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Climate change, human land use and future fires in the Amazon

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Abstract

There is increasing consensus that the global climate will continue to warm over the next century. The biodiversity-rich Amazon forest is a region of growing concern because many global climate model (GCM) scenarios of climate change forecast reduced precipitation and, in some cases, coupled vegetation models predict dieback of the forest. To date, fires have generally been spatially co-located with road networks and associated human land use because almost all fires in this region are anthropogenic in origin. Climate change, if severe enough, could alter this situation, potentially changing the fire regime to one of increased fire frequency and severity for vast portions of the Amazon forest. High moisture contents and dense canopies have historically made Amazonian forests extremely resistant to fire spread. Climate will affect the fire situation in the Amazon directly, through changes in temperature and precipitation, and indirectly, through climate-forced changes in vegetation composition and structure. The frequency of drought will be a prime determinant of both how often forest fires occur and how extensive they become. Fire risk management needs to take into account landscape configuration, land cover types and forest disturbance history as well as climate and weather. Maintaining large blocks of unsettled forest is critical for managing landscape level fire in the Amazon. The Amazon has resisted previous climate changes and should adapt to future climates as well if landscapes can be managed to maintain natural fire regimes in the majority of forest remnants.

Keywords: climate change, deforestation, fragmentation

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Introduction

Global climate change will stress the world's ecosystems in many ways as temperatures, precipitation and other climate variables shift in intensity, range and season. While considerable uncertainty remains about the form and rate of ecosystem responses, climate change can be expected to result in vegetation structure and composition shifts over time. Potential vegetation is strongly linked to regional climate but existing vegetation is often a function of established fire regimes (Bond *et al*., 2005). Human activity can greatly affect regional fire regimes; therefore, knowledge of expected climate changes alone are insufficient for predicting future vegetation responses. For many ecosystems, climate change will be the ultimate cause of vegetation shifts, but wildfire events will be the means of affecting these land cover changes.

The biodiversity-rich Amazon forest is one region of growing concern under scenarios of future climate change. Global climate models (GCMs) indicate a probable mean global warming of 1.8–4.0 °C by 2100 (IPCC, 2007). Regional warming may be much greater, potentially as high as 10 °C in the western Amazon (Cox *et al*., 2004). Several GCMs also predict moderate (IPCC, 2007) to severe (Cox *et al*., 2000) reductions in regional precipitation. High moisture contents and dense canopies have historically made Amazonian forests extremely resistant to fire spread. If the Amazon is now at risk of significant warming and drying, these forests could potentially transition from being highly resistant to fire ignition (Uhl & Kauffman, 1990) to becoming extensively flammable in the near future. However, as wildfire in the Amazon is almost exclusively driven by human activity (UNEP, 2002), anthropogenic land use and land cover

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change must be factored into any fire scenarios. Future fire regimes will be a product of both climate changes and human land management practices.

Climate, weather anomalies and climate change effects

The Amazon has evolved under the auspices of a relatively benign climate but it has not been unchanging. Paleoclimate analyses of Pleistocene drought conditions (cool and dry), which were once thought to have resulted in Amazon dieback, have shown that no dieback occurred. The forest has proven remarkably stable during extensive periods of previous global climate changes (Bush & Silman, 2004; Cowling & Shin, 2006). However, the general consensus of increasingly sophisticated GCMs is that the future climate of the Amazon will be warmer and potentially drier in some regions (IPCC, 2007), with uncertain effects on regional vegetation. In a study of nine GCM climate change forecasts under different future emissions scenarios, the Amazon was found to be among the regions most likely to experience 'novel' climate conditions by the end of the 21st century (Williams *et al*., 2007). The novel conditions were a result of higher temperatures than are currently experienced in this region, with potential loss of some climates from mountainous portions of the Amazon.

All global climate models include the Amazon by definition. However, an influential GCM (HadCM3) study by Cox *et al*. (2000) galvanized attention on the Amazon by predicting that future climate change could cause rapid dieback of much of the forest by 2050. Using an innovative composite model that coupled a GCM to an ocean carbon-cycle model and a dynamic global vegetation model (DGVM), they predicted that carbon-cycle feedbacks could exacerbate atmospheric CO_2 levels in the IPCC standard IS92a scenario (980 ppmv vs. 700 ppmv). This would result in an additional 1.5 °C (5.5 °C vs. 4 °C) increase in global-mean warming by 2100. In the Amazon, temperature increases (>9 °C) and much reduced annual precipitation (64%) forced a simulated dieback of the Amazon forest after 2050. By 2100, modeled climate conditions are so extreme that over 50% of the Amazon is expected to be bare ground (i.e. a desert) (Cox *et al*., 2004).

Given that precipitation recycling of evapotranspiration from Amazon forests is responsible for 25–50% of the region's rainfall (Eltahir & Bras, 1996; Li & Fu, 2004), the modeled dieback creates a positive feedback of ever lower rainfall as tree cover is reduced. This causes a domino effect of rapid forest retreat from the northeastern coast south and west toward the Andes, reducing modeled forest cover from 80% to 10% in the Amazon (Betts *et al*., 2004). By way of comparison, Salazar *et al*. (2007) forced a potential vegetation model with climate scenarios from 15 GCMs and estimated a 9% reduction in forest cover during the same time period and similar emission scenario.

