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Changing patterns of fire occurrence in proximity to forest edges, roads and rivers between NW Amazonian countries

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Abstract. Tropical forests in NW Amazonia are highly threatened by the expansion of the agricultural frontier and subsequent deforestation. Fire is used, both directly and indirectly, in Brazilian Amazonia to propagate deforestation and increase forest accessibility. Forest fragmentation, a measure of forest degradation, is also attributed to fire occurrence in the tropics. However, outside the Brazilian Legal Amazonia the role of fire in increasing accessibility and forest fragmentation is less explored. In this study, we compared fire regimes in five countries that share this tropical biome in the most north-westerly part of the Amazon Basin (Venezuela, Colombia, Ecuador, Peru and Brazil). We analysed spatial differences in the timing of peak fire activity and in relation to proximity to roads and rivers using 12 years of MODIS active fire detections. We also distinguished patterns of fire in relation to forest fragmentation by analysing fire distance to the forest edge as a measure of fragmentation for each country. We found significant hemispheric differences in peak fire occurrence with the highest number of fires in the south in 2005 vs. 2007 in the north. Despite this, both hemispheres are equally affected by fire. We also found difference in peak fire occurrence by country. Fire peaked in February in Colombia and Venezuela, whereas it peaked in September in Brazil and Peru, and finally Ecuador presented two fire peaks in January and October. We confirmed the relationship between fires and forest fragmentation for all countries and also found significant differences in the distance between the fire and the forest edge for each country. Fires were associated with roads and rivers in most countries. These results can inform land use planning at the regional, national and subnational scales to minimize the contribution of road expansion and subsequent access to the Amazonian natural resources to fire occurrence and the associated deforestation and carbon emissions.

1 Introduction

Fires in the tropics are a major consequence of the interaction between climate and human activity and are becoming an increasingly important ecological factor affecting forest extent and condition (Bowman et al., 2009; Cochrane, 2009). Fire degrades forests by changing their composition and structure (Barlow and Peres, 2004), altering essential ecological processes and functions such as nutrient or hydrological cycling, or modifying the rates at which these operate (Cochrane, 2003; Marengo et al., 2008b; Morton et al., 2007). Agricultural practices that use cutting and burning as a land management technique or use fire for land clearing or grazing are usually linked to tropical deforestation (Fearnside et al., 2009; Kirby et al., 2006; Lima et al., 2012; Nepstad et al., 2001). Most recently, large-scale industrial agriculture was related to the use of fire (Brando et al., 2013). Increasing demands for agricultural land and forest-related products has enhanced the link of fire to tropical deforestation by enabling conditions related to increased accessibility to forests (Barber et al., 2014; Laurance et al., 2002) and changing climatic patterns (Aragão et al., 2008; Flannigan et al., 2009; Malhi et al., 2008). Fires in the region can be broadly classified as maintenance, deforestation and forest fires with different temporal patterns related to climate conditions, but in some cases they are related to the ignition cause; i.e. maintenance fires in Brazil are lit every 2–4 years (Roy and Kumar, 2017)

Fire occurrence in the tropics has a particular pattern: in Latin America it has been established that north of the equator the fire season is between December and February while in the south it is between May and July (Chuvieco et al., 2008). However, unusual fire events have been occurring more frequently and more intensely in the Amazon Basin and are associated with extreme climatic events such as the El Niño Southern Oscillation (ENSO) (Aragão et al., 2007; Ray et al., 2005) or the warm tropical North Atlantic Oscillation (NAO) (Marengo et al., 2008a; Phillips et al., 2009) as well as to the occurrence of extreme drought years (Asner and Alencar, 2010; Brown et al., 2006; Lewis et al., 2011; Malhi et al., 2009; Marengo et al., 2008b).

In Amazonia, the increasing frequency and intensity of fires has many consequences locally, regionally and globally. Fluctuations in biomass in Amazonia have a significant impact on atmospheric concentrations of CO_2 (Phillips et al., 2009) which contribute to global warming. Fires also cause a reduction in above-ground biomass (Cochrane and Schulze, 1999; Kauffman and Uhl, 1990), primary production (Kinnaird and O'Brien, 1998), biodiversity and disruption of regional water and energy cycles (Salati, 1987; Salati and Vose, 1984). Despite the importance of fire occurrence in Amazonia, there is a lack of knowledge of the significance of both climate and landscape characteristics driving fire patterns, especially for those subregions outside the Legal Amazonia. Rainfall patterns in the Amazon Basin have high heterogeneity (Marengo, 1992; Marengo and Tomasella, 1998). Northwestern Amazonia, in particular, is one of the wettest tropical rainforest regions with ca. 3000 mm of rain per year and has a shorter dry season than south-western, south-eastern or central Amazonia (Malhi and Wright, 2004).

