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Hydropower dams, deforestation, and land use change: evidence from Brazil

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Summary

In Brazil, hydropower generates more than 60% of the national electricity supply, placing the country among the most hydropower-dependent economies worldwide. This study analyses the causal impact of dam construction on forest loss in Brazil using a new municipality-level panel dataset covering 379 dams, combined with high-resolution satellite data on forest coverage, among other variables. Applying modern staggered difference-indifferences estimators and a dynamic event study, we find that dams reduce forest cover by almost 9% percent in the municipality. No anticipatory effects are detected, but postconstruction losses increase steadily over time. Mechanism show that forest loss occurs mainly through cropland expansion, with smaller increases in pasture and higher agricultural value added. Effects are concentrated in the North and North-East regions, amplified in municipalities with high public land shares and unequal land distribution, consistent with persistent institutional legacies. Our results highlight that while hydropower enhances energy security, it entails substantial environmental costs requiring stronger land governance.

Keywords: Hydropower dam, forest loss, land use change; institution

JEL Classification: Q56, Q15, O13

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Abstract

In Brazil, hydropower generates more than 60% of the national electricity supply, placing the country among the most hydropower-dependent economies worldwide. This study analyses the causal impact of dam construction on forest loss in Brazil using a new municipality-level panel dataset covering 379 dams, combined with high-resolution satellite data on forest coverage, among other variables. Applying modern staggered difference-in-differences estimators and a dynamic event study, we find that dams reduce forest cover by almost 9% percent in the municipality. No anticipatory effects are detected, but post-construction losses increase steadily over time. Mechanism show that forest loss occurs mainly through cropland expansion, with smaller increases in pasture and higher agricultural value added. Effects are concentrated in the North and North-East regions, amplified in municipalities with high public land shares and unequal land distribution, consistent with persistent institutional legacies. Our results highlight that while hydropower enhances energy security, it entails substantial environmental costs requiring stronger land governance.

Highlights

- Hydropower dams in Brazil cause substantial and long-lasting deforestation, both direct and indirect
- Forest loss is stronger in municipalities with greater potential for soy and other high-value crops
- Event-study estimates show persistent post-construction deforestation across affected areas
- While hydropower advances renewable energy goals, it imposes long-term ecological costs

1 Introduction

Hydropower dams have significantly influenced socio-economic and environmental outcomes worldwide, offering benefits such as energy generation, irrigation, and flood control. However, their broader impacts reveal complex trade-offs for local communities and ecosystems (Dillon and Fishman, 2019; Severnini, 2023).

In Brazil, hydropower accounts for over 60% of the country's electricity supply, making it one of the most hydropower-dependent economies in the world (García et al., 2024). This reliance has enabled the country to reduce its dependence on fossil fuels, improve energy security, and expand electrification, particularly in urban and industrial areas (Balboni et al., 2023; Castro-Diaz et al., 2024). However, while the socio-economic benefits of hydropower projects are well documented, their environmental and social trade-offs remain a subject of considerable debate.

One of the most pressing concerns is the significant forest loss induced by dam construction and operation. In particular, its role in accelerating deforestation[1] has become a growing concern, raising questions about long-term sustainability. This is because the construction and operation of hydropower dams are not isolated events but rather catalysts of broader landscape transformations that reshape land use, disrupt ecosystems, and trigger complex economic responses (Dillon and Fishman, 2019; Nickerson et al., 2022).

Deforestation linked to hydropower development occurs through both direct and indirect mechanisms. The most immediate environmental impact arises from the clearing of vast areas for dam reservoirs, power stations, and transmission infrastructure, leading to the permanent loss of forest cover (De Faria et al., 2017). However, hydropower-induced deforestation extends well beyond these direct land-use changes. The infrastructure required to support dam construction, particularly roads, facilitates human settlement, logging, and agricultural expansion, increasing accessibility to previously undisturbed forests (Nickerson et al., 2022; Araujo et al., 2025). This process is particularly pronounced in the Amazon, where new roads linked

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[1] Throughout the paper, the terms deforestation and forest loss are used interchangeably to describe reductions in forested area.

to hydropower projects have opened up remote areas to land speculation and farming, accelerating land conversion at an alarming rate (Gollin and Wolfersberger, 2024). Also in Costa et al. (2025) estimates suggest that infrastructure expansion associated with hydropower has contributed significantly to deforestation, particularly in regions where economic incentives for land-use change are strong. Transportation networks further amplify deforestation when combined with technological advancements in agriculture, which lower production costs and increase profitability in newly accessible forested areas (Balboni et al., 2023; Farrokhi et al., 2025). By 2020, approximately 10% of Brazil's forests had been degraded, with hydropower-induced deforestation identified as one of the key drivers (Dillon and Fishman, 2019; Balboni et al., 2023). This loss of forest cover exacerbates biodiversity loss, disrupts carbon sequestration processes, and contributes to greenhouse gas emissions through the decomposition of organic matter in reservoirs (Fan et al., 2022).

Moreover, the construction of dams has profound socio-economic effects on local communities. While hydropower plants are often justified on the grounds of economic growth and electrification, the benefits are unevenly distributed. Many rural and indigenous communities, especially in the Amazon, continue to rely on expensive and polluting diesel generators despite their proximity to major hydroelectric plants (Mayer et al., 2022; Mandai et al., 2024). The infrastructure development associated with dams has also been shown to exacerbate land conflicts, displacement, and socio-economic inequalities (Gollin and Wolfersberger, 2024).

Conversely, small hydropower plants (SHPs) have emerged as an alternative approach to energy generation, frequently promoted as a lower-impact solution. However, recent studies challenge the assumption that SHPs are inherently more sustainable, demonstrating that when developed in clusters, they can collectively produce higher deforestation rates per unit of energy generated than large-scale projects (Nickerson et al., 2022). This paradox arises because SHPs require extensive supporting infrastructure, including roads and transmission lines, contributing to habitat fragmentation and environmental degradation over time. The cumulative effect of multiple small-scale projects necessitates a reassessment of energy policies that incentivize their expansion without fully accounting for their environmental footprint.[2]

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[2] Further research can be done in this direction, at the macro level, looking at the evaluation of having SHP

Given the magnitude of hydropower development in Brazil and its intersection with critical environmental and economic concerns, a comprehensive assessment of its broader impacts is essential.

This paper aims to go deeply in this direction, studying the relationship between hydropower expansion and deforestation patterns, using municipal-level data on 379 hydropower projects, alongside deforestation data. A key contribution of this study is the use of an original dataset on dams, which uniquely incorporates the construction period of each project. This novel approach enables a more precise identification of the effects under analysis, as it provides a more detailed and temporally accurate measure of the treatment variable. Additionally, the study examines heterogeneity in the impact of hydropower dams across regional and municipal characteristics, as well as across agricultural and institutional contexts. In the latter case, we focus on two institutional dimensions of property rights to capture how tenure regimes and historical patterns of land concentration could mediate the dams' environmental impacts.

We employ a Two-Way Fixed Effects (TWFE) model in a staggered difference-in-differences setting, as the different municipalities adopt the treatment at different points in time, to estimate the overall impact of the presence of a dam, establishing robust causal estimates of dam-induced deforestation at the municipal level. Then, we use an event study approach to look at the effect of having a dam in the municipality across the years. The findings suggest that dams reduce municipal forest cover by nearly 9%, with losses accumulating after construction and driven mainly by cropland expansion. Impacts are concentrated in the North and North-East and are amplified in areas with large shares of public land and in municipalities with a high rate of land inequality. Consistent with Nickerson et al. (2022) and Balboni et al. (2023), our results show that hydropower expansion strengthens energy security but entails significant environmental costs, calling for stronger land governance (Costa et al., 2025). Together, these contributions provide the most comprehensive causal evidence to date on how hydropower development drives forest loss in Brazil, enriching both the empirical literature on tropical deforestation and the broader debate on the environmental costs of infrastructure

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vs large infrastructures.

projects in the Global South.

The remainder of this paper is structured as follows. Section 2 provides a comprehensive literature review on the socio-economic and environmental impacts of hydropower projects. Section 3 outlines the data and descriptive statistics, while Section 4 outlines the methodological framework, detailing the econometric models. Section 5 presents the empirical results and discusses policy implications. Section 6 concludes.

2 Impacts of hydropower dams: a literature review

Hydropower development is often considered a driver of economic growth, rural electrification, and employment generation, particularly in emerging economies where electrification gaps persist, and economic growth is closely tied to infrastructure investment. Empirical studies highlight its positive effects on GDP growth, employment creation, and industrial expansion, as well as its contribution to energy security by reducing reliance on fossil fuels and stabilizing electricity prices (De Faria et al., 2017; Dillon and Fishman, 2019; Balboni et al., 2023; Castro-Diaz et al., 2024). While these benefits are substantial, their distribution is often highly uneven. In fact, hydropower projects tend to concentrate economic advantages in urban centers and industrial hubs, while rural and indigenous communities near dam sites frequently face displacement and environmental degradation with limited access to the energy infrastructure they host (Mandai et al., 2024).

