Global Change Biology (2004) **10,** 704–717, doi: 10.1111/j.1529-8817.2003.00772.x

Amazon drought and its implications for forest flammability and tree growth: a basin-wide analysis

DANIEL NEPSTAD*†, PAUL LEFEBVRE*, URBANO LOPES DA SILVA†, JAVIER TOMASELLA‡, PETER SCHLESINGER*, LUIZ SOLÓRZANO*, PAULO MOUTINHO† DAVID RAY* and JOSÉ GUERREIRA BENITO*

**The Woods Hole Research Center, PO Box 296, Woods Hole, MA 02543, USA, †Instituto de Pesquisa Ambiental da Amazônia, Av. Nazaré, 669, Centro, Belém, Pará 66035-170, Brazil, ‡Centro de Previsão de Tempo e Estudos Climáticos, Rodovia Presidente Dutra, Km 40, SP-RJ 12630-000, Cachoeira Paulista, SP, Brazil*

Abstract

Severe drought in moist tropical forests provokes large carbon emissions by increasing forest flammability and tree mortality, and by suppressing tree growth. The frequency and severity of drought in the tropics may increase through stronger El Niño Southern Oscillation (ENSO) episodes, global warming, and rainfall inhibition by land use change. However, little is known about the spatial and temporal patterns of drought in moist tropical forests, and the complex relationships between patterns of drought and forest fire regimes, tree mortality, and productivity. We present a simple geographic information system soil water balance model, called RisQue (Risco de Queimada – Fire Risk) for the Amazon basin that we use to conduct an analysis of these patterns for 1996–2001. RisQue features a map of maximum plant-available soil water (PAW_{max}) developed using 1565 soil texture profiles and empirical relationships between soil texture and critical soil water parameters. PAW is depleted by monthly evapotranspiration (ET) fields estimated using the Penman–Monteith equation and satellite-derived radiation inputs and recharged by monthly rain fields estimated from 266 meteorological stations. Modeled PAW to 10 m depth ($PAW_{10\,m}$) was similar to field measurements made in two Amazon forests. During the severe drought of 2001, $PAW_{10\,m}$ fell to below 25% of PAW_{max} in 31% of the region's forests and fell below 50% PAW_{max} in half of the forests. Field measurements and experimental forest fires indicate that soil moisture depletion below 25% PAW_{max} corresponds to a reduction in leaf area index of approximately 25%, increasing forest flammability. Hence, approximately one-third of Amazon forests became susceptible to fire during the 2001 ENSO period. Field measurements also suggest that the ENSO drought of 2001 reduced carbon storage by approximately 0.2 Pg relative to years without severe soil moisture deficits. RisQue is sensitive to spin-up time, rooting depth, and errors in ET estimates. Improvements in our ability to accurately model soil moisture content of Amazon forests will depend upon better understanding of forest rooting depths, which can extend to beyond 15 m. RisQue provides a tool for early detection of forest fire risk.

Keywords: evapotranspiration, fire, hydrology, NPP, rainforest, soil moisture, tropical, water balance

Received 5 December 2002; revised version and accepted 27 March 2003

Introduction

Drought is one of the most important components of global change in shaping the future of moist tropical forests. In 1997 and 1998, the 20th century's most intense El Niño Southern Oscillation (ENSO) event provoked severe droughts in Amazonia, Southeastern Asia, and Mexico, with large effects on forest fire, net primary productivity (NPP), and carbon storage. This drought depleted soil moisture in large forest areas, contributing to approximately 40 000 km^2 of fire in standing forests of the Amazon (Nepstad *et al.*, 1999a,

Correspondence: Daniel Nepstad, The Woods Hole Research Center, PO Box 296, Woods Hole, MA 02543, USA, fax +1 508 540 9700, e-mail: dnepstad@whrc.org

2001a; Mendoça *et al.*, (in press)) and 20 000 km^2 of forest fires in Eastern Kalimantan, Borneo (Siegert *et al.*, 2001). These understory fires killed forest trees containing approximately 0.4 Pg of carbon in the Amazon (Mendoça (in press)) – 5% of annual global human-induced emissions (Houghton, 1999). More than 1 Pg may have been released from the peat forests of Indonesia through the fires of 1998 (Page *et al.*, 2002). The 1997/1998 drought also elevated the annual mortality rate of central Amazonian trees by 60% (Williamson *et al.*, 2000). Drought- and fire-related tree mortality further increase forest flammability in a positive fire feedback loop that could exacerbate forest impoverishment in these regions (Nepstad *et al.*, 1995, 1999a, 2001a; Cochrane *et al.*, 1999). Model experiments also suggest that the 1997/1998 drought reduced NPP of Amazon forests by 20–30% (Potter *et al.*, 1998; Tian *et al.*, 1998). Moreover, a large throughfall exclusion experiment indicates that the most drought-sensitive component of NPP in a central Amazon forest is stemwood growth (Nepstad *et al.*, 2002); hence, drought-induced reductions of NPP appear to decrease inputs to this long-lived carbon storage pool, counteracting the net accumulation of carbon that has been suggested for some Amazon forests (Grace *et al.*, 1995; Malhi *et al.*, 1998; Phillips *et al.*, 1998).

Several processes may be driving an increase in drought frequency and severity in moist tropical forests of Amazonia and Southeast Asia. Tropical temperature increase associated with global warming may exacerbate drought effects by accelerating evaporation (White *et al.*, 1999; Costa and Foley, 2000). ENSO events have also become more frequent in recent years, perhaps because of the accumulation of heat-trapping gases in the atmosphere (Trenberth & Hoar, 1997; Timmermann *et al.*, 1999). Tropical land-use change may provoke drought and temperature increases over much of Amazonia through the changes in albedo and evapotranspiration (ET) that accompany forest conversion to cattle pasture (Nobre *et al.*, 1991; Lean *et al.*, 1996; White *et al.*, 1999; Costa & Foley, 2000; Silva Dias *et al.*, 2002), while smoke from fires may inhibit rainfall locally (Rosenfeld, 1999).

Given the importance of drought for moist tropical forests, our understanding of drought and its effects on moist tropical forests is surprisingly shallow. Field studies of drought effects on tropical mast fruiting, for example (Curran *et al.*, 1999; Wright *et al.*, 1999) are difficult to examine at larger spatial scales because of a lack of information on the spatial and temporal patterns of drought in the world's large tropical forest formations. Similarly, field studies of drought effects on forest flammability and forest fire (Kauffman *et al.*, 1988; Uhl *et al.*, 1988; Uhl & Kauffman, 1990; Cochrane & Schulze, 1998; Nepstad *et al.*, 1999a, b, 2001a; Alencar *et al.*, in press) and regional studies of tropical fire based upon remote sensing of fire and fire scar (Setzer & Pereira, 1991; Siegert *et al.*, 2001) are difficult to integrate and interpret for lack of regional analyses of drought patterns.

We present a simple geographic information system model of soil water for the Amazon basin, and employ this model together with field data to conduct an initial assessment of the basin-wide effects of drought on forest flammability and carbon uptake through tree growth. We validate the model through comparison with field measurements of soil moisture, and test the model's sensitivity to spin-up parameters, rooting depth, and errors in the estimation of ET.

Methods

General approach

We call our soil water balance model RisQue (from 'Risco de Queimada', meaning 'fire risk' in Portuguese) because of its original use in forest fire prediction for the Amazon region (Nepstad *et al.*, 1998, 1999a, b). RisQue is calculated at monthly time steps and has spatial resolution of 8 km. It begins with a map of maximum plant-available soil water (PAW_{max}) developed using soil texture profiles and empirical equations relating soil texture and critical soil water parameters. Soil moisture is depleted by ET as calculated using the Penman–Monteith equation with radiation inputs derived from the GOES-8 satellite. Soil moisture is recharged by fields of rainfall interpolated from a network of 266 weather stations. Forest rooting depth is poorly known for tropical forests, and for world vegetation generally (Canadell *et al.*, 1996; Schenk & Jackson, 2002a, b) and is assumed to average 10 m in the Amazon based upon previous deep soil studies in the region (Nepstad *et al.*, 1994, 2002; Hodnett *et al.*, 1995; Jipp *et al.*, 1998). We depart from the assumption that the capacity of plant root systems to absorb soil water decreases with depth as root length density diminishes (e.g. Jackson *et al.*, 2000) based upon our previous field measurements of soil water uptake from deep soil layers (Nepstad *et al.*, 1994; Jipp *et al.*, 1998). The ability of sparse roots in deep soil to absorb substantial amounts of water may be facilitated by arbuscular mycorrhizae (Nepstad *et al.*, 2001b) and hydraulic redistribution of soil moisture (Dawson, 1993).

