Land Use Policy 169 (2026) 108161

Contents lists available at ScienceDirect

Land Use Policy

journal homepage: www.elsevier.com/locate/landusepol

Property size linked to deforestation, fragmentation and forest fires in the Brazilian Amazon

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ARTICLE INFO

Keywords:
Canopy height
Farm size
Forest Code
Forest degradation
Land-use change
Legal Reserve
Riparian buffers

ABSTRACT

Deforestation and forest degradation within Brazilian Amazonia primarily occur on private lands, yet their extent varies considerably among properties. We combine high-resolution remote-sensing and land-ownership data to analyse relationships between property size and forest retention, canopy height, fragmentation and fire prevalence across 30,022 private properties within an $> 113{,}000\ km^2$ consolidated agricultural frontier. Bayesian geospatial regressions showed that larger properties retained a greater proportion of forest overall and within Riparian Protected Areas and were more likely to adhere to Legal Reserve requirements. Properties > 1000 ha contained 75.2% of all remaining forest, while ≤ 150 ha properties accounted for the largest shares of total deforestation (27.6%) and riparian losses (39.9%). Smaller properties were also associated with lower-stature forest canopy, higher forest fire densities, and greater landscape-scale (~100-km^2) fragmentation, primarily due to increased disaggregation of forest among, rather than within, landholdings. Smaller properties thus not only retained less forest, but this forest was more degraded, underscoring the need to tailor policy interventions within Brazilian Amazonia according to property size. Targeted subsidies and credit schemes could enable smallholders to farm a smaller portion of their land, restore forest and reduce reliance on degrading land-use practices (e.g., fires). Fines, embargoes, and stricter Forest Code enforcement may help preserve remaining forest and reduce fire use in largeholdings. Incentivising forest restoration across property boundaries, and offsetting forest cover deficits to remaining continuous forest tracts, would help minimise fragmentation. Implementing these measures will depend on collaboration between government agencies and NGOs to align enforcement, financing, and coordinate landscape-scale management.

1. Introduction

Anthropogenic activities have increasingly eroded forest resilience throughout Amazonia, raising concerns that the region is approaching a tipping point of irreversible ecosystem collapse (Lovejoy and Nobre, 2018; Boulton et al., 2022; Flores et al., 2024). Avoiding this transition requires immediate efforts to curb forest clearance, fires, fragmentation and degradation (Brando et al., 2025; Filho et al., 2025). As agricultural expansion and intensification are the major drivers of Amazonian forest disturbances (Lapola et al., 2023; Haddad et al., 2024), the fate of the region hinges on the effective governance of private agricultural properties. The Brazilian Amazon alone contains > 137 million hectares (Mha) of private landholdings, which have accounted for almost 70% of regional deforestation since 2003 (da Silva et al., 2023) and up to one-third of annual forest fire cover (Alencar et al., 2019). Nonetheless, these properties host one-quarter of all remaining native Amazonian vegetation within Brazil (da Silva et al., 2023), equivalent to 8.7 Gt of carbon (Freitas et al., 2017). Property-level activities will thus accumulate to shape future, system-wide forest dynamics throughout Brazilian Amazonia. Identifying the characteristics of agricultural land-use systems that drive variation in forest disturbance, and where governance interventions can be most effective, is thus essential to bolstering regional forest resilience and safeguarding the global carbon budget (Rajão et al., 2020; Gatti et al., 2021).

Private properties throughout Brazil are governed by the Native Vegetation Protection Law (Law 12651/2012; henceforth, 'Forest Code'). In the Amazon this typically requires landowners to preserve 80% forest cover within their property as a 'Legal Reserve' (LR);

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https://doi.org/10.1016/j.landusepol.2026.108161
Received 22 June 2025; Received in revised form 9 June 2026; Accepted 11 June 2026
Available online 16 June 2026

prohibits deforestation within several ‘Areas of Permanent Preservation’, including buffers around rivers, streams and springs (Riparian Preservation Areas, RPAs); and heavily limits agricultural fires and forest resource extraction (Soares-Filho et al., 2014). However, 45% of Amazonian properties are non-compliant with this law (Rajão et al., 2020). In 2012, several major changes to the Forest Code were enacted, with the aim of improving compliance. This included the establishment of the Cadastro Ambiental Rural (CAR), a compulsory scheme requiring landowners to register their property boundaries in a public database (Soares-Filho et al., 2014). While this has not curbed illegal deforestation and forest degradation – between 2012 and 2020, 54% of deforestation within Brazilian Amazonia occurred within designated LRs (da Silva et al., 2023) – the CAR has provided unprecedented insights into landholder activities. For instance, recent research using the CAR has shown that 2% of all properties account for 55% of all illegal deforestation within Brazilian Amazonia (Rajão et al., 2020).

Property size is posited as a key determinant of variation in Amazonian deforestation rates and forest degradation processes (e.g., fragmentation, fires, resource extraction), as it shapes land-use decisions, economic pressures and compliance incentives faced by landowners (D’Antona et al., 2006; Godar et al., 2014). Smallholdings, typically family-run and dependent on subsistence agriculture, operate under strong spatial and financial constraints, meaning many smallholders must deforest most of their property to maintain economic viability (Michalski et al., 2010; Assunção et al., 2015). Individual largeholders may deforest greater areas but can typically meet financial break-even thresholds while clearing a smaller proportion of their land and are disproportionately targeted by fines and market pressures that discourage illegal deforestation (Godar et al., 2014; Nepstad et al., 2014). Largeholders can also be more selective in the areas they deforest – potentially increasing within-property forest disaggregation – but have greater capacity to preserve large, contiguous forest tracts, whereas landscapes dominated by smallholdings may exhibit greater fragmentation if forest set-asides are not coordinated across property boundaries (Arima et al., 2016). Fires, now the main source of carbon emissions in Amazonia (Gatti et al., 2021; Basso et al., 2023), represent a cheap alternative to mechanised forest clearance and pasture regeneration, and may thus be more heavily relied upon by smallholders (Cammelli et al., 2020; Pivello et al., 2021). Likewise, smallholders may depend more on practices such as timber extraction, nontimber forest product collection and hunting for subsistence (these practices are difficult to quantify directly, but canopy height provides a good proxy for overall forest degradation; Peres et al., 2006).

Despite these plausible mechanisms, empirical evidence linking property size to deforestation and forest degradation rates in Brazilian Amazonia remains inconsistent. Michalski et al. (2010) found that larger and more recently titled properties retained more forest but based this on a localised sample of only 300 properties. Godar et al. (2014) reported higher, less fragmented and less degraded forest cover in Amazonian census tracts dominated by smallholdings – yet these were generally located in more recently established agricultural frontiers, where greater forest cover would be expected. Furthermore, by aggregating across census tracts, Godar et al. (2014) attributed broad-scale forest cover patterns to properties that often covered a small fraction of each focal area, potentially obscuring property-level variation. Indeed, more recently, Cabral et al. (2022) found a positive association between property size and forest retention across all individual properties in Brazilian Amazonia but did not report on forest degradation processes, nor the influence of other, potentially confounding, variables. Consequently, a broad-scale assessment that attributes land-use outcomes directly to individual landowners, while accounting for confounding factors such as property age, is still needed to clarify the relationship between property size, deforestation and a full suite of forest degradation processes.

Here, we analyse relationships between property size and patterns of deforestation, forest degradation (canopy height, fragmentation, and fires), and legal compliance within the Portal da Amazonia (PdA) – a $>$ 11.37 Mha consolidated agricultural frontier in southeastern Brazilian Amazonia (Fig. 1a). We investigate hypotheses across four topics and three scales (individual properties, landscapes, and the entire region), focusing on the cumulative impacts of historical land-use changes as of 2020 and long-term forest fire prevalence:

1. *Overall forest cover*: Larger properties retain a greater proportion of forest; are more likely to adhere to the 80% LR requirement; and maintain less degraded forest, reflected in higher average canopy height.
2. *RPAs*: Larger properties retain a greater proportion of riparian forest; are more likely to fully comply with RPA requirements (i.e., 100% forest cover in mandated buffers); and retain less degraded riparian forest.
3. *Forest fragmentation:* Within-property forest fragmentation increases with property size. However, at landscape scales, forest will be more fragmented where average property sizes are smaller.
4. *Forest fires:* Annual forest fire cover is greater in smaller properties. Accordingly, the proportion of forest remaining in 2020 that has previously burnt will also be higher in smaller properties.

We investigate each of these hypotheses using separate, spatially explicit Bayesian regression models. Then, based on the observed property size relationships, we discuss how policy interventions tailored according to property size could help increase forest cover and reduce forest degradation, fragmentation, and fires, drawing on evidence from previous socio-economic and land-management initiatives within Amazonia. In doing so, we identify both interventions targeting specific forms of disturbance and broader approaches that may simultaneously address multiple dimensions of forest loss and degradation.

2. Methods

2.1. Study area

The PdA comprises 17 municipalities in northern Mato Grosso, covering $>$ 11.37 Mha at the centre of the ‘Amazonian Arc of Deforestation’ (Figure S1.1). It is one of Amazonia’s oldest and most consolidated agricultural frontiers, but historically consisted almost entirely of submontane *terra firme* forest, with some natural savannahs, grasslands, and wetlands (Fig. 1a). In the late 1970s roads connected the region to expanding agricultural frontiers farther south, paving the way for Agrarian Reform Settlements (ARS) and agricultural colonists. By 2020 forest cover had declined to 61.6%, with most deforestation occurring before the mid-2000s (forest cover in 2005 = 65.8%). Converted areas primarily consist of cattle pasture, although croplands have become increasingly prevalent in recent years (Figure S1.2). The region contributes ~R$8.7 billion annually to Brazil’s GDP, primarily through beef and soybean production, and now houses ~282,000 people, with 32% residing in rural areas (IBGE, 2023). As is typical of Amazonian agricultural frontiers, the PdA contains peri-urban areas comprised of smallholdings branching from main roads, and areas farther from towns comprised of variable sized properties, including major largeholdings (Oliveira-Filho and Metzger, 2006; Arima et al., 2016; Fig. 1b). The representative structure of the region and deceleration of regional deforestation over recent decades – indicative of most landowners having realised the full extent of intended deforestation – means that relationships between property size, deforestation, and forest degradation within the PdA could provide insights into future trends within developing agricultural frontiers elsewhere.

