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Tel.: (12) 3625-4212



Influence of drainage status on soil and water chemistry, litter decomposition and soil respiration in central Amazonian forests on sandy soils

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Fabrcio Berton Zanchi^{1,2}; Maarten Johannes Waterloo²; Albertus Johannes Dolman²; Margriet Groenendijk²; Jrgen Kesselmeier³; Bart Kruijt⁴; Marcos Alexandre Bolson⁵; Flvio Jesus Luizao⁵; Antnio Ocimar Manzi⁵

¹Universidade Federal do Amazonas - UFAM, Instituto de Educao, Agricultura e Ambiente - IEAA

E-mail: faberzanchi@gmail.com

²VU University, Department of Hydrology and Geo-environmental Science

E-mail: {maarten.waterloo, han.dolman, margriet.groenendijk}@falw.vu.nl

³Max Planck Institute for Chemistry, Biogeochemistry Department

E-mail: j.kesselmeier@mpic.de

⁴Alterra, Wageningen University and Research

E-mail: bart.kruijt@wur.nl

⁵Instituto Nacional de Pesquisa da Amaznia - INPA

E-mail: {mbolson, fluizao, manzi}@inpa.gov.br

ABSTRACT

Central Amazonian rainforest landscape supports a mosaic of tall *terra firme* rainforest and ecotone *campinarana*, riparian and *campina* forests, reflecting topography-induced variations in soil, nutrient and drainage conditions. Spatial and temporal variations in litter decomposition, soil and groundwater chemistry and soil CO₂ respiration were studied in forests on sandy soils, whereas drought sensitivity of poorly-drained valley soils was investigated in an artificial drainage experiment. Slightly changes in litter decomposition or water chemistry were observed as a consequence of artificial drainage. Riparian plots did experience higher litter decomposition rates than *campina* forest. In response to a permanent lowering of the groundwater level from 0.1 m to 0.3 m depth in the drainage plot, topsoil carbon and nitrogen contents decreased substantially. Soil CO₂ respiration decreased from 3.7±0.6 μmol m⁻² s⁻¹ before drainage to 2.5±0.2 and 0.8±0.1 μmol m⁻² s⁻¹ eight and 11 months after drainage, respectively. Soil respiration in the control plot remained constant at 3.7±0.6 μmol m⁻² s⁻¹. The above suggests that more frequent droughts may affect topsoil carbon and nitrogen content and soil respiration rates in the riparian ecosystem, and may induce a transition to less diverse *campinarana* or short-statured *campina* forest that covers areas with strongly-leached sandy soil.

Keywords: Amazon rainforest; Drought experiment; *Campina*; Soil Nutrients; CO₂ Efflux.

Influência da drenagem na química do solo e da água, na decomposição da serapilheira e na respiração do solo em floresta de solo arenoso na Amazônia central

RESUMO

A floresta Amazônica central possui um mosaico de floresta tropical de terra firme, ecotones de *campinarana*, florestas ripárias e ecossistema de *campina*, que reflete a variação induzida pela topografia sobre as condições de solo, nutrientes e drenagem. Foram estudadas em florestas em solo arenoso as variações espacial e temporal da decomposição da liteira,

química do solo e da água do solo, e a respiração do solo, uma vez que a sensibilidade à seca estava sendo investigada em solos de vale pobremente drenado com um experimento de drenagem controlada. Pequenas mudanças foram observadas na decomposição da liteira e na química da água do solo devido à drenagem artificial. A floresta ripária experimentou maior taxa de decomposição do que o ecossistema de *campina*. Em resposta ao permanente rebaixamento do nível do lençol freático de 0.1 m para 0.3 m de profundidade, promovido pelo experimento de drenagem, o carbono e nitrogênio do solo superficial diminuíram substancialmente. A respiração do solo diminuiu de $3.7 \pm 0.6 \mu\text{mol m}^{-2} \text{s}^{-1}$ antes da drenagem para 2.5 ± 0.2 e $0.8 \pm 0.1 \mu\text{mol m}^{-2} \text{s}^{-1}$ em oito e onze meses depois da drenagem, respectivamente. Na parcela de controle, a respiração do solo permaneceu em $3.7 \pm 0.6 \mu\text{mol m}^{-2} \text{s}^{-1}$. Isso sugere que secas frequentes podem afetar o carbono e nitrogênio do solo superficial e as taxas de respiração dos ecossistemas ripários, e podem também induzir a transição dessas áreas para uma floresta de menor diversidade como a *campinarana* ou a vegetação de *campina* que cobrem áreas com solos arenosos fortemente lixiviados.