Among the most significant environmental costs of hydropower expansion is deforestation, which results from both direct and indirect mechanisms. The direct impact stems from the large-scale land clearing required for reservoirs, construction sites, and transmission lines, leading to the permanent loss of forest cover and biodiversity (Stickler et al., 2013). Beyond this immediate transformation, hydropower infrastructure triggers broader land-use changes through the construction of roads and transportation networks that facilitate access to previously remote areas, accelerating deforestation through agricultural expansion and settlement growth (Nickerson et al., 2022; Flecker et al., 2022). The expansion of road networks linked to dam projects has been identified as a major contributor to forest loss, with estimates suggesting that road-building since the 1990s has accounted for approximately 10% of total deforestation in Brazil (Gollin and Wolfersberger, 2024; Pfaff et al., 2007). Improved transportation infrastructure enhances market accessibility, which, combined with advances in agricultural technologies, further incentivizes the conversion of forests into farmland (Araujo et al., 2025). The profitability of cash crops such as soybeans amplifies this dynamic, as regions with greater connectivity to global markets experience higher rates of land conversion (Balboni et al., 2023; Farrokhi et al., 2025). Complementing this evidence, Costa et al. (2025) quantify the forest impacts of ten large hydropower plants built in the Brazilian Amazon between 2006 and 2011 using a synthetic control design. They estimate that dam construction explains about 13% of observed forest loss within 50 km of project sites, with effects concentrated in four plants (Belo Monte, Colíder, Jirau, Santo Antônio). Notably, at least 38% of the estimated loss is non-authorized—that is, outside licensed clearing—linking the largest impacts to episodes in which high-level political pressure overrode the environmental agency's technical recommendations; by contrast, projects with stricter adherence to licensing show negligible or even negative local effects. By separating authorized from non-authorized clearing and documenting sharp heterogeneity across plants, the study underscores both the potential effectiveness of Brazil's licensing framework and its vulnerability to political interference.

Beyond localized land conversion, the environmental footprint of hydropower projects extends to broader ecological processes. Large reservoirs alter hydrological cycles, impacting water availability, sediment transport, and soil fertility, which in turn contribute to land degradation and secondary deforestation in downstream areas (Dillon and Fishman, 2019). Changes in river flow disrupt riparian ecosystems, reducing agricultural viability in some regions while increasing land pressures in others where hydrological disruptions are less pronounced. De Faria et al. (2017), using a dataset on 56 Brazilian hydropower plants built between 1991 and 2010, show that the cumulative effects of dam construction extend well beyond the submerged areas, as landscapes surrounding reservoirs suffer from increased forest fragmentation, changes in microclimate, and the spread of invasive species. García et al. (2024) examine two large hydropower dams in the Brazilian Amazon and show that spatial spillovers are systematically underweighted in environmental impact assessments. Using household data from communities around the construction sites, they analyze changes in energy sources, access, and electricity prices, and identify their spatial (upstream/downstream, distance) and political (resettlement, negotiation) determinants. The authors report spatial injustices in energy access: households most adversely affected by construction, especially those in more distant upstream and downstream locations, those not resettled, and those not directly engaged in negotiations with dam developers, were significantly less likely to experience improvements in energy access or cleaner energy sources.

A particularly relevant aspect of hydropower-related deforestation is the role of small hydropower plants (SHPs), which are often perceived as an environmentally sustainable alternative to large dams. However, recent research challenges this assumption, demonstrating that SHPs can lead to disproportionately high deforestation rates per unit of energy produced. Nickerson et al. (2022) find that when these projects are developed in clusters, their cumulative impact on forest loss exceeds that of large-scale hydropower facilities, primarily due to the extensive road networks required for their construction and maintenance. Forest loss per megawatt of installed capacity is particularly severe in regions where multiple small-scale projects are developed nearby, highlighting the need to reassess policies that promote SHPs as a low-impact alternative. The interplay between hydropower expansion and agricultural intensification further exacerbates deforestation pressures. As roads associated with hydropower infrastructure improve accessibility, forested areas are increasingly converted into cropland, particularly for high-value export crops such as soybeans (Balboni et al., 2023).

A distinct strand of the literature highlighs how global demand for agricultural commodities accelerates deforestation by increasing the profitability of land conversion, particularly where transportation networks reduce costs and improve market access (Farrokhi et al., 2025). Economic competition for land near hydropower sites further amplifies these effects, as land values rise in response to improved connectivity and agricultural profitability (Boubekraoui et al., 2024). Bazzana et al. (2020) employ land-use intensity indicators to evaluate the extent to which hydropower development influences forest conversion, demonstrating that in many cases, dam expansion competes with agricultural activities for land resources, creating feedback loops that accelerate deforestation. Adamopoulos (2025), studying the effects of Ethiopia's 1997–2014 road expansion program on agricultural productivity and structural change, further illustrates how road investments linked to hydropower dams facilitate agricultural intensification, reinforcing land-use changes that contribute to further forest loss. These findings emphasize the necessity of incorporating spatially explicit methodologies in environmental impact assessments, as traditional evaluation frameworks frequently fail to capture the broader indirect effects of hydropower development on forest ecosystems.

2.1 Brazil environmental legislation

Beyond these environmental and socio-economic trade-offs, Brazil has experienced a long history of environmental legislation, with numerous policies to regulate land use, protect native vegetation, and address the ecological impacts of large infrastructure projects. Starting from 1934 with the Forest Code and then the Water Code, there was an introduction of restrictions on forest clearing and water use, but with weak enforcement. A major step forward came with the new 1965 Forest Code, which established Areas of Permanent Protection (APPs) and Legal Reserves.[3] The institutional framework was further strengthened by the National Environmental Policy (Law 6.938/1981, PNMA) and the 1988 Constitution, which enshrined the right to a balanced environment and assigned the state responsibility for ecosystem protection. Subsequent legislation, including the Public Civil Action Law (1985) and the Environmental Crimes Law (1998), introduced mechanisms for civil society oversight and administrative and criminal sanctions for environmental damage. In the 2000s, provisions such as Provisional Measure 2166/2001 increased the share of Legal Reserves in the Amazon, while the Dam Safety Policy (Law 12.334/2010) established registration and monitoring requirements for hydropower projects (World Bank, 2024; Brancalion et al., 2016). Finally, Complementary Law 140/2011 clarified the distribution of environmental responsibilities across federal, state, and municipal levels, and in 2025, Congress enacted the General Environmental Licensing Act (Law No. 15,190/2025), which introduced new categories of environmental licenses, expanded

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[3] Later reinforced by the 1989 reform.

self-declaration procedures, and harmonized rules across jurisdictions. Despite this gradual strengthening of the legal framework, enforcement has remained uneven, particularly in frontier regions of the Amazon, underscoring the importance of evaluating the environmental impacts of large-scale infrastructure projects (Barbosa et al., 2021).[4]

In line with thise, World Bank (2024) calls for the integration of more comprehensive environmental impact assessments that account for both direct and indirect land-use changes, emphasizing the importance of regulatory oversight and sustainable land management.

3 Data and descriptive statistics

We use the hydropower dataset constructed by Corbi et al. (2025), which integrates the National Dam Safety Information System (SNISB), that is the registry established by the National Dam Safety Policy (Law 12,334/2010) and administered by the National Water and Basic Sanitation Agency (ANA), with project-level documentation. SNISB records dams with multiple purposes (water supply, electricity generation, industrial and mining waste containment). We restrict the sample to hydropower plants, large investment projects that may provide economic and social benefits but also entail non-trivial, well-documented environmental costs. A key advantage of this dataset is its incorporation of official documents, work-authorization records from the National Electric Energy Agency (ANEEL) and information released by operating companies, which allows us to recover construction start dates and, in turn, to precisely reconstruct project timelines.[5,6]

The database covers 379 hydropower dams spanning 1889–2022. For each project, we observe both construction and operation periods, structured as a municipality-year panel, allowing for longitudinal analysis of changes in deforestation and economic activities over

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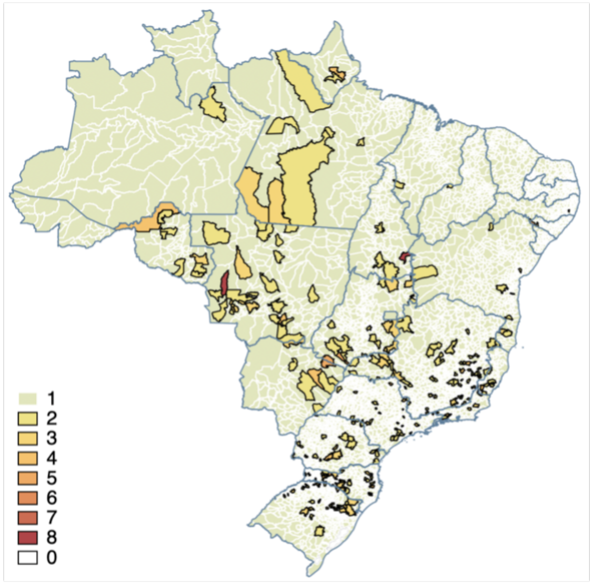
[4] See Barbosa et al. (2021) and Brancalion et al. (2016) for a comprehensive review on the environmental laws in Brasil.

