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To cite this article: Divino V Silvério *et al* 2022 *Environ. Res. Lett.* **17** 045012

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ENVIRONMENTAL RESEARCH
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OPEN ACCESS

RECEIVED
1 August 2021REVISED
18 January 2022ACCEPTED FOR PUBLICATION
21 February 2022PUBLISHED
21 March 2022

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Intensification of fire regimes and forest loss in the Território
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E-mail: dvsilverio@gmail.com**Keywords:** Amazon, deforestation, forest cover, fire, droughts, Indigenous people, seasonally floodedSupplementary material for this article is available [online](#)**Abstract**

The contemporary fire regime of southern Amazonian forests has been dominated by interactions between droughts and sources of fire ignition associated with deforestation and slash-and-burn agriculture. Until recently, wildfires have been concentrated mostly on private properties, with protected areas functioning as large-scale firebreaks along the Amazon's agricultural frontier. However, as the climate changes, protected forests have become increasingly flammable. Here, we have quantified forest degradation in the Território Indígena do Xingu (TIX), an iconic area of 2.8 million hectares where over 6000 people from 16 different ethnic Indigenous groups live across 100 villages. Our main hypothesis was that forest degradation, defined here as areas with lower canopy cover, inside the TIX is increasing due to pervasive sources of fire ignition, more frequent extreme drought events, and changing slash-and-burn agricultural practices. Between 2001 and 2020, nearly 189 000 hectares (~7%) of the TIX became degraded by recurrent drought and fire events that were the main factors driving forest degradation, particularly in seasonally flooded forests. After three fire events, the probability of forest loss was higher in seasonally flooded areas (63%) compared to upland areas (41%). Given the same fire frequency, areas that have not suffered with extreme droughts showed a 24% lower probability of forest loss compared to areas that experienced three drought events. Distance from villages and human density also had a marked effect on forest cover loss, which was generally higher in areas close to the largest villages. In one of the most culturally diverse Indigenous lands of the Amazon, in a landscape highly threatened by deforestation, our findings demonstrate that climate change may have already exceeded the conditions to which the system has adapted.

1. Introduction

For much of its history, forest fires in the Amazon basin occurred at intervals of 100 years (Sanford *et al* 1985, Meggers 1994, Bush *et al* 2007). Records from pre-Columbian times and for much of the

twentieth century suggest that forest canopies have functioned as ecological firebreaks by creating a moist understory microclimate (Sanford *et al* 1985, Piperno 1997, Nepstad *et al* 2006, Bush *et al* 2007). Although pre-Columbian Indigenous populations used fire to manage their lands (Heckenberger *et al* 2003,

Levis *et al* 2018), they also employed advanced techniques to prevent them from escaping into primary forests (Heckenberger *et al* 2003). Such ancient knowledge likely prevented widespread forest fires even during dry warm years, when Amazon forests tend to be more flammable. In recent decades, however, global changes have altered several key drivers of forest flammability, particularly in southern Amazonia (Alencar *et al* 2004, Brando *et al* 2014). The resulting intensification of fire regimes in the region may be pushing forests beyond a flammability tipping point, allowing wildfires to spread over vast areas (Pueyo *et al* 2010, Brando *et al* 2020a) with negative consequences for the region's local populations.

As climate changes in the Amazon region, forests have become increasingly susceptible to fire—including the protected areas (Indigenous lands and conservation units) that have long worked as large-scale firebreaks. In the past 20 years, extreme drought events have triggered major forest fires in the Amazon region even as deforestation rates have declined (Brando *et al* 2014, Aragão *et al* 2018), probably because human sources of fire ignition are so abundant in the landscape. There is growing evidence that forest degradation—areas with lower canopy cover, due in part to fire and drought stress—has now surpassed deforestation throughout much of the Amazon biome, both in terms of area and carbon emissions (Baccini *et al* 2017, Matricardi *et al* 2020, Kruid *et al* 2021). An outstanding question, though, is whether widespread forest fires will occur at the same rate inside Indigenous lands and conservation units, given the projected increases in regional droughts (Duffy *et al* 2015) and associated decreases in the forest's capacity to maintain cool, moist understory microclimates (Brando *et al* 2020b).

Conservation units and Indigenous lands have proven effective in preventing deforestation and forest degradation historically, but rapid climate changes and more frequent anthropogenic disturbances now pose serious threats to these protected forests (Walker *et al* 2020). Such novel disturbance regimes can cause major shifts in forest communities, potentially pushing the ecosystem towards another stable state (Hirota *et al* 2011, Scheffer *et al* 2015, Trumbore *et al* 2015, Flores *et al* 2017). As a result, forests may lose their capacity to store biomass and other crucial ecosystem services, such as stabilizing local and regional climate, or providing food and medicine for Indigenous communities living in the region (Levis *et al* 2017).

