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Environmental change and the carbon balance of Amazonian forests

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ABSTRACT

Extreme climatic events and land-use change are known to influence strongly the current carbon cycle of Amazonia, and have the potential to cause significant global climate impacts. This review intends to evaluate the effects of both climate and anthropogenic perturbations on the carbon balance of the Brazilian Amazon and to understand how they interact with each other. By analysing the outputs of the Intergovernmental Panel for Climate Change (IPCC) Assessment Report 4 (AR4) model ensemble, we demonstrate that Amazonian temperatures and water stress are both likely to increase over the 21st Century. Curbing deforestation in the Brazilian Amazon by 62% in 2010 relative to the 1990s mean decreased the Brazilian Amazon's deforestation contribution to global land use carbon emissions from 17% in the 1990s and early 2000s to 9% by 2010. Carbon sources in Amazonia are likely to be dominated by climatic impacts allied with forest fires (48.3% relative contribution) during extreme droughts. The current net carbon sink (net biome productivity, NBP) of +0.16 (ranging from +0.11 to +0.21) Pg C $year^{-1}$ in the Brazilian Amazon, equivalent to 13.3% of global carbon emissions from land-use change for 2008, can be negated or reversed during drought years [NBP = −0.06 (−0.31 to +0.01) Pg C $year^{-1}$]. Therefore, reducing forest fires, in addition to reducing deforestation, would be an important measure for minimizing future emissions. Conversely, doubling the current area of secondary forests and avoiding additional removal of primary forests would help the Amazonian gross forest sink to offset approximately 42% of global land-use change emissions. We conclude that a few strategic environmental policy measures are likely to strengthen the Amazonian net carbon sink with global implications. Moreover, these actions could increase the resilience of the net carbon sink to future increases in drought frequency.

Key words: carbon emissions, recovery, drought, fire, climate, secondary forests, deforestation.

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I. INTRODUCTION

Formal international efforts to understand climatic changes and to quantify the effects of land use on terrestrial ecosystems started during the late 1980s and early 1990s with the creation of the Intergovernmental Panel on Climate Change (IPCC) in 1988. The IPCC's role in the climate change arena was consolidated by the publication of its first Assessment Report in 1990 (IPCC, 1990), which linked human greenhouse gas emissions to global warming. The International Geosphere Biosphere Project was subsequently created in 1992 to predict and determine how these effects feed back to the atmosphere. In the same year, the first United Nations Conference on Environment and Development (UNCED) was held in Rio de Janeiro, Brazil. On this occasion, world leaders began to set the scene for tackling major environmental and socio-economic issues and for working towards a more sustainable planet. However, it was not until 1994, with the United Nations Framework Convention on Climate Change (UNFCCC), that we started to think about how to manage carbon emissions from terrestrial ecosystems for stabilizing atmospheric greenhouse gases concentrations at safer levels. The 1997 Kyoto Protocol international agreement was the keystone in limiting greenhouse gas emissions.

Within this context, Amazonia has consistently emerged as a vitally important component of the global climate system (Cox *et al*., 2000, 2004; Del Grosso *et al*., 2008; Grace, 2004; Nobre, Sellers & Shukla, 1991). According to the definition proposed by Eva & Huber (2005), this biome (Amazonia *sensu stricto*, the Guiana shields and the Gurupí regions) covers an area of about 5.3 million km^2 of humid lowland undisturbed rainforest, and accounts for 40% of the global tropical forest area. Because of its vast extent, any climate- or human-driven perturbations can have significant impacts on the global carbon cycle. Even small changes in the dynamics of this system may lead to regional or global climatic feedbacks (Nobre *et al*., 1991; Cox *et al*., 2000, 2004) and tipping points in the earth system (Lenton *et al*., 2008; Lenton 2011).

Given the global importance of Amazonia, it is important to demonstrate how climatic changes may impact and interact with human-driven changes and efforts to conserve ecosystems. Among key priority actions defined at the 13th session of the Conference of the Parties (COP13), in December 2007, we can cite mitigating carbon emissions through reducing emissions from deforestation and degradation (REDD) in the tropics (United Nations, 2008), and preserving fundamental ecosystem services, such as atmospheric decarbonisation, water supply and biodiversity. International summits such as the 2012 United Nations Conference on Sustainable Development (Rio + 20) and forthcoming UNFCCC meetings should take advantage of our growing understanding of the processes and feedbacks between Amazonia and the climate system to progress towards integrating climatic, ecological and human associations and their relative importance into climate-change mitigation policies.

To contribute to the global discussion on climate-change impacts, feedbacks and mitigation, this review has a general aim of presenting the state of our knowledge on how climatic changes may affect the Brazilian Amazon carbon budget (Fig. 1) and how these changes interact with anthropogenic perturbations to this ecosystem. We focus on the Brazilian Amazon as it holds approximately 4 million km^2 of forests (Houghton *et al*., 2000), covering approximately 75% of the total Amazonian area.

By reviewing the literature and providing new analyses of key environmental and climate datasets, we provide insights on the vulnerability of this biome to climatic and human-driven changes, and assess which of the main drivers of change can be managed. Our review integrates information across disciplines, which has been pointed as a crucial step to understanding the functioning of Amazonia and its responses to environmental pressures (Barlow *et al*., 2011). We start by presenting a brief overview of the main components of the Amazonian carbon cycle. We then provide a review of the role of undisturbed forests on the carbon balance of the Amazonian biome. Subsequently, we focus on the

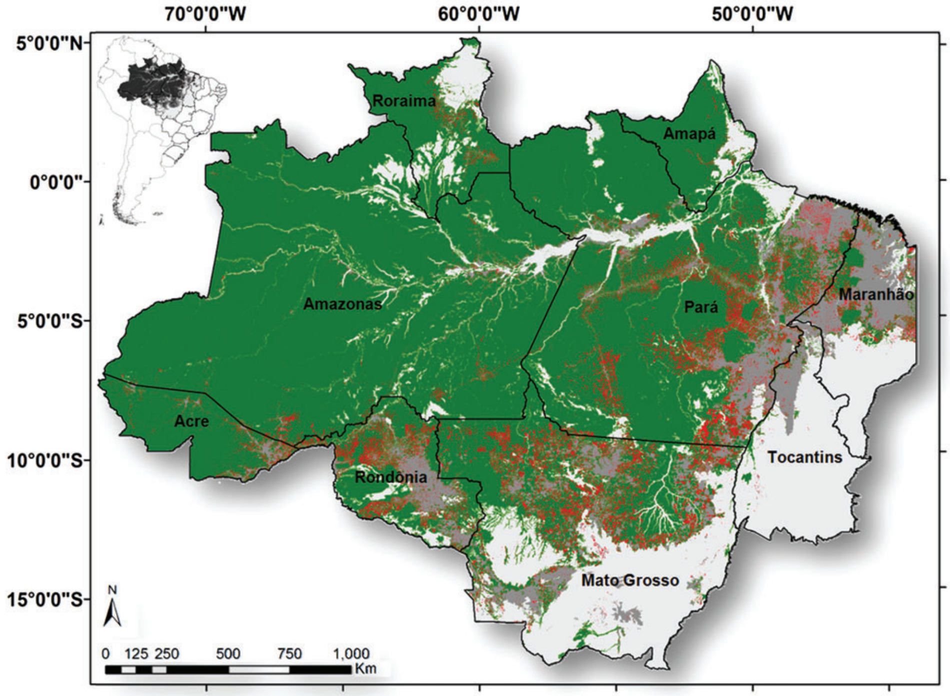

Fig. 1. Map of the studied region showing in green the undisturbed forest area, in red the deforested area from 2000 to 2010, in dark grey the deforested area prior to 2000 and in light grey non-forested areas. The deforestation data are from the INPE/PRODES (2010*b*) project. Inset: a map of South America highlighting the Amazon biome within the boundaries of the Brazilian Legal Amazonia. Black lines indicates the political boundaries of Amazonian states.

two key environmental factors affecting the stability of the Amazonian carbon cycle: climate variability and human activities. This section is intended to provide information on long-term trends in climate and land-cover change, as well as to review the causes and impacts of these two factors upon Amazonian carbon balance. Finally, we compile the numerical information presented in the previous sections to produce an approximated picture of the carbon balance of the Brazilian Amazonia in 2010, accounting for the effects of the multiple components of net biome productivity that control the strength of carbon sinks and sources in this biome.

II. AMAZONIA AND THE GLOBAL CARBON CYCLE

One of the key components of the global carbon cycle is net primary productivity (NPP), defined as the difference between photosynthesis (gross primary productivity, GPP) and autotrophic respiration (R_a). Amazonia alone contributes approximately 14% of all the carbon fixed by the global terrestrial biosphere and explains 66% of interannual variations of global NPP (Zhao & Running, 2010). Global terrestrial annual NPP values vary between 46 Pg C (Del Grosso *et al.*, 2008) and 62.6 Pg C (Grace, 2004).

The Amazonian carbon budget is directly affected by climate-induced extreme events, such as recent Amazonian droughts (Marengo *et al.*, 2011), and land-cover change. These two processes do not operate independently and may even reinforce each other (Cochrane, 2001; Cochrane & Laurance, 2002, 2008; Alencar, Solorzano & Nepstad, 2004; Aragão *et al.*, 2007, 2008; Poulter *et al.*, 2010), exacerbating the single forcing impact.