[5] There are various official data sources on dams at the national and global levels. However, none of these databases include information on the year construction started, nor do they provide data at a micro-definition level (municipality-year level).

[6] Official data sources on dams at the national and global levels, are, among others, the Global Georeferenced Database of Dams (GOODD), the International Commission on Large Dams (ICOLD), The World Bank Hydropower Projects Database, AQUASTAT, the Global Reservoir and Dam Database (GRanD), and the Global Dam Tracker (Zhang et al., 2023 – Sci Data).

time. On average, construction lasts about five years, while average operation exceeds 17 years. Hydropower plants are present in 20 of Brazil's 26 states, underscoring their broad geographic footprint (see Figure 4).[7] Additionally, seven municipalities host more than three dams,[8] underscoring regional disparities in hydropower infrastructure concentration.

Figure 1: Distribution of hydropower dams across Brazilian municipalities



Source: Author's elaboration based on the original dataset on dams. *Notes:* The figure represents the cumulative number of hydropower dams built in Brazil from 1889 to 2022.

Forest cover data, as well as pasture land, cropland and soybean areas, comes from MapBiomas, which monitors land cover through satellite imagery with an estimated accuracy of 95%. This high level of precision ensures that the annual rates of forest cover accurately reflect land-use changes, minimizing measurement bias and enabling robust assessments of both immediate and cumulative deforestation effects linked to dam construction and operation. Figure B.1 shows the evolutions of forest cover, pastureland, cropland and rivers across Brazilian municipalities, during the years.[9] A reduction in forest cover and an increase in

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[7] States with at least one hydropower dam: Alagoas, Amapá, Amazonas, Bahia, Espírito Santo, Goiás, Maranhão, Mato Grosso, Mato Grosso do Sul, Minas Gerais, Paraná, Pará, Pernambuco, Piauí, Rio Grande do Sul, Rio de Janeiro, Rondônia, Santa Catarina, São Paulo, Tocantins.

[8] Delmiro Gouveia (Alagoas), Comendador Levy Gasparian (Rio de Janeiro), Cassilândia (Mato Grosso do Sul), Aporé and Caçu (Goiás), Dianópolis (Tocantins) and Campos de Júlio (Mato Grosso).

[9] Source: https://plataforma.brasil.mapbiomas.org

both pastureland and cropland are evident.

The dataset includes also deforestation data from the PRODES project (INPE, 2000–2022) allowing us to replicate the analysis as robustness check.[10]

In addition to hydropower and land cover data, as agricultural control variables, the dataset includes livestock (total), arable land (Km²) and the agricultural production value (U.S. dollars).[11] The first variable sources from the Municipal Livestock Production (PPM), which details the number of sheep, cattle, horses, and goats at the municipality level from 1974 up to 2022.[12] To account for agricultural activity, we incorporate the other two agricultural control variables sourced from the Municipal Agricultural Production dataset (PAM). Arable land is used to construct a measure of agricultural expansion by computing the ratio of cropland area to total municipal land area (from 1988 to 2022). This variable provides a precise indicator of land-use intensity, enabling a clearer assessment of how hydropower development affects both direct deforestation and agricultural land dynamics.[13] The latter, agricultural production value, reflects total municipal output, capturing variations in productivity and economic performance, and thus the incentives driving land-use change, especially where hydropower expansion may interact with farming activities (Bazzana et al., 2020).

To check for heterogeneity in the hydropower dams' impact across different agricultural contexts, we also consider specific land destinations such as farming, pasture, cropland, and soybean areas, expressed in Km² and sourced from MapBiomas.[14] National land-use statis-

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[10] The PRODES project collaborates with the Ministry of the Environment and the Brazilian Institute of Environment and Renewable Natural Resources (IBAMA) and is funded by the Ministry of Science, Technology, and Innovations. Results are presented in Table 4 in Section 5.3.

[11] See Figure B.1 in the Appendix for the distribution of forest cover, pastureland, cropland and rivers across Brazilian municipalities, over the years.

[12] Poultry (e.g., chickens and hens) are excluded from the sample, as their production is typically associated with intensive farming systems rather than extensive pastures. This exclusion ensures a more accurate representation of livestock activities that directly impact land use and forest loss through pasture expansion.

[13] According to PAM (IBGE), this variable is reported by crop and by year: for temporary crops, it is the hectares sown in the reference year (e.g., soy, maize, rice), regardless of harvest outcome; for permanent crops, it is the hectares with mature plants expected to be harvested in the reference year (e.g., coffee, citrus, oil palm). Because reporting is per crop, multiple or sequential crops on the same plot may be counted more than once; consequently, PAM totals can exceed the municipality's physical cropped area. In the analysis we treated and replaced these cases with the same value as the municipal area.

[14] Farming follows the MapBiomas Level 1 class "3. Farming," which includes four subclasses: Pasture (3.1), Agriculture (3.2), Forest Plantation (3.3), and Mosaic of Uses (3.4). In this paper, cropland refers specifically to Agriculture (3.2), i.e., temporary and perennial crops, and excludes Forest Plantation and Mosaic of Uses. Pasture land refers to planted/improved pastures associated with livestock production.

tics from MapBiomas (1985-2022) show that agro-pastoral activities presented a 50% increase during the period, bringing the sector to occupy almost 30% of Brazil's territory.[15] Agricultural croplands expanded by 420,000 km², largely devoted to grains and sugarcane, reaching a total of 610,000 km² in 2022. Of this area, soybeans account for 350,000 km², four times their extent in 1985.[16] The Cerrado[17] remains the main agricultural region, accounting for almost 48% of Brazil's soybean area, while the Amazon experienced accelerated soy expansion in the last few years. Perennial crops tripled, led by coffee, citrus, and oil palm, which expanded sharply in Pará and Roraima regions. Consistent with Bustos et al. (2016) and Arias et al. (2017), roughly 64% of agricultural expansion is attributable to pasture-driven deforestation, 10% to the direct conversion of native vegetation into cropland, and 26% to intensification on already anthropic land.

The final dataset ranges from 2002-2022. Table 1 reports descriptive statistics for the main variables we accounted for. The treatment indicator shows that only a small share of municipality-year observations are exposed to dam construction and operation (3%), reflecting the staggered nature of hydropower expansion. Installed capacity varies widely across projects, ranging from small facilities to large dams exceeding 11.000 kW. Land-use variables from MapBiomas display substantial heterogeneity across municipalities: while average municipal forest cover is about 955 km², values range from zero to more than 156.000 km², a dimension equivalent to the entire territory of Tunisia; cropland and soybean areas also show large dispersion, with a maximum of 6.000 km² and 5.800 km², respectively. Expressed as municipal shares, forests account on average for 39% of total land, pastures for 30%, cropland for 11%, and soybeans for 7%. Socio-economic indicators highlight strong inequality across municipalities: mean GDP per capita is 5.590$, but the distribution spans from below 200$ to almost 400.000$.[18] Agricultural production value and livestock numbers, drawn from

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[15] Pastures alone grew by 610.000 km² reaching 1.64 million of tons km² in 2022, with the Amazon biome as the one with the largest pasture area (577.000 km² vs. 513.000 km² of the Cerrado).

[16] Brazil produced 46.2 million tons of soy grains in 2006 (IBGE, 2006) and 21.6 million tons in 1996 (Falcone and Rosenberg, 2025).

[17] The Cerrado is Brazil's vast tropical savanna biome, covering roughly 2 million km² across central Brazil.

[18] According to IBGE municipal GDP per capita data, Primeira Cruz (MA), Guamaré (RN), and Mirante (BA) are among the municipalities with the lowest per capita GDP in Brazil, whereas Presidente Kennedy (ES) consistently records the highest one.

PAM and PPM respectively, also reveal wide differences, consistent with the coexistence of small-scale farming and highly industrialized agribusiness.[19]

Table 1: Summary statistics

Variable	Mean	SD	Min	Max	Obs.
Source: Authors' elaboration (see text)					
I(dam)	0.03	0.18	0	1	211698
Dam capacity (Kw)	12.30	193.03	0	11200	211698
Source: MapBiomas					
Forest cover (km^2)	955.72	5095.48	3.12	154996.6	115311
Farming (km^2)	456.69	777.04	0	19741.07	115332
Pasture land (km^2)	297.10	642.62	0	19631.95	113211
Cropland (km^2)	93.65	294.58	0	6066.66	111300
Soybean area (km^2)	85.55	288.43	0	5854.007	67704
Source: IBGE					
GDP per capita ($)	5564.06	6938.11	139.62	398512.7	109840
Population density	91.80	411.44	.031	11123.13	115218
Source: PAM and PPM					
Agricultural production value ($)	15335.41	45321.67	0	2224649	114044
Arable land	126.90	365.53	.01	9294	114044
Livestock	44423.54	93560.32	1	2572716	114740
Soybean production value ($)	45298.11	163701	0	5809024	42215
Source: Atlas Agropecuario					
Public land	.16	.29	0	.99	115353
Source: Naritomi et al. (2012)					
Gini land inequality index	.80	.079	0	.94	103194

Notes: Authors' elaboration based on the dataset included in the main empirical analysis. Statistics refer to the 2002–2022 sample. See text for variable definitions and sources and Table A.1 in the Appendix for additional variables used for robustness analysis.

