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Research article

# Small forest patches and landscape-scale fragmentation exacerbate forest fire prevalence in Amazonia

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ABSTRACT

Over recent decades, forest fire prevalence has increased throughout the tropics, necessitating improved understanding of the landscape-scale drivers of fire occurrence. Here, we use MapBiomas land-cover and fire scar data to evaluate relationships between forest fragmentation, land-use, and forest fire prevalence in a typically consolidated Amazonian agricultural frontier: Portal da Amazonia, Mato Grosso, Brazil. Using zero-/zero-one-inflated Beta regressions, we investigate effects of forest patch (area, shape, surrounding forest cover) and landscape-scale variables (forest edge length, land-cover composition) on forest fire occurrence and density between 1985 and 2021. We show that fire density was greatest in small, complex forest patches. Small patches ($\leq$100 ha) were also the dominant contributors to annual, regional forest fire cover. At the landscape-scale (100 km$^2$), forest edge length and urban land cover were positively associated with forest fire occurrence and density. Furthermore, forest fires were most likely to occur in landscapes consisting of ~45% pasture cover, while fire density increased roughly linearly with pasture cover. Cropland cover was negatively associated with forest fire occurrence and density. Our findings indicate clear links between forest fragmentation and increased forest fire prevalence. This is cause for global concern, given that fragmentation rates throughout Amazonia are increasing, and fires are eroding the Amazon's capacity to act as a carbon sink. Efforts to minimise further fragmentation within Amazonia would likely help reduce forest fire prevalence. Within already fragmented regions, the conversion of pasture into crops, alongside targeted efforts to suppress fires within small forest patches and urbanized areas, may also limit fire prevalence.

## 1. Introduction

Fire is an increasingly important driver of forest loss and degradation worldwide (Curtis et al., 2018; Tyukavina et al., 2022), and a major contributor to changes in biomass carbon dynamics throughout forest regions (McDowell et al., 2020). Amazonia, the world's largest contiguous tropical rainforest domain, historically experienced forest fires only rarely (Goulart et al., 2017; Feldpausch et al., 2022) but has succumbed to marked increases in fire-driven degradation in recent decades (De Faria et al., 2017). Consequently, Southeastern Amazonia is now emitting more carbon than it absorbs (Gatti et al., 2021; Fawcett et al., 2022). The management of fire risk in Amazonian landscapes is therefore a pressing global concern, urgently requiring improved understanding of the determinants of fire occurrence at local to landscape scales (Pivello et al., 2021).

Fire plays a key role in many Amazonian agricultural practices, including slash-and-burn, forest clearing, and pasture regeneration (Morton et al., 2008; Barlow et al., 2019). These human-initiated fires often encroach into neighbouring forests (Cano-Crespo et al., 2015; Brando et al., 2019), leading to more prevalent forest fires within fragmented landscapes of the Amazon, where the density of forest edges is higher (Armenteras et al., 2013; Alencar et al., 2015; Silva-Junior et al., 2018). Recent pressure for agricultural intensification has further promoted the use of fire-based land management practices (Eufemia et al., 2022; Gatti et al., 2023) and accelerated the rate of fragmentation throughout Amazonia (Montibeller et al., 2020). Given the link between fragmentation and fire susceptibility in forest landscapes, it is vital to improve our understanding of how the composition and configuration of fragmented forests influence the risk and severity of forest fires. Identifying relationships between fire prevalence and landscape characteristics can enable managers to better direct their efforts to minimise future fires (Morton et al., 2013; Driscoll et al., 2021; Rosan et al., 2022).

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The creation of forest edges is well known to reduce humidity and increase temperatures within adjacent forest (Broadbent et al., 2008; Meza-Elizalde and Armenteras-Pascual, 2021). Accordingly, there is growing concern that forest fragmentation may not only increase the interface between forest and agricultural lands, making the spread of fire into forests more likely, but also simultaneously reduce the resilience of tropical forest remnants to fire, resulting in larger and more severe forest fires (Cochrane, 2002; Driscoll et al., 2021). In fragmented regions of the Amazon, edge-related desiccating effects may penetrate 1000–2700 m into the interior of remnant rainforest patches (Briant et al., 2010), aligning well with the maximum depth of edge-related fires (2–3 km: Cochrane and Laurance, 2002; Armenteras et al., 2013). Given that drought frequency and severity are increasing throughout Amazonia (Barkhordarian et al., 2019; Boulton et al., 2022), fragmentation is most likely exacerbating a background trend towards hotter, drier conditions within standing forests (Leite-Filho et al., 2021; Nunes et al., 2022). Indeed, reductions in the size and increases in the perimeter-area ratio of forest patches are known to reduce their microclimate buffering capacity (Ewers and Banks-Leite, 2013), potentially rendering them even more vulnerable to fire (Cochrane and Laurance, 2002; Guedes et al., 2020). This prospect is particularly concerning given that the average size of forest patches is decreasing within forest regions throughout the tropics (Taubert et al., 2018; Montibeller et al., 2020).

Fire prevalence depends not only on environmental conditions but also on human activities (Achu et al., 2021), particularly whether landholders incorporate fire into land management (Cano-Crespo et al., 2015). Many factors may influence the decision to use fire-based management practices. For instance, as forest clearance using fire is typically inexpensive compared to any other means, low-income landowners with smaller economies of scale may be more likely to set fires. On the other hand, agribusinesses, which typically own some of the largest properties in the Amazon, may use fires to clear considerably larger areas of forest, especially within expanding agricultural frontiers (Pivello et al., 2021). Furthermore, fire-based management practices throughout the Amazon are more commonly used within pasturelands than croplands, and while pastures often consist of relatively combustible shrub vegetation, croplands may host a variety of fire-resistant plants (Cano-Crespo et al., 2015; Rabin et al., 2015). Also, given that most Amazonian wildfires originate from anthropogenic ignition sources (Pivello, 2011), human population density (and/or proximity to urban areas and roads) is also likely to be an important factor influencing fire activity (Price et al., 2014; dos Reis et al., 2021; Achu et al., 2021). The composition of non-forest habitats in areas surrounding remaining forests is therefore expected to play a significant role in influencing the prevalence of forest fires.

Here, we investigate the factors associated with the occurrence and extent of forest fires within the ~113,000-km$^2$ Portal da Amazonia (PdA) agricultural frontier region of southeastern Brazilian Amazonia. Based on land-cover and burn scar classifications derived from satellite imagery, we quantify associations between landscape composition, configuration, and forest patch morphology with forest fire prevalence across a 37-year time-series. During this period, the PdA was converted from nearly continuous forest into a mosaic of pasture, cropland, and isolated forest remnants. Deforestation within the region largely followed a fishbone pattern, where small agricultural properties branch off from main roads, although the region also includes some major landholdings, containing large forest remnants, and protected areas of continuous forest, as is typical of Amazonian agricultural frontiers (Oliveira-Filho and Metzger, 2006; Arima et al., 2016, Fig. 1). The PdA thus constitutes an ideal case study for investigating how common land-uses and deforestation/fragmentation patterns may be contributing to recent increases in forest fire prevalence throughout Amazonian agricultural frontiers (Cochrane, 2002; De Faria et al., 2017).

We separately analysed patterns of forest fire within the PdA at the landscape and patch-scale using Bayesian zero- and zero-one inflated Beta regression, respectively. This enabled us to simultaneously investigate the influence of fragmentation and land-use metrics on both the density (i.e., proportion of forest burnt) and occurrence (i.e., forest fire density >0) of fires, while controlling for spatial clustering of fires and spatio-temporal variation in climate. We consider four main questions (1) is landscape configuration (i.e., the degree of forest fragmentation): associated with the occurrence and density of forest fires? (2) how does the composition of non-forest areas affect the prevalence of forest fires? (3) how does forest patch size and shape affect the occurrence and density of forest fires? and (4) are small forest patches disproportionately affected by fire in terms of the area of forest burnt at the regional scale?

