

Economic losses to sustainable timber production by fire in the Brazilian Amazon

Aline S. de Oliveira^{1,2} | Raoni G. Rajão³  | Britaldo S. Soares Filho² |
 Ubirajara Oliveira² | Lucas R. S. Santos² | Alexandre C. Assunção² |
 Richard van der Hoff^{4,5} | Hermann O. Rodrigues² | Sônia M. C. Ribeiro² |
 Frank Merry⁶ | Letícia S. de Lima⁷

¹Departamento de Engenharia Agrícola, Universidade Federal de Viçosa, Viçosa, MG, Brazil

²Centro de Sensoriamento Remoto, Universidade Federal de Minas Gerais, Belo Horizonte, MG, Brazil

³Laboratório de Gestão de Serviços Ambientais, Universidade Federal de Minas Gerais, Belo Horizonte, MG, Brazil

⁴Radboud University, Nijmegen, The Netherlands

⁵Universidade Federal de Minas Gerais, Belo Horizonte, MG, Brazil

⁶Conservation Strategy Fund, Washington, DC, USA

⁷Integrative Research Institute on Transformations of Human-Environment Systems, IRITHESys, Humboldt-Universität zu Berlin, Berlin, Germany

Correspondence

Raoni G. Rajão
 Email: rajao@ufmg.br

Funding information

Coordenação de Aperfeiçoamento de Pessoal de Nível Superior; Fundação de Amparo à Pesquisa do Estado de Minas Gerais; Conselho Nacional de Desenvolvimento Científico e Tecnológico; World Bank Group; Alexander von Humboldt-Stiftung

Although still the largest expanse of tropical rainforests in the world, the Amazon is suffering a declining capacity to deliver ecosystem services, to which the widespread use of fire is one of the main contributing factors. Even if fires directly affect the timber sector, most current logging practices often tend to increase rather than mitigate the problem. We argue that in order to involve the timber sector in fire mitigation policies in the Amazon it is crucial to assess the economic impact of fire on the sector. This paper describes EcoFire (Economic Cost of Fire), a spatially explicit model for valuing the economic losses to sustainable timber harvest operations in the Brazilian Amazon as a result of fire. To conduct this analysis, we have integrated a set of models that simulate the synergy between logging and fire spread and intensity. Our results show that fire affects roughly 2% of the timber production areas that would be harvested between 2012 and 2041. In burnt areas, fire causes losses on average of US\$39 ± 2 ha/year (equivalent annual annuity), which represents a loss of 0.8% of expected rents. Yet losses can reach up to US\$183 ± 30 ha/year in areas hit by recurrent fires that are near milling centres. The results indicate that some of the municipalities that are likely to accumulate most economic losses due to fire do not yet have local-level fire mitigation programmes. We therefore conclude that spatially explicit valuations of the economic impact of fire can pinpoint priorities to better target fire action plans as well as to engage local actors in integrated fire management practices.

KEYWORDS

collective fire management, economic losses by fire, fire mitigation policies, spatially explicit modelling, timber species distribution

1 | INTRODUCTION

Although still the largest expanse of tropical rainforests in the world, the Amazon is suffering a declining capacity to deliver ecosystem services such as climate regulation (Costanza et al., 1997; Baccini et al., 2017; Phillips & Brienen, 2017). Moreover, some scholars have warned that “negative synergies between deforestation, climate change and widespread use of fire indicate a tipping point for the Amazon system to flip to non-forest ecosystems in eastern, southern and central Amazonia” (Morton et al., 2013; Lovejoy & Nobre, 2018, p. 1). Forest fires are particularly detrimental at the deforestation frontier where forests are made vulnerable by logging activities, road construction and land occupation (Freifelder et al., 1998; Brando et al., 2012; Soares-Filho et al., 2012). In spite of increased governmental efforts in the 2000s to improve forest conservation policies (Rajão & Vurdubakis, 2013; Cunha et al., 2016), forest fire rates have remained high (INPE, 2016). According to Moutinho et al. (2016), landowners have responded to these policies by changing deforestation strategies, such as deforestation in smaller patches and more frequent burning to induce gradual degradation, in the hope of remaining undetected by monitoring systems and law enforcement. Some scholars have tried to measure the economic losses of forest fires in the Amazon as a counterweight to the perceived economic benefits of land use change for individual landowners (Nepstad et al., 2001; Andersen et al., 2002; Gerwing, 2002; Menton, 2003; Mendonça et al., 2004; Strand, 2017). However, these studies have not considered the spatial variability of these losses, which may be useful for public policy-making.

In this article we present EcoFire (Economic Cost of Fire), a spatially explicit model developed to estimate the economic losses in the forestry sector caused by forest fires between 2002 and 2041. More specifically, we simulate synergies between selective logging of native forests, fire spread and fire intensity by integrating EcoFire with FISC (Fire Ignition, Spread and Carbon components) and SimMadeira, two models that simulate fire and timber rents (in the absence of fire) that have already been established in the scientific literature (Merry et al., 2009; Brando et al., 2012, 2014; Soares-Filho et al., 2012). In this respect, we aim not only to raise awareness of the costs of forest fires for the timber industry, as has been done in other studies (e.g., Andersen et al., 2002), but more importantly to identify regions where the losses are particularly high and, subsequently, to aid policy-makers in the formulation of fire mitigation policies. In addition, we aim to advance current methods and tools for mapping economic losses from ecosystems services in the Brazilian Amazon. The next section opens our argument with a discussion of the available literature on fire mitigation policies. The third section elaborates our research approach, in which we explain and discuss the integration of the FISC, SimMadeira and EcoFire models as well as our assumptions and primary data sources. In the fourth section, we present the results of our analysis, followed by a discussion on the implications for fire mitigation policies in the region.

2 | THE POLITICS AND ECONOMICS OF FIRE MITIGATION IN THE BRAZILIAN AMAZON

Historically, disincentive-based policies by the federal government have been the most common response to reduce fires in Brazil (Morello et al., 2017). According to the Brazilian Forest Code (LEI 12651/2012), the use of fire is generally prohibited and, if justified, requires a fire management plan as well as prior consent (i.e., licensing) from the state government (art. 38). In practice, however, most fires occur without formal licensing and use insufficient preventive measures, and the few fines issued by the government are rarely paid (Rajão & Vurdubakis, 2013). Consequently, these regulations are inefficient in reducing forest fires (Morello et al., 2017), which has led to calls for the reformulation of current mitigation policies (Cammelli, 2013; Carmenta et al., 2013; Morello et al., 2017). Some existing fire prevention programmes led by governmental and non-governmental organisations with a stronger involvement of local communities have produced good results. These include the Register of Socio-environmental Commitment and Fire Brigade of the Land Alliance (Silvestrini et al., 2011), the “Green Flame” Project in Paragominas (Vilhena, 2016) and the “Green Municipalities” in Pará (Guimarães et al., 2013). In contrast to the punitive character of state-level and federal disincentive-based policies, these local programmes invest most of their efforts in educational, preventive and integrated fire management practices associated with specific land uses, such as pasture and agriculture (Myers, 2006). These programmes also tend to be more sensitive to the needs of small farmers who use fire as part of relatively sustainable slash-and-burn agriculture, while pressuring large farmers who use fire to deforest large tracts of land.

The successes of local fire mitigation programmes can be partially explained by their understanding of forest fires as a problem of collective action. While different economic actors collectively participate in regional land management processes (Tacconi et al., 2006; Cammelli, 2013), land users often do not consider the broader effects of fire. Nepstad and Alencar

(1999) argue that land users are so dependent on fire that it represents an inseparable component of management and expansion of agricultural frontiers. Furthermore, they often lack the willingness or knowledge to invest in fire control, which may have damaging consequences for adjacent economic activities. This is especially the case for large farmers who use fire as a way to reduce the cost of deforestation and pasture expansion (Nepstad & Alencar, 1999; Cammelli, 2013). In addition, forest fires also have damaging consequences for society, such as respiratory ailments and damage to livestock, pasture, crops, houses and forestry resources (Mendonça et al., 2004). The main challenge, therefore, is to make such damaging consequences clear to both landowners and policy-makers in order to incentivise them to adopt more proactive action to reduce forest fires. According to Zybach et al. (2009), for example, forest fire suppression costs can be 50 times lower than the total costs related to the damage that fire causes to society. For this purpose, some authors suggest that economic impact estimates could inform the establishment of preventive measures, such as agricultural credit schemes for rural production (Nepstad et al., 2001; Morello et al., 2017) or integrated fire management programmes (Myers, 2006).