If forest dieback were to occur, it is likely that the Amazon's fire regime would become one of extensive fires with extreme fire behavior until the affected forests were replaced with more fire-resilient or fire-adapted vegetation types. Before disappearance though, drought stress would make many Amazonian forests susceptible to burning every year. Recurrent fires would severely degrade forests and cause rapid unintentional deforestation. The permeability of the landscape to fire spread would likely devastate much of the region's agriculture. The El Niño fires of 1997–1998 in Roraima, Brazil, resulted in destruction of 80% of the state's staple crops. Similar fires in the state of Pará ran rampant, destroying fruit trees, pepper plantations, rubber tree plantations and sustainable forestry projects as well as forage, cattle and fencing. The smoke pall during the burning season resulted in weeks to months of airport closures in many Amazonian cities and caused numerous respiratory problems for regional populations (UNEP, 2002). With each fire's occurrence, future control of landscape level burning becomes more difficult (Cochrane, 2003). Without major changes in land use and fire management, fires of unprecedented scale and severity could occur. The dieback of the Amazon, outlined by Cox *et al*. (2000), would be preceded by years with thousands of wildfires burning millions of hectares of forest. Whole regions of previously damaged forests could be subjected to the cascading effects of many independent fires (Peters *et al*., 2004), leading to mass fire conditions (Quintere, 1993). These so-called firestorms create their own weather with hurricane force winds being sucked into the flames and can have chaotic fire spread over large regions (Cochrane, 2003). The devastation from such fires would have global climate consequences. Such apocalyptic scenarios of environmental devastation make it necessary to critically examine the climate model simulation driving speculation about a massive dieback of the Amazon.

Modeled vs. real climate

One obvious reasonableness check of model simulations is to evaluate of how well they recreate current climate conditions. Cox *et al*. (2000) predict substantially reduced precipitation in the Amazon. However, current period Amazonian precipitation amounts modeled by the HadCM3 GCM are inaccurate. Before initiating future climate change simulations, the HadCM3 simulations of annual Amazon precipitation are 25% too low (Huntingford *et al*., 2004). This underestimate is not spatially uniform. The northeastern Amazon, which has some of the highest rainfall rates in the world

(3000 mm yr^{-1}), is modeled as receiving only 700–1000 mm yr^{-1} (Gandu *et al*., 2004). These annual precipitation estimates are 60–70% too low (1800 mm yr^{-1}) (Cox *et al*., 2004) before climate change is taken into account. Actual rainfall data from 1960 to 1998 show no significant change in annual rainfall amounts across the Amazon. The eastern Amazon, thought to be most at risk of large precipitation reductions, has gotten steadily wetter (Malhi & Wright, 2004). A comparison of future Amazonian climate change in 21 GCMs showed only seven with reduced precipitation, while 13 predicted rainfall increases. The HadCM3 models were the most extreme, predicting Amazon rainfall reductions twice as large as any other models (Covey *et al*., 2003; Cox *et al*., 2004).

The seasonality implied by the HadCM3 GCM raises additional concerns. Although it does a good job of recreating the spatial pattern of dry-season-length-differences across the basin, the model overpredicts dry season length by approximately 1 month and substantially underpredicts monthly dry season rainfall. Modeled rainfall is near zero for June–August and is 1–2 mm day^{-1} lower than gauge observations in May (Li *et al*., 2006). In short, HadCM3 predicts longer and more intense dry seasons than actually exist in the Amazon for the current climate.

Establishment of a perpetual El Niño state is the driving force behind the HadCM3 predictions of future Amazonian rainfall reductions. Reduced precipitation and higher temperatures are calculated to reduce photosynthesis and increase plant respiration costs to the point that net primary productivity (NPP) becomes negative, initiating rapid tree death. Therefore, climate change remains a concern in the Amazon because dry season lengths during El Niño are predicted to increase by 1–4 months across the basin (Li *et al*., 2006). In the Amazon, 90% of forest burning may occur in El Niño years (Cochrane *et al*., 1999; Alencar *et al*., 2006). One way of evaluating the reliability of GCM predictions of future El Niño events is by examining how well historical ENSO variability (i.e. timing of El Niño and La Niña periods) is modeled. The Coupled Model Intercomparison Project (CMIP) looked at the realism of 20 GCM (including HadCM3) representations of ENSO variability. While some models predicted changes in climate to either a mean El Niño or La Niña state, most showed little or no change in ENSO variability. The CMIP results showed that the models that predicted the largest ENSO-like climate changes (El Niño or La Niña climate state) were the poorest at simulating historical ENSO variability. The most likely future ENSO climate scenario was concluded to be similar to today's climate, with no trend towards either mean El Niño or La Niña climate states (Collins, 2005), which is also the position of the IPCC (2007).

Modeled vs. real vegetation distribution

Coupling the Top-down Representation of Interactive Foliage and Flora Including Dynamics (TRIFFID) DGVM to the HadCM3 GCM was a significant innovation (Cox *et al*., 2000). Fully coupling the GCM and DGVM models better represents carbon-cycle feedbacks between model components. TRIFFID reduces the world's vegetation to five generic plant functional types (PFTs), specifically, broadleaf tree, needleleaf tree, C3 grass, C4 grass and shrub (Cox *et al*., 2000). PFTs compete against one another following a simple tree–shrub–grass dominance hierarchy (Betts *et al*., 2004). The entire Amazon forest is classified as being a single type, broadleaved forest.