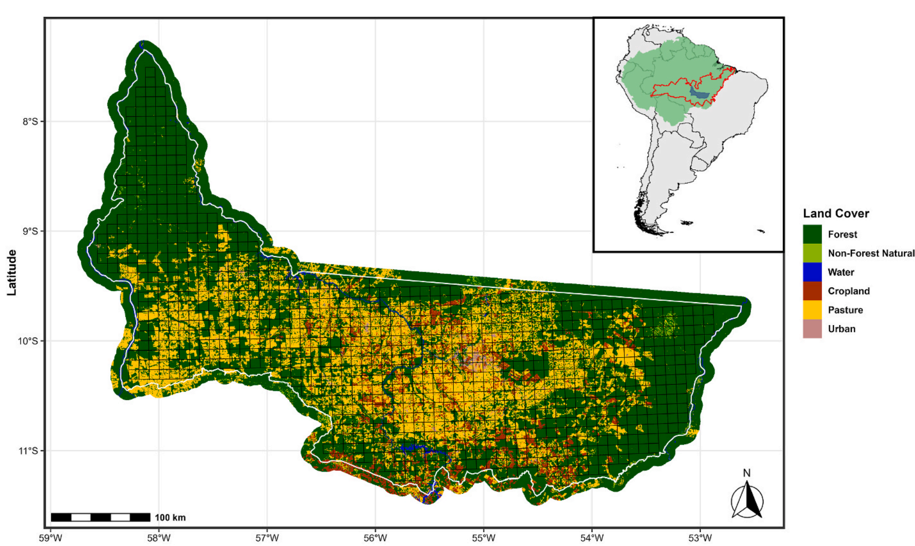


**Fig. 1.** Map of the land-cover composition of the Portal da Amazonia (PdA) region, northern Mato Grosso state, Brazil, in 2021. Land-cover is shown for the PdA (white border) plus a 10-km buffer in all directions. Black grid shows the 1006 10 km × 10 km quadrats (landscapes) into which the PdA was subdivided for our landscape-scale analysis. Inset map shows the position of the Portal da Amazonia (blue shading) within South America. The boundaries of the Amazon basin (green shading) and the 'Arc of Deforestation' within Brazilian Amazonia (red polygon) are also shown.

## 2. Methods

### 2.1. Study region

The PdA comprises 17 municipalities in northern Mato Grosso state, Brazil, spanning an area >113,000 km$^2$ at the centre of the Amazonian 'Arc of Deforestation' (Fig. 1). The region historically consisted almost entirely of continuous, tropical submontane *terra firme* forest, although small areas of savannah and grassland vegetation were also present (Oliveira-Filho and Metzger, 2006; Lees et al., 2013). However, in the late 1970s new roads connected the region to expanding agricultural frontiers further south, paving the way for agricultural resettlement programs and leading to widespread deforestation (Oliveira-Filho, 2001). Between 1985 and 2021 regional forest cover declined from 90.56% to 62.51%, with most deforestation occurring before the mid-2000s (forest cover in 2005 = 65.75%) (estimates derived from MapBiomas Annual 30-m Land-Cover classifications; Figures A.1-A.3; Souza et al., 2020). Forests were predominantly converted into cattle pastures, but also soybean, coffee, and fruit farms (IBGE, 2010; Gervazio et al., 2023).

In 2019 the PdA was home to ~282,000 people, with 32% of these residing in rural areas. Family farms account for 82% of all rural enterprises in the PdA and the region produces 5.7% (~R$8 million) of the Mato Grosso state gross domestic product, mostly through the dairy and livestock industries (Gervazio et al., 2023). The climate of the region is humid equatorial, with average annual temperatures of 24–26 °C and annual rainfall typically exceeding 2500 mm (RADAMBRASIL, 1983; Butturi et al., 2021). Elevations within the PdA range between 0 and 625 m (mean ± S.D. = 298 m ± 69.87; Valeriano and Rossetti, 2012). Soils are primarily composed of ultisols with some oxisols, as is typical in southeastern Amazonia (RADAMBRASIL, 1983). The region is bisected by the Rio Teles Pires, a primary tributary of the Rio Tapajós, which runs from the southern border to the northwestern tip of the PdA.

### 2.2. Data sources

#### 2.2.1. Fire and land cover data

We quantified the occurrence and density of fires within the PdA using MapBiomas Annual Fire Scar data (Collection 2), and classified land-cover using MapBiomas Annual Land-Cover data (Collection 7; Souza et al., 2020; brasil.mapbiomas.org), obtained at 30-m resolution for the 1985–2021 period (Figures A.1-A.5). MapBiomas classifies land-cover and fire scars based on monthly LandSat satellite images; for full methodological details of the land-cover classifications see Souza et al. (2020), and for the fire scar classifications see Alencar et al. (2022). Prior to analysis, we simplified the land-cover data by grouping all forest classes (henceforth, 'Forest'), and all classes of natural non-forest vegetation ('Non-Forest Natural'), open water ('Water'), cropland ('Crop'), pasture ('Pasture'), and urban areas and mines ('Urban') (Table A.1; Figures A.2, A.3).

#### 2.2.2. Climate data

Fires within Amazonia are highly seasonal, occurring almost exclusively within the dry season, and fire prevalence within many regions is highly correlated with the severity of dry season conditions (i.e., high temperatures, low precipitation) (Aragão et al., 2018; Carvalho et al., 2021; dos Reis et al., 2021). To control for spatio-temporal variation in dry season conditions, we obtained monthly precipitation and maximum daily temperature values for the 1985–2021 period from Terraclimate, at ~4 km resolution (Abatzoglou et al., 2018). We aggregated the monthly data to produce annual estimates of the mean maximum daily temperature of the driest quarter (Bio9) and total precipitation of the driest quarter (Bio17; Hijmans et al., 2005) (Figures A.5, A.6).

### 2.3. Forest fire density analyses

#### 2.3.1. Landscape-scale variables

To investigate how landscape composition and configuration influence patterns of forest fire, we divided the PdA region into 1006 10 km × 10 km (100 km$^2$) quadrats, or 'landscapes' (Fig. 1). We used 100 km$^2$ landscapes as these captured a wide range of landscape compositions and configurations, including areas of continuous forest and areas subject to varying levels of fragmentation (Fig. 1; Table 1) and have previously been used in comparable studies on associations between forest fragmentation and fire (Silva-Junior et al., 2018). Furthermore, previous research suggests that if the grid resolution is reasonable in comparison to the overall area of the study region, the exact cell size does not notably alter distribution of metrics characterizing fragmentation within Amazonian regions (Saito et al., 2011). We then quantified the density of forest fires within each landscape in each year (henceforth, 'landscape-year'), defined as the proportion of forest within each quadrat that had burnt.

We defined 10 landscape-scale predictor variables, calculated for each landscape-year using the corresponding annual land-use, fire, and

**Table 1**
Patch- and landscape-scale predictor variables considered for inclusion in our models of forest fire density, including the spatial extent over which each variable was extracted. The mean, standard deviation, and range of each variable are also shown; for the patch-scale metrics, summary statistics represent the subsampled dataset (i.e., the 185,000 forest patches that were retained for modelling of patch-scale forest fire density). Forest cover was not included in our models of landscape-scale forest fire density because it exhibited a near-perfect, negative correlation with pasture cover.

| Predictor Variable | Spatial Extent | Included in Model? | Mean ± S.D. (Range) |
|---|---|---|---|
| ***Patch-Scale Model*** | | | |
| Patch Area | Patch | Yes | 130.04 Ha ±870.65 (0.53–24,818.22 Ha) |
| Patch Shape | | Yes | 1.90 ± 0.52 (1.12–6.53) |
| Forest Cover | 1 km radius around patch, excluding patch itself | Yes, as Quadratic term | 35.43% ± 24.33 (0.00–99.43%) |
| Proportion of Surrounding Area Burnt | | Yes | 8.83% ± 17.24 (0.00–100.00%) |
| Bio9 (Mean Temp. of Driest Quarter) | 1 km radius around patch, including patch itself | Yes | 35.81 °C ± 1.09 (31.31–38.73 °C) |
| Bio17 (Total Precip. of Driest Quarter) | | Yes | 29.78 ml ± 21.23 (0.00–129.64 ml) |
| ***Landscape-Scale Model*** | | | |
| Total Edge Length | 10 × 10 km quadrat | Yes | 179.08 km ± 140.41 (0.00–1032.45 km) |
| Forest Cover | | No | 71.13% ± 28.16 (1.24–100.00 %) |
| Non-Forest Natural Cover | | Yes | 1.08% ± 2.74 (0.00–41.09 %) |
| Water Cover | | Yes | 0.40% ± 1.21 (0.00–30.20 %) |
| Crop Cover | | Yes | 1.51% ± 4.74 (0.00–59.53 %) |
| Pasture Cover | | Yes, as Quadratic Term | 25.57% ± 26.33 (0.00–96.59 %) |
| Urban Cover | | Yes | 0.35% ± 1.86 (0.00–40.91 %) |
| Bio9 (Mean Temp. of Driest Quarter) | | Yes | 35.57 °C ± 1.09 (31.81–38.70 °C) |
| Bio17 (Total Precip. of Driest Quarter) | | Yes | 30.99 ml ± 21.83 (0–127.62 ml) |
| Mean Proportion of Forest Burnt in Neighbouring Landscapes | 4 neighbouring 10 × 10 km quadrats | Yes | 1.04% ± 3.43 (0.00–64.58 %) |