2.2. Datasets

We obtained land tenure data for the PdA from Instituto Centro de Vida (ICV), an NGO that assists landowners throughout Mato Grosso

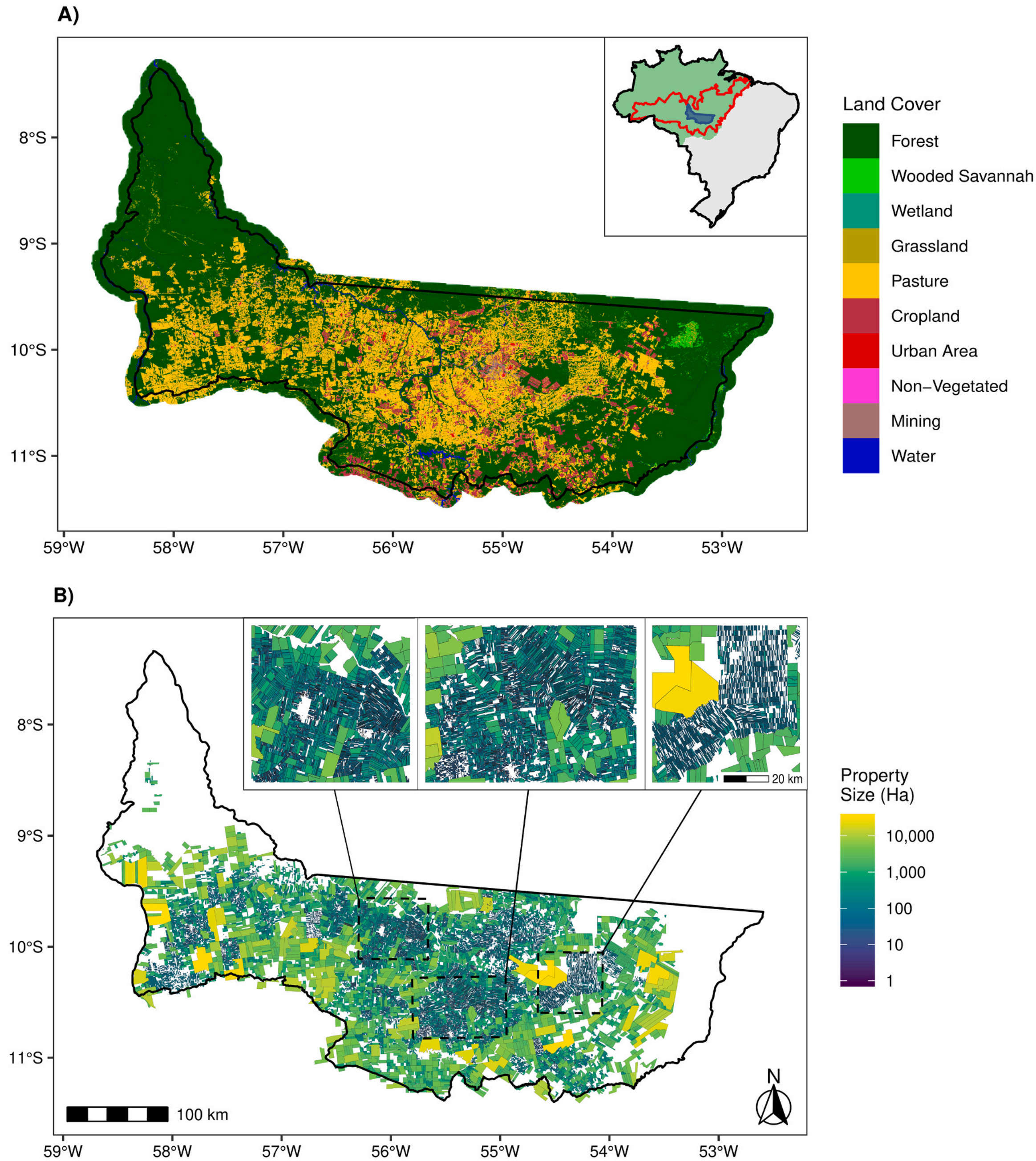

Fig. 1. A) Map of land-cover within the Portal da Amazonia (PdA) in 2020. Inset shows the position of the PdA (blue) within Brazil (grey), the Brazilian Amazon (green), and the Amazonian 'Arc of Deforestation' (red). B) Distribution of properties considered in this study. Inset maps depict three areas containing a high density of small properties.

with CAR registration (Fig. 1b). Property polygons were compiled from the CAR (accessed June 2023). Duplicate records and boundary overlaps were resolved by ICV, informed by extensive discussions with local landholders. To focus on agricultural properties and ensure reliability in estimates of associated land cover characteristics, we only analysed 30,022 properties (total excluded = 1676) where:

- *Land conversion had begun by 2020.* That is, forest cover was < 97.5% by 2020. This was intended to exclude properties titled post−2020 and properties that were fraudulently reported to gain subsidies (i.e., not actually owned by the registrant).
- *Converted areas were predominantly agricultural.* That is, pasture and/or crops covered the greatest proportion of converted areas within a property, thereby excluding properties dominated by urban areas and/or mines.
- *Water cover was < 10%.* This excluded: 1) properties flooded by hydroelectric reservoirs, as this prevented accurate estimation of agricultural land-use changes; and 2) aquaculture properties, as we could not definitively determine whether these properties were flooded or comprised of natural waterbodies.

We quantified land cover characteristics within each retained property in 2020 using MapBiomas 10 m Land-Use/Land Cover data (LULC; MapBiomas Project, 2023; Fig. 1a). As the Forest Code does not specify the exact vegetation type that landowners should set-aside, we grouped all native vegetation (forest, savannah, wetland, grassland) for analysis. However, as forest accounted for 97.2% of native vegetation within the analysed properties, we henceforth collectively refer to all native vegetation as 'forest' (Table S1.1).

We obtained hydrological data for the PdA from the State Environmental Agency (SEMA-MT, 2023). This included geo-referenced polygons representing all rivers, perennial streams, lakes and ponds, and geolocations for all headwater springs. We created buffers around each hydrological feature according to the legal RPA requirements, varying the buffer radius around stream and river segments (30–500 m buffers) according to their width, the buffer radius (50–100 m) around lakes and ponds according to their area, and constructing 50-m radial buffers around all springs (Table S1.2; Figure S1.3).

We obtained two 10-m resolution canopy height rasters for 2020 from Lang et al. (2023), containing mean canopy height estimates and associated estimation uncertainties (standard deviations), and masked out all non-forest areas according to the 2020 10-m LULC classification (Figure S1.4). We quantified fire occurrences, and classified the type of land cover burnt, using MapBiomas 30-m resolution Annual Fire Scar (Collection 3; Alencar et al., 2022) and LULC data (Collection 7; Souza et al., 2020) for 1985–2020. We also used the 30-m LULC data to estimate property ages (see Appendix S1); otherwise, the 30-m datasets and all derived metrics were exclusively used for forest fire analyses.

To control for accessibility, we obtained a road map for the PdA from the Humanitarian OpenStreetMap Team (HOTOSM, 2020), including all public roads circa 2020, and identified all urban areas from the 10-m (2020) and 30-m (annually 1985–2020) LULC classifications (Figures S1.5–1.6).

2.3. Property-level variables

As property-level response variables, we extracted eight land cover characteristics (Table 1): (1) Forest Cover: Proportional forest cover throughout each property (i.e., LR size). (2) LR Compliant: Whether each property met the ≥ 80% LR requirement. (3) RPA Forest Cover: Proportional forest cover across all RPAs within each property, where 1 indicates full compliance with RPA requirements. (4) Mean Forest Canopy Height and (5) Mean RPA Forest Canopy Height: Mean of the mean canopy height estimates for all remaining forest throughout each property and within its RPAs. To quantify uncertainty, we also calculated the pooled standard deviation for each canopy height metric:

$$\sigma_{Pooled,i} = \sqrt{\frac{\sum_{n=1}^{N_i} \sigma_n^2}{N_i}}$$

Where N_i is the total number of forest cells in property i (or its RPAs) and σ_n is the standard deviation of each cell's height estimate. (6) Forest Division Index (FDI): A measure of within-property forest disaggregation, defined as:

$$FDI_i = 1 - \sum_{p=1}^{n} \left(\frac{a_{i,p}}{A_i}\right)^2$$

Where A_i is the total forest area and $a_{i,p}$ is the area of forest patch p (Jaeger, 2000). This represents the probability that any two forested cells originate from different forest patches, enabling comparison among properties of different sizes and levels of forest cover: a 0 value indicates forest is concentrated in a single patch, while higher values indicate proportional increases in forest disaggregation. (7) Annual Forest Fire Density: Proportion of forest burnt in each year since each property was established (henceforth, 'property-years'), representing overall forest fire prevalence. (8) Remaining Forest Burnt: Proportion of forest remaining in 2020 that had been burnt at least once since each property was established, used as a measure of fire-driven forest degradation. Due to the temporal range of the fire data, we only considered fires since 1985.

In addition to Property Size (ha), we quantified nine other property-level predictor variables that could confound associations between property size and our property-level responses (Table 2): (1) Property Age and (2) Predates 1986: Using the 30-m LULC data (1985–2020), we identified the first year of deforestation within each property as a proxy for its establishment, defined as the first year forest cover dropped below 97.5% and decreased by ≥ 1% from the previous year (see Appendix S1 for details). We then calculated the age of each property in 2020, and annually for analyses of Annual Forest Fire Density. Because properties founded before 1986 could not be precisely dated, we also defined a binary variable indicating whether deforestation began before 1986. (3) Distance to Road and (4) Distance to Urban Area: Minimum distance (km) between each property and the nearest public road (2020 only) and urban area (in 2020 using 10-m LULC data, and annually for 1985–2020 using the 30-m data). As high-quality, historical roadmaps for the PdA were unavailable, we excluded Distance to Road from our Annual Forest

Table 1
Definitions and summaries of all property- and landscape-scale response variables, including the data source/s from which each metric was derived.