We examined the potential effects of patterns of drought that occurred from 1996 to 2001 on forest flammability using the empirical relationship between PAW to 10 m depth ($PAW_{10\,m}$) and leaf area index (LAI) measured at two of our field sites (Nepstad *et al.*, 1994,

2002) and the empirical relationship between LAI and the spread rate of experimental fires conducted in three Amazon forests. Finally, we examine the potential effects of drought on tree growth using an empirical relationship between PAW_{10m} and tree bole increment (Nepstad *et al.*, 2002).

Mapping PAW_{max}

We define PAW as the amount of water stored in the soil that is held at the matric potential of –0.01 MPa or less ('field capacity', SW_{fc}) minus soil water held at –1.5 MPa or less ('permanent wilting point', SW_{pwp}). PAW_{max} is therefore equal to SW_{fc} minus SW_{pwp}. PAW at any point in time is equal to the difference between total soil water ($\leq$field capacity) and SW_{pwp}. The permanent wilting point of Amazon tree falls below –1.5 MPa in some Amazon forests but this has little effect on the absolute amount of soil water available to the forest (Jipp *et al.*, 1998).

We mapped PAW_{max} by combining texture data from soil samples collected in the field with empirical relationships between soil texture and the critical soil physical parameters SW_{fc} and SW_{pwp}. Soil texture data were provided by the RADAMBRASIL (Brazil 1973–1983) project, and include 1494 soil profiles, with each profile divided into soil horizons for which texture analyses were conducted. Equations relating soil texture and carbon content to field capacity and permanent wilting were developed by Tomasella & Hodnett (1998) using soil water vs. matric potential curves for Amazon soils. We tested these soil texture equations against 40 unpublished curves of soil water vs. matric potential (E. Maklouf, unpublished data), and found that they performed well for soils with very high sand or very high clay content, while soil texture equations of Saxton *et al.* (1986) were better for estimating critical soil moisture values for intermediate textures. Thus, we employed the texture equation that was most appropriate to the texture of each RADAMBRASIL soil profile.

For each soil texture profile, we estimated the PAW_{max} (in mm) for each horizon using SW_{fc}, SW_{pwp} and the height of the horizon, and summed PAW_{max} for the horizons to estimate PAW_{max} for the entire sampled profile. Most of the RADAMBRASIL soil texture profiles extend to depths of only 1–2 m. We estimated PAW_{max} of soil below the sampling depth by assuming that the lowermost horizon extended uniformly downward. The scant soil physical data that are available for depths greater than 2 m (Jipp *et al.*, 1998) suggest that PAW_{max} declines with depth; hence, our estimate of PAW_{max} for the entire profile may have overestimated PAW at depth. Our estimates of PAW_{max} for soils underlaid by bedrock (e.g. Lithosols) included only the sampled soil profile, since forest root systems presumably extend little into the rock.

Rooting depth is poorly described for the Amazon basin, but extends to 8 m and greater where it has been examined. Field excavations of root systems and measurements of soil water changes across seasons indicate that rooting depths in two Amazon forests located on the Belterra clay formation extend to below 10 m depth (Nepstad *et al.*, 1994, 2002; Jipp *et al.*, 1998; L. Solorzano, unpublished data). Soil excavations in the Ducke Reserve (central Amazon), in Santana do Araguaia (southeastern Amazon), and near Rio Branco, Acre (southwestern Amazon), found roots to the maximum excavation depth of 8 m (D. Nepstad, unpublished data, E. Mendonza *et al.*, unpublished data). Moreover, we found that roots had penetrated buried plinthite rock layers 5 m thick in Paragominas (eastern Amazon, Nepstad *et al.*, 1994). (This rock is more porous than bedrock.) Given this uncertainty about rooting depths, we examine the effects of three rooting depths (5, 10, and 15 m) on spatial and temporal patterns of PAW and forest flammability.

PAW is also influenced by the depth to water table, the seasonal variation of this depth, and surface runoff. RisQue underestimates PAW in forests that receive subsurface water flow from neighboring regions of higher rainfall and that have water tables that rise into the rooting zone during the rainy season. RisQue overestimates PAW in forests where steep slopes and/or low infiltration rates are associated with surface runoff. Insufficient data is currently available to assess the magnitude of these errors.

Maps of PAW_{max} were derived from individual soil profiles by interpolating the point estimates of PAW_{max} within soil map units. We first simplified the soil map of SOTER (van Engelen & Wen, 1995), which divides the Amazon into 47 soil types, by combining texturally and pedogenically similar soil types into eight different soil groups. The point estimates of PAW_{max} were then kriged within each aggregated map unit. In regions of the Amazon for which soil texture data were not available, including most of the non-Brazilian Amazon, PAW_{max} was mapped within each soil category as the average PAW_{max} for that category where data were available.

ET and rainfall

Mean daily potential ET was estimated using the Penman–Monteith formulation (Monteith, 1965) with parameterizations provided by the ABRACOS climate experiment in the Amazon (Wright *et al.*, 1996). Basic meteorological data for ET was obtained from synoptic weather stations (i.e., stations that provide real-time measurements of air pressure, air temperature, relative humidity, and wind) from the global telecommunications system (GTS) network augmented by automatic weather stations of the Centro de Previsão de Tempo Clima (CPTEC) network. Since GTS synoptic stations provide data every 6 h, we used the values at 12 and 18 GMT to represent the diurnal cycle. In general, these synoptic times provide more frequent and dense observations, since not all the synoptic stations available operate at the four synoptic times, and they provide a better representation of the diurnal cycle during daylight hours, when most evaporation takes place. We acquired data from 266 weather stations, 129 of which were located within Amazonia.

Since synoptic weather stations do not provide solar radiation data, a daily integrated estimation of total solar radiation was derived using satellite estimates (VIS 4 channel of GOES) using the algorithm proposed by Ceballos e Moura (1997). The results obtained by this methodology are in close agreement with the values derived for the Amazon forest by Wright *et al.* (1996) and Shuttleworth (1988).

Rainfall data incorporated information from several sources. Besides synoptic data, we used rainfall data form the Brazilian Electrical Agency (ANEEL). Data was interpolated using kriging schemes to a spatial scale of 25 km.

Validation

RisQue output (PAW) was validated through comparison with temporal trends of PAW measured to 10 m depth in a forest near Paragominas, eastern Amazon (Nepstad *et al.*, 1994; Jipp *et al.*, 1998) and in a forest near Santarém, in the east-central Amazon (Nepstad *et al.*, 2002).

Sensitivity analysis

The sensitivity of RisQue to rooting depth and ET error was examined by calculating the spatial and temporal variation of PAW using three scenarios of rooting depth (5, 10, and 15 m), and five scenarios of ET error (errors of –15%, –5%, 0, + 5% and + 15%). We also tested the sensitivity of model output to initial PAW, including 0, 50%, and 100%, and PAW estimated by running RisQue for 6- and 12-year periods using ET and rainfall data for 1996–2001.

Forest flammability

We examined the relationship between patterns of PAW estimated by RisQue and patterns of forest

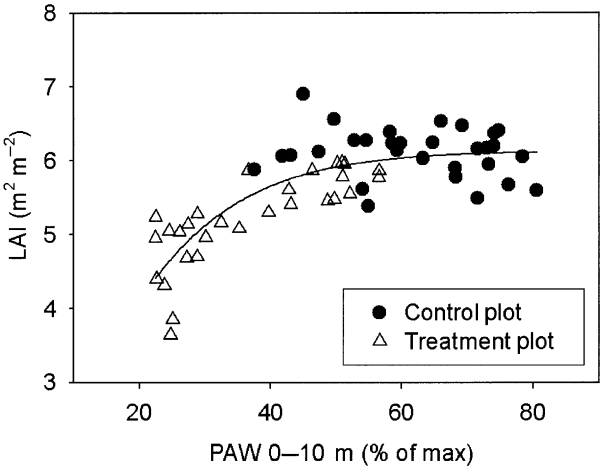

Fig. 1 Relationship between % PAW$_{\max}$ to 10 m depth and leaf area index (LAI) in the Tapajós National Forest, east-central Brazil (Fig. 4). Soil moisture measurements are made in deep soil shafts using time domain reflectometry and LAI is measured using the LiCor 2000 (Licor Inc., Lincoln, NE, USA). plant canopy analyzer. Points are from two 1 ha forest plots. The dry plot had 50% of rainfall excluded from 2000 to 2002 using understory panels. See Nepstad *et al.* (2002).

