Chapter 13
Fires in Amazonia

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13.1 Introduction

Tropical forest fires are an emerging important environmental issue of the twenty-first century. In Amazonia, this recent preoccupation is, in part, related to the fact that some global climate models predict an increase in the frequency and intensity of droughts (see e.g. Chap. 4), due to changes in atmospheric circulation induced by planetary warming (Li et al. 2006), which may turn the world's largest tropical forest into a more fire-prone system (Malhi et al. 2008). A reduction in rainfall is expected to exacerbate the synergism between climate, deforestation, and fires (Cochrane and Laurance 2002; Hutyra et al. 2005). This drought–deforestation–fire interaction may increase the likelihood of fires to leak into surrounding undisturbed forests, consequently magnifying the contribution of Amazonian fires to global carbon emissions from land use. Fire frequency in Amazonia induced by ongoing human activities has already been observed to have increased during

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periods of prolonged dry seasons during the 1998, 2005, and 2010 droughts (Aragão and Shimabukuro 2010a).

The Large Scale Biosphere-Atmosphere Programme (LBA) has studied the multiple facets of fire since the mid-1990s. These and other studies have focused on understanding the frequency of fire occurrence, their impacts on vegetation and consequent C emissions (Chaps. 5 and 6), as well as the effects of environmental changes on fire events and potential feedbacks between climate, deforestation, and fire (Uhl and Kauffman 1990; Cochrane and Schulze 1999; Rosenfeld 1999; Cochrane et al. 1999; Ackerman et al. 2000; Laurance et al. 2001a, b; Laurance and Williamson 2001; Cochrane and Laurance 2002; Barlow and Peres 2004a; Nepstad et al. 2004; Artaxo et al. 2005; Alencar et al. 2006; Aragão et al. 2007a, 2008; Bowman et al. 2009).

With increasing international demand for C emission reductions to avoid passing dangerous climate change thresholds, controlling the indiscriminate use of fires in the Amazon region can be an efficient strategy to reduce carbon emissions. The key challenge is to improve our understanding of fire regimes and on how they may change with future changes in land use and climate. This will improve our ability to forecast fire incidence at spatial and temporal scales that permit operational interventions for minimising the impacts of fires on carbon emissions, ecosystem services, and human health.

In this chapter, we start by providing an overview of the state of our knowledge on the spatial and temporal patterns of fires, focusing on fire incidence. We briefly introduce the history of the use of fire in Amazonia, including pre-Columbian fires, and depict its configuration in space and time with a particular consideration about its relationship with land use and land cover and the influence of climate seasonality and recent droughts on these patterns. We subsequently focus on the impacts of fire, examining the extent of burned forests during major droughts and describing the main impacts of fire on forest carbon stocks, forest structure, and composition as well as Amazonian people. We then review the main modelling approaches for quantifying and predicting fire occurrence. We conclude by providing a comprehensive view of the processes that influence fire occurrence, potential feedbacks, and impacts in Amazonia, centred on human actions, fire, deforestation, and climate and feedbacks among them.

Most of the analyses presented in this chapter refer to the Brazilian Legal Amazon (BLA), which is the administrative boundary defined by law by the Brazilian government, including not only the Amazon forest ‘biome’, but also part of the Cerrado (savanna) and Pantanal (hyperseasonal flooded cerrado) ‘biomes’ within the national frontiers of Brazil. However, some analyses and discussion are referred to Amazonia. In this case, we analysed the data in the context of the whole Amazon forest ‘biome’, which includes closed and open evergreen broadleaf lowland rainforests across the Amazon basin.

13.1.1 Fire Incidence

Deforestation has been for years the major green house gas (GHG) emission source in Brazil, contributing to c. 77 % of all GHG emissions of the country (MCT 2010). The deforestation process in BLA, which relies on clear cut of the native vegetation and the subsequent use of fire to remove the slashed material, was a key contributor for the 1,614,970 fire occurrences detected by the Terra/MODIS sensor between 2001 and 2010 in Brazil (Fig. 13.1a and b).

Despite a reduction in deforestation rates in 2010 by 64 % below its 5-year average from 2005 to 2009 (PRODES 2013), fire incidence has increased in 59 % of BLA with decreasing trends in deforestation rates (Aragão and Shimabukuro 2010a). Possibly this pattern relates to the high probability that deliberate fires, used for managing pastures and suppressing regrowth in deforested areas, were leaking into surrounding intact forests, helped by recent droughts, and by ongoing increase of forest edge area, number of fragments, and secondary forest area (Aragão and Shimabukuro 2010b).

To disentangle the potential interacting factors and processes, we first depict the historical usage of fire in Amazonia, from pre-Columbian time to recent days. We then examine the recent spatial and temporal footprint of fire in Amazonia, giving particular attention to the effects of land use and land cover, climate seasonality, and droughts on fire patterns.

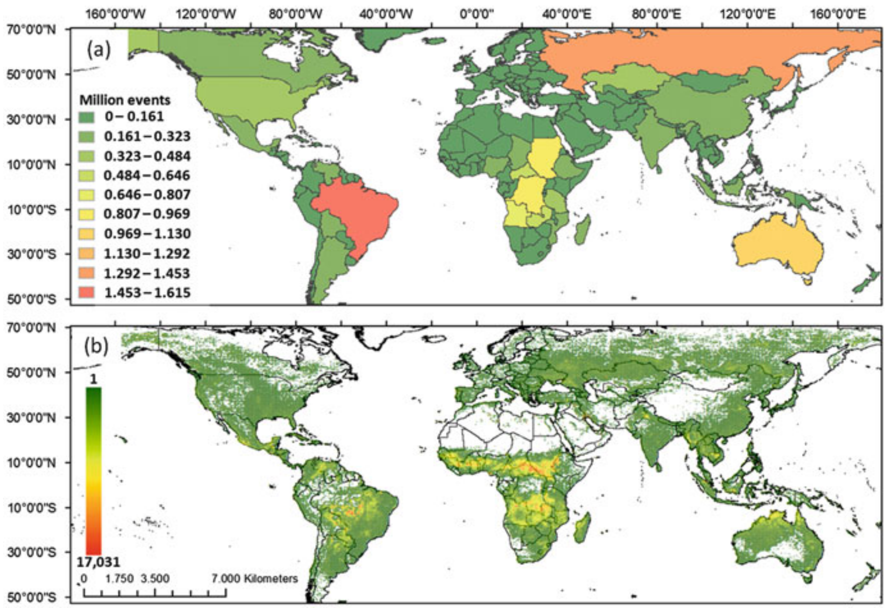

Fig. 13.1 Global view of (**a**) total number of active fire detections per country and (**b**) the spatial configuration of total number of active fires per 0.25° × 0.25° grid cells from Terra/MODIS sensor between 2001 and 2010

13.1.2 Fire Usage in Amazonia

Most of the historical evidence suggests that wildfires in tropical forests were rare, with return time intervals typically ranging from hundreds to thousands of years (Sanford et al. 1985; Meggers 1994; Bush et al. 2007). Despite the possibility of natural fire occurrence in pre-Columbian times, it is well accepted that the presence of charcoal is an indicative of the use of fire by humans (Bush et al. 2008).

Establishing the causes and consequences of paleo-wildfires in Amazonia is compromised by the small number of sites investigated and their spatial configuration. Radiocarbon dating of charcoal collected from soil samples around San Carlos de Rio Negro, Venezuela, indicates the occurrence of wildfire events at 6000, 3000, 1500, 650, 400, and 250 years ago (Sanford et al. 1985; Saldarriaga and West 1986). These dates match with dry climatic phases in the late Holocene as confirmed by pollen dating (Sanford et al. 1985).

In the past 50 years or so, fires have become more frequent (Bowman et al. 2009), with the vast majority of burning events resulting from human-lit fires (Cochrane and Schulze 1999; Cochrane et al. 1999; Uhl and Kauffman 1990). In the Amazon basin, fire is widely used for the initial conversion of extensive areas of natural vegetation into agricultural fields and pasture areas and for the subsequent suppression of secondary succession (Cochrane and Schulze 1999; Kodandapani et al. 2004; Giglio et al. 2006; Bowman et al. 2008; Sorrensen 2008).