Variable	Scale	Definition (Unit)	Mean ± S.D. (Range)	Source
Forest Cover	Property, Landscape	Total forest cover (%)	Property: 21.61% ± 23.84 (0 – 97.49%) Landscape: 45.48% ± 21.57 (1.77 – 95.17%)	MapBiomas 10 m LULC
LR Compliant	Property	≥ 80% total forest cover?	Yes = 1113 No = 28,909	MapBiomas 10 m LULC
Mean Forest Canopy Height	Property	Mean (± pooled S.D.) canopy height across all remaining forest (m)	19.47 m ± 4.26 (0 – 34.10 m)	Lang et al. (2023)
RPA Forest Cover (RPA Compliant)	Property	Forest cover across all RPAs (%)	42.78% ± 32.85 (0 – 100%) RPA Compliant: Yes = 788 No = 27,998	SEMA-MT, MapBiomas 10 m LULC
Mean RPA Forest Canopy Height	Property	Mean (± pooled S.D.) canopy height across all remaining forest in RPAs (m)	19.37 m ± 4.06 (2.15 – 32.64 m)	SEMA-MT, Lang et al. (2023)
Forest Division Index	Property, Landscape	Probability any two forest cells originate from different forest patches	Property: 0.32 ± 0.26 (0 – 0.97) Landscape: 0.56 ± 0.30 (<0.01 – 0.98)	MapBiomas 10 m LULC
Edge Density	Landscape	Density of forest edges (m/ha of landscape)	37.71 m/ha ± 17.10 (3.15 – 120.01 m/ha)	MapBiomas 10 m LULC
Annual Forest Fire Density	Property	Annual proportion of forest burnt, 1985–2020 (%)	3.02% ± 12.96 (0 – 100%)	MapBiomas 30 m LULC & Fire Scars
Remaining Forest Burnt	Property	Proportion forest remaining in 2020 that had previously burnt (%)	28.48% ± 32.33 (0 – 100%)	MapBiomas 30 m LULC & Fire Scars

Table 2
Definitions and summaries of all property- and landscape-scale predictor variables, including the data source/s from which each metric was derived.

Variable	Scale	Definition (Unit)	Mean ± S.D. (Range)	Source
Property Size	Property	Area of property (ha)	234.71 ha ± 971.11 (0.71 – 39,484.52 ha)	CAR/ICV
Weighted Mean Property Size	Landscape	Mean Property Size, weighted by Property Size (ha)	2839.93 ± 4743.74 ha (50.99 – 39,484.52 ha)	
Property Age	Property	Year forest cover first fell below 97.5% (Year; N/A if property predated 1986)	26.01 yrs ± 6.52 (0 – 35 yrs)	MapBiomas 30 m LULC for 1985–2020
Predates 1986	Property	Was forest cover already below 97.5% in 1985?	Yes = 14,414 No = 15,608	
Weighted Mean Property Age	Landscape	Mean Property Age, weighted by Property Size. Properties predating 1986 assigned 38 years (Years)	31.05 yrs ± 7.24 (4.03 – 38 yrs)	
Distance to Road	Property	Minimum distance between property edge and public road in 2020 (km)	0.61 km ± 2.42 (0 – 85.07 km)	HOTOSM (2020)
Road Density	Landscape	Density of roads in landscape in 2020 (m/ha)	2.18 m/ha ± 1.71 (0 – 14.11 m/ha)	
Distance to Urban Area	Property, Landscape	Minimum distance to nearest urban area (km)	Property: 17.56 km ± 15.37 (0 – 154.48 km) Landscape: 28.66 km ± 21.81 (0.55 – 145.88 km)	MapBiomas 10 m LULC for 2020 30 m LULC for 1985–2020
Predominant Land-Use	Property, Landscape	Do pasture or crops cover majority of converted areas?	Property: Pasture = 27,303 Crops = 2719 Landscape: Pasture = 564 Crops = 55	MapBiomas 10 m LULC for 2020 30 m LULC for 1985–2020
RPA Density	Property, Landscape	Density of required RPAs (ha/ha)	Property: 0.09 ha/ha ± 0.07 (0 – >0.99 ha/ha) Landscape: 0.09 ha/ha ± 0.03 (0.02 – 0.18 ha/ha)	SEMA-MT
In ARS	Property	Is property part of an ARS?	Yes = 19,691 No = 10,331	CAR/ICV
Overlaps ARS	Landscape	Does landscape contain ≥ 1 properties belonging to an ARS?	Yes = 169 No = 450	
Property Coverage	Landscape	Proportion of landscape covered by known properties (%)	96.47% ± 4.41 (80.36 – 100%)	
Proportion Surrounding Burnt	Property	Proportion area burnt in a 1 km radius around each property in each year (%)	6.44% ± 11.96 (0 – 100%)	MapBiomas 30 m LULC and Fire for 1985–2020

Fire Density model. (5) Predominant Land-Use: A categorical variable denoting which land-use type covered the greatest proportion of each property in 2020 (10-m data) and annually for 1985–2020 (30-m data), as the spatial and topographical requirements of croplands and pastures differ. (6) RPA Density: Proportion of each property covered by required RPAs. Greater RPA density may increase pressure to clear/degrade riparian forest due to spatial conflict with agriculture; may elevate fragmentation due to the requirement to preserve forest corridors; and may confer greater natural fire resistance due to associated waterlogged soils. (7) In ARS: Whether each property was part of an ARS. Because ARSs were created to provide land to economically vulnerable peoples, landowners within ARSs may face greater economic constraints than holders of autonomous properties of comparable size. (8) Latitude and (9) Longitude: The coordinates of the centroid of each property, used to control for natural spatial gradients in forest canopy height (Figure S1.4).

2.4. Landscape-level variables

We defined a novel methodology to group neighbouring properties into 'landscapes' of roughly equal area, while minimising inclusion of areas not covered by known properties. This involved grouping properties using a 10 km × 10 km grid across the PdA and snapping the boundaries of each group to the external edges of its outermost properties (Figure S1.8).

We quantified three landscape-level response variables (Table 1). (1) Landscape Forest Cover: Proportional forest cover in 2020. (2) Landscape Edge Density (m/ha): The density of forest edges, used to characterise landscape configuration capturing both forest disaggregation and patch complexity. (3) Landscape FDI: The Forest Division Index, providing a fragmentation measure that was comparable with our property-level metric and independent of forest cover.

Next, we derived eight landscape-level predictors, with comparable purposes to the property-level predictors (Table 2). We calculated mean property size and age within each landscape weighted by property size, as larger properties inherently influence a greater proportion of a landscape (Weighted Mean Property Size and Weighted Mean Property Age). For this purpose, properties predating 1986 were assigned an age of 38 years (i.e., the oldest any property could be accurately dated, plus three years). For each landscape, we then determined the density of roads (Road Density), predominant land-use type (Predominant Land-Use), density of RPAs (RPA Density), distance to the nearest urban area (Distance to Urban Area), and whether any constituent properties belonged to an ARS (Overlaps ARS). To further control for the inclusion of areas outside of known properties, we quantified the proportion of each landscape covered by properties (Property Coverage).

Table 3
Structure of all models. For property-scale models, the neighbourhood threshold distance of the CAR term included in the best performing model is shown. For zero-/zero-one-inflated Beta regressions, we identified the best performing neighbourhood threshold for each component separately: μ, p and q represent the Beta, zero-inflation, and one-inflation components, respectively.

Response Variable (Unit)	Scale	Model Family (Link)	Random Effects (Neighbourhood Structure)	Fixed Effects
Forest Cover (%)	Property	Zero-Inflated Beta (Logit)	CAR term (μ: 250 m; p: 10 m)	Log_{10} Property Size, Predominant LU, Property Age, Predates 1986, Distance to Urban Area, Distance to Road, In ARS
LR Compliant?	Property	Logistic (Logit)	CAR term (10 m)	
RPA Forest Cover (%)	Property	Zero-One-Inflated Beta (Logit)	CAR term (μ, p, q: 10 m)	Log_{10} Property Size, Predominant LU, Property Age, Predates 1986, Distance to Urban Area, Distance to Road, In ARS, RPA Density
Forest Division Index	Property	Zero-Inflated Beta (Logit)	CAR term (μ: 500 m; p: 10 m)	
Mean Forest Canopy Height (m)	Property	Gaussian (Identity)	CAR term (10 m)	Log_{10} Property Size, Predominant LU, Property Age, Predates 1986, Distance to Urban Area, Distance to Road, In ARS, RPA Density, Latitude:Longitude
Mean RPA Forest Canopy Height (m)	Property	Gaussian (Identity)	CAR term (10 m)	
Landscape Forest Cover (%)	Landscape	Beta (Logit)	iCAR term (Rook's Case)	Log_{10} W. Mean Property Size, W. Mean Property Age, Predominant LU, Distance to Urban Area, Road Density, Overlaps ARS, RPA Density, Property Coverage
Landscape Edge Density (m/Ha)	Landscape	Gaussian (Identity)	iCAR term (Rook's Case)	
Landscape Forest Division Index	Landscape	Beta (Logit)	N/A	
Annual Forest Fire Cover (%)	Property	Zero-One-Inflated Beta (Logit)	Year Property ID	Log_{10} Property Size, Predominant LU, Property Age, Predates 1986, Distance to Urban Area, Distance to Road, In ARS, RPA Density, Proportion Area Burnt in 1 km Radius
Remaining Forest Burnt (%)	Property	Zero-One-Inflated Beta (Logit)	CAR term (μ: 10 m; p: 1 km; q: 10 m)	Log_{10} Property Size, Predominant LU, Property Age, Predates 1986, Distance to Urban Area, Distance to Road, In ARS, RPA Density

2.5. Statistical analyses

We used separate regressions to model each response (Table 3). We excluded properties containing no forest in 2020 from canopy height, FDI and Remaining Forest Burnt models; excluded properties without required RPAs from RPA Forest Cover and Mean RPA Forest Canopy Height models; and excluded property-years with no remaining forest from Annual Forest Fire Density models. We checked for correlations between all continuous predictors but found no highly correlated variables (all $|r| < 0.7$). We subsequently scaled and centred all continuous predictors.

We modelled response variables on the (0,1) interval using Beta regressions, responses on the [0,1) interval using zero-inflated Beta regressions, and responses on the [0,1] interval using zero-one-inflated Beta regressions. Zero- and zero-one-inflated Beta regressions represent hurdle models, where observations on the interval (0,1) are modelled using Beta regression, while the probability of observing a 0 and conditional probability of observing a 1, $P(y = 1|y > 0)$, are modelled using logistic regressions (Liu and Kong, 2015). We used Gaussian regressions to model all continuous positive response variables; we also considered using a log-link, but the identity link provided a reasonable fit without producing negative expected values. To propagate uncertainty in canopy height estimates, we scaled the model precision for each observation by the inverse of $\sigma_{Pooled,i}$ (Table 3).

