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Environmental Research Letters

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Keywords: wildfires, tropical forests, stem mortality, necromass decomposition, combustion, CO_2 uptake, net emissions

Supplementary material for this article is available online

OPEN ACCESS

RECEIVED
31 March 2020

REVISED
22 July 2020

ACCEPTED FOR PUBLICATION
8 September 2020

PUBLISHED
26 October 2020

Abstract

Wildfires in humid tropical forests have become more common in recent years, increasing the rates of tree mortality in forests that have not co-evolved with fire. Estimating carbon emissions from these wildfires is complex. Current approaches rely on estimates of committed emissions based on static emission factors through time and space, yet these emissions cannot be assigned to specific years, and thus are not comparable with other temporally-explicit emission sources. Moreover, committed emissions are gross estimates, whereas the long-term consequences of wildfires require an understanding of net emissions that accounts for post-fire uptake of CO_2. Here, using a 30 year wildfire chronosequence from across the Brazilian Amazon, we calculate net CO_2 emissions from Amazon wildfires by developing statistical models comparing post-fire changes in stem mortality, necromass decomposition and vegetation growth with unburned forest plots sampled at the same time. Over the 30 yr time period, gross emissions from combustion during the fire and subsequent tree mortality and decomposition were equivalent to 126.1 Mg CO_2 ha^{-1} of which 73% (92.4 Mg CO_2 ha^{-1}) resulted from mortality and decomposition. These emissions were only partially offset by forest growth, with an estimated CO_2 uptake of 45.0 Mg ha^{-1}over the same time period. Our analysis allowed us to assign emissions and growth across years, revealing that net annual emissions peak 4 yr after forest fires. At present, Brazil's National Determined Contribution (NDC) for emissions fails to consider forest fires as a significant source, even though these are likely to make a substantial and long-term impact on the net carbon balance of Amazonia. Considering long-term post-fire necromass decomposition and vegetation regrowth is crucial for improving global carbon budget estimates and national greenhouse gases (GHG) inventories for tropical forest countries.

1. Introduction

Wildfires, defined here as uncontrolled understory fires affecting forested areas (Barlow *et al* 2020), were once absent or incredibly rare in humid tropical forests (Cochrane 2003, Mcmichael *et al* 2012). However, since the 1980s they have been growing in prevalence due to increases in deforestation, forest fragmentation, and widespread use of fire in land management (Goldammer and Seibert 1990, Cochrane *et al* 1999, Mouillot and Field 2005). These factors combined with changes in climate, including increased temperatures and drought frequency, heighten fire probability (Fernandes *et al* 2017, Silva Junior *et al* 2019). It is predicted that by 2050, the southern Brazilian Amazon will have 16% of its extent affected by wildfires (Brando *et al* 2020). Wildfires can lead to large changes in species composition and forest structure (Van Nieuwstadt and Sheil 2005, Barlow and Peres 2008, Balch *et al* 2011, Oliveras *et al* 2018), with up to 50% of all trees dying when a forest burns for the first time (Barlow *et al* 2003). This is particularly worrying, as in years of extreme drought, emissions resulting from wildfires can be greater than those from deforestation (Alencar *et al* 2006, Anderson *et al* 2015, Aragão *et al* 2018). Given the recent magnitude of tropical wildfires, refined temporal estimates of their associated emissions are crucial for improving national and global carbon budgets.

Although it is recognized that tropical wildfires can significantly contribute to global climate change (Page *et al* 2002, Nepstad *et al* 2008, Brando *et al* 2020), their carbon emissions remain absent from most national and global-level accounting systems. For example, the official Brazilian System for Registering National greenhouse gases (GHG) Emissions (SIRENE, in Portuguese) and the Brazilian System for Estimating Emissions of GHG (SEEG, in Portuguese) do not account for wildfire-related emissions (MCTIC 2017, Azevedo *et al* 2018). There are two key knowledge gaps. The first is spatial; despite recent advances in remote sensing techniques (e.g. Anderson *et al* 2015, Hawbaker *et al* 2017, Chuvieco *et al* 2018, Reiche *et al* 2018) fire-emission datasets such as the Global Fire Emission Database (GFED) (van der Werf *et al* 2010), still rely on burned area products that can underestimate low-intensity understorey wildfires in closed-canopy forests by up to 11 times (e.g. see Withey *et al* 2018). The second is temporal; most estimates of emissions focus on immediate emissions from combustion (e.g. Withey *et al* 2018) or estimates of committed emissions from mortality (Barlow *et al* 2003, Alencar *et al* 2006, Anderson *et al* 2015), but no studies have yet attempted to quantify the dynamics of post-fire forest carbon fluxes in humid tropical forests. This study addresses this second knowledge gap.