Throughout the tropics, road development increases the susceptibility of forests to deforestation and forest fragmentation by exposing forest edges to increasing levels of disturbances particularly in Amazonia (Barber et al., 2014; Cochrane and Barber, 2009). Fire is frequently used for clearing in fragmented forests and is largely associated with forest edges (Cochrane, 2001; Cochrane and Laurance, 2002). Distance from forest edges influences fire occurrence and intensity (Cochrane, 2003; Armenteras et al., 2013a). The combination of road development, forest deforestation and fragmentation makes tropical forests more vulnerable to fires, especially under expected climate change conditions (Cochrane, 2003). Recent studies showing forest accessibility (from both roads or rivers) as enabling conditions for fires and using fire as a proxy for deforestation are focused mostly in the Legal Amazonia (Adeney et al., 2009) or in the Brazilian tropical moist forest biome, largely ignoring the Amazonian territory under other sovereignties (Kumar et al., 2014). Little is known about factors which influence fire dynamics and patterns in NW Amazonia, or about the links between fires to deforestation and fragmentation in the Ecuador, Peruvian and Venezuelan parts of Amazonia with some data in Colombia (Armenteras and Retana, 2012; Armenteras-Pascual et al., 2011). The shared territory does not have the same land use policies and climate action plans nor the same economic or infrastructure development (i.e. road construction). Because of the high variability of both environmental conditions and human dimensions, there is an imperative need to untangle the regional dynamics across the different countries. In this study, we analysed the dynamics and patterns of fires in the most north-westerly part of the Amazon Basin to highlight regional differences in patterns of fire occurrence in relation to accessibility and forest fragmentation between neighbouring countries. To achieve this, we addressed the following questions. (i) Are temporal patterns of fire occurrence in NW Amazonian tropical forests influenced by its latitudinal position (north/south) and/or the country of occurrence? (ii) Are fire occurrences detected in NW Amazonia influenced by accessibility and do they differ between countries? (iii) Are there differences between countries regarding the effect of forest fragmentation (i.e. edge effect) on fire occurrence?

2 Methods

2.1 Study site

The study area (Fig. 1) corresponded to the north-western (80–65° W, 10° S–6° N) part of Amazonia shared by Colombia, Ecuador, Peru, Venezuela and Brazil. The most northerly and westerly limits were delimited using a biogeographic limit corresponding to the South American tropical and subtropical humid forest biome (UNEP, 2009). The total study area contained approximately 2 140 936 km^2 of land of which 582 612 km^2 are located north of the Equator and 1 558 324 km^2 to the south. The largest area belonged to Brazil (41 % or 885 459 km^2), followed by Peru (26 %, 559 709 km^2) and Colombia (21 % or 451 847 km^2) and finally Venezuela and Ecuador with a 8 % (173 966 km^2) and 3 % respectively (69 955 km^2).

2.2 Data sources and analyses

We used the following data sources:

- A forest/non-forest map for 2010 derived from the 25 m global PALSAR mosaics produced using JAXA data from Advanced Land Observing Satellite (ALOS)/Phased Array type L-band SAR (PALSAR), Japan Aerospace Exploration Agency. Accuracy was over 84.86 % with regional variations (Shimada et al., 2014).
- For fire data we used the remotely sensed active fire detections from MODIS (MCD14DL, from both Aqua and Terra satellites download from January 2003 to January 2015) through FIRMS (Fire Information for Re-