To incorporate climatic and direct human impacts on the net exchange of carbon between the biosphere and atmosphere in Amazonia, there is a need to describe the carbon budget in terms of net biome productivity (NBP) (Poulter *et al.*, 2010). NBP can be decomposed into its component fluxes (Equation 1) described as:

$$NBP = NEP + D. \quad (1)$$

Net ecosystem productivity (NEP) is defined as NPP minus R_h, where R_h is heterotrophic respiration (CO_2 efflux from the respiration of dead organic matter). In the present context, we consider NEP as the net carbon flux for undisturbed primary forests only, which includes the effects of natural disturbances, such as gap formation. Fluxes for human-driven disturbances are included in the human-induced disturbance term (D). This term can be expressed as:

$$D = -D_F - L - F + R_{DF} + R_L + R_F. \quad (2)$$

D_F, L and F are gross emission terms from deforestation, logging and forest fires (defined as fires that impact closed-canopy forests, without accounting for deforestation fires which are included in the D_F term) and R_{DF}, R_L and R_F are the uptake terms by recovering vegetation following deforestation, logging and forest fires, respectively. Note that all negative terms indicate emissions to the atmosphere (sources) and positive terms indicate uptake from the atmosphere (sinks). For coherence with the terms of the equations, throughout the text, all sources are preceded by a minus sign (−) and sinks by a plus sign (+).

Due to the large temporal and spatial scales that NBP is integrated over, limited accuracy of single NBP terms is a major drawback for estimating the net carbon balance of tropical regions. For Amazonia specifically, apart from deforestation emissions within the Brazilian Legal Amazon (Aguiar *et al.*, 2012), which has a monthly (INPE/DETER Project, 2010*a*) and an annual (INPE/PRODES Project, 2010*b*) wall-to-wall monitoring system, all other terms are highly uncertain. This problem is even more critical in other tropical forest nations.

In the following sections we specifically focus on depicting the causes and consequences of the environmental impacts and on estimating the potential contribution of the specific terms described in Equations 1 and 2 to the Amazonian carbon cycle.

III. THE ROLE OF UNDISTURBED AMAZONIAN FORESTS

Amazonian undisturbed forests are estimated to be a major tropical carbon sink in the global carbon budget (Pan *et al.*, 2011). Quantifying the environmental impacts on the overall carbon balance of Amazonian forests requires information on how NPP and NEP change over space and time (Wright, 2012).

NPP of Amazonian forests tends to be higher in the western than in the eastern portion of the basin (Malhi *et al.*, 2004, 2009*a*; Aragão *et al.*, 2009). This variation is related to higher soil fertility in the younger alluvial soil of western Amazonia in comparison the highly weathered 'nutrient-poor' soils in eastern Amazonia (Quesada *et al.*, 2010). In Amazonia, NPP pattern is strongly related to soil P availability and soil texture, and varies between $+9.3 \pm 1.3$ Mg C ha^{-1} year^{-1}, at a white sand plot, and $+17.0 \pm 1.4$ Mg C ha^{-1} year^{-1} at very fertile anthropogenic 'dark-earth' sites, with an overall average of $+12.8 \pm 0.9$ Mg C ha^{-1} year^{-1} (Aragão *et al.*, 2009; Mercado *et al.*, 2011). The higher NPP of western Amazonian forests indicates that these forests are capable of assimilating and fixing in their biomass more atmospheric carbon over time than the areas in the eastern flank. This east–west differentiation is likely to be reflected in the responses of these forests to environmental change. Understanding this variability is critical to model the basin-wide spatial distribution of carbon uptake and to evaluate how this process can be affected by environmental change in the future.

The most comprehensive estimates of NEP in undisturbed Amazonian forests are based on the RAINFOR network of long-term 1-ha plots (Malhi *et al.*, 2002*a*). This network covers approximately 140 plots spread widely across Amazonia. The latest estimates from this network, with the respective 95% confidence interval (CI) in parentheses, suggested that on average the mature forests of Amazonia have contributed to a sustained sink of +0.89 (+0.65 to +1.12) Mg ha^{-1} year^{-1} or +0.47 (+0.34 to +0.59) Pg C year^{-1} for the whole 5278747 km^2 of undisturbed broadleaf evergreen forests (*Terra Firme* forest) since at least the 1980s (Phillips *et al.*, 2009). Because all other flux values presented herein are for the Brazilian Amazon, we scaled this value down to represent the sink over 3.3×10^6 km^2 for 2010 (Gloor *et al.*, 2012). Thus, the estimated carbon sink in undisturbed *Terra Firme* forest in the Brazilian Amazon in 2010 was +0.30 (+0.22 to +0.37) Pg C year^{-1}. This corresponds to around 6.4% of the global gross land carbon sink of $+4.7 \pm 1.2$ Pg C year^{-1} for 2008 (Le Quéré *et al.*, 2009) or 7.5% of $+4.0 \pm 0.7$ Pg C year^{-1} for the period between 2000 and 2007 (Pan *et al.*, 2011).

The undisturbed forest carbon sink may be slightly higher if we incorporate the productivity of undisturbed flooded forests (*Várzea* and *Igapó* forests). Floodplain forests tend to be more productive than *Terra Firme* forests, with an average net above-ground biomass accumulation of +1.04 Mg C ha^{-1} year^{-1}, with $\pm$ 0.37 Mg C ha^{-1} year^{-1} standard error (Baker *et al.*, 2004*a*). The floodplain area of the Amazon basin covers 350000 km^2 (Richey *et al.*, 2002), and could therefore contribute to an extra net above-ground biomass accumulation of $+0.04 \pm 0.01$ Pg C year^{-1}. The true value for this term, however, remains highly uncertain and its accuracy depends on better quantification of the floodplain area of major Amazonian rivers, better estimates of the variation in forest NEP as a function of stand development stages (stand age) along dynamic meandering western Amazonian rivers, and on understanding the partitioning between above- and below-ground components. In this review, we focus on the terrestrial component of the carbon balance and do not attempt to quantify the magnitude of CO_2 efflux from rivers and the carbon fraction that is transported to the ocean. According to Gloor *et al.* (2012) the balance between the above processes may contribute to a sink of approximately +0.07 Pg C year^{-1}.

IV. FACTORS AFFECTING AMAZONIAN CARBON BUDGET

Environmental change poses an increasing risk for the maintenance of carbon stocks and productivity of Amazonian forests. These changes are controlled by climate events such as droughts, and human activities such as deforestation (slash and burn) and selective logging. However, changes driven by forest fires can be a result of both climate and human activities acting together. Quantifying the magnitude of the effects of climate and direct human activity on the stability of the Amazonian natural carbon sink has been a scientific priority for many multi-disciplinary research groups in recent years. The growing concern about the impact of environmental changes in Amazonia reflects: (*i*) the predictions from previous global climate models (GCMs), indicating increased drought frequency in the region (Li, Fu & Dickinson, 2006) and (*ii*) the recent occurrence of two major droughts in this region in the last decade (Marengo *et al.*, 2011). The severity of the resulting drought impacts over Amazonia, moreover, may be aggravated by synergy with other anthropogenic forcing factors such as deforestation and fires (Cochrane *et al.*, 1999; Laurance & Williamson, 2001; Hutyra *et al.*, 2005; Aragão *et al.*, 2008; Cochrane & Laurance, 2008).

(1) Climatic changes

(*a*) *Long-term climatic trends*

To provide an overview of long-term climatic changes in Amazonia we analysed temperature and rainfall outputs (between the years 1900 and 2100) from the Intergovernmental Panel for Climate Change (IPCC) Assessment Report 4 (AR4) model ensemble (http://www.ipcc-data.org/obs/ar4_obs.html), for three distinct emission scenarios (A2, A1b, B1). As most of the IPCC models tends to underestimate current rainfall in Amazonia (Malhi *et al.*, 2009*b*), we normalized the data with observational data from the Climate Research Unit (CRU) (http://www.ipcc-data.org/obs/ar4_obs.html).

Our analysis of IPCC AR4 annual temperature data for Amazonia shows a positive trend in temperature for all three emission scenarios analysed, with a clear increase during the 21st Century (Fig. 2A). This positive trend in temperature is also evident in the observational CRU dataset since the mid-1970s (Fig. 2A). The rate of decadal increase from 1976 to 1998 based on the CRU climatology was estimated to be 0.26 with $\pm 0.04^{\circ}$C standard error (Malhi & Wright, 2004).

During the 200-year time period evaluated (1901–2100), no trends in mean annual precipitation were observed for this region in any of the scenarios analysed (Fig. 2B). This result is corroborated by the lack of a temporal trend in rainfall observed in the CRU dataset as well as by an analysis of rainfall measured by ground stations in the Brazilian Amazonia; 77% of the stations did not show temporal trends in rainfall (Satyamurty *et al.*, 2010). River data, by contrast, indicated a 20% increase in discharge from 1900 to 2010, which is likely to reflect similar trends in annual mean net precipitation (Gloor *et al.*, 2012). Further evaluation of long-term rainfall trends must be carried out to improve our understanding of future ecological changes in this biome. However, this same dataset shows an increased trend in inter-annual variation of rainfall over the last decades (Gloor *et al.*, 2012), and it is likely that our analysis of rainfall is missing spatially dependent trends, such as the rainfall reduction in eastern Amazonia previously reported for 70% of the 23 IPCC/AR4 models used (Li *et al.*, 2006; Malhi *et al.*, 2008).