If such near-term shifts in ecosystem stable states were to occur in Amazonia, the southeastern region would likely experience the first, most drastic changes in vegetation structure, functioning, and diversity. The climate of this region is already relatively drier and hotter than other parts of Amazonia. In addition, large-scale deforestation and global climate changes

have already triggered a lengthening of the dry season in the region (Fu *et al* 2013). Furthermore, several climate models project an even drier and hotter climate for the region in the coming decades (Cox *et al* 2000, Duffy *et al* 2015). Such climatic changes increase the likelihood of widespread forest fires that, if left unchecked, could cause abrupt changes in vegetation structure (Brando *et al* 2014, 2020a). Floodplain forests are particularly vulnerable to such rapid changes in fire regimes, since common understory fires can kill entire tree stands from the root, with limited chances for recovery (Flores *et al* 2017). Given the intensification of fire regimes in southeastern Amazonia, it is necessary to better understand the extent of forest cover changes and why these changes have occurred.

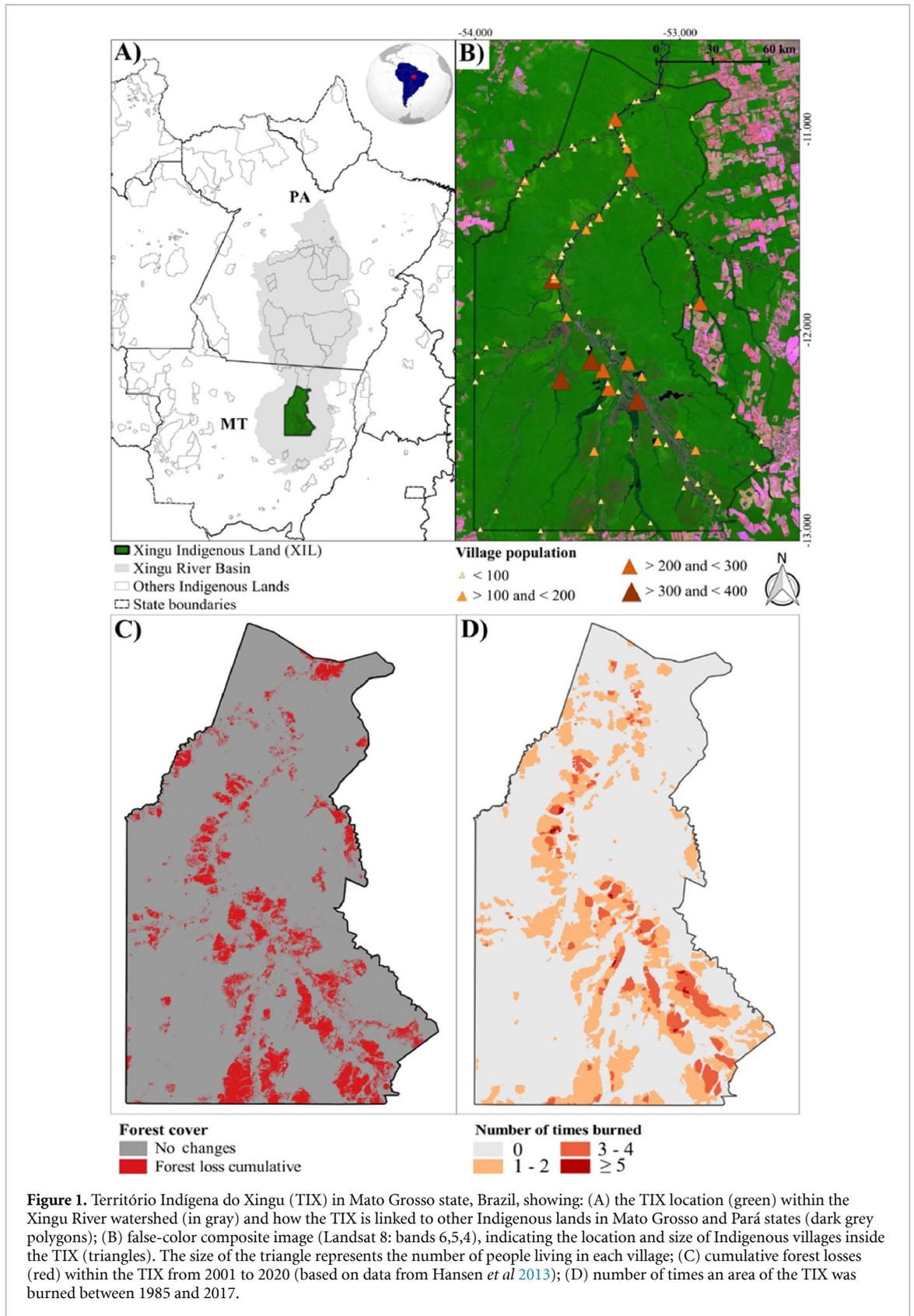
Forest degradation associated with global changes could have serious consequences for Indigenous communities. In Brazil, for instance, drought-fire interactions now threaten the Xingu Indigenous Territory (TIX, Portuguese acronym)—an iconic 2.8 Mha reserve in the southeastern Amazon, that is home to over 6000 Indigenous people from 16 distinct ethnic groups. The TIX has experienced rapid climatic changes due to high deforestation rates outside the park since the 1970s (Silvério *et al* 2015). Since the 2000s, it has also contended with more frequent and intense droughts, and widespread forest fires, marking a shift from previous decades. Although climate models predict an average increase in dry season length and a consequent decrease in average precipitation in the TIX, we still know relatively little about how interactions between droughts, fires, and land management by Indigenous populations will affect the future of these forested protected areas.