In Amazonia, during wildfire events, the immediate emissions from combustion of leaf litter and woody debris are likely to be dwarfed by the committed emissions resulting from tree mortality and subsequent decomposition. Tree mortality remains above-baseline levels for at least 7 yr after the fires (Silva *et al* 2018). The subsequent decomposition of these dead trees will lead to CO_2 being emitted over decades later (Chambers *et al* 2000). These longer-term emissions could be partially or completely offset throughout a largely unquantified phase of post-fire regeneration, which is initially dominated by pioneers (Barlow and Peres 2008, Berenguer *et al* 2018), but later by slow growth higher wood density tree species (Silva *et al* 2018). Without quantifying these processes, it is not possible to assign CO_2 emissions from wildfires to specific years, limiting our ability to compare emissions resulting from wildfires to those resulting from other sources, such as deforestation. This lack of temporal detail also hinders effective tracking of country-level emissions targets under the Paris commitments. Furthermore, a better understanding of the temporal progression of wildfire-related emissions would allow us to estimate their influence on the airborne fraction of CO_2 in the atmosphere, elucidating previously unknown sources and sinks.

Here, we provide the first evidence-based assessment of the temporal basis of gross and net CO_2 emissions resulting from Amazonian wildfires. We use a unique field-based dataset of trees, palms and lianas in four different regions in the Brazilian Amazon, where stem mortality, growth and recruitment have been assessed since 2009. We focused on CO_2 fluxes resulting from growth and decomposition of woody components, which store the largest Carbon content with the longest residence time in the forest. We develop a novel statistical approach to estimate year-to-year net CO_2 emissions from burned forests. For all four regions, nearby undisturbed forests were considered as our baseline for forest dynamics, allowing us to separate the marginal influence of fires from confounding drought effects, and other variation across sites. We address the following questions: (*i*) What is the temporal pattern of gross CO_2 emissions resulting from fire-induced stem mortality and decomposition? (*ii*) What is the contribution of post-fire stem recruitment and growth to long-term CO_2 uptake? (*iii*) What is the multi-decadal net CO_2 fluxes of burned forests given the relative contribution of combustion and decomposition-related CO_2 emissions and post-fire CO_2 uptake? To answer question (*i*), we have used empirical models (Silva *et al* 2018) to describe post-fire stem mortality rates, incorporating a decomposition constant rate previously estimated for the central Amazon (Chambers *et al* 2000). For question (*ii*), we have used the Chapman-Richard function to model post-fire tree growth and estimate how much CO_2 is taken up by vegetation over time. For question (*iii*), we have used data from questions (*i*) and (*ii*) to model the net CO_2