Anthropogenic activities can facilitate and directly increase the spread of fire into forest systems (Cardoso et al. 2003) by creating and enlarging forest edges and by disturbing forests through selective logging, which increases forest flammability (Uhl and Bushbacher (1985); Nepstad et al. 1999; Cochrane and Laurance 2002; Alencar et al. 2006). Secondary forests, regrowing on deforested areas (e.g. Lucas et al. 2000), are also vulnerable to spreading fire as they can become rapidly desiccated and flammable during dry periods (Ray et al. 2005).

The recent relatively large-scale settlement of humans in Amazonia was a result of large government development projects (see e.g. Chap. 18). This process inevitably involved the frequent use of fire to clear forests and agricultural residue (e.g. Cardoso et al. 2003; Nepstad et al. 2004). The expansion of the agricultural frontier, therefore, led not only to an exponential increase in population in Amazonia (Aragão et al. 2014), which can be directly associated with fire ignition sources, but also to fragmentation of the natural vegetation matrix (cerrado, forest), which increased the susceptibility of forests to fires.

Land use dynamics and related fire patterns in the region may vary according to the price of agricultural commodities and due to various biophysical and socio-economic factors, such as planned settlement, changes in infrastructure and accessibility, as well as government policies (Sorrensen 2008; Brondizio and Moran 2008; Carmenta et al. 2011). Clearing for intensive agriculture is usually characterised by repeated burning. This process can take up to 3 years for achieving the complete removal of the slashed vegetation (Morton et al. 2006).

Several studies have demonstrated a temporal association between fire and deforestation, in the Brazilian Amazon and elsewhere (Sorrensen 2000, 2004; Bowman et al. 2008; Sorrensen 2008; Aragão et al. 2008; Morton et al. 2013). This relationship is consistent with the fact that burning events in Amazonia are usually man-made. Spatial patterns of fire occurrence are also expected to follow the patterns of forest conversion and subsequent land use (Lima et al. 2012). Despite the extensive evaluation of the temporal links between fire and deforestation, there is still a need for better understanding the spatial structure of the association between fire and deforestation and evaluating how past and present land use and land cover change (LULCC) influence these spatial patterns.

13.1.3 Fire, Land Use, and Land Cover

Natural fires are rare in the Amazon forest 'biome' (Cochrane 2001; Bush et al. 2007), but common in the Cerrado 'biome' (Ramos-Neto and Pivello 2000), in the south and east of BLA (Fig. 13.2a). In the Cerrado, the long dry season period (5–7 months) with rainfall lower than 100 mm month^{-1} (Sombroek 2001) makes climatic conditions suitable for natural fires. This 'natural' pattern of low fire incidence in the rainforest of Amazonia and high incidence in the Cerrado appears to have changed in recent years.

Deforestation in BLA was responsible for the transformation of around 760,000 km^2 of pristine seasonal, open, and closed canopy forests into pastures for cattle ranching and agricultural lands by 2012 (PRODES 2013; Chap. 15). This value corresponds to c. 15 % of the original forested area of BLA. Forest conversion in BLA has mostly affected the contact zones between the Cerrado and Amazon forest 'biomes', expanding deep into the forest where access routes were available (e.g. Fig. 13.2b). The spatial configuration of fires detected by satellites is highly linked to the distribution of deforestation (Fig. 13.2c). This is expected because of two widespread land use practices, acting as ignition sources for fires in the region: (1) land clearing by slash and burn and (2) management of pastures using fire.

Fire recurrence time, as the time needed for fire to strike in the same area, has been reduced to around 5–15 years (Cochrane et al. 1999; Alencar et al. 2006) because of the amplification of land use in Amazonia. Within BLA, Mato Grosso (MT) has been the state with the highest fire occurrence, detected by MODIS/TERRA satellite, with an average (mean $\pm$ standard deviation) of 1432 $\pm$ 1838 and 754 $\pm$ 1109 active fires per month detected between November 2000 and July 2011 in the biomes Amazonia and Cerrado, respectively (Fig. 13.3).

Observing fire patterns across 'biomes' within BLA for the same period, it is clear that despite the dominance of fires in the contact zone between Amazonia and Cerrado 'biomes' (1443 $\pm$ 1716 active fires), closed and open broadleaf forests are also exposed to extensive fire occurrence (1692 active fires) (Table 13.1).

In the BLA region, as a whole, fire incidence has been directly related to deforestation (Aragão et al. 2008). The annual rates of deforestation in BLA have

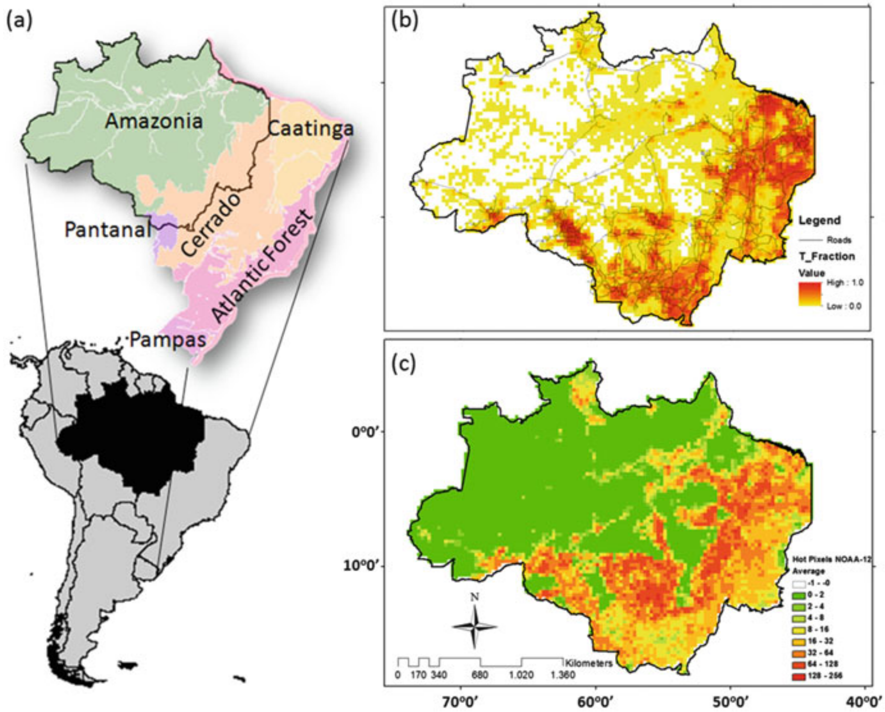

Fig. 13.2 (**a**) Map of South America, displaying the Brazilian Legal Amazon (BLA, *black* limits) and a detailed distribution of Brazilian 'biomes'. (**b**) Fraction of the total grid cell area (0.25° × 0.25°) that has been deforested by 2007, where 0 means no deforestation and 1 complete conversion of the grid cell. (**c**) Mean number of active fires detected by the NOAA-12 sensor from 1998 to 2006 in each grid cell with similar spatial resolution as (**b**)

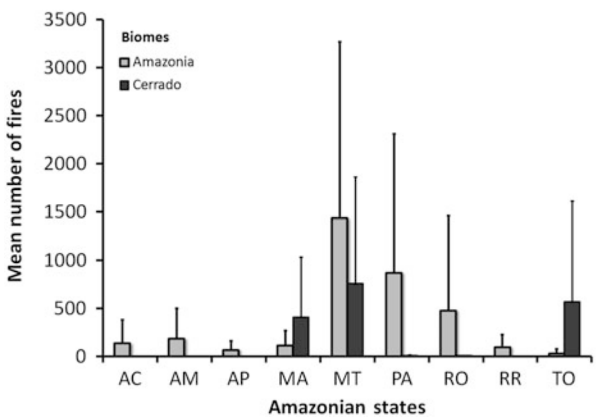

Fig. 13.3 Mean number of fires per month that occurred in each Amazonian state from November 2000 to July 2011. Error bars correspond to the standard deviation

Table 13.1 Mean (±SD) number of active fires detected by MODIS/Terra in 0.25° × 0.25° grid cells analysed in each forest type (count) in Brazilian Amazonia from November 2000 to July 2011

	Contact zone	Open broadleaf	Closed broadleaf	Seasonal semi-deciduous	Seasonal deciduous	Campinarana
Mean	1443	1055	637	112	31	4
Sd	1716	2072	870	158	52	4
Count	129	123	126	129	79	42
Percent of the total	100	95	98	100	61	33

Percent of the total, the percentage of grid cells within each forest type that have at least one fire recorded within the analysed period