The disintegrated character of fire management practices is confirmed by scholars suggesting that, in spite of damaging consequences, the forestry sector contributes to increasing occurrences of forest fires. Furthermore, even sustainable practices have not been able to mitigate the problem. By augmenting fuel loads on the ground and opening forest understories, selective logging increases the vulnerability of forests to fire (Soares-Filho et al., 2012). Furthermore, roads built by loggers often provide access to land grabbers and cattle ranchers who use fire to clear lands (Freifelder et al., 1998; Brando et al., 2012). Since the use of fire has both costs and benefits, although borne differently by different actors, it is crucial to demonstrate the economic consequences of forest fires on the provision of forest products and services as key information to decision-makers from both public and private sectors (Gerwing, 2002; Menton, 2003).

Some studies have already estimated the economic impact of forest fires on sustainable timber production. Andersen et al. (2002), for example, measured fire-induced forest cover loss in agricultural areas and the effects on the values of timber. The authors assumed that sustainable timber supply is worth US\$ 28 ha/year and that destructive fire entails a loss of 100% for a period of 50 years following the fire event (Andersen et al., 2002, p. 178). Alternatively, Mendonça et al. (2004) integrated data from the literature to assess losses in agriculture, costs of respiratory illnesses, forest resource losses, and CO₂ emissions from forest fires. They calculated a total average yearly loss ranging between US\$ 90 million and US\$ 5 billion during the period 1996–1999 for the entire Brazilian Amazon. For these years, forest losses represent on average 0.5% of these total losses, equivalent to a value of US\$ 5 ha/year (Mendonça et al., 2004). Although these studies are an important starting point, there are some limitations. Both approaches have disregarded the way in which different fire intensities may have varying effects on different tree species and the quality of timber, instead treating the Amazon forest as a homogeneous ecosystem for which they calculate a single average marginal economic loss. According to Andersen et al. (2002), for example, it is “obviously infeasible here to attach a different value to each of the several hundred million hectares of Amazon forest” (p. 170). Although such estimates provide valuable information, they are insufficient to guide policies aimed at reducing the impact of forest fires, since they do not consider the spatial variability of their occurrence, the economic impact and ecological effects. By extension, they do not provide information about where preventive measures may be most cost effective.

3 | METHODS

3.1 | General approach

To demonstrate the spatial variability of economic losses caused by fire in sustainable timber production, we have developed the spatially explicit model EcoFire. In this paper, sustainable timber production refers to reduced impact logging (RIL), which corresponds to the legal norms and practices (CONAMA, 2009) for minimising the ecological impacts in the areas of timber concessions (details presented in Section 3.2). The EcoFire model processes and combines spatial data on the occurrence and intensity of forest fires, data on the impact variation on different tree species, and economic data on timber production in the Amazon in order to estimate the economic losses. We simulated fire occurrence and intensity by using the FISC model developed by Silvestrini et al. (2011) and Soares-Filho et al. (2012). In addition, we simulated timber production and rents by using the SimMadeira model developed by Merry et al. (2009). Both models were adapted to provide data for the entire Amazon region, with a spatial resolution of 1 km², and to facilitate integration with the EcoFire model (Figure 1), details of which will be given in the following sections. These new versions of the FISC and SimMadeira models that integrate with EcoFire were developed by Soares-Filho, Lima et al. (2017) and Soares-Filho, de Oliveira et al. (2017) (available at <http://amazones.info>). To establish the relationship between fire and timber, EcoFire consists of a set of heuristics (i.e., empirical parameters) that represent the economic impact of different fire intensities on different commercial tree species. For instance, low-intensity fires can reduce 5% of the selling price when they reach commercial timber coming from mature trees (Figure 1; see Section 3.3 for details).

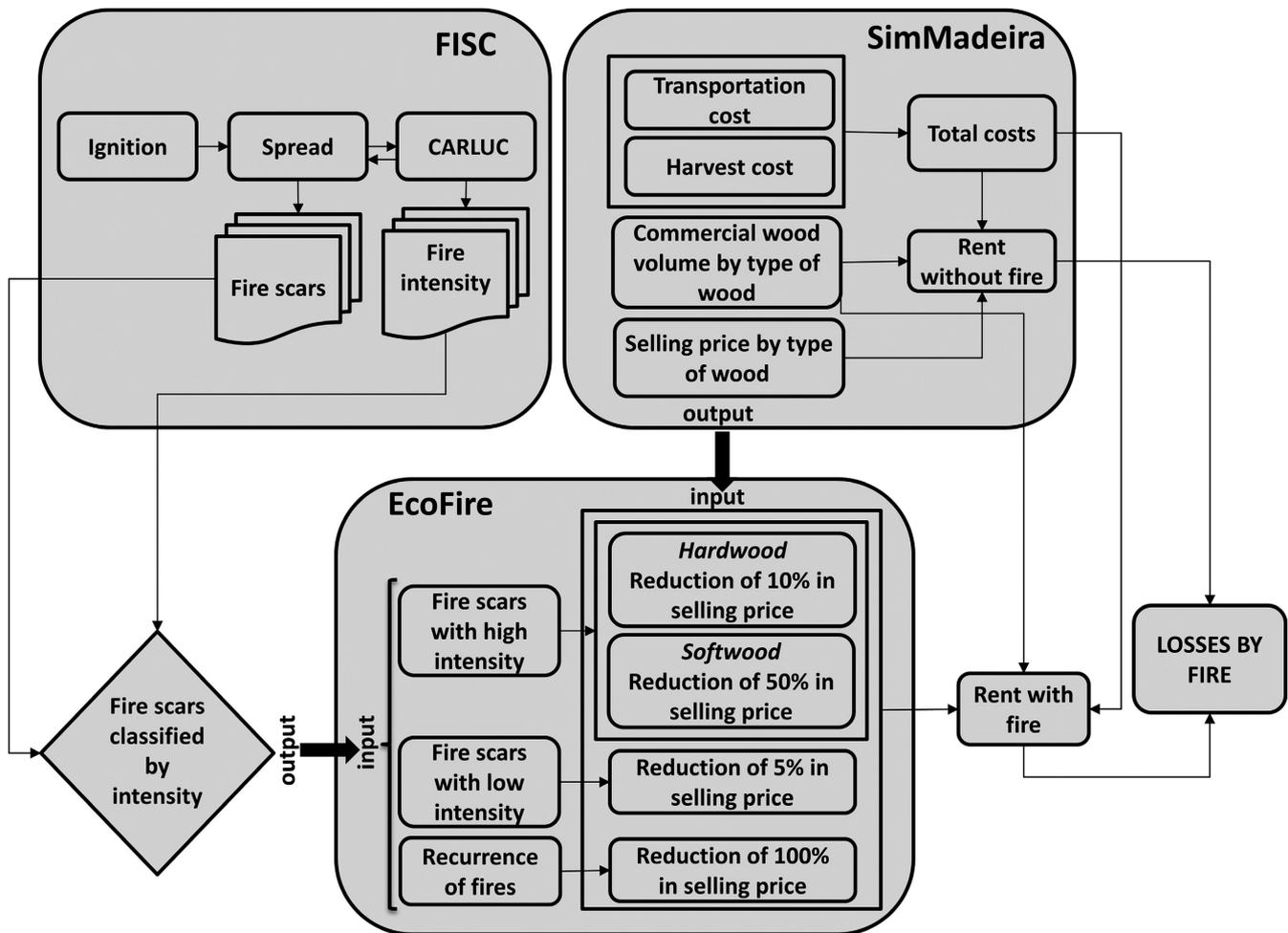


FIGURE 1 Integration of the models and heuristics derived from fieldwork used by EcoFire to calculate the economic losses to the sustainable timber production in the Amazon.

The annual net revenues (rents) and economic losses of sustainable timber are presented as the equivalent annual annuity (EAA). Based on the net present value (NPV), for which we used an interest rate of 5%, the EAA derives the annual uniform value of a project/activity that is evenly spread over its lifespan. The reference period of our analysis was chosen on the basis of current legislation in the state of Mato Grosso (Decreto no. 2015[2014]). This legislation states that a full production cycle of sustainable timber covers on average a 30-year period (art. 9-II). Choosing a starting year that facilitates the integration of the models used in our analysis, we therefore calculate the economic losses for the period 2012–2041. At the same time, this legislation prohibits timber harvests in cases of fire recurrence (i.e., more than once) within a 10-year period (art. 25). To account for the possibility of an economic impact from fires in preceding years, we estimate fire occurrence for the period of 2002–2041. Correspondingly, EcoFire assumes a 10-year period as the maximum duration that fire entails timber losses, because this model takes into account that 90% of biomass losses could recover within this time interval as is estimated by the CARLUC component in the FISC model (Soares-Filho et al., 2012).