Northeast Amazonian vegetation is simulated by Cox *et al*. (2000) to be dominated by 50 million hectares of grass in areas that are currently broadleaved forest due to the model's erroneously dry regional climate conditions (Betts *et al*., 2004). The expediency of the top-down DGVM approach is adequate for global modeling of vegetation effects on climate, as vegetation acts upon climate collectively. However, the converse is not true. Climate changes result in individual responses by plants that vary as functions of species, age, soils, topography, microclimate and a host of other factors (Korner, 2000). The crude simplifying assumptions for global modeling make regional estimation of vegetation responses tenuous at best.

Regional climate models (RCMs) have much higher spatial resolutions that can better represent local topography, geographic features and land cover changes at this scale (Cook & Vizy, 2006). Regional modeling studies have shown the Amazon to be highly resistant to climate change, with little vegetation change for reductions in rainfall as high as 60% (Cowling & Shin, 2006) and low variability in vegetation assemblages whether the ecosystems gained or lost carbon (Levi *et al*., 2004). In intercomparisons, most models with dynamic vegetation components fail to predict an Amazon dieback (Friedlingstein *et al*., 2006). Even the fully coupled HadCM3LC GCM, as developed by Cox *et al*. (2000), shows no evidence of future Amazon dieback if the climate change simulation is initiated with current climate conditions (Huntingford *et al*., 2004).

Modeled and real responses of vegetation to drought and temperature increases

Drought is a potential threat to tropical forests. Extreme drought conditions during the El Niño of 1997–98 are known to have increased mortality of large emergent trees on upland slopes in Indonesia. Similar findings have recently come from a simulated drought experiment in the Amazon where, after 3.2 years of 60% reduction in wet season rainfall, mortality of emergent canopy trees increased substantially (Nepstad *et al*., 2007). Although large trees are impacted by severe drought, smaller trees appear to be more resistant.

The constraints of the TRIFFID DGVM mandate that the Amazon responds as a monoculture of constant height and universal rooting depth (3 m). A key ecophysiological driver of Amazon dieback is widespread depletion of soil water. This effectively shuts off photosynthesis, making the net carbon balance negative. TRIFFID has no capacity to emulate the drought deciduous behavior of many tropical trees (Cox, 2001), so severely drought stressed trees are all assumed to die. The Amazon basin is a vast network of rivers. Discharging 20% of the globe's riverine fresh water, the Amazon has over 1000 tributaries. A large percentage of the forest has direct access to groundwater and is therefore not adequately modeled by fixed soil water limits.

In the more extensive upland *terra firme* forests where the model could apply, forests have been shown to access much deeper soils (>10 m; Nepstad *et al*., 1994). A blanket 3 m rooting depth is a poor analogue for Amazonian forests, which extend rooting depths in response to dropping water table depths (Jipp *et al*., 1998) and derive 75% of soil water uptake from depths below 2 m during the dry season (Nepstad *et al*., 1994). Through direct comparison with eight flux towers in the Amazon, Hasler & Avissar (2007) show that current GCMs and RCMs overestimate dry season water stress. Ichii *et al*. (2007) constrained rooting depths of Amazonian forests using the BIOME-BGC terrestrial ecosystem model and satellite imagery (MODIS). Using observed weather data, they modeled gross primary production and varied rooting depth model parameters to more closely match observed patterns of the MODIS enhanced vegetation index (EVI). They found that areas experiencing dry seasons of 1–2, 3–4 and 5–6 months required soil depths of 1–3, 3–5 and 5–10 m, respectively. In 2005, an extensive drought, one of the most extreme on record, did not cause trees in the western Amazon to reduce leaf area, in fact the trees increased their effective leaf area while rivers literally ran dry. This indicates that the vegetation was never water limited and instead responded as if it were normally light limited by clouds. The authors suggest that Amazon forests are more resilient than many ecosystem models assume (Saleska *et al*., 2007).

Land cover change and climate

Land cover changes can affect climate through several processes. Standard land cover change scenarios in the Amazon (IPCC, 2007) may have little or no global effect on climate but may have substantial regional impact, potentially raising Amazonian temperatures 2 °C by 2100 (Feddema *et al*., 2005). Numerous studies have investigated the hydrological effects of complete deforestation of the Amazon. These studies indicate much reduced evapotranspiration and diminished precipitation recycling (*e.g*. Shukla *et al*., 1990; Nobre *et al*., 1991). Because recycling of evapotranspiration is responsible for 25–50% of regional precipitation (Eltahir & Bras, 1996; Li & Fu, 2004), even partial deforestation reduces evapotranspiration and is expected to reduce regional precipitation (Salati & Nobre, 1991; Costa & Foley, 2000). However, recent indications are that regional rainfall might actually increase over deforested areas due to local convection processes (Da Silva & Avissar, 2006). These effects may be scale dependent, requiring large amounts of heterogeneous regional deforestation (D'Almeida *et al*., 2006). The net effect on precipitation will be a combination of greenhouse gas-related climate change and land cover effects (Johnson & Cochrane, 2003). A model combining these factors showed Amazonian precipitation increasing by 0.28 mm day^{-1} for a doubling of atmospheric CO_2, however, deforestation simultaneously reduced average precipitation levels by 0.73 mm day^{-1}, resulting in net drying (Costa & Foley, 2000).