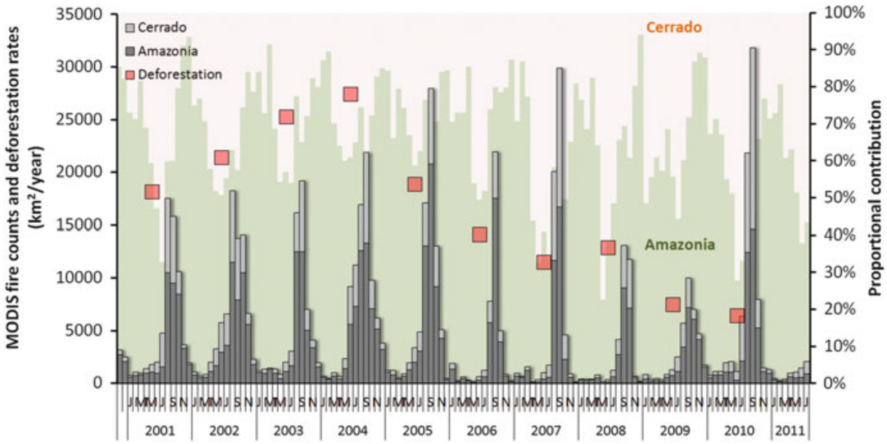

Fig. 13.4 Total number of monthly active fire occurrences within the domain of the Brazilian Legal Amazonia. *Grey bars* correspond to the sum of fire occurrence in the Cerrado (*light grey*) and Amazon forest (*dark grey*) 'biomes'. The *coloured areas* correspond to the proportional contribution of fires in the Cerrado (*light magenta*) and in the Amazon forest (*light green*) to the total number of fire events detected in each month. *Magenta squares* correspond to annual deforestation rates

decreased from 21,400 km^2 years^{-1} (1980s) to 4571 km^2 recorded in 2012 (PRODES 2013). Nevertheless, this significant reduction in deforestation did not involve a proportional reduction in fire incidence (Fig. 13.4). This result corroborates the study by Vasconcelos et al. (2013), which, analysing MODIS active fire data from 2003 to 2012 for the state of Amazonas, did not observe the deforestation–fire relationship proposed by Aragão et al. (2008). Moreover, Aragão and Shimabukuro (2010a) also quantified a decoupling between fire and deforestation in BLA, analysing active fire data from AVHRR and MODIS sensors.

The decoupling between fire and deforestation in the past 10 years may be related to the fact that secondary forests are not included in forest loss monitoring programmes (PRODES 2013). Their conversion to agricultural use, as reported for

BLA (Fearnside et al. 2007), is not accounted for as new deforestation, while fires associated with this conversion are quantified. In addition, the increased number of forest fragments and area of forest edges (Broadbent et al. 2008), which are more vulnerable to escapee fires, and the increased frequency of droughts in recent decades (Marengo et al. 2011) could also have contributed to an absolute increase in fire occurrences.

In contrast to the positive trends observed in 42 % of the forested area in BLA from 1998 to 2006, in the Cerrado within the BLA, fire occurrence has decreased during the same period (Aragão et al. 2013). The Cerrado covers a climatic region with high risk of fires (Arima et al. 2007); however, once the natural vegetation has been cleared by fire, mechanised intensive agriculture tends to reduce fires (use for burning residues) in comparison with agro-pastoral land use. Therefore, robust predictions of fire risk require explicit information on land use (see Sect. 4.3) as a key additional driver to the climatic and some infrastructural variables (e.g. distance to roads, connectivity to markets, and population density) currently used in fire probability models available (Laurance et al. 2002).

To date, we have achieved a reasonable understanding of fire patterns that accompany the conversion of woody vegetation to agro-pastoral use in Amazonia; however, the direct influence of different land uses on the spatial patterns of fire incidence still need to be adequately considered in future work.

13.1.4 Fire, Climate Seasonality, and Droughts

In addition to land use and cover change, climate seasonality can become extremely relevant for determining fire occurrence in Amazonian forests if human-related ignition sources are active. Rainfall, temperature, and relative humidity (Cardoso et al. 2003; Sismanoglu and Setzer 2005), plant available water (PAW) (Nepstad et al. 2004), and vapour pressure deficit (VPD) inside the canopy (Ray et al. 2005) are some of the most important factors directly related to forest fires in Amazonia. The seasonality of these variables, which normally co-vary, defines the period of occurrence and the intensity of fires (the intensity also depends on the availability of fuel loads). Overall, fires tend to intensify during July, August, and September (Fig. 13.4). This period corresponds to the dry season in most of the Amazon area, with rainfall lower than 100 mm month^{-1}, high VPD, and low PAW, especially in the south and east of the region.

Severe droughts, moreover, can exacerbate fire incidence and severity, as observed recently in 1997/1998, 2005, and 2010. The majority of droughts in the region are associated with extreme El Niño events, which is characterised by the anomalous warming of the equatorial Pacific Ocean near the coast of Peru (Marengo 1992; Uvo et al. 1998; Ronchail et al. 2002; Marengo 2004). Recent El Niño events occurred in 1982/1983, 1986/1987, and 1997/1998 (Fig. 13.5). During the last decade, contrarily, droughts in Amazonia have been associated with anomalously warm waters in the tropical Atlantic Ocean, following the Atlantic

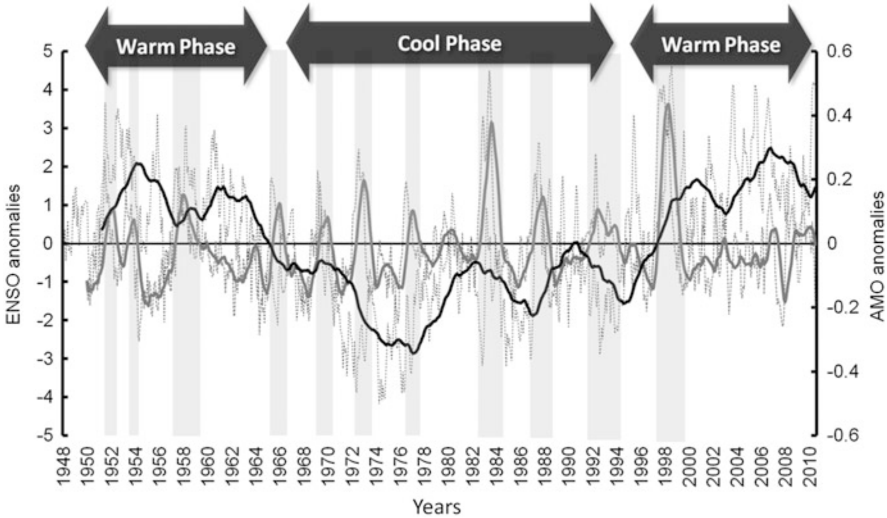

Fig. 13.5 Monthly sea surface temperature anomalies in the tropical Pacific Ocean (*dotted grey line*), related to the El Niño Southern Oscillation (ENSO) event and measured using the Multivariated El Niño Index (MEI) and in the north Atlantic Ocean (*slashed black lines*) represented by the Atlantic Multidecadal Oscillation index (AMO). Thick lines represent moving averages of 12 months for the ENSO (*grey*) and 40 months for the AMO (*black*)

Multidecadal Oscillation (AMO) cycle (Li et al. 2006; Good et al. 2008; Marengo et al. 2008). The AMO was identified as a partial driver of the 1997/98 drought and the main driver of the 2005 and 2010 droughts (Marengo et al. 2008). Amazonia constantly experiences El Niño or AMO-driven cycles that can cause droughts (Fig. 13.5). As some of these droughts can manifest as extreme events (see e.g. Chap. 4), refining our current understanding on how fire patterns respond to such extremes is critical for predicting future impacts of fire on Amazonian ecosystems and human populations.

El Niño-driven droughts normally affect northern Amazonia during the boreal winter, which corresponds to the dry season in parts of South America, north of the Equator. Moreover, eastern Amazonia is affected by these El Niño-driven droughts during the austral winter, due to the opposite timing of the dry season. AMO-driven droughts are related to droughts in the south-west of Amazonia during the Austral winter (Saatchi et al. 2013).

The 1997/1998 El Niño-driven drought created perfect conditions for the widespread occurrence of extensive wide-spreading fires. The total area of forests burned by understory fires in BLA, for instance, was 14 times higher than during an average non-El Niño year (Alencar et al. 2004, 2006). These fires followed the drought pattern affecting areas in northern (Barbosa and Fearnside 1999) and south-eastern flanks of BLA (Alencar et al. 2004).