climate data (Table 1). We quantified landscape composition as the percentage cover of each land-cover category. We quantified the configuration of forest within each landscape-year using the total length of forest edges (km); we also considered a variety of other configuration metrics but found that they were highly correlated with forest cover, pasture cover or total edge length (Table A.2). To control for climatic variation, we extracted the mean values of Bio9 and Bio17 for each landscape-year. Finally, as fires in the PdA exhibited spatial clustering, and the regions that experienced the greatest prevalence of fires varied among years (Figure A.2), we calculated the mean forest fire density estimates from all quadrats which shared a border with a given landscape in each year (i.e., rooks-case adjacency), to control for spatio-temporal autocorrelation.

### 2.3.2. *Patch-scale variables*

To investigate how the morphology and surroundings of individual forest patches influence their vulnerability to fire, we quantified the density of fires within each forest patch in each year (henceforth, 'patch-years'), measured as the proportion of each patch that was burnt. As private landowners in Brazil are required to set-aside forest buffers along rivers and streams, many forest patches were connected by riparian forest corridors (Fig. 1; Lees and Peres, 2008). We therefore delineated forest patches using the marker-controlled watershed transformation (Lefebvre et al., 2012), separating areas connected by corridors ≤150 m wide. A full description of this method is available in Noble et al. (2023). We subsequently excluded forest areas >25,000 ha, which we defined as continuous forests, and forest remnants <0.5 ha, as these typically represented extensions of larger forest patches that were artificially isolated due to the resolution of the land-cover data. Based on these criteria, we identified 1,537,825 patch-years within the PdA region across the 1985–2021 period.

We extracted seven patch-scale predictor variables using the corresponding annual land-cover, fire, and climate data (Table 1). We quantified patch area (ha) and shape, defined as the ratio between the longest chord of a patch and the diameter of a circle of equivalent area, whereby a value of one indicates a maximally compact (i.e., circular) patch, while higher values indicate proportional increases in patch complexity (Lefebvre et al., 2012). We opted to use this measure of patch shape as it is independent of patch size, thus enabling us to disentangle the effects of patch size and shape, whereas the perimeter-area ratio is negatively correlated with patch size (Malcolm, 1994). We measured surrounding forest loss as the percentage forest cover within a 1-km radius of each patch-year, and quantified the percentage area burnt within the same radial buffer, to control for surrounding fire prevalence. Finally, to control for climatic variation, we extracted the mean values of Bio9 and Bio17 within a 1-km radius of each patch-year (including the patch itself). We tested a series of other buffer sizes (0.25, 0.5, 2.5, 5 and 10 km), but the metric values from the 1-km radial buffers were highly correlated with those from all other buffer sizes (Table A.3).

To minimise spatial autocorrelation in patterns of fire among forest patches, we opted to randomly subsample our patch-year data, retaining 5000 patches from each year for analysis (185,000 patches in total). Subsampling was performed using an adaptation of the Metric Uniform Design Algorithm, outlined by Bowler et al. (2021), which aims to maximise both the representativeness of metric space and the geographic distance between retained sites (see Appendix B for details).

### 2.3.3. *Statistical analysis*

We analysed landscape- and patch-scale forest fire density separately in all cases. Prior to analysis, we tested for correlation among all predictor variables using Pearson's $r$. At the landscape scale, forest and pasture cover exhibited near-perfect negative correlation ($r = -0.97$) and, as we were interested in the effect of land-use *change* on forest fires, we retained only pasture cover for analysis. All other variables exhibited weak-to-moderate correlations ($r$: Landscape $\leq |0.56|$; Patch $\leq |0.14|$). We then scaled and standardised all remaining predictor variables.

Both our landscape- and patch-scale measures of forest fire density were [0,1] bounded and included many zeroes (i.e., absence of fire), while our patch-scale measures also included many ones (i.e., entire patch burnt) (Figure C.1). Considering this, we opted to model landscape-scale measures of forest fire cover using a zero-inflated beta distribution (Ospina and Ferrari, 2012):

$$f\left(y_{S=L,i,j} \,\middle|\, \eta_{S=L,i,j}\right) = \begin{cases} \left(1 - p_{S=L,i,j}\right) & \text{if } y_{S=L,i,j} = 0 \\ p_{S=L,i,j}\text{Beta}\left(\mu_{S=L,i,j}, \phi_{S=L}\right) & \text{if } y_{S=L,i,j} \in (0,1) \end{cases}$$

and our patch-scale measures of fire cover using a zero-one-inflated beta distribution (Liu and Kong, 2015):

$$f\left(y_{S=P,i,j} \,\middle|\, \eta_{S=P,i,j}\right) = \begin{cases} \left(1 - p_{S=P,i,j}\right) & \text{if } y_{S=P,i,j} = 0 \\ p_{S=P,i,j}q_{S=P,i,j} & \text{if } y_{S=P,i,j} = 1 \\ p_{S=P,i,j}\left(1 - q_{S=P,i,j}\right)\text{Beta}\left(\mu_{S=P,i,j}, \phi_{S=P}\right) & \text{if } y_{S=P,i,j} \in (0,1) \end{cases}$$

Here, $S$ represents the scale of analysis (Landscape = $L$; Patch = $P$); $y_{S,i,j}$ represents the density of forest fires within landscape or patch $i$ in year $j$; $\mu_{S,i,j}$ and $\phi_S$ represent the mean (i.e., the expected value) and variance of the beta distribution, respectively (Ferrari and Cribari-Neto, 2004); $p_{S,i,j}$ represents the probability of observing a non-zero value; and $q_{S=P,i,j}$ represents the conditional probability of observing a one, i.e., $P\left(y_{S=P,i,j} = 1 \,\middle|\, y_{S=P,i,j} \neq 0\right)$.

The above distributions enabled us to simultaneously investigate the factors that influence: 1) the density of forest fires where they occur $\mu_{S,i,j}$; 2) the probability of forest fire occurrence $p_{S,i,j}$; and 3) the probability of an entire forest patch being burnt $q_{S=P,i,j}$. Therefore, we modelled $\mu_{S,i,j}$, $p_{S,i,j}$ and $q_{S=P,i,j}$ as the outcomes of a series of fixed and random effects, using logit link functions (Table 1):

- *Landscape-Scale:* we modelled $\mu_{S=L,i,j}$ and $p_{S=L,i,j}$ against linear effects of each landscape-scale predictor. As forest fire prevalence may vary non-linearly with pasture cover (due to differences in the likelihood of anthropogenic ignition depending on the pasture/forest ratio in the landscape), we included a quadratic term of pasture cover in both components. Finally, we included a random effect of year in each component to control for interannual variation in forest fire density resulting from factors other than land-cover and climatic conditions (e.g., changes in political climate; Caetano, 2021).
- *Patch-Scale*: we modelled $\mu_{S=P,i,j}$, $p_{S=P,i,j}$ and $q_{S=P,i,j}$ as the outcomes of the linear effects of each of our patch-scale predictors. We also included a random effect of year in each component and included surrounding forest cover as a quadratic term to investigate possible non-linear effects of surrounding forest loss. Finally, we included an interaction between patch shape and area in each component, as the perimeter-area ratio of a patch (and thus the proportion of a patch that is exposed to edge-related desiccating effects) is a function of both its size and complexity, and an equivalent increases in our patch complexity metric had a greater impact on the perimeter-area ratio of smaller forest patches (Figure A.8).