We included as fixed effects in each model all predictors hypothesised to influence the response (Table 3). To determine the optimal format of the property size effect in models of each response (linear, $\log_{10}$-transformed, quadratic), we compared model versions based on the Watanabe-Akaike Information Criteria (WAIC), retaining the model with the lowest WAIC. Next, we determined whether including a spatial random effect improved model performance, as land-use changes are known to spread in a contagion-like manner within Amazonia (Rosa et al., 2013; Richards, 2018). We considered the inclusion of a Conditional Autoregressive term (CAR term) in property-level models and an intrinsic Conditional Autoregressive term (iCAR term) in landscape-level models, as CAR terms routinely caused overfitting. For property-level responses, we fitted five spatial models based on an adjacency graphs with neighbourhood threshold distances of 10, 100, 250, 500, 1000 m, as the exact scale of spatial dependency was unknown. For landscape-level models we used a rook's case adjacency graph. The model with the lowest WAIC (spatial or non-spatial) was retained for inference. Model selection was conducted separately for each component of zero- and zero-one-inflated Beta regressions (Table 3).

Annual Forest Fire Density models including a CAR term failed to converge. Therefore, we instead considered the inclusion of a fixed effect of the proportion area burnt within a 1-km radius of each property-year (this was well correlated with fire cover in 100-m to 10-km radii; Table S1.3). We also included random effects of Property ID, to control for temporal autocorrelation within properties, and Year, to control for temporal variation in other factors known to influence forest fire prevalence (e.g., climate; Noble et al., 2025).

We fitted our models using the Integrated Nested Laplace Approximation via the 'INLA' R package (v24.06.27; Rue et al., 2009), which provides approximate Bayesian inference. We always used the default priors. We assessed model fits using posterior predictive checks and residual diagnostics and inferred 'significance' of coefficients whenever their 95% posterior Credible Intervals (CIs) excluded zero. Full details provided in Appendix S1.

2.6. Sensitivity checks

To verify that the observed property size relationships were not artefacts of our selection criteria or covariate estimation, we performed a suite of sensitivity checks, repeating all property-level models while varying the properties or covariates included. These checks fell into three categories:

- *Sample Dependence*: To ensure trends were not biased by our selection criteria (see Section 2.2), we repeated our models including all 31,698 known properties in the PdA.
- *Property Age Definition:* As we could only *estimate* property age and could not accurately date properties before 1986, we repeated our models: (1) only including properties whose age could be reliably estimated (i.e., excluding pre−1986 properties); (2) using a more sensitive rule for defining Property Age and Pre−1986 (first year forest cover dropped below 100% and decreased by ≥0.1% from previous year); and (3) using a more conservative criteria (forest cover <95%, decrease ≥2%).
- *Property Accessibility:* To evaluate whether omitting Distance to Road from the Annual Forest Fire Density model biased our results, we

repeated all other models without Distance to Road and re-estimated the annual fire-density model without Distance to Urban, allowing us to assess whether controlling for accessibility influenced the observed relationships.

2.7. Regional summaries

To characterise relationships between property size, deforestation and forest fires across the entire PdA, we binned all properties into five size classes: very small (≤150 ha), small (150 - 400 ha), medium (400 - 1000 ha), large (1000 - 2500 ha) and very large (>2500 ha). These groupings align with the Forest Code definition of smallholdings (typically ≤400 ha within Amazonia; Michalski et al., 2010). For each class, we then calculated the: 1) total area deforested; 2) total area deforested within RPAs; 3) total area of forest remaining in 2020 that had previously burnt; and 4) total area of forest burnt at least once since 1985. We also expressed these statistics as proportions (e.g., proportion deforested) and quantified the contribution of each size class to the regional total area affected (e.g., proportion of the total area deforested across all properties). As forest cover varied temporally, we calculated the proportion of forest burnt in each year separately. We did not create regional summaries of forest fragmentation, as our landscape-scale analyses performed a similar function, nor mean forest canopy height, as natural spatial gradients in canopy height would confound inference on property size effects.

3. Results

The 30,022 properties analysed ranged between 0.71 and 39,484.52 ha (median = 51.75 ha; IQR = 29.12–109.55 ha) and covered a total of 7.03 Mha, or 60.4% of the PdA. Most properties were ≤ 150 ha (23,967; 79.8%). Small properties accounted for the next largest proportion (3477; 11.6%), while medium (1134), large (996) copand very large properties (448) accounted for 3.8%, 3.3% and 1.5% of landholdings, respectively (Fig. 1b).

3.1. Overall forest cover

By 2020, PdA landowners had deforested a total of 3.49 Mha within their properties, leaving 50.4% forest cover. On average, each property retained 21.6 ± 23.8% forest cover. However, by all measures considered here, forest retention increased with property size (Fig. 2a-d).

At the regional scale, proportional forest cover consistently increased with property size, from 20.0% across all very small properties to 65.0% across all very large properties. Very small properties also accounted for the largest share of regional forest loss (27.6%, 0.97 Mha), despite representing only 17.3% (1.22 Mha) of the total landholding area. For comparison, very large properties accounted for 25.7% (0.91 Mha) of regional deforestation but covered more than twice the area of very small properties (2.59 Mha, 36.9%). Small properties also made a disproportionately large contribution to regional deforestation (0.59

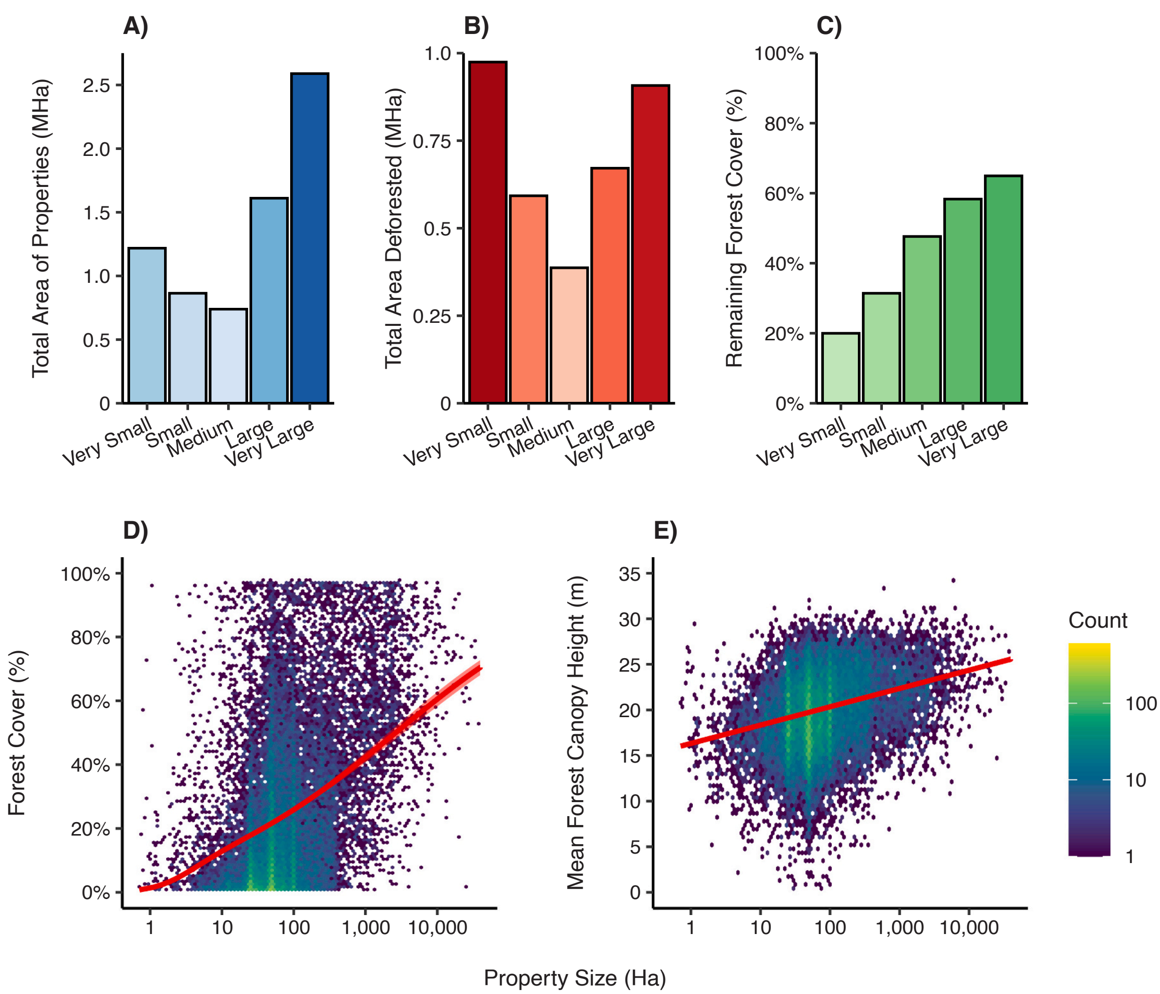

Fig. 2. A) Total area of and B) total area deforested across all properties in each size class, in 2020. C) Proportional forest cover across all properties in each class. D) Average marginal effect of property size on property-level forest cover. E) Average marginal effect of property size on mean canopy height of remaining forest in each property. In D and E, solid red lines represent the means, and red shaded areas the 95% Credible Intervals, of the modelled relationships.

Mha, 16.8%) relative to their total area (0.86 Mha, 12.3%). Medium-sized properties accounted for a roughly equal fraction of deforestation (10.9%, 0.39 Mha) and total property area (10.5%, 0.74 Mha), while large properties made a disproportionately small contribution, accounting for 22.9% (1.61 Mha) of private lands and 19.0% (0.67 Mha) of deforestation (Fig. 2a-c; Table S2.1).

