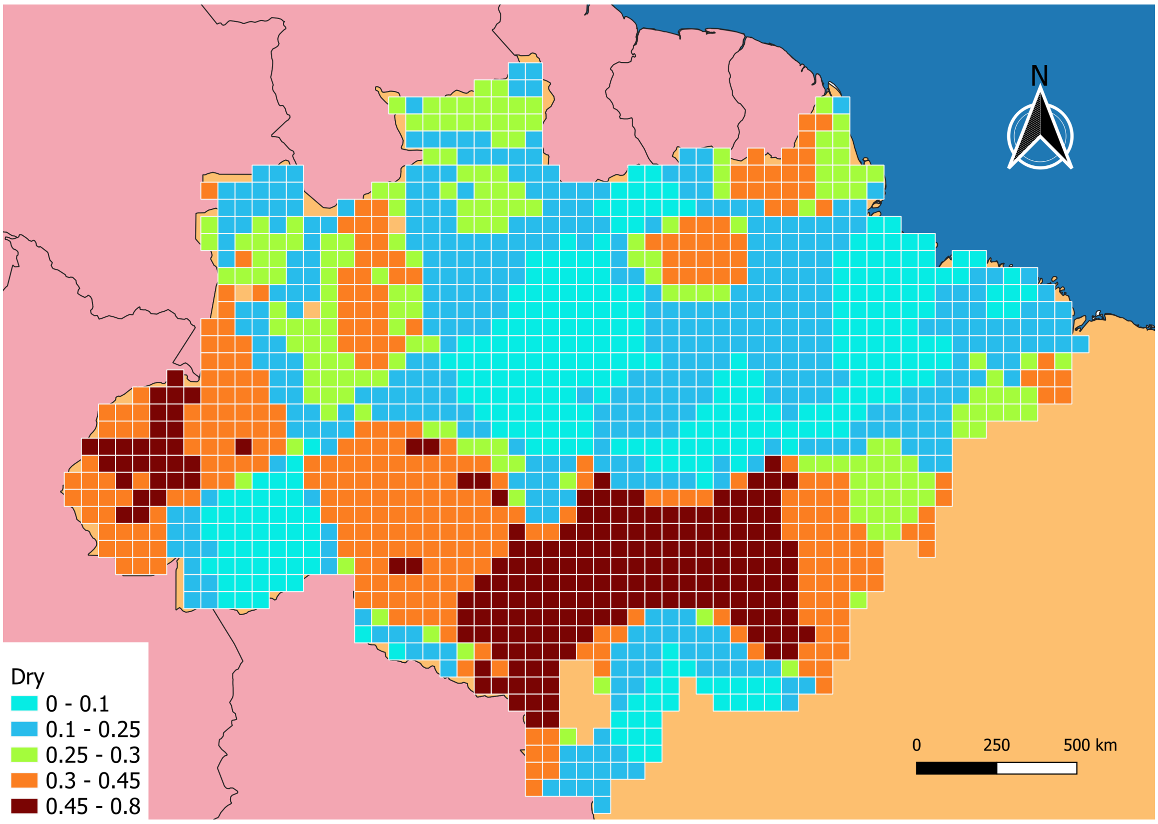
Pasture Growth (int) corrresponds to $\widehat{PGint\_ts}$. Pasture Growth (tot) corrresponds to $\widehat{PGint\_tot}$

4.2 Measuring Intensive and Extensive Pasture Growth

The primary goal of this paper is to investigate how ranchers on private lands react differently to drought than ranchers on public lands, particularly those who have already settled on public lands at the time of drought, as opposed to those who invade public lands in response to a drought. Unfortunately, we cannot observe the boundaries of public land farms since these ranchers are settled irregularly. Therefore, we cannot use public and private farms as our unit of analysis to distinguish between expansion and invasion of public land.

To address this issue, we use information on land tenure and land use to distinguish between in-

Fig. 3: Average Drought Conditions from 2003 to 2022



tensive and extensive pasture growth. Intensive pasture growth measures the expansion of pasture on land adjacent to previously established pastureland, reflecting expansion by existing ranchers, while extensive pasture growth captures the creation of new pasture on previously unoccupied lands. Accordingly, for each 0.5° cell and year t, we classify growth in pastureland that intersects with pastureland in $t-1$ as intensive and growth in pastureland that does not intersect with pastureland in $t-1$ as extensive (Figure 4).

Using these definitions, we construct two aggregate measures of pasture growth for each tenure type: (1) *Intensive-pasture growth* ($PGint$), which includes all new pasture on private lands and only extensive growth on public lands; and (2) *Total-pasture growth* ($PGtot$), which includes all observed new pasture on private and public land. Assuming that all private lands are already settled and that extensive growth in public lands results from new ranchers invading unoccupied public lands, the measure $PGint$ captures the effect of droughts on already settled ranchers. In contrast, the measure $PGtot$ captures the total effect of droughts on pasture expansion, including the invasion of public land. We provide additional details on the interpretation these measures under different assumptions in the Appendix.

We can observe that pasture growth is larger on private lands when we consider just the observed intensive growth in public lands, but it is larger in public lands when we consider the total

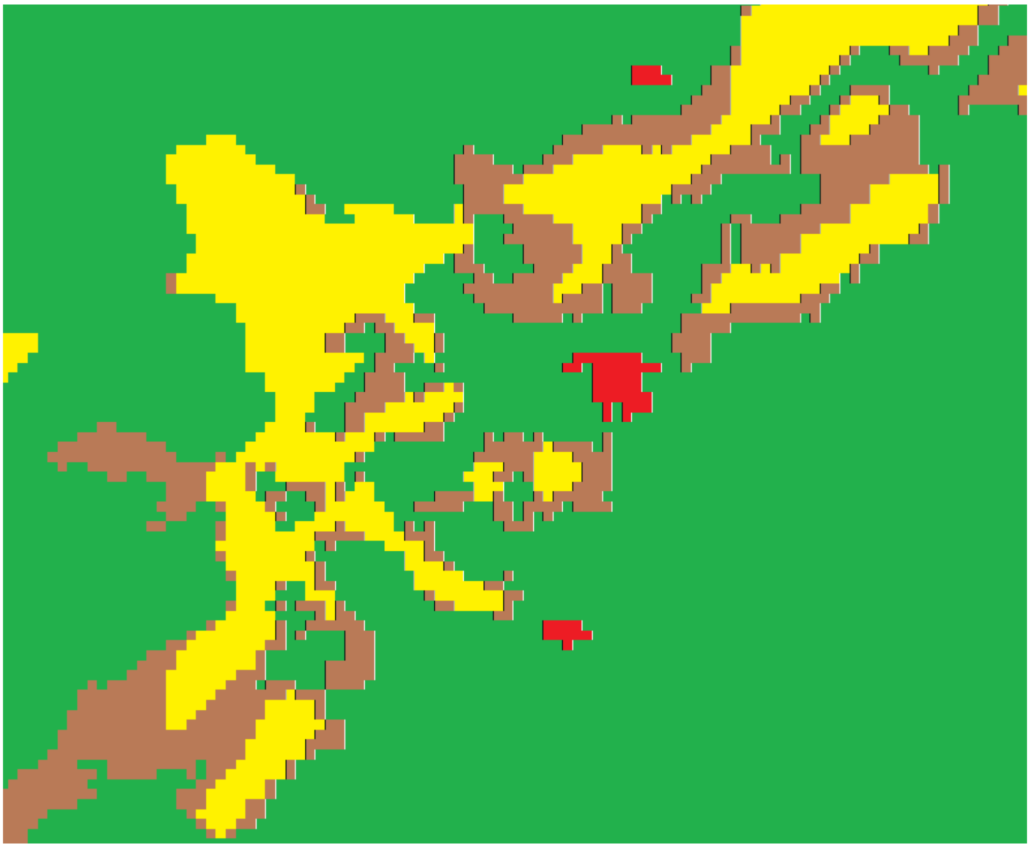
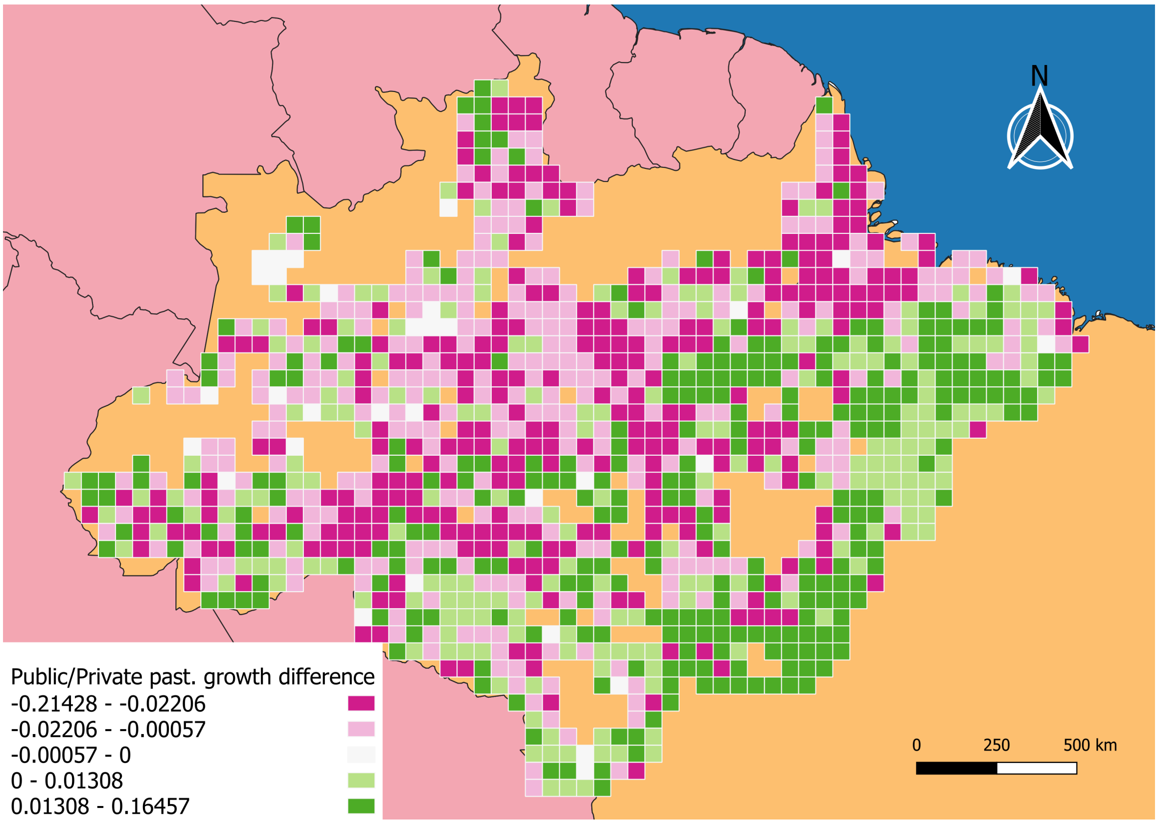


Fig. 4: **Observed Intensive and Extensive Pasture Growth.** Yellow represents pasture in year $t-1$, brown represents observed intensive pasture growth in year t, and red represents observed extensive pasture growth in year t.

Fig. 5: Average Difference between Public and Private Pasture Growth Rates



Considering only observed intensive growth for public lands

growth in those lands (Table 1. Figure 5, in turn, shows the average difference between public and private pasture growth rates over the studied period, considering only the observed intensive growth for public lands. Green cells observed more public land pasture growth, and pink cells observed more private land pasture growth. Generally, average private land pasture growth is more concentrated in the northern portion of the map, where average drought conditions were milder.