It has been postulated that the Amazon might have multiple stable states with the current forest promoting resilience of itself through precipitation recycling. A threshold at 25–30% deforestation would lead to a state change causing the loss of remaining forests and conversion of the Amazon to a stable grassland state (Alcock, 2003). This model presumes tight coupling between precipitation amounts and total forest cover such that any precipitation reduction would lead to a concomitant loss of canopy cover. The model does not account for stored moisture accessible by deep rooting forests, forest regeneration processes or the existence of comparable closed canopy forests across a range of average annual precipitation amounts and dry season lengths.

Fire and climate

Climate will affect the fire situation in the Amazon directly, through changes in temperature and precipitation, and indirectly, through climate-forced changes in vegetation and fuel composition and structure (Pausas & Bradstock, 2007). These effects will occur at a range of temporal and spatial scales. If temperatures increase and precipitation is reduced, then potential fuels that are normally too wet to burn will dry more quickly and more often, thereby increasing the susceptibility of forests to burning. The frequency of drought will be a prime determinant of both how often forest fires occur and how extensive they become. El Niño years have generally been associated with the largest fire events in the Amazon (Cochrane *et al*., 1999; UNEP, 2002; Alencar *et al*., 2006) but intense drought and fire conditions, such as occurred in the southwest Amazon in 2005, need not be associated with El Niño (IPCC, 2007).

Over longer time scales, vegetation responses to climate change may drive changes in regional fire regimes. Forest dieback would likely result in periods of extensive and intense burning until reduction of forest fuels. Savanna (cerrado) vegetation would succeed forests when nearby but, throughout much of the basin, grass or scrub vegetation would dominate. These ecosystems would be characterized by frequent low intensity fires that would reinforce climate exclusion of mature forest species.

Fire and human land use

Fire regimes of many of the world's ecosystems are largely a function of human activity, which either promotes or suppresses fire on the landscape. We have changed the frequency, displaced the seasonal timing and altered the severity of fires over several millennia. The natural fire regime for tropical moist and tropical humid forests is one of little or no fire. Fire return intervals of 1000 years or more are quite likely (Hammond & ter Steege, 1998). Fires undoubtedly occurred in the Amazon before human involvement, especially within the transition forests that form the ecotone between the Amazon's core and savanna regions, but it is likely that tree populations went generations between fire events. Conditions throughout much of the Amazon now differ substantially from historical patterns and fire is almost exclusively a side effect of human activities. Shifts in the frequency, intensity and pattern of forest fires in the tropics represent a shift in the fire regime.

The fire situation in the Brazilian Amazon has changed dramatically since the early 1970s. Brazil built a network of roads linking the Amazon to the rest of the country, opening the land for colonization and changing the nature of fire in this region. The large influx of people, deforestation to support new land uses and continued road construction have combined to fragment the region's forests. Fire is the primary tool used to establish and maintain wide stretches of pastures and agricultural lands along the expanding road network. More than 15% (INPE, 2005) of the Amazon has been cleared in the last several decades, with forests being felled and burned at >2 million ha yr^{-1} in recent years. Annually, roughly another 20 million hectares of cleared lands are intentionally treated during the dry season with fire (UNEP, 2002) as cleared lands are often reburned every 2–3 years for management purposes (Kauffman *et al*., 1998).

Selective logging is a prevalent land use in the Amazon, along with subsistence farming, ranching and commercial agriculture. These logging activities do not clear the land of forest. Commercially valuable tree species are selectively removed. If left undisturbed, these forests would recover to preharvest levels of biomass. Extraction rates can be as low as a few trees per hectare but many forests are heavily damaged during road building and tree felling activities. Forests can be logged multiple times for additional tree species as regional timber markets develop (Uhl *et al*., 1997). Logging also catalyzes deforestation by opening roads into unoccupied government lands and protected areas that are subsequently colonized by ranchers and farmers (Veríssimo *et al*., 1995). Logged forests have open canopies that allow greater air movement and heating that rapidly dry potential fuels. Intact forests are resistant to fire encroachment after more than a month without rain, but selectively logged forests may become flammable in as few as 6–8 rainless days (Uhl & Kauffman, 1990). Fire can spread twice as fast in damaged forests (Cochrane *et al*., 1999) and the regional presence of large areas of logged forests results in much greater areas being affected by wildfires (Cochrane *et al*., 2004).

Fire and wildlife

Fire has causes little direct mortality of wildlife but the indirect loss of habitat is just as deadly (UNEP, 2002). To date, there have only been a few studies on a limited number of taxa upon which to judge the response of wildlife to fires in the Amazon, but the impact appears widespread and severe (Peres *et al*., 2003; Barlow *et al*., 2007). More studies are needed in order to understand the breadth of the impact of fires on wildlife and how it responds to fire disturbances of different spatial scales and temporal return intervals.

The fire dynamic

Moist Amazonian tropical forests do not appear to have tell-tale signs of evolutionary adaptation to fire (e.g. thick bark, serotiny, heat or smoke-related germination) that would indicate fire has been an evolutionary selective pressure on these tree species (Uhl & Kauffman, 1990). Aside from fortuitous traits, such as the resprouting capacity of many tree species (Kauffman, 1991) or the deeply buried meristems of ancient monocot species (e.g. palms), the diverse Amazonian tree species are remarkably vulnerable to even the weakest of fires. Initial fires in intact closed canopy forests spread as thin, slowly creeping, ribbons of flames a few tens of centimeters in height. Little besides leaf litter is consumed but fire-contacted seedlings and saplings are killed. Smaller trees (<30 cm in diameter) are at high risk of mortality because most Amazonian trees have very thin bark. However, bark thickness increases with tree diameter (Uhl & Kauffman, 1990). Typical fires may kill 40% of the trees (>10 cm diameter), but reduce living biomass by as little as 10%, as few large trees are killed (Cochrane & Schulze, 1999).