Regional-scale deforestation trends were reflected in our property-level forest cover model: among properties that retained some forest, forest cover increased with $\log_{10}$ property size, and smaller properties were more likely to be completely deforested (Table S2.2). With all other variables held constant, 100, 1000 and 10,000 ha properties were expected to retain 25.9%, 42.3% and 60.6% forest cover, respectively (Fig. 2d). Accordingly, larger properties were more likely to comply with the 80% LR requirement (Table S2.3). However, only 1113 (3.7%) properties retained $\geq$ 80% forest cover, while more than twice as many (8.4%, 2533) were entirely deforested (Fig. 2d).

The mean canopy height of remaining forest was positively associated with property size. Specifically, a ten-fold increase in property size was associated with an average 2.01-m increase in mean forest canopy height (Fig. 2e; Table S2.4).

3.2. Riparian preservation areas

In total, 27,998 properties contained legally required RPAs. These areas covered 589,223 ha in total, 207,969 ha of which had been deforested by 2020, leaving 64.7% forest cover. On average, landowners retained 42.8% ± 32.9 forest cover within their required RPAs. Only 788 (2.8%) landowners fully complied with the RPA requirements, while over three-times as many (2764, or 9.9%) completely deforested their RPAs.

Regional scale RPA forest cover consistently increased with property size, nearly doubling from the smallest (39.9%) to the largest (76.9%) property size class. Very small properties accounted for 39.88% of regional deforestation within RPAs, the largest amount of any class, despite containing only 19.5% of the total RPA area. Conversely, large and very large properties made disproportionately small contributions to riparian forest loss, accounting for 15.3% and 23.0% of RPA deforestation but 20.85% and 35.2% of the regional RPA area, respectively (Fig. 3a-c; Table S2.5).

Similar patterns were apparent at the property-level: 87.3% of properties retained some RPA forest cover, and among these, forest retention increased with property size. Holding all other variables constant, our model indicated that RPAs in 100, 1000 and 10,000 ha properties were expected to comprise 51.2%, 63.8%, and 74.5% forest, respectively. However, RPA forest cover was negatively associated with property size among properties $<$ 3 ha (Fig. 3d). This bifurcation reflects the relative magnitude of property size effects in the model one- and zero-inflation components. Specifically, property size was positively

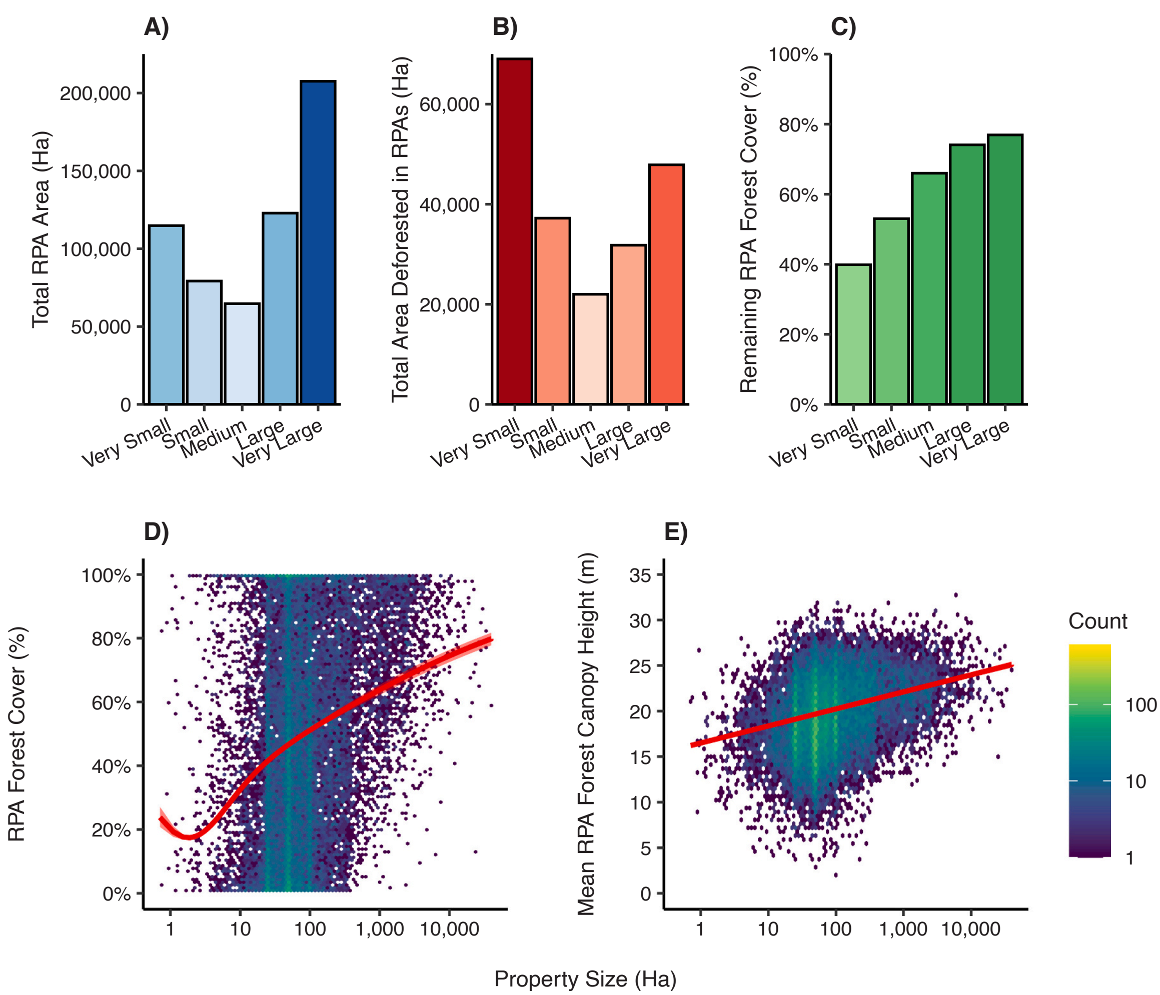

Fig. 3. A) Total area of and B) total area deforested within Riparian Protected Areas (RPAs) across all properties within each size class, in 2020. C) Proportional forest cover within RPAs across all properties within each class. D) Average marginal effect of property size on RPA forest cover within each property. E) Average marginal effect of property size on mean canopy height of remaining forest within RPAs in each property. In D and E, solid red lines represent the means, and red shaded areas the 95% Credible Intervals, of the modelled relationships.

associated with the probabilities of retaining both 100% and 0% riparian forest cover, with the latter effect being of lower magnitude (Table S2.6). Although seemingly paradoxical, this likely reflects the fact that riparian areas are typically smaller in smallholdings, making both complete clearing and complete preservation easier to achieve.

The mean canopy height of remaining forest within RPAs was positively associated with property size. Specifically, a ten-fold increase in property size was associated with a 1.87 m increase in mean riparian forest canopy height (Fig. 3e; Table S2.7).

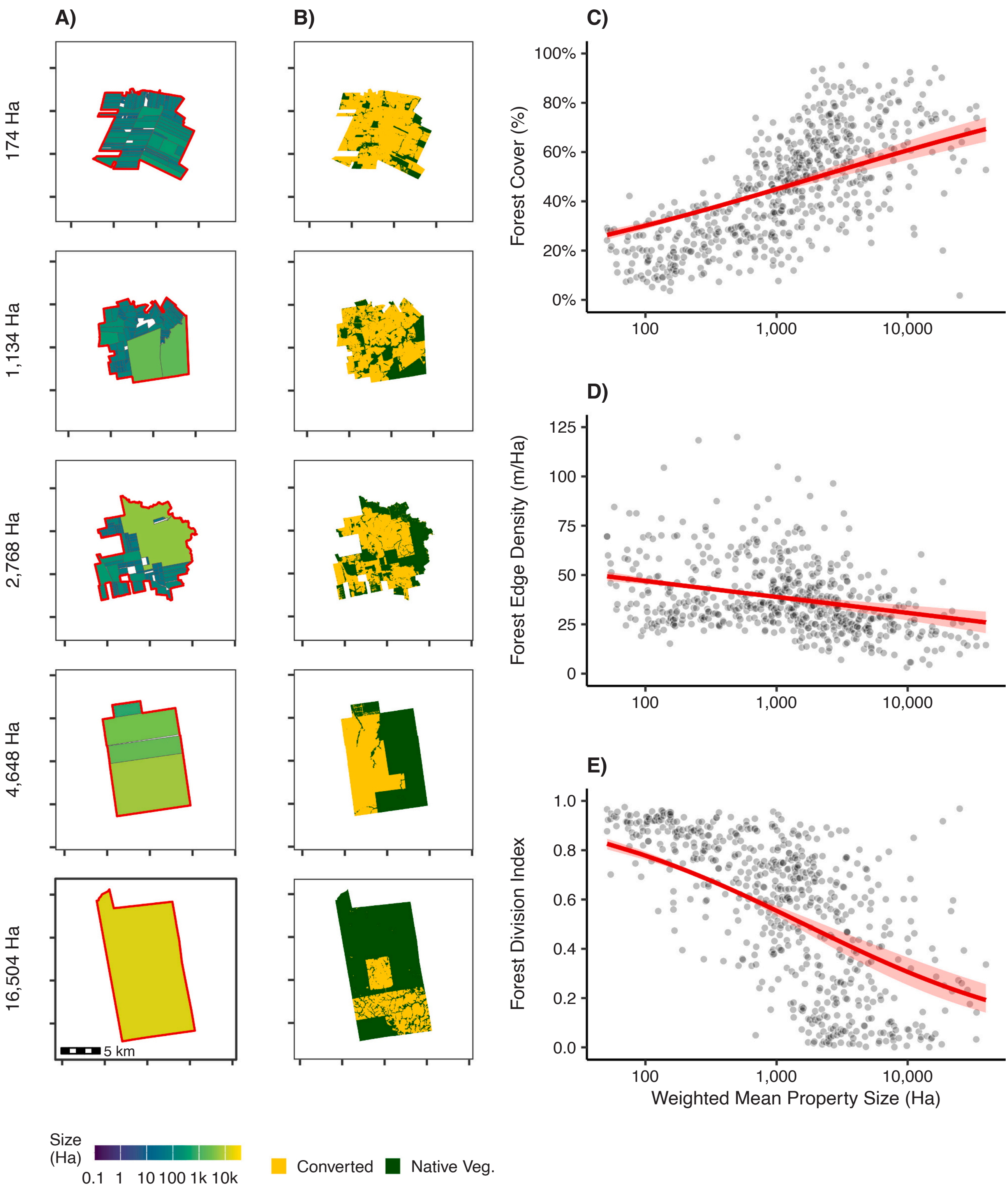

Fig. 4. A) Five examples of property groupings used to analyse landscape-scale patterns of forest fragmentation. Properties colour-coded according to their $\log_{10}$-transformed area. Landscape-level weighted mean property sizes are shown along the y-axes. B) Forest cover within the five example landscapes in 2020. Right-hand column: Average marginal effects of $\log_{10}$ weighted mean property size on landscape-scale C) forest cover, D) forest edge density and E) FDI. Solid red lines represent the means, and red shaded areas the 95% Credible Intervals, of the modelled relationships.