4.3 Estimation Strategy

We estimate the differential response of farmers to drought in public and private lands within the same cell using the following model:

$$y_{cit} = \beta_0 + \beta_1 Dry_{ct} + \beta_2 Priv_{ci} + \beta_3 Dry_{ct} \cdot Priv_{ci} + \gamma_c + \delta_t + \varepsilon_{cit}, \tag{1}$$

where y_{cit} denotes pasture growth in cell c, year t, and tenure i, Dry_{ct} is a dummy variable that equals to one if cell c experiences a drought in year t, and $Priv_{ci}$ is a dummy variable that equals one if land tenure i in cell t is private. Cell- and year-fixed effects are represented by γ_c and δ_t, respectively. We measure pasture growth as the difference in logs of pastureland area in years t and

$t-1$ using either the intensive or total pasture-growth measures, $PGint$ and $PGtot$. We define $Dry_{ct} = 1$ when a cell's SPEI is one standard deviation below its historical mean. Thus, we estimate whether drought shocks in year t induce pasture expansion in public and private lands, considering a cell's usual climate.

Our coefficients of interest are β_0 through β_3. The constant β_0 captures pasture expansion on public land in non-drought years, while $\beta_0 + \beta_2$ captures pasture expansion on private land in non-drought years. Thus, β_2 represents differential baseline pasture expansion between farmers on private and public land, providing a direct test of whether the "opportunity-cost-of-land" effect dominates the "probability-of-getting-evicted" effect from our conceptual model. Similarly, the coefficient β_1 indicates how farmers respond to drought years on public land, while $\beta_1 + \beta_3$ indicates how farmers respond to droughts on private lands. Thus, β_3 represents how farmers on private land respond differently to droughts than those on public land, providing a direct test of the main hypothesis from our conceptual model.

The cell fixed effects control for all factors common to private and public farms within the same cell. In essence, it allows us to compare how differently public farmers react to droughts compared to private farmers within the same cell.

We exploit quasi-experimental variation in droughts to causally identify the differential impact of droughts on public and private pasture expansion. To interpret our estimates as causal, there must be no time-varying unobserved variables correlated with the difference between public and private pasture expansion and droughts. This assumption would be violated if droughts are more (or less) likely to occur in cells with differential trends in public and private pasture expansion.

As we compare pasture growth across public and private lands within cells, we trimmed our sample to only include cells with more than one square kilometer of pasture in both public and private lands. In doing so, we avoid distortions from percentage changes in places with small baseline pastureland and ensure we are using cells with comparable public and private pastureland. Similarly, we only included cells with at least 10% of forest cover in 2001 in both land tenure types, ensuring that pastureland has not saturated an entire cell, thereby preventing pasture expansion. We cluster our standard errors to account for spatial autocorrelation, clustering neighboring cells into groups of four, equalling 100km by 100km squares. We also use Conley spatial standard errors as a robustness exercise, with the code made available by Hsiang 2010.

5 Results

Our main results are presented in Figure 6 and Table S.2. We estimate that droughts increase the growth rate of intensive pasture expansion on public lands by 0.88 percentage points, an increase of roughly thirty percent over the baseline growth rate in non-drought years. In contrast, we find a small, insignificant increase of 0.04 percentage points in the growth rate of intensive pasture expansion on private lands in response to droughts. This 0.84 percentage-point difference between public and private land is statistically significant at the 1% significance level and robust across both measures of pasture growth ($PGint$ and $PGtot$). The results derived from using $PGint$ as our outcome variable predict a lower baseline expansion rate for public lands under no drought conditions and a larger response to dry conditions compared to the results using $PGtot$. Considering the intensive pasture expansion variable, the point estimates predict that public land farmers expand their pasture by 3.9% during droughts, compared to 3% in non-drought years. Farmers in private lands expand their pasture, on average, by 3.2% regardless of the dry conditions.[6]

Consistent with our conceptual model, these results provide evidence that farmers on public lands react more to droughts than those on private lands. However, as our conceptual model highlights, there are several possible explanations for this result, including fewer incentives to invest and manage the land sustainably, better opportunities to deforest and claim lands during droughts, or different socioeconomic characteristics across public land and private land farmers. We explore some of these mechanisms in the next section.

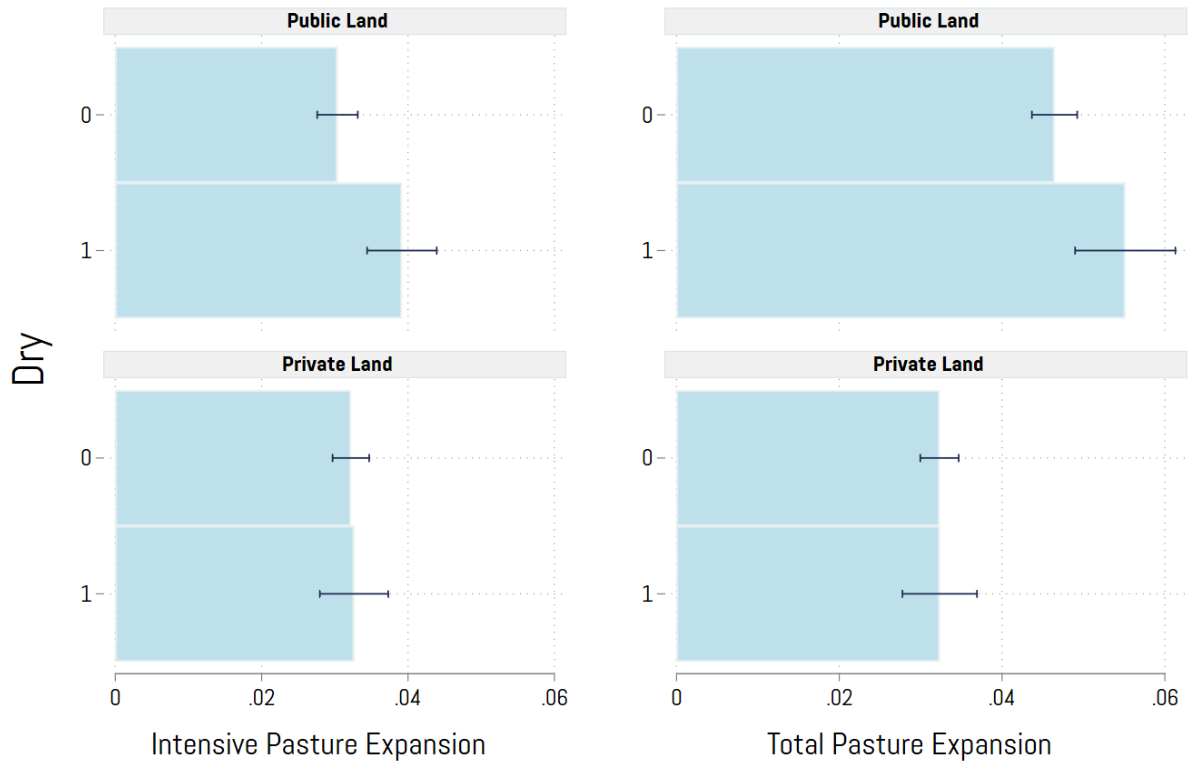
5.1 Mechanisms

5.1.1 Pasture Degradation

In this section, we explore some possible mechanisms for the observed difference in pasture expansion across public and private land farmers during drought years. First, we investigate the role of pasture degradation between droughts and farmers' land use response. Economic theory postulates that negative productivity shocks, such as droughts, increase the value of outside option activities and thus reduce investment in pastureland. However, in a context of an extensive production system with low capital input and abundant land, drought-induced pasture degradation may lead ranchers to look for new lands.

[6] As we discuss in the Appendix, some intensive growth on public lands may be mistakenly classified as extensive growth. In this case, the increase in pasture expansion on public lands using $PGint$ is underestimated. Since expansion growth on public lands using $PGint$ is larger than when using $PGtot$, it must be that both measures underestimate the pasture growth in public lands (See third line of Table S.1). Further, since the effect of droughts on intensive expansion on public land is greater than the effect on private land, both measures also underestimate the difference in pasture expansion growth between public and private lands.

Fig. 6: Droughts and pasture expansion



Our conceptual framework in Section 3 predicts that land degradation should trigger similar responses among farmers on both public and private lands. However, the model does not explicitly account for investment in pasture restoration. Since farmers on public lands face weaker incentives to restore degraded pastures, they may respond more strongly to degradation by expanding into new areas instead. To examine these dynamics, we first analyze how droughts influence pasture degradation rates, and then assess how degradation, in turn, affects pasture expansion under each land tenure regime.

Table 2 shows the impacts of droughts on pasture degradation rate [7]. Droughts increase the pasture degradation rate by 0.5 percentage points, with no different impacts across public and private land pastures. Pasture on private lands, on average, suffers 0.6 percentage points more degradation each year, which could be a result of private farms being older or public land ranchers abandoning

[7] The Mapbiomas dataset includes data on pasture quality and classifies the pasture into three categories: good, medium, and degraded. We calculate the pasture degradation rate as

$$PDeg_{c,i,t} = \frac{p^{gm}_{c,i,t} + p^{gd}_{c,i,t} + p^{md}_{c,i,t}}{p^{G}_{c,i,t-1} + p^{M}_{c,i,t-1}}$$

where $p^{gm}_{c,i,t}$ is the pasture area classified as good in $t-1$ and medium in t, $p^{gd}_{c,i,t}$ is the pasture area classified as good in $t-1$ and degraded in t, $p^{md}_{c,i,t}$ is the pasture area classified as medium in $t-1$ and degraded in t, $p^{G}_{c,i,t-1}$ is the total pasture area classified as good in $t-1$, and $p^{M}_{c,i,t-1}$ is the total pasture area classified as medium in $t-1$.

the land faster. However, private land pasture could face more visible degradation if those farmers employ a pasture rotation system. In this case, other parts of their pasture area would also regenerate more. The second column of table 2 includes the pasture regeneration rate [8] as a control, and the results do not change significantly.

Tab. 2: Droughts and pasture degradation

	Pasture degradation	
Dry	0.0048***	0.0039***
	(0.0013)	(0.0012)
Private Land	0.0062***	0.0061***
	(0.0011)	(0.0011)
Dry*Private Land	-0.0005	-0.0009
	(0.0014)	(0.0014)
Constant	0.0417***	0.0527***
	(0.0007)	(0.0010)
Observations	26814	26557
R^2	0.4670	0.5062
Regeneration Past. Controls	No	Yes

Standard errors in parentheses
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

Next, we examine how pasture degradation influences pasture expansion across different land tenure types. Table 3 presents these results. For both the full sample and the public land subsample, the estimated effect of droughts on pasture expansion declines once we control for the degradation rate, suggesting that degradation operates as a mediating mechanism. In all samples, higher degradation rates are associated with greater pasture expansion, indicating that ranchers tend to clear new areas when their existing pastures deteriorate.