Palavras-chave: Floresta Amazônica; Experimento de drenagem do lençol freático; *Campina*; Nutrientes do solo; Efluxo de CO₂.

1. INTRODUCTION

The Amazon region (6.6 million km²) contains more than half of Earth's remaining tropical rain forest (Houghton et al., 1996), which accounts for 30 – 50% of the total global primary production (Dixon et al., 1994). Over the past decades about 750,000 km² of the Brazilian Amazon has been deforested (Nobre and Borma, 2009) which, 11.968 km² y⁻¹ between 2007 and 2008 (INPE, 2008). The large-scale conversion of rain forest to agricultural land may cause changes in local, as well as global water and carbon cycles (Taylor and Lloyd, 1992). Changes in the hydrological cycle may potentially lead to dangerous positive feedback effects in the carbon cycle through tree mortality (drought), changes in CO₂ emissions and in the forest CO₂ sink strength, decreased biomass production and reduced evapotranspiration by the remaining forest. Increases in the frequency of forest fires during dry periods may also be expected as a consequence of climate change induced droughts (Aragão et al., 2008). These effects enhance changes in global climate even further reported by Cox et al., (2000, 2004) a strong climate feedback towards reduced vegetation and soil carbon, predicting decreases of about 128 Gt C for the period between 1860 and 2100.

Climate models predict a global temperature rise of 1 to 6°C until the 21st century (IPCC, 2007). This warming, and the associated enhanced heterotrophic respiration, may force changes in Amazon rain forest that could lead to a future climate without analogue in the recent past (Maslin et al., 2005).

Several Global Circulation Models (GCM) simulations have predicted a dieback of rain forest in the Amazon Basin (Cox et al., 2004; Huntingford et al., 2008). The GCM showed an enhanced of soil carbon emissions, whereas reducing the size of the respiring soil carbon pool until the absolute respiration flux as a consequence of global warming. Other GCM simulations of deforestation and subsequent savannisation of part of the Amazon region indicate enhanced forest fragmentation, with associated edge effects, and an increase in temperature and decrease in precipitation (Costa and Foley, 2000; Oyama and Nobre, 2003; Cox et al., 2000; Lewis et al., 2011; Li et al., 2006). This scenario is likely to cause further mortality and ecosystem decay (Laurance et al., 2002; Hutyra et al., 2005). Recently, Nobre and Borma (2009) have identified two thresholds for the stability of the Amazon forest-climate equilibrium associated with global warming and deforestation, which are perceived as the main threats for Amazon forests. Higher risks of forest dieback or savannisation over

large portions of the Amazon Basin was foreseen beyond a global warming of 3 - 4°C, or when deforestation in the Amazon Basin would exceed 40% of the initial forest area.

These models studies all treat the rain forest as a homogeneous cover of the dominant, well-drained upland *terra firme* forest, which grows on clayey to loamy soils (Oxisols, Ultisols) and covers about 70% of Amazonia (Luizão, 1996). This forest type has close to optimal photosynthesis conditions under current climate and soil moisture conditions. In central Amazonia, unconsolidated sedimentary layers have been dissected by rivers and creeks over time, which lead to formation of a landscape with rather flat plateaus being abruptly separated by steep slopes from broad swampy river valleys (Chauvel et al., 1987; Waterloo et al., 2006). Different forest types have developed within this landscape in response to topography-controlled distribution of soil types (clay content), phreatic levels and nutrient availability (Singer and Aguiar, 1986; Walker, 1987; Laurance et al., 1999). As a consequence, the Amazonian *terra firme* forest landscape is actually made up of several different forest types, that each may cover substantial areas. These types include seasonally inundated *varzea* or *igapó* valley forests along the major rivers, *campinarana* ecotone areas and riparian valley forests along the smaller rivers and low-statured *campina* forests *sensu* Anderson et al. (1975) that occur on strongly leached sandy soils (Prance and Schubart, 1978). *Campinarana* and *campina* forests cover approximately 6% of Amazonia (McClain et al., 1997; Luizão et al., 2007). The presence of such a variation of forest types (*campinarana* and *campina*) in the landscape may reduce the overall sensitivity of its carbon cycle with respect to a faster carbon turnover or changes in ecosystem evapotranspiration and groundwater recharge (Richey et al., 2002) from all different Amazonian forests.