The drought in 2005, conversely, was driven by the warming up of the tropical Atlantic. This drought also led to a large reduction in rainfall during the dry season

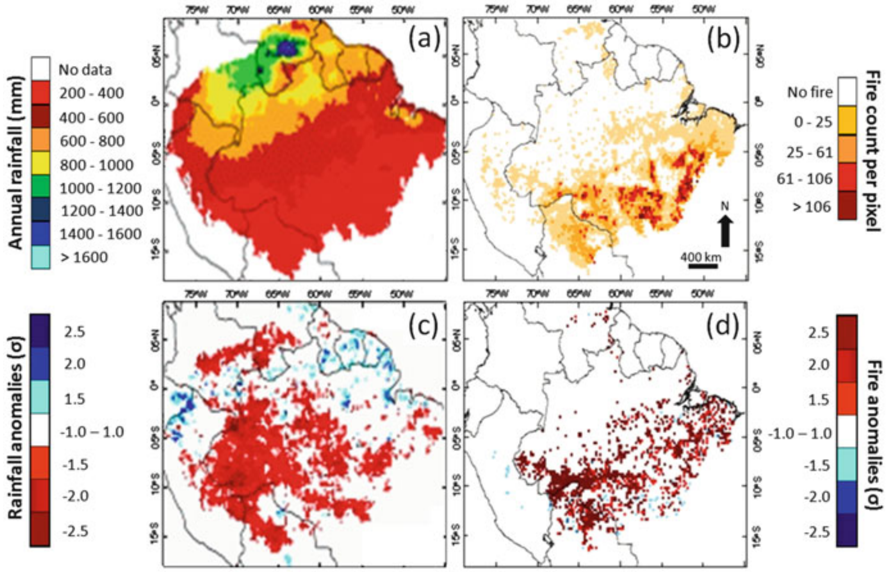

Fig. 13.6 *Top left panel* (**a**) shows the long-term (1997–2006) mean of accumulated rainfall for the months of July, August, and September (dry season in most of the Amazon region). *Top right panel* (**b**) shows the same as top *left panel* but for active fire dataset. *Bottom left panel* (**c**) displays rainfall anomalies (units in standard deviation of the long-term mean) for the trimester of July, August, and September of 2005 and the *bottom right panel* (**d**) shows the same as *bottom left* using the active fire dataset. Adapted from Aragão et al. (2007a)

(July, August, September) (Fig 13.6a and c). This anomalous water shortage created ideal conditions for the widespread occurrence of fires in south-western Amazonia (Aragão et al. 2007a; Cardoso and Oliveira 2007). Fire occurrence increased by 33 % in relation to the long-term average with anomalies reaching values larger than two standard deviations (σ) of the mean (Fig. 13.6b and d).

In 2010, despite the drought being more severe than in 2005, fire incidence was 26 % lower. A total of 34,484 active fires were recorded for the Amazon forest 'biome' in 2005 in comparison with 25,612 in 2010. Cerrado areas within BLA were more affected by fire than the Amazon forest 'biome' (Fig. 13.4). The lower number of fires in 2010 in relation to 2005 was probably a reflexion of the 66 % reduction in deforestation rates in comparison to 2005, limiting ignition sources. Nonetheless, the number of fires normalised by the area of deforested land in BLA increased from 1.83 fires km^{-2} of deforested land in 2005 to 3.97 fires km^{-2} of deforested land in 2010. Morton et al. (2013) quantified a 22 % increase in the area affected by understory forest fires from 2005 to 2010. These results clearly indicate that fire counts detected from satellites are not restricted to forested areas quantified by the INPE/PRODES deforestation programme and other land cover types may be increasingly exposed to fire impacts (Aragão and Shimabukuro 2010b; Lima et al. 2012).

The key to understanding fire in Amazonia is that although drought exacerbates fire occurrence, the ignition of fires is man-made; even during extreme droughts Amazonian forests would not be affected by natural wildfires. This is evident by the lack of significant positive fire anomalies in the Peruvian Amazon (low human occupation/activity) during the 2005 drought (Fig. 13.6d), as opposed to fire anomalies observed in eastern and southern BLA (high levels of human activity)

Where ignition sources are present, two climatic variables are critical in shaping monthly and annual fire incidence behaviour: monthly rainfall and the length of the dry season. Monthly rainfall explains around 60 % of the variance in fire incidence between 1997 and 2006 (Fig. 13.7a). Rainfall lower than evapotranspirative loss (mean c. 103.4 $\pm$ 9.1 mm month^{-1} (Shuttleworth et al. 1989; Malhi et al. 2002; Cox et al. 2004; Rocha et al. 2004; Hutyra et al. 2005) causes water deficit, which is correlated with a high fire incidence. Over 50,000 active fires per month were detected in one single month for the whole BLA during the 2005 drought event. The relationship between fire and monthly rainfall follows an exponential decay function, decreasing fire incidence with increase in rainfall (Aragão et al. 2008), as for example observed between 2003 and 2012 in Amazonas State (Vasconcelos et al. 2013).

At the annual scale, the length of the dry season explains c. 70 % of the variance in the maximum monthly number of active fires (Fig. 13.7b). As the length of dry season length increases, and consequently enhancing the water deficit of many forests in Amazonia, which are common features of droughts in the region, leaf shedding is exacerbated (Alencar et al. 2004; Phillips et al. 2009). This boost of organic matter on the forest floor, with associated increase in canopy gaps (Ray et al. 2005), favours the rise of temperature inside the forest and the reduction in soil and litter moisture. With the increase in drying the accumulated combustible fuel on the forest floor during extended dry periods conditions becomes ideal for the spread

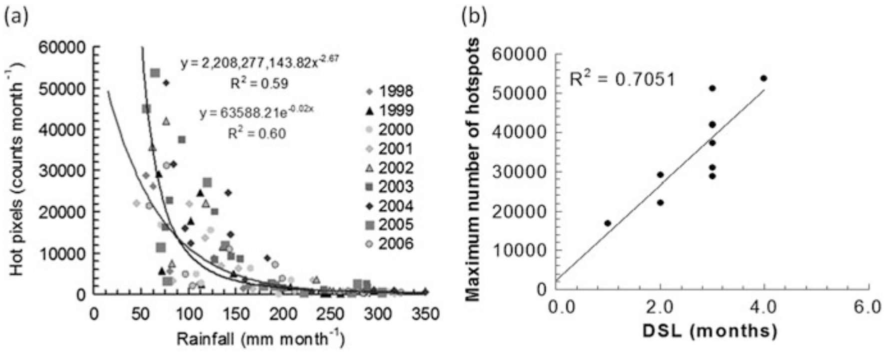

Fig. 13.7 (**a**) Relationship between monthly active fire detections (*hot pixels*) and monthly rainfall and (**b**) relationship between the maximum cumulative number of active fire detections in each year analysed and the dry season length (DSL). Each dot in the graph corresponds to the month with the maximum number of fires within a year. Both analyses considered data from NOAA-12 from 1998 to 2006

of fires into forest (Uhl and Kauffman 1990; Cochrane and Schulze 1999; Cochrane et al. 1999; Barlow and Peres 2004b; Nepstad et al. 2004; Ray et al. 2005).

Understanding relationships between fire incidence, human activities, and climate in Amazonia may offer an approximation of the expected changes in fire activity and an indication of the likelihood of forests to be impacted by fires during these extreme events under future climate conditions.

13.2 Fire Impacts

13.2.1 Extent of Burned Areas

Fire in the late Holocene has been associated with the increased adoption of agriculture (c. AD 200 and AD 800), with El Niño-related droughts (c. AD 800 and AD 1000–1100), and with insolation minima (Bush et al. 2008; Mauas et al. 2008). Fires that occurred in this period and in non-drought years were likely to be small fires that almost always extinguished themselves, at most, within 100 m inside the forest (Uhl and Kauffman 1990). However, in the early 1970s, colonisation and settlement projects in BLA changed this pattern.

The beginning of the large-scale burned area estimates is associated with the use of satellite imageries for detecting anomalous high temperature and fire plumes. One of the first studies for BLA (c. 5×10^6 km^2) suggested that in the dry season of 1987, 350,000 independent fires were detected, possibly corresponding to about 200,000 km^2 of area burned (Setzer and Pereira 1991).