This relationship is slightly stronger for public lands: a one–percentage–point increase in the degradation rate leads to an expansion of approximately 0.31% in pasture area on public lands, compared to 0.27% on private lands [9]. These findings are consistent with the hypothesis that ranchers operating on public lands have weaker incentives to invest in pasture restoration and, consequently,

[8] Pasture regeneration rate is calculated analogously to the pasture degradation rate. It is calculated as

$$PReg_{c,i,t} = \frac{p^{mg}_{c,i,t} + p^{dg}_{c,i,t} + p^{dm}_{c,i,t}}{p^{M}_{c,i,t-1} + p^{D}_{c,i,t-1}}$$

where $p^{mg}_{c,i,t}$ is the pasture area classified as medium in $t-1$ and good in t, $p^{dg}_{c,i,t}$ is the pasture area classified as degraded in $t-1$ and good in t, $p^{dm}_{c,i,t}$ is the pasture area classified as degraded in $t-1$ and medium in t, $p^{M}_{c,i,t-1}$ is the total pasture area classified as medium in $t-1$, and $p^{D}_{c,i,t-1}$ is the total pasture area classified as degraded in $t-1$.

[9] This difference is statistically significant, as reported in Table S.3 in the Appendix.

rely more on expanding their grazing area when degradation occurs.

Finally, the effect of droughts on public intensive pasture expansion remains significant when we control for degradation, but it disappears when we consider the total pasture expansion. This result is evidence that there are still other mechanisms through which droughts affect intensive public pasture expansion, but other mechanisms likely affect extensive expansion in the opposite direction.

Tab. 3: Droughts, degradation and pasture expansion

	Intensive Pasture Expansion					
	(1)	(2)	(3)	(4)	(5)	(6)
Dry	0.0046	0.0032	0.0092**	0.0074**	0.0001	-0.0008
	(0.0030)	(0.0029)	(0.0036)	(0.0034)	(0.0030)	(0.0029)
Degrad. rate		0.3158***		0.3159***		0.2656***
		(0.0259)		(0.0376)		(0.0387)
	Total Pasture Expansion					
	(1)	(2)	(3)	(4)	(5)	(6)
Dry	0.0044	0.0031	0.0086*	0.0069	0.0001	-0.0008
	(0.0034)	(0.0033)	(0.0044)	(0.0043)	(0.0030)	(0.0029)
Degrad. rate		0.2707***		0.3050***		0.2656***
		(0.0298)		(0.0418)		(0.0387)
Observations	26814	26814	13389	13389	13395	13395
Sample	Full	Full	Public	Public	Private	Private

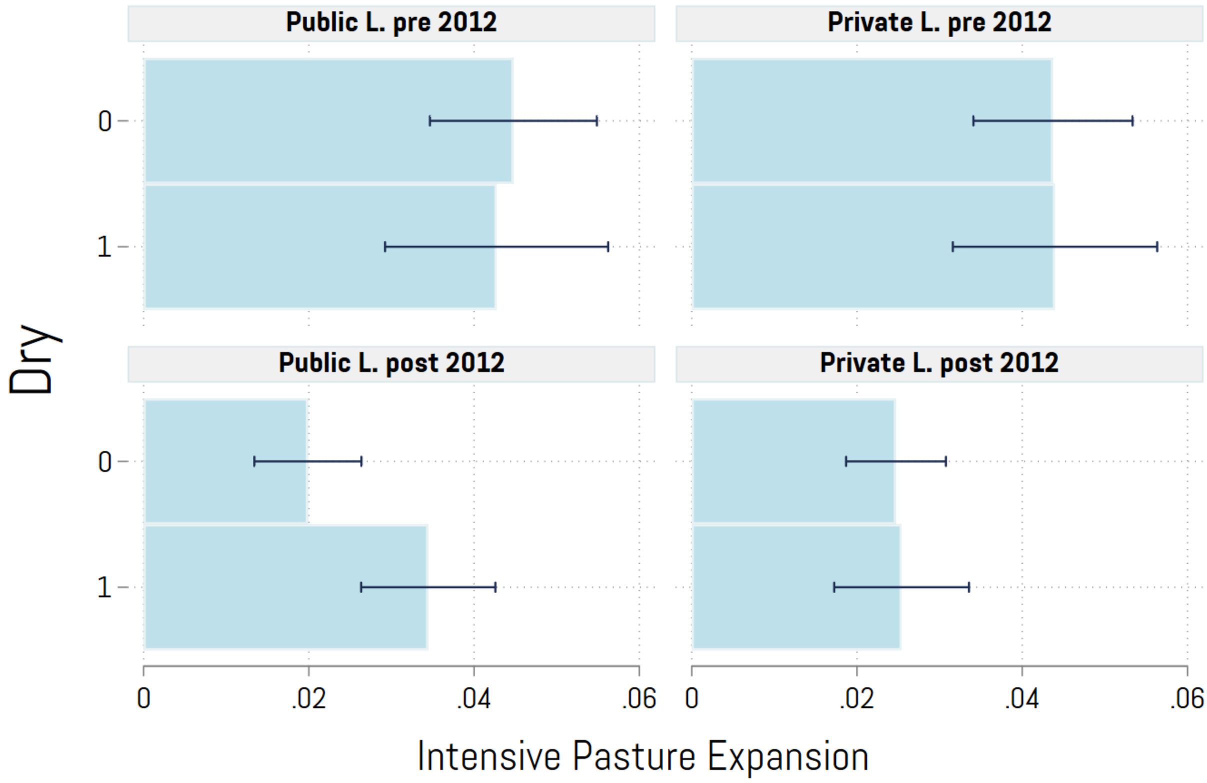
Standard errors in parentheses
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

5.1.2 Loosening of environmental regulation

In 2012, the Brazilian Congress enacted a new forest code that, among other measures, grandfathered in irregular properties in public lands established before 2008. A. A. Sant'Anna and Costa 2021 studied the effects of this loosening in environmental regulation and found evidence that this bail-out created incentives for farmers to violate the law in the expectation of receiving future amnesties. At the same time, Brazilian law makes it easier to acquire land tenure if the land is being used productively (L. J. Alston, Gary D Libecap, and Mueller 1999). Therefore, by making the vegetation more flammable and reducing the cost of deforestation, in the context of post-2012 loosening of environmental regulation, droughts may create the perfect opportunity for farmers in public lands to expand their productive area and their claim to the land.

We test the hypothesis that there was a break in public farmers' response to drought before and after 2012. Figure 7 shows the results of a regression that allows the impacts of droughts on

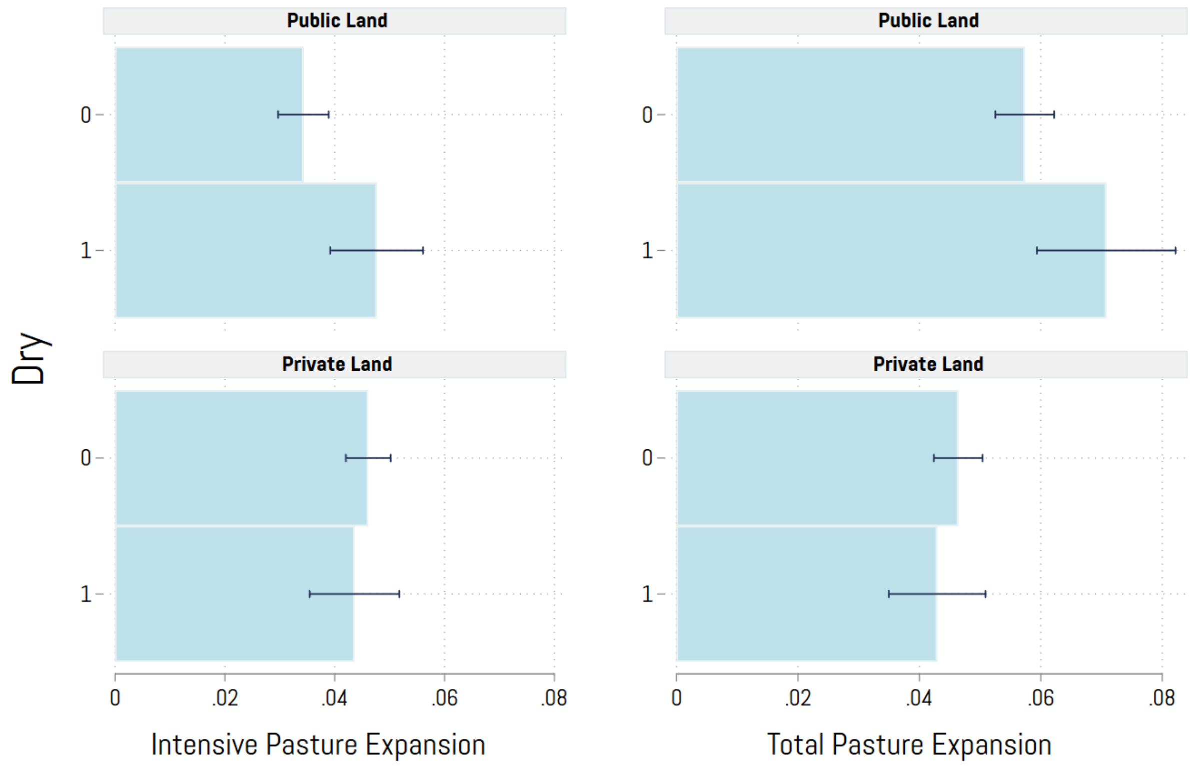
Fig. 7: Droughts and intensive pasture expansion pre and post-2012



intensive pasture expansion to differ before and after 2012. We observe that droughts have little to no effect on intensive public pasture expansion in the pre-2012 period, whereas it has a significant impact in the post-2012 period. Droughts seem to have no impact on private expansion in either period. These results are robust when we use total pasture expansion as our independent variable. Figure S.1 in the Appendix presents the total pasture expansion results.

These results are consistent with the hypothesis that the relaxation of environmental regulations after 2012 created stronger incentives for farmers on public lands to expand their properties, particularly by taking advantage of drier conditions to clear new areas. However, we cannot conclusively attribute the observed differences in responses across periods solely to this policy change, as other factors also varied between the two periods—most notably, the frequency of droughts. The earlier period (2003–2011) experienced milder climatic conditions, with cells under drought only 17.5% of the time, whereas in the later period (2012–2022), drought frequency increased to 37%. Consequently, part of the observed shift could reflect differences in climatic variability rather than regulatory changes alone, or limited statistical power in the earlier period due to fewer drought events.