3.2 | Localisation of forest fires (FISC model)

Fire Ignition, Spread and Carbon components is a process-based understory fire model developed for tropical forests (Silvestrini et al., 2011; Soares-Filho et al., 2012). Since the initial version was implemented for the Xingú region in the state of Mato Grosso, we expanded the FISC model to simulate fire ignition and propagation processes in the entire Amazon biome. Furthermore, we expanded the spatial resolution from 10.2 ha (320 m × 320 m) to 25 ha (500 m × 500 m) to accommodate the larger area of analysis.

The ignition and fire spread components of FISC are inter-related to provide data on fire occurrence, namely the location of fire ignition and the subsequent spread of fire across the landscape that result in forest fire “scars.” The fire

ignition component in FISC simulates hot pixels¹ as a function of land use that are modulated by spatial determinants (e.g., distance to deforested land, roads and towns, and elevation), land-use restrictions (e.g., protected areas) and climatic seasonality represented by monthly data on vapour pressure deficit (VPD) (inputs, see Supporting Information Table S1). Following the fire ignition, the fire spread component employs a cellular automata model to simulate fire propagation as a function of distance to ignition sources, terrain features (e.g., declivity, obstacles, different land uses), fuel loads and wind direction (inputs, see Supporting Information Table S1). Furthermore, this component includes data on forest climatic conditions and availability of fuel loads (e.g., dry wood) from the CARLUC model (see below). Both components (ignition and spread) require probability maps for the simulations (e.g., of climatic data). FISC uses, among other techniques, logistic regression to generate these maps and probability density functions, which define where fires are likely to occur and spread. For the calibration of the fire ignition components, we compared the monthly number of simulated and observed hot pixels (NOAA; INPE, 2016) between 2004 and 2010 (see Supporting Information, Figure S2). For the validation of fire spread data we compared burned area metrics, spatial distribution and scar size between the simulated and observed data of Morton et al. (2013) for the years 2002–2010 (see Supporting Information, Figures S3 and S4). For 2002–2010, the scars simulated from FISC present a difference of 18% (lower) in relation to average burnt areas in the map of Morton et al. (2013) (see details in Supporting Information; Figures S1 and S3).

FISC also contains a carbon and land use change component, the CARLUC model (Hirsch et al., 2004), for simulating fuel load dynamics, forest regrowth and carbon emissions. The calibration of CARLUC was based on field observations in the Tapajós National Forest (for more details see Brando et al. 2014; and Supporting Information Figures S5, S6, and Table S3). CARLUC simulates fire intensity dynamically, based on the amount of available fuel load, and the fire spread and combustion heat (Byram, 1959). As fire intensity has a direct impact on timber production (see Section 3.3), we define thresholds of fire intensity according to the work of Brando et al. (2014), which relates fire intensity with the tree mortality. In this way, we define forest fires of high intensity as being higher than 400 kW/m, when tree mortality in general exceeds 50%; fire events with values equal to or below this threshold are considered low intensity as they cause 10%–20% tree mortality. For instance, during two drought years, 2005 and 2010, roughly 82% and 65% of fire scars simulated for those years, respectively, presented high-intensity values, reaching more than 800 kW/m. Therefore, FISC allows us to investigate the changes in fire regime, such as fire frequency, extent and interval, to simulate post-fire damage, for example, burnt area, type of vegetation affected and the recurrence of fire (Figure 2).

The occurrences of fire simulated by FISC are closely related to the expansion of the agricultural frontier and the consequential forest fragmentation (Silvestrini et al., 2011; Soares-Filho et al., 2012). To take this into account, deforestation data for the years 2012–2015 were obtained from the Programme for the Estimation of Deforestation in the Brazilian Amazon from the Brazilian Institute for Space Research (INPE, 2015). For the remaining years (2016–2041), we assumed a constant yearly deforestation rate of 5,000 km², which corresponds to the deforestation target of the National Climate Change Plan and the value detected by PRODES in 2012. As such, the recent increase in deforestation rates from 2013 and 2016 was disregarded, which renders our approach more conservative. This resulted in a spatially explicit simulation of deforestation and fire scars in the Amazon between 2012 and 2041 (see Supporting Information Figure S7).

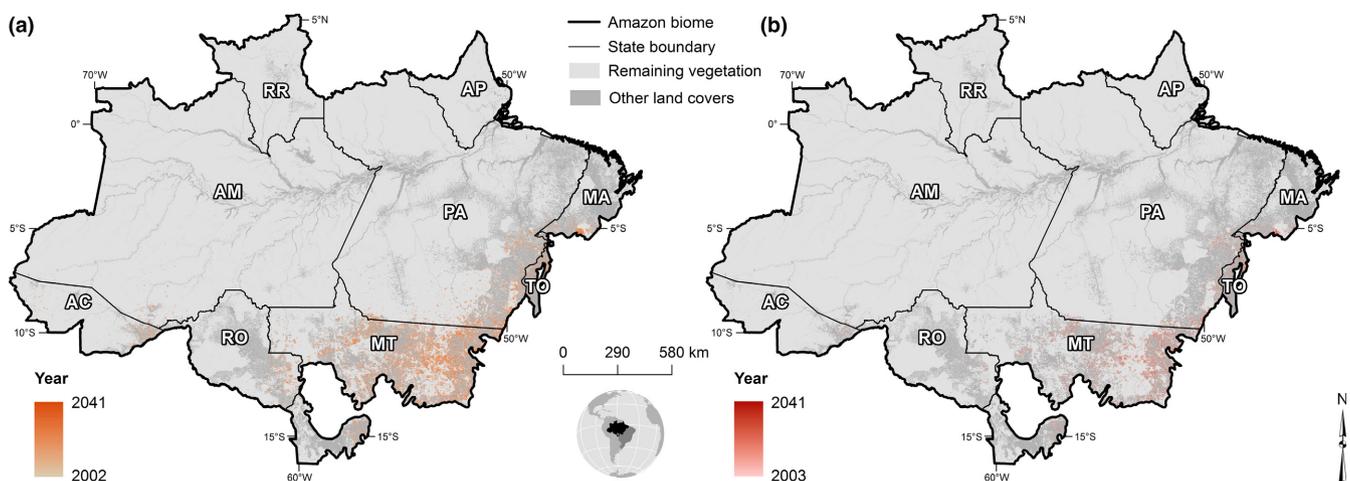


FIGURE 2 Fire occurrence (a) and recurrence (b) output from the FISC model.

Source: Data calculated by the authors using the FISC model.

3.3 | Rents and timber production (SimMadeira model)

The SimMadeira model simulates the timber production of native forests in the Amazon (with a spatial resolution of 1 km²) based on RIL (Reduced Impact Logging). We understand RIL to be a form of sustainable forest production, which refers to forms of harvest planning and logistics that maximise productive efficiency while minimising the impacts on timber production. RIL reflects the norms and practices proposed by the Brazilian government for timber concessions. In this case, (1) timber production may not exceed 0.86 m³ ha year⁻¹ and involves the adoption of forest management units; (2) annual harvest areas are defined; and (3) protection occurs against re-cutting during the harvest cycle (CONAMA, 2009). SimMadeira calculates sustainable logging rents (parameters, see Supporting Information Table S4) based on production costs (see Supporting Information Tables S5 and S6) and timber market prices (see Supporting Information Table S7) in the Amazon (Merry et al., 2009). According to Brazilian resolution number 406/2009 (CONAMA, 2009), timber harvest cannot occur in protected areas and therefore SimMadeira does not envisage timber extraction in these areas.

For the development of EcoFire, SimMadeira was extended to provide robust geographically differentiated estimates of sustainable timber rents for 40 timber genera (each including one or more species) based on their ecological distribution (see Supporting Information Table S7). These genera build on definitions and valuations developed by the Institute of Man and the Environment of Amazonia (IMAZON), which provide the most complete data source currently available (Pereira et al., 2010). We use data on the occurrence of tree species from online databases of species occurrences (CRIA - Specieslink [2015] and Global Biodiversity Information Facility [GBIF, 2015]) to model the distribution of these species based on their ecological niches (see Section 2.1 of the Supporting Information for more details).