It is also important to consider that the mean annual precipitation is a poor metric for understanding the impacts of water shortage on tropical forests. Because of the high

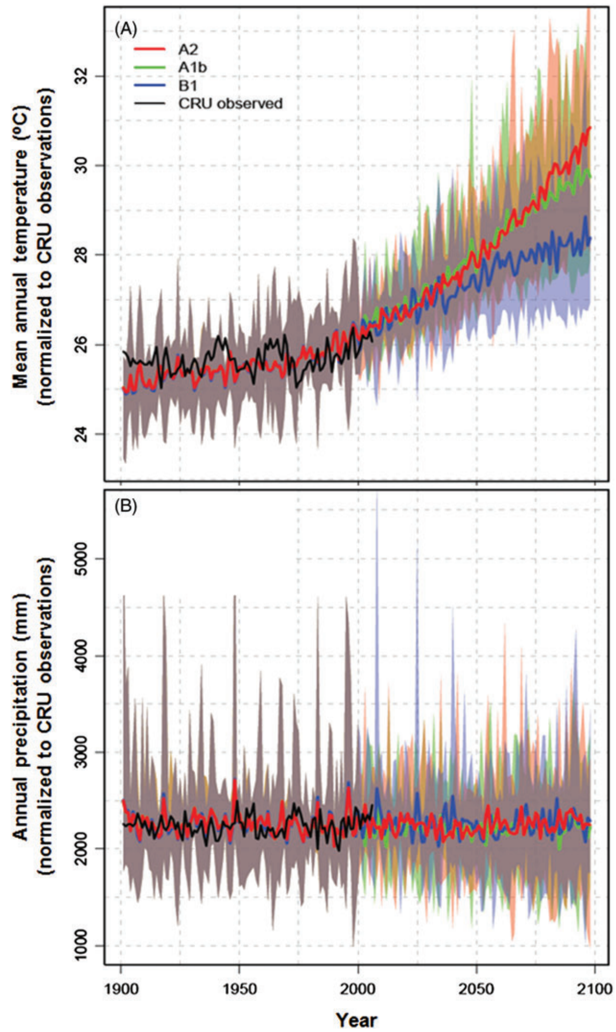

Fig. 2. Predicted changes in mean annual temperature (A) and cumulative annual rainfall (B) based on the outputs of all Intergovernmental Panel Climate Change Fourth Assessment Report (AR4) models (IPCC, 2007). Results are for three different scenarios A2, A1b and B1 in comparison to the Climate Research Unit (CRU) observed climatology (http://www.ipcc-data.org/obs/ar4_obs.html). According to the IPCC AR4, the A2 scenario describes a world with elevated population growth and slow economic development and technological change. A1b scenario assumes a world with very rapid economic growth, a global population that peaks in the mid-21st century and rapid introduction of new and more efficient technologies with a balance across all sources. B1 on the other hand, describes a convergent world, with the same global population as A1b, but with more rapid changes in economic structures towards a service and information economy.

rainfall rates in Amazonia, water deficit and consequent impacts on vegetation will only occur if rainfall input is lower than outputs from evapotranspiration. This negative balance normally takes place during the dry season and cannot be represented by the total annual rainfall or rainfall anomalies. A better representation of forest water stress can be achieved by estimating the maximum cumulative

Table 1. Time series of maximum cumulative water deficit (MCWD) for the Amazon Basin, Eastern Amazonia, and Western Amazonia (values are averages of the full IPCC AR4 GCM ensemble, $N = 32$)

	Amazon Basin			Eastern Amazonia			Western Amazonia		
Decade	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max
1900	−217	−319	−164	−290	−405	−210	−158	−307	−102
1910	−215	−319	−157	−287	−433	−204	−159	−290	−100
1920	−217	−320	−155	−292	−424	−207	−157	−308	−97
1930	−209	−325	−161	−278	−412	−203	−148	−312	−88
1940	−210	−330	−158	−281	−453	−207	−150	−267	−97
1950	−217	−323	−163	−284	−381	−207	−161	−330	−106
1960	−217	−335	−163	−288	−452	−219	−160	−328	−99
1970	−214	−300	−147	−291	−396	−192	−149	−265	−86
1980	−213	−305	−154	−284	−407	−206	−153	−295	−92
1990	−225	−347	−159	−300	−402	−210	−163	−341	−92
2000	−221	−332	−155	−299	−445	−207	−155	−335	−93
2010	−229	−362	−144	−313	−438	−195	−160	−367	−83
2020	−229	−350	−145	−310	−426	−184	−161	−346	−86
2030	−234	−354	−149	−321	−461	−198	−165	−349	−83
2040	−237	−371	−141	−325	−487	−184	−166	−374	−75
2050	−242	−395	−154	−333	−491	−200	−169	−380	−91
2060	−247	−401	−153	−341	−540	−192	−176	−400	−87
2070	−247	−385	−145	−339	−544	−180	−176	−391	−82
2080	−255	−416	−151	−353	−571	−193	−182	−402	−82
2090	−257	−425	−144	−355	−629	−154	−185	−432	−82

Data are in mm and represent the decadal mean, minimum and maximum values. More negative values indicate drier conditions.

water deficit (MCWD), which has been shown to be a reliable descriptor of droughts in Amazonia (Aragão *et al.*, 2007; Malhi *et al.*, 2009*b*; Philips *et al.*, 2009; Anderson *et al.*, 2010).

Based on the IPCC rainfall estimates, we estimated the long-term changes in the MCWD to provide a more comprehensive evaluation of potential shifts in dry-season rainfall. The MCWD corresponds to the most negative value of the accumulated water deficit (WD) reached for a given year. This calculation relies on the assumption that moist tropical canopies transpire at approximately 100 mm month^{-1} at a constant rate. This value is based on the mean (±S.D.) evapotranspiration of 103.4 ± 9.1 mm month^{-1} estimated for different locations and seasons in Amazonia (Shuttleworth, 1989; Malhi *et al.*, 2002*b*; Cox *et al.*, 2004; da Rocha *et al.*, 2004; von Randow *et al.*, 2004; Hutyra *et al.*, 2005). Hence, when precipitation (P) is lower than 100 mm month^{-1} the forest starts experiencing water deficit. The following rule was applied to calculate the cumulative water deficit (WD) for each month (n), with evapotranspiration (E) fixed at 100 mm month^{-1} (Aragão *et al.*, 2007): if $WD_{n-1} - E + P_n < 0$, then $WD_n = WD_{n-1} - E + P_n$; otherwise, $WD_n = 0$.

The MCWD corresponds to the most negative accumulated value of WD among all the months in each one of the years. Note that the 100 mm threshold must be applied with caution as evapotranspiration may increase following an increase in temperature and consequent rise in vapour pressure deficit or alternatively, decrease following an atmospheric CO_2 increase, as a result of increased water-use efficiency (Gloor *et al.*, 2012).

This analysis demonstrated that despite no observed changes in annual precipitation, there is a trend towards increased dry season intensity, as represented by the MCWD (Table 1). The intensification of the dry season can be observed in the basin as a whole, but it is particularly apparent when the eastern and western flanks are analysed separately (Table 1, Fig. 3). This result provides a new perspective on possible climate-change impacts in the region, highlighting not only a high risk of increased droughts in eastern Amazonia, but also in western Amazonia. Western Amazonia has previously been described as a low-risk region in terms of rainfall reduction (Malhi *et al.*, 2008), but these predictions are likely due to limitations in the usage of total rainfall estimates as a descriptor of droughts.

The increases in drought frequency predicted by the GCMs analysed here are associated with changes in global atmospheric circulation patterns (Li *et al.*, 2006). This dry-season rainfall reduction in Amazonia is likely to impact forest functioning (Phillips *et al.*, 2009) and exacerbate the synergistic effects of climate and anthropogenic forcing such as deforestation, edge creation, selective logging and fires (Hutyra *et al.*, 2005; Cochrane & Laurance, 2008), which will be further explored below.

(*b*) *Causes and impacts of climatic extremes*

Anomalies in sea surface temperatures (SSTs) of the tropical Pacific Ocean, related to El Niño Southern Oscillation (ENSO) events, have long been recognized as a main cause of Amazonian droughts. El Niño-related droughts have occurred in 1982/1983, 1986/1987 and 1997/1998

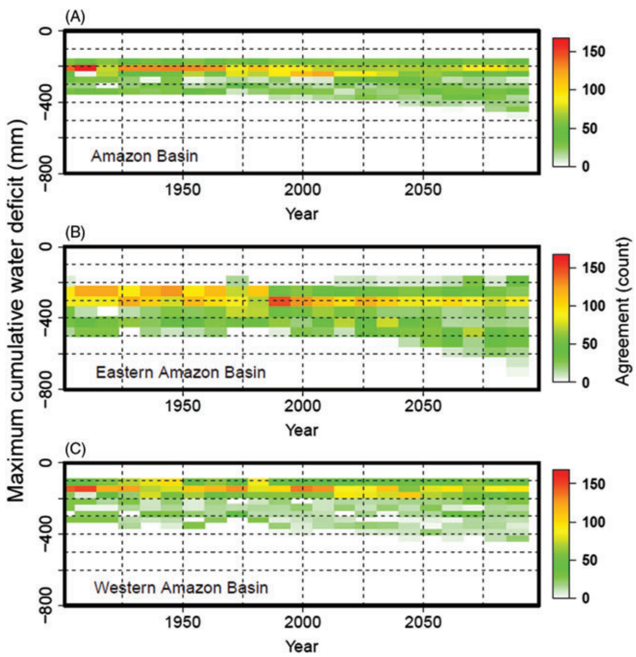

Fig. 3. Model agreement (for the full IPCC AR4 GCM ensemble, $N = 32$) in maximum cumulative water deficit (MCWD) for (A) the Amazon Basin, (B) Eastern Amazonia, and (C) Western Amazonia. Red indicates higher agreement between the model ensemble, and green indicates lower agreement. The count reflects the number of data points in agreement within a 20 year × ~30 mm window. Note that more negative values indicate drier conditions.