Fig. 8: Droughts and pasture expansion in the more humid subsample

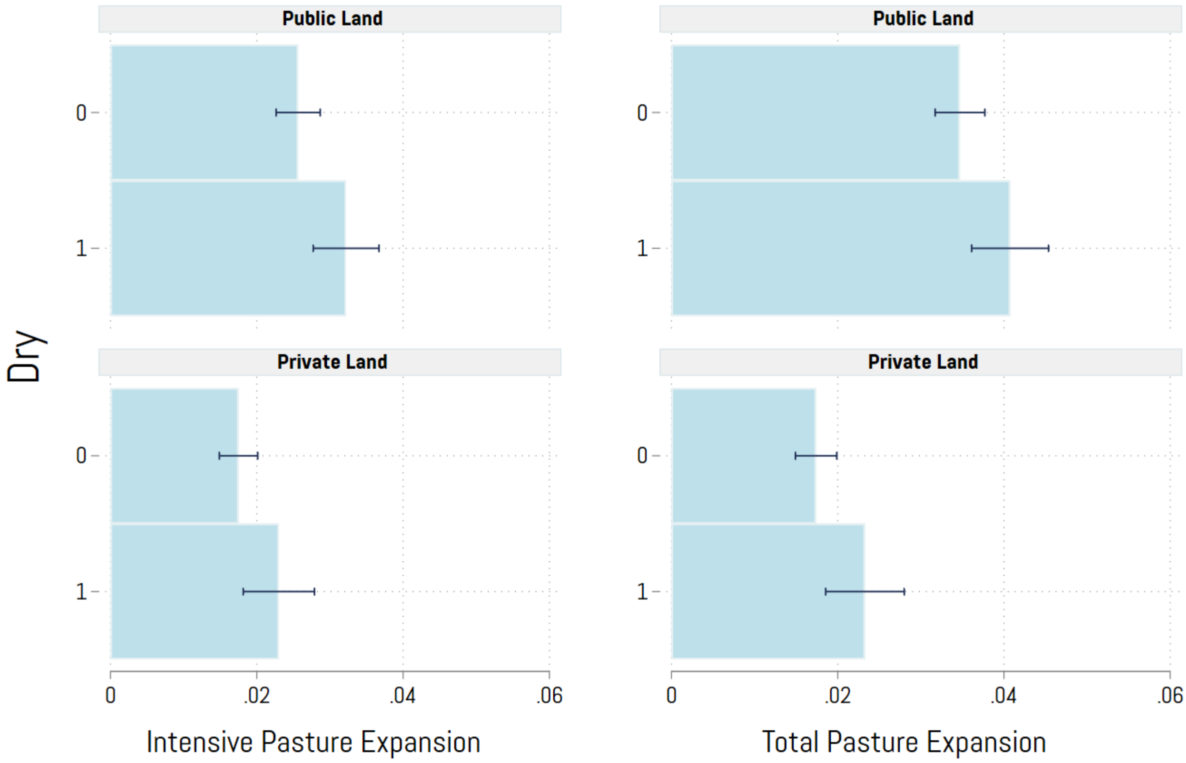


5.2 Spatial Heterogeneity by Humidity

We also examine whether the effect of droughts on pasture expansion differs between public and private lands across regions with distinct baseline climatic conditions. The drought variable, based on the Standardized Precipitation–Evapotranspiration Index (SPEI), measures how climate conditions deviate from the historical average for each location and month. In this section, we test whether the impact of droughts varies between relatively humid and drier regions within our sample. To characterize baseline climate conditions, we use the Climate Hazards Group InfraRed Precipitation with Stations (CHIRPS) dataset (Funk et al. 2015) to compute each cell's average annual rainfall for the period 1981–2000. We then divide the sample into two groups—below- and above-median historical rainfall—where the median across all cells is 2,022 mm.

Using the same specification as in Equation 1, we analyze how farmers on public and private lands respond to droughts within each climatic subsample. Figure 8 reports the results for the more humid regions. The patterns closely resemble those observed in Figure 6, which includes the full sample. In humid areas, droughts lead to significant pasture expansion on public lands, both in terms of intensive and total expansion measures, while farmers on private lands show no discernible response to drought conditions.

Fig. 9: Droughts and pasture expansion in the drier subsample



The results for the drier subsample, shown in Figure 9, reveal a different pattern. Farmers on public lands continue to expand their pasture area during drought years; however, in these regions, private land farmers also become responsive to droughts. As a result, the difference in responses between public and private landholders is no longer statistically significant. One possible explanation is that the drier baseline conditions reduce incentives for private land ranchers to invest in pasture restoration. In predominantly rainfed systems, maintaining productivity under dry conditions can be costly, so even private landholders may find it more profitable to expand into new areas rather than invest in existing pastures.

Overall, the results indicate that most observed differences in private and public land ranchers' behavior come from the more humid subsample. While ranchers in public lands always respond to droughts, ranchers in private lands respond only in the drier subsample. This difference in private ranchers' behavior across the subsamples could indicate that higher rainfall levels give ranchers more incentives to invest in the land and restore degraded pastures. However, ranchers in public lands still react to droughts in the humid subsample, as the eviction possibility still discourages any investment in the land. The results indicate that the difference in responses to droughts of ranchers across both tenures also depends on the baseline climate conditions of the area.

5.3 Robustness Exercises

5.3.1 Alternative Outcome Variable: Forest to Pasture Conversion

In this section, we test whether our results are robust using another variable as our outcome of interest. Specifically, we use the forest-to-pasture conversion as our alternative outcome variable. The Mapbiomas dataset allows us to observe forested pixels in year $t-1$ converted into pasture in year t. As before, we consider intensive forest-to-pasture conversion if the new pasture area is spatially connected with the pasture area in $t-1$, and total forest-to-pasture conversion accounts for all observed forest-to-pasture conversion.

Table 4 presents the results of estimating equation 1 with our outcome of interest being forest to pasture conversion in square km. The results are similar for both the intensive and total forest-to-pasture conversion variables. Overall, the coefficients have the same sign as our pasture expansion specification but are measured with more noise. This noise could come from measurement errors derived from using remote sensing imagery for land use classification, as described by Alix-Garcia and Millimet 2023.

Tab. 4: Droughts and forest to pasture conversion

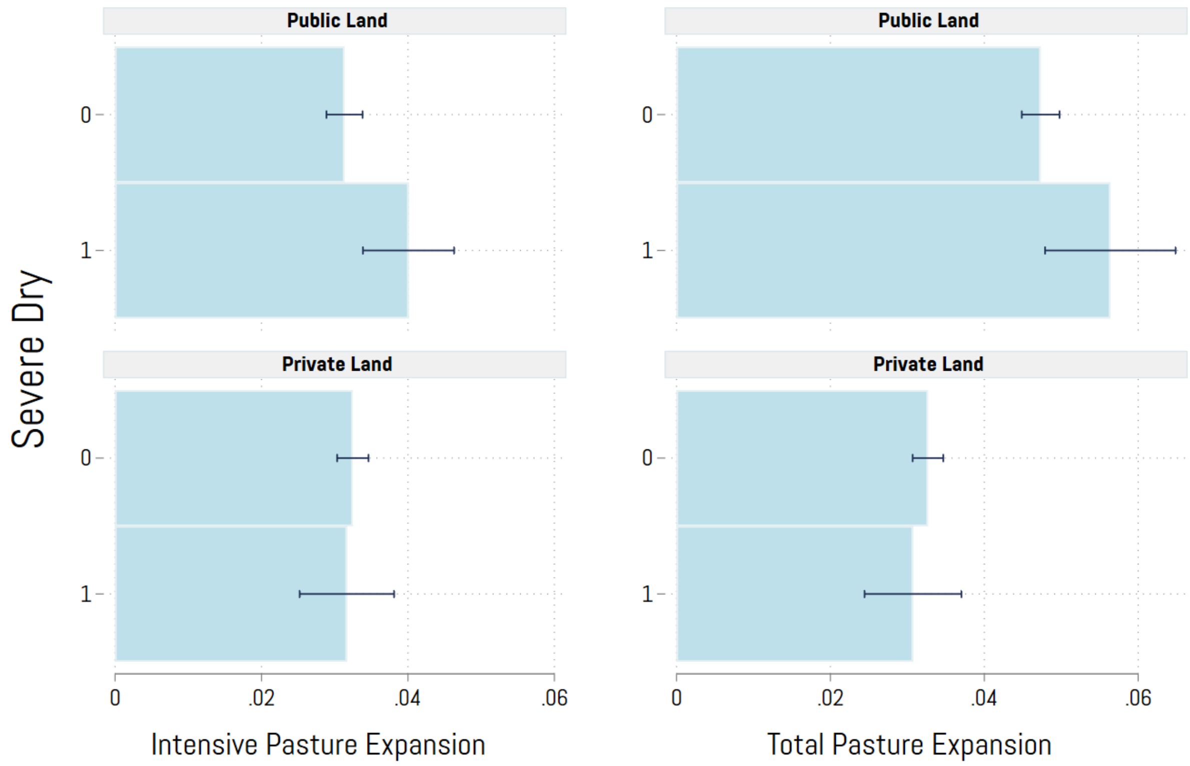
	Forest to Past. Int.	Forest to Past. Tot.
Dry	0.4996	0.5046
	(0.3272)	(0.3727)
Private Land	3.6069***	3.9726***
	(0.5680)	(0.6567)
Dry*Private Land	-0.6990*	-0.8734*
	(0.4141)	(0.4864)
Constant	5.9229***	6.8437***
	(0.2950)	(0.3390)
Observations	26814	26814
R^2	0.3723	0.3718

Standard errors in parentheses
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

5.3.2 Alternative Independent Variable: Severe and Extreme Droughts

We also investigate whether our results are sensitive to drought severity. Our main specification defines a drought episode for a given cell c at time t if the SPEI index is below one standard deviation of its historical average for cell c at time t. In this section, we test the robustness of our results to

Fig. 10: Severe Droughts and pasture expansion



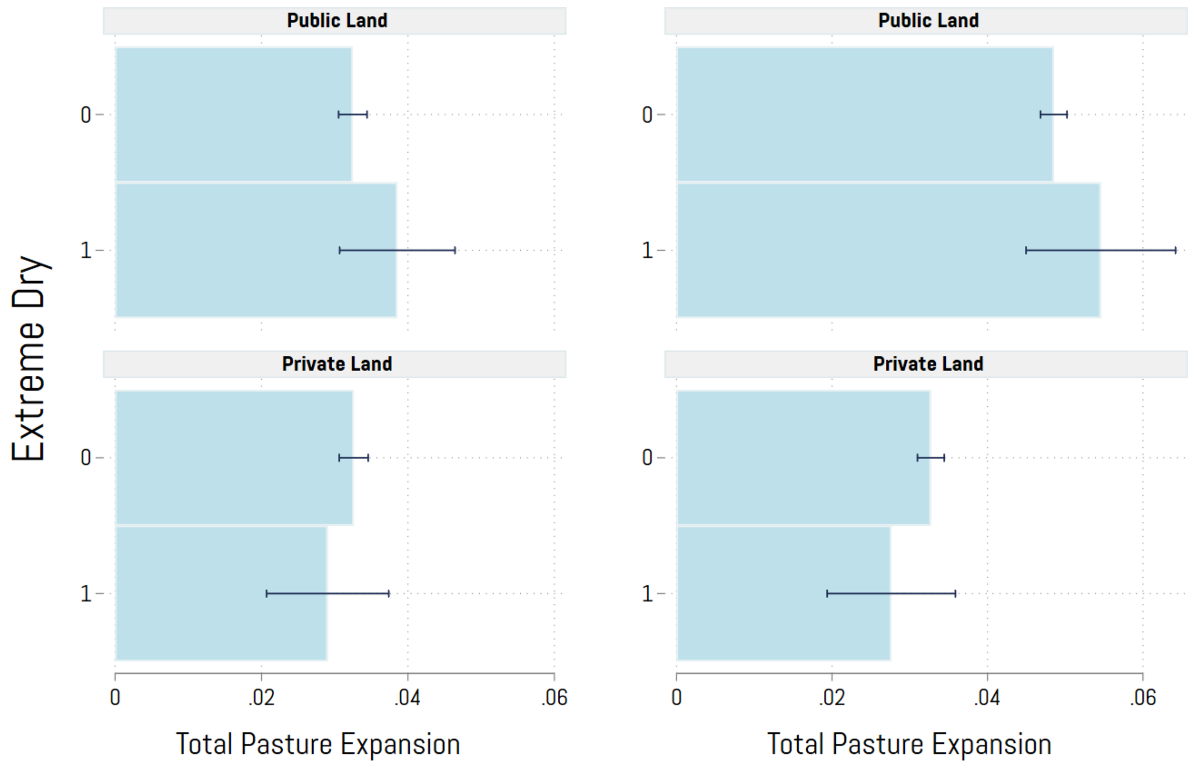
this SPEI index threshold.