At both scales, we assumed consistent precision among estimates of forest fire density, specifying $\phi_S$ as the outcome of an intercept term only, with a log link. Both models thus included the terms:

$$\text{Logit}\left(\mu_{S,i,j}\right) = \alpha_{S,\mu} + \upsilon_{S,\mu,j} + X_{S,i,j}\beta_{S,\mu}$$

$$\text{Logit}\left(p_{S,i,j}\right) = \alpha_{S,p} + \upsilon_{S,p,j} + X_{S,i,j}\beta_{S,p}$$

$$\text{Log}(\phi_S) = \alpha_{S,\phi}$$

While the patch-scale model also included:

$$\text{Logit}\left(q_{S=P,i,j}\right) = \alpha_{S=P,q} + \upsilon_{S=P,q,j} + X_{S=P,i,j}\beta_{S=P,q}$$

where $\alpha_{S,c}$ denotes intercept terms in component $c$ ($c = \mu, p, \phi, q$); $\beta_{S,c}$ represents fixed effects slopes; $\upsilon_{S,c}$ denotes random effects of year; and $X_S$ is the fixed-effects design matrix, containing all scale-specific covariate values (Table 1). We set weakly informative priors for both models. Specifically, we used logistic (0,1) priors for the intercepts of $\mu_S$, $p_S$ and $q_{S=P}$ components; normal (0,5) priors for the intercepts of $\phi_S$ components; normal (0,2.5) priors for fixed effect slopes ($\beta$); and exponential (1) priors for random effect variances.

#### 2.3.4. Model fitting and inference

We fitted our models using '*brms*' in R (Bürkner, 2017; R Core Team, 2023), using four chains of 2500 Markov Chain Monte Carlo iterations, each with a 2500-iteration warmup. Convergence was assessed based on R-hat values being <1.05 (Vehtari et al., 2021). We assessed model fit using a series of posterior predictive and residual checks and inferred 'significance' of coefficients whenever their 95% Bayesian posterior Credible Intervals excluded zero. To visualise results, we plotted the average marginal effects of each predictor variable on $\mu_{S,i,j}$, $p_{S,i,j}$ and $q_{S=P,i,j}$.

#### 2.3.5. Temporal variation in forest fire density

To investigate whether patterns and drivers of forest fires vary across different stages of agricultural frontier development (Ribeiro et al., 2024), we repeated our patch- and landscape models using data from: 1) before 2005 only, a period when forest cover within the PdA was rapidly declining, indicating substantial agricultural expansion; and 2) from 2005 onwards, when large-scale deforestation within the region had decelerated considerably, and the frontier could thus be considered 'established' (Figure A.4). Results of these models are reported in Appendix D and their implications are outlined in the Discussion.

### 2.4. Regional areal analysis

To evaluate the effects of forest fragmentation on regional forest fire cover, we also quantified the relative contribution of different patch size classes to the overall forest area burnt throughout the entire PdA. For this we used the full patch-level dataset (see Section 2.3.2); that is, we excluded forest areas >25,000 ha and <0.5 ha but did not perform any subsampling. First, we grouped all patches from each year into five bins based on area: 0.5–10; 10–100; 100–1000; 1000–10,000; and 10,000–25,000 ha. We then calculated the total area of forest, and the total area of forest burnt, within each bin. Next, for each year, we calculated the percentage contribution of the forest patches within each bin to 1) the annual total area of all forest patches across the region; and 2) the annual total area of forest burnt across all forest patches in the region. Then, for each bin in each year, we subtracted the percentage contribution to the total area of forest patches from the percentage contribution to the total area of forest patches burnt. This provided a relative measure of the contribution of patches in each bin to regional forest fire cover ('relative contribution'), where positive values would indicate that patches in a given bin made a disproportionately large contribution to regional forest fire cover (i.e., accounted for a greater percentage of regional forest *fire* cover than they did regional forest cover), and vice versa for negative values. Finally, to provide insight into temporal trends in the relative contribution of different sized patches to regional forest fire cover, we performed separate linear regressions of year (predictor) against our relative contribution measures (response) from each patch size bin.

## 3. Results

### 3.1. Fire density analyses

#### 3.1.1. Landscape-scale model

We obtained data for 37,222 landscape-years (1006 100-km$^2$ quadrats in each of 37 years) within the PdA. The density of forest fires within each landscape-year ranged from 0 to 0.942 (mean ± S.D = 0.010 ± 0.047) and forest fires occurred within 52.58% of landscape-years (Figure C.1). Posterior predictive and residual checks suggested that our landscape-scale model fit the data well (Figure C.2).

Both total forest edge length (mean $\beta_p$ = 0.827; $\beta_\mu$ = 0.220; Fig. 2A–C.3) and urban land cover (mean $\beta_p$ = 0.073; $\beta_\mu$ = 0.031; Figure C.3, C.4) were significantly positively associated with the probability of forest fire occurrence and the density of forest fires where they occurred. Pasture cover exhibited a significant inverted-U shaped relationship with the probability of forest fire occurrence (mean Pasture $\beta_\mu$ = 0.239; Pasture$^2$ $\beta_\mu$ = −0.128; Figure C.3), with fire occurrence probability decreasing rapidly either side of ~45% pasture cover (Fig. 2B). In the subset of landscape-years where forest fires did occur, pasture cover exhibited a significant curvilinear relationship with forest fire density (mean Pasture $\beta_\mu$ = 0.239; Pasture$^2$ $\beta_\mu$ = −0.128; Figure C.3), increasing with pasture cover to ~70% but decreasing slightly thereafter (Fig. 2B).

Both cropland cover (mean $\beta_\mu = -0.051$; $\beta_p = -0.304$; Fig. 2C, C.3) and water cover (mean $\beta_\mu = -0.041$; $\beta_p = -0.061$; Figures C.3, C.5) were significantly negatively associated with the probability of forest fire occurrence and the density of forest fires where they occurred. Interestingly, natural non-forest cover was significantly positively associated with the density of forest fires within landscape-years where fires occurred (mean $\beta_\mu = 0.071$) but negatively associated with the probability of forest fire occurrence (mean $\beta_p = -0.194$; Figures C.3, C.6).

#### 3.1.2. Patch-scale model

After subsampling, the mean (±SD) fire density across all patch-years within the PdA was 0.080 (±0.250) (full dataset = 0.083 ± 0.260). Forest fires were absent from 81.91% of all patch-years (full dataset, 86.80%). Of those patch-years where forest fires did occur, 28.35% were burnt entirely (46.55%). Posterior predictive and residual checks suggested that our patch-scale model fit the data reasonably well but tended to underestimate forest fire density >0.15 (Figure C.10). This was likely due to the concentration of observations below this threshold - forest fire density was <0.15 within 44.11% (24.39%) of all patch-years in which fires occurred (Figure C.1) - but also suggests that additional factors not included in our models may play important roles in dictating patch-level fire cover (see Discussion, section 4.1).

The area and shape of forest patches were significantly associated with forest fire density in three ways. Firstly, forest fire occurrence probability increased with both patch area (mean $\beta_p = 1.100$; Fig. 3a, C.11) and patch shape complexity (mean $\beta_p = 0.100$; Fig. 3b and C.11), with an interaction such that the association between patch shape and fire occurrence became increasingly negative with increasing patch area (mean Area:Shape $\beta_p = -0.579$). This interaction meant that among patches larger than ~600 ha, the probability of fire occurrence was negatively associated with patch shape complexity (Figure C.14). Secondly, within patch-years where fires did occur, fire density decreased with increasing patch area (mean $\beta_\mu = -0.052$) and increased with increasing patch shape complexity (mean $\beta_\mu = 0.073$) (Figures, 3A, 3B, C.11), with no significant interaction effect (Figure C.11). Finally, patch area (mean $\beta_q = -17.072$), shape (mean $\beta_q = -0.145$), and the interaction between area and shape (mean $\beta_q = -15.097$) exhibited large negative associations with the probability of an entire forest patch being burnt in a given year, suggesting that small, compact patches were considerably more likely to be entirely burnt than patches of any other