Here, we evaluated forest cover changes in the TIX from 2001 through 2020, and whether fire occurrence and extreme drought events played a role in the observed changes. Our two hypotheses were that: (a) forest degradation inside the TIX is increasing due to increased frequency of forest fires, drought events, and slash-and-burn agriculture as populations grow; and (b) forest degradation is higher in seasonally flooded forests than in upland forests, independent of ignition sources.

2. Material and methods

2.1. Study area

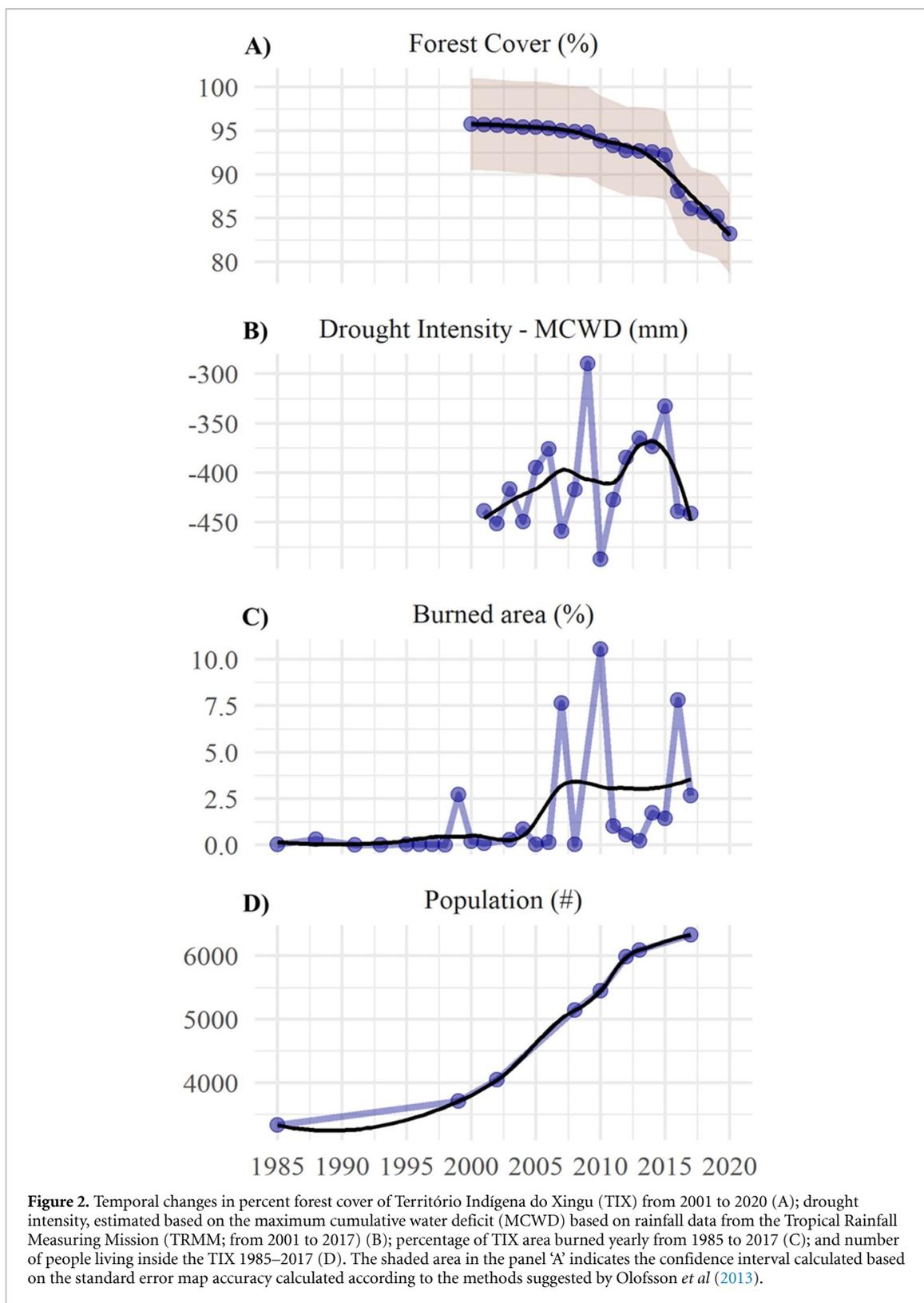
The study area was the Território Indígena do Xingu (TIX), located in northern Mato Grosso state, Brazil. The TIX comprises the Xingu Indigenous Park (2.6 Mha) and three other Indigenous Lands (TI): Batovi (5159 ha), Pequizal do Naruvôtu (27 980 ha) and Wawi (150 328 ha). Together, these areas form a mosaic of 2.83 million hectares (figure 1), home to 16 ethnic groups representing seven distinct language families (ISA 2011). The total population of



the TIX was 3331 people in 1985 (UNIFESP (Universidade Federal de São Paulo) 1983) and increased to 6326 people by 2017 (SIASI/SESAI 2017) (figure 2).