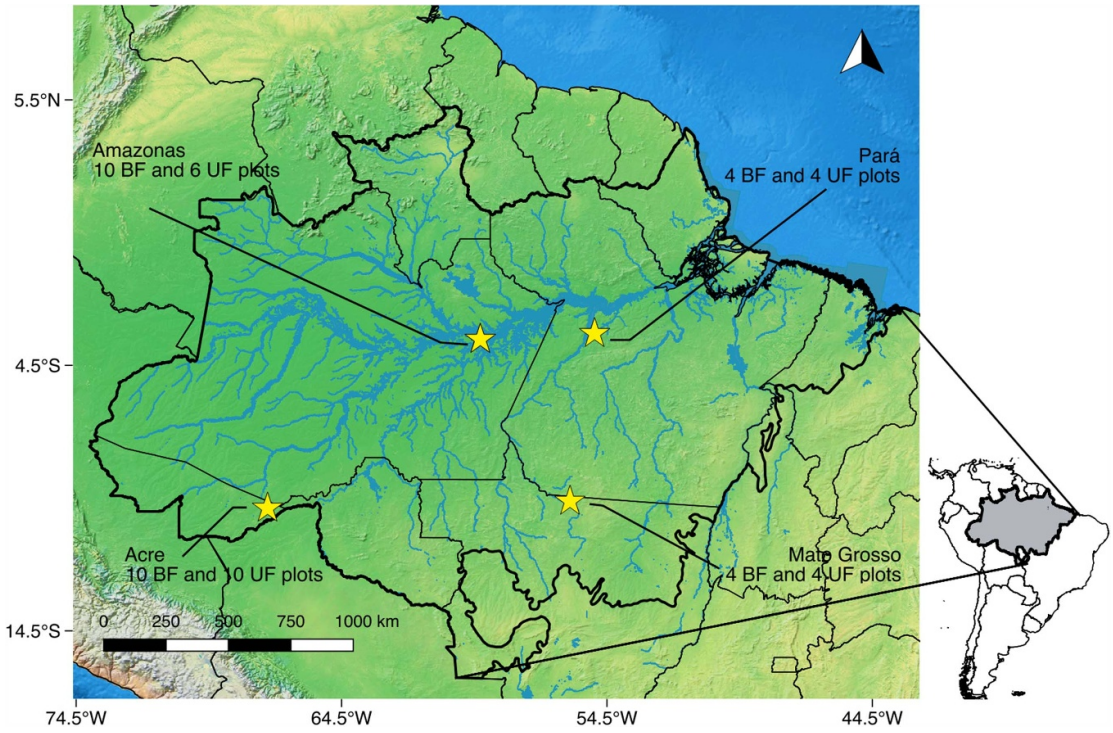

Figure 1. Location of our four regions: Amazonas state where wildfires occurred during the 2015/2016 El Niño and forest censuses started by the same time (2015), Pará state where the latest wildfires occurred in 1998 and the earliest in 1985, and forest censuses started in 2010, Mato Grosso state where wildfires occurred during the 2005 drought and forests censuses started in 2009, and Acre state where wildfires occurred during the 2005 drought and censuses started in 2009. Forest censuses data are archived in ForesPlots.net (Lopez-Gonzalez *et al* 2009).

fluxes following Amazonian wildfires over a 30 yr period and evaluate the model by conducting an uncertainty analysis. Finally, we compare our estimates of decomposition-derived emissions with previous estimates of combustion-related emissions

2. Methodology

2.1. Study region and field measurements

Our dataset was collected in four different regions across the Brazilian Amazonia (figure 1), with permanent plots (0.25 ha) located in both burned (BF, $n = 27$) and unburned (UF, $n = 24$) *terra firme* primary forests (table S.M. 1). Burned forests were only affected by fire once, between 1 and 30 yr prior to sampling. Plots in unburned forests were located near burned ones (1.3–34.6 km) and sampled at the same time. In all plots, we measured all live stems (trees, palms and lianas; 7527) $\geq$10 cm of diameter at 1.3 m height. Aboveground biomass (AGB) were estimated according to Silva *et al* (2018), using specific allometric equations for trees (Chave *et al* 2014), palms (Goodman *et al* 2013) and lianas (Gerwing and Farias 2000). The AGB was estimated for all live stems in the plots, with the use of specific wood density and diameter for trees and only diameter for palms and lianas. We quantified plots AGB growth by adding the AGB of stems recruited with the AGB gain of live stems within censuses. The plot-level AGB losses due to stem mortality were quantified by adding the AGB of all dead stems (downed and standing) within censuses. The number of times each plot was revisited varied (2–6 times), as well as the time interval between censuses (1–4 yr). Corrections at the plot level were applied in order to account for stems recruitment and mortality not measured between censuses, as well as for stem-level growth prior to mortality, following Talbot *et al* (2014).

2.2. Estimating gross CO_2 emissions

2.2.1. Fire-induced aboveground necromass production.

Aboveground necromass production (AGNp, Mg ha^{-1} y^{-1}) is defined as being the same as the annual AGB loss due to stem mortality, from all causes, including downed and standing dead stems. The fire-induced AGNp (fAGNp) is determined by subtracting the AGNp of the control, unburned plots from AGNp of each burned plot after the fires,

$$\mathrm{fAGNp}_{(i)} = \mathrm{AGNp}_{\mathrm{BF}(i)} - \overline{\mathrm{AGNp}_{\mathrm{UF}}} \qquad (1)$$

where $\mathrm{AGNp}_{\mathrm{BF}(i)}$ refers to annual AGNp of the ith plot of burned forest and $\overline{\mathrm{AGNp}_{\mathrm{UF}}}$ refers to the average annual AGNp of all unburned forest plots measured in the same region at the same time of BF plots.

This allows us to exclude the influence from spatial (e.g. soil fertility) and temporal (e.g. droughts) drivers on fAGNp.

We used a non-linear least squares regression and a standard exponential decay function to model fAGNp,

$$\mathrm{fAGNp}_{(t)} = \mathrm{fAGNp}_{(t=0)} \cdot \mathrm{e}^{(-kt)} \quad (2)$$

where t is years since fire, and k is the rate at which fAGNp reduces over time. The regression analysis was done using the *nls* function from the *stats* R package (R CoreTeam 2019), S.M.2.

2.2.2. Aboveground necromass decomposition.

2.2.2.1. Removing combusted necromass from subsequent decomposition emissions in burned forests.

Combustion during understory wildfires removes c. 73% of forest necromass stocks (Withey *et al* 2018). The vast majority of this necromass would have been emitted at a later date during decomposition. To avoid accounting for this loss twice (as both combustion and decomposition), we estimated the decomposition that would have occurred at each year over the 30 yr (figure S.M.3) (available online at stacks.iop.org/ERL/15/114023/mmedia). This estimate was based on published estimates of the combustion completeness of coarse woody debris (CWD), fine woody debris (FWD) and leaf litter stocks in central Amazonia (Withey *et al* 2018). The decomposition of the AGN stocks that were combusted was done by the following equation:

$$\mathrm{cAGNd}_{(t)} = b.\mathrm{cAGN}.\, e^{(-b.t)} \quad (3)$$

where b is a constant decomposition rate estimated for unburned forests in central Amazonia (Chambers *et al* 2000), cAGN is combusted necromass stock that would have been emitted by decomposition, and t years since fire. The removal of cAGNd from total decomposition emissions is demonstrated in next section equation (4).