The EcoFire model evaluates the economic losses for different types of timber affected by fires of different intensity. The SimMadeira model differentiates these genera of commercial timber on the basis of their densities and resistances, distinguishing between hardwood (i.e., high density, high commercial value and high resistance to fire) and softwood (i.e., low density, low commercial value and low resistance to fire). Although classifications already exist in the literature (Melo et al., 1990; Dias & Lahr, 2004), they do not cover all genera/species listed by IMAZON. For this reason, four forest engineers independently classified each commercial timber genus/species from the IMAZON list as either hardwood or softwood. The list contains both genera, such as *Aspidosperma* spp., with many tree species that go under the common name of “Peroba,” and tree species such as *Mezilurus itauba* (Itaúba) that correspond to one genus. In this way, our classification draws on the market experience of these forest engineers to ascertain the impact of fires on timber production. Despite using a different approach, our classification based on forest engineer experience corresponds to 70% and 83% of comparable data presented by Dias and Lahr (2004) and (Nahuz et al., 2013), respectively. According to technical standards, high-density timber has a value of 835 kg/m³ or higher, while low-density timber has lower values.

The commercial timber volume for each tree species was based on volume data provided by Merry et al. (2009). Since these data are based on remote sensing biomass maps, the values only represent indirect measurements of commercial timber volumes, which is a common challenge in the available literature (Goetz et al., 2009; Baccini et al., 2012; Mitchard et al., 2014). To calculate the volume of each type of timber, we used our species distribution maps (see Figure S10) to locate the pixels representing each timber type, after which we allocated the commercial timber volumes of each pixel to the timber types present according to their relative abundance (see supporting Information Figures S11 and S12). The relative abundance of tree species was calculated based on an Amazon-wide distribution of 4,962 tree species estimated by ter Steege et al. (2013).

We calculated the potential value of timber production by multiplying the timber volume estimates by the respective prices. The gross revenues of each timber genus were allocated to the softwood and hardwood categories (see above) to calculate timber losses accordingly in the EcoFire model (see supporting Information Figure S13). Roughly 14% of the harvested volume and 20% of the gross revenue stem from hardwood and the remainder from softwood timber types. The rent or stumpage value (i.e., residual value for the landowner) from harvest timber was obtained by deducting all harvest costs from the gross revenue. The rent π_j for a cell j was calculated by SimMadeira as follows:

$$\pi_j = \sum_{\text{for each } g \text{ in } j} [(p_{jg} * V_{jg}) - [(TC_j + HC_j + PC_j) * (1 + I)]] \quad (1)$$

where p_j is the location-specific price of the timber type g in the cell j , V is the commercial volume of the timber type g , TC is the transportation cost of round wood from a specific cell j to the location of the nearest milling centre, HC and PC are the harvest and processing costs, respectively, and I is a social discount rate (the model assumes an inflation-adjusted rate of 5%). Transportation costs range from 0.05 to 2,564 US\$/m³/km (see Supporting Information Table S6), depending on land-use types (e.g., deforested area, public forest without designation), road paving conditions (e.g., paved, unpaved,

four-lane road) or waterway conditions (e.g., navigable waterway, limited navigability, navigability only in rainy season). All costs remained unchanged in relation to the original version of SimMadeira, while rents were between 4.3% and 10% higher in the extended version due to a more refined representation of timber distribution and volumes (Merry et al., 2009). In the SimMadeira model, prices are taken as fixed throughout the entire harvest cycle due to difficulties of accurately forecast timber prices as well as other input prices that would also affect rents (e.g., fossil fuel prices affecting transportation costs), which also introduces some degree of uncertainty to our analysis. Moreover, timber prices are estimated as a weighted average value for a small sample of genus and species, since the available literature does not provide standardised data on variation in species, class and density in different regions in the Amazon.

3.4 | Economic losses from destructive fire (EcoFire model and heuristics)

The data on fire occurrence from FISC and the data on timber revenues from SimMadeira were inserted in the EcoFire model, but our analysis of economic impacts required additional data on the relation between fire and timber production to complete the input data. Since few studies provide such data in consideration of species variability, or differentiate types of forest fires (Andersen et al., 2002; Gerwing, 2002; Menton, 2003; Mendonça et al., 2004), our analysis obtained these data from questionnaires. We conducted 30 interviews with farmers, forest engineers, forest rangers, loggers and sawmill owners in the Sinop region in the state of Mato Grosso (see questionnaire in the Supporting Information and Table S9). This municipality was selected due to its high density of logging activities and fire occurrence (Silvestrini et al., 2011; Morton et al., 2013; INPE, 2016).

According to these questionnaires, there are three major drivers of fire-related economic losses in timber production: (1) fire intensity; (2) fire recurrence; and (3) the resilience to fire of individual timber species. These findings correspond with reports in the available literature on the relation between fire and forest damage (Holdsworth & Uhl, 1998; Nepstad et al., 1999; Barlow et al., 2012; Brando et al., 2012, 2014). The majority of interviewees (75%) pointed out that low-intensity fire reduces the average selling price of 1 m³ of softwood and hardwood by around 5%. In the case of high-intensity fires, most interviewees (83%) reported an average price reduction of 10% for hardwood, while price reductions for softwood reach up to 50% due to reduced tree resilience (Figure 1). Finally, respondents reported substantial economic losses to timber production regardless of fire intensity, which accounts for state legislation (i.e., Decreto no. 2015[2014]) that prohibits timber harvests in cases of fire recurrence (i.e., more than once) within a 10-year period. Extending this to the entire Amazon, we assume that the recurrence of fires entails a 100% loss of commercial value during this prohibitive period (Figure 1). Although the small number of respondents may pose limitations for generalisation, the introduction of these “heuristics” to our EcoFire model (Figure 1) introduces a novelty to existing literature on modelling the losses of fire.

In general, EcoFire estimates the economic losses in a number of consecutive steps. First, SimMadeira provides the EAA of sustainable timber production in the absence of fire for the period 2012–2041. To account for the uncertainties of the model, we considered bounds corresponding to $\pm 15\%$ variation in timber prices (see rents in the absence of fire in Supporting Information Figures S14 and S15). EcoFire then calculates the economic impact of fire on each pixel, taking into account the timing of fire events in relation to the harvest year and the set of heuristics related to fire damage (Figure 1). Since the economic impact varies according to the type of fire, we used the annual fire scar data from FISC classified by intensity for the period 2002–2041. The results, therefore, represent the economic impact of 40 years of fires in a full timber production cycle (see Section 3.1). EcoFire calculates economic losses based on the commercial value of timber (see Supporting Information Figure S13). In this way, economic losses to timber net revenue (rents) as a result of fire are calculated by assuming that harvest costs remain the same, and by subtracting these costs from the decreased commercial values of fire-affected timber yields. In cases where net rents are negative, we set the net rent to zero since those areas are not profitable. The model estimates effective losses (losses in simulated burnt areas that eventually would end up logged) and “potential values/losses” as if all burnt areas would be logged in the near future. In the same way that it is useful to consider the total amount of CO₂ that would be emitted if the Amazon were to be completely deforested, it is also relevant to consider the impact of fire on timber as if all those areas affected would be logged.

4 | RESULTS

Our spatially explicit analysis indicates a substantial and growing decoupling between sustainable timber production and fire events in the Brazilian Amazon. While forest fires are predicted to occur on 11.1 millions of hectares (Mha) of newly burnt areas and 3.8 Mha on recurrent areas between 2012 and 2041, we estimate that only 693 ± 168 thousand ha will occur in productive areas (see Supporting Information Figure S16). Conversely, these forest fires will affect only 2% of the

total cumulative area (i.e., 48 ± 7 Mha) designated for sustainable timber production. We also observed a declining share of affected areas in the reference period. Even though the annual fire-affected area would increase to 31 ± 11 thousand ha by 2024, the expansion of timber production grows at a faster rate in areas not prone to fire, hence the percentage of the harvest area affected by fire reduces to 0.5% by the end of the period, in 2041 (Figure 3).