The soil of the region is mainly typical dystrophic red–yellow latosol and typical dystrophic floss neossol (Santos *et al* 2018). The Xingu River basin covers

approximately 520 000 km² (figure 1(A)), with an average flow rate of between 2582 and 9700 m³ s⁻¹ (Latrubesse *et al* 2005). The average air temperature is 26 °C in summer and 28 °C in winter, with annual precipitation varying between 1485 and 2547 mm.



2.2. Datasets and analysis

To quantify changes in forest cover within the TIX and assess the main factors driving these changes, we integrated ground data (Indigenous village location, population density, ground data points for different land cover types) with remote sensing time series (e.g. data on forest cover, burned area, drought intensity, land cover). Using these variables, we develop a

regression model to quantify the relative importance of different predictors of forest cover change. Additional details on the datasets and analytical approach used to test our hypotheses are provided below.

2.2.1. Changes in forest cover

We calculated forest loss based on the annual forest change data produced by Hansen *et al* (2013) and

available from the Global Forest Watch (GFW) platform. We used version 1.8, which includes annual forest cover from 2001 through 2020. The GFW provides the canopy cover for the year 2000 (the reference map) at 30 m resolution for all areas with tree canopies >5 m, and annual forest loss between 2001 and 2020 (defined as areas stand-replacement disturbance or nearly complete removal of tree canopy cover at the Landsat pixel scale). For this analysis, we considered forest areas as those with percent tree cover $\geq 20\%$ in 2000. We defined forest degradation as the long-term reduction in canopy cover (to below 20%) associated both with human activities (Indigenous slash and burned systems), forest fires, and drought stress over the study period (2001–2020). Although GFW is well documented and has been widely used to understand forest dynamics at various scales, inconsistencies exist due to differences in Landsat sensor technology and algorithm adjustments (GFW 2021). We assessed the accuracy of the GFW product for the TIX using high-resolution images from Google Earth that host images with resolution >2 m. We reported the confusion matrix (table S1 available online at stacks.iop.org/ERL/17/045012/mmedia) in terms of estimated area proportions, and the estimated area for each class associated with confidence interval following the methods recommended by Olofsson *et al* (2014). Additional details about this dataset and the accuracy of this product for the TIX region are provided in the supplementary information (figures S1–S3; tables S1 and S2).

2.2.2. Type of forest

To differentiate seasonally flooded forests from *terra firme* forests (upland forests that grow on nonflooded soils), we relied on the map of wetland extent in the Amazon Basin produced by Hess *et al* (2015). The data was generated based on synthetic aperture radar images from the Japanese Earth Resource Satellite-1 at 90 m spatial resolution.

2.2.3. Drought frequency

We calculated the number of drought events for the period between 2001 and 2017 based on the maximum climatic water deficit (MCWD). To calculate MCWD, we used precipitation data from the Tropical Rainfall Measuring Mission satellite, assuming that humid tropical forests (under normal conditions) have a monthly evapotranspiration of 100 millimeters. If the monthly rainfall is less than 100 mm, the water deficit accumulates to the next month (Aragão *et al* 2007). To calculate the number of drought years, we used the lower 0.1 quantile, considering areas where MCWD exceeded 10% (−471.16 mm) in a year as having experienced a drought. Based on this analysis, we created a single raster image where values represent the total number of drought events

occurring in any given pixel over the course of the time series (2001–2017).

2.2.4. Fire frequency

We calculated the fire frequency (i.e. the number of times a given area burned between 1985 and 2017) based on two different approaches to burn scar mapping: (a) we produced a time series of annual fire scar maps, created manually based on ground information and Landsat images. To map the fire scars, we manually drew the perimeter of the fires scar (hand digitizing) by visual interpretation of vegetation changes based on red-green-blue (RGB) composites of Landsat images. We used images available for the dry season (June–September), which corresponds to the fire season and the period when most cloud-free images are available; and (b) we used a published burned area map generated by Morton *et al* (2013) for the region, which estimated the extent of understory forest fires in the dry season (June–August) from 2001 to 2012. To assemble a complete time series, we merged the two maps and created a final raster representing a count of the number of times each pixel burned during the study period.

2.2.5. Distance from villages and population density

We obtained the location of each Indigenous village in the TIX from Fundação Nacional do Índio and verified their location by comparing with RGB images from the Sentinel-2 satellite. We developed a map of population density by combining the location of villages for the TIX with data on the number of people living in each village in 2017, obtained from the Special Indigenous Health District (DSEI Xingu 2017), Special Secretariat for Indigenous Health (SIASI/SESAI 2017) and the Army Geographic Database (BDGEX 2019). These institutions are linked to the Ministry of Health, which is responsible for Indigenous health in Brazil (SIASI/SESAI 2017). Based on these two data layers, we delineated 2 km buffer zones around each village (ranging from 2 to 40 km) and calculated the population density within each zone.