3.3. Forest fragmentation

Across the 27,521 properties that retained some forest, the median FDI was 0.30 (IQR = 0–0.55) and 9.0% of properties contained only one forest patch (i.e., had a FDI of 0). Smallholders were more likely to concentrate all remaining forest in one patch, but forest retention was typically low within these properties (median = 4.4%; IQR = 0.5–17.6%), and the observed lack of forest disaggregation may simply reflect the fact that there was little forest left to fragment. Among properties containing two or more forest patches, property size was not

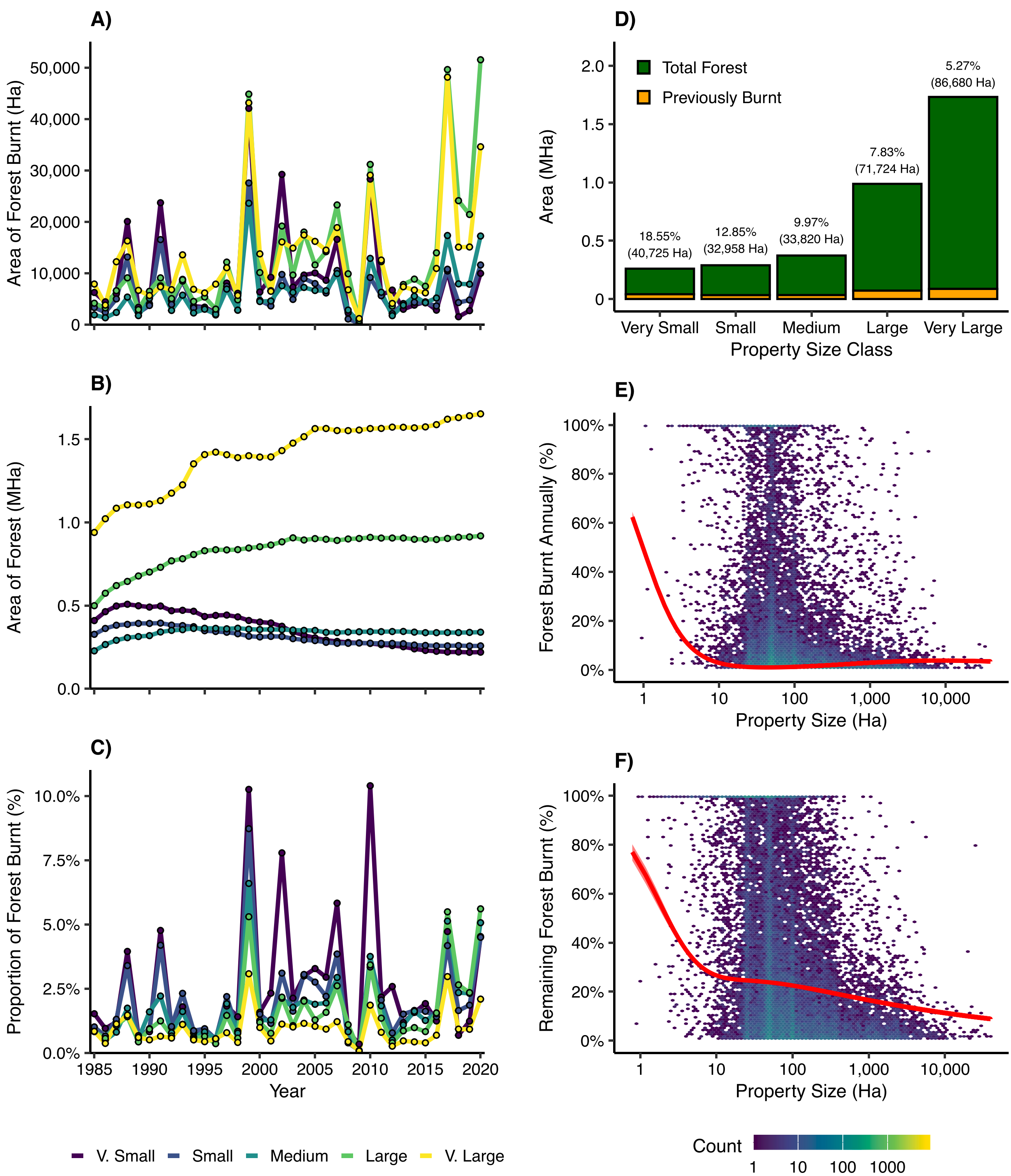

Fig. 5. Annual A) total area of forest burnt, B) total area of forest, and C) proportion of forest burnt across all properties in each size class between 1985 and 2020. D) Area and proportion of remaining forest across all properties within each size class in 2020 that had previously burnt. E) Average marginal effect of property size on the annual proportion of forest burnt within each property-year. F) Average marginal effect of property size on the proportion of forest remaining within each property in 2020 that had previously burnt. Solid red lines represent the means, and red shaded areas the 95% Credible Intervals, of the modelled relationships.

associated with forest disaggregation (Table S2.8). Overall, our model suggested that within-property forest disaggregation increased rapidly with property size from 1 ha (expected FDI = 0.04) to 50 ha (0.29) but only marginally thereafter (Figure S2.1).

We identified 619 property landscapes, with weighted mean property sizes ranging between 51.0 and 39,484.5 ha (median = 1325.0 ha, IQR = 438.7–2971.1 ha). Each landscape on average covered 99.21 ± 41.54 km^2, while property coverage averaged 96.5 ± 4.4% (Fig. 4a-b). Weighted mean property size was positively associated with landscape-level forest cover but negatively associated with forest edge density and forest disaggregation (Fig. 4c-e; Tables S2.9–11). All other variables held constant, our models suggested that landscapes with weighted mean property sizes of 100, 1000 and 10,000 ha were expected to contain 30.1%, 44.9%, and 60.7% forest cover; have forest edge densities of 46.97, 38.89 and 30.81 m/ha; and have LANDSCAPE FDI scores of 0.78, 0.55, and 0.31, respectively. Together these trends indicate that remnant forest cover tended to be broken up into smaller patches within landscapes dominated by smallholdings. In fact, comparing two equal-area landscapes, the landscape comprised of smaller properties would be expected contain less forest but more forest edges.

The negative association between weighted mean property size and LANDSCAPE FDI contrasts with the somewhat positive property-level relationship. Furthermore, within 68.1% of the 598 landscapes that contained multiple properties, the LANDSCAPE FDI was significantly higher than the size-weighted-mean FDI of the constituent properties (Weighted T-test: $p < 0.05$), with the magnitude of this difference negatively associated with mean property size (Figure S2.2). This suggests that the higher levels of forest fragmentation in smallholder-dominated landscapes was driven by the disaggregation of forest among, rather than within, properties.

3.4. Forest fires

In total, 1.56 Mha of forest burnt at least once between 1985 and 2020 across all properties, and annual forest fire cover averaged 48,185 ± 38,084 ha (1.5 ± 1.1%). Of the 1.45 Mha of forest burnt before 2020, 1.18 Mha (81.6%) was cleared. Nonetheless, 7.7% (265,907.16 ha) of remaining forest in 2020 had burnt at least once since 1985.

Properties > 1000 ha were the majority contributors to forest fire cover: large and very large properties accounted for 28.0% and 26.7% of forest burnt at least once between 1985 and 2020, and 27.1% and 32.6% of forest remaining in 2020 that had previously burnt, respectively. Very small, small and medium-sized properties accounted for 19.3%, 13.4% and 12.6% of forest burnt at least once between 1985 and 2020, and 15.3%, 12.4% and 12.7% of previously burnt remaining forest, respectively (Fig. 5a; Tables S2.12–13).

From 1996 onwards, large properties consistently contained around twice as much forest as the three smaller classes, while very large properties contained almost twice as much forest again throughout the time-series (Fig. 5b). Consequently, in contrast to areal fire cover, regional forest fire densities consistently decreased with property size. Median annual forest fire density between 1985 and 2020 was more than twice as high across all very small properties (1.7%) as across all very large properties (0.8%; Fig. 5c). Furthermore, the proportion of remaining forest in 2020 that had previously burnt was more than three-times higher across all very small properties (18.6%) than all very large properties (5.3%; Fig. 5d).

At the property-scale, property size was negatively associated with the 1) annual proportion of forest burnt and 2) probability of all forest being burnt, in property-years where forest fires occurred (20.8% of property-years); and the 3) proportion of remaining forest previously burnt and 4) probability of all remaining forest having previously burnt, among properties containing at least some previously burnt forest in 2020 (87.55% of properties). However, forest fires were more likely to occur in larger properties, and these properties were also more likely to contain some previously burnt forest in 2020 (Tables S2.14–15). Overall, expected annual forest fire density decreased rapidly with property size between 1 and ~45 ha, from 49.2% to 1.0%, but then increased slightly thereafter (density in 10,000 ha property = 3.8%; Fig. 5e). The expected proportion of remaining forest previously burnt decreased rapidly with property size between 1 and 10 ha, from 71.6% to 26.6%, and decreased at a slower rate thereafter (proportion in 10,000 ha property = 11.3%; Fig. 5f).

3.5. Sensitivity checks

The 95% CIs of the back-transformed PROPERTY SIZE coefficients (β) from our main models overlapped with those of the corresponding coefficients from every sensitivity analysis; the corresponding mean coefficient estimates were consistently similar; and all associations were qualitatively identical (i.e., variables showing a 'significant' positive association with property size in the main models also showed positive associations in every sensitivity check, and vice versa; Tables S2.16–18). This suggests that the property size relationships we report here reflect genuine trends, rather than artefacts of our selection criteria, our inability to date properties before 1986, the specific thresholds used to estimate property age, or our inability to fully control for property accessibility in our ANNUAL FOREST FIRE DENSITY model.