During the El Niño event in 1997/1998, it was estimated that in Roraima state (total area of 224,299 km^2) fires burned over an area between 33,000 km^2 and 38,144 km^2 (UNDAC 1998; Barbosa and Fearnside 1999). From this total, three studies quantified understory fires in forested areas: between 7800 and 9200 km^2 (Barbosa 1998), 11,730 km^2 (INPE 1999) and between 11,394 km^2 and 13,928 km^2 (Barbosa and Fearnside 1999). It was estimated that a total of 39,000 km^2 of forest in the whole BLA was affected by understory fires during this drought (Alencar et al. 2006; Mendonça et al. 2004)

During the 2005 drought, in Acre state (total area of 152,581 km^2), the epicentre of the drought, c. 3700 km^2 burned in previously deforested areas and 2800 km^2 corresponded to understory forest fires (Shimabukuro et al. 2009). Recent estimates of Amazonian forests affected by fires suggested a repeated fire activity in 16 % of all understory fires from 2002 to 2010 (Morton et al. 2013). Moreover, these results indicated that 73 % of the forests affected by fires in 2010 did not burn previously. This result is in agreement with the results of Alencar et al. (2011), who using Landsat data showed that 72 % of the fire-affected forest burned only once during a 23-year study period.

Taking advantage of the extensive database of satellite products, we carried out an analysis of burned areas in Amazonia from 2001 to 2012 by using the MODIS

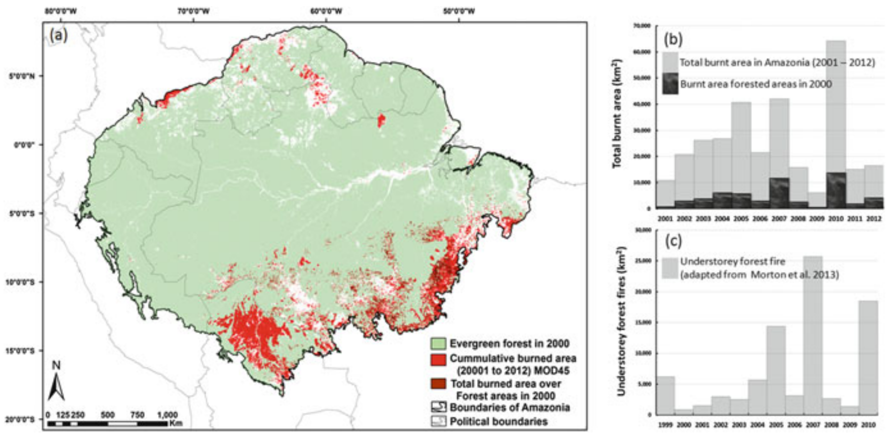

Fig. 13.8 (**a**) Spatial distribution of the cumulative burned area from 2001 to 2012 based on the MODIS MOD45 c5.1 product. The boundaries of Amazonia cover an area of c. 6.76 million km^2. (**b**) Extent of annual burned areas for Amazonia and over primary forests areas in 2000 based on MOD45 collection 5.1. (**c**) Extent of understory forest fires, adapted from Morton et al. (2013)

product MOD45 collection 5.1, for the geographical boundaries defined by Achard et al. 2005 (Fig. 13.8a). Our results showed peaks in burned areas in 2005 (40,500 km^2), 2007 (42,000 km^2), and 2010 (64,000 km^2) (Fig. 13.8b). Considering only the areas that were primary forest or recently deforested, by masking the data outside forest boundaries in 2000, we quantified that burned areas associated with these two land covers peaked in 2004, 2007, and 2010 with an area of c. 6000 km^2, 11,500 km^2, and 13,600 km^2, respectively (Fig. 13.8b). Based on this, c. 4.5 % of Amazonia has burned at least once in the last 12 years. Our results also indicate that c. 60,000 km^2 of the burned area recorded during the studied period was related to forest conversion, land maintenance, and fire leakage to forests. It is interesting to note that, although monitoring an area extent less than half that considered in our analysis, Morton et al. (2013) detected peaks in burned forests of 14,300 km^2, 25,600 km^2, and 18,500 km^2 in 2005, 2007, and 2010, respectively (Fig 13.8c). The differences between the estimates provided by these two independent studies indicate that there are still high uncertainties related to the detection of burned areas, particularly over primary forests. Cloud coverage, data availability for only part of the area, the time window selected, and methods used for detecting burn scar, especially in forest areas, are a major source of uncertainty for quantifying the area burnt (Box 1).

13.2.2 *Impact Fires on the Structure, Composition, and Carbon Stocks of Forests*

The amplified incidence of large forest areas affected by fire in recent years, because of the leakage of agricultural fires into surrounding forests, has caused large changes in the structure and composition of these forests as well as in the maintenance of their carbon stocks. Despite the lack of quantification, episodes of augmented fire incidence and leakage were probably happening since the start of colonisation in Amazonia. However, only after the 1982/1983 El Niño drought event that Uhl and Bushbacher (1985) have first assessed the influence of fires on logged forests in Amazonia. Globally, the relevance of this issue increased after the simultaneous droughts and forest fires in Amazonia and south-east Asia in 1997/1998. This event brought to light that (1) these severe and sometimes recurrent fires (Fig 13.9a, b, and c) result in high levels of tree mortality, which can initiate a process of 'savannisation' or transformation to secondary forest of primary forests (Cochrane et al. 1999; Malhi et al. 2008; Barlow and Peres 2008; Xaud et al. 2013), (2) fires could promote a positive-feedback cycle, where forests that burn once become increasingly flammable and are likely to succumb to a more severe recurring fire (Cochrane et al. 1999), and (3) fires could be emitting significant levels of CO_2 globally (Nepstad et al. 1999).

In transitional semi-deciduous Amazonian forests between evergreen rainforest and cerrado small trees are highly vulnerable to low-intensity understory fires. About 50 % of stems with a diameter at breast height (DBH) < 10 cm can die within 1 year after fire (Balch et al. 2011). Larger trees can also suffer high levels of fire-induced mortality and biomass loss, which tends to increase with fire intensity

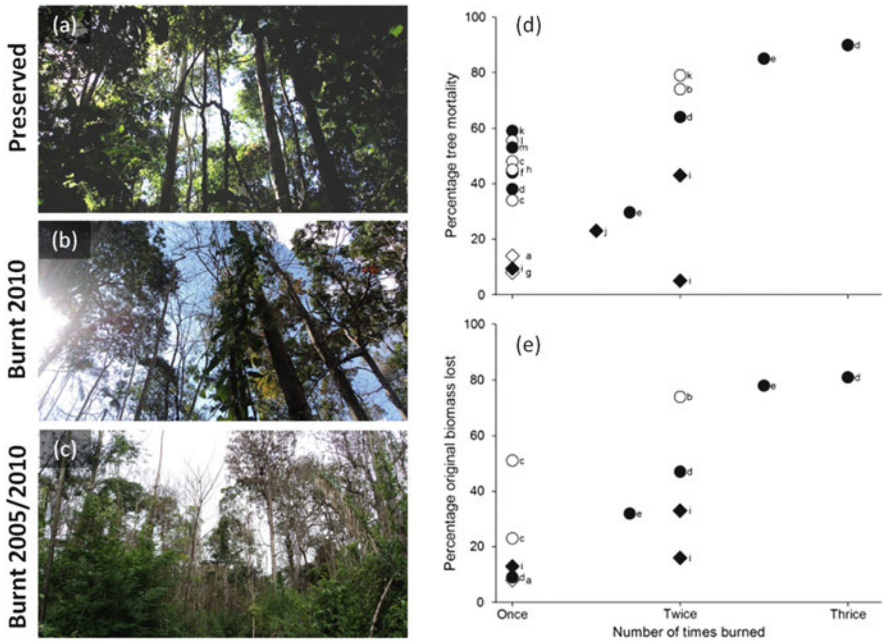

Fig. 13.9 Photos on the *left* show examples of intact forests (**a**), once-burned (**b**), and twice-burned forests (**c**) in Acre State. *Left panels* show the effect of consecutive burns on tree mortality (**d**) and biomass (**e**). Panels on the *left* are from Barlow et al. (2012)

and decrease in return time interval (Brando et al. 2012; Barlow et al. 2012, Fig. 13.9d and e). Substantial variation in the vulnerability of trees to fire has been observed in Amazonia (Balch et al. 2011). This variation is dependent on species-specific traits that can protect trees against fire damage. For instance, Brando et al. (2012) quantified that < 20 % of individuals with bark thicker than 18 mm died from fire damage. Moreover, mortality decreases as tree diameter and height increase and species with dense wood survive better than species with light wood. The large variation in fire-induced tree mortality, determined by the canopy and fire characteristics, is also reflected in the results of biomass loss in Amazonian forests (Fig. 13.9d and e).