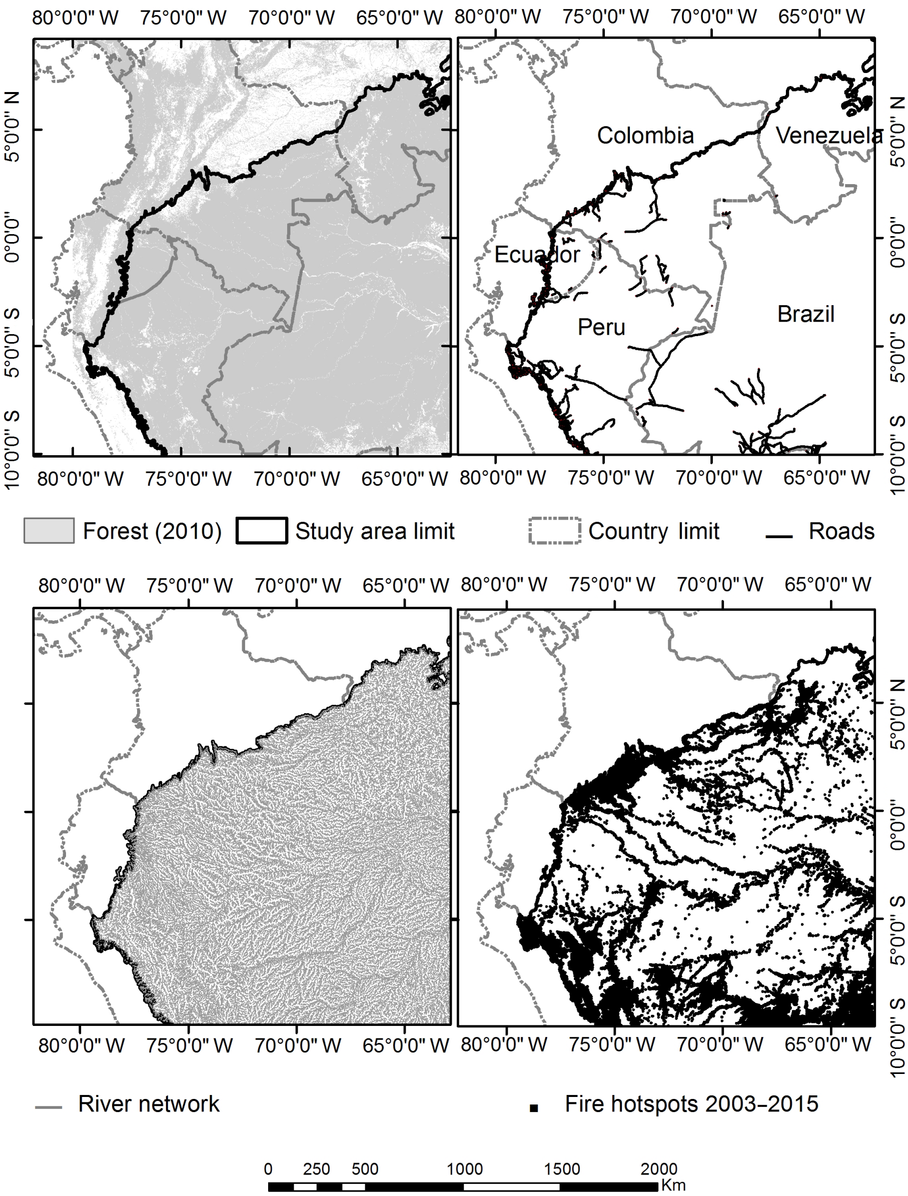

Figure 1. Location of the study area, roads and river network and fire hotspots detected for the period 2003–2015 and a portion of the territory shared amongst countries.

source Management System: Archiving and Distributing MODIS Active Fire Data, Collection 6). We only used data with confidence levels over 30 % (nominal and high confidence fires as applied in (Armenteras et al., 2016; Chen et al., 2013a) We standardized fire occurrence by the area in km^2 of the unit of analysis, so we used fire density (number of occurrences per 1000 km^2) as a fire variable.

– Roads from CIESIN-ITOS (Center for International Earth Science Information Network, 2013). This data set is the best publicly available information up to 2010 for the region. The database was built from public domain road data and has some topology corrected at the national level. The roads are joined topologically at the country borders. The approximate scale is 1 : 250 000. This database shows no roads for Venezuela in the Amazonia; therefore, we removed Venezuela from the analysis of accessibility by roads.

– USGS HydroSheds data were used for the river network, and a consistent hydrological data set from 3 arcsec resolution for the region was publicly available. The data set was built from the high-resolution elevation data obtained during a Space Shuttle flight for NASA's

Shuttle Radar Topography Mission (SRTM) (Lehner et al., 2008).

- To examine the temporal patterns of fire occurrence, we first explored the long-term patterns of fire (January 2003 to January 2015) to describe both intra-annual and interannual variability. We tested for differences in fire occurrence between those occurring north and south of the Equator using a paired two sample *t* test. We also used an ANOVA test to check for differences in fire occurrence between latitudinal position and another ANOVA test to evaluate the differences between countries.

We explored the effect of accessibility on fire occurrence by analysing the proximity of detected fires to rivers and roads. We calculated the distance of each fire hotspot (the point coordinates were the centre of the 1 km pixel) to the closest river and road. We followed the approach presented by Kumar et al. (2014) and built cumulative frequency distributions (CFDs) per country for each set of distances to quantify the annual probability of the occurrence of fire within a given distance of each transportation means. Kumar et al. (2014) built a grid spacing of 0.5 km as reference. To evaluate the observed distributions of distances to road or river networks we followed the procedure layout by Kumar et al. (2014). A regularly spaced 1×1 km square grid was created across the study area, including Colombia, Peru, Ecuador, Venezuela and Brazil. Next, distances from all locations in this grid to the road or river networks were calculated. These distance distributions represented our null models (i.e. the distance distributions that would result if there was no association between fires and those networks) against which observations should be compared. Finally, we applied a non-parametric Kolmogorov–Smirnov test to check for differences between the CFD of the observed distances and that of the corresponding null model on a per-country level. The two-sample Kolmogorov–Smirnov statistics (hereafter, D-statistics) measure the maximum distance between the two CFD curves being compared. The D-statistics index can vary from zero (both CFD curves show a complete overlap; i.e. they match exactly) to one (the two CFD curves do not overlap).