The dataset also includes key geographic and topographic variables that are used to compute the Coarsened Exact Matching (CEM). In particular: i) the global slope[20] (in degrees) as a measure of the average terrain steepness within each unit of analysis, derived from the Shuttle Radar Topography Mission (SRTM) dataset at 500m resolution; ii) the distance to

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[19] Sorriso (MT) and São Desidério (BA) are the two municipalities with the highest agricultural production value while São Félix do Xingu (PA) is the one with the highest number of livestocks in absolute terms, but also per inhabitant (almost 17 animals per person).

[20] Jarvis A., H.I. Reuter, A. Nelson, E. Guevara, 2008, Hole-filled seamless SRTM data V4, International Centre for Tropical Agriculture (CIAT), available from http://srtm.csi.cgiar.org.

water[21] (in meters) derived using World Vector Shorelines combined with rivers and lakes from World Data Bank 2 (via Natural Earth) dataset and, iii) the global elevation[22] (in meters) representing the mean altitude of each unit, also sourced from the SRTM dataset. These variables are critical in controlling for geographic heterogeneity and ensuring comparability across treated and control units in the analysis. By integrating these high-resolution datasets, this study offers a comprehensive and detailed evaluation of the trade-offs between hydropower expansion, deforestation, and agricultural transformation in Brazil.

4 Econometric Specifications

To assess the impact of hydropower dam construction and operation on deforestation and land-use changes in Brazil, we employ a Two-Way Fixed Effects (TWFE) model and an event study approach to capture effects along the timeline.

Our empirical setting corresponds to a staggered difference-in-differences design, since municipalities are treated if they are exposed to a dam construction project at some point during the observation period. In such settings, conventional TWFE estimates may conflate heterogeneous treatment effects across cohorts and periods (Wooldridge, 2025).[23]

We complement the baseline specification with an event-study analysis *à la* Callaway and Sant'Anna (2021) which serves two key purposes. First, it allows us to test the validity of the parallel trends assumption by examining pre-treatment dynamics. Second, it provides a valid picture of the dynamic adjustment path, showing how deforestation responds in the years before, during, and after dam construction and operation.[24]

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[21] Wessel, P., and W. H. F. Smith, A Global Self-consistent, Hierarchical, High-resolution Shoreline Database, J. Geophys. Res., 101, B4, pp. 8741-8743, 1996.

[22] Jarvis A., H.I. Reuter, A. Nelson, E. Guevara, 2008, Hole-filled seamless SRTM data V4, International Centre for Tropical Agriculture (CIAT), available from http://srtm.csi.cgiar.org.

[23] See also Nagengast and Yotov (2025) for an application.

[24] This approach is particularly relevant in a staggered adoption context, as it avoids biased aggregation across treatment cohorts and provides a more accurate description of the timing and persistence of dam-induced land-use changes.

4.1 Two-Way fixed effects model in a staggered difference in differences setting

Our baseline specification is:

$$Y_{it} = \alpha I(dam)_{it} + \mu_i + \gamma_t + \epsilon_{it} \tag{1}$$

where the dependent variable Y_{it} is annual forest cover, measured in Km², for municipality i in year t. In this baseline specification (eq. (1)), the key explanatory variable is a treatment indicator, $I(dam)_{it}$, that equals one from the first year of dam construction until the end of the sample period. This specification includes municipality fixed effects (μ_i), which absorb all time-invariant municipal characteristics such as geography, climate, or long-term land-use potential, and year fixed effects (γ_t), which control for macroeconomic shocks, national policies, or climatic events that simultaneously affect all municipalities. ϵ_{it} is the error term. In equation (2), we extend the model by incorporating time-varying controls that capture agricultural dynamics, given their central role in land-use change in Brazil.

$$Y_{it} = \alpha I(dam)_{it} + \phi Controls_{it} + \mu_i + \gamma_t + \epsilon_{it} \tag{2}$$

Specifically, as baseline controls, we include per capita GDP (in U.S. dollars) and the logarithm of population density to account for broader socio-economic conditions. In addition, we include the share of arable land on total municipal area (a proxy for agricultural expansion), the value of agricultural production (in U.S. dollars) capturing the economic intensity of farming, and the value of livestock on the total municipal area from the Municipal Livestock Production (PPM) dataset. These variables help isolate the direct impact of dams from concurrent agricultural dynamics.

Finally, equation (3) introduces interaction terms between the treatment indicator and characteristics of municipalities and dams ($Interactions_{it}$).

$$Y_{it} = \alpha Interactions_{it} + \phi Controls_{it} + \mu_i + \gamma_t + \epsilon_{it} \tag{3}$$

These interactions make it possible to investigate potential heterogeneity in the effects of hydropower development: for instance, impacts may differ depending on the region in which the dam is constructed, the size of the municipality, or the timing in which the dam has been constructed. Moreover, we disentangle the impacts according to baseline land use, agricultural intensity, or the type of agricultural production (e.g., soybean production).[25] This step enables us to explore whether dams located in agriculturally consolidated regions generate systematically different deforestation outcomes.

To ensure robustness in our analysis of the impact of hydropower dams on deforestation, we employ Coarsened Exact Matching (hereafter, CEM) as a non-parametric method that reduces imbalance in covariates between treated and control units by temporarily coarsening the data into strata and ensuring only comparable observations are retained for analysis. In our context, this approach helps mitigate selection bias by controlling for topographical and geographical differences across municipalities, such as slope, elevation, and distance to the coast. By matching treated municipalities (those with hydropower dam projects) to control municipalities (those without dams) on these specified criteria, we ensure that observed differences in forest loss are more likely to be attributable to the presence of hydropower dams rather than other confounding factors. Moreover, as further robustness checks, we account for different measures of our forest cover outcome variable: i) standard deviation; ii) variation with respect to the previous year and ii) deforestation coming from the PRODES dataset.[26]

4.2 Event-study approach

To complement the TWFE estimates, we implement an event-study design in the spirit of Callaway and Sant'Anna (2021).[27] This approach explicitly accounts for the staggered timing of treatment by interacting cohort indicators with time periods relative to dam construction and operation. Specifically, equation (4) models forest cover at the municipality-year level

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[25] To further explore the relationship between agriculture and dams, we examine the impact of hydropower dams on various agricultural land variables rather than focusing solely on forest cover. Hence, based on equation (2), we use the three different outcome variables: farming, as the sum of pasture land and cropland, but we also distinguish between pasture land and cropland.

[26] Results are presented in subsection 5.3.

[27] We also follow Wooldridge (2023), Roth (2024), Borusyak et al. (2024), and de Chaisemartin and Li (2025) for robustness checks.

as a function of the interaction between the cohort G_i to which municipality i belongs and the period t (measured as years relative to the start of dam construction), yielding dynamic treatment effects γ_{gt} that correspond to the average treatment effect on the treated (ATT) for cohort G_i at time t.

$$Y_{it} = \sum \gamma_{gt} * G_i * T + \phi Controls_{it} + \mu_i + \gamma_t + \epsilon_{it} \tag{4}$$

As in the TWFE framework, we include municipality fixed effects (μ_i) to absorb time-invariant local heterogeneity and year fixed effects (γ_t) to control for common shocks. This specification allows us to test for the validity of the parallel trends assumption by examining pre-treatment dynamics and to trace the evolution of deforestation impacts in the years before, during, and after dam construction. In a staggered adoption context, the event-study design is particularly important because it avoids biased aggregation across treatment cohorts and provides a transparent picture of the timing and persistence of dam-induced land-use changes.

5 Results

5.1 Hydropower development and forest loss: main estimates

Table 2 presents the baseline TWFE estimations, without (column 1) and with controls (columns 2-4). The results indicate that the presence of an hydropower dam significantly decreases forest cover, with coefficients ranging from 95 to 90.8 ($p < 0.05$). These findings confirm that a hydropower dam directly contributes to deforestation by clearing land for reservoirs, roads, and related infrastructure (Balboni et al., 2023; Araujo et al., 2025). Specifically, adding no controls (column 1), the effect is slightly greater.[28] Once we include some controls that, as the literature outlines, are possible causes of forest loss, the magnitude of the coefficients slightly decreases, but they are still negative and highly significant.[29] As the

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[28] The estimation sample begins in 2002 because municipal-level information on control variables (e.g., per capita GDP and population density data) is only consistently available from that year onward.