Primary forests affected by successive fire events tend to undergo a complete turnover in species composition (Barlow and Peres 2008, Fig. 13.10). Fire-induced mortality and consequent gap formation favour the establishment of fast-growing pioneer species (Barlow and Peres 2008). These species usually have lower wood density than slow-growing, late-successional species (Baker et al. 2004). This characteristic cannot only directly increase the vulnerability of these forests to recurrent fire but also indirectly create a feedback, as trees with low wood density are more susceptible to mortality during droughts (Phillips et al. 2009) and as a result the increase in organic debris can facilitate fire spread into these forests during subsequent drought events.

Annually recurring experimental fires over 5 years have reduced the number and diversity of regenerating stems, and the species pool tended to change towards cerrado-like vegetation in MT (Massad et al. 2013). The reduction in species

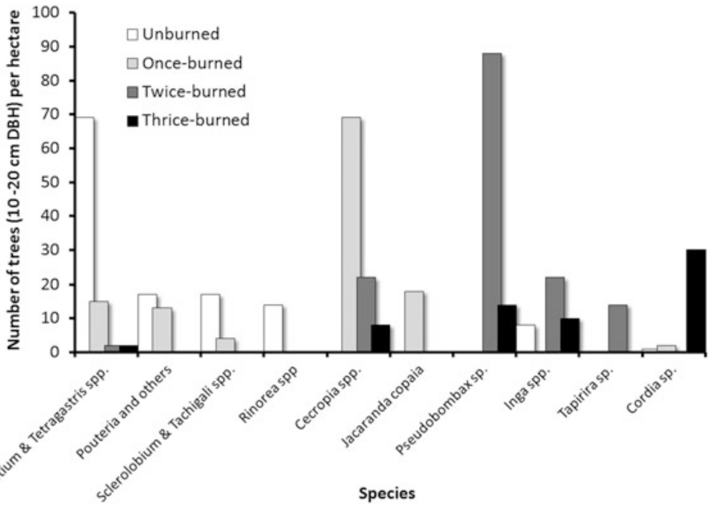

Fig. 13.10 Tree species and genera from the 10–20 cm DBH size class (and shrubs and samplings <10 cm DBH) that were most abundant in each burn treatment, showing a high degree of turnover in community composition with each additional burn. All species (or genera) with a density greater than 10 trees per hectare are shown for trees ≥10 cm DBH, in once-, twice-, and thrice-burned forest plots. Data in the figure are from (Barlow and Peres 2008)

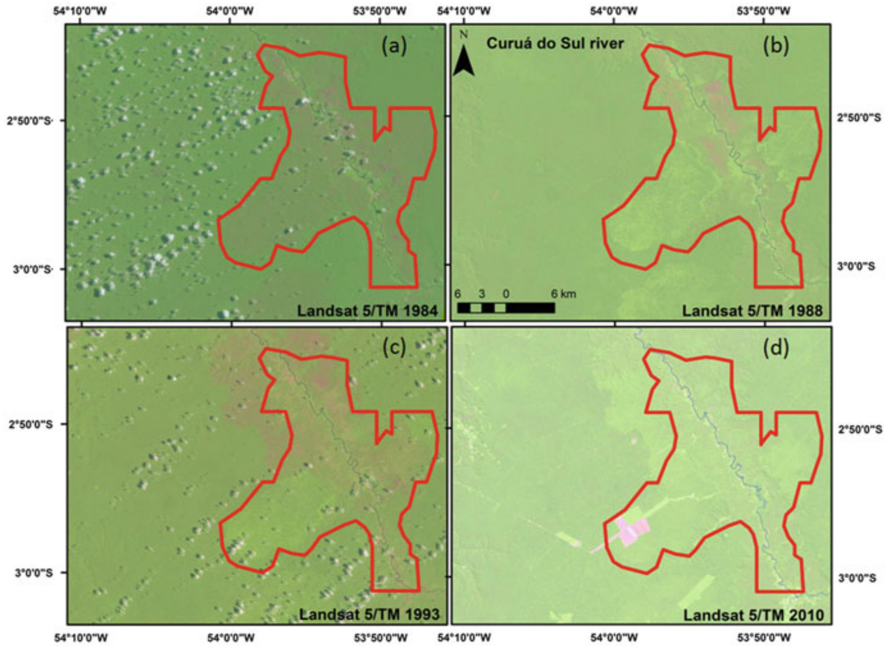

Fig. 13.11 Landsat 5/TM images, path/row 227/62, composition RGB 543, showing the change in the reflectance of a forest area affected by fires during the El Niño in 1983 (Nelson 1994), highlighted by the *red polygon*. Burned forests exhibit *higher shades* of *red colour* or *lighter shades* of *green* when compared with healthy forests (*darker green*). (**a**) Image acquired on 24th August 1984, 1 year after the fires. (**b**) Image acquired on 3rd August 1988, 5 years after the fire. (**c**) Image acquired on 20th October 1993, 10 years after the fire. (**d**) Image acquired on 29th June 2010, 27 years after the fire

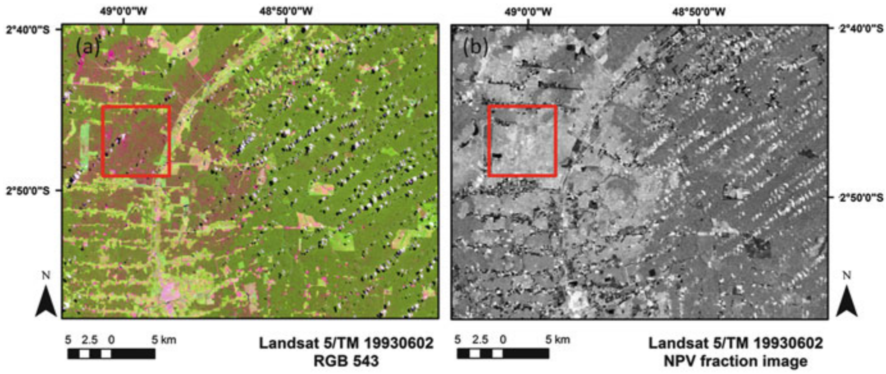

Fig. 13.12 Example of a forest burned area (*left* image) and the non-photosynthetic vegetation fraction image (*right* image) for the same area and date evaluated in the pioneer work of Cochrane and Souza (1998). Burned forests are highlighted by the *red rectangle* and appear brightened in the NPV fraction image. Landsat 5, path/row: 223/62, 2nd of June 1993

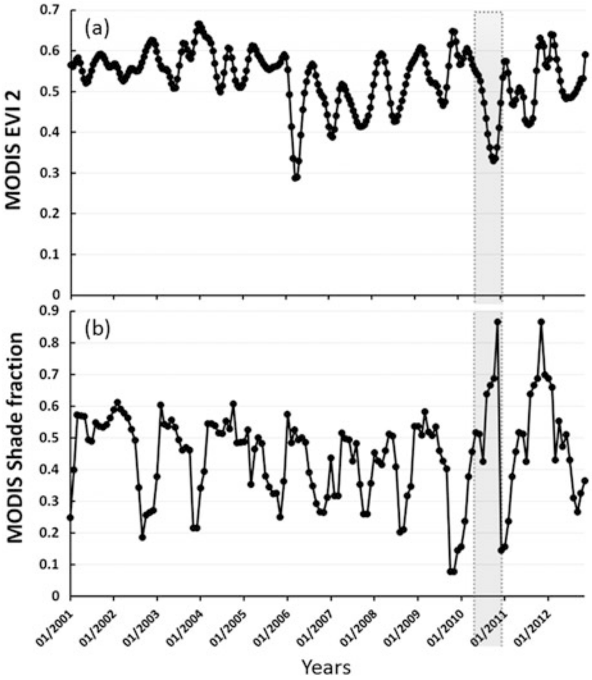

Fig. 13.13 Example of a forest affected by fires in 2010, detected by two-image processing techniques applied to MODIS data in Southern Para State. (**a**) The Enhanced Vegetation Index (EVI) 2, derived from MODIS trajectory in time acquired through the website: http://www.dsr.inpe.br/laf/series/ exhibiting the decrease in the index after 2008, due to the forest degradation probably by logging, and lowest values at the end of the dry season in 2010, after the forest fires. (**b**) Same area as "a" showing the increase in the shade fraction, derived from the unmixing model, due to the burn scar in the forest

diversity in burned plots can enhance the susceptibility of remaining species to herbivory (Massad et al. 2013), which can slow down or impede the long-term recovery of fire-affected forests. So, in regions along the Amazon forest—Cerrado ecotone, fire may be an important factor shaping the boundary between the two 'biomes' (Staver et al. 2011; Silvério et al. 2013).