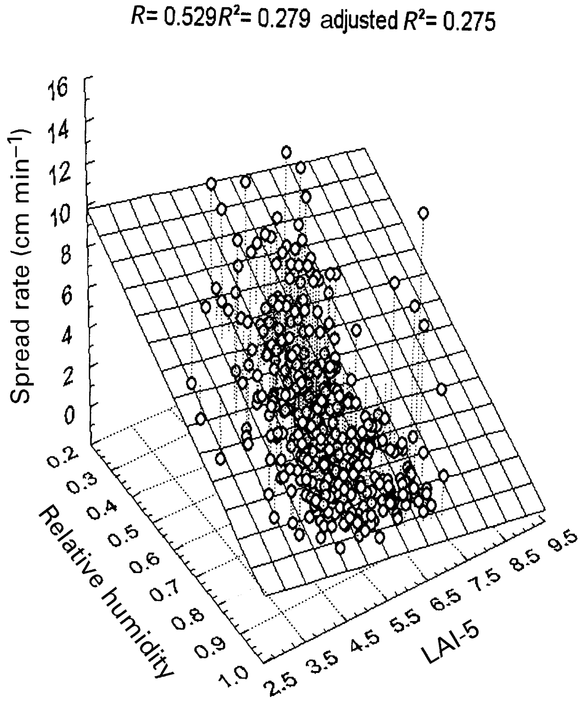

Fig. 2 Relationship between the rate of fire spread, leaf area index (LAI), and relative humidity of the forest understory for two Amazon forests. Each point represents an experimental fire conducted in either a tall, dense forest (Paragominas) or in a transitional forest (Santana do Aragauia, described in Nepstad *et al.* (1994).

flammability using empirical relationships between PAW and LAI derived from field measurements in a large, rainfall exclusion experiment underway in the east-central Amazon (Fig. 1, Nepstad *et al.*, 2002). In this experiment, soil water is monitored to 11 m depth and LAI is measured monthly.

LAI reduction in response to soil water depletion is a strong predictor of forest flammability. Canopy thinning allows more solar radiation to penetrate to the forest floor, drying out the fine fuel layer, and increasing air temperatures (Uhl and Kauffman, 1990; Nepstad *et al.*, 1995, 1999b). Experimental fires ignited in two Amazon forests indicate that LAI is strongly associated with the rate of fire spread (Fig. 2).

We employ RisQue estimates of PAW together with these field measurements of LAI, PAW, and forest flammability to provide an initial assessment of spatial and temporal patterns of forest flammability in the Amazon. We assume that a PAW reduction to 25% of its maximum value, which corresponds to an LAI reduction of 25% at our east-central Amazon field site (Fig. 1), is a sufficient level of drought stress to increase forest susceptibility to fire.

Carbon uptake in tree stemwood

Drought stress also inhibits primary productivity of the forest by reducing LAI (Fig. 1) and by provoking stomatal closure and reduced photosynthetic potential (Nepstad *et al.*, 2002). Although leaf production and belowground production appear to vary little in response to soil water deficits, tree stemwood growth is quite sensitive to PAW (Fig. 3, Nepstad *et al.*, 2002). Since stemwood is a long-lived carbon pool relative to litter and fine roots, its sensitivity to soil water may have important implications for the carbon budget of the Amazon basin. We provide an initial estimate of the effect of drought on carbon uptake in tree stemwood by assuming that the stemwood increment declines two-fold as PAW falls to 25% of PAW_{max} (Fig. 3).

Results

PAW_{max}

Assuming a 10-m rooting depth across the basin, PAW_{max} varied 10-fold, from a low of <250 mm of water in soils underlain by bedrock (Lithosols) to

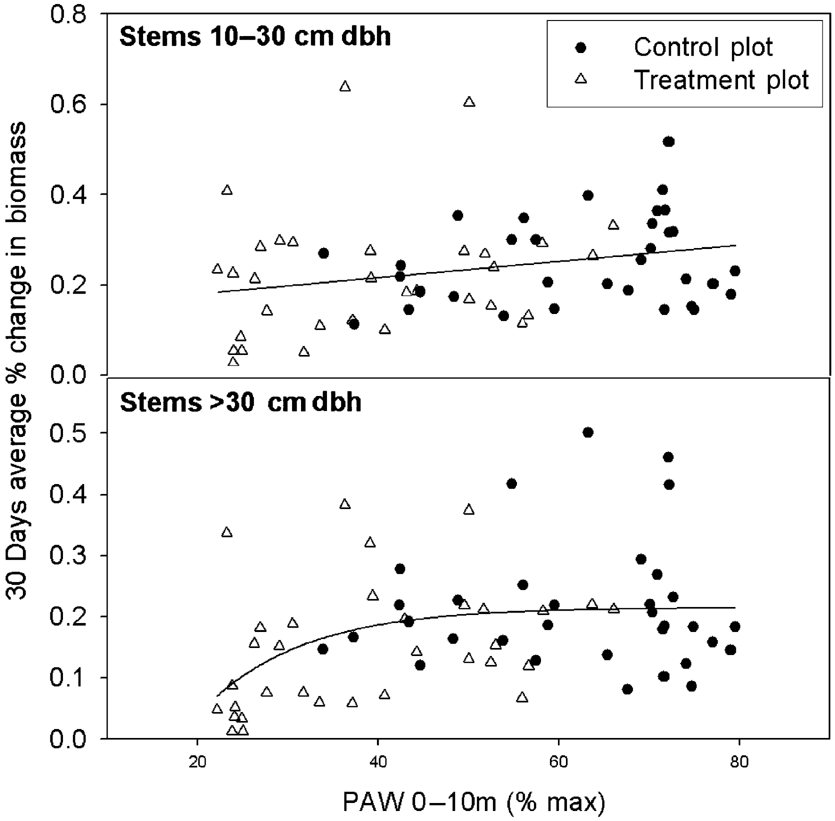

Fig. 3 Relationship between % PAW_{max} and tree growth for two size classes from the Tapajós National Forest (Nepstad *et al.*, 2002). Trees greater than 30 cm diameter represent two-thirds of the biomass of this forest.

2750 mm in some of the Podzolic soils (Ultisols in the US taxonomy) of the eastern and west-central Amazon (Fig. 4). The average $\mathrm{PAW}_{\max}$ for the region was 1496 mm, which is equivalent to approximately 1 year of ET. The paucity of soil texture profiles in non-Brazilian portions of northern and western Amazonia restricts the reliability of the $\mathrm{PAW}_{\max}$ estimates for these regions. These gaps in soil data have little effect on our estimates of severe soil moisture depletion, since rainfall is quite high in the northern and western portions of the region.

Evaporation and rainfall

Annual ET varied from 700 mm in the cloudy northwestern region of the Amazon to more than 2000 mm in relatively hot, sunny southeastern corner of the region (Fig. 5). Interannual variation of ET was high, ranging from an Amazon-wide average of 1612 mm in 1997 and 1680 mm in 1996 and 1999, to a high of 1728 in 2001.

Annual rainfall during the study period ranged from a low of 300 mm in regions of southern and southeastern Amazonia to a high of more than 6000 mm in the northwestern Amazon (Fig. 6). Amazon-wide annual rainfall varied from an average of 1660 mm in 2001 and 1830 in 1998 to a high of 2050 in 1996.

Meteorological station coverage for the region is fairly uniform, but sparse. There is an average of one station per 55 000 km^{-2} of the Amazon, with important gaps in the east-central portion of the region and in the northwest (Fig. 7).

Validation

Modeled PAW was highly correlated with PAW measured at two Amazon sites (Fig. 8, $r^2 = 0.67$, $P < 0.001$). The slopes of the modeled vs. measured PAW line and the 1 : 1 line were not significantly different ($F = 0.0998$; $P = 0.75$). The difference between the intercepts of these curves was significant, however ($F = 28.1875$; $P < 0.0001$), indicating that RisQue tends to underestimate PAW at the validation sites. Forty-seven percent of the validation points were within 20% of 1 : 1 line, with the greatest errors close to PAW = 0.