In order to establish a measure of forest fragmentation and to determine whether there were any differences between countries regarding the edge effect on fire occurrence (i.e. fires occurring more frequently near the forest edge), we calculated the distance of fires (i.e. pixel centre) to the forest edge (considered as the pixel edge), taking into account distances both towards the interior and the exterior of the forest). We used the 2010 forest map to establish the forest edge. Similarly to the tests for accessibility to roads and rivers, we built CFD curves for the edge distances and compared them (*k*-sample Anderson–Darling test) both with their respective null models of distances to edges and between countries.

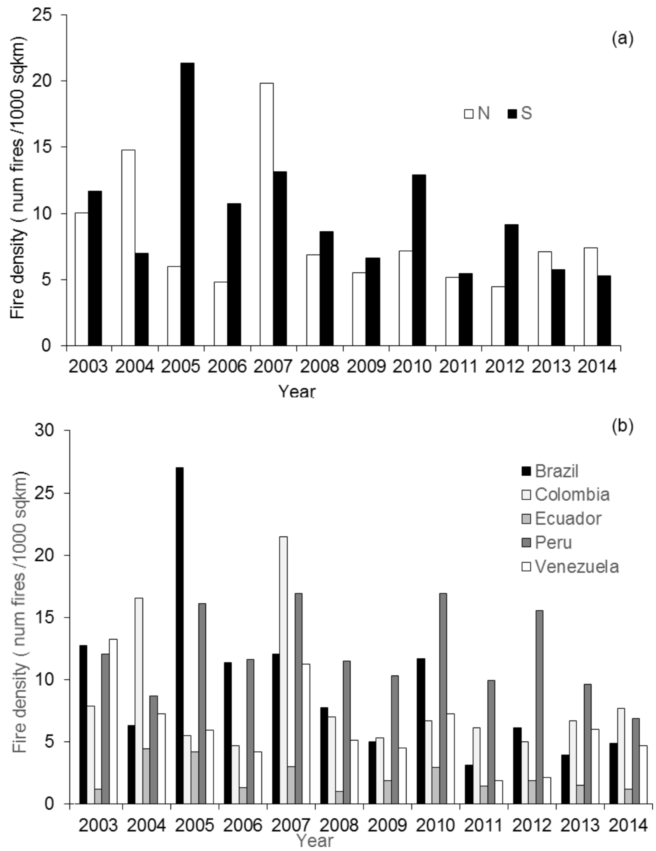

Figure 2. Interannual variability of satellite-detected active-fire density (number/1000 sq km) per latitudinal position (north, south) **(a)** and country **(b)**.

3 Results

There was high interannual variability of MODIS active fire detections (Fig. 2a). The year with the highest density of fires in the north was 2007 while 2005 was the year with the highest number of fires in the south. Concerning the years with fewer fires detected, the north had fewer in 2012 while the south showed the lowest number of fires in 2011. Despite the different patterns in terms of annual average fires there was no significant difference (t test $= 1.0$, $p = 0.17$) between the average annual density to the north (mean $\times$ standard deviation: 7.5×4.6 fires/1000 km^2) and to the south of the Equator (9.0×4.8 fires/1000 km^2). However annual fire density was significantly different between countries (Fig. 2b, ANOVA $F = 8.0$, $p<0.01$).

Seasonal differences between the northern and southern parts of the study area were clear (Fig. 3a) with the main fire season occurring between December and March to the north and between July and October to the south of the Equator. In terms of monthly variability between countries (Fig. 3b), both Colombia (4.0 fires/1000 km^2) and Venezuela (2.0 fires/1000 km^2) presented February as the fire peak month of the year, as expected since both have