Drought experiments in Amazonia have been exclusively conducted in the plateau *terra firme* forests. The drought studies by Meir et al., (2008), Nepstad et al., (2002) and Davidson et al., (2008), suggesting that tall *terra firme* forest on well-drained soils are not very vulnerable to reductions in rainfall over periods of less than three years, but do respond to longer periods of drought (Lewis et al., 2011), which cause increases in tree mortality and up to 60% reduction in above-ground net primary productivity (Nepstad et al., 2007; Brando et al., 2008). Meir et al., (2008, 2009) also reported 20-30% reduction in the leaf area index, 30-40% in transpiration rates and a 20% reduction in soil CO₂ emissions. .

Whilst the well-drained Oxisols and Ultisols on slopes and plateaus have a significant clay fraction, the poorly-drained valley soils consist of predominantly of pure quartz sand (Spodosols) and have little capacity to retain water or nutrients (Singer and Aguiar, 1986; Chauvel et al., 1987). Riparian and ecotone *campinarana* forests grow on these poor sandy soils and experience almost permanently saturated soil conditions, which are maintained by a continuous flow of groundwater from slope and plateau areas, where higher groundwater levels are maintained by recharge during wet periods (Hodnett et al., 1997; Tomasella et al., 2008).

Malhi and Phillips (2004) have reported that tropical forests are dynamic ecosystems of constantly shifting composition and structure. As the riparian forest, and to a somewhat lesser extend the ecotone *campinarana* forest, is adapted to near-permanent saturated soil conditions, a frequent or permanent lowering of the groundwater level due to changes in upland groundwater recharge may induce changes in the valley soil and forest composition and functioning. Indeed, Walker (1987) already perceived desertification of Spodosol areas in the Amazon as a real threat if climate should get drier.

In central Amazonia, short-statured *campina* (heath) forests grow on strongly-leached sandy soils, with a texture similar to the valley soils, but with lower phreatic levels. The main characteristics that distinguish the *campina* forest type from other forest types are its low species richness and dominance of a single or very few tree species, its unusual physiognomy

- shorter stature, multi-branched and tortuous trees and the presence of bushes with scleromorphic leaves (Anderson et al., 1975; Anderson, 1981; Richards, 1996).

Cowling et al. (2004) suggest that the predominant mechanisms for maintenance of forest in cool periods are the reduced evapotranspiration and lowered respiration costs and that, conversely, tropical rain forests may now be near an upper temperature threshold where these physiological mechanisms become positive feedbacks that could induce forest dieback. Changes in the frequency of valley soil saturation conditions may induce changes in soil carbon and nutrient retention capacities, partly because of the poor soil nutrient retention capacity (Chauvel et al., 1987). This could potentially lead to local shifts in the composition of riparian forest towards adjacent lowland ecotone *campinarana* forest or to *campina* forest e.g. *savannization*; (Cox et al., 2000; Oyama and Nobre, 2003) with its particular carbon cycling. If valley forest evapotranspiration is reduced due to such a shift in forest composition towards *campina* forest, this may lead to a further increase in air temperature. This is in line with reports that the tropical rain forest will increase its relative tolerance due to the temperature rise and that it is inevitable that tropical forest composition will change in response to atmospheric change, as particular species are favoured by higher CO₂ concentrations and changing climate conditions (Malhi and Phillips, 2004).

CO₂ gas in the soil, which eventually evades as soil respiration, can be generated by various processes. These include biotic processes, such as respiration of roots, microbes or macro-fauna in the soil. Such biotic processes are affected by temperature, as well as by moisture content. Under very dry soil conditions the ecosystem is known to have lower respiration rates (Sotta et al., 2004; Davidson et al., 2008). This mechanism was explained by Linn and Doran (1984), who observed that under very dry soil conditions the substrate diffusion through water films around soil particles to microbial active cells becomes limited. Soil CO₂ can also be generated through chemical processes, such as oxidation of soil organic matter, which may lead to either fast or slow carbon loss, depending on aeration status, soil pH and the presence of chemical components to interact with soil organic matter (and litter).

For a better understanding of the implications of climate change on the rain forest carbon cycle in central Amazonia, we need to study changes in soil carbon (Phillips et al., 2009), nutrients and heterotrophic respiration in response to lowered groundwater levels in the sandy riparian zone, which is likely to be most affected by reduced rainfall. The results presented here describe observed variations in soil respiration, groundwater levels, soil nutrients and litter decomposition in adjacent riparian forest, *campinarana* forests and in a separate *campina* forest. In addition, we report on a study of the drought response of soil in a saturated valley area where the phreatic level was artificially lowered using drainage tubes. This study allows assessment of the rate and magnitude of changes that may typically occur during extended periods of drought in sandy valley soils in central Amazonia and may provide clues on potential changes in the vegetation that is adapted to high phreatic levels as maintained by continuous groundwater flow from the plateaus and slopes.