Sensitivity analysis

RisQue output of PAW is highly sensitive to spin-up parameters, rooting depth, and ET error. Given the close balance between rainfall and ET in much of the region (Figs 5 and 6), the level of PAW used to initiate the model can influence model output for up to 10 years. We therefore initiated the model with initial PAW equal to 50% of $\mathrm{PAW}_{\max}$, with a 12-year spin-up time using

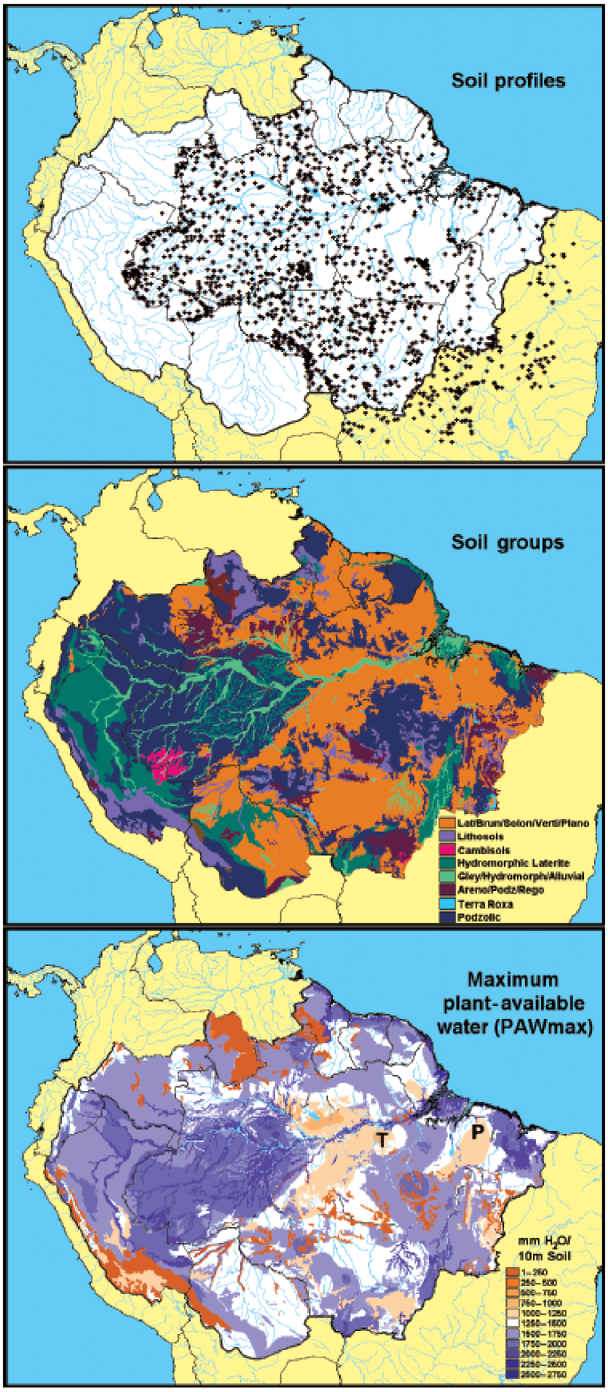

Fig. 4 (a) Locations (+) of RADAMBRASIL soil profiles used to develop empirical relationships between soil texture properties and critical soil water parameters. (b) Map of groups of soil classes based on texture properties, used to constrain the spatial interpolation of profile data in development of (c) the map of maximum plant available water ($\mathrm{PAW}_{\max}$) for 10 m rooting depth. Validation sites are indicated for the Tapajós National Forest (T) (Nepstad *et al.*, 2002) and Paragominas (P) (Nepstad *et al.*, 1994).

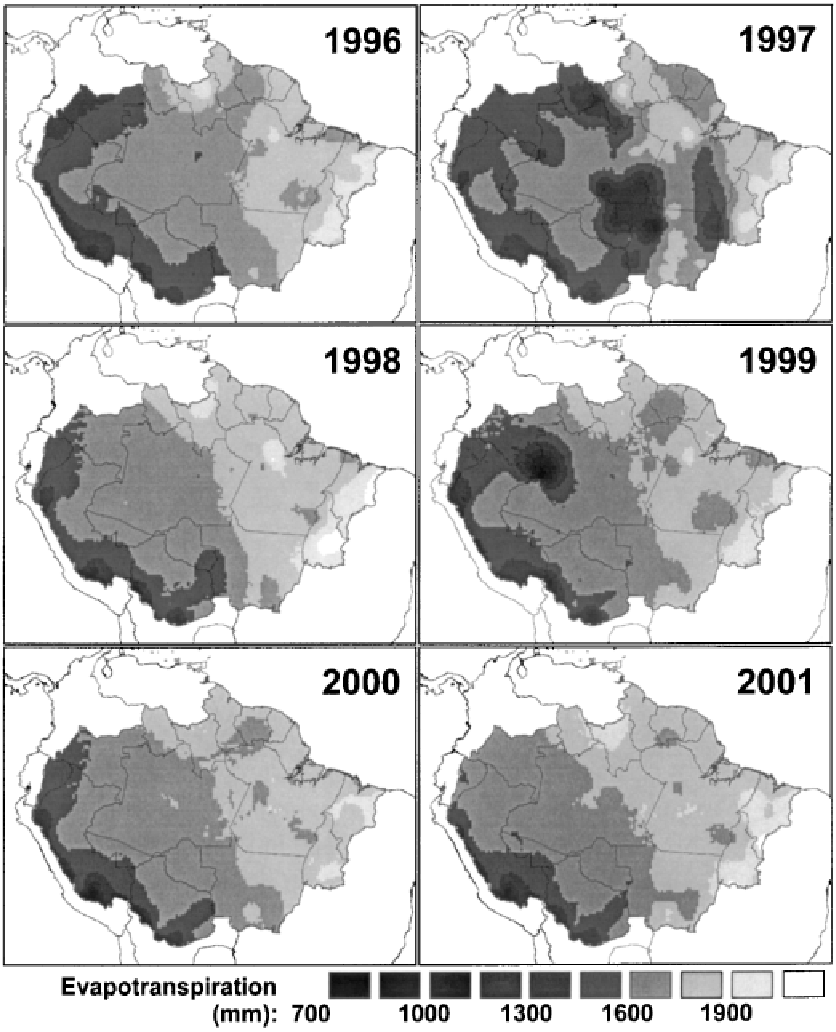

Fig. 5 Annual total evapotranspiration in millimeters of water per year.

two repetitions of rain layers and ET layers for the 6-year study period (Figs 5 and 6).

Rooting depth influences PAW by affecting PAW_{max} and the amount of moisture that is potentially stored in the soil to sustain ET during periods of low rainfall. A halving of rooting depth from 10 to 5 m results in a doubling of the Amazon forest area in which PAW drops below 25% of PAW_{max}, during most years (Figs 9 and 10). Using a rooting depth of 5 m, RisQue estimates that virtually the entire Amazon had dropped below 50% of PAW_{max} by the end of 2001 (Fig. 9).

ET errors also have a large influence on RisQue estimates of PAW. A 15% error in ET results in approximately a twofold change in the area of severe soil water depletion ($\leq$25% PAW_{max}, Figs 9 and 11). When ET is reduced by 15%, the area of moderate soil water depletion ($\leq$50% PAW_{max}) changes little for the driest month of the 6-year study period, December 2001 (Fig. 9).

RisQue underestimates PAW along the Andes mountain range because of the strong influence of the desert meteorological stations along the Pacific coast on the interpolated rain fields (Figs 6 and 7).

Patterns of drought-induced forest flammability

These results allow us to conduct an initial analysis of the potential effect of soil moisture depletion on forest flammability for the 1996–2001 study period. If we assume a 10 m rooting depth (Figs 9 and 10), no correction of ET (Figs 9 and 11), and a 12-year spin-up period with an initial PAW equal to 50% of PAW_{max}, then important spatial and temporal patterns of forest flammability emerge. First, there is a strong seasonal march of soil water depletion, as illustrated with data from 2001 (Fig. 12). The Amazon dry season begins in May along the southern rim of the region, advancing northward so that the dry season along the northern