We define severe and extreme droughts as periods when the SPEI index falls below 1.5 and 2 standard deviations, respectively, from its historical average for cell c at time t. Overall, our findings are robust to these alternative drought definitions. Figures 10 and 11 display the results for severe and extreme droughts, respectively. Compared to the baseline results, the patterns remain largely consistent; however, Figure 11 shows a slight reduction in pasture expansion on private lands during extreme droughts. This difference, though, is small and not statistically significant.

5.3.3 Including CAR poor lands

In this section, we reintroduce the CAR poor lands into our data. As described in section 4.1, CAR poor lands are properties registered in CAR with more than 5% of their area overlapped with other properties. Since these properties are more likely to be invaded public lands, we introduce them in our data in the public lands category. The results, shown in Figure 12, remain consistent with those obtained when CAR poor lands are excluded, indicating that their inclusion does not significantly affect our findings.

Fig. 11: Extreme Droughts and pasture expansion



5.3.4 Fixed Effects

We also investigate whether our results are robust to different fixed effects specifications. Figure 13 shows the results of regressions that include all the combinations of cell and year-fixed effects for intensive pasture expansion. The fixed effects make the *Dry* coefficient more precisely estimated, but the *Dry*Private* interaction coefficient is stable over all combinations of fixed effects. Therefore, even if the basis effect of droughts on pasture expansion depends on the specification, the different response between private and public land farmers is robust in all specifications. These results are similar when considering the total pasture expansion, as shown in figure S.2 in the Appendix.

We also investigate whether our results are robust to the recent DiD estimators. More specifically, we compare the de Chaisemartin et al. 2022, henceforth CDiD, and two-way fixed effects estimators. Since our baseline model in equation 1 does not include unit fixed effects (which would be tenure-by-cell fixed effects) and the recent DiD estimators, such as CDiD, always include unit fixed effects, we compare the CDiD estimator to the following two-way fixed effects model:

$$y_{cit} = \beta_0 + \beta_1 Dry_{ct} + \gamma_{ci} + \delta_t + \varepsilon_{cit} \tag{2}$$

Fig. 12: Droughts and pasture expansion including CAR poor lands

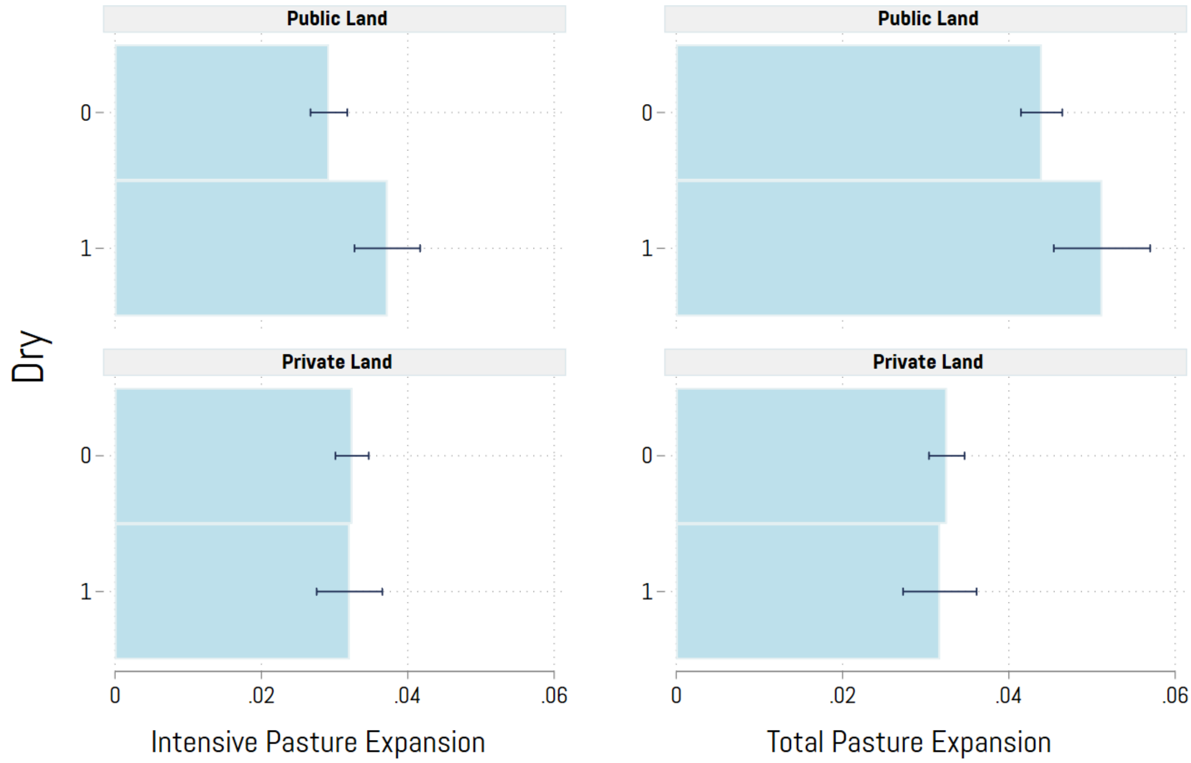
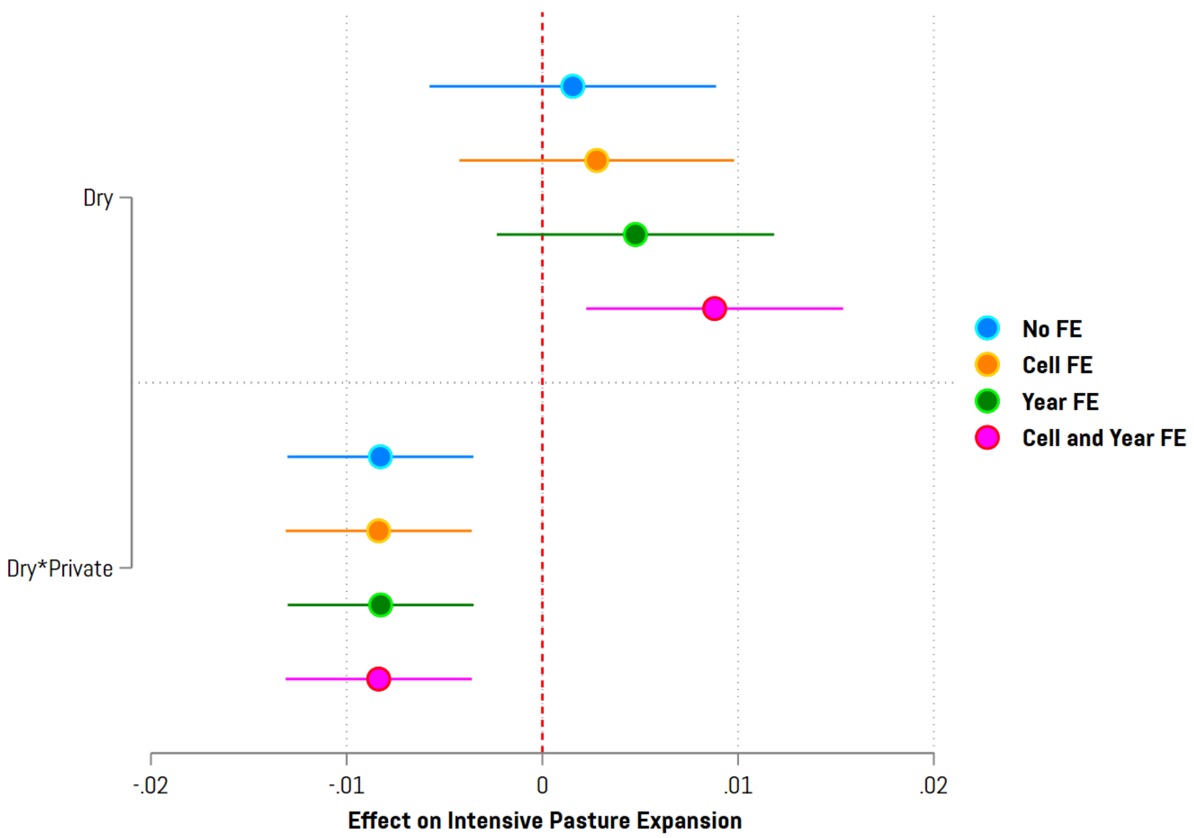


Fig. 13: Different fixed effects specifications, intensive expansion



where γ_{ci} is the unit (tenure-by-cell) fixed effect. We estimate the equation 2 and CDiD models for each private and public subsample.

The results are shown in table 5. Compared to the CDiD, the TWFE estimator underestimates the impacts of droughts on both private and public lands. However, the CDiD estimator still predicts a larger impact of droughts on public lands compared to private lands. The CDiD estimator predicts that droughts lead to a pasture expansion of 1.31% or 1.11% for the intensive and total expansion metrics used, respectively. The same estimator predicts an expansion of 0.32% of pasture on private lands. It is worth noting, however, that compared to the CDiD, the TWFE estimator greatly underestimates the impacts of droughts on private lands (0.01% vs. 0.32%). Nonetheless, for both estimators, droughts' impact on private lands is not statistically different than zero at the 5% significance level. On the other hand, both estimators predict a positive and statistically significant impact of droughts on public lands pasture expansion, regardless of the metrics used.

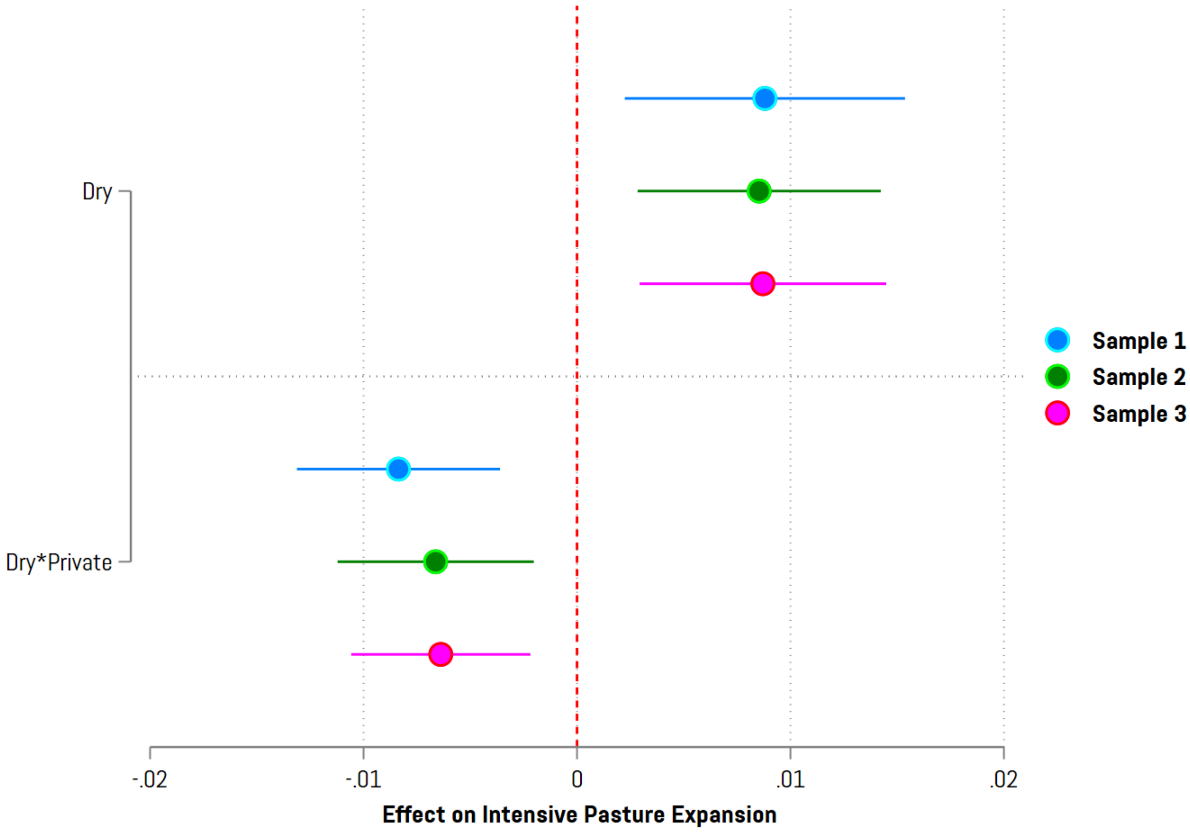
Tab. 5: TWFE and de Chaisemartin et al. 2022 estimates