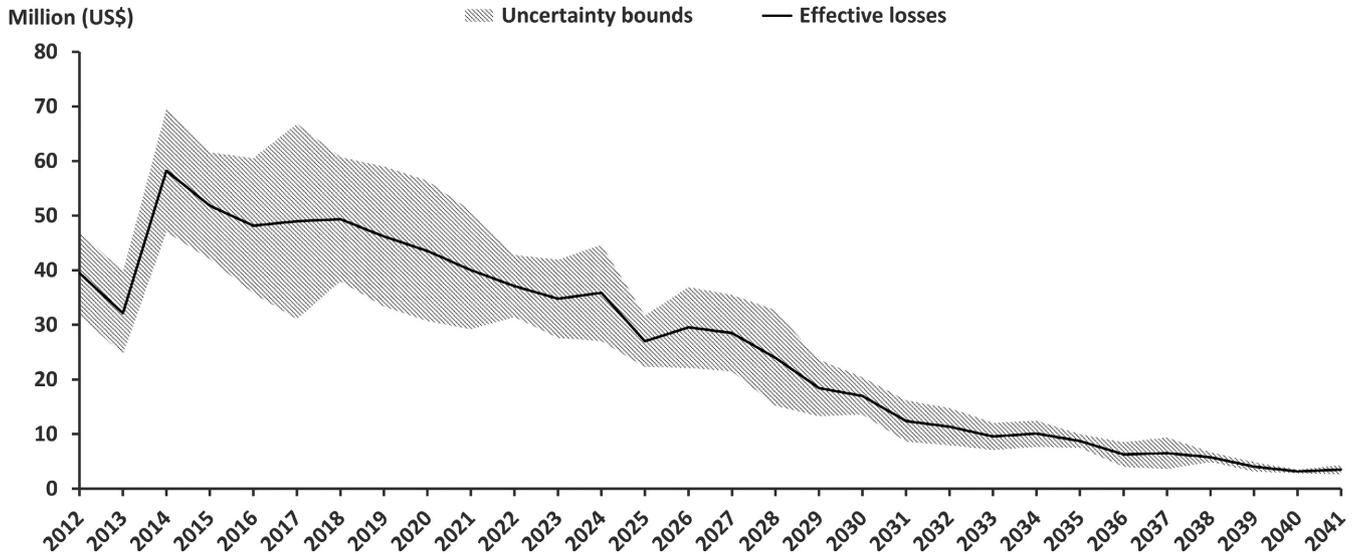


FIGURE 3 Annual economic losses by fire to sustainable timber production and uncertainty bounds ($\pm 15\%$ of average timber prices).
Source: Data calculated by the authors using the EcoFire model.

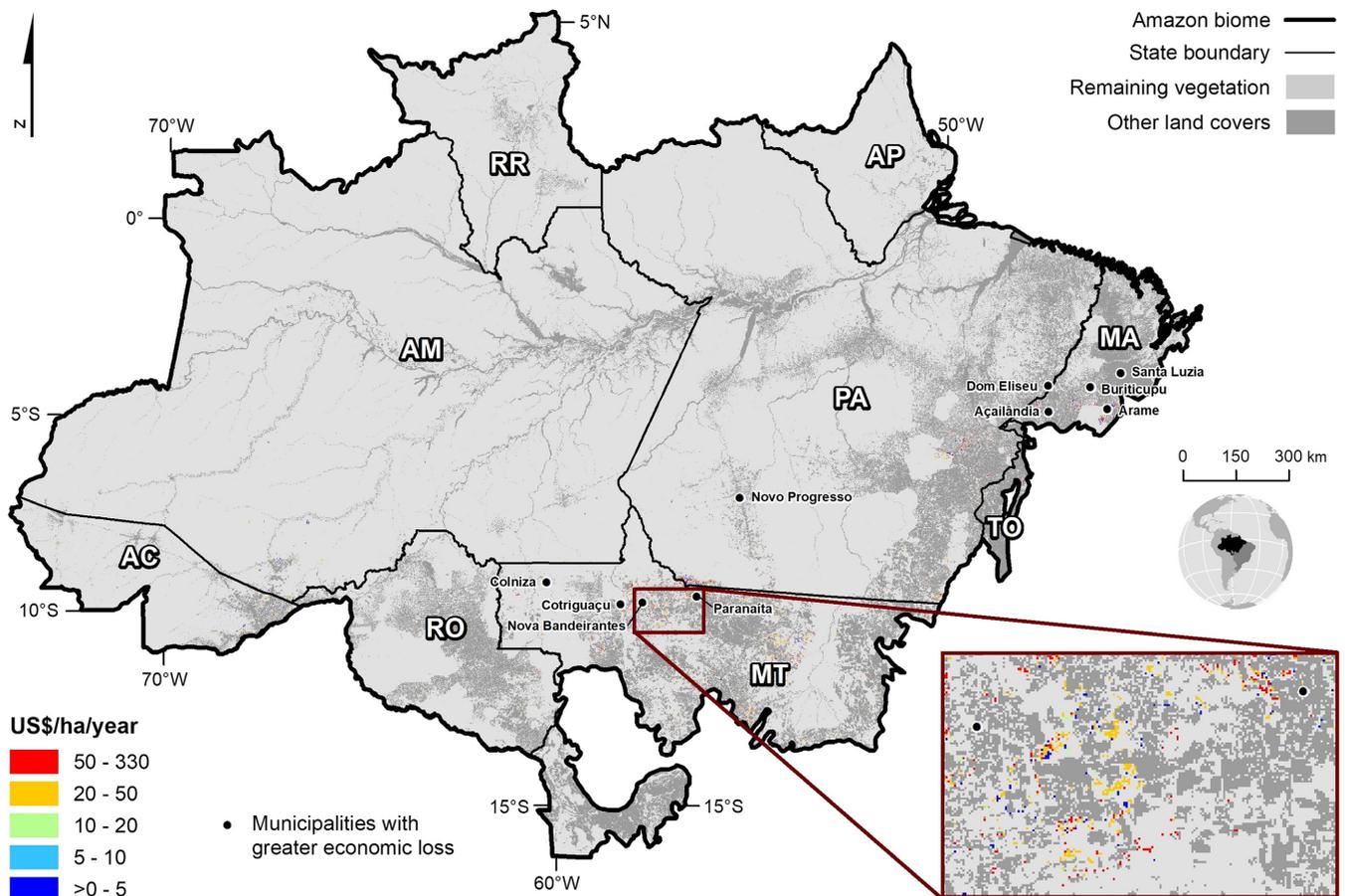


FIGURE 4 Economic losses by fire to sustainable timber production, highlighting the region of Sinop.
Source: Data calculated by the authors using the EcoFire model.

Between 2012 and 2041, the total net revenue (rent) of sustainable timber production is estimated to increase from US\$ 431 ± 91 million in 2012 to US\$ 913 ± 360 million in 2041, peaking at US\$ 1.04 ± 0.3 billion in 2023 (see Supporting Information Figure S15). Concurrently, the economic impact of forest fires peaks at US\$ 54 ± 11 million (8%) in 2014 and steadily declines to US\$ 2.7 ± 0.8 million (0.3%) by the end of the logging cycle (Figure 3). The gradual reduction of the impact of forest fires corresponds with the lower deforestation rates projected by FISC. Moreover, these observations indicate that most economic losses from forest fires occur before 2020, after which values decrease at an average rate of 10% per year. In the policy scenario considered, the decrease in the economic losses from fire reflects a migration of sustainable timber production away from the agricultural frontier, where forest fires are more likely due to high levels of fragmentation and dryness, towards more remote areas where valuable tree species are more abundant (Broadbent et al., 2008).

These impacts are not evenly distributed over the Amazon region. In the affected areas, the average economic losses induced by forest fire were estimated to be US\$ 39 ± 2 ha/year, which represents a loss of 0.8% of the expected rent. We found that most forest fires concentrate near the agricultural frontier or major roads in the north-western Mato Grosso, Eastern Pará, and south-central Maranhão states (Figure 4). Moreover, these regions may incur costs of up to US\$ 183 ± 30 ha/year, especially in areas hit by recurrent fires near milling centres.

Although the aggregate economic losses may be very low (see previous paragraph), these observations indicate that the impact of forest fires at a local scale can be substantial. These results are slightly amplified when estimating the potential economic losses of forest fires (if all burnt areas were to be logged in the near future). On average, the potential economic losses by forest fire for sustainable timber production were estimated at US\$ 726 ± 193 million, with an NPV of US\$ 689 ± 184 million, only 4% of the total net revenue (US\$ 1.52 ± 0.2 billion) from a 30-year logging cycle in the absence of fire (Figure 5). By contrast, economic losses were absent in large expanses of the Northwestern Amazon, where there is a paucity of forest fires due to low agricultural activity and where high transportation costs (e.g., few roads) render sustainable forest production largely unprofitable.