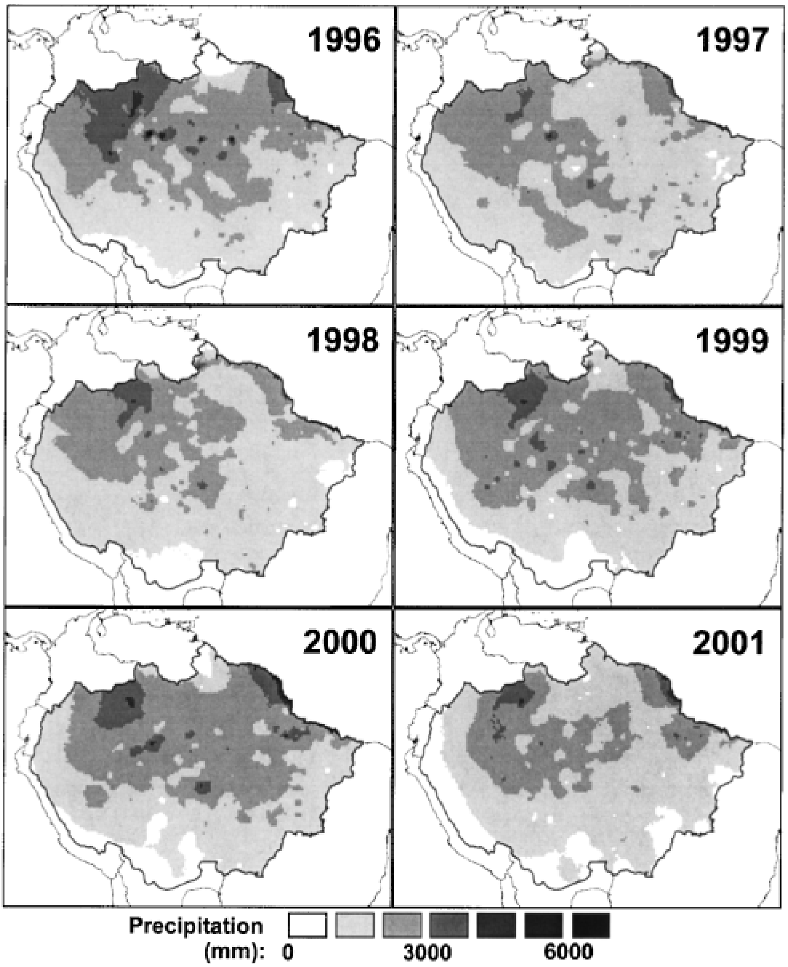

Fig. 6 Annual total precipitation in millimeters per year. Note: desert meteorological stations located along the Pacific coast, outside of the Amazon (Fig. 7) caused an underestimate of rainfall in the interpolated (kriged) rain field of the far western Amazon.

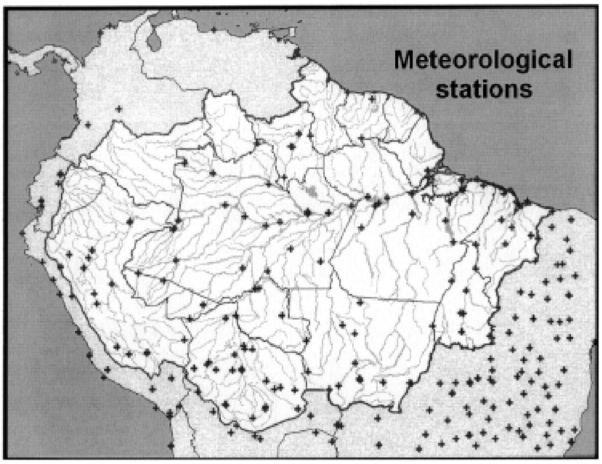

Fig. 7 Locations of meteorological stations (+) used to create maps of precipitation and evapotranspiration.

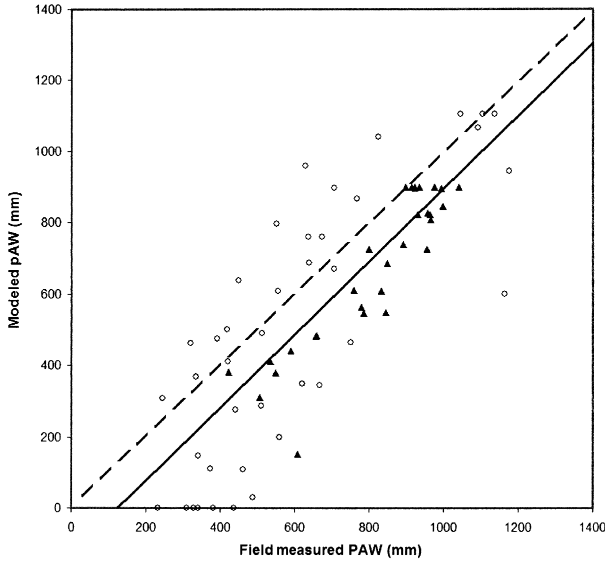

Fig. 8 Correspondence between field-measured and modeled % PAW_{max} for the Tapajós and Paragominas validation sites.

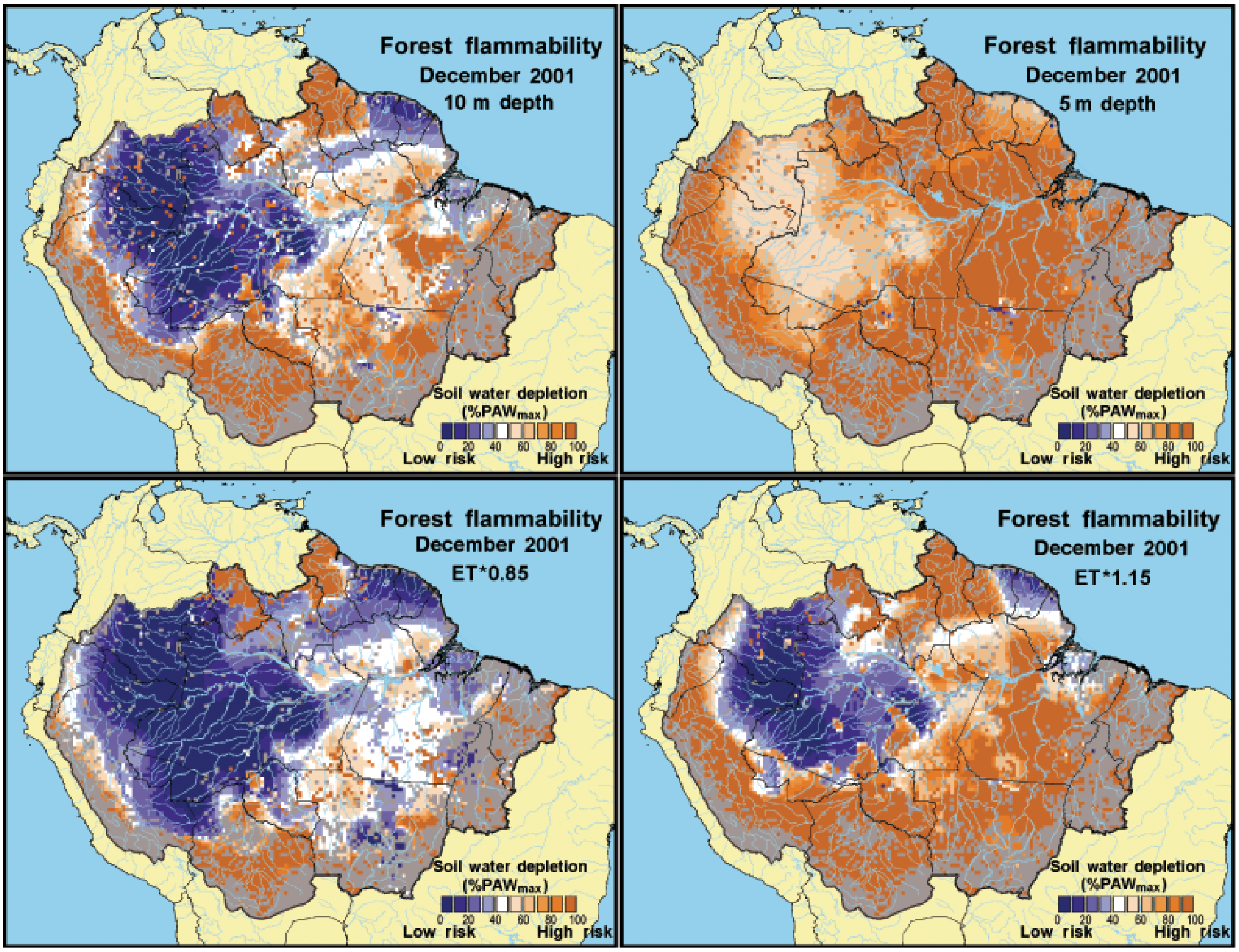

Fig. 9 Model output maps of % PAW$_{max}$ for December 2001, showing the effect of manipulating soil depth and evapotranspiration parameters. (a) Soil depth of 10 m, evapotranspiration unaltered; (b) soil depth of 5 m, evapotranspiration unaltered; (c) soil depth of 10 m with 15% depression in evapotranspiration; (d) soil depth of 10 m with 15% increase in evapotranspiration. See Note in Fig. 6 concerning underestimates of rainfall (and, hence, PAW) in the western Amazon.