2. MATERIAL AND METHODS

2.1. Site description

Three measurement sites were established in the *Reserva do Cuieiras* - ZF2 (2° 36' S, 60° 12' W), which includes the Igarapé Asu catchment (Waterloo et al., 2006). The Reserve belongs to the *Instituto Nacional de Pesquisa da Amazonia* (INPA) and is about 70 km north of Manaus along the Manaus - Boa Vista highway. The *Reserva do Cuieiras* study site has extensively been described by Araújo et al., (2002, 2010). The topography is typical for central Amazonia consisting of rather flat plateaus (90-105 m a.s.l.) incised by broad swampy valleys (45-55 m a.s.l.) with moderately steep slopes (15°-30°; Waterloo et al., 2006). The

soils on slopes and plateaus are Oxisols and Ultisols with a significant clay fraction and cover 57% of the area (Chauvel et al., 1987; Rennó et al., 2008). The valley soils cover the remaining 43% of the area and consist of strongly leached quartz sands (Spodosols) with a low capacity to retain water or nutrients (Chauvel et al., 1987; Brinkmann, 1985; Waterloo et al., 2006). The dominant forest type on the well-drained, clayey soils on slopes and plateaus is tall *terra firme* rain forest with tree heights varying between 25 m and 45 m (Oliveira et al., 2008; Rennó et al., 2008). The poorly-drained broad valleys are covered by *campinarana* and riparian forests, which are characterised by tree heights of up to 25 m and have a higher abundance of palms. Phreatic levels within the research area ranged from well over 30 m below the soil surface on the plateau to less than 0.1 m depth in the riparian forest (Tomasella et al., 2008).

The drainage experiment was conducted in riparian forest using drainage and control plots established 50 m apart along the Igarapé Asu river in the swampy valley bottom (Figure 1), where the phreatic level is usually at less than 0.1 m below the soil surface (Tomasella et al., 2008). Both plots had similar soil, vegetation composition, hydrologic and micro-climatic conditions. Ecotone *campinarana* forest grows on sandy soils in the zone between the riparian forest and the tall *terra firme* forest in central Amazonian catchments (Costa et al., 2005; Drucker et al., 2008). This forest type shares many plant species with riparian forest and covers an area of about 750,000 km² in the Amazon (Hutyra et al., 2005). A *campinarana* forest plot was established close to the riparian plots, about 300 m away from the Igarapé Asu stream. This site had a lower phreatic level at an average of 0.8 m below the surface (Figure 2).

A fourth plot, representing short-statured *campina* forest growing on well-drained, strongly-leached white sands at a higher topographic position in the landscape, was established in INPA's *Reserva de Campina* (2° 59' S, 60° 03' W) located along the BR-174 highway 45 km north of Manaus, Amazonas State, Brazil. The size of the reserve is approximately 9 km² and the vegetation consists predominantly of lowland evergreen rain forest, but also includes patches of stunted heath forest (*campina*, 0.03 km²) and taller heath forest (*campinarana*, 0.04 km²). The *campina* forest at this site is characterised by small patches of trees and shrubs that are surrounded by areas of bare soil *i.e.* sun *campina sensu* (Anderson et al., 1975). Tree heights range between 4 m and 7 m and between 10 m and 18 m for *campina* and *campinarana i.e.* shade *campina sensu* (Anderson et al., 1975) forests, respectively (Luizão, 1996; Luizão et al., 2007). The phreatic level is deeper than at the other sites at about 3 m below the surface and the coarse grained soil is conducive to excessive drainage and is poor in nutrients (Prance and Schubart, 1978).

The climate in central Amazonia does not vary much, the annual average temperature is 26.7°C and relative humidity is about 80%. Annual rainfall, measured at the Ducke rain forest reserve near Manaus (1966-1992), amounts to 2442 mm, with a standard deviation of 306 mm. The rather weak dry season is between June and November (Hodnett et al., 1997; Araújo et al., 2002; Waterloo et al., 2006).

2.2. Measurements

Drought experiments in other parts of central Amazonia used throughfall exclusion to simulate drought e.g. (ESECAFLOR - Meir and Grace, 2005; Nepstad et al., 2002). Throughfall exclusion could not be used to simulate drought in valleys because of the constant supply of groundwater from the plateaus. Hence, we designed a method to simulate drought by artificially lowering the phreatic level without excluding rainfall. To achieve such lowering of the phreatic level, we installed two agricultural drainage tubes in parallel with 15 m spacing (Figure 1). Installation of these tubes started on March 1st and finished on March 10, 2007. Each tube had a length of 30 m and 0.2 m diameter and was installed ~1 m below

the soil surface. To avoid having to cut through large roots, the tubes were positioned at least 4 meters away from large trees. Backfilling of the trenches was done in such a way to preserve the original soil profile as much as possible.