2.2.6. Main rivers

We identified the main rivers through information from the Army Geographic Database (BDGEX 2019). We extracted the theme ‘Trecho Massa D’agua’ from vector topographic charts at 1:250 000 scale. Distances from the main rivers were calculated based on buffers (ranging from 2 to 50 km) from the main river channel to simulate the main means of transportation used by Indigenous peoples in the TIX. We converted these vector layers into rasters and applied the ‘minimum value’ function to integrate all values in a single raster image. For these analyses we use the packages ‘raster’ (Hijmans 2019), ‘sp’ (Bivand *et al* 2013), and ‘sf’ (Pebesma 2018) in the R program v.3.6.0 (R Core Team 2019).

2.2.7. Land use/land cover map

We created a 30 m resolution land cover map for the TIX based on supervised classification of Landsat image mosaics for July 2001 (Landsat-5) and July 2017 (Landsat-8). To create this map, we used GPS data collected in the field (111 points for 2001 and 120 points for 2017) for six different vegetation classes for training the supervised classification. The vegetation classes were defined as follows: (a) *bare soil*—soil surface with no vegetation as, for example, the sandy areas close to rivers and roads; (b) *grassland*—open vegetation dominated by grasses and sparse trees; this includes ‘sapezal’ areas, which are areas of arrested forest succession dominated by the native grass species *Imperata brasiliensis* Trin. (Gramineae) and managed by Indigenous people to produce thatch. Increased fire frequency favors the expansion of areas dominated by sapezal, which tends to increase ecosystem flammability, thus representing a positive feedback mechanism (Schwartzman *et al* 2013); (c) *water*—surface covered by water; (d) *young secondary forest (capoeira)*—secondary vegetation in the initial stages of succession; (e) *old secondary forest (capoeira)*—secondary vegetation in more advanced stages of succession; (f) *mature forest*—forest with no evidence of recent disturbances. We used 228 independent points to assess the map accuracy (117 for 2001 and, 111 points for 2017) that were collected based on expert identification of images from Landsat 5 (for 2001) and Sentinel 2 (2017), and the known locations of each indigenous village (opportunistic points). Additional details on our classification, analytical approach, and validation are provided in the *supplemental information* (Land cover map section, table S2 and figure S3).

2.2.8. Data analysis

To test our hypotheses, we used a generalized linear model with a binomial distribution. We considered forest loss (between 2001 and 2020) as our response variable as a function of six predictors: (a) type of forest (flooded forests vs. upland forests); (b) fire frequency (0–7, between 1985 and 2017); (c) population density (0.78–42.8 people km⁻² as of 2017); (d) distance from villages (2–50 km); (e) distance from the main rivers (2–50 km); and (f) number of extreme drought events (0–6, between 2001 and 2017). We had previously calculated autocorrelation between the predictor variables and found no significant limitation (table S3). All data were standardized to a 1 km scale, then we sampled 20% of pixels randomly to run the model, and help to satisfy desirable design criteria for spatial statistical modeling, including accounting for spatial autocorrelation (explained in detail in the *supplementary information*).

First, we developed a global model with all variables and possible interactions. We then performed a model selection evaluating the contribution of each variable to the performance of the model (using

the *dredge* function of the *MuMin* package in the R program; table S4). We assumed that the ‘best model’ was the one with the lowest value of the Akaike Information Criterion (Bozdogan 1987). Finally, we interpreted the parameters of the best model and used predictive graphs to examine the relative importance of each predictor variable in determining the likelihood of forest cover loss (table S5). All statistical analyses were performed using the R program v.3.6.0 (R Core Team 2019).

3. Results

The forest cover in the TIX dropped 7% from 2001 (25 930.42 ± 1,179.09 km²; ±CI) to 2020 (24 044.34 ± 1,104.05 km²) (figures 1, 2 and S2). The annual rate of forest loss between 2001 and 2015 averaged 0.23% (6512 ha yr⁻¹), but that rate increased to 4.13% (116 941 ha) and 1.93% (54 648 ha) in 2016 and 2017, respectively. Additionally, approximately 16% (453 041 ha) of the TIX area burned at least once between 1984 and 2017, and 12.5% (353 939 ha) burned more than once. Between 1984 and 2006, the average annual burned area was 0.3% (8484 ha yr⁻¹) of the TIX but it increased to 3.4% (96 271 ha yr⁻¹) between 2007 and 2017. The peak annual burned area occurred in 2010, when more than 10% of the TIX’s forests burned (figures 1 and 2).

Our analysis suggests that several variables influenced the rate of forest cover loss inside the TIX. All six variables (forest type, fire frequency, population density, distance from rivers, and droughts frequency) were important predictors of forest loss from 2001 to 2020 in our final statistical model (table S5). Fire occurrence was the most important predictor of forest cover loss, but interactions between fire, droughts, and forest type were also significant (table S5). Our full model explained 22% of the observed decline in forest cover (pseudo $r^2 = 0.22$; table S5).

The recurrence of forest fires during drought (vs. non-drought) years, and in flooded (vs. upland) forests, strongly influenced the likelihood of forest loss (figure 3). Forests that did not experience extreme droughts, even if they were burned multiple times, had much higher probability of maintaining higher canopy cover compared with forests that burned multiple times during drought years (figure 3(B)). In flooded forests, the interaction between fire recurrence and drought events more than doubled the likelihood of forest losses (figure 3(A)). In the case of upland forests, the incidence of four or more forest fires increased the probability of forest loss to >57%. In the case of flooded forests, just three fire events raised the probability of forest loss to >63% (figure 3(A)). These results indicate that the recurrence of forest fires is a key process driving forest loss in the region and confirms that flooded forests are particularly vulnerable to fire-drought interactions (figure 3).