(Marengo, 1992, 2004; Uvo *et al.*, 1998; Ronchail *et al.*, 2002). Recent droughts, however, have been associated to tropical north Atlantic SST anomalies, possibly related to the Atlantic Multidecadal Oscillation (AMO) (Li *et al.*, 2006; Marengo *et al.*, 2008). Anomalies related to the AMO have been implicated as a causal factor of the severe 2005 drought that affected Amazonia (Aragão *et al.*, 2007; Marengo *et al.* 2008) and were also one of the principal drivers of the 2010 drought (Marengo *et al.*, 2011). The AMO anomalies are influential in suppressing rainfall in southern and western Amazonia, while ENSO anomalies normally reduce rainfall in north and eastern Amazonia (Marengo *et al.*, 2008; Saatchi *et al.*, 2013).

Drought-induced water stress on intact forests causes a sequence of effects that reduce the overall capacity of the forest system to uptake atmospheric CO_2 and induce tree mortality (Phillips *et al.*, 2010; van der Molen *et al.*, 2011). Drought can directly decrease the photosynthetic capacity of forests by promoting stomatal closure and/or inducing leaf shedding. Stomatal closure can occur either when guard cells respond to changes in relative humidity or vapour pressure deficit, or when hydraulic gradients within the plant, from the root to the atmosphere, become too large due to soil moisture stress. These processes, consequently, will reflect in lower aboveground (Nepstad *et al.*, 2004; Phillips *et al.*, 2009; da Costa *et al.*, 2010) and belowground biomass production (Metcalfe *et al.*, 2008).

Modelling analyses suggests that El Niño-induced droughts may cause reductions in Amazonian NEP ($NEP = NPP - R_h$) by the additive effect of declines in photosynthesis during the drought and subsequent increases in heterotrophic respiration in the following wet season (Tian *et al.*, 1998; Zeng *et al.*, 2008). Based on field observations, long-term impacts on R_h fluxes are expected because of widespread drought-induced tree mortality increasing the decomposing pool (Williamson *et al.*, 2000; Phillips *et al.*, 2009). Phillips *et al.* (2009) estimated that the Amazon was a source of −1.60 (95% CI from −2.63 to −0.83) Pg C as a consequence of the 2005 drought. This was, in part, due to reduced NPP and mostly related to carbon emissions from tree mortality [approximately −1.1 (95% CI from −2.04 to −0.49) Pg C]. It is important to note that these emissions would be spread across approximately 30 years, based on an exponential wood decomposition rate of 0.17 year^{-1}, assuming no moisture limitation (Chambers *et al.*, 2000). Assuming this decomposition rate, the actual annual emission 1 year after the drought would be around −0.18 (95% CI from −0.32 to −0.07) Pg C and would steadily reduce over time. Adjusting this value to be proportional to the area of the Brazilian Amazon we estimate an immediate drought effect of −0.11 (95% CI from −0.20 to −0.04) Pg C year^{-1}.

One important issue is that Chambers *et al.*'s (2000) value does not take into account spatial variation of decomposition rates in Amazonia. Decomposition rates of wood debris are uncertain across Amazonia due to limited data. Higher rates of decomposition were estimated for western Amazonia in relation to the eastern part of the region (Chao *et al.*, 2009) (Table 2). Chao *et al.* (2009) found an average decomposition rate of 0.18 (95% CI from 0.16 to 0.21) year^{-1} for 27 plots across Amazonia. This value is only marginally higher than the value presented in Chambers *et al.* (2000). However, the regional variation presented by Chao *et al.* (2009), which ranges from 0.13 ± 0.003 to 0.32 ± 0.017 year^{-1} (Table 2), suggests that the accuracy of long-term carbon emission estimates from drought-induced tree mortality depend on an explicit consideration of the spatial configuration of the drought.

Drought alone may therefore weaken or reverse the undisturbed Amazonian forest net carbon sink depending on its magnitude, extent and location. However, to elucidate the long-term duration of this effect, more continuous monitoring of these forests is needed. Moreover, reconciliation of ground measurements, ecosystem models and satellite data would allow a better understanding of large-scale and long-term effects of droughts (Saatchi *et al.*, 2013).

(2) Human-induced changes

(*a*) *Long-term land-use and land-cover trends*

Historically in the Brazilian Amazon, deforestation rates, defined here as closed canopy forested areas cleared each year excluding the clear cut of regenerating forests (INPE/PRODES Project, 2010*b*), are determined by three main factors: (*i*) population growth, driven mainly by external

Table 2. Coarse wood debris decomposition rates derived from data presented in Chao *et al.*, 2009 and estimation of carbon emissions 1 year after drought based on the different decomposition rates and assuming tree mortality biomass carbon as presented in Phillips *et al.* (2009)

		Decomposition rates		Emissions 1 year after drought		
Region	*N*	($year^{-1}$)	S.E.	Pg C	Low 95% CI	High 95% CI
NW	2	0.30	0.034	0.29	0.13	0.53
SW	3	0.32	0.017	0.31	0.13	0.56
S	6	0.22	0.015	0.22	0.10	0.40
E	16	0.13	0.003	0.14	0.06	0.25

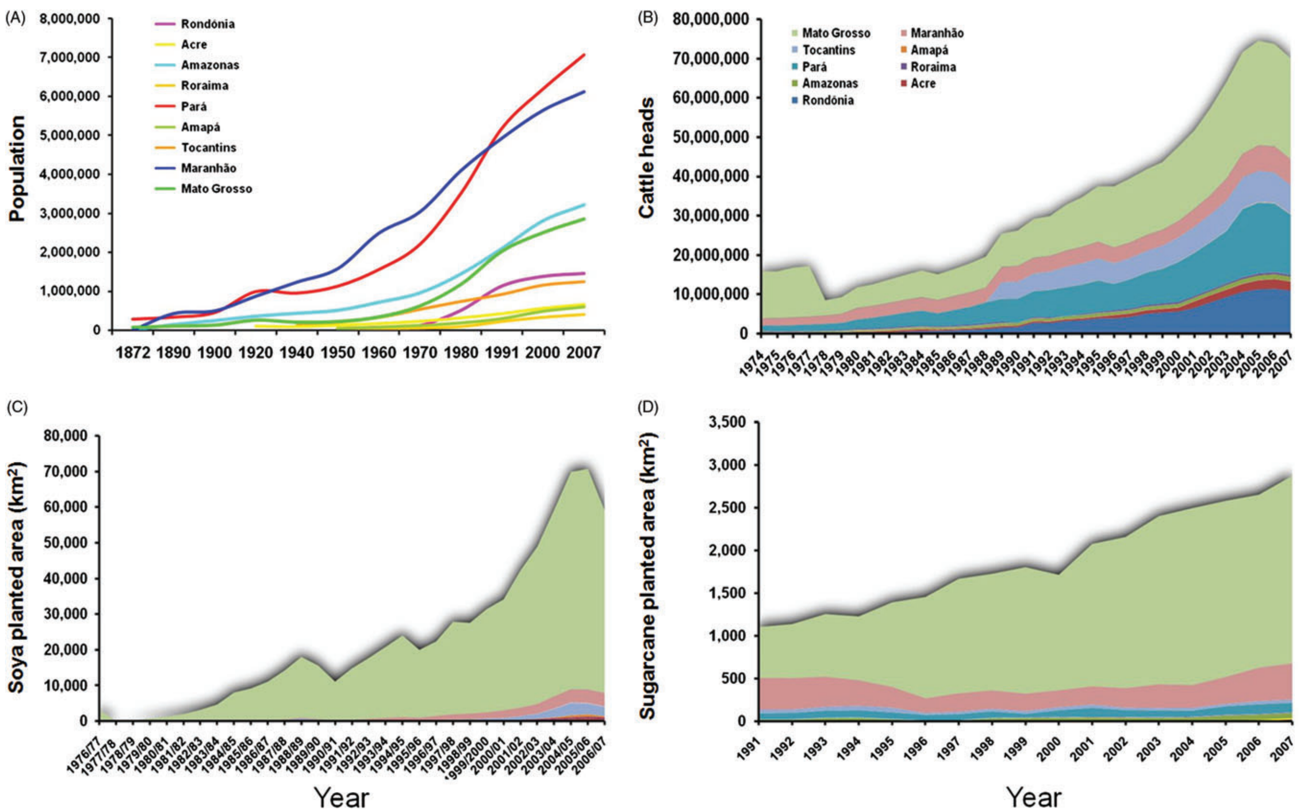

Fig. 4. Evolution over time of Amazonian population (A), cumulative number of cattle heads (B), cumulative planted area of soya beans (C), and cumulative planted area of sugarcane (D). All figures show the values separated by states. Data are from the Brazilian Institute for Geography and Statistics (IBGE - http://www.sidra.ibge.gov.br/).

colonization initiatives, (*ii*) overexploitation of resources and large-scale agricultural expansion focusing on economic growth policies, and (*iii*) lack of economic and technical support for smallholders (Becker, 2005). During the last 135 years, the population in the Brazilian Amazon grew from 336000 people in 1872 to 23600000 in 2007 (IBGE, 2010; Fig. 4A), with 70% currently living in urbanized areas (Padoch *et al.*, 2008).