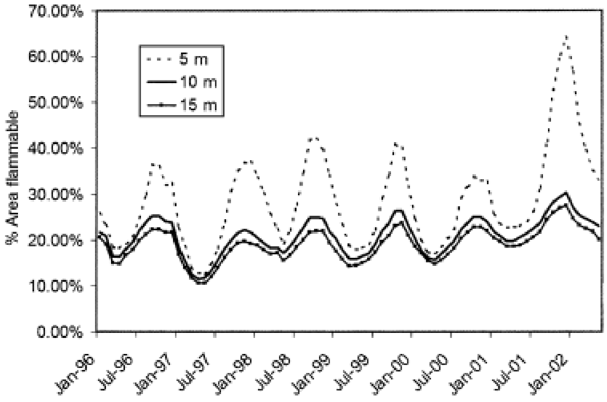

Fig. 10 Sensitivity of model to variations in rooting depth.

rim begins only in December. Hence, the end of the dry season, when soil water depletion is highest, occurs in September along the southern rim, in December along south-central Amazonia, and in March along the northern rim (Fig. 12, Table 1).

If the assumptions of this model run are accurate, then approximately 31% of the forests of Amazonia had less than 25% PAW$_{max}$ in December 2001, and may have been susceptible to fire (Fig. 12, Table 1). Moderate soil moisture depletion ($\leq$50% PAW$_{max}$) affected 51% of Amazon forests during this month (Table 1, Fig. 12).

Most of these flammable forests are located along the southern and eastern margins of the region, where the Amazon's agricultural and logging frontier is concentrated (Nepstad *et al.*, 1999a). Hence, the forests that are most susceptible to fire because of drought-induced reductions in LAI (Figs 1 and 2) are the most likely to

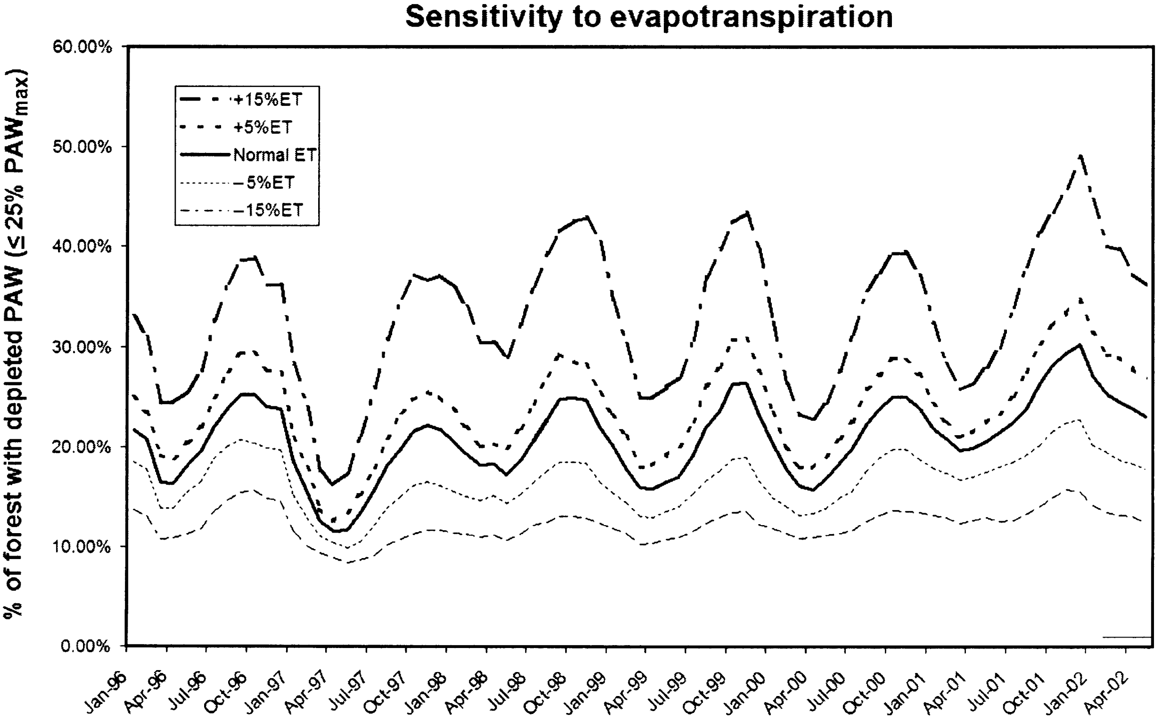

Fig. 11 Sensitivity of model to variations in evapotranspiration.

suffer logging and further increases in flammability (Uhl & Kauffman, 1990), and they are also the most likely to be ignited through escaped agricultural fires (Nepstad *et al.*, 1999b, 2001a).

Drought-induced reductions in tree growth

An initial estimate of the effect of severe soil moisture deficits on annual Amazon-wide carbon storage in tree stemwood was made by reducing the monthly stemwood increment from 0.2% of aboveground biomass to 0.1% of aboveground biomass during months when PAW fell below 25% of PAW_{max} (Fig. 3). Aboveground biomass was mapped using 1600 RADAMBRASIL (1968) forest inventories, to which allometric equations were applied (Brown & Lugo, 1982). The resulting forest biomass map (U. Lopes, unpublished data) provides values of forest biomass that are intermediate to those values summarized by Houghton *et al.*, (2002) for the Brazilian Amazon. Stemwood production tallied for 2001, the driest year that we studied, was 0.19 Pg of carbon less than stemwood production that would be expected during a year without drought-induced inhibition of stemwood production (i.e., without forested pixels that fell below 25% PAW_{max}). Hence, drought can potentially double the flux of carbon to the atmosphere associated with forest clearing and regrowth (0.2 Pg, Houghton *et al.*, 2000) by merely inhibiting tree growth, and can therefore counteract the net carbon sink that has been suggested by some studies for the region (Grace *et al.*, 1995; Phillips *et al.*, 1998).

Discussion

This study points to the widespread effect of drought on Amazon forests, and the vulnerability of Amazon forests to small declines in rainfall or increases in ET. Rainfall and ET are nearly equal across the Amazon during most years, with total rainfall falling below ET during years of severe drought. Such droughts may become more common if ENSO events continue to be frequent and severe, if rainfall is inhibited by deforestation or smoke, and if warming trends continue. Increases in ET of only 15% or similar reductions in rainfall can lead to severe soil moisture deficits over roughly half of the Amazon (Fig. 9).

The increase in forest flammability associated with severe drought poses one of the greatest threats to the ecological integrity of Amazon forests. By the end of 2001, we estimate that nearly one-third of the forests of this region had depleted all but 25% of plant-available soil moisture to a depth of 10 m (Fig. 9, Table 1). Milder drought during the 1997/1998 ENSO episode were associated with widespread burning that may have affected 40 000 km^{-2} of standing forest (Nepstad *et al.*, 1999a, 2001a; Mendoça *et al.*, in press). We do not