Fires spread slowly, on the order of 0.25 m min^{-1}, due to moist conditions under the forest canopy. Late in the day, as temperatures drop and relative humidity levels rise, fires often die out, residing only in a few smoldering logs. If weather conditions permit, the smoldering remains of the previous day's fires reignite by mid-to-late morning. Fire lines may move only 100–150 m a day but can keep burning this way for weeks or months, as weather permits (Cochrane *et al*., 1999). The quantity, condition and distribution of large fuels (fallen boles, crowns and large branches) determine reignition probability because of their ability to shelter fires during periods when conditions are insufficient for flaming combustion and fire spread. Because of heavy slash loads, logged forests are more likely to sustain fires over extended time periods (Cochrane & Schulze, 1999). Fuel loading makes fires in these forests very intense (Uhl & Kauffman, 1990), highly degrading sites and making them more vulnerable to recurrent fires.

Subsequent fires in previously burned forests are more ecologically severe. Forests lose much of their remaining canopy cover. An intact forest rarely exceeds 28 °C on the hottest days, but after fire or logging opens the canopy, temperatures may approach 38 °C under similar conditions (Uhl & Kauffman, 1990). Forest fuels dry quickly, making the forest susceptible to new fires. Mortality and tree-fall result in greater fuel loading after initial and subsequent fires. Flame lengths, flame depths, spread rates, residence times and fireline intensities are all significantly higher in recurrent fires (Cochrane *et al*., 1999). A second fire can kill another 40% of the original trees, this time corresponding to 40% of the living biomass (Cochrane & Schulze, 1999). Canopy cover is further reduced and fuel loads increase again. Burning closed canopy evergreen forests creates a positive feedback in both fire susceptibility and fire severity. This process can continue until complete deforestation occurs and a grassland or scrub ecosystem replaces the forest (Cochrane *et al*., 1999).

Fragmentation and land cover change

Landscape fragmentation exposes remaining forests to increasing levels of disturbance along edges. This can result in increased mortality to at least 100 m from forest edges (Laurance *et al*., 1997), which can raise wind speeds, insolation rates and fuel loads (Nascimento & Laurance, 2004). Ongoing deforestation and regional development plans that provide colonists access to new portions of the Amazon are expected to increase annual deforestation rates by 25% by 2020 (Laurance *et al*., 2001) further fragmenting remaining forests and increasing fire frequency.

Landscape fragmentation and land cover change interact synergistically to expose more forest area to wildfire (Cochrane, 2001). In the past, agricultural plots and pastures existed as islands of easily flammable vegetation (e.g. grasses) within a matrix of forest that was too moist to burn. As selective logging and periodic forest fires damage the forest's moisture retaining capacity, the landscape matrix becomes increasingly porous, allowing fire to spread more easily across the landscape. This situation is opposite that in many regions of the world where land conversion to agriculture results in a reduction of landscape connectivity and fire spread. The driving factors are changes in fuel availability (primarily a function of fuel moisture) and fuel continuity. Fire spread rates in tropical moist forests are generally <1 m min^{-1} (Cochrane *et al*., 1999) while those in open pastures, depending on wind speeds and grazing intensity, can exceed 100 m min^{-1} (Cheney & Sullivan, 1997). Even when fuel breaks do occur (e.g. roads), there are rarely trained fire suppression personnel available to capitalize on these features. Analogous systems outside of the tropics might be found in temperate rainforests and swamp lands. The effects of fragmentation and logging on fire spread are illustrated in Fig. 1. More exposed edges mean more forest area is likely to burn. Unplanned settlement of forest interiors, as occurs when settlers follow loggers into new forest areas (Veríssimo *et al*., 1995), is even more damaging (Fig. 1d) as fire spread rates may double in degraded forests and the active burning period lengthens such that much more area becomes vulnerable to fire.

Managing fire in Amazonian landscapes

Fire frequency in tropical forests is a function of location, proximity to deforested edges is the prime determinant of risk because nearly all fires are anthropogenic in origin (Cochrane, 2001; Cochrane & Laurance, 2002). The new fire dynamic for many tropical forests within a few kilometers of deforested edges is one of frequent fire incursions and increasing fire severity. Portions of the eastern Amazon now experience forest fires every 5–10 years (Cochrane *et al*., 1999; Alencar *et al*., 2006), which prevents any significant regeneration of canopy trees. Rainforests are vulnerable to recurrent fires and