2.2.2.2. Decomposition of annual necromass inputs.

Over a decadal-time scale, changes in stem mortality mean there is a decreasing amount of aboveground necromass being produced in burned forests (fAGNp). After the first year, the fraction of AGN that decomposed at each year is added with the fractions decomposed in the previous years. Therefore, relative to unburned forests, the total losses by decomposition fAGNd (Mg CO_2 ha^{-1} y^{-1}), occurring in burned forests at a given time t is the sum of all decomposed fractions (present and previous) minus the cAGNd at a given time t (see equation (3)),

$$\mathrm{fAGNd}_{(t)} = \left(\sum_{t=1}^{t}\left(b\,.\,\mathrm{fAGNp}_{(t)}\,.\,\mathrm{e}^{(-b.t)}\right)\right) - \mathrm{cAGNd}_{(t)} \quad (4)$$

with all symbols defined as above. fAGNd was then converted into gross CO_2 emissions, using 0.5 as the biomass to carbon conversion factor (Penman *et al* 2003) and then by multiplying the value obtained by 3.67 (the ratio between C and CO_2 molecular weights) as the CO_2 conversion factor.

2.3. Estimating CO_2 uptake by stem growth and recruitment

Aboveground biomass growth (AGBg) is defined as the annual increment in AGB due to stem growth plus the AGB of recruited stems. Here, to estimate the fire-induced changes to growth rates (fAGBg) we used a similar relationship as equation (1),

$$\mathrm{fAGBg}_{(i)} = \mathrm{AGBg}_{\mathrm{BF}(i)} - \overline{\mathrm{AGBg}_{\mathrm{UF}}} \quad (5)$$

where $\mathrm{AGBg}_{\mathrm{BF}(i)}$ is the annual AGBg of every ith plot of burned forest and $\overline{\mathrm{AGBg}_{\mathrm{UF}}}$ is the average annual AGBg of all unburned forest plots measured in the same region at the same time of BF plots. To model the process of post-fire forest growth according to the pattern observed in fAGBg over the years since fire, we fitted a Chapman-Richard growth function (Richards 1959), which is widely used in forestry to model tree population growth (Pommerening and Muszta 2016). We used a non-linear least squares regression to estimate the function parameters (S.M.2) as per fAGNp. We used the first derivative of this function to model the annual AGB growth rates of forests after the fire,

$$\mathrm{fAGBg}_{(t)} = \mathrm{gmax} \cdot \left(1 - \mathrm{e}^{(-gt)}\right)^{(c-1)} \cdot \left(c \cdot g \cdot \mathrm{e}^{(-gt)}\right) \quad (6)$$

where *gmax* is the maximum growth the forest could reach corresponding to the inflection point of the cumulative function, g is the mean growth rate, c is a nondimensional parameter controlling the curve shape and the location of the inflection point, and other symbols defined as above. fAGBg is converted to CO_2 uptake as per fAGNd. We used CO_2 uptake instead of sequestration as the longevity of this sink remains uncertain.

2.4. Net CO_2 emissions and the relative contribution of combustion

Net CO_2 emissions were calculated by subtracting the modelled CO_2 uptake from the modelled CO_2 gross emissions. We compared the relative contributions of our estimates of net and gross CO_2 emissions (derived just from necromass decomposition) with published estimates of immediate CO_2 emissions deriving from the combustion of CWD, FWD and leaf litter in central Amazonia (Withey *et al* 2018). We used the cumulative values of each emission component to estimate their relative contribution over the 30 yr.

For all analysis, we quantified and propagated uncertainties throughout the model outputs (see supplementary material S.M.4).

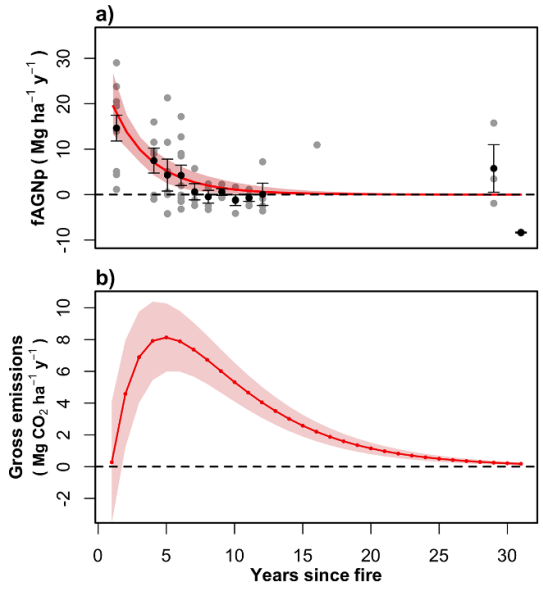

Figure 2. (**a**) Fitted model for predicting fire-induced necromass production (fAGNp). The grey dots are estimated fAGNp from field observations ($n = 61$) derived from the comparison between burned forest plots and locally measured unburned forest plots (equation (1)), black dots with bars are the mean ± standard error, and the red shaded area represents the 95% CI of the model. The dashed black line represents the necromass baseline in undisturbed primary forests. (**b**) Total CO_2 gross emissions (solid line) resulting from the sum of previous and present emissions per year and the subtraction of cAGN.