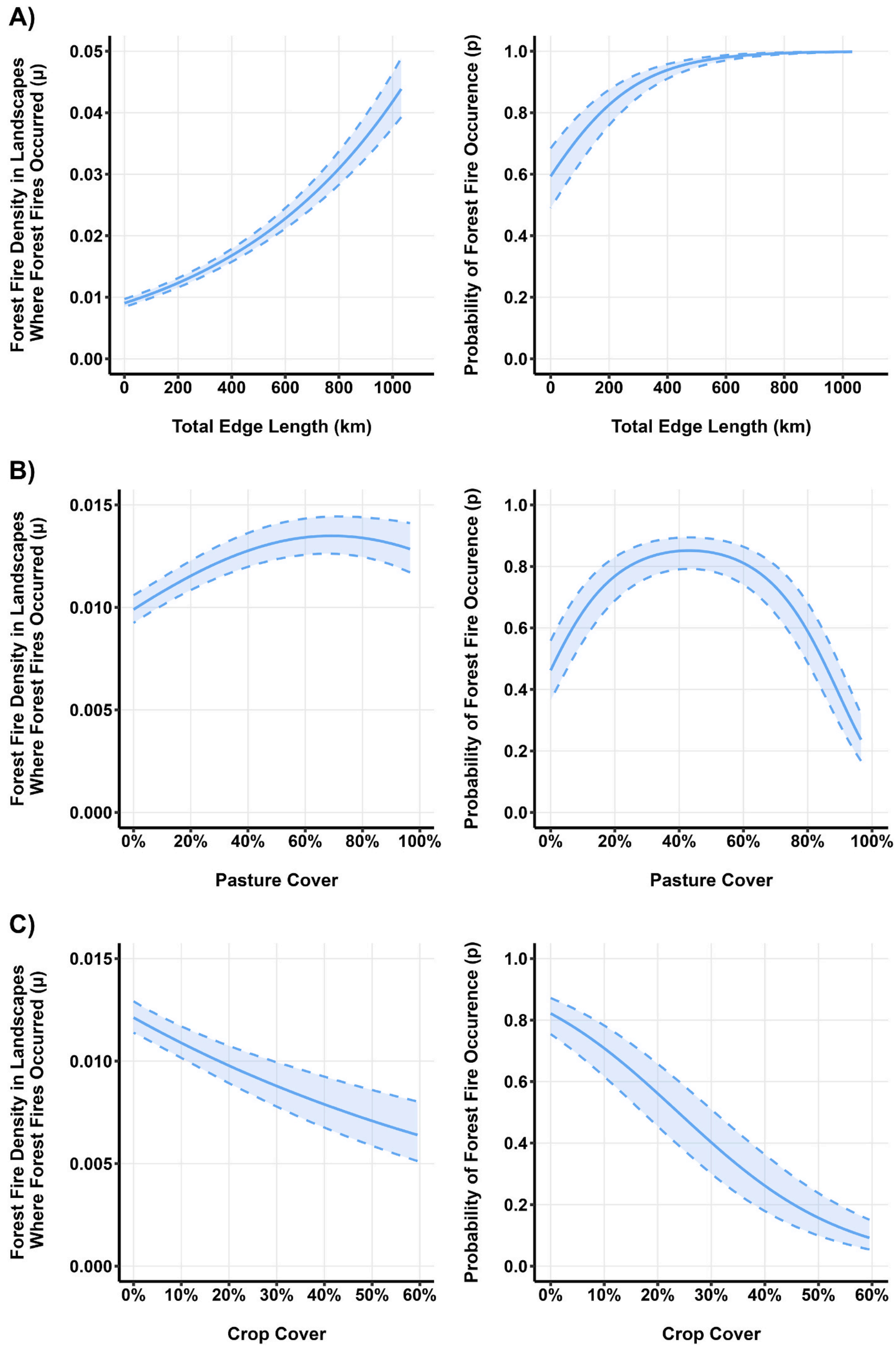


**Fig. 2.** Average marginal effects of A) total forest edge length (km); B) pasture cover (%); and C) cropland cover (%) on landscape-scale forest fire density. Left column shows the effect of each variable on the density of forest fires within landscapes where forest fires did occur. Right column shows the effect of each variable on the probability of forest fire occurrence (i.e., the probability that at least one forest raster cell within a landscape was burnt).

size or shape configuration (Figures C.11-C.14). Taken together, these relationships indicate that although fires were most likely to occur within large, relatively compact forest patches, where fires did occur, fire density tended to increase within decreasing patch size and increasing patch shape complexity. Furthermore, not only were fires most likely to occur in patches that were both small *and* complex in shape, but these patches were also more likely to be burnt entirely. The observed trends are likely linked to how the perimeter-area ratio of forest patches, and thus the relative extent of edge-related desiccating effects, varies with patch size and shape. Specifically, small forest patches typically have a greater perimeter-area ratio than large forest patches, regardless of their shape complexity, while patches that are both small *and* complex have the greatest perimeter ratio of all forest remnants, and the relative extent of desiccating edge effects will thus be highest among these patches (Figure A.8).

The proportional cover of forest surrounding forest patches was also significantly associated with forest fire density in three ways. Firstly, the probability of forest fires occurring within a patch-year increased with surrounding forest cover to ~45% but decreased thereafter (mean $\beta_p$: Forest = 0.924; Forest$^2$ = −0.915). Secondly, among patch-years where fires did occur, fire density increased exponentially with surrounding forest cover (mean $\beta_\mu$: Forest = 0.027; Forest$^2$ = 0.489) and, thirdly, the

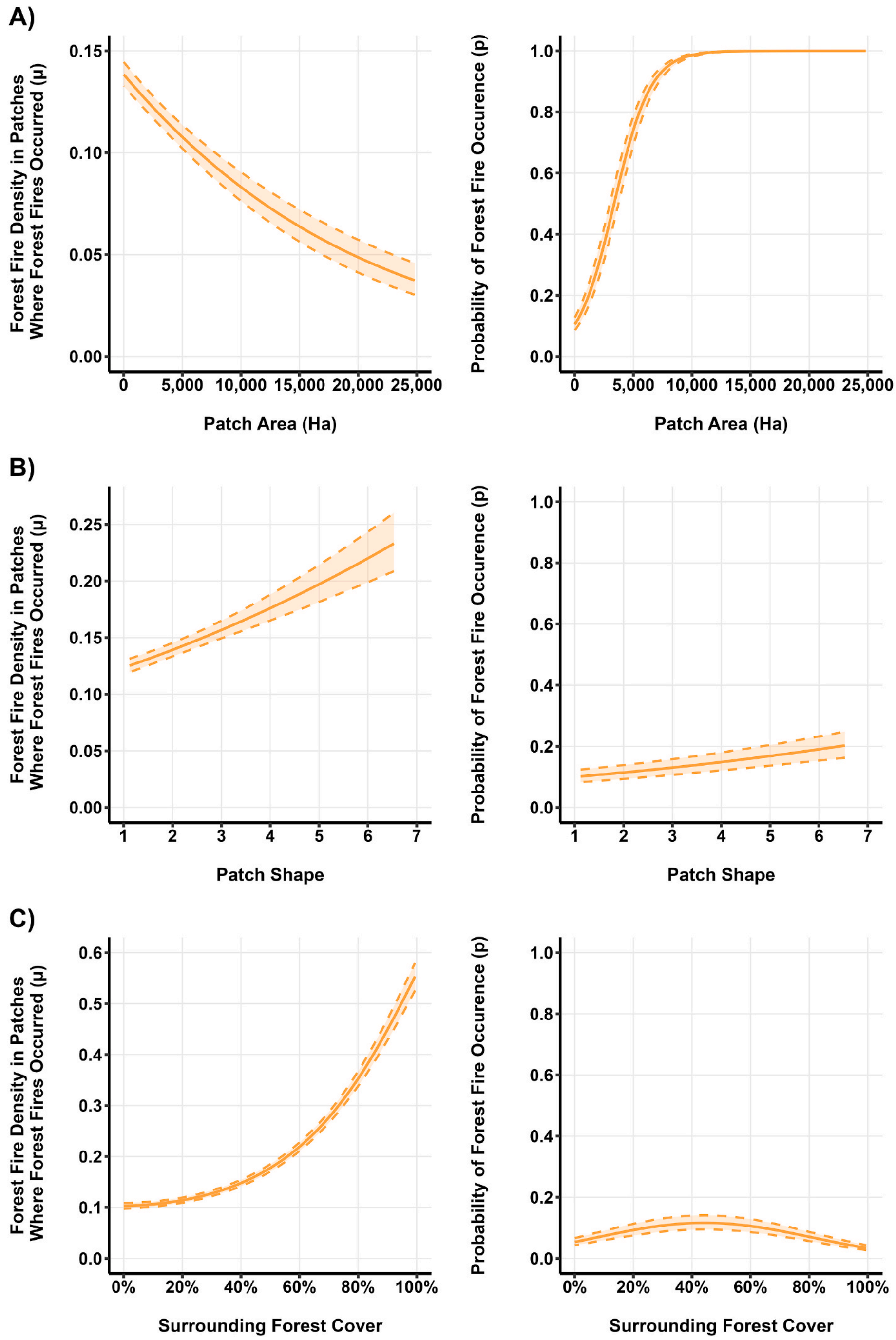


**Fig. 3.** Average marginal effects of A) forest patch area (ha); B) forest patch shape; and C) surrounding forest cover (1-km radius) on patch-scale forest fire density. Left column shows the effect of each variable on the density of forest fires within patches where forest fires did occur. Right column shows the effect of each variable on the probability of forest fire occurrence (i.e., the probability that at least one forest raster cell within a patch was burnt).