Fire-induced modifications in the canopy structure and floristic characteristics are precursors of changes in ecosystem processes. In forests submitted to experimental annual fires total net primary productivity (NPP) was reduced by 15 % in years following the fire and autotrophic respiration was reduced by 4 % in comparison with the adjacent intact forest (Rocha et al. 2013). Litter production in these fire-affected forests can be reduced by 50 % (4.3 Mg ha^{-1} $year^{-1}$) in comparison with that in intact forests across Amazonia (Balch et al. 2008). Decomposition rates in areas affected by successive annual fires have tended to be slower (Silveira et al. 2009), resulting, potentially, from a drier microclimate and lower litter moisture (Balch et al. 2008). In contrast, a single low-intensity fire had no such effect (Silveira et al. 2009).

The decline in litter production after successive burns seems to suppress the spread of fires, even when microclimatic conditions are favourable (Balch et al. 2008). However, the inhibition of fire by shortage of fine fuel loads may not be sustained in the long term, as delayed tree mortality (Barlow et al. 2003) can increase fuel availability in subsequent years.

One of the most uncertain components of Amazonian forest fire impacts is the magnitude of short- and long-term carbon emissions and potential implications for CO_2 levels in the atmosphere and consequent global warming. Quantification of carbon emissions from understory forest fires is still lacking, preventing accurate estimates of the real contribution of this component to the global carbon cycle. Recently, van der Werf et al. (2010) estimated for the period between 1997 and 2009 that globally, fires were responsible for an annual mean carbon emission of 2.0 Pg C year^{-1}, with South America contributing 14.5 %. Of this, about 8 % appears to have been associated with forest fires, based on estimates from the Global Fire Emission Dataset (GFED) product for South America.

In years not affected by droughts forest fire emissions in Amazonia are likely to be small, e.g. Alencar et al. (2006) estimated a negligible amount of C emission from forest fires in the BLA varying between 0.001 and 0.011 Pg C for 1995. This, however, changes in drought years. During the El Niño event that occurred in 1997/1998, forest fires in Roraima state alone (around 25 % of the total burned area estimated for BLA) were responsible for emissions of c. 0.03 Pg C (Barbosa and Fearnside 1999). Committed gross forest fire emissions, which include all carbon stocked in the dead biomass associated with the fire event that will be released through decomposition along several years, for the southern part of Amazonia added between 0.024 and 0.165 Pg C to this amount (Alencar et al. 2006). During the 2005 drought mean committed gross forest fire emissions for the states of MT, Rondônia, and Acre (total area of 1,293,515 km^2 or 25 % of BLA) were estimated to be 0.21 (0.04–0.34) Pg C (Aragão et al. 2007b).

Studies on the long-term effect of forest fires in Amazonia are few. Barlow et al. (2003) suggested that mortality, especially from large trees, could increase after 3 years of a fire event, possibly doubling the amount of biomass loss and consequent carbon emissions. To address the many hiatuses in our knowledge, we suggest that future work should concentrate on (1) quantifying short- and long-term carbon dynamics in burned forests; (2) developing systematic mapping of the whole spatial extent of burned forests; (3) estimating the recovery rates of carbon stocks and species composition in forests affected by fire; and (4) better quantifying the burning efficiency for live and dead components of the biomass including charcoal formation rates. Integration of field-based surveys, remote sensing information, and ecological modelling is critically important for progressing towards a more accurate estimate of the contribution of forest fires to the global carbon cycle.

13.3 Modelling Fire Occurrence in Amazonia

Current rates of human-induced environmental changes and climate variability, in addition to predictions regarding the future climate in Amazonia, indicate that the conditions for increased forest fire frequency and propagation have already been established. With this in mind, models for forecasting fire occurrence are critical for quantitatively estimating the alteration in the magnitude and spatio-temporal configuration of fires. In addition, the forecast at the correct scale would allow operational actions for potentially avoiding fire episodes and their consequent impacts on carbon emissions, ecosystem services, and human health.

Models have been developed to predict fire risk for Amazonia, aiming to provide information for preemptive actions and for evaluating potential changes in fire pattern as a response to environmental changes (Cardoso et al. 2003; Nepstad et al. 2004; Alencar et al. 2004; Sismanoglu and Setzer 2005; Silvestrini et al. 2011; Chen et al. 2011). These studies have advanced the understanding of spatial and temporal dynamics of fires.

The Brazilian Centre for Weather Forecasts and Climate Studies (CPTEC/INPE) operates a system to predict daily fire risk in South America (Justino et al. 2002; Sismanoglu and Setzer 2002). Their approach is based on meteorological information about cumulative precipitation, minimum relative humidity, and maximum temperature of the preceding 120 days of the prediction date. In addition, active fire data detected by using AVHRR and MODIS sensors are also used (CPTEC 2013). This is a unique systematic product and probably the most comprehensive to date. Forecasts of fire risk are produced daily and can be consulted at http://www.inpe.br/queimadas/risco.php.

Other approaches exist too. Cardoso et al. (2003) developed a model for evaluating the impact of forest conversion on fire occurrence. Along with climate, variables related to human activities are included to predict contemporary patterns of fire incidence. The authors used active fire data from the non-interpolated Automated Biomass Burning Algorithm (ABBA), based on GOES-8 (Prins et al. (1998), to calibrate the fire risk model as a function of total and minimum precipitation, distance from paved roads, forest cover, and deforestation. The model was applied under two scenarios to estimate fire occurrence within a $2.5° \times 2.5°$ grid cell for the dry seasons of 1995 and 1997: one scenario with moderate deforestation rates, following the deforestation model developed by Laurance et al. (2001a), and another, extreme scenario, where forest conversion to degraded pastures and paved roads are present in each grid cell analysed. The analysis of both cases indicated that the frequency and spatial configuration of fires in Amazonia are susceptible to extensive changes related to agricultural development.

Nepstad et al. (2004) developed the fire susceptibility system RisQue to map the vulnerability of Amazonian forests to fire in response to the 1997/1998 El Niño drought. RisQue derived information about PAW, which the authors considered as the main driver of fire risk, based on information about soil properties, climate, evapotranspiration, and land use (especially selective logging). The resulting monthly maps of fire risk with a spatial resolution of 8 km showed that small declines in rainfall and increases in evapotranspiration could significantly augment fire risk during drought periods.

More recently, Silvestrini et al. (2011) integrated climate (VPD) with land use variables to model mid-twenty-first century fire responses to climate change and land use. Maps based on VPD to forecast fire risk were integrated with an annual probability of anthropogenic fires. Distance to deforested areas, distance to forest, distance to urban areas, distance to roads, elevation, and protected areas were the key variables used to calculate an annual anthropogenic fire probability. Calibration and validation of this model was based on the night-time AVHRR/NOAA-12 active fire data. The model indicated that extremely wet areas in the north-western Amazonia might become vulnerable to the spread of fires under future climate change. Furthermore, fire occurrence might double in Amazonia by 2050 if trends in climate change and deforestation rates were sustained at the levels that prevailed in the early 2000s.

Chen et al. (2011) have recently proposed a model based on the fact that fire intensity is strongly correlated to changes in sea surface temperatures of the Pacific and Atlantic Oceans. However, the use of it as a fire-risk warning system is limited as the best spatial resolution achievable is 5° by 5°, and the information does not allow accurate spatial planning for effective actions to prevent and curtail fire. Importantly, the model does not account for the influence of human-ignition sources, which have been consistently associated with fire occurrence in the region.

All of these studies have produced significant advances in modelling methods and understanding of spatial and temporal dynamics of fires. It is necessary that future models combine relevant climatic, anthropogenic, and biophysical variables that best forecast fires in the Amazon. Large amounts of freely available satellite-derived and geospatial information allow refining operational systems for analysing fire risk in Amazonia. For effective application of these models for supporting activities to restrain fire in Amazonia, the choice of the spatial resolution is critical and must be tailored to Amazonian geographic and political conditions.