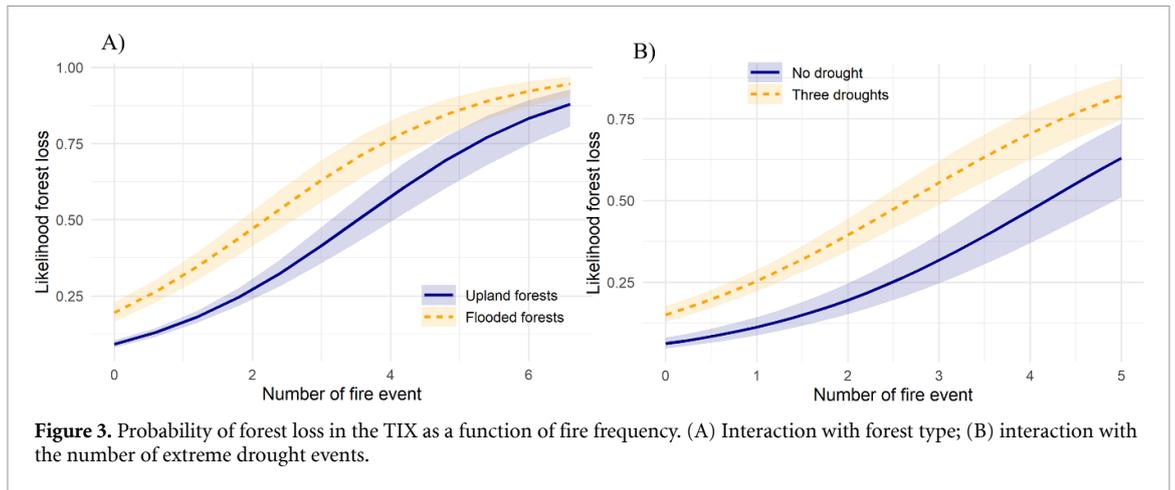


Figure 3. Probability of forest loss in the TIX as a function of fire frequency. (A) Interaction with forest type; (B) interaction with the number of extreme drought events.

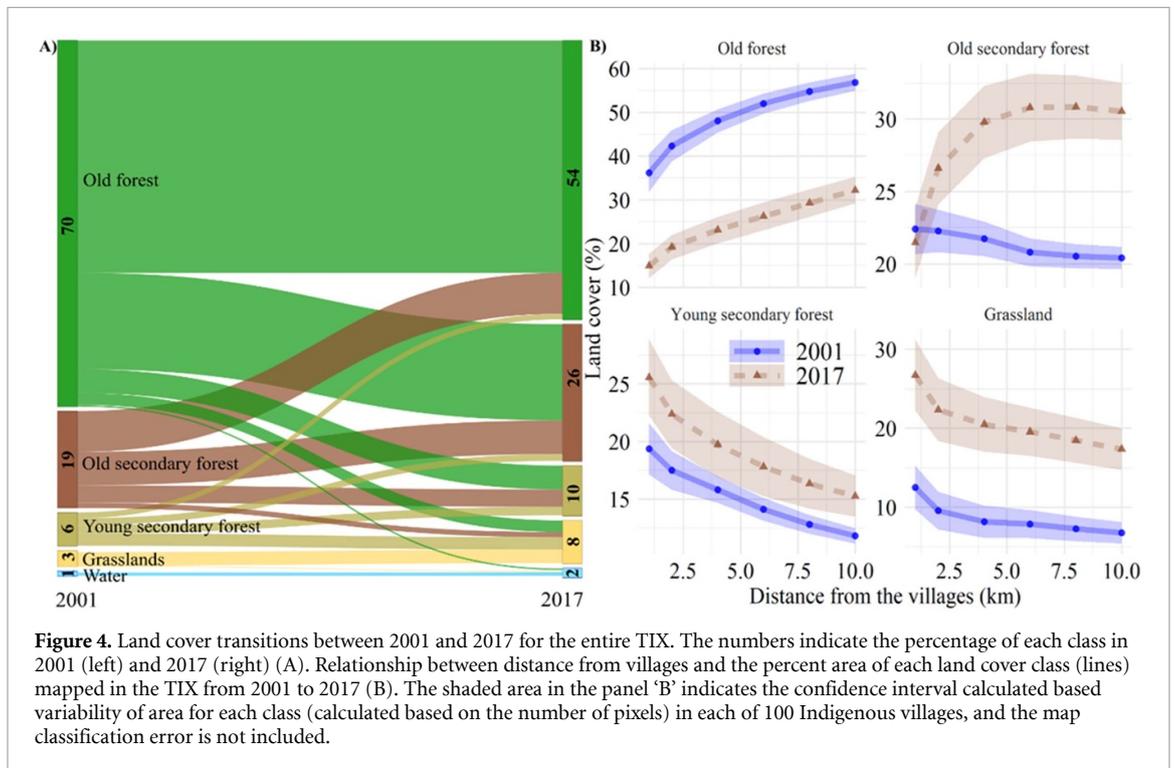


Figure 4. Land cover transitions between 2001 and 2017 for the entire TIX. The numbers indicate the percentage of each class in 2001 (left) and 2017 (right) (A). Relationship between distance from villages and the percent area of each land cover class (lines) mapped in the TIX from 2001 to 2017 (B). The shaded area in the panel 'B' indicates the confidence interval calculated based on variability of area for each class (calculated based on the number of pixels) in each of 100 Indigenous villages, and the map classification error is not included.