Soil respiration was measured at all plots with a single automated soil CO₂ flux system (LI-8100, LI-COR, Nebraska, USA) with a 0.2 m diameter chamber. The system was set up to measure soil CO₂ flux, as well as soil moisture content and soil temperature (both measured at 5 cm depth) at 30-minute intervals. The instrument was consecutively used in the various plots. In each plot, measurements were made over periods of four to seven consecutive days at three locations in the plot before it was moved to the next plot. In this way information on both spatial and temporal variations could be obtained for each plot and average values reported for each plot therefore include both variations in time and space.

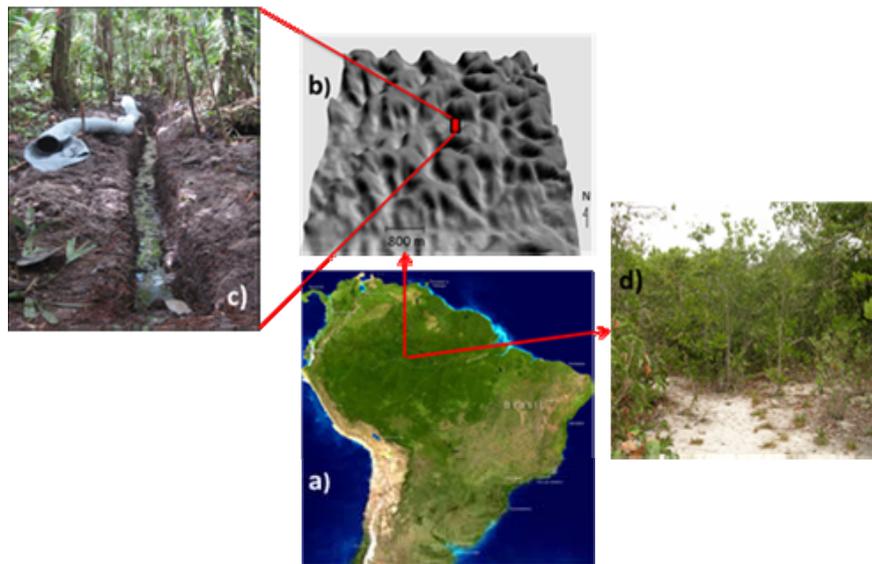


Figure 1. Location of research sites and trench installation: a) South America View from Google maps with arrow pointing to the ZF2 experimental site in central Amazonia, b) riparian zone (SRTM image) with broad swampy valleys and moderately steep slopes in the Cuieiras Reserve where the drainage experiment was done, c) is the drainage tube installation in trench at 1 m depth in the riparian forest plot and d) is the short-statured *campina* reserve forest.

Decomposition of leaf litter was measured according to the litter bag method of Bock et al. (1957). We used sixty litter bags with 1 mm mesh size and larger lateral holes to allow entrance of larger insects. Each bag contained 5.75 g of dried (at 70°C) fresh leaf material collected from trees in the same plots. The experiment started at all sites on 6 December 2006, when the bags were installed on the ground surface and spread out in 12 blocks of 5 bags each. Five bags were retrieved randomly at each site every month to determine dry weight loss. A double exponential model, used earlier by Luizão and Schubart (1987) and based on that of Carpenter (1982), was used to estimate times of 50% and 95% initial mass loss rates, based on measured mass loss rates over the period of a year.

Two piezometers were installed 2 m apart in the drained plot with their screens at 0.75-1.25 m below the surface. The piezometers were installed parallel to the drainage tubes at a distance of 4 m from the tubes using hand-operated bailer boring equipment (Eijkelpamp Agrisearch Equipment, The Netherlands) on March 7, 2007. A single piezometer was installed in the control plot, with its screen at a depth of 1.62-2.12 m below the surface on

March 16, 2007, about 50 m away from those in the drained plot. The fourth piezometer was installed in the *campinarana* forest at this site on the March 8, 2007.

A fifth piezometer was installed in the *campina* forest plot on January 26, 2007, with its screen at 3.62-4.12 m below the surface, reflecting the much lower phreatic level in this plot. The screen depths were such to remain below minimum dry season phreatic levels. Groundwater levels were measured weekly with an acoustic sounding device (Eijkelpamp Agrisearch Equipment, The Netherlands). In addition, groundwater samples were collected at monthly intervals from all piezometers, after flushing at least three times the tube volume using an electric pump to sample fresh groundwater.