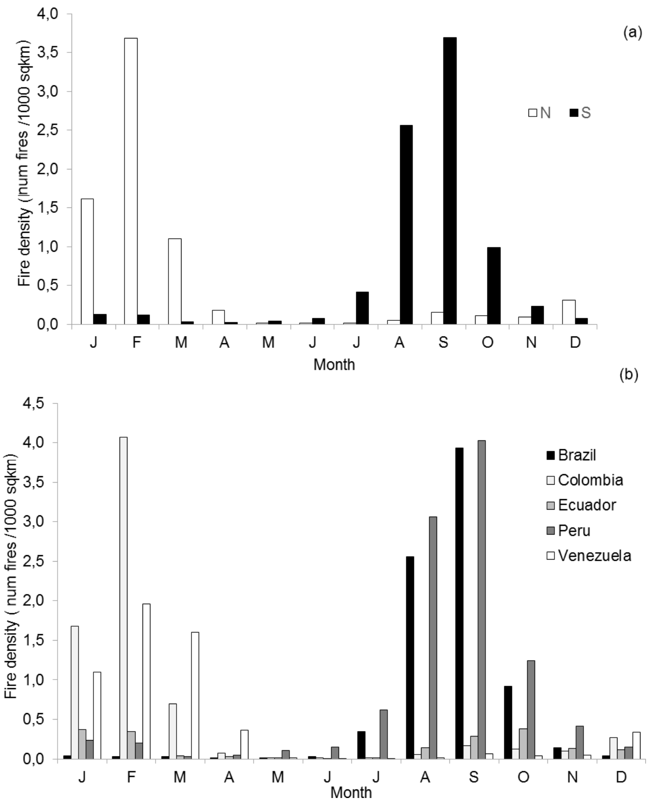

Figure 3. Monthly average satellite-detected active-fire density (number/1000 sq km) per latitudinal position (north, south) **(a)** and country **(b)**.

the highest proportion of their territory in the north. Brazil (3.9 fires/1000 km^2) and Peru, which are mostly located to the south of the Equator had their fire peak in September (4.0 fires/1000 km^2). Ecuador, despite having most of its territory in the Southern Hemisphere had two peaks in January (0.37 fires/1000 km^2) and October (0.38 fires/1000 km^2). The results of the Kolmogorov–Smirnov tests (Supplement 1) used to compare the CFDs of the active fires distances to transportation networks and the null model for distance of the territory within each country, the higher the values the higher the differences between those. For all countries and both roads and rivers, the pattern of fire occurrence is significantly different to their null model and, thus, for each case the fire pattern was related to both roads and rivers (Supplement 1).

Fig. 4 shows the comparison between the CFDs of the observed distributions of distances from fires to the closest rivers (Fig. 4a) and roads (Fig. 4b). There are significant differences between distributions of distances from fires to rivers (Fig. 4a; k-sample Anderson–Darling test, $p<0.001$) and also to roads (Fig. 4b, k-sample Anderson–Darling test, $p<0.001$). A comparison between Fig. 4a and b with their corresponding null-model curves in Fig. 4c and d indicates that most fires were much closer to the river and the road networks than a null model would suggest. In addition to the k-sample Anderson–Darling tests, we computed pairwise comparisons between curves within each of the two sets of CFD in order to evaluate the magnitude of their differences. The results are shown in Tables S4 and S5 of Supplement 2, corresponding to comparisons between curves in Fig. 4a and c, and indicate that differences went from 0.01 to 0.39 (average value of 0.2) for distances from fires to rivers (Figs. 4a and Table B1) and from 0.09 to 0.3 (average value of 0.21) for distances from rivers to roads (Fig. 4b and Table S3).

Figure 4a shows that 80 % of fires in Ecuador were within 300 m of the closest rivers, whereas for Colombia this figure increased to 500 m and remained nevertheless below 1 km for the other countries. Figure 4b, in addition, indicates that a large proportion of distances from fires to roads was below 10 km for all countries. In turn, a comparison between Fig. 4c and d also shows that null models for rivers and roads behaved differently. The CFDs of the null model for rivers (Fig. 4c) showed a strikingly similar set of curves for all five countries, suggesting that distances from fires to rivers were similarly distributed regardless of the country. Although a k-sample Anderson–Darling test of that data sets in Fig. 4c yielded a significant $p<0.001$, the magnitude of the pairwise differences between null-model curves in Fig. 4c was very small (0.01–0.08, average value of 0.039; see Table S3, Supplement 2) compared to the differences in Fig. 4a (Table S4), corroborating their apparent similarity. That is, although the p value indicates that the effect (i.e. the difference between curves) exists, the magnitude of that difference in this case is very small (Sullivan and Feinn, 2012). On the other hand, a visual inspection of the distribution of null-model distances from fires to the closest roads (Fig. 4d) showed noticeable differences between countries, a fact that was confirmed by a k-sample Anderson–Darling test ($p<0.001$) and by the relatively large magnitude of the pairwise differences between null-model CFD curves (0.15–0.48, average value of 0.3; see Table S5). A final examination of the curves in Fig. 4d points out that the country with the highest road density (null-model locations closest to roads) in Amazonia was Ecuador, followed by Peru, Colombia and Brazil.

In relation to the occurrence of fires in the deforestation frontier and where the forest is fragmented, all countries presented a significant relation to the distance to the forest edge (Supplement 3). There were significant differences between countries regarding the CFDs of the distances of the active fire detection to the forest edge (Supplement 3), both towards the inside of the forest (Fig. 5a; Anderson–Darling test, $p \leq 0.01$) and the outside of the forest (Fig. 5b; Anderson–Darling test, $p \leq 0.01$). The vast majority of fires occurred within 500 m outside (Fig. 5b) of the forest edge, with Colombia presenting the most fires occurring close to the edge (e.g. 80 % within 250 m outside the forest or in the forest).