3. Results

3.1. Temporal pattern of gross CO_2 emissions from fire-induced stem mortality and decomposition

Immediately after fires, necromass production rates increased by 22.4 ± 4.5 Mg ha^{-1} y^{-1} above the levels of unburned forests (>4-fold the UF plots' necromass), and then declined over time at a constant rate of 0.32 ± 0.08 yr^{-1} (figure 2(a); tables S.M.6 and 7). Overall, the nonlinear regressions fit the fAGNp field data well (RSE = 6.63 Mg ha^{-1} y^{-1}; df = 61). The initial necromass stock was the most important parameter for short-term changes in fAGNp, while the contribution from reduction rates ('*k*' in equation (2)) increased over time and was the most important parameter for the long-term changes (figure S.M.8).

One year after fires, the CO_2 emissions from necromass decomposition occurred at the rate of 0.27 ± 1.95 Mg ha^{-1} y^{-1}, as a result of low necromass stocks. New necromass stocks are produced in subsequent years as a result of delayed stem mortality, triggering new decomposition processes that will emit CO_2 (figure S.M.5). Gross CO_2 emissions reached their peak 5 yr after fire (8.13 ± 1.1 Mg CO_2 ha^{-1} y^{-1}), and then decreased over time, approaching the baseline levels 30 yr after the fire event (figure 2(b)).

3.2. Temporal pattern of gross CO_2 uptake due to post-fire recruitment and growth

AGB growth in burned forests increased above baseline levels accumulating the maximum of 22.5 ± 7.41 Mg ha^{-1} in 30 yr. AGBg slowly declined and reached baseline levels between 20 and 25 yr after the fire (figure 3). When burned forests AGBg peaked, CO_2 was taken up at the maximum rate of 5.59 ± 1.33 Mg ha^{-1} y^{-1} (figure 4(a)).

The nonlinear model fit the fAGBg data well (RSE = 2.21 Mg ha^{-1} y^{-1}; df = 50). However, the nondimensional parameter related to the time and size of the growth peak ('*c*' in equation (6)) had the greatest variation (17.9 ± 18.4; tables S.M.6 and 7). All the three parameters in the Chapman-Richard function (*gmax*, *k*, *c* in equation (6)) controlling the forest growth had similar contributions (S.M.9) at the maximum growth (inflection point).

3.3. Multi-decadal net CO_2 flux from burned forests: comparing the contribution of combustion and decomposition-related CO_2 emissions with post-fire CO_2 uptake

The balance between gross emissions and uptake results in net CO_2 emissions that peaked 4 yr after the fire, with the release of 7.51 ± 1.39 Mg CO_2 ha^{-1} y^{-1} to the atmosphere (figure 4(a)). After that, the net CO_2 emissions decline sharply due to increases in CO_2 removals. Net CO_2 emissions converged with baseline levels towards the end of the 30 yr period. However, when we combined our estimates of CO_2 emissions resulting from dead-wood decomposition with those from the combustion of woody debris and leaf litter (33.64 Mg CO_2 ha^{-1}, see Withey *et al* 2018), both cumulative gross and net CO_2 emissions remained above baseline levels (figure 4(b)). We, therefore, estimate a cumulative gross emission of c. 126.1 Mg CO_2 ha^{-1} for 30 yr after a fire event. Cumulative CO_2 uptake only offsets 35% of these emissions (45.0 Mg CO_2 ha^{-1}) within the same timeframe. Decomposition-related emissions account for approximately 58% (47.4 Mg CO_2 ha^{-1}) of total net emissions. The inclusion of net decomposition-related emissions doubles the emission estimates from combustion 7 yr after the fire.

4. Discussion

4.1. Improving emission estimates from Amazonian wildfires

Our approach provides a calibrated method for integrating Amazonian wildfires into national and global emission databases. At present, in humid tropical forests, GFED focuses on emissions from deforestation fires and assumes that wildfires are carbon neutral in the long term, with regrowth offsetting respiration of woody debris and leaf litter (Landry and Matthews 2016). Also, the currently omitted CO_2 emissions and removals, from post-fire stem