Three sets of six vacuum-type ceramic cup soil water samplers were installed at depths of 0.1, 0.2, 0.3, 0.4, 0.5 and 0.6 m in the drained, control, *campinarana* and *campina* forest plots to obtain samples from soil moisture and shallow groundwater. Before installation, all cup samplers were rinsed with a diluted HNO₃ solution (pH=1) to remove any sorbed ions and flushed with distilled water until the EC of water used for flushing fell below 2 µS cm⁻¹. Samples from the three sets of cup samplers in each plot were bulked, yielding single samples for each depth after each sampling event.

Water samples were analysed for Dissolved Organic Carbon (DOC), Dissolved Inorganic Carbon (DIC) and inorganic ions. DOC/DIC samples were filtered in situ with pre-ashed glass fiber filter (0.7 µm Whatman GFF, USA) and then stored in a 25 ml glass bottle with a Teflon lid to prevent contamination. HgCl₂ (at 300 µM) was added for preservation of the sample. DOC and DIC concentrations were analysed shortly after sampling on a TOC analyser (TOC5000A; Shimadzu, Japan) in the laboratory at INPA. Separate samples were collected for analyses of major cations (Na⁺, K⁺, Mg²⁺, Ca²⁺, NH₄⁺) and anions (Cl⁻, SO₄²⁻, PO₄³⁻, NO₃⁻, NO₂⁻). These samples were filtered using disposable cellulose-acetate filters (0.47 µm, Whatman GFF, USA) and stored in high-density 100 ml polyethylene bottles. Sample preservation was achieved by adding few drops of a 100 mg L⁻¹ Thymol solution and then analysed on a Dionex ion chromatograph (DX500, USA).

Soil samples were collected with a hand auger set (Eijkelpamp Agrisearch Equipment, The Netherlands) at three locations within each plot to account for spatial variation. These samples were then bulked according to depth (0-0.5, 0.5-0.10, 0.20-0.30, 0.30-0.40 and 0.60-0.70 m) to limit the number of samples to be analysed. Sampling occurred both in control and drained plots just before installation of the drainage tubes (February 7, 2007) and sixteen months after the installation of the drainage tubes (July 18, 2008). Soil samples were collected in a similar fashion in the *campina* plot. All samples were stored in plastic bags for subsequent drying and analyses in the laboratory.

Soil sample analyses were done at the *Laboratório Temático de Solos e Plantas* – LTSP of INPA according to the methods described by Silva (1999). Analyses of %C and %N were done on finely ground soil sample triplicates using an Elemental Analyser (NA 1500 W Series 2, Fisons Instruments, UK). Extraction of exchangeable P, K, Fe, Mn and Zn was done according to the Mehlich I procedure, whereas exchangeable Ca and Mg were extracted in a 1N KCl solution as described in Silva (1999). Potential acidity (exchangeable Al³⁺H) was measured in a 0.5N Ca-acetate solution at a pH of 7. The obtained solutions were analysed with an atomic absorption spectrometer (1100B, PerkinElmer, USA). Soil pH and pH_{KCl} were measured in 1:2.5 soil-water and 1:2.5 soil 1N KCl solutions after equilibrating for 1 h and repeated shaking (Silva, 1999). The pH was measured with a laboratory pH meter (mPA210/mPA-210P, Tecnocon, Brazil) calibrated with pH buffers of 4.00 and 7.00. Electrical conductivity (EC) was measured with a WTW conductivity meter (315i EC-meter, USA), calibrated against a 1460 µS cm⁻¹ 0.01M KCl solution.

At the ZF2 site, 30-minute precipitation totals were measured at 51 m above ground level on top of a micro-meteorological tower with a tipping bucket rain gauge (EM ARG-100, UK;

0.2 mm resolution) connected to a CR10X datalogger (Campbell Scientific, USA). Rainfall was measured with a similar system, but at about 3 m above the canopy in the *campina* forest.

3. RESULTS AND DISCUSSION

3.1. Soil drainage status

Time series of groundwater levels in the riparian forest drainage and control plots, the *campinarana* plot and in the *campina* forest plot are shown in Figure 2. Visual inspection of the drainage and control plots before the start of the experiment showed that the soil was completely saturated and water was on the surface at both plots. Groundwater level measurements in the control plot started only two months after installation of the drainage tubes and the phreatic level remained close to the surface at all times. The mean water level depth was 0.12 ± 0.06 m ($n=36$). Two days after the installation of the tubes (10 March 2007) the phreatic level in the drained plot had already decreased to 0.21 m and the phreatic level remained afterwards below that of the control plot, varying between depths of 0.24 m and 0.45 m below the surface, with a mean depth of 0.35 ± 0.04 m ($n=36$).