4. Discussion

We show that property size is a key determinant of regional forest disturbance dynamics within the PdA agricultural frontier, identifying clear associations with forest retention, canopy height, fragmentation, and fires. Larger properties retained more forest – both overall and within RPAs – and were more likely to comply with LR requirements. In fact, 75.2% of all remaining forest was concentrated in properties > 1000 ha, while smallholdings ≤ 150 ha cleared more forest than any other size class. Smaller properties were also associated with higher forest fire densities, greater landscape fragmentation, and shorter canopy heights – patterns long hypothesised in Amazonia but empirically demonstrated here for the first time (Oliveira-Filho and Metzger, 2006; Arima et al., 2016; Cammelli et al., 2020). Together these trends indicate that larger properties not only retain more forest, but that this forest is less degraded and thus likely to be of greater ecological value (Haddad et al., 2015; Lapola et al., 2023), be more resilient to climate change (Alencar et al., 2015; Giardina et al., 2018) and store more aboveground carbon per unit area (Rappaport et al., 2018; Gatti et al., 2021). From a governance perspective, our findings underscore the need to tailor policy interventions within Brazilian Amazonia according to property size (Assunção et al., 2015), combining strong protection of remaining forest within largeholdings with targeted policies that address the disproportionate contributions of smallholdings to forest clearance and degradation. Furthermore, as > 99% of landowners in the PdA – and 45% of landowners throughout Brazilian Amazonia (Rajao et al., 2020) – are already non-compliant with the Forest Code, governance emphasis must be placed on forest restoration alongside preservation.

4.1. Forest retention and restoration

Many of the observed associations between property size and forest disturbance likely reflect persistent financial inequalities between smallholders and largeholders, including the greater deforestation observed on smaller properties. Land-use revenue in post-colonial Amazonian frontiers mainly arises from non-forest activities, meaning forest retention incurs opportunity costs (Garrett et al., 2017). This is exacerbated by land conversion overheads that do not necessarily scale linearly with area and low agricultural productivity, with Amazonian cattle pastures typically yielding < 1 head/ha (Antonaccio et al., 2018). While these constraints affect properties of all sizes – perhaps explaining the generally low levels of LR and RPA compliance observed (3.7% and

2.8%, respectively) – they disproportionately limit smallholders' capacity to retain high forest cover while maintaining long-term viability (Michalski et al., 2010; Assunção et al., 2015). Largeholders – typically established agribusinesses that have historically received greater agricultural subsidies (Binswanger, 1991; Oliveira and Marquis, 2002) – likely have greater capacity to absorb start-up costs and farm a smaller proportion of their land, in expectation of long-term profitability.

We note that the resolution of the LULC data (10 m) may have limited our ability to discretise changes in forest cover across LR and RPA borders, leading to underestimates of compliance (Nunes et al., 2019). However, many previous studies in Brazil have also reported clear positive associations between property size, forest/native vegetation retention and Forest Code compliance, not just in Amazonia (D'Antona et al., 2006; Michalski et al., 2010; Godar et al., 2012; Okida et al., 2021; Cabral et al., 2022) but also in the neighbouring Cerrado biome (Richards and VanWey, 2016; Stefanes et al., 2018). Our results thus appear to be consistent with a widespread pattern within Brazil's major deforestation hotspots, in which financial constraints fundamentally limit smallholders' forest retention capacity.

Notably, Cabral et al. (2022) found that although properties > 2500 ha retained the most forest they also deforested the greatest area throughout Brazilian Amazonia. This discrepancy from our own findings likely reflects the lower areal representation of > 2500 ha properties in the PdA (36.9% vs. 46.3% in Brazilian Amazonia), alongside differences in frontier age (Godar et al., 2014), commodity prices (Probst et al., 2020), and land-tenure security (Pacheco and Meyer, 2022). Accordingly, we do not dispute that curbing deforestation within larger properties will also be key to slowing deforestation across Amazonia. Crucially, however, smallholdings are now experiencing the fastest increases in both deforestation rates and absolute losses across Amazonia, while deforestation on largeholdings is decelerating (Godar et al., 2014; Cabral et al., 2022). As a mature, consolidated frontier, the PdA may therefore foreshadow a broader shift in deforestation toward smaller properties in newer frontiers.

Targeted subsidies could ease smallholder financial constraints and reduce net forest loss, particularly if conditioned on environmental targets (Wunder, 2007; Börner et al., 2014; Pinho et al., 2014). This approach has shown promise: in 2008, Brazil's Central Bank restricted rural credit based on Forest Code compliance, contributing to marked reductions in Amazonian deforestation (Assunção et al., 2020). However, the 2012 legislative revision of the Forest Code weakened these incentives for smallholders, granting amnesties to properties ≤ 4 fiscal modules ($\sim\leq 400$ ha in Amazonia) with pre−2008 LR deficits and reducing restoration requirements for RPAs cleared before 2008, scaled by property size (e.g., properties ≤ 400 ha need only restore ≤ 15 m buffers, versus up to 500 m previously). In the PdA, $> 99\%$ of properties ≤ 400 ha began deforesting before 2008, and these collectively accounted for 54.05% (1.15 Mha) and 51.09% (0.11 Mha) of the regional LR and RPA deficits in 2020, respectively. The 2012 revision has thus severely eroded legal incentives to restore forest across the PdA and Amazonia more broadly (Soares-Filho et al., 2014). Furthermore, while many Amazonian landowners are legally required to balance deficits, estimates suggest only 6% of these are actively restoring forest (Azevedo et al., 2017). Compliance-based incentives alone therefore appear insufficient, highlighting the need for complementary interventions to promote restoration among both obligated and exempt landowners.

Tiered credit schemes – providing upfront financing with declining interest or debt as incremental restoration or environmental targets are met – could align livelihood security with conservation outcomes, regardless of current Forest Code compliance. Although this would initially finance illegal deforesters, given the high cost of active restoration in Amazonia (US$536–1327 ha^{-1}; Stickler et al., 2013), aligning final credit-based targets with Forest Code requirements could ultimately strengthen the law by lowering the barriers to compliance for low-income smallholders. This could be complemented by payments for ecosystem services (PES) schemes, which have achieved some success in Brazilian Amazonia: Bolsa Verde (a national, state-funded PES) and Bolsa Floresta (an NGO-run PES in Amazonas) provide regular payments to smallholders in exchange for maintaining standing forest and adopting sustainable land uses, reducing deforestation and poverty in participating areas (Cisneros et al., 2022; MacNeill and Drummond, 2025). While these schemes have focused on primary forest retention, a similar approach could be used to incentivise retention of restored forest once properties achieve compliance.

At all stages, PES schemes could help PdA smallholders shift from cattle ranching to specialty (e.g., citruses) and staple crops (e.g., maize, beans), which generate returns per hectare orders of magnitude greater than cattle production (Garrett et al., 2017), reducing the land needed to remain viable. However, the biodiversity consequences of such transitions in Amazonia are also determined by management practices, rather than land-use type alone. Low-intensity, well-managed pastures that retain structural heterogeneity and maintain connectivity with forest remnants can be less harmful to biodiversity than simplified, input-intensive cropping systems (Moura et al., 2013; Solar et al., 2015; Estrada-Carmona et al., 2022). Correspondingly, transitions from pasture to mechanised agriculture have been associated with net biodiversity declines in some Amazonian taxa, albeit smaller than those associated with all other common land-use changes in the region (Nunes et al., 2022). Although the higher per-hectare returns of crops may help spare land and create opportunities for forest restoration, these gains must be weighed against the biodiversity costs of potentially intensified management within croplands. Pasture-to-crop transitions thus should not be framed as inherently beneficial, and PES and credit schemes should instead incentivise improved management alongside carefully targeted land-use transitions, with the optimal strategies evaluated at local scales. In this context, further research is needed to identify the conditions under which shifts to cropland deliver net benefits for forest amount, resilience and biodiversity.

Regardless of the exact conditions of tiered credit and PES programs, educating landowners on the tangible benefits of forest cover could help increase uptake. For example, many Amazonian landowners value riparian buffers for irrigation, shade, and reliable water for cattle (Norris and Michalski, 2009), which may explain why 79.3% of PdA properties had higher RPA than overall forest cover. Promoting awareness of upland forest ecosystem services – including pollination, pest control, soil fertility, and reduced fire risk – could similarly encourage broader forest retention and restoration (Metzger et al., 2019; Pinillos et al., 2021). These messages could be delivered by the same government agencies or NGOs administering credit or PES programs.

While conditional credit and PES schemes could also incentivize largeholders to preserve and restore forest, their greater financial resilience and access to private credit is likely to limit uptake (Assunção et al., 2020). Because large properties tend to have closer links to domestic and international markets, they are likely to be more responsive to legal pressures and market risks, making punitive enforcement potentially more effective. In the 2000s, larger properties were disproportionately targeted by both violation notices (Dalla-Nora et al., 2014) and the beef and soy moratoria (Nepstad et al., 2014), and subsequently made disproportionately large contributions to Amazonia's deforestation slowdown (Godar et al., 2014), consistent with fines and embargoes being effective in promoting forest retention on largeholdings. Furthermore, previous work suggests that 76% of Amazonian landowners would only engage in forest restoration if compelled by fines or market pressures (Azevedo et al., 2017). Fines could therefore both incentivise largeholders to restore forest cover deficits and indirectly encourage smallholders to pursue credit-backed restoration, provided penalties are scaled to landowners' financial capacity to avoid exacerbating existing inequalities. This 'carrot-and-stick' approach has already shown success in motivating Amazonian landowners to develop restoration plans, but effectiveness depends on consistent enforcement (Börner et al., 2015; Santiago et al., 2018). However, while the CAR has nominally enabled tracking of the actions of individual landowners, as of 2025, only 9% of property boundaries reported to the CAR had been ratified, although this presents a rapid increase from 3.3% in 2024 (Lopes et al., 2025). Therefore, while the means is in place, strengthening enforcement will require continued acceleration of CAR application processing. To help maximise immediate effectiveness of these efforts, municipal, state, and NGO involvement could help prioritise high-deforestation properties for CAR ratification.