[29] Including dam capacity as an additional control outline that larger dams are positively correlated with forest cover due to scale effects, while the main treatment coefficients remain negative and significant, showing that the deforestation effect is not driven by dam size (see Table A.2 in the Appendix).

average forest cover of a municipality is around 955 km² for all samples, this implies that, on average, having a dam in a municipality reduces forest cover by almost 10% of the mean in column (1), while in columns (2)–(4) roughly a decrease of 9.61% to 9.43% of the mean.[30] The magnitude of our results is in line with Costa et al. (2025).

Additionally, when examining the control variables, the arable land ratio shows a strong negative association with forest cover. Similarly, livestock intensity exhibits a significant effect, indicating that areas with high livestock densities experience more pronounced deforestation. Finally, higher income and population density are both associated with lower forest stocks: a 10% increase in GDP per capita corresponds to about 3.8 km² of forest loss, while a 10% increase in population corresponds to roughly 18 km². These negative coefficients are consistent with greater economic and demographic pressure on land (De Faria et al., 2017; Fan et al., 2022).

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[30] Table A.3 in the Appendix reports placebo regressions assigning treatment five to seven years before actual dam construction. Coefficients are small and mostly insignificant, confirming the absence of anticipatory effects and supporting the validity of the parallel trends assumption.

Table 2: Baseline estimates: impact of hydropower dams on forest cover

Dep. var.: Forest Cover (km^2)	(1)	(2)	(3)	(4)
I(dam)	-95.251**	-91.837**	-91.502*	-90.132*
	(47.289)	(45.787)	(47.563)	(46.269)
GDP per capita (log)		-38.961***		-32.570***
		(4.797)		(4.530)
Population density (log)		-181.188***		-180.428***
		(36.720)		(38.123)
Agri prod. value (log)			-11.805***	-7.966***
			(1.510)	(1.290)
Arable land (ratio)			-0.426***	-0.212*
			(0.124)	(0.116)
Livestocks (mun. share)			-0.703***	-0.625***
			(0.090)	(0.082)
Time FE	YES	YES	YES	YES
Municipalities FE	YES	YES	YES	YES
Adj. R-sq	0.998	0.998	0.997	0.986
Observations	113169	107645	109554	104265
Municipalities	5389	5389	5319	5319

Notes: TWFE regressions in a Staggered DiD setting (*jwdid* Stata command). Robust standard errors clustered at the municipality level in parentheses. All control variables are measured at the municipality-year level. *** $p < 0.01$, ** $p < 0.05$, * $p < 0.1$.

In Table 3, we examine the heterogeneity of the treatment at the regional, municipal, and dam characteristic level. All specifications include agricultural and socio-economic controls plus municipality and time fixed effects. Column (1) distinguishes dams located in northern versus southern regions; column (2) differentiates between small and large municipalities,[31] while columns (3) distinguish dams constructed before versus after 2010. Results show that in the North, the presence of a dam is associated with a 23.64% reduction in forest cover relative to the mean of the region, while southern regions display a small, negative, and non-significant effect on forest cover.[32] This regional divergence is in line with evidence that land-use in the Amazon is particularly sensitive to infrastructure shocks, due to road expansion, land speculation, and weak institutional enforcement (Yanai et al., 2020). By contrast, southern Brazil is characterized by more consolidated agricultural systems and higher governance capacity,

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[31] Large municipalities is equal to 1 if the municipality's area is above the median; 0 otherwise.

[32] That is -437.984 km^2 on 1.852 km^2 which is the average forest cover in Northen regions.

which may explain the absence of significant deforestation effects (Barona et al., 2010; Escobar, 2019). The concentration of negative impacts in large municipalities supports the view that extensive landholdings and land-tenure concentration facilitate large-scale forest clearing (Yanai et al., 2020), while small municipalities, often with more fragmented land structures, show positive responses.

Moreover, older dams display stronger negative impacts than recent ones, which can be interpreted as the cumulative result of long-run processes of agricultural expansion (De Faria et al., 2017) but also as evidence that dams built before the Dam Safety Law (2010) and the Native Vegetation Protection Law (2012) often bypassed environmental safeguards (Brancalion et al., 2016). These patterns align with evidence that insufficient enforcement of environmental rules has exacerbated the consequences of large infrastructure in the Amazon (Pinto et al., 2024; Escobar, 2019). More recent projects have been subject to stricter environmental licensing and mitigation standards, though enforcement has remained partial and has been weakened under recent political administrations (Barbosa et al., 2021).

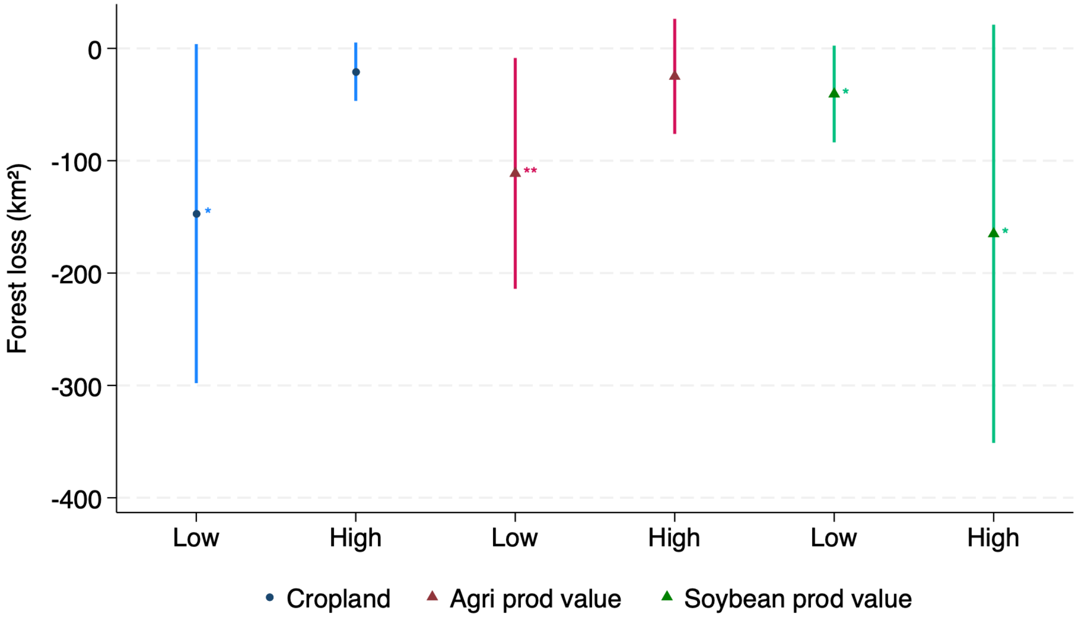
Table 3: Heterogeneous effects by regional, municipal and structural characteristics

	Regional characteristics	Municipal characteristics	Dams characteristics
Dep. Var.: Forest cover (km^2)	(1)	(2)	(3)
I(dam)*North	-437.984**		
	(222.614)		
I(dam)*South	-7.688		
	(12.612)		
I(dam)*Large mun		-139.355**	
		(65.906)	
I(dam)*Small mun		21.934***	
		(4.213)	
I(dam)*Old			-115.288*
			(68.921)
I(dam)*Recent			-41.030
			(25.180)
Socio-economic controls	YES	YES	YES
Agricultural controls	YES	YES	YES
Time FE	YES	YES	YES
Municipalities FE	YES	YES	YES
Adj. R-sq	0.988	0.987	0.987
Observations	104250	104250	104250
Municipalities	5304	5304	5304

Notes: TWFE regressions in a Staggered DiD setting. Robust standard errors clustered at the municipality level in parentheses. Control variables measured at the municipality-year level. North vs South is equal to 1 if the municipality is located in the North, North-East, 0 otherwise. Large municipalities is equal to 1 if the municipality's area is greater than the median level, 0 if it is below. Recent dams is equal to 1 if dam construction started after 2010; 0 otherwise. Extended results in Table A.4 in the Appendix. *** $p < 0.01$, ** $p < 0.05$, * $p < 0.1$.

Figure 3 examines further heterogeneity existing across different agricultural contexts. Specifically, we disaggregate the impact of dams on forest cover by categorizing municipalities as below (low) or above (high) the median based on their share of arable land, total agricultural production value, or soybean production value. Results show that dams drive sizable forest loss in areas with limited arable land, where forest is still abundant, but have no effect where cropland is already consolidated. Forest loss is also concentrated in municipalities with lower levels of agricultural production value. By contrast, where agricultural production is already high, the effect is smaller and not significant. This pattern may reflect the fact that in high-production value municipalities most of the forest has already been converted, while in low-production areas dams open new opportunities for agricultural expansion. In the case of soybeans, both low and high soybean municipalities show negative and significant effects, though magnitudes differ: 41 km² for low value of soybean areas and a much larger 165 km² for high soybean municipalities. This suggests that soybean cultivation is a particularly important mechanism linking dam construction to forest loss, with larger impacts in already consolidated soybean regions. Overall, these results suggest that hydropower dam induced deforestation is most pronounced in municipalities characterized by a relatively small share of cropland, lower overall agricultural production value, higher soybean production value, or a combination of these factors.