probability of an entire forest patch burning increased roughly linearly with surrounding forest cover (mean $\beta_q$: Forest $= 2.385$; Forest$^2$ $= -0.954$) (Fig. 3C–C.11). Overall, these associations suggest that among patch-years where fires did occur, the proportional area burnt tended to be greatest within patches surrounded predominantly by forest, but that the probability of fire occurrence was considerably higher in patches where roughly half of the surrounding forest cover had already been lost in the corresponding year.

#### 3.1.3. Surrounding forest fire and climate

As we only wished to control for effects of the mean proportion of forest burnt within neighbouring landscapes (landscape-scale model), the prevalence of fires surrounding individual forest patches (patch-scale model), and climatic conditions (both models) on forest fire density, we do not report on these effects here. However, the corresponding coefficients and average marginal effects are presented in Appendix C.

### 3.2. Regional areal analysis

Both 0.5–10 Ha (mean relative contribution ± SD = 22.80 ± 10.30%) and 10–100 Ha (14.73 ± 5.31%) forest patches made disproportionately large contributions to annual regional forest fire cover in all 37 years. The relative contribution of 0.5–10 Ha patches to regional forest fire cover decreased significantly through time ($\beta = -0.59$; $p <$

0.001), but no significant temporal trend was apparent in the relative contribution of 10–100 Ha patches (Fig. 4; Figure C.19; Tables C.3, C.4). The disproportionate contribution of small patches to regional forest fire cover becomes even clearer when considering all patches <100 Ha, which accounted for >50% of annual regional forest fire cover in each of 27 years (mean across all years ± SD = 63.10 ± 15.55%), despite never accounting for more than 33% of the total forest patch area in any year (25.56 ± 3.16%; Fig. 4; Table C.3).

In 28 of the 37 years considered, 100–1000 Ha patches made disproportionately small contributions to regional forest fire cover (mean relative contribution ± SD = −4.57 ± 6.87%), although their relative contribution increased significantly over time (β = 0.39; p < 0.001). Similarly, patches of 1000–10,000 Ha made disproportionately small contributions to regional forest fire cover in all but one year (mean relative contribution ± SD = −22.10 ± ), but these relative contributions increased significantly through time (β = 0.29; p = 0.03). The largest forest patches in the region (10,000–25,000 Ha) made disproportionately small contributions to regional annual forest fire cover in all but two years (mean relative contribution ± SD = −10.87 ± 6.94%), although there were no patches in this size class in 1989, preventing assessment in this year. There was no significant temporal trend in the relative contribution of 10,000–25,000 Ha patches to regional forest fire cover (Fig. 4; Figure C.19; Tables C.3, C.4).

## 4. Discussion

While it has long been suggested that small tropical forest patches are particularly vulnerable to fire (Cochrane and Laurance, 2002; Guedes et al., 2020), our results represent some of the first empirical evidence for a strong negative association between forest patch size and fire density (Maillard et al., 2020). Furthermore, we show that small forest patches (0.5–100 ha) consistently made disproportionately large contributions to the annual area of forest patches that was burnt throughout the Porta da Amazônia region of southeastern Amazonia. Indeed, in almost three-quarters of the 37 years considered, patches of <100 Ha actually accounted for the majority of annual regional forest fire cover.

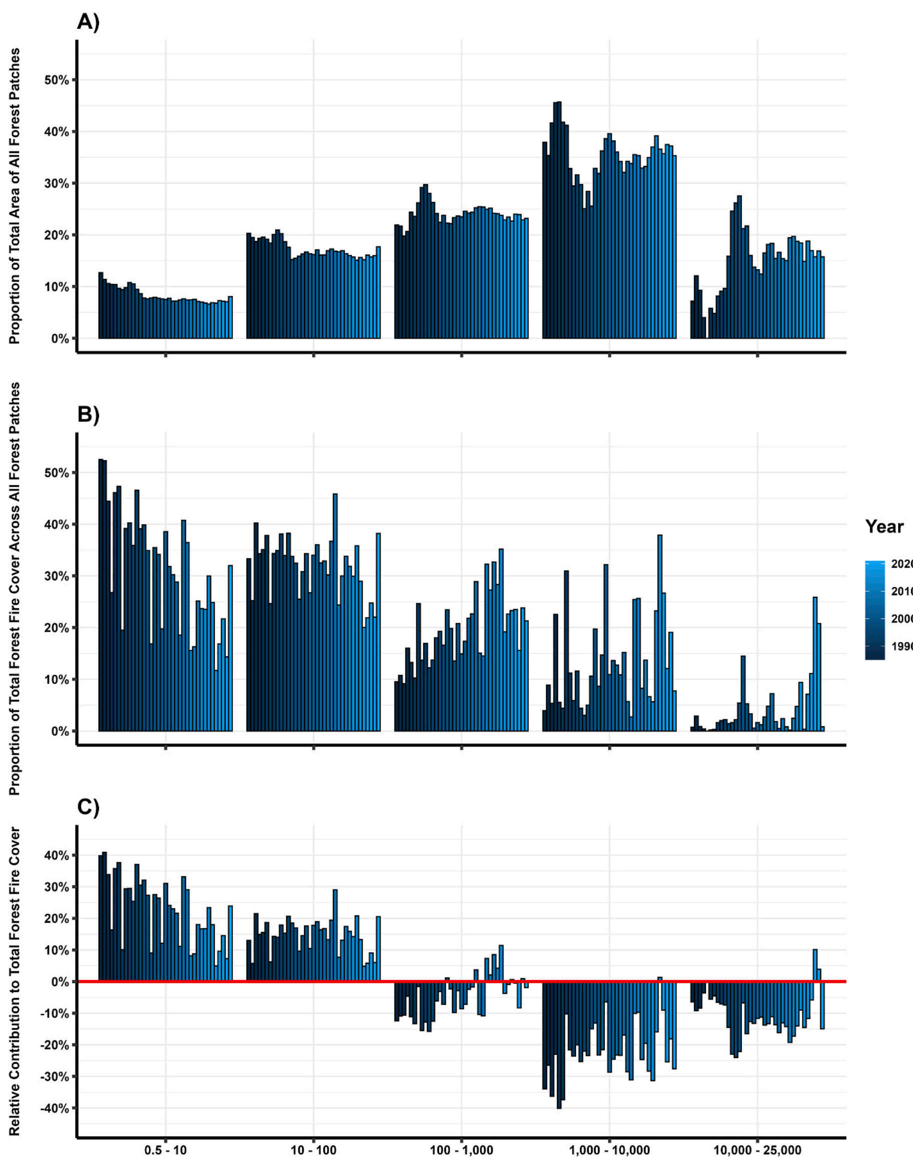


**Fig. 4.** Annual values (for each year between 1985 and 2021) of: A) the proportion of the area of all forest patches within the Portal da Amazonia region accounted for by forest patches of different size; B) the proportion of the area of forest burnt across all forest patches in the PdA region accounted for by patches of variable sizes; and C) the relative contribution of forest patches of different size to (that is, the statistics presented in section B of this figure, minus the statistics presented in A). The red line in C denotes a relative contribution of 0 (i.e., equal contribution to the total area of forest patches and the total area of forest patches burnt in a given year).