Despite widespread illegal deforestation in the PdA, 1031 properties held LR surpluses totalling 107,811.31 ha – 84.45% of which was on properties $\geq$ 1000 ha – placing these forests at risk of legal deforestation and making their retention a policy priority. One potential approach is the sale of Environmental Reserve Credits, allowing properties with LR surpluses to sell credits to those with deficits (Soares-Filho et al., 2014); in a best-case scenario, this could offset 10% of the LR deficits of all properties $>$ 400 ha in the PdA (1.08 Mha). In February 2025, Brazil launched its first national Environmental Reserve Credit market, lowering barriers to participation. Given slow CAR ratification, local NGOs (e.g., ICV) could help identify surplus-holding landowners – particularly the 22.64% of $\geq$ 1000 ha that met the LR requirements – and promote engagement. NGOs or private entities could also directly purchase surpluses, as demonstrated by the SIMFlor programme (www.simflor.org), which will invest R$1 billion with the aim of protecting 500,000 ha of LR surpluses in the Amazon. Given the already severe extent of deforestation in the PdA, the region represents a particularly high-impact target for such investments.

4.2. *Forest degradation, fires and fragmentation*

The measures described above could also help reduce forest degradation, as financial constraints trap many Amazonian smallholders into degrading land-use cycles (Garrett et al., 2017), likely contributing to the lower stature and higher fire densities of forests in smaller properties. Reduced canopy heights often reflect timber or forest resource extraction to offset agricultural shortfalls, a strategy typically adopted by smallholders in Brazilian Amazonia when under acute cash constraints (Amacher et al., 2009) or where high forest retention limits agricultural revenue, as shown in Ecuadorian Amazonia (Vasco et al., 2017). Escaped agricultural fires further induce canopy tree mortality in adjacent forests (Pontes-Lopes et al., 2026), yet fire-based management is typically the only feasible option for smallholders due to high upfront costs of mechanisation (Guedes et al., 2012). While this has previously led to suggestions that smallholders disproportionately contribute to fire prevalence in Amazonia (Pivello et al., 2021), we now provide empirical evidence supporting this. Importantly, punitive fire restrictions tend to only exacerbate inequality without reducing fire use among smallholders (Carmenta et al., 2019). In contrast, PES and credit schemes could support fire-free mechanised management, reduce reliance on forest extraction, and promote crop production, as fires are more frequently used in, and escape from, pastures than croplands in Amazonia (Cano-Crespo et al., 2015) and forest fire densities in the PdA are negatively associated with crop cover (Noble et al., 2025).

However, while mechanisation and crop production increases potential returns, they can also increase vulnerability to escaped fires from surrounding areas (e.g., due to accumulated fuel loads), often nullifying revenue benefits for smallholders in fire-prone landscapes (Cammelli et al., 2020). Although our results suggest that a given forest hectare is more likely to be burnt if located within a smaller property – whether this was subsequently cleared or retained – properties $>$ 1000 ha account for the majority of forest fire cover throughout the PdA, mirroring patterns across the Arc of Deforestation (Schelly et al., 2021). Sustained reductions in forest fire prevalence will therefore require parallel efforts to curb fire use by largeholders and implement landscape-scale fire management, alongside smallholder-focused interventions.

Because largeholders generally have greater financial capacity and can afford mechanised management, the extensive burning on properties $>$ 1000 ha likely reflects cost-minimizing choices rather than necessity, unlike among smallholders. Therefore, as with forest retention, fines and embargoes may be most effective for reducing fire use in largeholdings, given that most fires in Amazonia are set illegally (Pivello et al., 2021). This requires stronger enforcement, though prioritising monitoring of large properties in the PdA could improve efficiency. Largeholders will also necessarily play a central role in any landscape-scale fire management strategy. The recent adoption of Integrated Fire Management (IFM) approaches in Brazil offers a promising approach by coordinating prevention, prescribed burning, suppression, and post-fire recovery across entire regions (Oliveras-Menor et al., 2025). IFM programmes have already helped reduce fire prevalence in the Brazilian Cerrado (Franke et al., 2024), a region with clear socio-economic similarities to Amazonia, and Chiapas, Mexico (Neger et al., 2024), although IFM schemes are context-dependent and thus requires tailored, local-to-regional scale coordination. In the PdA, an IFM scheme administered by NGOs (e.g., ICV) or municipal government/s, could facilitate cooperation between neighbouring properties to coordinate prescribed burns and install fire breaks to fragment fuel loads (Franke et al., 2024), and convey the health risks of fire (Carmenta et al., 2017; Damm et al., 2024). Furthermore, given that Amazonian forest fragmentation has been linked to increased forest fire prevalence (Armenteras et al., 2013), including the PdA (Noble et al., 2025), efforts to reduce forest fragmentation would likely further reduce fire prevalence.

Notably, the heightened forest fragmentation in smallholding-dominated landscapes of the PdA likely reflects a lack of landscape-scale coordination rather than financial constraints, as it was mainly driven by the disaggregation of forest among, rather than within, properties. This reinforces long-standing theory linking forest configuration to agricultural colonisation processes and property boundary density within Brazilian Amazonia (Oliveira-Filho and Metzger, 2006; Arima et al., 2016) and Cameroon (Mertens and Lambin, 1997) and aligns with empirical evidence linking smallholding-dominated landscapes to increased fragmentation in the Argentine Chaco (Gasparri and Grau, 2009). In contrast, however, Godar et al. (2014) reported lower fragmentation in smallholding-dominated census tracts across Brazilian Amazonia, but this may reflect their location in newer frontiers where agricultural expansion is still ongoing. Trends in the PdA are more representative of landscape configuration after full agricultural consolidation, although similar studies in other Amazonian regions would be valuable. Nonetheless, the consistency of our results with longstanding hypotheses and cross-regional empirical evidence suggests that the disaggregation of forest across many small properties is a structurally predictable driver of landscape fragmentation.

Although there is debate around the nature of fragmentation impacts generally (Fletcher et al., 2018; Fahrig et al., 2019), small Amazonian forest patches host fewer species across numerous taxa (Benchimol and Peres, 2013; Palmeirim et al., 2021; Noble et al., 2023) and fragmented Amazonia landscapes are more susceptible to drought and fires (Broadbent et al., 2008; Armenteras et al., 2013; Noble et al., 2025). Reducing forest fragmentation will require coordinated landscape planning across property boundaries, particularly among smallholdings. The optimal approach would be to reduce forest cover deficits and forest fragmentation simultaneously, by connecting forest remnants across property boundaries via incentivised, collaborative restoration, ensuring continuity of LRs. Ideally, landowners would also maximise the width of any cross-property forest interfaces to minimise edge effects (e.g., by centring LRs around shared vertices where multiple property boundaries intersect) – this would likely benefit average canopy heights, which are known to decline with proximity to the forest edge (Maeda et al., 2022). Multiple landowners could also collectively purchase and restore forest in abandoned agricultural lands or set-aside pre-existing forest areas. This could be encouraged by promoting the value of Environmental Reserve Credits, which could enable the maintenance of existing, sizable forest patches on largeholdings by combining the LR deficits of numerous properties into contiguous forest tracts. NGOs and local government would have a critical role in education of and mediation between smallholders.

Furthermore, rural investors often hold on to purchased large-holdings to then parcel these areas out at higher resale value. While research on land speculation in Brazil is limited and we cannot directly assess its effects, evidence from the USA suggests that parcelization typically reduces forest cover and increases fragmentation (Haines et al., 2010; Zhu and Liu, 2019), and this would appear to align with the observed property size associations throughout the PdA. This practice could be mitigated by encouraging landowners to sell forest as Environmental Reserve Credits instead of clearing or subdividing it. To compete with the potential profits from selling to agricultural entities, the price of these credits may need to be subsidised by the government or NGOs. Alternatively, NGOs could directly purchase such properties, sell the credits themselves or establish conservation units to ensure long-term forest persistence.

5. Conclusion

Property size was associated with all assessed aspects of forest disturbance in the PdA. Larger properties maintained more forest, with taller canopies, and were associated with lower landscape fragmentation, whereas smallholdings made disproportionate contributions to regional deforestation and forest fire prevalence, and landscape fragmentation was particularly pronounced where land was subdivided among numerous smallholdings. Policy must therefore be tailored by property size: smallholders, constrained by limited capital and low productivity, would likely benefit most from tiered restoration credits and payments for ecosystem services, which support forest retention, fire-free mechanised management, and transitions to higher-value crops; larger properties, with greater financial capacity, may instead be more responsive to fines, embargoes, and stricter Forest Code enforcement to curb illegal deforestation and fire use. Landscape-scale planning is essential to reduce fragmentation and forest fire prevalence, improving connectivity through collaborative restoration across property boundaries, aggregation of LR deficits via Environmental Reserve Credits, and the implementation of recently established IFM strategies. Effective implementation of these recommendations will require participation of multiple governance bodies: municipal and state agencies for enforcement and CAR ratification, federal programs to support credit and PES schemes, and NGOs to provide education, mediation, and direct forest protection. Tailored property-level interventions, combined with landscape-scale cooperation, offer a viable pathway to restore forests, reduce degradation, and sustain livelihoods throughout the entire PdA region. Furthermore, while we focused on a single frontier, many of the relationships observed in the PdA mirror patterns reported elsewhere in Amazonia and elsewhere and align with longstanding land-use theory, suggesting these dynamics may have far-reaching applicability. Nonetheless, we recommend similar studies in other tropical regions to assess the generality of our findings, particularly regarding links between property size and forest fires, degradation and fragmentation.

CRediT authorship contribution statement

Ciar D. Noble: Writing – review & editing, Writing – original draft, Visualization, Software, Methodology, Formal analysis, Data curation, Conceptualization. **James J. Gilroy:** Writing – review & editing, Supervision, Conceptualization. **Weslei Butturi:** Writing – review & editing, Data curation. **Carlos A. Peres:** Writing – review & editing, Supervision, Project administration, Conceptualization.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgement

C.D.N.'s PhD at the University of East Anglia is funded by a University of East Anglia Science Studentship. C.A.P. is supported by a Frontiers Planet Prize from the Frontiers Research Foundation. The authors would like to thank the staff of the UEA High Performance Computing Cluster (ADA), and Instituto Centro de Vida (ICV) for access to geospatial data.

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.landusepol.2026.108161.

Data availability

Data will be made available on request.

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