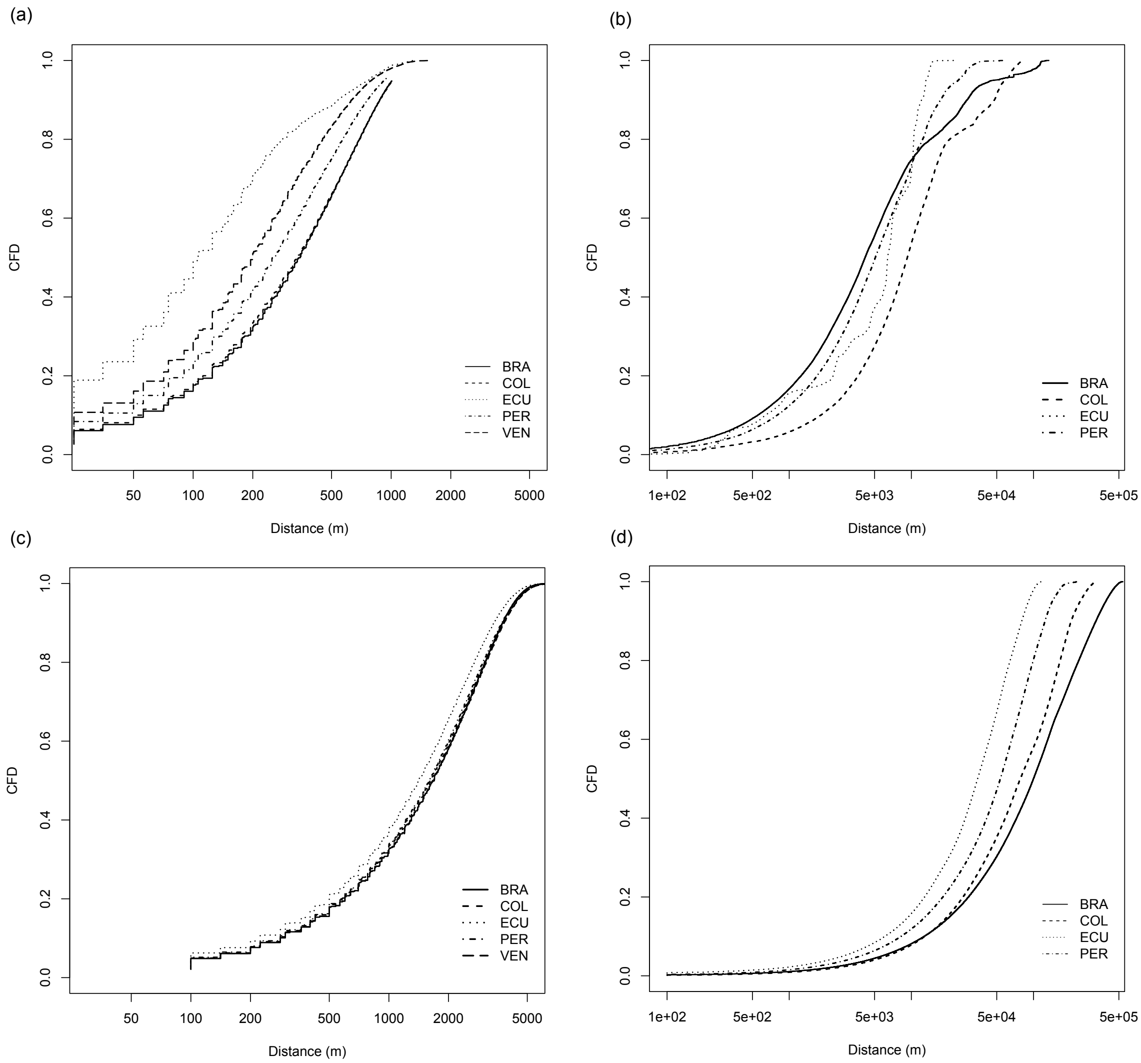

Figure 4. Cumulative frequency of the observed closest distance of fires to rivers **(a)** and roads **(b)** for each country and of the null model for distances to rivers **(c)** and roads **(d)** for each country.