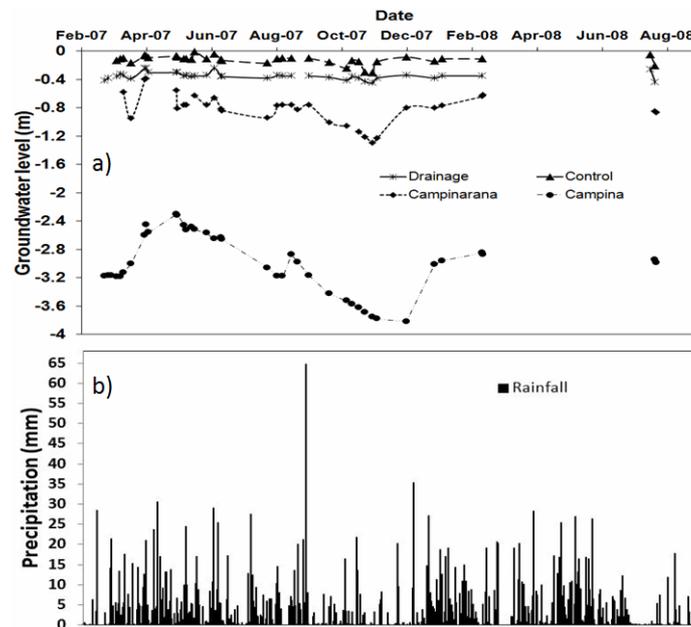


Figure 2. a) Changes in phreatic levels at the four research plots as observed from weekly measurements; b) hourly rainfall in the Igarapé Asu catchment (ZF2).

The installation of the drainage tubes therefore effectively lowered the phreatic level permanently by 0.23 m in the drained plot as compared to that in the control plot. Seasonality in the phreatic level is virtually absent in the riparian forest that is fed by a constant supply of deep groundwater originating from the plateau (Hodnett et al., 1997; Tomasella et al., 2008). However, due to the sandy, permeable nature of the soil, sudden drops can occur during dry periods, such as in October-November 2007 when the phreatic level in the control plot fell to a depth of 0.31 m below the surface in response to dropping groundwater levels on the plateau. However, these dry periods generally do not last longer than a few weeks under the current climate regime in central Amazonia. The phreatic level in the drainage plot seems to exhibit less variation than that in the control plot, but also did show a response to drought in periods of low rainfall (October-November, 2007).

The *campinarana* plot, located between the plateau/slope and riparian area, experienced lower phreatic levels ranging between depths of 0.39 m and 1.30 m below the surface, with a mean depth of 0.81 ± 0.21 m ($n=32$). Different combinations of soil texture, drainage status and nutrient availability impact on the forest cover in central Amazonia, with tall *terra firme* rainforest on clayey soil and short statured *campina* forest on white sands occupying both ends of the spectrum in non-flooded areas with respect to the cycling of carbon, mineral nutrients and species composition (Guillaumet, 1987; Walker, 1987; Luizão et al., 2004). Our studies in riparian, ecotone *campinarana* and *campina* forests that grow on sandy soils with little capacity to retain nutrients and under varying drainage conditions, indicate that there are clear differences in litter decomposition, soil and water chemistry and soil CO₂ respiration.

The phreatic level in the *campina* forest was lowest, with depths ranging between 2.3 m and 3.8 m below the surface and with a mean depth of 2.99 ± 0.42 m ($n=39$, Figure 2). Phreatic levels in the *campinarana* and *campina* plots were therefore much lower than those experienced by the riparian forest and also showed a much more pronounced seasonal variation, especially in the *campina* forest plot (Figure 2). The well-drained *campina* forest is an infiltration zone due to its high position in the landscape (Prance and Schubart, 1978), whereas the *campinarana* and riparian forests receive groundwater from upland areas (exfiltration zones) and therefore experience much higher phreatic levels (Hodnett et al., 1997; Waterloo et al., 2006; Tomasella et al., 2008). A slight to moderate reduction in rainfall presumably would not reduce evapotranspiration by the tall *terra firme* forest on the higher parts in the landscape, but may reduce groundwater recharge in these upland areas significantly causing a lowering of the phreatic level below the plateaus. This in turn would limit the flow of groundwater to the valley, reducing stream discharge and causing a higher temporal variation in the phreatic levels in the valley.