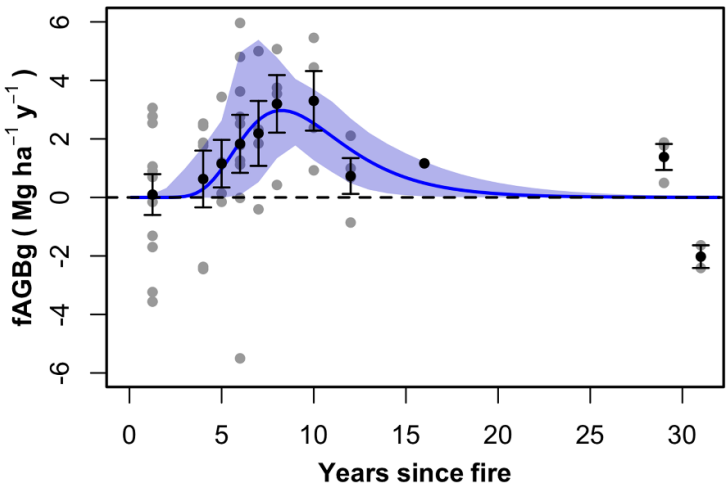

Figure 3. Fitted model (Chapman-Richard) for predicting fire-induced biomass growth (fAGBg). The grey dots are estimated fAGBg from field observations ($n = 50$), derived from the comparison between burned forest plots and locally measured unburned forest plots (equation (5)), black dots and bars are the mean ± standard error, and blue shaded area represent the 95% CI of the model. The dashed black line represents the baseline growth in undisturbed primary forests.

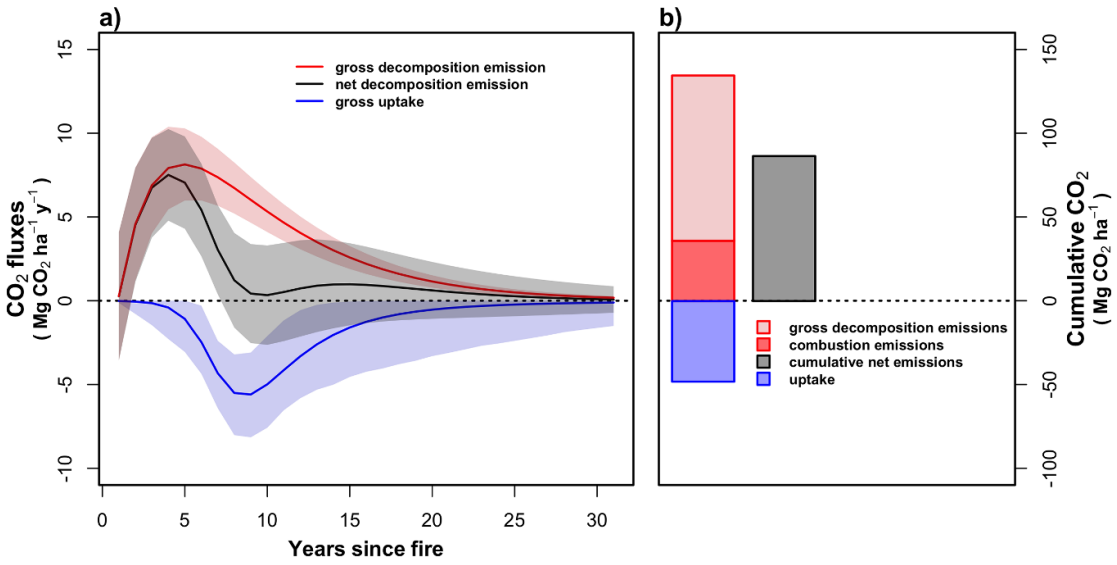

Figure 4. (**a**) CO_2 fluxes (Mg ha^{-1} y^{-1}) from wildfires. Gross emissions (red line) are the total emissions derived from necromass decomposition each year after the burn, CO_2 uptake (blue line) is the CO_2 taken up through above-ground biomass growth, and net CO_2 (black line) is the balance between gross CO_2 emissions and uptake. (**b**) Cumulative CO_2 (Mg ha^{-1}) emissions and uptake over 30 yr. Emissions from combustion (dark red) represent a single emission during the burn while gross decomposition emissions (light red) are the cumulative decomposition from all necromass stocks produced in 30 yr, accounting for 73% of total gross emissions. Uptake (blue) offsets part (35%) of total emissions resulting in above baseline values (81 Mg CO_2 ha^{-1}) of net emissions (grey).

mortality, growth and recruitment, in SIRENE and SEEG could be resolved by employing the approach proposed here. The omission of wildfire-related emissions is important: for example, while the emissions from the 2.6 million hectares of Amazonian forests affected by the 1998 wildfires (Alencar *et al* 2006) would have been ignored by GFED, SEEG and SIRENE, our analysis suggests that this single event would have emitted 0.17–0.25 Pg of CO_2 to the atmosphere by 2030, even without considering subsequent recurrent wildfires or deforestation. These are equivalent to 18%–27% of Brazil's intended contribution in 2030 (i.e. 0.9 Pg of CO_2, see UNFCCC 2016) under the Paris agreement. Following these estimates, the CO_2 emissions resulting from 2010 and 2015–2016 wildfires, if properly accounted for, would have direct implications for Brazil's ability to meet its National Determined Contribution (NDC). Furthermore, as these emissions databases can be used for fire representation in dynamic global vegetation models, the omissions shown here may significantly impact the carbon budget of tropical countries, if complemented by accurate wildfire mapping.