4 Discussion

Our results indicate the temporal pattern of fire occurrence in NW Amazonian tropical forests was determined by its latitudinal position (north/south) and by the country. Thus, intense fires seasons in the Northern Hemisphere were almost the opposite to what is expected in Southern Hemisphere Amazonia in terms of temporal variability (see Fig. 2). Fire dynamics is strongly influenced by climate, and indeed the dry season (∼ July–September) in the southern Amazonia corresponded to a wet season in northern Amazonia, and this is well established. For example, 2004/2005, 2006/2007 and 2009/2010 were El Niño years, affecting the dry season in the Northern Hemisphere, while increased Atlantic sea surface temperatures (SST) in the Atlantic Ocean were responsible for the 2005 and 2010 droughts during the dry season in the Southern Hemisphere (Phillips et al., 2009; Saatchi et al., 2013). The Atlantic Multidecadal Oscillation (AMO), for instance in 2004, 2005, 2007 and 2010, also influenced fire patterns, with strongly positive effects north of the Equator (Chen et al., 2011). Despite the intra-annual variability, the interannual comparison through the average annual fire density for the time period studied did not differ significantly between the north and south. This indicated that regions north and south of the Equator may differ as to when fire occurs but they do not show differences in the intensity and land affected, as the two hemispheres have been equally affected by fire.

Our results also indicated differences in fire patterns between countries. In 2005, fire density was higher in Brazil in association with increased SST in the Atlantic Ocean; in particular the state of Acre was the epicentre of the drought (Aragão et al., 2007; Chen et al., 2011). In the case of the states of Amazonas and Acre, AMO had a stronger posi-

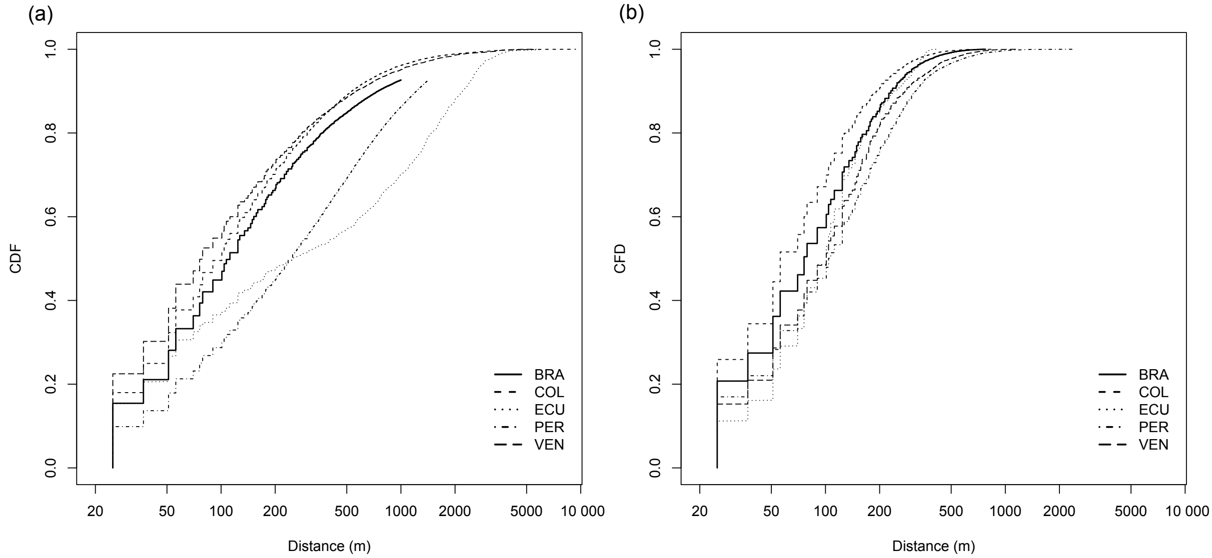

Figure 5. Cumulative frequency of the observed closest distance of fires to the forest edge occurring outside **(a)** and inside the forest **(b)** in 2010 for each country.

tive correlation in 2004, 2007 and 2010 (Chen et al., 2011). Colombia had higher fire density in 2004 and 2007 during two dry seasons associated with El Niño (Armenteras-Pascual et al., 2011) and also influenced by the AMO (Chen et al., 2013b). For Ecuador, only 2004 and 2005 stood out as relative higher density years for this country and are likely associated with the AMO in 2004 and the SST-associated 2005 drought. Venezuela, the only country in this study with all its territory in the Northern Hemisphere, presented a high density of fires in 2004 in association with AMO (Chen et al., 2011) and in the 2007 El Niño year following the same pattern as Colombia. Finally, in 2005 and 2010 Peru presented high densities of fires as expected, being in the Southern Hemisphere. However, Peru stood out in 2007 (a year with particularly more fires in the Northern Hemisphere) for excessive fire density and showed another peak in 2012. The first peak could be associated with the AMO and the 2012 peak with the SST or La Niña year (Marengo et al., 2013), but this also might indicate that, apart from climate, there are other factors influencing the occurrence of fire in this country.