Intensive Pasture Expansion					
		Estimate	SE	LB CI	UB CI
Public	TWFE	0.0092	0.0036	0.0021	0.0162
	CDiD	0.0131	0.0042	0.0050	0.0213
Private	TWFE	0.0001	0.0030	-0.0059	0.0061
	CDiD	0.0032	0.0024	-0.0014	0.0078
Total Pasture Expansion					
		Estimate	SE	LB CI	UB CI
Public	TWFE	0.0086	0.0044	0.0000	0.0173
	CDiD	0.0111	0.0051	0.0010	0.0211
Private	TWFE	0.0001	0.0030	-0.0059	0.0061
	CDiD	0.0032	0.0030	-0.0026	0.0090

5.3.5 Sample Trimming

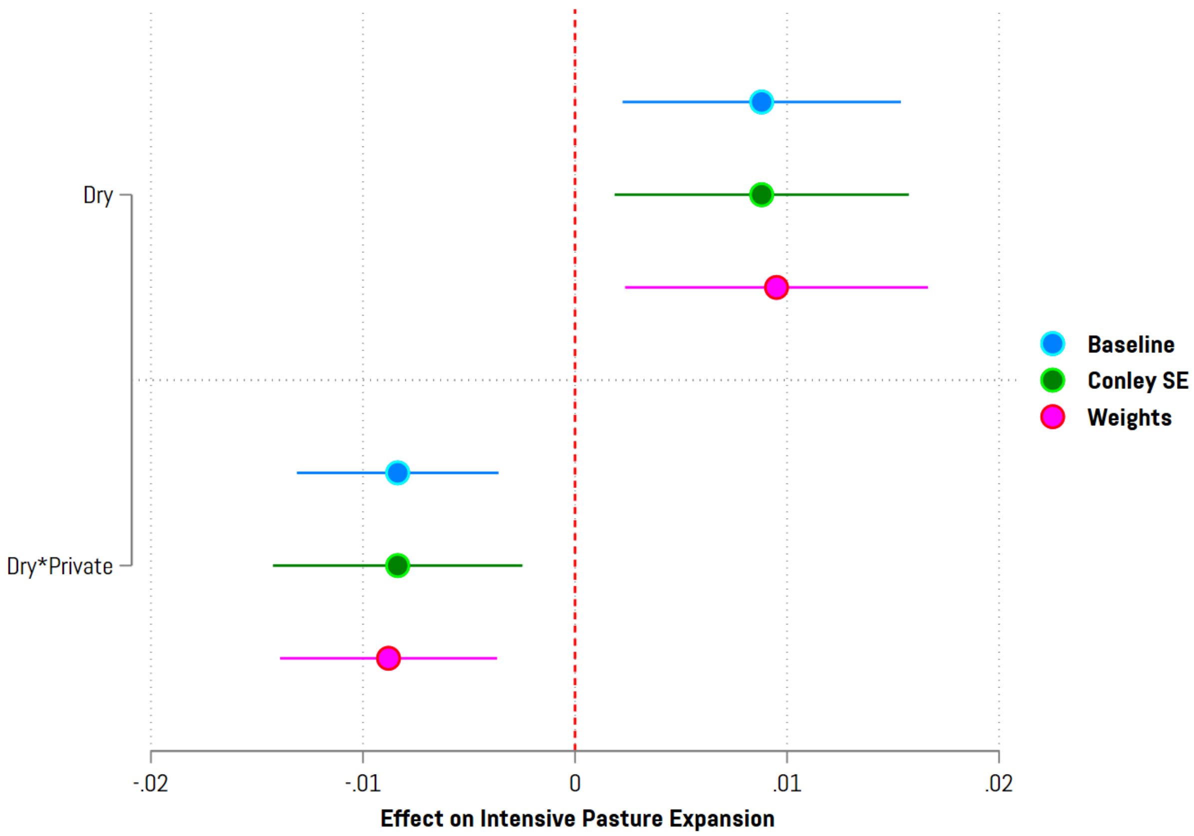
We also investigate whether our results are driven by the specific thresholds in pasture and forest cover we use for trimming. In our original specification, we trim our sample to include cells in which both public and private lands have at least 1 km^2 of pasture in the previous year and at least 10% of forest cover in 2001. We find that changing the pasture thresholds to 3 km^2 and 5 km^2, and the forest cover in 2001 to 15% and 20% do not change our results. Figure **??** shows the results for intensive pasture expansion. The results for total expansion, which are similar, are presented in figure S.3 in the Appendix.

Fig. 14: Different trimming samples, intensive expansion



S1 corresponds to a sample in which both public and private lands within a cell have at least 1 km^2 of pasture in the previous year and at least 10% of forest cover in 2001. S2 corresponds to a sample with thresholds of 3 km^2 of pasture and 15% of forest cover in 2001. S3 corresponds to a sample with thresholds of 5 km^2 of pasture and 20% of forest cover in 2001.

Fig. 15: Conley SE and weights, intensive expansion



5.3.6 Conley Spatial Standard Errors and Weights

Considering that cells have different compositions of pasture and forests within their public and private lands, we want to compare how pasture growth responds to dry conditions in cells with a similar composition of pasture and forest across different land tenures. Here, we test whether giving greater weight to cells with similar public and private land cover compositions changes our results. Let $p_{pub} = \frac{pasture}{pasture+forest}$ in public lands, and $p_{priv} = \frac{pasture}{pasture+forest}$ in private lands. Our weights take the form $1 - |p_{pub} - p_{priv}|$, thus giving more weight to more balanced cells. Figure 15 shows that these weights do not change our results significantly. Figure 15 also shows that using Conley standard errors with a 200 km threshold does not impact our standard errors significantly. Figure 15 in the Appendix shows the same exercise for total expansion, with similar results.

6 Concluding Remarks

This paper investigates how farmers under different land tenure regimes respond to climate shocks. Ranchers in public lands expand their pastureland more during drought years than private land ranchers, consistent with the premise that higher eviction risk causes ranchers to discount the future and avoid investing in pasture restoration. By expanding their pastureland during droughts, farmers in public lands mitigate the immediate negative effects of climate stress by bringing forestland into use. However, the results show that these differences are largely driven by the more humid regions of the Amazon: ranchers in public lands consistently respond to droughts across all climate zones, whereas private land ranchers do so only in the drier areas. This suggests that higher rainfall gives private landholders stronger incentives to invest in their properties and restore degraded pastures. Yet even in humid regions, ranchers in public lands still expand pasture rather than restore it, as eviction risk continues to discourage long-term investment. Investigating the mechanisms behind these results, we find that farmers in public lands respond more strongly to pasture degradation, reinforcing the theoretical argument that tenure insecurity amplifies short-term, extensification-based adaptation to drought.

We also find that the effects of droughts in public land pasture expansion remain positive and statistically significant even after controlling for pasture degradation, which is evidence that other mechanisms are at play. We investigate whether the environmental code modifications of 2012 gave farmers in public lands incentives to claim more land by expanding their productive area. We find that our results are driven by the post-2012 period. These findings are consistent with farmers in public lands using droughts as an opportunity to deforest and claim land after the environmental deregulation after 2012 and could help explain why the effect of droughts on public pasture expansion remains positive and statistically significant after controlling for pasture degradation. Nevertheless, we cannot undoubtedly attribute these post-2012 results to environmental deregulation since there are other important differences between these two periods. For instance, the pre-2012 period was characterized by milder climate conditions, whereas droughts were more frequent in the post-2012 period.

However, we must interpret these results with caution. We observe more pasture expansion in public lands during drought years, but our data does not allow us to conclude that these observed discrepancies are caused by land tenure. Farmers in public and private lands are inherently different from each other in socioeconomic terms. For instance, our results could be driven by different mitigation strategies by farmers of distinct income groups if farmers in public lands are poorer on average than those in private lands. Furthermore, our work focuses on droughts' short-term impacts across the different land tenure classes. Our results cannot tell us what the differences in deforestation across land tenure would be if the whole region became drier in the long term.

Our work presents new opportunities for future research. First, there is a need to better understand the differences between farmers in public and private lands at the farm level and how socioeconomic disparities could drive different responses to droughts. Second, future work should investigate the long-term effects of droughts on land-use decisions. A drier climate should induce land use conversion to drought-tolerant agricultural activities. However, differences in land tenure could induce diverse incentives for this long-term adaptation to a drier climate.

Overall, these results provide evidence that the increased occurrence of droughts under climate change expands the pressure on public land forests. Land use change driven by even short-term climate deviations can contribute to the erosion of the local environment. Since public lands cover more than two-thirds of the Amazon, the public forests represent a gigantic carbon stock, and their vulnerability makes it challenging to curb climate change. Our results should guide policymakers to increase the monitoring of public lands during drought years, design mechanisms to better enforce public property rights, and impose a tax on extensive cattle ranching. However, they do not provide information that either land tenure would perform better in environmental measures under permanent drier conditions.

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Appendices

A Measuring Intensive and Extensive Pasture Growth: Details

One practical challenge of this paper is that we do not observe the boundaries of public land farms since these ranchers are settled irregularly. To overcome this obstacle, we impose a set of assumptions that affect the interpretation of our empirical results.

To motivate this discussion, suppose we observe two ranchers, i and j. Rancher i is settled in private lands, and Rancher j in public lands. Suppose we also observe their land plots in years t and $t+1$ and suppose the ranchers expand their pasture land from year t to $t+1$. We call this growth in the ranchers' pasture *intensive* pasture growth. Ideally, we would like to compare both ranchers' intensive pasture growth.

Suppose now that we observe, close to rancher's k land, a newcomer rancher k in year $t+1$. Rancher k was not settled in the area during year t and decided to create a new ranch on previously unoccupied public land in year $t+1$. This pasture growth that comes from rancher k's new activity we call *extensive* pasture growth.

The primary goal of this paper is to investigate how ranchers on public lands react differently to drought shocks compared to ranchers on private lands. It is important to note that this problem involves ranchers already settled in public lands when the drought hits, as farmer j; it does not involve the problem of public land invasion by landgrabbers, as farmer l. As we cannot observe well-defined properties in public lands, we cannot directly observe ranchers j and l. That is, we cannot directly observe intensive and extensive growth in public lands. Therefore, we cannot use public and private farms as our unit of analysis and compare the land-use change across farms of different land tenures.

However, we observe land tenure, that is, whether a given area is public or private land, and we also observe land use. Let us define the observed intensive and extensive pasture growths using these two pieces of information. Define observed intensive growth in year t, $\widehat{IG}_t$, as the growth in pasture that intersects pastureland in year $t-1$, and extensive growth in year t, $\widehat{EG}_t$, as the growth in pastureland that does not intersect with pastureland in year $t-1$ (Figure 4).