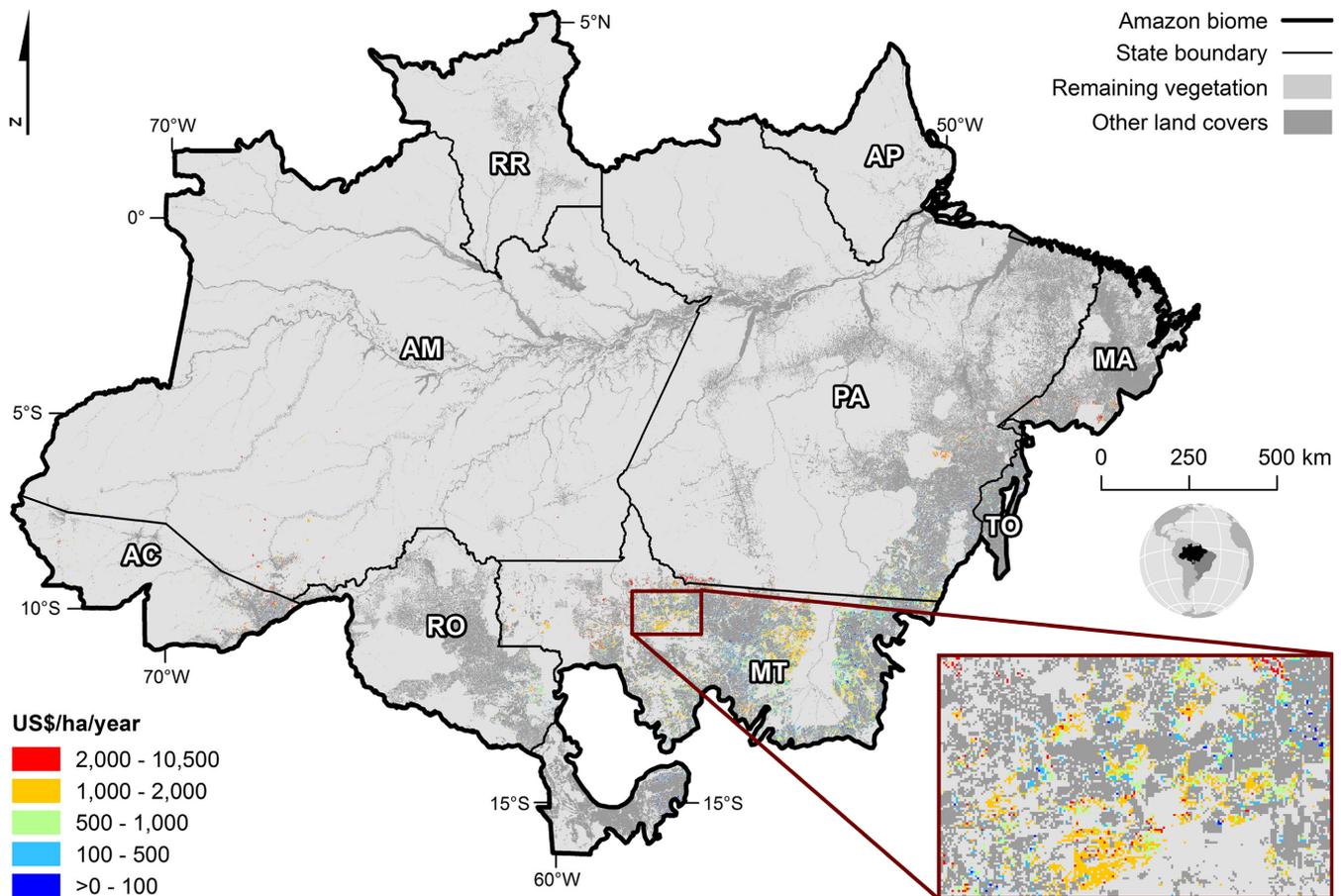


FIGURE 5 Potential economic losses (net present value, NPV) by fire to the sustainable timber production, highlighting the region of Sinop. *Source:* Data calculated by the authors using the EcoFire model.

5 | DISCUSSION AND CONCLUSION

Our spatially explicit analysis of the economic impact from forest fires for sustainable timber production reveals higher economic losses than suggested by other scholars. Mendonça et al. (2004), for example, estimated the yearly total loss in timber production at US\$ 1–13 million, while our study indicates an annual average loss of US\$ 29 ± 4 million. Moreover, Andersen et al. (2002) reported that fire causes average losses to timber production of US\$ 28 ha/year, while we estimate a loss of US\$ 39 ± 2 ha/year. The main difference between these estimates is to be found in diverging data sources and methodologies. For instance, our analysis includes fire scars where forest caught fire in forest understories, differentiates between different tree species, and considers survival variation based on tree lifespan, which differs from studies that measure the losses only by the mortality of trees at a constant rate for the whole forest. More importantly, our study has identified specific regions where economic losses are very high and drive up the average costs induced by forest fires. In these regions, the economic losses far exceed the relatively low shares of average net revenues (i.e., 4%) estimated for the entire Amazon region.

The most important factor that drives up economic losses from forest fires is the proximity of sustainable timber production to the agricultural frontier, where different economic practices (i.e., sustainable logging and agriculture) compete. Feeding these assumptions into our model (see Section 3.2), we found that such damaging consequences are most likely to be concentrated in the northwestern Mato Grosso, Eastern Pará, and south-central Maranhão states. Moreover, economic losses tend to be highest in the first few years of the 30-year logging cycle, after which sustainable logging activities are likely to move away from the agricultural frontier.

These findings have important implications for decision-makers in both private and public sectors in terms of forest fire prevention. First, preventive efforts could target the fire “hotspots” identified by our analysis to reduce economic losses to sustainable timber production. With respect to fire mitigation programmes deployed in the Amazon (see Section 2), we found little evidence that they target the fire “hotspots” identified in our analysis. For instance, only two of the 10 most affected municipalities (Dom Eliseu in the state of Pará and Paranaíta in Mato Grosso) are part of regional fire mitigation programmes. By contrast, we could not find local programmes in the municipalities of Arame, Buriticupu or Santa Luzia in Maranhão, and Colniza, Cotriguaçu and Nova Bandeirantes in Mato Grosso, even though they are likely to experience the heaviest fire losses in the region (see Supporting Information Tables S10, S11 and S12). In addition to spatial prioritisation of fire prevention efforts, we also argue that such efforts must be quick to materialise to avoid most economic losses to sustainable timber production.

The limitations of incentive-based and disincentive-based policies indicate that alternative approaches to fire prevention need to be considered. By exposing the potential economic losses, our study empowers the logging sector as an important ally of fire mitigation strategies (Nepstad et al., 2001; Mendonça et al., 2004). At the same time, however, the negative economic impacts of forest fires may include a myriad of other economic activities that also need to be included in integrated fire management programmes (Myers, 2006). Unless a broader agreement is reached with rural producers, fire mitigation policies are unlikely to succeed (Tacconi et al., 2006; Cammelli, 2013; Carmenta et al., 2013).

We recognise that our spatially explicit analysis needs refinement in some respects to better reflect the complexity of estimating economic impacts. For instance, our assumptions were constrained by a lack of parameters related to field experiments, scarce literature on the reaction of various tree species to different fire-intensity damage, and a lack of robust data on the occurrence of tree species in the Amazon. At the same time, however, our study merely scratches the surface of providing economic impact data on forest fires. We have assessed only one of 17 ecosystem services identified by Costanza et al. (1997), namely the provision of raw materials, and only one product within that category (i.e., timber), which suggest that the economic losses from forest fires may be much higher than estimated in our study. In this respect, our analysis demonstrates an important first step is to advance geographically differentiated impact assessments.

ACKNOWLEDGEMENTS

We are thankful to EMATER – SINOP (Empresa de Assistência Técnica e Extensão Rural do Estado do Mato Grosso) for logistics for fieldwork. The authors also thank the forestry engineers, forest producers and officials from the Secretary of Environment from Sinop, the Brazilian Institute of Environment and Renewable Natural Resources (IBAMA), Sinop Fire Department, the Brazilian Company of Agricultural Research (Embrapa Agrossilvipastoril, Sinop-MT) and the Federal University of Mato Grosso. This study was carried out with the support from Norway through the World Bank, and has benefited from detailed feedback by Jon Strand, Michael Tomann, Yoanna Kraus, Jeffrey Vincent and Johannes Schielein. We also acknowledge the support from CAPES (Coordenação e Aperfeiçoamento de Nível Superior), FAPEMIG (Minas

Geraias State Research Foundation) and CNPQ (Conselho Nacional de Desenvolvimento Científico e Tecnológico) and the Humboldt Foundation.

ENDNOTE

¹ Heat signals from fire as observable from satellite-based infrared sensors.