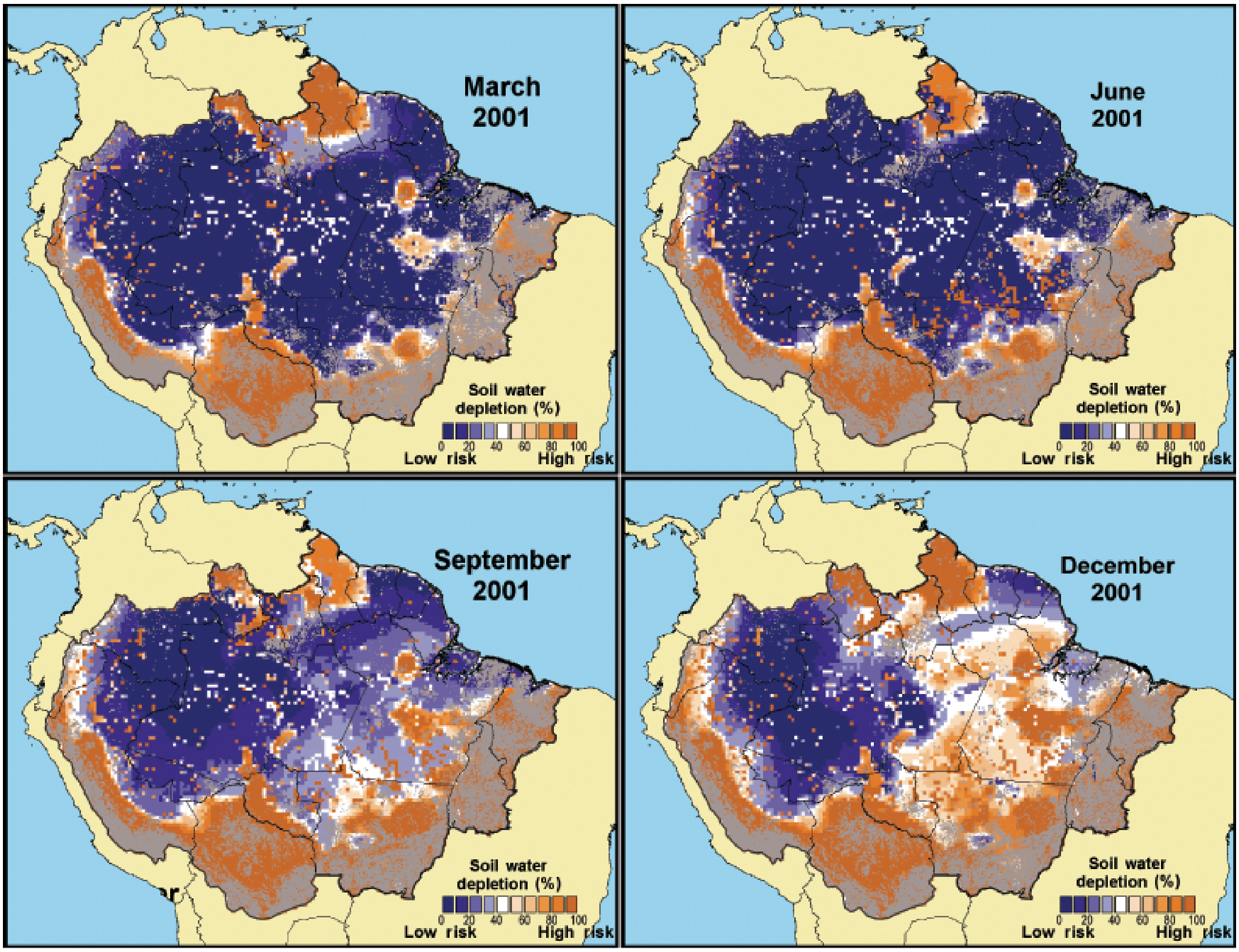

Fig. 12 % PAW_{max} for March, June, September, and December 2001 showing seasonal variability of soil moisture deficits. See Note in Fig. 6 concerning underestimates of rainfall (and, hence, PAW) in the western Amazon.

know the areal extent of forest understory fire in 2001 or 2002, but active fires detected by the weather satellite (NOAA-12, 14) were more abundant in 2002 than during any of the previous 6 years. Forest understory fires provoke high levels of tree mortality (Holdsworth & Uhl, 1997; Cochrane & Schulze 1998; Barlow *et al.*, 2003; O. Carvalho & D. Nepstad, unpublished data), and therefore trigger a large release of carbon to the atmosphere that is similar in magnitude to the annual release of carbon associated with deforestation and forest regrowth of 0.2 Pg (Houghton *et al.*, 2000; Mendoça *et al.*, in press) while increasing the probability of recurrent fire (Nepstad *et al.*, 1995, 2001a; Cochrane *et al.*, 1999).

Drought can also increase carbon emissions to the atmosphere by inhibiting NPP. The recent finding that tree stemwood growth is one of the most sensitive components of NPP to drought (Nepstad *et al.*, 2002) signifies that the long-term (decade-scale) storage of carbon in Amazon forest tree wood can be reduced by drought, corroborating the predictions of modeling experiments (Potter *et al.*, 1998; Tian *et al.*, 1998). Tree growth is also essential to polycyclic forest management in the Amazon region. Hence, greater drought would reduce the viability of sustainable forest management in the region, which is currently advanced as a promising basis for the regional economy (Barreto *et al.*, 1998; Verissímo *et al.*, 2002). We did not evaluate the Amazon-wide carbon emissions that may result from drought-induced tree mortality (Williamson *et al.*, 2000), which may substantially increase drought effects on forest carbon content and forest flammability, and further reduce the viability of forest timber management.

We present these findings as a preliminary assessment of the spatial and temporal patterns of drought in the Amazon and their implications for forest flammability and forest tree growth. There are many uncertainties present in this analysis that should be the themes of further research. We have validated our

Table 1 Percent of the forests of the Amazon by category of soil water depletion

% PAW_{max}	0–25%	25–50%	50–75%	75–100%
Mar-96	16	9	7	68
Mar-97	10	10	9	70
Mar-98	17	11	15	57
Mar-99	15	12	9	64
Mar-00	15	10	7	68
Mar-01	18	7	9	66
Jun-96	19	7	6	67
Jun-97	13	10	8	68
Jun-98	18	9	9	63
Jun-99	17	10	7	67
Jun-00	17	9	6	68
Jun-01	20	7	7	67
Sep-96	24	6	17	52
Sep-97	20	10	18	52
Sep-98	25	11	19	46
Sep-99	23	8	15	54
Sep-00	22	7	13	58
Sep-01	27	7	22	44
Dec-96	22	8	14	56
Dec-97	20	12	21	47
Dec-98	22	10	17	51
Dec-99	23	7	15	55
Dec-00	22	7	14	56
Dec-01	31	20	22	26

PAW_{max}, maximum plant-available soil water.

PAW model with field measurements from only two forests. We have assumed, conservatively, that the forests of the Amazon do not have root-impeding layers that would make them more susceptible to drought except for small patches of Lithosols (Fig. 4). And we have not attempted to map those forests that have shallow water tables supplied by subsurface flow from regions of high rainfall, where our model would underestimate PAW, nor have we estimated surface runoff. These are some of the topics warranting further investigation to better understand drought and its effects on Amazon forests.

Acknowledgements

This research was funded by grants from the National Aeronautics and Space Agency (NASA) Large-Scale Biosphere Atmosphere Experiment (LBA-ECO), Conservation International – Center for Applied Biodiversity Science (CI-CABS), the US Agency for International Development (USAID-Brazil), the US National Science Foundation, and the Programa Piloto pela Conservação das Florestas Tropicais Brasileiros (Ministry of Science and Technology, Brazil). The Woods Hole Research Center and Instituto de Pesquisa Ambiental da Amazônia provided institutional support. Carlos Nobre, Ane Alencar, and Gustavo Negreiros provided helpful input to the research.

References

Alencar A, Solórzano L, Nepstad D (in press) Modeling forest understory fire in an eastern Amazon landscape. *Ecological Applications*.

Barlow J, Lagan BD, Haugaasen T (2003) Large tree mortality and the decline of forest biomass following Amazonian wildfires. *Ecology Letters*, **6**, 6–8.

Barreto P, Amaral P, Vidal E *et al*. (1998) Costs and benefits of forest management for timber production in eastern Amazonia. *Forest Ecology and Management*, **108**, 9–26.

Brown S, Lugo AE (1982) The storage and production of organic matter in tropical forests and their role in the global carbon cycle. *Biotropica*, **14**, 161–187.

Canadell J, Jackson RB, Ehleringer JR *et al*. (1996) Maximum rooting depth of vegetation types at the global scale. *Oecologia*, **108**, 583–595.

Ceballos JC, Moura GBA (1997) Solar radiation assessment using Meteosat 4-VIS imagery. *Solar Energy*, **60**, 209–219.

Cochrane MA, Alencar A, Schulze MD *et al*. (1999) Positive feedbacks in the fire dynamic of closed canopy tropical forests. *Science*, **284**, 1832–1835.

Cochrane MA, Schulze MD (1998) Forest fires in the Brazilian Amazon. *Conservation Biology*, **12**, 948–950.

Costa MH, Foley JA (2000) Combined effects of deforestation and doubled atmospheric CO_2 concentrations on the climate of Amazonia. *Journal of Climate*, **13**, 18–34.

Curran LM, Caniago I, Paoli GD *et al.* (1999) Impact of El Niño and logging on canopy tree recruitment in Borneo. *Science*, **286**, 2184–2188.

Dawson TE (1993) Hydraulic lift and water use by plants: implications for water balance, performance and plant–plant interactions. *Oecologia*, **95**, 565–574.

Grace J, Lloyd J, McIntyre J *et al.* (1995) Carbon dioxide uptake by an undisturbed tropical rain forest in southwest Amazonia, 1992–1993. *Science*, **270**, 778–780.