Based on these definitions, let us define the following two estimators of intensive pasture growth:

$$\widehat{PGint}_{ts} = 1(s = pub) \cdot (\widehat{IG}_{ts}) + 1(s = priv) \cdot (\widehat{IG}_{ts} + \widehat{EG}_{ts}) \tag{3}$$

$$\widehat{PGtot}_{ts} = 1(s = pub) \cdot (\widehat{IG}_{ts} + \widehat{EG}_{ts}) + 1(s = priv) \cdot (\widehat{IG}_{ts} + \widehat{EG}_{ts}) \tag{4}$$

where s indicates if the land tenure is public or private.

The variable $\widehat{PGint}_{ts}$ considers only the observed intensive growth as the actual intensive growth in public lands. Assuming all private lands are already settled, all pasture growth in these lands is intensive growth. For private lands, both observed intensive and extensive growth are considered the actual intensive growth. The underlying assumption is that all observed extensive growth in public lands comes from new ranchers invading public lands.

Analogously, the variable $\widehat{PGtot}_{ts}$ considers that all observed growth is intensive for both land tenures. The underlying assumption in this case is that the number of ranchers in public lands is fixed, and there are no newcomers. The assumption for private lands is the same as before.

Finally, considering that we want to use $\widehat{PGint}_{ts}$ and $\widehat{PGtot}_{ts}$ to estimate the impacts of droughts on intensive public and private pasture growth, it is important to understand the implications for our estimations when the assumptions above are not met. More precisely, we would like to use these variables to estimate the impacts of droughts on the pasture growth difference between public and private lands. Keeping the assumption that all growth in private lands is intensive, we assume all growth in private lands is estimated correctly since these variables capture the same growth in private lands. Let us now consider three cases for public lands: all observed extensive growth is extensive, all observed extensive growth is intensive, and finally, some but not all observed extensive growth is intensive. Let us also consider b_{pub}, b_{priv}, and b_{EG} the effect of droughts on intensive public land pasture growth, private land pasture growth, and extensive public land pasture growth, respectively. We analyze how changing these three assumptions affects the estimation of b_{pub} and $b_{priv} - b_{pub}$ if we use $\widehat{PGint}_{ts}$ and $\widehat{PGtot}_{ts}$ as our pasture growth variables.

Let us consider the first case. If all observed extensive growth is extensive, then the variable $\widehat{PGint}_{ts}$ precisely captures the intensive pasture growth on public lands. Therefore, b_{pub} and $b_{priv} - b_{pub}$ are precisely estimated since these different assumptions do not affect the estimation of b_{priv}.

However, in this case, $\widehat{PGtot}_{ts}$ includes as intensive growth in public lands some extensive growth. The implications for the estimation bias of b_{pub} depend on the signs of b_{pub} and b_{EG}, and the bias of $b_{priv} - b_{pub}$ depend on the signs of b_{pub} and b_{EG}, and also whether $b_{priv} > b_{pub}$ or $b_{priv} < b_{pub}$. Consider b^*_{pub} and $b_{*priv-b*pub}$ the respective estimator of each coefficient. The bias in each one of these cases is summarized on the right-hand side of table S.1. First, consider that both b_{pub} and b_{EG} are negative. Since $\widehat{PGtot}_{ts}$ adds the extensive growth to the intensive growth, both negative effects compound, and the negative effect of droughts on intensive public land pasture is overestimated, i.e., $b^*_{pub} < b_{pub} < 0$. In this case, if $b_{priv} > b_{pub}$, then $b_{priv} - b^*_{pub} > b_{priv} - b_{pub} > 0$ and our estimator overestimates the positive difference between private and public land pasture growth. However, if $b_{priv} < b_{pub}$, then $b_{priv} - b_{pub} < b_{priv} - b^*_{pub} < 0$, and our estimator underestimates the negative difference between private and public growth.

Consider now that b_{pub} is negative, but b_{EG} is positive. Since $\widehat{PGint_{ts}}$ includes extensive pasture growth in its composition, b_{EG} reduces b_{pub} negative effect and b^*_{pub} is underestimated. If $b_{priv} > b_{pub}$, then $b_{priv} - b_{pub} > b_{priv} - b^*_{pub} > 0$ and our estimator underestimates the positive difference between private and public land pasture growth. However, if $b_{priv} < b_{pub}$, then $b_{priv} - b_{*pub < b_{priv} - b_{pub} < 0}$, and our estimator overestimates the negative difference between private and public growth.

We can apply the same logic to the other cases. If $b_{pub} > 0$ and $b_{EG} \leq 0$, b^*_{pub} is underestimated and $b^*_{priv} - b^*_{pub}$ is overestimated and underestimated if $b_{priv} > b_{pub}$ and $b_{priv} < b_{pub}$, respectively. If $b_{pub} > 0$ and $b_{EG} > 0$, then b^*_{pub} is overestimated. Now, $b^*_{priv} - b^*_{pub}$ will be underestimated if $b_{priv} > b_{pub}$ and overestimated if $b_{priv} < b_{pub}$.

Let us now consider the second case in which all observed extensive growth is actually intensive growth. Now, the variable $\widehat{PGtot_{ts}}$ precisely captures all the intensive growth, and the estimators derived from it should be unbiased. However, now $\widehat{PGint_{ts}}$ fails to consider the change in intensive pasture growth that we observe as extensive. Therefore, b^*_{pub} will underestimate b_{pub} regardless of the sign. The bias in this case is summarized on the left-hand side of table S.1.

The bias of $b^*_{priv} - b^*_{pub}$ depends on the sign of b_{pub} and whether $b_{priv} > b_{pub}$ or $b_{priv} < b_{pub}$. If $b_{pub} < 0$ and $b_{priv} > b_{pub}$, then $0 < b_{priv} - b^*_{pub} < b_{priv} - b_{pub}$, and $b^*_{priv} - b^*_{pub}$ is underestimated. If $b_{priv} < b_{pub}$, then $b_{priv} - b^*_{pub} < b_{priv} - b_{pub} < 0$ and $b^*_{priv} - b^*_{pub}$ is overestimated. Analogously, $b^*_{priv} - b^*_{pub}$ will be overestimatedif $b_{priv} < b_{pub}$ and underestimated if $b_{priv} > b_{pub}$, considering that $b_{pub} > 0$.

Finally, considering that a fraction , $0 << 1$, of all observed extensive pasture growth in public lands is actually intensive growth, then both variables fail to capture the correct intensive growth. The same cases discussed before apply and the bias is summarized in table S.1. In sum, the estimators derived from $\widehat{PGint_{ts}}$ and $\widehat{PGtot_{ts}}$ can both simultaneously underestimate or overestimate the actual effects b^*_{pub} and $b^*_{priv} - b^*_{pub}$, therefore, together they do not represent upper and lower bounds. The estimator derived from $\widehat{PGint_{ts}}$ will always underestimate b^*_{pub} if $\alpha > 0$, and the estimator derived from $\widehat{PGtot_{ts}}$ may overestimate or underestimate it depending on the sign of b_{EG}. The bias of $b^*_{priv} - b^*_{pub}$ depend on whether $b_{priv} < b_{pub}$, and on the sign of b_{EG} in the case of $\widehat{PGtot_{ts}}$. Ultimately, comparing the sign and magnitude of b^*_{pub} and $b^*_{priv} - b^*_{pub}$ derived from both variables helps us shed light on which case is more likely to be accurate.

Regardless of the assumptions, the variable $\widehat{PGtot}_{ts}$ captures the overall difference in pasture growth across public and private lands. It may shed light on how each type of tenure incentivizes land use conversion regardless of the type and number of farmers in each type of land. Even though this last variable may not perfectly capture how farmers settled in both types of land respond to climate shocks, it may still contribute to important policy implications.

Tab. S.1: Intensive pasture growth estimators and estimation bias

		$\widehat{PGint}$ if $\alpha > 0$					$\widehat{PGtot}$ if $\alpha < 1$		
		b^*_{pub}	$b^*_{priv} - b^*_{pub}$				b^*_{pub}	$b^*_{priv} - b^*_{pub}$	
			$b_{priv} > b_{pub}$	$b_{priv} < b_{pub}$				$b_{priv} > b_{pub}$	$b_{priv} < b_{pub}$
b_{pub}	< 0	U-	U +	O-	b_{EG}	< 0	O-	O+	U-
						≥ 0	U-	U +	O-
b_{pub}	> 0	U +	O +	U-	b_{EG}	≤ 0	U +	O+	U-
						> 0	O+	U +	O-

U represents an underestimated coefficient and O represents an overestimated coefficient. The signs represent whether the coefficient is positive or negative, ie, U- represents an underestimation of a negative coefficient, and O+ represents an overestimation of a positive coefficient

B Supplementary Tables

Tab. S.2: Droughts and pasture expansion

	Intensive Past.	Total Past.
Dry	0.0088***	0.0087**
	(0.0033)	(0.0042)
Private Land	0.0018	-0.0141***
	(0.0021)	(0.0017)
Dry*Private Land	-0.0084***	-0.0087***
	(0.0024)	(0.0030)
Constant	0.0304***	0.0464***
	(0.0014)	(0.0014)
Observations	26814	26814
R^2	0.2242	0.2296

Standard errors in parentheses
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

Tab. S.3: Droughts and pasture degradation

	Intensive Past.	Total Past.
Degrad. rate	0.3942***	0.3420***
	(0.0309)	(0.0342)
Private Land	0.0049**	-0.0134***
	(0.0023)	(0.0022)
Degrad. rate*Priv. Land	-0.1605***	-0.1089***
	(0.0387)	(0.0392)
Constant	0.0160***	0.0342***
	(0.0016)	(0.0016)
Observations	26814	26814
R^2	0.2375	0.2380

Standard errors in parentheses
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