According to Becker (2005), the population escalation started in the 1960s and was boosted by infrastructure development (e.g. Belém to Brasília and Brasília to Rio Branco roads) and colonization initiatives led by the government. From late 1970s onwards, however, the growth was controlled by financial opportunities related to agribusiness, starting with the rise of cattle ranching (Fig. 4B), followed by the growth of soya plantations, which reached maximum expansion during the mid-2000s (Fig. 4C) (IBGE, 2010). Sugarcane for biofuel production has so far played a minor role in this conversion process, but it is increasing in regions such as Mato Grosso, (Fig. 4D). Carbon emissions from direct use of fossil fuels in Amazonia are likely to be small, even with the growing population. Nobre (2008) estimated an annual emission from fossil fuel sources of −0.5 t C per capita in Brazil. If we consider the size of the human population in Amazonia for 2007, as presented above, we estimate an annual emission of approximately −0.01 Pg C. This value is likely to be even lower as the per capita average used here includes well-developed cities in the south-east of

Table 3. Deforestation rates and cumulative area deforested taken from the INPE/PRODES (2010) project, number of fragments and edge length are based on the relationships developed in this study and the secondary forest age and area are based on Neeff *et al.* (2006) models

	Deforestation				Secondary forest	
Year	Rates ($km^2 year^{-1}$)	Cumulative (km^2)	Fragments (number)	Edge length (km)	Age (years)	Area (km^2)
1976	19705	128651	2601	150704	4.9	46447
1977	19695	148346	2832	158405	4.9	57334
1978	23854	172200	3138	168261	5.0	68730
1979	15507	187707	3355	174995	5.0	75321
1980	19666	207373	3652	183923	5.0	82936
1981	19656	227029	3975	193302	5.0	89857
1982	19646	246674	4326	203154	5.0	96200
1983	19636	266310	4708	213503	5.0	102054
1984	19626	285936	5124	224374	5.1	107489
1985	19616	305552	5576	235792	5.1	112560
1986	19606	325158	6067	247785	5.1	117314
1987	19596	344755	6602	260381	5.1	121786
1988	32745	377500	7603	282875	5.1	128722
1989	23900	401400	8428	300510	5.1	133413
1990	13800	415200	8944	311190	5.1	135997
1991	11200	426400	9386	320135	5.2	138031
1992	13786	440186	9961	331500	5.2	140463
1993	21940	462126	10949	350424	5.2	144181
1994	7852	469978	11326	357456	5.2	145468
1995	27077	497055	12728	382806	5.2	149750
1996	20014	517069	13874	402692	5.2	152767
1997	15028	532097	14803	418300	5.2	154957
1998	19685	551782	16113	439663	5.2	157733
1999	17487	569269	17375	459555	5.2	160118
2000	18226	587727	18813	481528	5.2	162556
2001	18165	605892	20345	504178	5.2	164883
2002	21651	627543	22335	532570	5.2	167566
2003	25396	652939	24919	567918	5.2	170599
2004	27772	680711	28087	609264	5.3	173782
2005	19014	699725	30486	639294	5.3	175888
2006	14286	714011	32422	662827	5.3	177432
2007	11651	725662	34092	682659	5.3	178670
2008	12911	738573	36042	705330	5.3	180017
2009	7464	746037	37221	718778	5.3	180786
2010	6451	752488	38270	730607	5.3	181444

Deforestation rates between 1980 and 1988 were derived from values of cumulative deforested area that were linearly interpolated between 1979 and 1989.

Brazil, which are likely to be the major contributors to this average.

Based on INPE/PRODES data (INPE/PRODES Project, 2010*b*), we estimate that human activities have converted a total of approximately 752000 km^2 (an area about three times the UK surface area) of pristine forests into pastures for cattle ranching and agricultural lands by 2010 (Table 3). This corresponds to approximately 15% of the original area of the Brazilian Amazon. Note that part of the total converted land has regenerated into secondary forests (Fearnside & Guimaraes, 1996; Houghton *et al.*, 2000; Neeff *et al.*, 2006). The cumulative deforestation (D_c), which is equivalent to the total area ever deforested over time, is a result of continuous land clearance with rates decreasing from 21400 $km^2 year^{-1}$ during the 1980s to around 17000 $km^2 year^{-1}$ during the 1990s and 2000s. Since 2004, rates of deforestation have been decreasing steadily and by 2010 a new low deforestation record was achieved (6451 km^2) (INPE/PRODES Project, 2010*b*; Table 3).

(*b*) *Causes and impacts of human activities*

Human activities in Amazonia generate a mosaic of land uses and a gradient of degradation that can be classed as: (*i*) deforestation, (*ii*) fragmentation, (*iii*) forest regrowth, (*iv*) selective logging, and (*v*) forest fires. The impacts of these processes are not mutually exclusive, and in many cases one enhances the impact of others.

(*i*) *Deforestation.* The net emission from deforestation is the balance between the annual carbon released by the deforestation process (D_F) and the annual uptake by regenerating vegetation (R_{DF}) (Houghton *et al.*, 2000). Over the period between 1989 and 1998, Houghton *et al.* (2000) estimated an annual net flux of −0.18 (ranging from −0.10 to −0.26) Pg C year^{-1} for the Brazilian Amazon. This same process was estimated by Defries *et al.* (2002) to release to the atmosphere annually −0.14 (ranging from −0.079 to −0.27) Pg C year^{-1} in the 1980s and −0.26 (ranging from −0.16 to −0.46) Pg C year^{-1} during the 1990s [we have removed the 7% addition to this value used by Defries *et al.* (2002) to represent cryptic processes, such as selective logging, as we discuss these processes separately below]. This latter emission estimate corresponds to 17% of the global land use carbon emissions based on the value of -1.5 ± 0.7 Pg C year^{-1} proposed by Le Quéré *et al.* (2009) for the period 1990–2005. It is important to note that deforestation rates by 2010 have decreased in the Brazilian Amazon by 62% (INPE/PRODES, 2010*b*) relative to the average value between 1990 and 1999, which is likely to lead to a similar decrease in net deforestation emission. Assuming this 62% reduction in the net deforestation emission, we estimate that by 2010 net emissions from deforestation for the Brazilian Amazon have been reduced to around −0.10 (ranging from −0.06 to −0.17) Pg C year^{-1} or 9% of the global land-use carbon emissions proposed by Le Quéré *et al.* (2009) for 2008 (-1.2 ± 0.7 Pg C year^{-1}). The range of values estimated from our simple calculation is comparable to a recent estimate of net emissions for the Brazilian Amazon. Based on the average of different emission models, this study estimated that net deforestation emissions varied between −0.10 and −0.15 Pg C year^{-1} for 2009 (Aguiar *et al.*, 2012). However, directly relating deforestation rates to carbon emissions may underestimate this value. According to Loarie, Asner & Field (2009) the advance of the deforestation frontier into areas of higher biomass densities can add up to −0.04 Pg C year^{-1} to the carbon emissions estimates (based on a 2000–2007 annual deforestation rate), which may explain why our values are at the lower boundaries of the estimates of Aguiar *et al.* (2012).

Using an approximate value for secondary forest regrowth of +0.05 Pg C year^{-1} during the 1990s, based on fig. 4 in Houghton *et al.* (2000), we estimate a gross deforestation flux of −0.31 (ranging from −0.21 to 0.51) Pg C year^{-1} for the 1990s. Using the value of forest regrowth for 2010 of +0.06 (ranging from +0.03 to +0.10) Pg C year^{-1}, estimated in this study (discussed below), we estimated a gross carbon source of −0.16 (ranging from −0.12 to −0.23) Pg C year^{-1} in 2010.

(*ii*) *Fragmentation.* Despite drastic reductions in deforestation rates, many remaining forests continue to be affected by the formation of forest edges and fragmentation. Edge formation and fragmentation are important landscape metrics that have implications for biodiversity and conservation (Laurance *et al.*, 2002; Tabarelli, Lopes & Peres, 2008), including management of anthropogenic fires, abundance of invasive species and disruption of interactions between plants and fauna (Broadbent *et al.*, 2008). To show the temporal dynamics of fragmentation in Amazonia from 1976 to 2010 we developed two exponential models, based on values from 1997 to 2002 published by Broadbent *et al.* (2008), to estimate the length of forest edges (E), and the number of fragments (F), as a function of cumulative deforested area (D_c) for the whole period (INPE/PRODES, 2010*b*). The length of forest edges was estimated based on Equation 3 ($r^2 = 0.90$, $P < 0.05$, $N = 4$).

$$E = 108829e^{-0.0000025Dc} \quad (3)$$

The length of forest edges has increased from 150704 km in 1976 to 730607 km in 2010 (Table 3), with a linear trend in rate of edge formation of around 17650 km year^{-1} ($r^2 = 0.96$, $P < 0.001$, $N = 35$ years).

Based on Equation 4 ($r^2 = 0.92$, $P < 0.05$, $N = 4$) we estimated, moreover, that the number of fragments has increased from 2601 in 1976 to 38270 in 2010, with a linear trend in rate of fragment formation of approximately 1000 new fragments per year ($r^2 = 0.90$, $P < 0.001$, $N = 35$ years) (Table 3).

$$F = 1617e^{-0.0000042Dc} \quad (4)$$

The evaluation of the potential impacts of this extensive fragmentation of the Amazonian landscape still needs further research (Broadbent *et al.*, 2008). However, some of the key aspects of impacts on the carbon cycle will be discussed in Section V.

(*iii*) *Secondary forest regrowth.* Carbon uptake from secondary forest regrowth (R_{DF}) is a key element of carbon budget estimates in tropical forests. In this review, it is related only to areas naturally regenerating over deforested lands and does not include recovery from other disturbances such as drought, fire and logging, which are accounted for here as separate terms. It also excludes plantations for timber and biofuels.