At the landscape scale, we found that forest fire density was greater within more fragmented landscapes, where the total length of forest edges was higher, in line with previous research in central (Silva-Junior et al., 2018) and northwestern Amazonia (Armenteras et al., 2013). Importantly, however, we show that land-use composition also exerts important effects on forest fire prevalence. Landscape-scale forest fire density was positively associated with pasture and urban land-cover and negatively associated with cropland cover. Patch-scale fire density also tended to increase with surrounding forest cover. Ongoing anthropogenic activities within non-forest areas thus likely play a major role in dictating patterns of forest fire prevalence (Pivello et al., 2011; 2021). Particularly stringent fire prevention measures may thus benefit forests near urban areas (Price et al., 2014; Singh et al., 2022), and transitioning existing pasture lands towards crop production, such as the advancing frontline of soybean monoculture, would further limit the occurrence of forest fires (Cano-Crespo et al., 2015). Future studies seeking to understand the effects of fragmentation on fire prevalence should therefore also consider the land-cover composition of focal regions. Notwithstanding, the effects of forest fragmentation on fire prevalence observed here represent cause for major concern, given that the average forest patch size is decreasing, and forest edge density is increasing throughout Amazonia and many other tropical regions (Taubert et al., 2018; Montibeller et al., 2020; Fischer et al., 2021).

### 4.1. *Effects of forest fragmentation on forest fire prevalence*

The effects of increased edge length and reduced patch size on fire density are fundamentally linked to elevated solar radiation in a more desiccated and hotter forest understorey (Broadbent et al., 2008; Meza-Elizalde and Armenteras-Pascual, 2021). These microclimatic alterations reduce vegetation moisture content, increase tree mortality, and promote shrub growth, increasing the fuel load and fire susceptibility of forest edges (Laurance et al., 2011; Berenguer et al., 2014; Benchimol and Peres, 2015). As the perimeter-area ratio of a patch tends to increase with decreasing patch size, edge effects influence a greater proportion of forest within smaller patches (Malcolm, 1994; Figure A.8). Accordingly, reducing the size of individual forest patches, and increasing the total length of forest edges, likely enable fires to more readily propagate throughout forest areas (Cochrane and Laurance, 2002; Armenteras et al., 2013; Numata et al., 2017).

Given the physical nature of these effects, it is unlikely that the observed associations between forest fire prevalence and both forest patch size and forest edge length are exclusive to our study region, nor Amazonia. Indeed, Maillard et al. (2020) found that fires most frequently occurred in small (<20 Ha) forest patches across a variety of forest types within Santa Cruz, Bolivia, including Amazonian rainforest. Similarly, Guedes et al. (2020) showed that burn probabilities were generally higher within smaller forest patches in the Atlantic Forest of Brazil. Forest fragmentation has also been cited as a likely contributor to elevated forest fire prevalence seen in rainforests in South-East Asia (Nikonovas et al., 2020) and Africa (Zhao, 2021; Wimberly et al., 2024), as well as in temperate forests in Spain (Roman-Cuesta et al., 2009) and the USA (Brudvig et al., 2012). Interestingly, however, fragmentation has been linked to reduced fire spread in the Brazilian Cerrado (Rosan et al., 2022) and several temperate (Portugal: Azevedo et al., 2013; USA: Breininger et al., 2006) and sub-artic (Siberia: Wirth et al., 1999) regions. However, this tends to be because native vegetation in these regions is naturally prone to burning, and is typically replaced by a relatively inflammable matrix, which acts as a fire break (Driscoll et al., 2021; Rosan et al., 2022). Conversely, in tropical rainforest regions, recurrent fires are rare at evolutionary timescales (Goulart et al., 2017; Feldpausch et al., 2022) and native vegetation is typically converted into *more* flammable land-cover types (e.g., pasture; Cano-Crespo et al., 2015; Driscoll et al., 2021). It thus seems likely that efforts to minimise fragmentation, in terms of both the creation of forest edges and small forest patches, will have major benefits for reducing forest fire prevalence in the tropics. Furthermore, given that fragmentation has been shown to reduce fire prevalence where natural vegetation is converted into less flammable land cover types, it seems likely that fire breaks placed within matrix areas may be beneficial for forest fire suppression within fragmented tropical forest landscapes. As data on fire suppression measures within the PdA were unavailable we were unable to assess the efficacy of fire breaks for fire suppression within fragmented landscapes, but this should constitute a topic for future research.

There are other possible reasons why forest fire density is higher within more fragmented landscapes and smaller forest patches. First, forest fire prevalence is often correlated with deforestation rates in Amazonia, in part because fire is typically used as a tool to clear forests (Mataveli et al., 2022; Ribeiro et al., 2024). Thus, active deforestation frontiers – which are likely to be more fragmentated than regions with little to no pre-existing forest loss – are expected to succumb to more frequent fires due to widely available anthropogenic ignition. However, we found that the positive association between forest fire density and total forest edge length, and the negative association between fire density and patch size, held when only considering the period after 2005 (see Appendix D), when deforestation rates within our study region had decelerated considerably (Figures A.1, A.4, A.5). Furthermore, in another study in southern Amazonia (including Mato Grosso), Cano-Crespo et al. (2015) found that trends in burned area were disconnected from deforestation rates, and that fires that escaped from agricultural lands into forests were major contributors to overall forest fire cover, especially in areas dominated by pasture land, as is true of much of the PdA (Figure A.1-A.3).

Second, a disproportionate number of forest patches within our dataset had burnt entirely, and these patches were almost always smaller than 100 ha (Figure C.12), which could suggest that landowners preferentially clear small forest patches, as has previously been found in the Amazon (Stickler et al., 2013; Tulloch et al., 2015). This may explain why our patch-scale model tended to underestimate fire densities >0.15 (i.e., certain small forest patches were targeted for clearance) and could also contribute to the observed decline in the relative contribution of the smallest patches (0.5–10 Ha) to regional forest fire through time. Specifically, as the Brazilian Forest Code required landowners within Amazonia to preserve 80% forest cover within their properties (Soares-Filho et al., 2014), and as regional forest cover region exhibited a near perfect, negative correlation with year ($r = 0.96$, Figures A.1-A.3), it may be that small forest patches increasingly represented the last remaining forest in certain areas/landholdings. Small patches may thus have become increasingly important for Forest Code adherence over time, making them less likely to be cleared. However, given that the negative association between patch size and fire density was also apparent when only considering the period after 2005, when deforestation had decelerated (see Appendix D), and patches of <100 Ha consistently made disproportionately large contributions to regional forest fire cover throughout the time series, we would argue that our results still indicate toward the heightened influence of edge effects within small patches contributing to increased fires susceptibility in these areas.

Further evidence for an association between increases in the relative extent of edge effects and an increased susceptibility of forest to fire is further supported by the observed associations between patch complexity and forest fire density. Among small forest patches, fire density increased with patch complexity, likely because, among patches of comparable size, the proportion of a patch exposed to edge related desiccating effects increases with shape complexity (Malcolm, 1994; Cochrane and Laurance, 2002). Interestingly, the most complex patches in our dataset (shape index >2) tended to be forest corridors, typically along perennial streams. As the Brazilian Forest Code requires landowners preserve forest buffers around all rivers and perennial streams (Soares-Filho et al., 2014), many of the most complex patches in the region were subject to an additional level of legal protection and should thus have been less likely to be intentionally burnt or otherwise degraded. Considering this, our finding of a positive association between patch shape complexity and fire density among small forest patches could thus serve as particularly strong evidence for the role of edge effects in dictating patch-scale fire density and could indicate that small, complex patches are particularly susceptible to escaping fires from neighbouring agricultural lands (Cano-Crespo et al., 2015). However, forest fire density in patches larger than ~600 ha actually decreased with increasing patch complexity. This is likely because variation in patch shape has much smaller impact on the perimeter-area ratio of large patches, and thus proportion of a patch exposed to edge effects (Malcolm, 1994). Alternatively, as large patches with complex shapes tended to represent riparian buffers along major waterways (Fig. 1), fires may have been inhibited by high soil moisture levels in these areas, while the Forest Code may also have been better enforced in these regions given the importance of large waterways to regional hydrological dynamics.