ORCID

Raoni G. Rajão  <http://orcid.org/0000-0002-1133-4837>

REFERENCES

- Andersen, L. E., Granger, C. W., Reis, E. J., Weinhold, D., & Wunder, S. (2002). The dynamics of deforestation and economic growth in the Brazilian Amazon. *Canadian Journal of Agricultural Economics/Revue Canadienne*, *51*, 437–442. <https://doi.org/10.1017/CBO9780511493454>
- Baccini, A., Goetz, S. J., Walker, W. S., Laporte, N. T., Sun, M., Sulla-Menashe, D., Hackler, J., Beck, P. S. A., Dubayah, R., Friedl, M. A., Samanta, S., & Houghton, R. A. (2012). Estimated carbon dioxide emissions from tropical deforestation improved by carbon-density maps. *Nature Climate Change*, *2*, 182–185. <https://doi.org/10.1038/nclimate1354>
- Baccini, A., Walker, W., Carvalho, L., Farina, M., Sulla-Menashe, D., & Houghton, R. A. (2017). Tropical forests are a net carbon source based on aboveground measurements of gain and loss. *Science*, *358*, 230–234 <https://doi.org/0.1126/science.aam5962>
- Barlow, J., Parry, L., Gardner, T. A., Ferreira, J., Aragão, L. E. O. C., Carmenta, R., Berenguer, E., Vieira, I. C. G., Souza, C., & Cochrane, M. A. (2012). The critical importance of considering fire in REDD+ programs. *Biological Conservation*, *154*, 1–8. <https://doi.org/10.1016/j.bioc.2012.03.034>
- Brando, P. M., Balch, J. K., Nepstad, D. C., Morton, D. C., Putz, F. E., Coe, M. T., Silverio, D., Macedo, M. N., Davidson, E. A., Nobrega, C. C., Alencar, A., & Soares-Filho, B. S. (2014). Abrupt increases in Amazonian tree mortality due to drought-fire interactions. *Proceedings of the National Academy of Sciences*, *111*, 6347–6352. <https://doi.org/10.1073/pnas.1305499111>
- Brando, P. M., Nepstad, D. C., Balch, J. K., Bolker, B., Christman, M. C., Coe, M., & Putz, F. E. (2012). Fire-induced tree mortality in a neotropical forest: The roles of bark traits, tree size, wood density and fire behavior. *Global Change Biology*, *18*, 630–641. <https://doi.org/10.1111/j.1365-2486.2011.02533.x>
- Broadbent, E., Asner, G., Keller, M., Knapp, D., Oliveira, P., & Silva, J. (2008). Forest fragmentation and edge effects from deforestation and selective logging in the Brazilian Amazon. *Biological Conservation*, *141*, 1745–1757. <https://doi.org/10.1016/j.biocon.2008.04.024>
- Byram, G. (1959). Combustion of forest fuels. In K. Davis (Ed.), *Forest fire: Control and use* (pp. 61–89). New York, NY: McGraw-Hill.
- Cammelli, F. (2013). *Smallholders collective action and fire risk in the Brazilian Amazon*. Firenze, Italy: University of Firenze.
- Carmenta, R., Vermeylen, S., Parry, L., & Barlow, J. (2013). Shifting cultivation and fire policy: Insights from the Brazilian Amazon. *Human Ecology*, *41*, 603–614. <https://doi.org/10.1007/s10745-013-9600-1>
- CONAMA (2009). *Resolução n. 406 de 2 de fevereiro de 2009. Estabelece parâmetros técnicos a serem adotados na elaboração, apresentação, avaliação técnica e execução de Plano de Manejo Florestal Sustentável - PMFS com fins madeireiros, para florestas nativas e suas form.* Brazil: CONAMA.
- Costanza, R., D'Arge, R., De Groot, R., Farber, S., Grasso, M., Hannon, B., & Raskin, R. G. (1997). The value of the world's ecosystem services and natural capital. *Nature*, *387*, 253–260. <https://doi.org/10.1038/387253a0>
- CRIA - Specieslink (2015). *Ocorrência de espécies [online]* <http://slink.cria.org.br>.
- Cunha, F. A. F. de S., Börner, J., Wunder, S., Cosenza, C. A. N., & Lucena, A. F. P. (2016). The implementation costs of forest conservation policies in Brazil. *Ecological Economics*, *130*, 209–220 <https://doi.org/10.1016/j.ecolecon.2016.07.007>
- Decreto no 2015 (2014). *Planos de Manejo Florestal Sustentável Madeireiro no Estado de Mato Grosso. [online]*. Publicado no DOE em 12 fev 2014. Available from: http://www.normasbrasil.com.br/norma/decreto-2152-2014-mt_265586.html [Accessed 15 Nov 2016].
- Dias, F. M., & Lahr, F. A. R. (2004). Strength and stiffness properties of wood esteemed through the specific gravity. *Scientia Forestalis*, *65*, 102–113.
- Freifelder, R. R., Vitousek, P. M., & D'Antonio, C. M. (1998). Microclimate change and effect on fire following forest-grass conversion in seasonally dry tropical woodland1. *Biotropica*, *30*, 286–297. <https://doi.org/10.1111/j.1744-7429.1998.tb00062.x>
- GBIF (2015). *Data species occurrences [online]*. Available from: <http://www.gbif.org/occurrence> [Accessed 10 Apr 2018].
- Gerwing, J. J. (2002). Degradation of forests through logging and fire in the eastern Brazilian Amazon. *Forest Ecology and Management*, *157*, 131–141. [https://doi.org/10.1016/S0378-1127\(00\)00644-7](https://doi.org/10.1016/S0378-1127(00)00644-7)
- Goetz, S. J., Baccini, A., Laporte, N. T., Johns, T., Walker, W., Kelldorfer, J., Houghton, R. A., & Sun, M. (2009). Mapping and monitoring carbon stocks with satellite observations: A comparison of methods. *Carbon Balance and Management*, *4*, 2. <https://doi.org/10.1186/1750-0680-4-2>
- Guimarães, J., Veríssimo, A., Amaral, P., Pinto, A., & Demachki, A. (2013). *MUNICÍPIOS VERDES: Caminhos para a sustentabilidade*. Belém, Brazil: Imazon.