4.2. The importance of avoiding further degradation in burned forests

Across the 30 yr period, burned forests acted as net CO_2 source, and cumulative net emissions were far higher than uptake. The average net annual emissions of burned forests over a 30 yr period were 1.52 Mg CO_2 ha^{-1} y^{-1}, which is approximately 36% of the estimated annual sink of old-growth secondary forests across tropical American rainforests (Suarez *et al* 2019). These long-term positive emissions are supported by the non-recovery of biomass stocks to pre-disturbance levels shown in Silva *et al* (2018). However, despite these emissions, burned forests also remain an important part of any strategy to mitigate carbon losses from degradation. Allowing burned forests to regrow offsets 35% of all decomposition- and combustion-related emissions over the 30 year period, and, unlike secondary forest, does not require expensive tree planting or incur opportunity costs from the abandonment of agricultural land. The protection of burned forests from further disturbances and/or clearance may also offer other important ecosystem services, such as maintenance of hydrological cycling (Brando *et al* 2019), as well as providing habitat for biodiversity—albeit at a lower level than in undisturbed primary forests (Berenguer *et al* 2014, Barlow *et al* 2016, Ferreira & Lennox *et al* 2018, França *et al* 2020). Yet, protecting these forests from clearance has recently become more challenging—since 2012, deforestation rates have risen 16% on average (PRODES 2020) and burned forests are often located at the agricultural frontier where they may be more susceptible to clearance. Likewise, protecting burned forests from further disturbances is far from straightforward, since burned forests are more vulnerable to windstorms (Silvério *et al* 2019) and are increasingly susceptible to repeated fires (Cochrane *et al* 1999, Alencar *et al* 2011, Morton *et al* 2013, Da Silva *et al* 2018), which is likely to be exacerbated by climate change (Fonseca *et al* 2019). If burned forests burn again, the consequences for CO_2 emissions are likely to be far worse. These recurrent fires are often much more intense, leading to much higher levels of tree mortality (Cochrane *et al* 1999, Barlow and Peres 2004), a high turnover of species composition towards pioneer species (Barlow and Peres 2008), and slower rates of post-fire carbon uptake through regrowth (Balch *et al* 2013).

4.3. Quantifiable uncertainties

While we present the first temporal estimate of emissions from Amazonian wildfires, we also recognise that many uncertainties remain. These include particularly the uncertainties associated with the growth parameters, especially relating to the phase when burned forests reach their peak of CO_2 uptake relative to unburned forests, where the confidence intervals were especially high ('*c*' in equation (6); table S.M.7). There are many reasons for such high uncertainty: post-disturbance growth is a complex process, and post-disturbance growth rates are known to vary significantly by species (Berenguer *et al* 2018), across regions (e.g. Poorter *et al* 2016), and can be affected by environmental factors including fire intensity and canopy openness (Balch *et al* 2013, Brando *et al* 2019), or even climate change or climate anomalies (Phillips *et al* 2009, Elias *et al* 2020). Although we tracked mortality over time in our burned plots, additional variability could have stemmed from the lack of samples in forests before they burned (e.g. França *et al* 2016). However, these pre-fire samples are only achievable by chance or through experimental fires, and the data-spread (figures 2 and 3) suggests our field observations are representative of some of the main environmental gradients within Amazonian forests (Johnson *et al* 2016). Finally, temporal limitations in the dataset represent a further source of uncertainty and our estimates of emission and regrowth are highly uncertain beyond 15 yr since fire. Narrowing this uncertainty remains challenging, as many of the sites impacted by 1980s and 1998 El Nino events have either been deforested, selectively logged, and/or burned again (e.g. see Bullock *et al* 2020).

Decomposition rates are also a source of uncertainty. We propagated the decomposition rate uncertainty measured in undisturbed forests, as decomposition rates in burned forests are unknown. Yet, the decomposition rates in burned forests may differ due to (1) drier microclimate brought on by changes in forest structure and canopy openness (Uhl and Kauffman 1990, Barlow and Peres 2008); (2) changes in decomposer community structure, including invertebrates (Ashton *et al* 2019) and microbes, relating to changes in pH and microclimate (Carvalho *et al* 2016); (3) changes in the litter quality, especially as wood density negatively affects decomposition rates in undisturbed forests (Chambers *et al* 2000, Chao *et al* 2009), and at least part of the mortality is related to short-lived lower wood density species that colonise rapidly after fires (Silva *et al* 2018); and (4) stem mode of death, which impacts wood decomposition rates because dead stems standing and suspended from the ground have much slower decomposition rates than downed stems (Gora *et al* 2019). None of these potential drivers of change in decomposition rates has been previously investigated or quantified in burned humid tropical forests.