Land use, traditionally, follows a cycle of slash and burn deforestation, cultivation and abandonment and in many cases eventual clearance again. Land-use transitions are considered relatively constant over time in many carbon budget analyses (Fearnside & Guimaraes, 1996; Houghton *et al.*, 2000; Neeff *et al.*, 2006) and hence changes in secondary forest area (SF_a) can be modelled as a function of deforested area (Neeff *et al.*, 2006). To estimate temporal changes in SF_a, we developed a statistical model ($r^2 = 0.82$, $P < 0.01$, $N = 7$, Equation 5), based on published values of SF_a over time (Neeff *et al.*, 2006) as a function of cumulative area deforested in Amazonia at each time interval (INPE/PRODES, 2010*b*).

$$SF_a = 76430 \ln D_c - 852742 \quad (5)$$

We estimated that SF_a has increased from 68730 km^2 in 1978 to 181444 km^2 in 2010 (Table 3). INPE produced an independent SF_a quantification of 198843 km^2 for 2008, based on the classification of Landsat imagery for the whole Brazilian Amazonia (INPE/TerraClass, 2011), which indicates that our estimates underestimate the SF_a

in Amazonia for this same year by approximately 10% ($SF_a = 180017\,km^2$). On the other hand, the Global Forest Resource Assessment 2010 (Food and Agriculture Organization of the United Nations, 2010) reports for the Brazilian Amazonia a SF_a of $232797\,km^2$, based on values reported for 2002. This value is 39% higher than our estimate for the same year. Interestingly, in this same report, the area of secondary forest reported for Colombia was 45% larger than the Brazilian estimates. These high area values may explain, in part, the strong secondary forest contribution to the net carbon budget of South America presented in Pan *et al.* (2011).

The average age of secondary forests in the Brazilian Amazonia varies between 4.4 and 4.8 years old (Neeff *et al.*, 2006). In these young secondary forests, the mean net above-ground carbon accumulation varies between $+1.5$ and $+5.5\,Mg\,C\,ha^{-1}\,year^{-1}$, with a mean of $+3.5\,Mg\,C\,ha^{-1}\,year^{-1}$ (Houghton *et al.*, 2000). However, some studies report a net above-ground carbon accumulation (NEP) of $+5.6\,Mg\,C\,ha^{-1}\,year^{-1}$ for areas up 12 years old (Feldpausch *et al.*, 2004), about an order of magnitude greater than in old-growth forests. Using the range of values of net above-ground carbon accumulation proposed by Houghton *et al.* (2000) multiplied by our secondary forest area, we calculated that the recovery of secondary forests (R_{DF}) contributed to a net carbon sink of $+0.06\,Pg\,C\,year^{-1}$ (with a minimum and maximum range of $+0.02$ and $+0.09\,Pg\,C\,year^{-1}$, respectively) for 1998 and of $+0.06\,Pg\,C\,year^{-1}$ (with a minimum and maximum range of $+0.03$ and $+0.10\,Pg\,C\,year^{-1}$, respectively) for 2010. Houghton *et al.* (2000) estimated the R_{DF} term to be approximately $+0.05\,Pg\,C\,year^{-1}$ by 1998. Wall-to-wall quantification of changes in secondary forest areas is an important next step for improving estimates of carbon sinks in Amazonia. Moreover, an explicit quantification of the annual rates of carbon accumulation in these areas using permanent plots is urgently required.

(*iv*) *Selective logging*. Selective logging (L) is poorly represented in the carbon emission estimates. Selective logging for timber, charcoal production, and fuel wood harvest are all considered forest degradation activities (i.e. any human activity that reduces forest biomass from its potential). This activity was considered to account for 4–7% of total deforestation emissions in previous estimates for Amazonia (Houghton *et al.*, 2000, Defries *et al.*, 2002, respectively). The only large-scale study of the extent of selective logging in the Brazilian Amazon estimated that an area of $19823\,km^2\,year^{-1}$ and $12075\,km^2\,year^{-1}$ were logged in 1999 and 2002, respectively (Asner *et al.*, 2005). The gross carbon emission from this activity averaged $-0.08\,Pg\,C\,year^{-1}$. Carbon emissions from selective logging can be offset by regeneration within these areas after about 40 years (Blanc *et al.*, 2009). The recovery term for the logging component (R_L), hence, seems to be small (around $+0.002\,Pg\,C\,year^{-1}$) in comparison to the regeneration of deforested areas. A fundamental aspect of this process is that selective logging activities are in general related to wood trading in international and Brazilian markets (Foley *et al.*, 2007). As a consequence the direct impact on carbon emissions is likely to be much lower, as the extracted wood is turned into long-lived products, as opposed to deforestation.

(*v*) *Forest fires*. Forest fire emission is another key component of the REDD policy that needs further attention. Natural forest fires in Amazonia are historically rare (Cochrane, 2003; Bush *et al.*, 2007) with a recurrence interval varying between 400 and 700 years during Pre-Columbian time associated to extreme droughts (Meggers, 1994). It is very likely, however, that several wet forests have never been burnt (McMichael *et al.*, 2012). During the last 40 years, the expansion of the agricultural frontier and intensification of land conversion through deforestation and subsequent land-use activities have turned fire frequency into an annual process in Amazonia, prone to leaking to adjacent undisturbed forests during drought years (Aragão *et al.*, 2008). This increase in ignition sources in recent decades exacerbates the risk of forest fires in Amazonia.

Estimates of forest fire emissions (F) for Amazonia specifically are rare. At present, there is a lack of systematic estimates of fire-affected forest areas, uncertainty in combustion fractions of live and dead biomass, and long-term monitoring of post-disturbance dynamics within permanent plots. One previous study suggested that forest fire emissions during non-drought years are small, varying between -0.001 and $-0.011\,Pg\,C$ for 1995 (Alencar, Nepstad & Vera Diaz, 2006). Fire emissions, nevertheless, are exacerbated by droughts. Few estimates have quantified the effect of droughts on this component. Phulpin *et al.* (2002) estimated a total area of $6980\,km^2$ of burnt forests with a corresponding gross emission of $-0.02\,Pg\,C$ for Roraima State, northern Amazonia, during the 1997/1998 severe El Niño event. For southern Amazonia, during the same period, Alencar *et al.* (2006) estimated a total of $26000\,km^2$ of forests affected by fires, with committed gross emissions varying between -0.024 and $-0.165\,Pg\,C$. Long-term recovery of carbon stocks in these areas (R_F) is unknown. These areas may or may not be a long-term source of carbon. Assuming that fire-affected forests have a similar recovery time as selective logged areas, the mean recovery term for the fire component would also be small ($+0.002\,Pg\,C\,year^{-1}$).

Carbon losses from tree mortality in fire-affected forests, however, appear to be cumulative through time, with an increase in large tree mortality 3 years after fire (Barlow *et al.*, 2003). This effect can potentially double current estimates of biomass loss and committed carbon emissions from these low-intensity fires in tropical forests (Barlow *et al.*, 2003). However, a number of studies show that the mortality of above-ground biomass is highly variable across the Amazon (e.g. Barlow *et al.*, 2012). Understanding the variability of short- and long-term dynamics of burned areas and quantifying the spatial extent of burned forests will help to resolve the large uncertainties surrounding forest-fire emission estimates currently available. Estimates of post-disturbance recovery from logging and forest fires are also critically lacking in the literature and are not explicitly included in any previous estimates of net carbon balance in

Amazonia. Despite these uncertainties, the inclusion of this term in future estimates will strengthen the accuracy of the Amazonian carbon budget.

V. SYNERGISMS AND FEEDBACKS

The aforementioned human and climatic drivers of environmental change do not operate independently, and must be integrated in a comprehensive way if we want to predict accurately the future of Amazonian forests. In this section, we describe the main feedbacks and synergistic interactions induced by the processes discussed above.

Drought, forced by changes in SSTs, naturally reduces forest productivity and increases tree mortality and leaf shedding (Nepstad *et al.*, 2004; Phillips *et al.*, 2009). This leads to a positive feedback in fire incidence, because droughts tend to increase stocks of organic matter on the ground, and increase canopy openings (Ray, Nepstad & Moutinho, 2005). The latter favours the amplification of incident radiation reaching the ground, followed by temperature rise within the canopy. As a result, the accumulated organic material on the ground is rapidly dehydrated, increasing the vulnerability of natural forests to fire. The probability of forest fires augments even more if forests exposed to drought are already degraded by edge effects, selective logging, fragmentation, and previous understorey fires (Uhl & Kauffman, 1990; Cochrane & Schulze, 1999; Cochrane *et al.*, 1999; Barlow & Peres, 2004; Nepstad *et al.*, 2004; Ray *et al.*, 2005).

The synergism among fire, deforestation and drought becomes clear when the total annual values of active fires from the Moderate Resolution Imaging Spectroradiometer (MODIS) for the Amazon forest biome (excluding Cerrado areas), within the Legal Brazilian Amazonia boundaries, is plotted against deforestation rate for the period between 2001 and 2010 (INPE/PRODES, 2010*b*). Despite being well established, the association between Amazonian deforestation and fire (Aragão *et al.*, 2008) is prone to change during extreme droughts (Fig. 5).

Droughts may increase the average rate of fire occurrence in relation to non-drought years by a factor of 1.7 (Fig. 5). The average number of fire counts per km^2 of deforested land increases from 0.96 to 2.82 counts km^{-2} from the non-drought to the drought years, respectively. The increase in fire detection during drought years is likely to be associated with increased persistence of land management and slash fires as well as with the leakage of these fires into adjacent forests and other vegetation.

Land management and deforestation fires are usually considered in carbon emission calculations, but fires in the surrounding forest areas are not (Balch *et al.*, 2010). This unquantified component not only impacts estimates of the Amazonian carbon budget, as described in Section IV.2*b.v*, but also affects the permanence of carbon stocks within a REDD context.