- Hirsch, A. I., Little, W. S., Houghton, R. A., Scott, N. A., & White, J. D. (2004). The net carbon flux due to deforestation and forest re-growth in the Brazilian Amazon: Analysis using a process-based model. *Global Change Biology*, *10*, 908–924. <https://doi.org/10.1111/j.1529-8817.2003.00765.x>
- Holdsworth, A., & Uhl, C. (1998). *O fogo na floresta explorada e potencial para a redução de incêndios florestais na Amazônia*.
- INPE (2015). *Projeto PRODES: Monitoramento da floresta Amazônia brasileira por satélite [online]*. Available from: <http://www.obt.inpe.br/prodes/index.php> [Accessed 13 Jun 2017].
- INPE (2016). *Programa queimadas: Monitoramento por satélites [online]*. São José dos Campos - SP, Brazil: Instituto Nacional de Pesquisas Espaciais. Available from: <https://queimadas.dgi.inpe.br/queimadas/> [Accessed 12 Jun 2017].
- LEI No 12.651 (2012). *Novo Código Florestal Brasileiro. Publicado no DOU de 28 de maio de 2012*. Capítulo IX Art 38. Estabelece normas para a proibição do uso do fogo e para o controle de incêndios. Brazil.
- Lovejoy, T. E., & Nobre, C. (2018). Amazon tipping point. *Science Advances*, *4*, eaat2340. <https://doi.org/10.1126/sciadv.aat2340>
- Melo, J. E., Coradin, V. R., & Mendes, J. C. (1990). Classes de densidade para madeiras da Amazônia brasileira. In: *Congresso florestal Brasileiro* (pp. 695–705). Campos do Jordão, Brazil: Anais do Congresso Florestal Brasileiro de Campos do Jordão.
- Mendonça, M. J. C., Vera Díaz, M. del. C., Nepstad, D., Seroa da Motta, R., Alencar, A., Gomes, J. C., & Ortiz, R. A. (2004). The economic cost of the use of fire in the Amazon. *Ecological Economics*, *49*, 89–105. <https://doi.org/10.1016/j.ecolecon.2003.11.011>
- Menton, M. C. (2003). Effects of logging on non-timber forest product extraction in the Brazilian Amazon: Community perceptions of change. *International Forestry Review*, *5*, 97–105. <https://doi.org/10.1505/IFOR.5.2.97.17419>
- Merry, F., Soares-Filho, B., Nepstad, D., Amacher, G., & Rodrigues, H. (2009). Balancing conservation and economic sustainability: The future of the Amazon timber industry. *Environmental Management*, *44*, 395–407. <https://doi.org/10.1007/s00267-009-9337-1>
- Mitchard, E. T. A., Feldpausch, T. R., Brienen, R. J. W., Lopez-Gonzalez, G., Monteagudo, A., ... Phillips, O. L. (2014). Markedly divergent estimates of Amazon forest carbon density from ground plots and satellites. *Global Ecology and Biogeography*, *23*, 935–946. <https://doi.org/10.1111/geb.12168>
- Morello, T. F., Parry, L., Markusson, N., & Barlow, J. (2017). Policy instruments to control Amazon fires: A simulation approach. *Ecological Economics*, *138*, 199–222. <https://doi.org/10.1016/j.ecolecon.2017.03.043>
- Morton, D. C., Le Page, Y., DeFries, R., Collatz, G. J., & Hurtt, G. C. (2013). Understorey fire frequency and the fate of burned forests in southern Amazonia. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *368*, 20120163. <https://doi.org/10.1098/rstb.2012.0163>
- Moutinho, P., Guerra, R., & Azevedo-Ramos, C. (2016). Achieving zero deforestation in the Brazilian Amazon: What is missing? *Elementa: Science of the Anthropocene*, *4*, 125.
- Myers, R. L. (2006). *Convivendo com o Fogo-Manutenção dos ecossistemas e subsistência com o manejo integrado do fogo*. Tallahassee, FL: The Nature Conservancy-Iniciativa Global para o Manejo do Fogo.
- Nahuz, A. R., Miranda, M., Ielo, P., Pigozzo, R., & Yojo, T. (2013). *Catálogo de madeiras brasileiras para a construção civil*. São Paulo, Brazil: IPT.
- Nepstad, D. C., & Alencar, A. A. (1999). *Flames in the rain forest: Origins, impacts and alternatives to Amazonian fires*. Brasília, DF, Brazil: World Bank.
- Nepstad, D., Carvalho, G., Cristina Barros, A., Alencar, A., Paulo Capobianco, J., Bishop, J., Moutinho, P., Lefebvre, P., Lopes Silva, U., & Prins, E. (2001). Road paving, fire regime feedbacks, and the future of Amazon forests. *Forest Ecology and Management*, *154*, 395–407. [https://doi.org/10.1016/S0378-1127\(01\)00511-4](https://doi.org/10.1016/S0378-1127(01)00511-4)
- Nepstad, D. C., Moreira, A., & Alencar, A. (1999). *A floresta em chamas: Origens, impactos e prevenção de fogo na Amazônia*. Brasília, DF, Brazil: Programa Piloto para a Proteção das Florestas Tropicais do Brasil.
- Pereira, D., Santos, D., Vedoveto, M., Guimarães, J., & Veríssimo, A. (2010). *Fatos florestais da Amazônia*. Belém, Brazil: Imazon.
- Phillips, O. L., & Brienen, R. J. W. (2017). Carbon uptake by mature Amazon forests has mitigated Amazon nations' carbon emissions. *Carbon Balance and Management*, *12*, 1. <https://doi.org/10.1186/s13021-016-0069-2>
- Rajão, R., & Vurdubakis, T. (2013). On the pragmatics of inscription: Detecting deforestation in the Brazilian Amazon. *Theory, Culture & Society*, *30*, 151–177. <https://doi.org/10.1177/0263276413486203>
- Silvestrini, R. A., Soares-Filho, B. S., Nepstad, D., Coe, M., Rodrigues, H., & Assuno, R. (2011). Simulating fire regimes in the Amazon in response to climate change and deforestation. *Ecological Applications*, *21*, 1573–1590. <https://doi.org/10.1890/10-0827.1>
- Soares-Filho, B. S., de Oliveira, A. S., Rajão, R. G., Oliveira, U., de Santos, L. R. S., Assunção, A. C., Rodrigues, H. O., de Oliveira, A. R., Merry, F., Costa, W. L., Brando, P., Ribeiro, S. M. C., & Teixeira, I. L. da S. (2017). *Economic valuation of changes in the Amazon forest area: Economic losses by fires to sustainable timber production*. Belo Horizonte, Brazil: UFMG (Centro de Sensoriamento Remoto).
- Soares-Filho, B. S., Lima, L. S., Oliveira, A. S., Rodrigues, H. O., Ribeiro, S. M. C., Oliveira, U., Costa, W. L. S., Oliveira, A. R., Merry, F., Teixeira, I. L., & Figueira, D. da S. (2017). *Economic valuation of changes in the Amazon forest area: Value maps for timber*. Belo Horizonte, Brazil: UFMG (Centro de Sensoriamento Remoto).
- Soares-Filho, B., Silvestrini, R., Nepstad, D., Brando, P., Rodrigues, H., Alencar, A., Coe, M., Locks, C., Lima, L., Hissa, L., & Stickler, C. (2012). Forest fragmentation, climate change and understory fire regimes on the Amazonian landscapes of the Xingu headwaters. *Landscape Ecology*, *27*, 585–598. <https://doi.org/10.1007/s10980-012-9723-6>
- Strand, J. (2017). Modeling the marginal value of rainforest losses: A dynamic value function approach. *Ecological Economics*, *131*, 322–329. <https://doi.org/10.1016/j.ecolecon.2016.09.019>

- Tacconi, L., Moore, P. F., & Kaimowitz, D. (2006). Fires in tropical forests – what is really the problem? lessons from Indonesia. *Mitigation and Adaptation Strategies for Global Change*, 12, 55–66. <https://doi.org/10.1007/s11027-006-9040-y>
- ter Steege, H., Pitman, N. C. A., Sabatier, D., & Baraloto, C., ..., Silman, M. R. (2013). Hyperdominance in the Amazonian tree flora. *Science*, 342, 1243092 <https://doi.org/10.1126/science.1243092>
- Vilhena, L. (2016). *Paragominas lança 'Projeto Chama Verde'* [online]. Prefeitura de Paragominas. Available from: http://paragominas.pa.gov.br/noticias/meio_ambiente/87-paragominas_lanca_projeto_chama_verde [Accessed 20 Jun 2017].
- Zybach, B., Dubrasich, M., Brenner, G., & Marker, J. (2009). *US wildfire cost-plus-loss economics project: The 'One-Pager' checklist* (p. 20). Tucson, AZ: Wildland Fire Lessons Learned Center.

SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article.

- Table S1.** Input variables used in ignition and spread components.
- Table S2.** Climatic variables used in the CARLUC model.
- Table S3.** Principal equations of CARLUC.
- Table S4.** Parameters of the SimMadeira model.
- Table S5.** Input variables used in SimMadeira model.
- Table S6.** Transportation costs used in SimMadeira model.
- Table S7.** Tree genera/species and respective mean prices for roundwood.
- Table S8.** Bioclimatic variables.
- Table S9.** List of interviewees.
- Table S10.** Municipalities with the greatest economic loss by fire.
- Table S11.** Municipalities participating in the “Green Municipalities Project”.
- Table S12.** Municipalities most attended by the fire brigade of the Aliança da Terra.
- Figure S1.** Calibration of the ignition component.
- Figure S2.** Validation of the ignition component.
- Figure S3.** Validation of simulated forest fire scars.
- Figure S4.** Size distribution of simulated and observed forest fire scars.
- Figure S5.** Calibration of CARLUC.
- Figure S6.** Spatial distribution of fire intensities 2005 (a) and 2010 (b).
- Figure S7.** Simulated fires scars and deforestation for the Brazilian Amazon.
- Figure S8.** Species distribution modelling.
- Figure S9.** Steps in SimMadeira to calculate timber volume and prices.
- Figure S10.** Number of tree genera of softwood (a) and hardwood (b).
- Figure S11.** Share of commercial timber genus in relation to the total tree population.
- Figure S12.** Commercial volumes for hard (a) and softwoods (b).
- Figure S13.** Commercial values for hard (a) and softwoods (b).
- Figure S14.** Potential equivalent annual annuity (EAA) of sustainable logging rents.
- Figure S15.** Total net revenue per year.
- Figure S16.** Timber production per year.

How to cite this article: de Oliveira AS, Rajão RG, Soares Filho BS, et al. Economic losses to sustainable timber production by fire in the Brazilian Amazon. *Geogr J.* 2018;00:1–13. <https://doi.org/10.1111/geoj.12276>