Although vegetation is the most disturbance sensitive carbon pool in the forest (Berenguer *et al* 2014), uncertainties could also be reduced by evaluating other components of forest carbon cycle. For example, FWD and leaf litter, which corresponds to 34% of total NPP in undisturbed forests (Malhi *et al* 2009), is assumed to decompose at the rate of CWD. This makes the decay time of FWD and litter in our model longer (5 yr) than that expected

(6 months–2 yr; Malhi *et al* 2011), causing a delay in the emissions. Moreover, not all carbon from woody debris and leaf litter is released as CO_2 to the atmosphere; part of it is biologically transformed and locked up in the soil or leached to groundwater. The net dissolved organic carbon (DOC) export from forest soil is, however, a very small component of the forest carbon cycle (0.003%–1.9% of total NPP; Malhi *et al* 2009). While burned forests soil carbon pool does not differ from unburned forests (Berenguer *et al* 2014), increases in DOC may be expected for burned forests. Further carbon release can be also expected through CH_4 emissions from termite's activity and anaerobic decay of wood and litter. However, anaerobic activity increase is unlikely in free-draining *terra-firme* forests where oxygen is not limited, and the production of CH_4 in *terra-firme* forests represents a small component of carbon cycle (0.005%–0.06%; Malhi *et al* 2009), and the sources have not been identified (Do Carmo *et al* 2006). While changes in CH_4 emission due to termites is a possibility, this has not been investigated in burned humid tropical forests.

Another important set of uncertainties go beyond our approach and relate to the spatiotemporal scaling of our results. For example, wildfires are mostly missed by active fire counts and estimates of burned area derived from satellite measures (Anderson *et al* 2015), meaning that we lack a reliable large-scale and historical mapping of fire scar coverage across Amazonia. Furthermore, even if fires are mapped with the use of improved techniques (Morton *et al* 2011, Anderson *et al* 2015, Withey *et al* 2018), pre-fire forest conditions will play an important role in determining fire intensity and mortality (Barlow *et al* 2012, Brando *et al* 2016). Forests that have experienced disturbances from logging or fires prior to the satellite era may harbour large fuel loads, resulting in more intense fires, albeit with lower initial carbon stocks. However, this source of uncertainty may remain unresolved due to the lack of both, on-the-ground and remote sensing data. Mortality is also likely to be higher near forest edges (Brando *et al* 2019), where necromass accumulation is higher.

5. Conclusion

Most estimates of wildfire-related CO_2 emissions account for committed emissions without considering the temporal evolution of stem mortality, the time taken for the subsequent decomposition of dead biomass, and the amount taken up by regrowth. By incorporating long-term field-data on biomass gains and losses, we developed an approach that addresses these knowledge gaps, showing that decomposition-related emissions make a significant contribution to the total CO_2 emitted and are only partially offset (~35%) by post-fire forest regrowth in 30 yr. Our approach allows the scaling-up of the net CO_2 emissions resulting from wildfires across the Amazon basin, providing a way of incorporating them into both national and global carbon budgets and databases. This, however, depends on the enhancement of forest fire detection and mapping.

Acknowledgments

We thank P. Brando, L. Paolucci and A. Alencar for the constructive discussions which helped to improve the manuscript. C.V.J.S acknowledges the Ph.D. scholarship granted from Lancaster University Faculty of Science and the grant from Quality-Related Strategic Priorities Funding (Research England). The authors thank multiple funding agencies for financial support that made the fieldwork possible: CNPq 574008/2008-0, 458022/2013-6, 305054/2016-3, 400640/2012-0, 309247/2016-0, CNPq-CAPES 441659/2016-0 [PELD-RAS], CNPq-PrevFogo-IBAMA 441949/2018-5 [SEM-FLAMA], and 442650/2018-3 [ACREQUEIMADAS], MCIC-CNPq 420254/2018-8 [RESFLORA]; FAPESP 2015/50392-8, 2018/15001-6, 2016/21043-8 [ARBOLES]; Embrapa SEG:02.08.06.005.00; The Nature Conservancy—Brasil; CAPES scholarships, the UK Darwin Initiative 17-023; NERC NE/F01614X/1; NE/G000816/1; NE/F015356/2; NE/l018123/1; NE/K016431/1; NE/P004512/1; NE/N012542/1, H2020-MSCA-RISE (691053-ODYSSEA); the Climate and Biodiversity Initiative of BNP Paribas Foundation (project BIOCLIMATE); and the Inter-American Institute for Global Change Research (IAI) SGP-HW0016 [MAP-FIRES]. We are grateful to all field collaborators, assistants, property owners who contributed to field logistic and data collection. Finally, we thank the Brazilian Research Institutes (INPE, INPA, EMBRAPA, LBA Santarém, ICV) for providing logistical support during the data collection and analysis.

Data availability statement

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

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