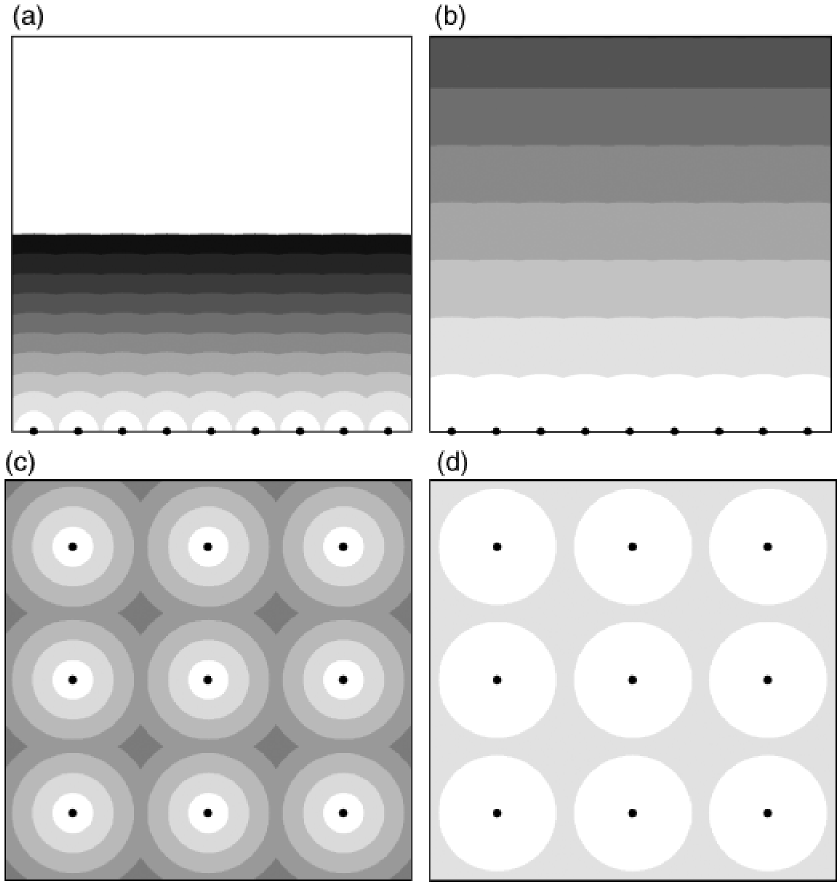
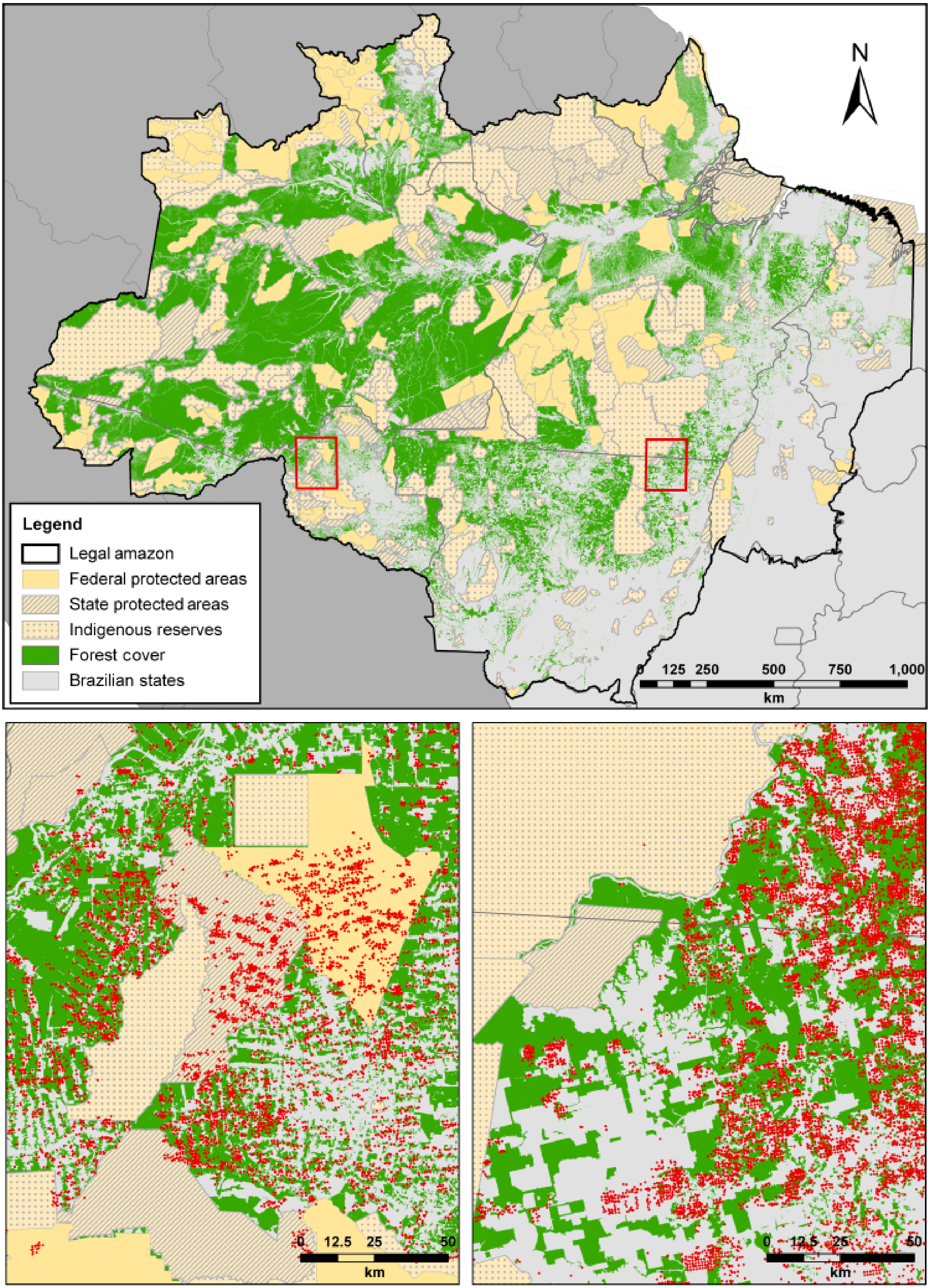