Hodnett MG, Pimental da Silva L, da Rocha HR *et al.* (1995) Seasonal soil water storage changes beneath central Amazonia rainforest and pasture. *Journal of Hydrology*, **170**, 233–254.

Holdsworth AR, Uhl C (1997) Fire in Amazonian selectively logged rain forest and the potential for fire reduction. *Ecological Applications*, **7**, 713–725.

Houghton RA (1999) The annual net flux of carbon to the atmosphere from changes in land use1850–1990. *Tellus*, **51B**, 298–313.

Houghton RA, Lawrence KT, Hackler JL (2002) The spatial distribution of forest biomass in the Brazilian Amazon: a comparison of estimates. *Global Change Biology*, **7**, 731–746.

Houghton RA, Skole DL, Nobre CA *et al.* (2000) Annual fluxes of carbon from deforestation and regrowth in the Brazilian Amazon. *Nature*, **403**, 301–304.

Jackson RB, Schenk HJ, Jobbagy EG (2000) Belowground consequences of vegetation change and their treatment in models. *Ecological Applications*, **10**, 470–483.

Jipp P, Nepstad DC, Cassel K *et al.* (1998) Deep soil moisture storage and transpiration in forests and pastures of seasonally-dry Amazonia. *Climatic Change*, **39**, 395–412.

Kauffman JB, Uhl C, Cummings DL (1988) Fire in the Venezuelan Amazon 1: fuel biomass and fire chemistry in the evergreen rainforest of Venezuela. *Oikos*, **53**, 167–175.

Lean J, Bunton CB, Nobre CA *et al.* (1996) The simulated impact of Amazonian deforestation on climate using measured ABRACOS vegetation characteristics. In: *Amazonian Deforestation and Climate* (eds Gash JHC, Nobre CA, Roberts JM, Victoria RL), pp. 549–576. John Wiley & Sons, New York.

Malhi Y, Nobre AD, Grace J *et al.* (1998) Carbon dioxide transfer over a central Amazonian rain forest. *Journal of Geophysical Research*, **103**, 31593–31612.

Mendoça M, Diaz M, Nepstad D *et al.* (2004) The economic cost of the use of fire in the Amazon. *Ecological Economics* (in press).

Monteith JL (1965) Evaporation and environment. *Symposia of the Society for Experimental Biology*, **19**, 205–234.

Nepstad DC, Carvalho GO, Barros AC *et al.* (2001a) Road paving, fire regime feedbacks, and the future of Amazon forests. *Forest Ecology and Management*, **154**, 395–407.

Nepstad DC, Carvalho CJRd, Davidson EA *et al.* (1994) The role of deep roots in the hydrological and carbon cycles of Amazonian forests and pastures. *Nature*, **372**, 666–669.

Nepstad DC, Jipp P, Moutinho PRdS *et al.* (1995) Forest recovery following pasture abandonment in Amazonia: canopy seasonality, fire resistance and ants. In: *Evaluating and Monitoring the Health of Large-Scale Ecosystems* (ed. Rapport D), pp. 333–349. Springer-Verlag, New York.

Nepstad DC, Moreira AG, Alencar A. (1999b) *Flames in the Rain Forest: Origins, Impacts and Alternatives to Amazonian Fire. The Pilot Program to Conserve the Brazilian Rain Forest*. World Bank, Brasília, Brazil.

Nepstad DC, Moreira AG, Veríssimo A *et al.* (1998) Forest fire prediction and prevention in the Brazilian Amazon. *Conservation Biology*, **12**, 951–953.

Nepstad DC, Moutinho PRdS, Dias-Filho MB *et al.* (2002) The effects of rainfall exclusion on canopy processes and biogeochemistry of an Amazon forest. *Journal of Geophysical Research*, **107**, 1–18.

Nepstad DC, Moutinho P, Markewitz D (2001b) The recovery of biomass, nutrient stocks, and deep soil function in secondary forests. In: *The Biogeochemistry of the Amazon Basin and its Role in a Changing World* (eds McClain M, Victoria R, Richey R), pp. 139–155. Oxford University Press, Oxford.

Nepstad DC, Veríssimo A, Alencar A *et al.* (1999a) Large-scale impoverishment of Amazonian forests by logging and fire. *Nature*, **398**, 505–508.

Nobre CA, Sellers PJ, Shuklam J (1991) Amazonian deforestation and regional climate change. *Journal of Climate*, **4**, 957–988.

Page SE, Siegert F, Rieley JO *et al.* (2002) The amount of carbon released during peat and forest fires in Indonesia during 1997. *Nature*, **420**, 61–65.

Phillips OL, Malhi Y, Higuchi N *et al.* (1998) Changes in the carbon balance of tropical forests: evidence from long-term plots. *Science*, **282**, 439–442.

Potter CS, Davidson EA, Klooster SA *et al.* (1998) Regional application of an ecosystem production model for studies of biogeochemistry in the Brazilian Amazon. *Global Change Biology*, **4**, 315–333.

Rosenfeld D (1999) TRMM observed first direct evidence of smoke from forest fires inhibiting rainfall. *Geophysical Research Letters*, **26**, 3105–3108.

Saxton KE, Rawls WJ, Romberguer JS *et al.* (1986) Estimating generalized soil–water characteristics from texture. *Soil Science Society of America Journal*, **50**, 1031–1036.

Schenk H, Jackson R (2002a) Rooting depths, lateral root spreads, and below-ground/above-ground allometries in plants of water-limited environments. *Journal of Ecology*, **90**, 480–494.

Schenk H, Jackson R (2002b) The global biogeography of roots. *Ecological Monographs*, **72**, 311–328.

Setzer AW, Pereira MC (1991) Amazonia biomass burnings in 1987 and an estimate of their tropospheric emissions. *Ambio*, **20**, 19–22.

Shuttleworth WJ (1988) Evaporation from Amazonian Forests. *Proceedings of the Royal Society of London B*, **233**, 321–346.

Siegert F, Ruecker G, Hinrichs A *et al.* (2001) Increased damage from fires in logged forests during droughts caused by El Niño. *Nature*, **414**, 437–440.

Silva Dias MAF, Rutledge S, Kabat P *et al.* (2002) Cloud and rain processes in biosphere-atmosphere interaction context in the Amazon region. *Journal of Geophysical Research*, **107**, 8072, doi:10.1029/2001JD000335.

Tian H, Melillo JM, Kicklighter DW *et al.* (1998) Effect of interannual climate variability on carbon storage in Amazonian ecosystems. *Nature*, **396**, 664–667.

Timmermann A, Oberhuber J, Bacher A *et al.* (1999) Increased El Niño frequency in a climate model forced by future greenhouse warming. *Nature*, **395**, 694–697.

Tomasella J, Hodnett MG (1998) Estimating soil water retention characteristics from limited data in Brazilian Amazonia. *Soil Science Society of America Journal*, **163**, 190–202.

Trenberth KE, Hoar TJ (1997) El Niño and climate change. *Geophysical Research Letters*, **24**, 3057–3060.

Uhl C, Kauffman JB (1990) Deforestation, fire susceptibility and potential tree responses to fire in the eastern Amazon. *Ecology*, **71**, 437–449.

Uhl C, Kauffman JB, Cummings DL (1988) Fire in the Venezuelan Amazon 2: environmental conditions necessary for forest fires in the evergreen rainforest of Venezuela. *Oikos*, **53**, 176–184.

van Engelen VWP, Wen TT (1995) *Global and National Soils and Terrain Databases (SOTER). Procedures Manual (Revised Edition)*. UNEP-ISSS-ISRIC-FAO, Wageningen (also published as World Soil Resources Report 74 Rev.1).

Verissímo A, Cochrane MA, Souza Jr, C (2002) National forests in the Amazon. *Science*, **297**, 1478.

White A, Cannell MGR, Friend AD (1999) Climate change impacts on ecosystems in the terrestrial carbon sink: a new assessment. *Global Environmental Change*, **9**, S21–S30.

Williamson GB, Laurance WF, Oliveira AA *et al.* (2000) Amazonia tree mortality during the 1997 El Niño drought. *Conservation Biology*, **14**, 1538–1542.

Wright SJ, Carrasco C, Calderón O *et al.* (1999) The El Niño southern oscillation, variable fruit production, and famine in a tropical forest. *Ecology*, **80**, 1632–1647.

Wright IR, Nobre CA, Tomasella J (1996) Towards a GCM surface parameterisation for Amazonia. In: *Amazonian Deforestation and Climate* (eds Gash JHC, Nobre CA, Roberts JR, Victoria RL), pp. 473–504. John Wiley and Sons, Chichester, UK.