The increased fragmentation (Broadbent *et al.*, 2008) and secondarization of Amazonian forests (Table 3) are expected to amplify fire incidence. Aragão & Shimabukuro (2010*b*) quantified an increased trend in fire incidence in approximately 60% of areas with decreased deforestation trends in Amazonia from 1998 to 2007. This pattern has a high probability of being associated with leakage of fires to adjacent forests, fragments and secondary forests (Aragão & Shimabukuro, 2010*a*). One key message is that if REDD policies are implemented, obeying current requirements, the permanence of carbon stocks is not guaranteed, because protected carbon stocks can be removed by fire. Therefore, curbing fire occurrence in the region is a priority.

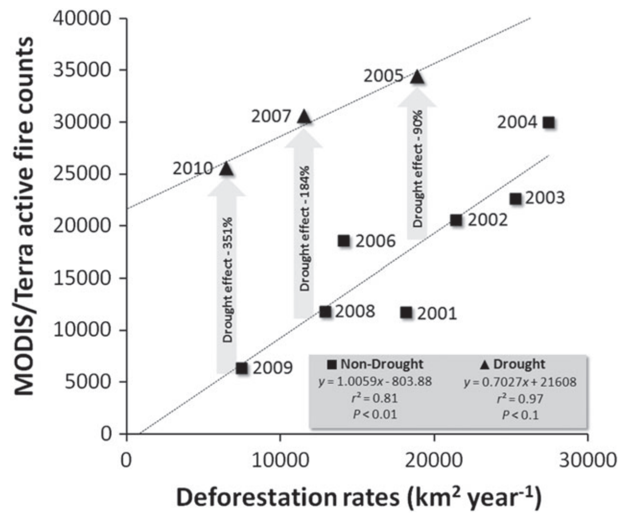

Fig. 5. Relationship between fire incidence and deforestation rates based on MODIS/Terra afternoon active fire detection data for the Legal Brazilian Amazonia. Active fire data represent fire occurrence in the Amazon biome within the boundaries of the Brazilian Legal Amazonia, excluding the Cerrado biome areas and all areas deforested before the year 2000. Triangles and squares represent drought and non-drought years, respectively. The values within the grey arrows represent the per cent change in the number of fires from a non-drought to a drought year. Note that the magnitude of changes increases with a decrease in deforestation rates.

Another important ecological process to be considered in this feedback between land use, drought and fire is that areas affected by successive fires undergo a complete turnover in species composition (Barlow & Peres, 2008). Most of the species colonizing the burnt areas are fast-growing species (Barlow & Peres, 2008), with lower wood density (Baker *et al.*, 2004*b*), and are consequently more susceptible to mortality during droughts (Phillips *et al.*, 2009).

Large areas of deforestation can also catalyse further degradation events in the remaining forest by directly inducing a reduction in local and regional precipitation (Nobre *et al.*, 1991; Laurance & Williamson, 2001; Laurance *et al.*, 2002; Silva Dias, Cohen & Gandu, 2005; Costa *et al.*, 2007; Cochrane & Laurance, 2008; Spracklen, Arnold & Taylor, 2012). The reduction in local precipitation is induced by two non-exclusive mechanisms. First, deforestation changes the physical properties of the surface, leading to a phenomenon known as 'vegetation breeze' (Cochrane &

Laurance, 2008). Clearings tend to increase surface warming leading to increased upward motion of air. This process reduces local air pressure, favouring the drawing of moist air from adjacent forests into the clearing. This air rises up and eventually condenses into convective clouds that can generate rainfall over the opened areas, delivering dry air back over the forest. Second, clearings reduce leaf area index and replace deep-rooted vegetation with shallow-rooted grasses, inducing a direct reduction of evapotranspiration rates (Spracklen *et al.*, 2012). This impact is likely to affect the formation of convective clouds, consequently reducing rainfall.

The deforestation effect may be exacerbated by the presence of smoke from fires in the atmosphere, which is also associated with a reduction in local rainfall (Rosenfeld, 1999; Ackerman *et al.*, 2000; Andreae *et al.*, 2004; Artaxo *et al.*, 2005). According to Andreae *et al.* (2004), the smoke from deforestation fires inhibits surface heating and evaporation, suppressing the formation of convective clouds and rainfall. This smoky air also changes the cloud's microphysics by reducing droplet size in comparison to clean air, which in turn inhibits the onset of precipitation.

Together, land-use and land-cover change as well as climate can create positive feedbacks in which drought occurrence and their impacts stimulate each other in a potentially vicious cycle of intensification. Changes in drought patterns, as predicted by models, and ongoing deforestation and degradation are expected to decrease the resilience of Amazonian forests to further environmental change.

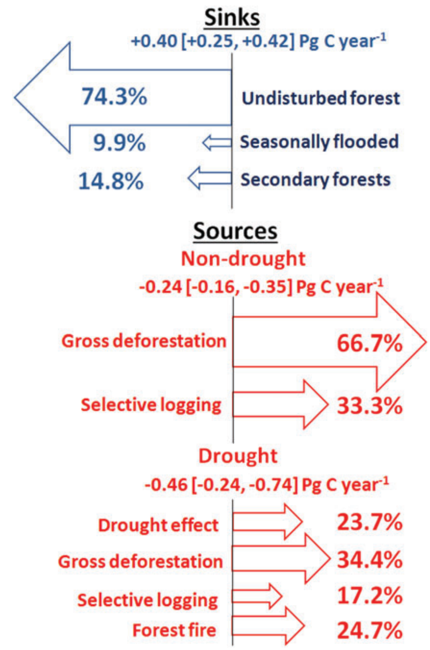

Fig. 6. Proportional contribution of each component to the total carbon sinks (blue) and carbon sources (red). The sources are separated into non-drought years and drought years. The magnitudes of the sink and the source are also given in Pg C year^{-1} with minimum and maximum ranges in parentheses. Note that sinks representing recovering from logging and forest fires, contributing to 1% of the total sink, are not represented with an arrow in the figure.

VI. RECONCILING THE COMPONENTS OF NET BIOME PRODUCTIVITY

In this review, we compared and integrated across recent findings from the literature and analysed available datasets to provide a comprehensive assessment of how climate and humans can influence the balance between major carbon sinks and sources in Amazonia. Sinks and sources of carbon can be estimated by adding their major components, as proposed in Equations 1 and 2. Below, we combine the estimates provided in previous sections, taking the year 2010 as a reference, to quantify the relative contribution of the terms (Fig. 6) and discuss the role of the Brazilian Amazon on the global carbon budget.

(1) Carbon sinks

The undisturbed *Terra Firme* forests of the Brazilian Amazon are estimated to provide an average carbon sink (between 1980 and 2004) of +0.30 (ranging from +0.22 to +0.37) Pg C year^{-1} (Table 4). There are no published estimates of the carbon sink in Amazonia for 2010, so we present a comparison with values for tropical Americas that have been recently published. The undisturbed forest carbon sink in Brazilian Amazonia corresponds to 52% (2000–2005) and 71% (2000–2007) of all undisturbed forest carbon sink in tropical Americas as estimated by Malhi (2010) and Pan *et al.* (2011), respectively. The greater contribution in comparison to the Pan *et al.* (2011) estimate is likely to be due to our use of a pre-drought estimate of net sink. This sink is larger if we consider the contribution of seasonally flooded forests in the overall undisturbed forest carbon sink.

Proportionally, undisturbed *Terra Firme* forests contribute 74.3% of the total Brazilian Amazon carbon sink [+0.40 (ranging from +0.25 to +0.42) Pg C year^{-1}]. Secondary forest regrowth accounts for 14.8%, while all other sinks, including seasonally flooded forests (9.9%) and recovery from logging and forest fire (1%), sum to 10.9% (Fig. 6).

Pan *et al.* (2011) estimated a tropical America secondary forest sink of +0.86 Pg C year^{-1}. Surprisingly, our value for the secondary forest sink in the Brazilian Amazonia is only 6% (+0.06 Pg C year^{-1}) of this value, which is consistent with the criticisms raised by Wright (2012). Our estimate also seems to agree with other published information (Houghton *et al.*, 2000; INPE/TerraClass, 2011) in terms of area, growth rates or carbon sequestration, as discussed previously. On the other hand, Pan *et al.*'s (2011) growth rates are also comparable with the literature; therefore, this implies that the area of secondary forests reported by FRA 2010, used by Pan *et al.* (2011), is responsible for the differences between the studies. We know that the area reported for Brazil in the FRA 2010 is 39% higher than our estimates. This may also explain why the relative importance of the secondary forest

Table 4. Carbon sinks in the Brazilian Amazonia

Carbon sinks		Component	Period	Fluxes Pg C year^{-1} Mean	Uncertainties Pg C year^{-1} Low	High	References	Description
1	Undisturbed forests	NEP	1980–2004	0.30	0.22	0.37	Phillips *et al.* (2009)	Adjusted to Brazilian Amazon area
2	Seasonally flooded forests	NEP	—	0.04	0.03	0.05	This study	Area: Richey *et al.* (2002), NEP: (Baker *et al.* (2004*a*))
3	Secondary forests	R_{DF}	2010	0.06	0.03	0.10	This study	Area: this study; growth rates: Houghton *et al.* (2000)
4	Recovery logging	R_L	—	0.002	0.001	0.003	This study	Estimated assuming 40 years recovery time (Blanc *et al.*, 2009) and uncertainty range of ±50%
5	Recovery forest fire	R_F	—	0.002	0.001	0.003	This study	Estimated assuming 40 years recovery time (Blanc *et al.*, 2009) and uncertainty range of ±50%
6	Total	—	Sum of 1–5	**0.40**	**0.25**	**0.42**	—	—

NEP, net ecosystem productivity.
All values are gross C sinks, except sinks 1 and 2 that include the effect of plot-scale natural disturbances, such as gap formation from tree mortality (not related to drought) and recovery.

sink in relation to the undisturbed forest sink differs when comparing our results with those of Pan *et al.* (2011).