Fig. 1 Forest fragmentation, forest degradation and fire spread. Part (a) shows 10 days of fire spread into a flammable forest (fire progression $0.25\,m\,min^{-1}$; 7 h active burning per day (Cochrane *et al.*, 1999) from nine hypothetical evenly spaced fires along a 2 km section of road. Roughly half the area burns in 10 days. Shading illustrates fire progression for each individual day. Part (b) shows the same effects in a selectively logged forest, which burns much more rapidly (7 days for 100%) due to increased fire spread rates and longer daily burning periods; average daily spread 2.9 times greater than in unlogged forest (Cochrane *et al.*, 2004). Part (c) shows the effects if fires were placed in the forest (100% in 5 days), which rarely occurs in intact forests. Part (d) shows the same effects in selectively logged forests which are frequently settled along organized logging roads (100% burned in <2 days).

frequent fires progressively replace forest species with fire tolerant vegetation. This transition can take decades to centuries to be completed, depending on fire frequency (Bowman, 2000). To date, fire regime changes have been spatially constrained by road networks and associated human land use. Climate change, if severe enough, could alter this situation, potentially changing the fire regime of vast portions of the Amazon forest.

Fire risk management needs to take into account landscape configuration, land cover types, disturbance history, climate and weather (UNEP, 2002). It is believed that one-third of the Brazilian Amazon became susceptible to fire during the El Niño of 2001 (Nepstad *et al.*, 2004). Fire susceptibility is not synonymous with fire vulnerability though. With spread rates of only $15\,m\,h^{-1}$ and limited hours for active fire spread each day (Cochrane *et al.*, 1999), it can take 10 days or more to burn 1 km deep into an intact forest. Fire spread rates fundamentally limit the amount of area that can be burned in any given year. Extensive areas of forest can only be burned by many widely distributed fires (Fig. 2).

Maintaining large blocks of unsettled forest is critical for managing landscape level fire in the Amazon. For this reason, Brazil's vast and expanding network of protected areas (Veríssimo *et al.*, 2002) is key for limiting the spread of both deforestation and forest fire (Fig. 2). These benefits depend on the efficacy of the protected areas, however, as they presume that forests inside the boundaries are less like to be burned and deforested. Fire management efforts should be centered around education and include alternatives to fire use, law enforcement, policy changes and planned development (Cochrane, 2003). Separating regions of selective log-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ging from those of fire-dependent agriculture and zoning development so as to reduce forest fragmentation would help to reduce fire spread. The frontier nature of much of the Amazon and the thousands of intentional fires set each year make locating escaped forest fires very difficult until they have become very large. Fire suppression activities are difficult, expensive and largely ineffective, so they should not be viewed as a substitute for proactive fire management efforts (UNEP, 2002).

The future of fire in the Amazon

There is increasing consensus among GCMs that the climate in the Amazon will continue to warm over the next century. Temperatures in the western Amazon are expected to rise more than in the east while the east is expected to get somewhat drier and the west even wetter (IPCC, 2007). Expected changes in extreme weather events include more days with intense precipitation in central Amazonia and weaker precipitation intensity in the north-east (Hegerl *et al*., 2004).

Model-derived expectations that, other than the north-west region, the Amazon will experience longer periods between rainfall events are of import for fire (Tebaldi *et al*., 2006). This is critical because fire susceptibility is more closely related to the time since last rain than total rainfall amounts (Uhl & Kauffman, 1990; Cochrane & Schulze, 1999). Malhi & Wright (2004) developed a useful dry season index (DSI) based on several decades of climate data that, at least in a relative sense, provides a composite descriptor of dry season length and intensity. Forest fire risk should be expected to increase with increasing DSI. Increasing temperatures augment evapotranspiration demands and increase DSI, as would decreasing precipitation, especially in the dry season. Unless precipitation rates increase sufficiently to offset expected evapotranspiration demands due to rising temperatures, the DSI is likely to increase. Calculation using the CRU TS 2.1 climate dataset (Mitchell & Jones, 2004) shows a slight but nonsignificant decrease in DSI from 1960 to 2002 for the Amazon. Regional comparisons also show net decreases in DSI (0.39–0.46 mm yr^{-1}; $P<0.005$) for all

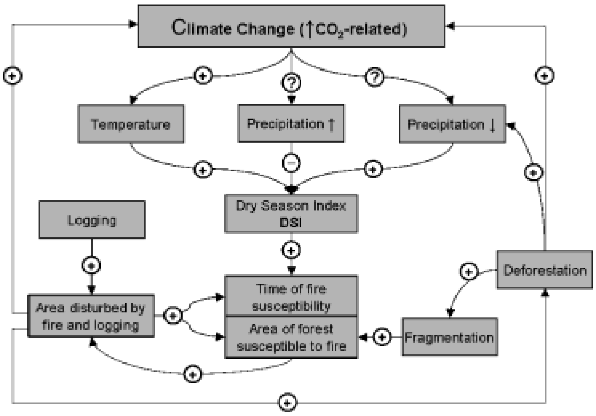

Fig. 3 Schematic relationships between climate, land use and fire.

portions of the basin with the exception of the Andes of Peru, Ecuador and Columbia, which has a slightly increasing DSI (0.19 mm yr^{-1}; $P<0.05$). DSI does not track directly with fire susceptibility though, because fuel moisture is driven by subcanopy humidity levels (Messina & Cochrane, 2007), which are largely decoupled from above-canopy conditions (90% subcanopy moisture comes from evapotranspiration; Moreira *et al*., 1997). The qualitative relationships between climate, land use and fire are shown in Fig. 3.