C Supplementary Figures

Fig. S.1: Droughts and total pasture expansion pre and post-2012

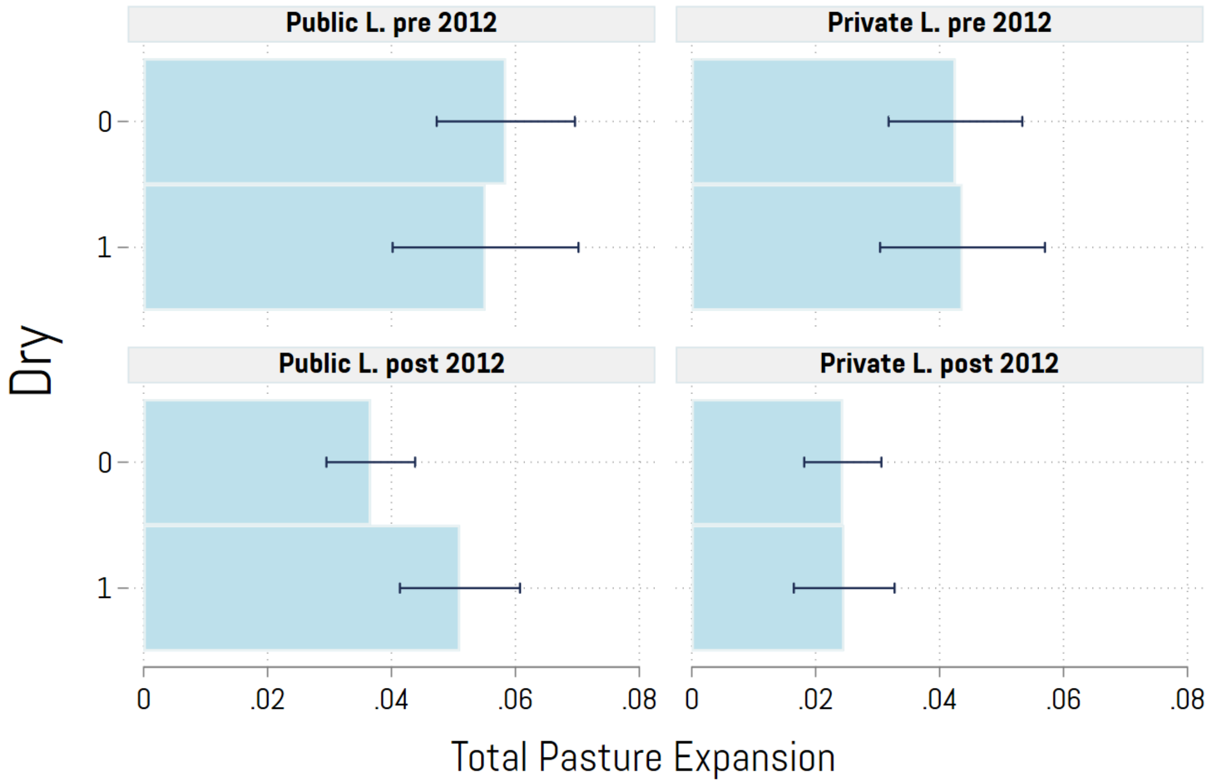


Fig. S.2: Different fixed effects specifications, total pasture expansion

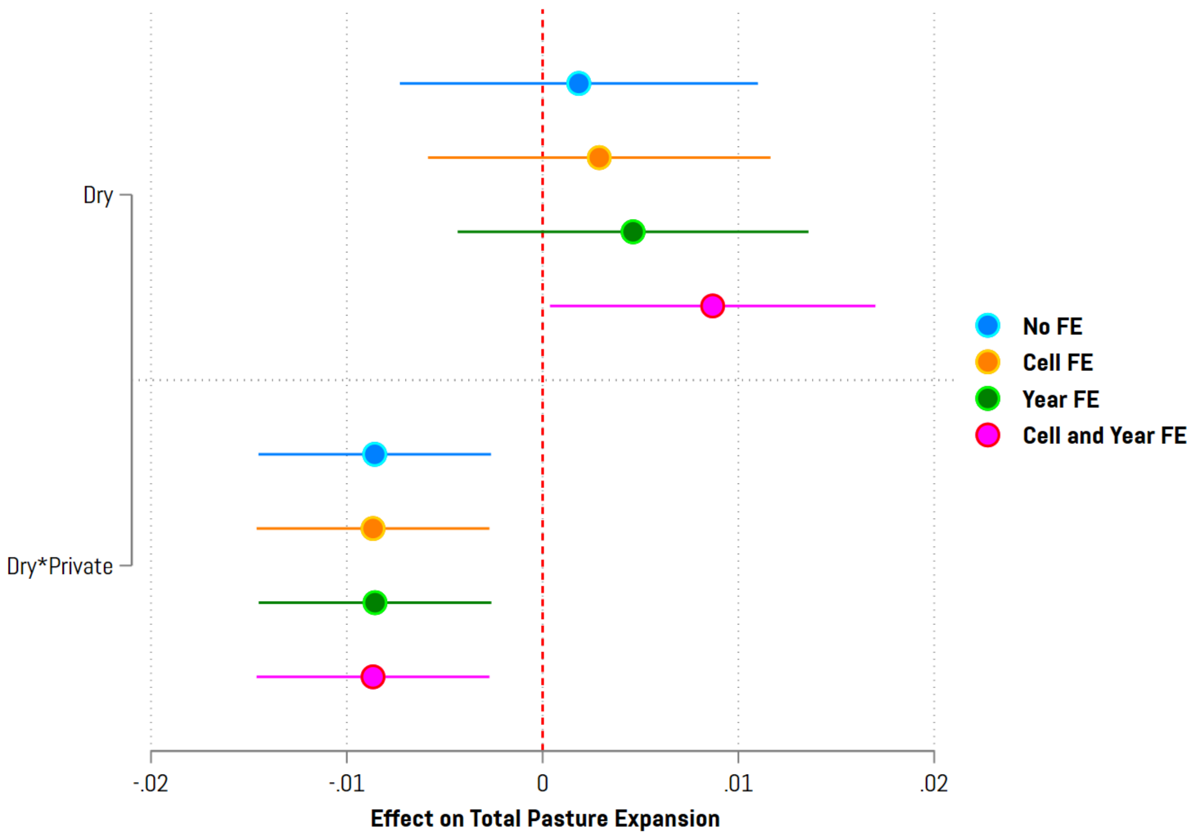
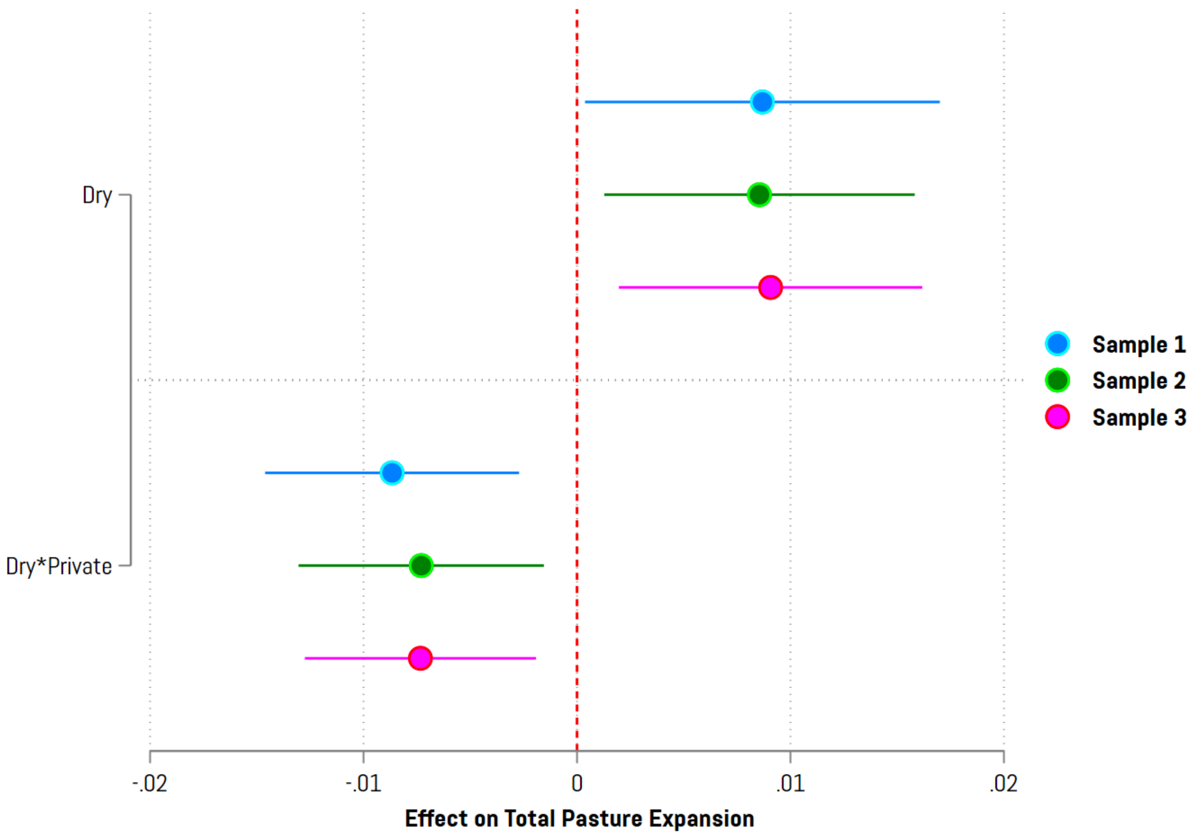
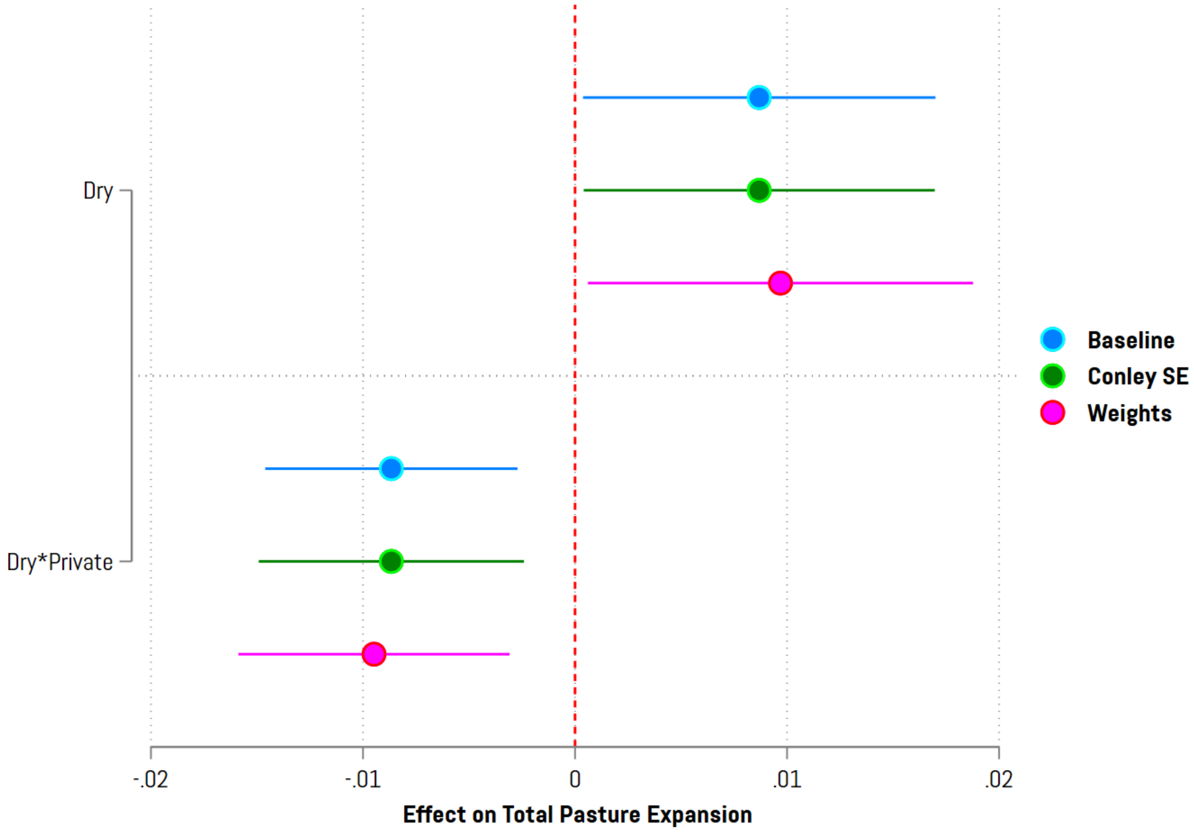


Fig. S.3: Different trimming samples, total expansion



S1 corresponds to a sample in which both public and private lands within a cell have at least 1 km^2 of pasture in the previous year and at least 10% of forest cover in 2001. S2 corresponds to a sample with thresholds of 3 km^2 of pasture and 15% of forest cover in 2001. S3 corresponds to a sample with thresholds of 5 km^2 of pasture and 20% of forest cover in 2001.

Fig. S.4: Conley SE and weights, total expansion



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