Regarding the influence of accessibility on fire occurrence, we also found (as obtained in a recent study in Legal Amazonia, Kumar et al., 2014) that fires were associated with roads, most of them within 10 km but with a lower 75 % of the 90 % found in Legal Amazonia. This was likely due to the unavailability of data on unmapped and newly developed roads. Unlike the Legal Amazonia study, we did not look at the official and unmapped/unofficial roads because this information is not available yet and nor is the year by year road development for most NW Amazonian countries. However, contrary to Kumar et al. (2014), we found that fires are also strongly associated with rivers, in particular for Colombia, Venezuela and Peru, where most fires occur within 1 km of the closest river. The fact that we also obtained this result in Brazil, where Kumar et al. (2014) did not find this association between fires and rivers, is probably due to the fact that they only accounted for navigable rivers. Our study considered the whole river network given the fact that many colonist in the frontier use small boats to access resources in the forest.

Our results revealed differing relationships between roads, fragmentation and deforestation between countries. The opening of roads in Ecuador and Peru, related to the oil industry (Espinosa et al., 2014; Finer et al., 2015; Finer and Jenkins, 2012; Finer and Orta-Martínez, 2010; Mäki et al., 2001), might be an explanation for increased fire occurrence and deforestation in these two countries. The subregions in which some of these developments have occurred have also reported the highest forest loss, particularly in 2009 and 2010 (Potapov et al., 2014). Colombia contains a large area of the undeveloped Amazon and fire is used as a tool to open the colonization frontier. Fire is also used as a pasture management tool once the frontier advances and basic road infrastructures are developed (Armenteras et al., 2013b; Dávalos et al., 2014). In Venezuela, although not included in the study due to the poor-quality road data, detected fires most likely resulted from expansion of the agricultural frontier (Pacheco et al., 2014).

Despite the fact that forest is associated with fire (Cochrane, 2003), there was little evidence outside Brazilian Amazonia of fire as an edge effect at large scales (Cochrane and Laurance 2002). Our results showed that, in this part of the Amazon, most fires occurred close to the edge and as such, fire occurrence was strongly linked in all countries to forest fragmentation. The distance to which fire edge effects were detected in our study (within 2 km of forest interior edge) coincided with previously recorded distances of fire influence of at least 2–3 km in other areas of Amazonia (Armenteras et al., 2013a; Cochrane and Laurance, 2002) or with the 1–2.7 km at which edge desiccating effects penetrated into fragmented forests (Briant et al., 2010). Our results also aligned with other studies in Brazil and Colombia concluding fire frequency increases at the forest edge (Cochrane 2001; Cochrane and Laurance 2002; Armenteras et al., 2013a). Some studies argue that the majority of fires in Brazil are agricultural fires or escaped fires from managed pastures (Cano-Crespo et al., 2015). It is likely that the countries with a higher percentage of fires resulting in deforestation are those countries, such as Colombia, that have most fires closest to the forest edge and less agricultural development (Armenteras et al., 2013a). Nevertheless, whether a forest fire is an unintended escaped fire or a fire used for forest conversion to another land use, the strong association of fires with both accessibility and fragmentation is an important result worth highlighting for the different countries. Indeed, if burning mostly occurs along forest edges and is also associated with increased access to the forest, all tropical forest edges in all countries are becoming increasingly more exposed to further disturbances. As such there might be different levels of impacts and different causes but there are common ecological consequences such as an increased desiccation affecting forest structure and composition, degradation of these forests, a decrease in living biomass and finally a reduction of their capacity to act as a carbon sink (Balch et al., 2015; Harper et al., 2005).

5 Conclusions

This study showed that, within the same tropical forest biome, there were clear differences between countries in terms of timing of peak fire season (different years, different peaks per country) and that accessibility was associated with increased fire occurrence. Forest edge effects occurred equally in all countries and it might be worthwhile addressing them either regionally or for each individual country, since they are causing forest degradation. All our results underscored the influence, not only of climate, but likely of the strong socio-economic factors (van der Werf et al., 2004) in increasing fires that drive deforestation. Future management plans for NW Amazonia should consider the potential and synergistic edge effects derived from infrastructure development plans and national climate adaptation and mitigation policies. More frequent fires along increasingly fragmented forests may also have other undesired cascade effects in terms of forest degradation and emissions. Subsequent forest loss should be addressed in the context of Reducing Emissions from Deforestation and Forest Degradation (REDD) strategies or other policy mechanisms implemented locally.

Data availability. All data is publically available from their original sources as stated in the methods section.

The Supplement related to this article is available online at https://doi.org/10.5194/bg-14-2755-2017-supplement.

Author contributions. Dolors Armenteras and Javier Retana conceived the idea, designed the analysis, performed the data analysis and wrote the manuscript; Joan Sebastian Barreto and Roberto Molowny-Horas analysed the data. Karyn Tabor contributed to writing part of the manuscript. All authors contributed to revising the manuscript.

Competing interests. The authors declare that they have no conflict of interest.

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