Figure 2: Heterogeneous impacts across agricultural contexts



Notes: TWFE regressions in a Staggered DiD setting. Robust standard errors clustered at the municipality level in parentheses. Socio-economic control variables measured at the municipality-year level. *High* stands for high level while *Low* stands for low level of the interacted variables. Extended results in the Appendix (Table A.5). *** $p < 0.01$, ** $p < 0.05$, * $p < 0.1$.

To further explore the agriculture–dam–deforestation nexus, we analyse the effect of hydropower dams on a set of agricultural land variables used as dependent outcomes. The results (see Table A.6) show that hydropower dams are associated with positive and significant increases in farming. When splitting the effect between cropland and pasture land, results are significant for cropland only.[33,34]

5.2 Public versus private land and the legacy of inequality

In this section, we exploit two complementary sources to capture the institutional and property-rights context shaping deforestation dynamics. First, we classify land as public or private using the *Atlas da Agropecuária Brasileira*, which compiles cadastral and geospatial data from the National Institute of Colonization and Agrarian Reform (INCRA) and other official registries to provide a contemporary map of property-rights regimes at the municipal level. Second, we follow Naritomi et al. (2012), who document how Brazil's colonial extractive episodes have left a persistent legacy of unequal land distribution, measured here through the land Gini from the 1996 Agricultural Census.

Figure 3 shows that the impact of dams on forest cover varies sharply across these institutional settings. In municipalities with a high share of public land (i.e., above 50% of the total municipal area), forest losses reach nearly 400 km² (i.e., almost 9% of the municipal area), whereas the effect is close to zero and statistically not significant where public land is limited. This pattern suggests that weak governance and unclear property rights in public lands amplify land conversion pressures following hydropower expansion. Turning to land equality, municipalities with high land equality also experience sizable forest losses, but estimates are not statistically significant. By contrast, in municipalities with a lower level of land equality, the effect of forest loss is 46 km².[35] Overall, these findings highlight that the institutional context, particularly the extent of public lands and the distribution of land ownership, may play a critical role in mediating the environmental costs of large infrastructure projects (Costa

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[33] Robust results in Table A.7 in the Appendix.

[34] The socio economic control variables reveal a positive association between GDP per capita and agricultural expansion, consistent with the literature showing that rising incomes stimulate agricultural intensification and greater commodity production (Carreira et al., 2024). Similarly, population growth correlates with greater agricultural land use in absolute terms (Boubekraoui et al., 2024).

[35] Extended resuls in Table A.8 in the Appendix.

Figure 3: Institutional determinants of deforestation after dam construction

Notes: TWFE regressions in a Staggered DiD setting. Robust standard errors clustered at the municipality level. Country controls and agricultural controls measured at the municipality-year level. Public High and Public Low are dummy variables equal to 1 if the share of public land in a municipality is above or below 50% of the total municipal area, respectively. Equal High and Equal Low are dummy variables equal to 1 if the land Gini coefficient is above (High) or below (Low) the median. *** $p < 0.01$, ** $p < 0.05$, * $p < 0.1$.

et al., 2025).

5.3 Testing the sensitivity of the results to alternative models and data

To verify the reliability of the baseline estimates, we conduct a battery of robustness checks reported in Tables 4. Panel A shows that the estimated negative impact of dams on forest cover remains remarkably stable across different specifications when we consider the standard deviation (SD) of forest cover as the outcome variable. In fact, introducing socio-economic controls and subsequently agricultural controls does not substantially alter the size or significance of the coefficients, which consistently point to a statistically significant decline in forest cover following dam construction and operation. In Panel B, we implement a matching procedure (CEM) to mitigate concerns about functional form assumptions or sample imbalance. The negative and significant coefficients confirm that the effect of dams on forest loss is not driven by specific model choices, but rather represents a robust empirical regularity. Panel C provides additional robustness by examining variation in forest cover from $t-1$, measured in km^2. The results again confirm a significant reduction in forested areas across specifications, thereby reinforcing the substantive relevance of the findings in terms of absolute land conversion. Finally, Panel D uses satellite-based PRODES deforestation data as the outcome variable. The results remain consistent: municipalities affected by dam construction and operation experience higher levels of deforestation, even when alternative datasets and measurement approaches are employed. Importantly, the magnitudes are comparable to those obtained using MapBiomas, underscoring the external validity of the analysis.

Table 4: Robustness checks: alternative specifications and data sources

Panel A:	*Forest cover (SD)*		
	(1)	(2)	(3)
I(dam)	-0.019**	-0.018**	-0.018*
	(0.009)	(0.009)	(0.009)
Adj. R-sq	0.998	0.997	0.998
Observations	113169	107645	104265
Municipalities	5389	5389	5319
Panel B:	*Forest Cover (Matching)*		
	(1)	(2)	(3)
I(dam)	-52.699**	-43.184*	-38.725*
	(24.617)	(22.620)	(21.084)
Adj. R-sq	0.985	0.978	0.982
Observations	10297	9688	9272
Municipalities	577	487	481
Panel C:	*Variation in forest cover (km^2)*		
	(1)	(2)	(3)
I(dam)	8.067**	8.786**	8.919**
	(3.189)	(3.507)	(3.570)
Adj. R-sq	0.563	0.559	0.559
Observations	107460	101962	98723
Municipalities	5373	5373	5301
Panel D:	*Deforestation (PRODES)*		
	(1)	(2)	(3)
I(dam)	119.403**	112.596**	110.681**
	(46.871)	(45.657)	(46.114)
Adj. R-sq	0.985	0.986	0.986
Observations	113211	107685	104265
Municipalities	5391	5391	5319
Socio-economic controls	NO	YES	YES
Agricultural controls	NO	NO	YES
Time FE	YES	YES	YES
Municipalities FE	YES	YES	YES

Notes: TWFE regressions in a Staggered DiD setting. Robust standard errors clustered at the municipality level in parentheses. Control variables measured at the municipality-year level. *** $p < 0.01$, ** $p < 0.05$, * $p < 0.1$.

Taken together, these robustness checks suggest that the observed impacts are not sensitive to model specification, outcome definition, or data source. The convergence of evidence across multiple approaches strengthens the causal interpretation of our findings and confirms that dam construction and operation exert a persistent and detrimental effect on forest conservation.

5.4 Event-study estimates of forest loss

To further investigate the dynamics of forest loss around dam construction and operation, we implement the Callaway and Sant'Anna (2021) with the augmented inverse probability weighting (AIPW) estimator, which combines regression adjustment with propensity weighting to obtain doubly robust estimates of dynamic effects.[36]

The results, displayed in Figure 4, consistently show that the parallel trends assumption is not violated: pre-treatment coefficients fluctuate around zero, with wide confidence intervals that include zero, suggesting the absence of systematic anticipatory effects of dams on forest cover. In particular, the figure emphasizes the cumulative nature of the impact: while short-term exposure yields negligible effects, longer exposure horizons generate increasingly negative average treatment effects, with point estimates surpassing 100 km^2 after seven to eight years.[37]

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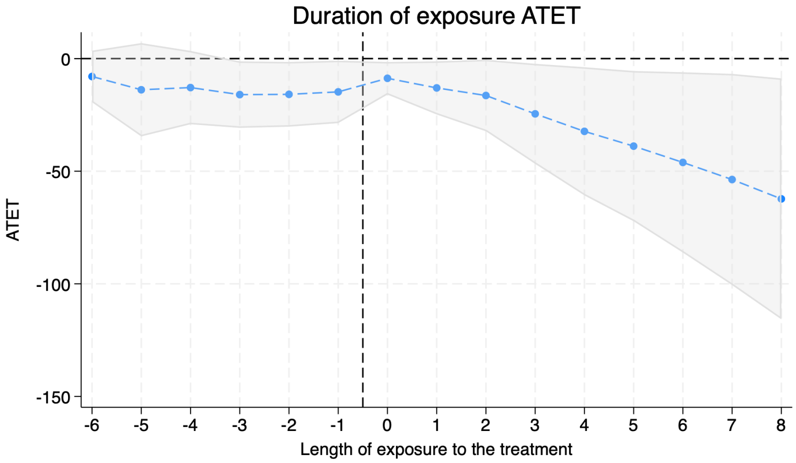
[36] In the Appendix robust results applying de Chaisemartin and D'Haultfoeuille (2023) and Callaway and Sant'Anna (2021).

[37] Appendix Figure B.2 reports robustness check results using other estimators, all confirming the baseline findings.

Figure 4: Dynamic effects of dam construction and operation on forest cover



Notes: Robust standard errors are clustered at the municipality level. Always-treated units are excluded from the estimates. Confidence intervals are reported at the 90% level. Estimates are obtained using the *xthdidregress aipw* command in Stata.