Our statistical model indicates that forest cover losses are generally higher for areas with higher population density, particularly in areas that are periodically flooded (tables S5 and S6). Specifically, we observed a higher probability of fire-related forest loss where population density was higher, especially in the case of flooded forests (figure S5). In areas burned once and with low population densities (e.g. ~ 5 people km^{-2}), the probability of fire-induced forest loss was 20%, but this probability increased to $\sim 90\%$ when population density increased to 30 people km^{-2} (figure S5). For flooded forests, the probability of forest loss was 50% at low population densities and nearly 100% at high population densities (figure S5).

While old forest cover near villages was relatively low in 2001, it declined even further by 2020 (figures 4 and S6). For instance, the forest cover in areas close to

the villages (1 km away) decreased from 82% in 2001 to 54% in 2020. Forest cover losses during this interval decreased as distance from the villages increased. For example, areas up to 10 km away from villages showed a reduction of only about 21% in forest canopy between 2001 and 2020 (figure S6).

Our land cover classification for the entire TIX showed that the area covered by mature forests (e.g. forests with no evidence of recent disturbances) decreased from 70% in 2001 to 54% in 2017 (figure 4(A)). This reduction in mature forest cover was accompanied by an increase in old secondary forest (from 19% in 2001 to 26% in 2017) and in grasslands (from 3% in 2001 to 8% in 2017; figure 4(A)). Young secondary forests also increased from 6% in 2001 to 10% in 2017 (figure 4(A)). These transitions in land cover type were strongly influenced by distance from the villages.

We observed more transitions from primary forests into old secondary forests and grassland near villages (figure 4(B)). As distance from the villages increased, likewise, primary forests became more dominant and younger secondary forests and less dominant. Finally, the grassland class increased between 2001 and 2017, but probability of finding it decreased slightly with distance from the villages (figure 4(B)). This shows that areas with greater influence on Indigenous activities may not be recovering fast enough to become mature forests.

4. Discussion

Our study investigated recent patterns and processes associated with forest degradation in the TIX during the 21st century. Our results point to an acceleration of forest canopy loss (our proxy for forest degradation) in the TIX between 2001 and 2020. We show that the main drivers of forest loss were associated with drought–fire interactions and distance from larger villages. Our findings also indicated that the probability of forest loss was substantially higher in flooded forests compared with upland ones. These synergistic effects of multiple natural and anthropogenic factors are already driving large-scale forest degradation across one of the world’s largest Indigenous lands. Avoiding further large-scale forest mortality as the climate changes will require decisive new approaches to fire management in the region. Indigenous populations of the TIX have a profound knowledge of fire management, including practices to prevent fires from escaping into primary forests (Heckenberger *et al* 2003). Some of these practices could help to prevent widespread forest fires, including the expansion of the fire brigades in the region.

Projections of future forest cover in the TIX rely mostly on process-based models that lack a realistic representation of forest fires or drought–fire interactions. Nevertheless, fire recurrence during droughts was the primary determinant of forest loss in the TIX in our study. After three consecutive fires, for instance, over half of the forests succumbed to a degraded state that could persist for years or decades. One hypothesis is that reduced forest resilience under a changing climate facilitates the establishment of more open vegetation following recurrent fires, with the predominance of grasses (Trumbore *et al* 2015, Nobre *et al* 2016). Given that grasses are highly flammable, new fires could prevent forest regrowth, as demonstrated by controlled fire experiments in which grass invasion triggered high intensity forest fires even during non-drought years (Silvério *et al* 2013, Brando *et al* 2014). Our study suggests that a similar process is apparently underway in the TIX. Over 350 000 ha of forests burned more than once during the study period, representing over 12% of the TIX’s total forested area. Grass invasion into these burnt forests may create a positive feedback in which fires facilitate

the permanent establishment of flammable grasses, especially if the climate changes—associated both with land-use change and greenhouse gas accumulation in the atmosphere—continue to reduce forest resilience to disturbances (Cochrane and Schulze 1999, Silvério *et al* 2013, Brando *et al* 2020a). Hence, fire suppression efforts and controlled fires may be key strategies for fostering forest recovery and driving an overall reduction in vegetation flammability inside the TIX.