13.4 Conclusions

Historically, fire occurrence was rare in Amazonia. However, with the increasing rates of settlement of humans in the region fire became a common feature of the system. Humans provide ignition sources for fire, mainly associated with large-scale deforestation, slash-and-burn, and management of pastures. The combination of the presence of ignition sources with extreme droughts has enhanced the flammability of natural ecosystems in Amazonia in recent years. During the droughts in 1998, 2005, and 2010, vast areas of forest were affected by fires.

The impact of fires on Amazonian ecosystems is large, with changes in forest structure and species composition, carbon stocks, and human health. Fires can also be part of a complex feedback loop that can increase the effects of climate and human-induced environmental changes (Fig. 13.14). The quantification of long-term impacts is still understudied and requires the implementation of permanent field plots associated with the knowledge of the age and intensity of fire-affected areas.

Remote sensing has been a critical tool in accessing the frequency of fire events and the extent of burned areas. Future work must combine field and remote sensing information to produce a more synoptic quantification of the extent of fire impacts. Moreover, operational programmes must be put in place to monitor the long-term

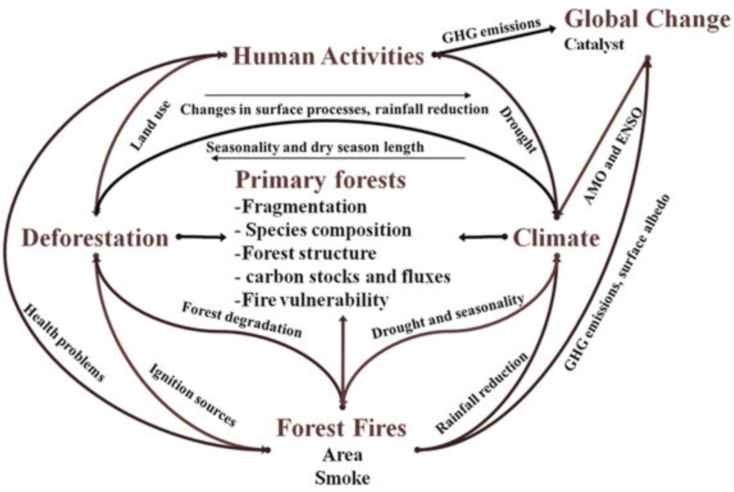

Fig. 13.14 Diagram depicting plausible feedbacks between land use, climate, fire, forests, and humans

impacts of fires in Amazonia. This would be a crucial step towards a better understanding of the resilience of ecosystems to fires in terms of biodiversity, carbon stocks, and ecosystems services. This is essential as climate change may exacerbate drought incidence and intensity in the region. So, understanding the functioning of disturbed systems may provide a clue to the consequences of environmental change on Amazonian forests, critically important for the management of these ecosystems in the future.

Box 1: Challenges for Estimating Burnt Forest Areas in the Satellite Era

Advances in mapping. The detection of burnt areas in Amazonia was first carried out by testing the relationships of fire pixels, detected in the 1-km^2 infrared channels of the Advanced Very High Resolution Radiometer (AVHRR) with the area burnt (Myers 1989; Setzer and Pereira 1991). However, high uncertainty and problems related to the method were detected (Fearnside 1990). Reflectance data from optical sensors (e.g. AVHRR) began to be used later for estimating large-scale burnt areas (e.g. Setzer et al. 1994; Razafimpanilo et al. 1995) and to describe the temporal patterns and trends of fire occurrence in Amazonia (e.g. Prins and Menzel 1994; Holben et al. 1996). Matson and Holben (1987) evaluated the possibility of using the 1-km^2 spatial resolution Normalised Difference Vegetation Index (NDVI) for detecting burnt areas at the subpixel scale. Their study found that burned forests had consistent lower values of NDVI than undisturbed forests.

(continued)

Nelson (1994) mapped areas of forest affected by fires during the El Niño in 1983 in BLA. Burn scars from this fire event were still visible in the Landsat 5/TM image in 1984 and in 1988. Forest reflectance values in the area affected by the fire in 1983 did not recover, and it is likely that reoccurrence of fires, logging, or other causes affected these areas subsequently (Fig. 13.11).

More robust methods of image processing and classification were subsequently developed (Cochrane and Souza 1998; Souza et al. 2003; Matricardi et al. 2010). One of the most used methodologies to separate intact forests from burned forests in Amazonia is based on the linear mixture model using three end members: photosynthetic vegetation (such as green leaves), non-photosynthetic vegetation (NPV), such as exposed tree branches, and shade, given by low reflectance areas in all channels representing shaded areas in the canopy. The NPV fraction image provided adequate means to separate unburned and recently burned forests, but old burned forests were not completely differentiated from intact and recently burned forests (Fig. 13.12). The information provided by the linear mixture model was then combined in the Normalised Difference Fraction Index (NDFI) that allows the detection and mapping of burned forests using Landsat 5 and 7 images (Souza et al. 2005).

The use of multiple dates, taking advantage of the high temporal resolution of MODIS images, brought considerable advances in detecting burned forests. Anderson et al. (2005) and Shimabukuro et al. (2009) have accurately identified burned forests in Amazonia (Fig. 13.13) by analysing the MODIS intra-annual variability of shade fraction images derived from a linear mixture model. Burned pixels exhibit lower reflectance (darker surfaces) and higher proportion of shade which make it possible to separate them from pixels that represent unburnt forest. Recently, Morton et al. (2011) used both intra- and inter-annual mean NDVI, derived from MODIS data to create a burn damage and recovery algorithm (BDR) to separate burned forests from selective logging and deforestation.

Uncertainties. Early estimates of burned area derived from active fire information and retrieved by thermal sensors have several limitations. They overestimate the burned area for the following reasons (Setzer and Pereira 1991): (1) small fires (less than 50 m^2) with high flame temperatures can be detected by the thermal sensor and by converting the fire pixel size (c. 1 km^2) directly to burned area would lead to an overestimation; (2) false fire pixel detection, in sandy areas, rocks, and bare soils that can reach high temperatures; (3) false detection in contrasting surface temperatures (e.g. boundaries between forest and bare ground); and (4) thermal sensor saturation generating false fire detection in regions close to the original fire. Conversely, omissions in fire pixels have also been observed due to (1) cloud coverage,

(continued)

(2) atmospheric attenuation due to smoke derived from fires obscuring detection, (3) low-intensity fire lines occurring over grasslands or pastures areas, and (4) timing of acquisition in relation to the start–end time of the fire (Schroeder et al. 2005, 2008a, b).

Although the methods have improved progressively through time by using reflectance data and series of intra- and inter-annual images, there are still many sources of uncertainties. The main factor affecting optical data is the presence of cloud shades, which produces a spectral signature similar to burned areas (low reflectance, low vegetation index, and high shade values). Although cloud and cloud shades filtering methods are available, if the algorithm is too restrictive, areas can remain with no data over long periods (weeks and months). On the other hand, if the algorithm is too permissive, commission error will occur. Another source of uncertainty is related to the seasonal dynamics of the forest. There is a lack of field data for tracking natural seasonal changes in the canopy structure. These changes, particularly in years of prolonged droughts, may be detected by the sensor of a satellite and may be erroneously interpreted due to the lack of knowledge and field data. For example, MODIS sensor data, which allows multi-temporal procedures for monitoring the forest canopy with medium spatial resolution, have been available only since 2000. Not enough time has passed since to have allowed an adequate evaluation of its use for detecting natural phenomena, such as extreme droughts.

Many global initiatives for mapping burned area started in the early 2000s (e.g. GLOBSCAR project, Global Burned Area GBA-2000, ATSR-2 World Fire Atlas, Global VGT burnt area product—L3JRC, MODIS burned area product—MCD45). However, differences among these products are evident (Simon et al. 2004; Giglio et al. 2005; Jain 2007; Chuvieco et al. 2008; Chang and Song 2009). For example, according to GLOBSCAR, 4333 km^2 of forests were burned in Brazil in the year 2000; GBA-2000 detected 846 km^2 (Tansey et al. 2004). It is expected that in the next years new methods will emerge, as there is a considerable increase in the freely available long-term time series of remotely sensed data.

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