To complement the analysis on forest outcomes, we also estimate event-study specifications using cropland, soybean cultivation, and pasture as dependent variables, thereby tracing the agricultural responses induced by dam construction rather than direct forest loss.[38] The event-study results reveal clear differences across agricultural outcomes. For cropland, we find a steady and significant increase in area after dam construction, with effects accumulating year after year. This pattern confirms that dams foster agricultural expansion by easing access to new land and incentivizing conversion. In the case of soybean lands, we also observe an upward trend, though estimates are noisier and confidence intervals wider, suggesting a strong but more variable response of cash crop production to infrastructure shocks. Finally, pasture shows only a delayed increase, becoming more pronounced several years after treatment but still not significant. This lag is consistent with slower adjustments in livestock activities relative to crop-based expansion. Taken together, these findings demonstrate that forest losses following dam construction are mediated primarily through cropland expansion and, to a lesser extent, soybean cultivation and pasture growth.

Overall, the dynamic estimates reveal a clear pattern of forest decline following dam con-

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[38] In Figure B.3, all related results.

struction and operation. No significant changes in forest cover occur before dams become operational, ruling out reverse causality or anticipation effects. Instead, after construction, forest cover decreases persistently and monotonically, consistent with infrastructure-driven pressures such as road opening, settlement expansion, and agricultural conversion. These results reinforce the baseline findings and demonstrate that the environmental costs of hydropower development materialize gradually but accumulate substantially over time.

6 Conclusions

This paper investigates the environmental impacts of hydropower dams in Brazil, combining data on 379 dams with high-resolution land-use and socioeconomic indicators. Using difference-in-differences estimators suitable for staggered adoption settings, complemented by dynamic event-study designs, we document robust causal effects of dam construction and operation on forest cover.

Three central findings emerge. First, dam construction and operation lead to persistent and economically large reductions in forest cover. Across specifications, the estimated losses range from 90 to over 96 km^2 per municipality, corresponding to almost 9% of mean forest area. Dynamic estimates reveal no evidence of anticipatory clearing before construction, but consistent post-treatment declines that accumulate over time, exceeding 100 km^2 after eight years of exposure.

Second, we provide new evidence on the mechanisms through which dams reshape land use. In treated municipalities, we observe significant expansions of cropland and soybean production, accompanied by more modest increases in pasture. These results suggest that infrastructure-driven accessibility accelerates the conversion of forests into intensive agricultural lands.

Third, the impacts are highly heterogeneous across space and institutional contexts. Forest losses are concentrated in the North and North-East regions, where enforcement is weak. In contrast, highly consolidated areas of the South and Southeast exhibit no responses. Moreover, the interaction between dams and land tenure regimes is decisive: municipalities with high shares of public land experience sharp declines in forest cover, whereas those with predominantly private and more equally distributed land show no significant effect.

Our results have important policy implications. While dams contribute to energy security and growth, their indirect environmental costs are substantial. The findings call for integrated planning of energy and land-use policies, particularly in regions where governance capacity is limited. They also suggest that strengthening property rights and land governance institutions can attenuate environmental damage, reducing the scope for speculative land grabbing on public land. More broadly, the evidence emphasizes the need to incorporate indirect and long-run mechanisms, including agricultural expansion and livestock production, into environmental licensing and compensation schemes for infrastructure projects.

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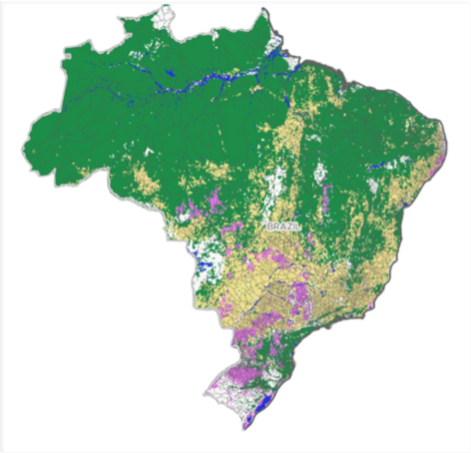
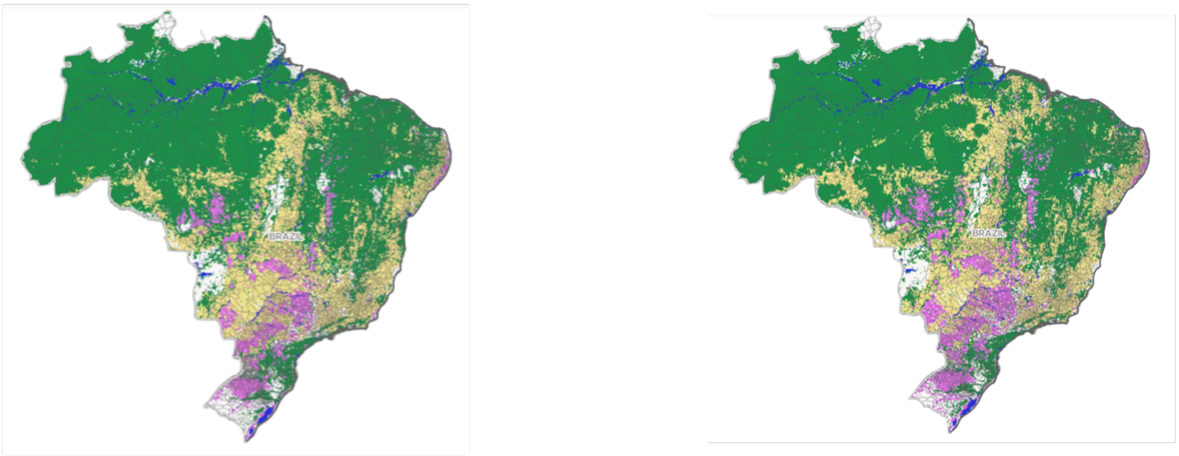
Appendix

Figure B.1: Forest cover, pastureland, cropland and rivers across Brazilian municipalities.



(a) 2000 year



(b) 2010 year

(c) 2020 year

Source: elaborations' from MapBiomas data. *Notes*: Forest cover is in green, pastureland in yellow, cropland in pink, rivers/lakes in blue.

Table A.1: Summary statistics - extended

Variable	Mean	SD	Min	Max	Obs.
Source: Geo-vars					
Slope	2.515383	1.714739	.114004	12.27699	115130
Elevation	451.4647	284.9965	3.932541	1587.793	115130
Distance to coast	326197.7	300405.7	753.4628	1492325	115130
Distance to water	52972.29	45644.25	685.5504	267537.2	115130

Notes: Authors' elaboration.

Table A.2: Estimated effects of dams on forest cover controlling for dam capacity

Dep. var.: Forest Cover (km^2)	(1)	(2)	(3)
I(dam)	-95.049**	-93.602**	-93.348**
	(45.670)	(47.345)	(46.183)
Dam capacity (mun. share)	24.710	16.868	24.504*
	(15.419)	(14.313)	(14.846)
Socio-economic controls	YES	NO	YES
Agricultural controls	NO	YES	YES
Time FE	YES	YES	YES
Municipalities FE	YES	YES	YES
Adj. R-sq	0.997	0.997	0.997
Observations	107645	109554	104265
Municipalities	5389	5319	5319

Notes: TWFE regressions in a Staggered DiD setting. Robust standard errors clustered at the municipality level in parentheses. Control variables measured at the municipality-year level. *** $p < 0.01$, ** $p < 0.05$, * $p < 0.1$.

Table A.3: Estimated effects of dams on forest cover - Placebo test

	5 years	6 years	7 years
Dep. var.: Forest Cover (km^2)	(1)	(2)	(3)
I(dam)	-59.214*	-39.590	-44.618
	(34.674)	(24.410)	(27.953)
Socio-economic controls	YES	YES	YES
Agricultural controls	YES	YES	YES
Time FE	YES	YES	YES
Municipalities FE	YES	YES	YES
Adj. R-sq	0.997	0.997	0.997
Observations	102879	102683	102588
Municipalities	5249	5239	5234

Notes: TWFE regressions in a Staggered DiD setting. Robust standard errors clustered at the municipality level in parentheses. Control variables measured at the municipality-year level. *** $p < 0.01$, ** $p < 0.05$, * $p < 0.1$.