Populations in the Amazon are becoming increasingly urban but per capita deforestation rates for rural residents have increased. Large investments for regional development are expected to expand the frontier and accelerate deforestation rates (Laurance *et al*., 2001). Deforestation rates, especially in the southern Amazon, are tied to the price of soybeans. The United States interest in biofuels has shifted much planting from soybean to corn, causing soy prices to rise and giving incentive for expanded Amazon deforestation for soybean cultivation (Laurance, 2007). Brazil is entering an agroenergy economic period which may lead to rapid frontier expansion in the center-west (soybean – biodiesel) and north regions (sugar cane – biofuel). This period may continue until technical advancements in cellulosic energy extraction make Amazonian bioenergy

Fig. 2 Protected forests in the Brazilian Amazon. Despite the extensive deforestation and development occurring in the Amazon, Brazil, has been creating an expanding network of various Federal and State conservation areas and Indigenous Lands. Current protected areas encompass roughly 37% of the region and these areas are being expanded to approximately 48% of the basin. Preserving large blocks of intact forest is crucial for managing fire at the landscape scale in the Amazon. Forest cover data from INPE Prodes (2005). Protected area data provided by Instituto do Homem e Meio Ambiente da Amazônia (IMAZON) and Instituto Socio-Ambiental (ISA). Inset maps illustrate variability in efficacy of protected areas. Red dots represent hotspot detections from the MODIS sensor aboard Terra and Aqua satellite for the period of July–October 2007 (http://maps.geog.umd.edu/firms/). Most protected areas are successful at near or complete exclusion of fire while others have little effect on fire incursion, with effectiveness being directly related to their ability to prevent human settlement and land conversion within their borders.

production less viable, potentially leading to an economic bust and land abandonment (Sawyer, 2008).

Although it appears unlikely that the Amazon will experience extensive dieback, warmer and potentially drier, conditions will likely make the forest more susceptible to burning in any given year. Fragmentation will expose more forests to fire and the accumulation of disturbed forests from logging activities and previous fires will exacerbate this problem. As long as agricultural land use relies on fire as a primary land clearance and maintenance tool, ignition sources along forest edges will never be limiting. There will be no question whether forest fires will occur; it will only be a matter of when and where. Variables governing widespread landscape fires will include drought, forest fragmentation, forest disturbance history and the amount of time fires are allowed to burn. Without significant changes in the way landscapes are being managed, the risk of forest fires and general fire spread among all human land uses will increase, potentially devastating the ecological and economic bases for human wellbeing in the region.

The future is not all bleak. Brazil's ambitious implementation of protected areas and sustainable use forests (Veríssimo *et al*., 2002) has the potential to protect much of the Amazon from runaway fragmentation and fire. Protected areas must function as more than paper parks though and sustainable use areas need to provide sufficient economic returns to local populations to warrant their protection from conversion to agriculture. Fire regimes throughout much of the Amazon are rapidly changing but the arrangement of these altered fire regimes is spatially tied to human land use and not spread homogeneously across the basin. The Amazon has been incredibly resistant to previous climate changes and should continue to adjust to future climate changes, but will only do so if the landscape is managed to allow both forests and human populations to coexist.

The future of climate and vegetation modeling in the Amazon

GCMs are currently limited by the scale at which they run and care must be taken when extrapolating local or regional results from climate simulations. Finer scale analyses are needed in order to support locally relevant decisions. Ideally, GCM results should be used to parameterize RCMs that have much higher spatial resolution and can incorporate topography and geographic features that are climatically relevant at that scale of analysis. GCMs need to be expanded to incorporate the reciprocal effects of land cover on climate. This requires accurate data layers on dynamic land cover changes and vegetation feedbacks for proper parameterizations. Without such additions, GCM simulations will be fundamentally limited in their power to represent climate at scales relevant to policy and management.

Current DGVMs are limited by necessary generalization of the world's vegetation into a few broad classes (PFTs). Because these models are not process driven, they are limited in their ability to predict how ecosystems will change. Broad generalization is adequate for work at a global scale, but modeled responses of large regional changes in vegetation structure type should be treated with skepticism. Anomalies are more likely indications of model weaknesses than imminent ecological disasters. Such findings should provide impetus for detailed work with regional experts to redefine more accurate PFTs. As research advances from broad questions to more detailed analyses, the level of sophistication and detail of the vegetation components must advance as well. Instead of relying on invariant parameters, DGVMs should make use of observations to derive inverse solutions to parameter values. For example, instead of assuming a constant rooting depth across a region, use observations and calculations of water demand to solve for actual rooting depth under drought conditions (cf. Ichii *et al*., 2007).

As GCMs become more sophisticated and increasingly linked to vegetation and land cover-related processes, greater interdisciplinary collaboration is needed between the modeling community and other earth scientists. Detailed knowledge of ecosystems and their responses to climate are needed to develop accurate and useful products that will themselves drive future studies of the ecosystems and processes involved. As climate, geomorphology and biology become integrated into models, the resultant simulations will transcend GCMs to become global biophysical models.

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