Based on the Lund-Potsdam-Jena Dynamic Global vegetation Model (LPJ), Poulter *et al.* (2010) estimated a comparable Amazonian sink of +0.60 Pg C year^{-1} for a baseline period (2003–2005). This value corresponds to the total area of Amazonia and not only Brazilian Amazonia, and it is, hence, expected to be higher. However, model representation of secondary forest regrowth, multiple forest types, and drought effects (including CO_2 feedbacks on stomatal conductance) need further evaluation. Our total gross carbon sink estimated for the Brazilian Amazonia [+0.40 (+0.25 to +0.42) Pg C year^{-1}], is likely to mitigate around 33.3 (20.8–35.0) % of global emissions from land-use change (−1.2 Pg C year^{-1} for 2008; Le Quéré *et al.*, 2009).

(2) Carbon sources

We estimate an overall gross carbon source during non-drought years of −0.24 (ranging from −0.16 to −0.35) Pg C year^{-1} for the Brazilian Amazonia (Table 5). The greatest single carbon source in our estimate was related to gross deforestation: −0.16 (−0.12 to −0.23) Pg C year^{-1} accounting for (66.7%) of total emissions. During drought years, however, carbon emissions increase by approximately twofold [−0.46 (−0.24 to −0.74) Pg C year^{-1}]. Drought-related carbon emissions from tree mortality and forest fires during extreme drought events become critically important, contributing to 48.3% of the total Brazilian Amazonia emissions (Fig. 6).

Degradation fluxes, through logging and fire, moreover, can be highly important sources. With the increased probability of droughts in the region these two factors together can account for 41.9% of the total gross emissions during drought years, while deforestation is estimated to account for 34.4% (Fig. 6).

Emissions from drought and fire are expected to contribute significantly to the fluxes during extremely dry years; however, deforestation may still be the dominant flux in decadal-scale estimates. We are conservatively reporting drought values estimated from ground measurements for 2005 (Phillips *et al.*, 2009) and assume that this impact was similar for the 2010 drought, despite indications of a larger impact during the latter drought (Lewis *et al.*, 2011). Similarly, our values for fire emissions are based on an analysis for the 1997/1998 drought (Phulpin *et al.*, 2002; Alencar *et al.*, 2006).

The analysis of the relative importance of carbon sources provides useful insights on future emissions. It indicates that if deforestation continues to decrease and approaches zero in the near future (Nepstad *et al.*, 2009), and drought frequency increases as demonstrated in our analysis, disturbance fluxes, such as degradation from drought, fire and logging may maintain relatively high levels of carbon emissions in the region. Gross emissions for non-drought years, including only deforestation and logging, correspond to 20.0% of the 2008 global land-use change emission estimate (Le Quéré *et al.*, 2009), but it is likely to increase to 38.3% relative contribution during droughts, when direct drought effect on vegetation and fires control the fluxes.

Table 5. Carbon sources in the Brazilian Amazonia. Annual drought effect is estimated based on the exponential decay rate (Chambers *et al.*, 2000) applied to the total committed emissions from tree mortality (Phillips *et al.*, 2009) for the first year after the 2005 drought. Gross deforestation emission is the sum of the estimates of net carbon emission from deforestation presented in Defries *et al.* (2002) with the carbon assimilated by regrowth of 0.06 Pg C $year^{-1}$ estimated in this study (sink 3, Table 4) minus 0.02 Pg C $year^{-1}$ (that was originally add to the net carbon emissions to compensate for cryptic disturbances emission, such as selective logging)

Carbon sources		Component	Period	Fluxes Pg C $year^{-1}$ Mean	Uncertainties Pg C $year^{-1}$ Low	High	References	Description
7	Drought effect	—	2005	0.11	0.04	0.20	This study	Chambers *et al.* (2000) exponential decay rates applied to Phillips *et al.* (2009) committed carbon emissions from tree mortality adjusted to the Brazilian Amazon area.
8	Gross deforestation	D_F	2010	0.16	0.12	0.23	This study	Defries *et al.* (2002) values, converted to gross deforestation based on Table 4 estimates of secondary forest regrowth fluxes and adjusted for 62% decrease in deforestation rates
9	Logging	*L*	1999–2002	0.08	0.04	0.12	Asner *et al.*, 2005	Uncertainty range of ±50%
10	Forest fire	*F*	1998	0.11	0.04	0.19	Alencar *et al.* (2006); Phulpin *et al.* (2002)	Sum of the two estimates
	Total drought	—	Sum of 7–10	**0.46**	**0.24**	**0.74**		
	Total non-drought	—	Sum of 8–9	**0.24**	**0.16**	**0.35**		

Poulter *et al.* (2010) estimated a gross carbon source of approximately −0.73 Pg C $year^{-1}$ using the LPJ model for the baseline period (2003–2005) for the whole of Amazonia. Once we adjust the LPJ values to account for the 62% reduction in deforestation, the gross flux of approximately −0.28 Pg C $year^{-1}$ seems to agree with our estimates of gross emissions, including deforestation and logging for non-drought years.

(3) Impact of environmental changes on the net biome productivity of the Amazon

NBP of the Brazilian Amazon in 2010 is estimated to be +0.16 (ranging from +0.11 to +0.21) Pg C $year^{-1}$, indicating a net sink of carbon equivalent to 13.3% of the global carbon emissions from land-use change for 2008 (Le Quéré *et al.*, 2009). Note that for NBP, positive values indicate a net sink and negative values a net source. The strength of this net sink can be related to the large reduction in deforestation rates. During drought years, conversely, the direct impact of drought on tree mortality and fires is estimated to negate the potential of Amazonia to decarbonise the atmosphere, as the carbon sink is overcome by the carbon sources [NBP = −0.06 (+0.01 to −0.31) Pg C $year^{-1}$]. These results indicate that, with the reduction in deforestation rates, Amazonia's important carbon sink is likely to be reversed during extreme droughts. The range of variation (maximum minus minimum) for the carbon sinks and sources during non-drought years is equal to 43 and 82% of the mean values, respectively. This highlights the fact that despite high uncertainty in both components, carbon sources, especially from degradation, are responsible for major uncertainties in the carbon budget of Amazonian forests. This becomes even more critical for drought years, where the variation between minimum and maximum values of the estimated sources increases to 108%. This is related to the uncertainties surrounding the direct impact of drought on tree mortality and the impact of forest fires.

VII. CONCLUSIONS

(1) We provided a comprehensive analysis of the key components affecting the net biome productivity of the Brazilian Amazonia using available data from the literature and novel data analysis.

(2) We demonstrated that the Amazonian climate is likely to experience increases in temperature and water stress in both eastern and western flanks of the area according to the IPCC model ensemble. This result, contrary to previous studies, highlights the vulnerability of western Amazonia to droughts. The impacts of droughts in western Amazonia is likely to intensify in the future as a response to decreasing anthropogenic aerosol levels over the Atlantic Ocean, with consequent increases in sea surface temperature (Cox *et al.*, 2008). Rainfall information from models, however, should be interpreted with caution due to contradictions between models and some field data.

(3) Managing the climate system is difficult; however controlling some of the components that feedback or

respond to its variability is feasible. Curbing deforestation, the primary source of carbon emissions in the Brazilian Amazonia by 2010, contributed to a decrease of $0.15\,\mathrm{Pg\,C\,year^{-1}}$ in comparison to the 1990s deforestation emission values.

(4) Four to five decades of high deforestation rates left behind a legacy of fragmentation, increased forest edges, and degraded forests that are more vulnerable to the direct impacts of climate change and human activities, especially forest fires.

(5) Climatic impacts allied with forest fires are likely to be the dominant flux (48.3% relative contribution) of carbon during extreme droughts in Amazonia. Reducing fires may, hence, be the most important next step for minimizing emissions, avoiding further degradation of forests and restraining dangerous biosphere–atmosphere feedbacks.

(6) Investing in programs that aim to protect existing secondary forests and increasing their area is the logical action to strengthen the forest carbon sink. Doubling the area of these forests would help the Amazonia gross forest sink to offset approximately 42% of global land-use change emissions. The protection and development of new areas can potentially be financed by derivates of the REDD mechanism and will probably be much more efficient after the national level registry of all Amazonian rural properties, when owners can be identified and different land uses and available land areas can be quantified.

(7) With a few strategic environmental measures the current net carbon sink of $+0.16\,\mathrm{Pg\,C\,year^{-1}}$ in the Brazilian Amazonia can become more vigorous, increasing the resilience of its net carbon flux to potential increases in drought frequency. It is clear that a reduction in deforestation rates can minimize the impacts of fire and drought in Amazonia. However, we still need to improve our understanding of the magnitude of disturbance fluxes, as the relative importance of these components for the overall carbon budget of the biome increases with decreased deforestation impact.

(8) Future research must focus on the quantification of long-term spatially explicit impacts of selective logging, drought and fire as well as of forest regrowth. This must be achieved by reconciling field observations with remote sensing data.

(9) A better quantification of carbon sources, especially during droughts, would reduce uncertainties surrounding the carbon balance of tropical regions. Minimizing uncertainties is critical to evaluate accurately the potential impacts of future environmental changes and inform mitigation and adaptation policies.

(10) Systematic monitoring of the most uncertain components of the Amazonian carbon budget is therefore fundamental for improving our capacity to quantify, manage and maintain the ecosystem services provided by Amazonia to the Earth system at local, regional and global scales.

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