Nonetheless, data on the reason for, and intended extent of, fires was not available for our study region, and the extent to which human agency drives some of the observed relationships between patch size and fire prevalence thus remains unclear. Considering this, future research on Amazonian forest fires should seek to incorporate factors that may influence the individual actors' decision to set fires, and how this contributes to variability in fire prevalence among landscapes subject to different levels of fragmentation. This could, for instance, include agricultural property size. Smallholders have lesser economic capabilities compared to largeholders and agribusinesses and are perhaps more likely to use fire for land management, as it is relatively inexpensive. That said, largeholders typically have the capacity to burn considerably more forest than individual smallholders (Carmenta et al., 2019; Pivello et al., 2021).

It is also noteworthy that although fire *density* tended to be higher in small patches, forest fires were more likely to *occur* within large patches. We propose that this is due to the increased interface between forest and other land uses, with greater potential for fire spread near agricultural land. This would also explain why forest fires were more likely to occur in landscapes with more forest edges. The occurrence of fires within large forest patches in the PdA and elsewhere in the Amazon is therefore a concern, particularly from the perspective of biodiversity, given that larger forest patches host a greater number of species across several taxa (Benchimol and Peres, 2013; Palmeirim et al., 2021; Noble et al., 2023). Nonetheless, the potential impacts of fire spread on biodiversity in small forest patches cannot not be overlooked either, given that these features are often vital to the maintenance of landscape connectivity (Tulloch et al., 2015) and regional beta-diversity (Dambros et al., 2024).

### 4.2. Effects of land-cover composition on forest fire prevalence

The extent of forest loss had important effects on forest fire occurrence and density at both the patch and landscape scales. Among those landscapes where forest fires did occur, forest fire density increased with decreasing forest cover. This is in line with previous studies (e.g. Armenteras et al., 2013; Silva-Junior et al., 2018), but we note that we modelled pasture cover rather than forest cover due to the near-perfect negative correlation between the two. This negative relationship between forest cover and forest fire density is consistent with the effective fire inhibiting effect of large forest protected areas and indigenous territories that remain largely undisturbed (Nepstad et al., 2006; Silvestrini et al., 2011), and the considerably higher level of fire resistance of continuous forest areas compared to areas with pre-existing forest loss (Nikonovas et al., 2020). This explanation is supported by the fact that we also observed a similar association between forest cover and forest fire density when only considering the period after 2005, when deforestation rates in the PdA had declined massively (see Appendix D), suggesting increased fire prevalence in regions with pre-existing forest loss is likely not solely due to ongoing deforestation in these areas. Interestingly, at the patch scale we found the opposite pattern, with elevated fire density in patches surrounded by highly forested landscapes. This could be because these patches are more likely to be deforested, given that natural ignition sources are unlikely (Pivello, 2011). This pattern of burning likely leads to erosion of landscape connectivity, with further negative impacts on biodiversity.

Interestingly, at both the patch and landscape scale, fires were most likely to occur where forest cover had been roughly halved. This suggests that biomass burning is still the primary method of forest clearing in regions undergoing agricultural frontier expansion. Thus, there may be more benefit, in terms of carbon retention, in allocating fire prevention efforts to areas where ~50% of forest has already been lost, rather than continuous forest areas that likely retain stronger natural immunity to wildfires (Cochrane and Laurance, 2002; Armenteras et al., 2013).

Croplands within the PdA region have rapidly expanded, although pastures continue to dominate deforested areas. At the landscape scale, forest fire density decreased with increasing cropland cover. Furthermore, this association held true both before 2005, when large-scale forest clearance was widespread, and thereafter, when deforestation rates declined (Appendix D). Importantly, fire is less often used as a management tool within croplands than in pastures and fire is less likely to escape from croplands, likely because farmers are incentivized to protect crops by preventing fires, whereas fire is still instrumental in suppressing shrub succession in low-yield pastures (Cano-Crespo et al., 2015). It should, however, be noted that croplands in this region are increasingly dominated by soy monoculture. The rapid expansion of soy cultivation across southern Amazonia, which primarily replaces pre-existing low-yield cattle pastures (Song et al., 2021), follows an economic logic in terms of land-use revenue but subsequently suppresses the use of fire in land management, which is consistent with the clear negative effect of cropland on fire occurrence detected in this study.

The impact of natural non-forest vegetation and urban cover is a further consideration. The PdA is in a transitional zone where any edaphic enclave of natural non-forest land-cover is dominated by savannah-like scrublands (Table A.1), resembling those of the neighbouring Cerrado biome. Fire has played an integral role in the natural history of the Cerrado, where most native herbaceous species evolved in a fire-climax ecosystem and can rapidly resprout post-burn (Gomes et al., 2018; Durigan, 2020). Considering this, our finding that forest fire density increased with increasing natural non-forest land-cover is perhaps unsurprising. Interestingly, the incidence of forest fires decreased with increasing surrounding natural non-forest vegetation, perhaps because human density and therefore anthropogenic ignition sources are rarer in these areas (Pivello, 2011). Indeed, increased urban land cover, likely associated with more heavily settled areas and anthropogenic ignition sources, was positively associated with both increased fire density and incidence. This is in line with previous research showing that roads and population density are well correlated with the prevalence of fires within Amazonia (dos Reis et al., 2021; Singh et al., 2022) and other tropical regions (Price et al., 2014; Achu et al., 2021).

### 4.3. Conclusions

Our findings suggest that forest fragmentation is driving an increase in the prevalence of forest fires in northern Mato Grosso, likely due to a more desiccated and hotter understorey resulting from forest edge creation and reductions in forest patch size (Cochrane, 2002; Armenteras et al., 2013; Numata et al., 2017). Although this study focused on the 113,000 $km^2$ PdA region, other analyses within Amazonia (Armenteras et al., 2013; Silva-Junior et al., 2018) and elsewhere (Nikonovas et al., 2020; Zhao, 2021; Wimberly et al., 2024) have found similar associations between forest fragmentation and increased fire prevalence, lending greater support to the generalization power of our findings. This is cause for global concern, given that carbon emissions resulting from forest degradation, including fragmentation and fires, are already three times higher than those from deforestation within Amazonia (Qin et al., 2021); remaining tracts of Amazonian forest are being increasingly fragmented (Taubert et al., 2018; Montibeller et al., 2020); and our findings could indicate toward a positive feedback loop between fragmentation and forest fire prevalence. Furthermore, given that the frequency and severity of drought conditions are already increasing throughout Amazonia due to climate change (Barkhordarian et al., 2019; Boulton et al., 2022), the observed associations between fragmentation and fire prevalence may worsen in coming years. Concerted efforts must thus be made to minimise the creation of new forest edges and preclude the spread of agricultural fires across pre-existing forest edges. Targeted efforts to suppress fires in small forest patches, perhaps through the preferential placement of fire breaks, could considerably reduce overall forest fire spread. That said, further research is required to determine which socioeconomic factors may contribute to the high prevalence of fires among small patches, such as the preferential targeting of these sites for land clearance (Stickler et al., 2013; Tulloch et al., 2015) or the possibility that smallholders may preferentially use fire-based land management due to their relatively low cost (Carmenta et al., 2019; Pivello et al., 2021).

Importantly, forest fire occurrence was relatively limited within large, continuous tracts of forest (i.e., landscapes with near 100% forest cover). While this may be in part due to a low levels of human activity in these areas, undisturbed continuous forests also retain high levels of natural fire resistance, as the preservation of an intact forest canopy generally translates into a moister, cooler understorey within continuous forests, providing a greater capacity to buffer against atmospheric climate change (Cochrane, 2002; Cochrane and Laurance, 2002; Ewers and Banks-Leite, 2013; Nunes et al., 2022). Preserving continuous tracts of forest is thus of paramount importance in the face of increasing drought frequency within Amazonia (Leite-Filho et al., 2021; Nunes et al., 2022). In areas with pre-existing forest loss, however, we would suggest that fire prevention efforts should be best targeted towards landscapes where remaining forest cover loss is already ~50%, as this is where fire incidence is highest. At the same time, identifying more specific characteristics of areas exposed to high fire risk is critically important to inform fire suppression efforts. Finally, our results underscore the importance of considering the wider landscape context when examining forest fragmentation impacts on fire, as the relative cover of different non-forest land-use classes has an important mediating effect on forest fire prevalence.

## CRediT authorship contribution statement

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

## 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.

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## Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2025.124312.

## Data availability

Code required to extract and analyse the publicly available data is provided in a dedicated repository (https://zenodo.org/records/13627983). Pre-processed data can be made available upon request.

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