The likelihood of forest loss by fire was higher where extreme droughts had occurred. This result indicates that the interaction with other degradation factors (climate change and biotic factors) is already putting pressure on the forests in the region. The projected intensity and frequency of droughts may thus be proxies for the potential strength and extent of future forest losses. Global models predict a new climate regime for the Amazon with longer dry seasons (Malhi *et al* 2008), higher temperatures, and a greater vapor pressure deficit (McDowell *et al* 2018, Barkhordarian *et al* 2019), suggesting that the forests of the region will experience greater water stress. The result could be widespread tree mortality, especially when water stress is accompanied by intense fires. Large trees, which store the highest amounts of carbon and contribute to climate regulation through evapotranspiration, are also the most vulnerable to these extreme droughts (Rowland *et al* 2015). In burned forests, they also become extremely vulnerable to disturbances such as windstorms and other edge effects associated with a more open canopy (Silvério *et al* 2019). As these trees die and lose foliage during extreme droughts, they add a large amount of fuel to the forest floor, which again favors the occurrence of high intensity forest fires (Nepstad *et al* 2004, Brando *et al* 2014, Balch *et al* 2015). Such interactions between drought events and forest fires can amplify forest degradation, pushing the ecosystem to a ‘grassy’ stable state, maintained by a positive fire feedback (Hirota *et al* 2011, Staver *et al* 2011).

Our study corroborated previous ones documenting the greater vulnerability of seasonally flooded forests compared to upland forests (de Resende *et al* 2014, Flores *et al* 2014). The higher flammability of flooded forests during droughts is attributed to a greater amount of fuel material that can easily spread smoldering ground fires (Flores *et al* 2014, Almeida *et al* 2016), a process that is consistent with our results. Moreover, in flooded forests, tree recruitment tends to be limited by soil impoverishment and the flooding period may restrict plant growth (Flores *et al* 2017, 2020). The new precipitation regime, already evident in the southeastern Xingu Basin, thus has immediate repercussions for the future of flooded forests (due to their lower resilience compared to surrounding upland forests) and places them at greater risk of transitioning to herbaceous vegetation after fires (Flores *et al* 2017).

In our study, human density in the villages contributed to explaining forest losses, as well as reductions in mature forest area closer to villages in the TIX. Two processes may explain these results. First, many Indigenous peoples used to live a semi-nomadic lifestyle centered around small, temporary settlements across the landscape. As these groups have transitioned to living in fixed villages, they have also intensified subsistence agriculture near the villages (Schmidt *et al* 2021). This process relates partly to limited availability of suitable lands for new villages as the TIX population expands, but also to the fact that some of these groups were resettled in the TIX in the early 1970s. Second, as the fire regime intensifies (i.e. more frequent, intense fires) it has increased the time needed for secondary forests to recover after abandonment from slash-and-burn agriculture, as found in a recent study conducted at the Kawaiwete and the Ikpeng villages in the TIX (Schmidt *et al* 2021). Deforestation outside Indigenous lands can also increase fire ignition sources and contribute to regional climate changes (Silverio *et al* 2015), which may overwhelm the inhibitory effect that tropical forests have historically played in reducing sources of ignition and acting as firebreaks in the region (Staver *et al* 2011, Brando *et al* 2014). This regional connectivity underscores the importance of preserving tropical forests both inside and outside conservation units and Indigenous territories.

5. Conclusions

Our results revealed that drought-driven fires already pose a serious threat to forest integrity in the TIX. The flooded forests of the TIX are less resilient and more vulnerable to fire degradation than upland forests, placing them at greater risk of transitioning to herbaceous vegetation. Forest losses were also associated with a higher human density and distance from the villages, since human populations occupy vulnerable areas (flooded forests) and can contribute to increasing ignition sources. If the trends observed here continue, they will exacerbate the negative effects of extreme climatic events on the Xingu region, increasing the intensity and frequency of fires, and transforming the region's forests. The persistence of these forests will ultimately depend on adapting the ancient fire management practices of Indigenous peoples to novel climatic conditions in the region. Reducing ignition sources and the accidental spread of forest fires will be crucial, both from the surrounding landscape (e.g. agricultural lands) and from within Indigenous territories.

Our findings reveal how the TIX is now exposed to unprecedented drought and fire regimes, which have driven cumulative forest cover losses. Moreover, increasing population density inside the TIX implies

that protecting the culturally and biologically diverse ecosystems of the Xingu may require the expansion of the area or existing protections on Indigenous territories to help mitigate forest losses in the region. This highlights the need for holistic and interdisciplinary analyses to understand how the interaction between current climate changes and human activities are modifying tropical forest ecosystems. Such an integrated understanding could contribute to the design of new management strategies to prevent vegetation loss and minimize the negative consequences for Indigenous peoples, who depend on forest resources for their livelihoods.

Data availability statement

The data that support the findings of this study are available upon reasonable request from the authors.

Acknowledgments

We thank all members of the research team at the Instituto de Pesquisa Ambiental da Amazônia (IPAM), the Instituto Socioambiental (ISA), and the Graduate Program of Ecology and Conservation at the Universidade do Estado do Mato Grosso (UNEMAT). This study was supported by grants from the Conselho Nacional de Desenvolvimento Científico e Tecnológico—CNPq/Prevfogo-Ibama (#442710/2018-6) and the Gordon and Betty Moore Foundation (#9957). CNPq provided a research scholarship to M T F (#380219/2021-2) and the Coordenação de Aperfeiçoamento de Pessoal de Nível Superior (CAPES/BRASIL; FINANCE CODE 001) provided scholarships to R S O and H K A.

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