Table A.4: Estimated effects of dams on forest cover - Heterogeneity (extended results)

	Regional characteristics	Municipal characteristics	Dams characteristics
Dep. Var.: Forest cover (km²)	(1)	(2)	(3)
I(dam)*North	-437.984**		
	(222.614)		
I(dam)*South	-7.688		
	(12.612)		
I(dam)*Large mun		-139.355**	
		(65.906)	
I(dam)*Small mun		21.934***	
		(4.213)	
I(dam)*Old			-115.288*
			(68.921)
I(dam)*Recent			-41.030
			(25.180)
GDP per capita (log)	-30.807***	-31.937***	-32.538***
	(4.589)	(4.481)	(4.525)
Population density (log)	-178.591***	-179.166***	-180.358***
	(37.834)	(38.011)	(38.141)
Agri prod. value (log)	-7.933***	-7.935***	-8.024***
	(1.263)	(1.282)	(1.315)
Arable land (mun. share)	-0.223*	-0.187	-0.213*
	(0.114)	(0.118)	(0.115)
Livestocks (mun. share)	-0.627***	-0.629***	-0.625***
	(0.082)	(0.082)	(0.082)
Socio-economic controls	YES	YES	YES
Agricultural controls	YES	YES	YES
Time FE	YES	YES	YES
Municipalities FE	YES	YES	YES
Adj. R-sq	0.988	0.987	0.987
Observations	104250	104250	104250
Municipalities	5304	5304	5304

Notes: TWFE regressions in a Staggered DiD setting. Robust standard errors clustered at the municipality level in parentheses. Control variables measured at the municipality-year level. North vs South is equal to 1 if the municipality is located in the North, Northeast, 0 otherwise. Large municipalities is equal to 1 if the municipality's area is greater than the median level, 0 if it is below. Recent dams is equal to 1 if dam construction started after 2010; 0 otherwise. *** $p < 0.01$, ** $p < 0.05$, * $p < 0.1$.

Table A.5: Heterogeneity across agricultural contexts

Dep. Var.: Forest cover (km2)	(1)	(2)	(3)
I(dam)*Cropland L	-147.070*		
	(76.965)		
I(dam)*Cropland H	-20.792		
	(13.243)		
I(dam)*Agri prod val L		-111.294**	
		(52.410)	
I(dam)*Agri prod val H		-24.942	
		(26.132)	
I(dam)*Soybean prod val L			-40.637*
			(21.955)
I(dam)*Soybean prod val H			-165.055*
			(94.940)
Socio-economic controls	YES	YES	YES
Time FE	YES	YES	YES
Municipalities FE	YES	YES	YES
Adj. R-sq	0.997	0.986	0.984
Observations	107645	107645	107645
Municipalities	5389	5389	5389

Notes: TWFE regressions in a Staggered DiD setting. Robust standard errors clustered at the municipality level in parentheses. In Agricultural controls are not included due to the high correlation with the interaction variables. *** $p < 0.01$, ** $p < 0.05$, * $p < 0.1$.

Table A.6: Effects of dams on farming, pasture land and cropland

	Farming (km^2)	Pasture land (km^2)	Cropland (km^2)
	(1)	(2)	(3)
I(dam)	93.912**	51.423	85.470*
	(43.839)	(46.421)	(44.951)
GDP per capita (log)	40.440***	-0.769	38.257***
	(5.025)	(4.981)	(5.180)
Population density (log)	171.691***	81.889**	181.496***
	(36.527)	(37.568)	(39.437)
Socio-economic controls	YES	YES	YES
Agricultural controls	NO	NO	NO
Time FE	YES	YES	YES
Municipalities FE	YES	YES	YES
Adj. R-sq	0.985	0.978	0.982
Observations	107665	105646	101854
Municipalities	5390	5289	5099

Notes: TWFE regressions in a Staggered DiD setting. Robust standard errors clustered at the municipality level in parentheses. Agricultural controls are not considered in the Table as they are highly correlates with the dependent variables. Socio-economic control variables are measured at the municipality-year level. *** $p < 0.01$, ** $p < 0.05$, * $p < 0.1$.

Table A.7: Effects of dams on farming, pasture land and cropland

	Farming (SD)	Pasture land (SD)	Cropland (SD)
	(1)	(2)	(3)
I(dam)	0.121**	0.080	0.114*
	(0.056)	(0.072)	(0.060)
Socio-economic controls	YES	YES	YES
Agricultural controls	NO	NO	NO
Time FE	YES	YES	YES
Municipalities FE	YES	YES	YES
Adj. R-sq	0.985	0.978	0.982
Observations	107665	105646	101854
Municipalities	5390	5289	5099

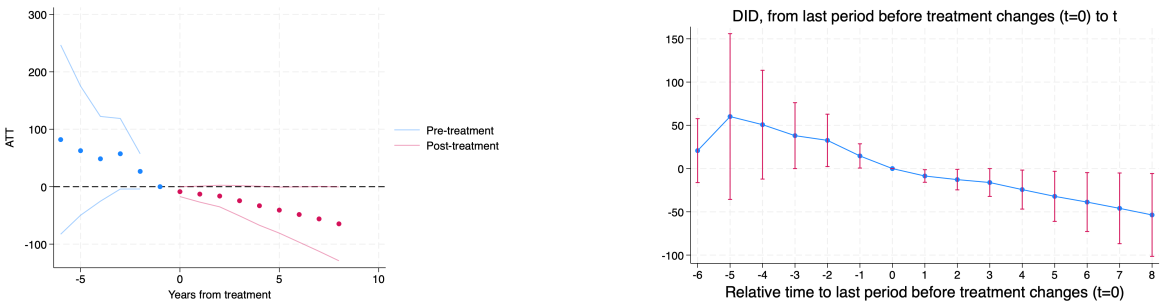
Notes: TWFE regressions in a Staggered DiD setting. Robust standard errors clustered at the municipality level in parentheses. Agricultural controls are not considered in the Table as they are highly correlates with the dependent variables. Socio-economic control variables are measured at the municipality-year level. *** $p < 0.01$, ** $p < 0.05$, * $p < 0.1$.

Table A.8: Estimated effects of dams on forest cover - Public vs private land and Gini index of land inequality (extended results)

	All regions		CO-NE-N regions	
Dep. var.: Forest cover (km^2)	(1)	(2)	(3)	(4)
I(dam)*Public H	-338.328*		-422.979*	
	(177.892)		(229.026)	
I(dam)*Public L	-13.921		-59.547	
	(14.751)		(50.037)	
I(dam)*Equal H		-150.144		-495.116
		(98.679)		(307.446)
I(dam)*Equal L		-44.593**		-89.923**
		(21.457)		(43.090)
GDP pc (log)	-37.795***	-38.948***	-55.349***	-54.212***
	(4.790)	(4.819)	(9.396)	(9.484)
Population density (log)	-180.072***	-181.005***	-243.332***	-237.000***
	(36.475)	(36.718)	(54.294)	(54.118)
Time FE	YES	YES	YES	YES
Municipalities FE	YES	YES	YES	YES
Adj. R-sq	0.997	0.998	0.997	0.995
Observations	107645	107645	52943	52943
Municipalities	5389	5389	2650	2650

Notes: TWFE regressions in a Staggered DiD setting. Robust standard errors clustered at the municipality level in parentheses. Socio-economic control variables measured at the municipality-year level. Columns 1 and 2 consider all regions while columns 3 and 4 Center-West, North-East and North regions only. Public H and Public L are dummy variables equal to 1 if the share of public land in a municipality is above or below 50% of the total municipal area, respectively. Equal H and Equal L are dummy variables equal to 1 if the land Gini coefficient is above (H) or below (L) the median. *** $p < 0.01$, ** $p < 0.05$, * $p < 0.1$.

Figure B.2: Event studies - other methods

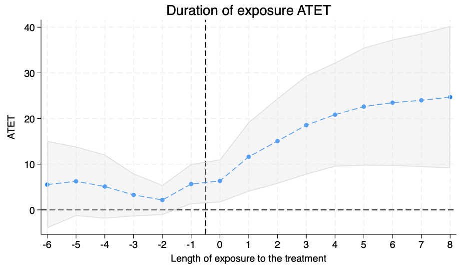


(a) Wooldridge (2025)

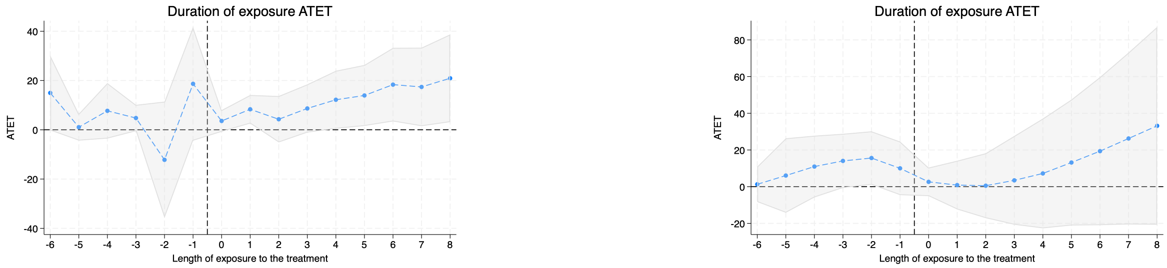
(b) De Chaisemartin & D'Haultfœuille (2025)

Notes: in Figure (a), *jwdid* with *hettype(event)* and *never* options have been used. In Figure (b), *didmultiplegtdyn* stata command has been used. Robust standard errors clustered at the municipality level.

Figure B.3: Event study analysis on cropland, soybean areas and pasture land.



(a) Cropland



(b) Soybean areas

(c) Pasture land

Notes: Robust standard errors are clustered at the municipality level. Always-treated units are excluded from the estimates. Confidence intervals are reported at the 90% level. Estimates are obtained using the command *xthdidregress aipw* in Stata.

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