3.2. Litter decomposition rates

Litter decomposition rates observed in the riparian forest plots indicated a rapid initial loss of leaf mass in the first month (up to 25% of the initial mass loss for individual samples), followed by a more gradual decline to about 73% mass loss a year after installation (Figure 3). The decomposition rate was much lower in the *campina* forest, where only 35% of litter mass was lost in the first year. About 50% of weight loss occurred after 210 days in both control and drainage plots, whereas in the *campina* forest the 50% mass loss occurred only after 480 days. The data indicated that 95% mass loss would occur after 925 days in the control plot, 955 days in the drainage plot and only after 1825 days in the *campina* forest plot (Figure 3), it was similar to those observed by Luizão and Schubart (1987), who obtained corresponding estimates of 218 days (50% of weight loss) and 1006 days (95% of weight loss), for dry season conditions based on a 150-day measurement period.

Luizão and Schubert (1987) observed that litter decomposition was faster on the plateau than in the valley of this *terra firme* landscape and that decomposition rates were higher during the wet season (50% mass loss in 32 days) than during the dry season. The latter observation is not supported by our measurements that started early in the wet season and continued throughout the dry season. According to Luizão et al. (2004), litter turnover rates on plateau and slope were not significantly different from those in the valley in spite of lower N concentrations in the upper soil layer in the valley, which they attributed to other processes, such as leaching, removing litter from the soil surface.

A permanent decrease in phreatic levels or a change in the frequency and lengths of droughts may impact the local decomposer community, and thereby decomposition rates of leaf litter. However, no significant differences were found between decomposition rates in the drained plot, where the phreatic level was artificially lowered by about 0.3 m, and the control plot (Figure 3). This may have several causes. First, the decomposition experiment started

four months before drainage tubes were installed and about 40% of the initial litter mass had therefore already been lost before the phreatic level was lowered. Furthermore, the size of the drained plot may have been too small (edge effects – micro-climate and vegetation) and the period of study after drainage started (1 year) too short for adaptation of the decomposer community to the new drainage conditions in the plot.

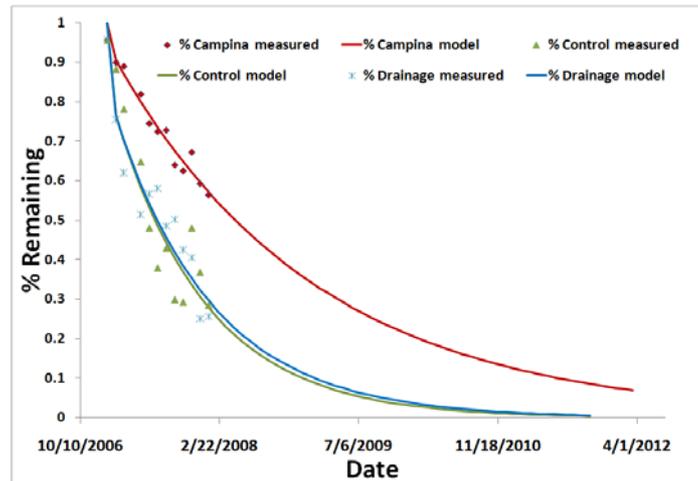


Figure 3. Remaining mass fraction of organic material in litterbags installed in riparian forest drainage and control plots (ZF2) and in the *campina* forest plot.

There may be several reasons for the slow decomposition observed in *campina* forest. Tree species in *campina* forest do have sclerophyllous leaves that may be more difficult to decompose and studies have shown that the decomposition activity of fungi is also suppressed in this environment, where ectomycorrhizae are instrumental in the cycling of nutrients (Singer and Aguiar, 1986). Walker (1987), Luizão and Schubart (1987) and (Luizão, 1996) have reported that even the high annual rainfall, which percolates rapidly through the nutrient-poor sandy soils, may not be sufficient to transport sufficient nutrients for the synthesis of adequate quantities of structural proteins (endoplasmatic reticulum, membranes, histones, etc.) in order to balance intense assimilation. This could be a cause for a relative excess of cellulose and lignin in the leaves. The first step in litter decomposition is leaching, when nutrients and organic compounds dissolve in water and move into the soil. Compounds that are easily leached from litter include potassium, sugars and amino acids. Subsequent leaf litter weight loss is related to termite activity in *terra firme* rainforest (Luizão and Schubart, 1987), which can also degrade such resistant substances as lignin (Butler and Buckerfield, 1979). When litter gets increasingly fragmented over time, it reveals new surfaces for bacteria and fungi to attack (Chapin III et al., 2002). The presence and activity of termites may be low in nutrient-poor *campina* forest as compared to those in other rain forest types, which may slow down litter fragmentation and decomposition in *campina* forest, causing low carbon levels in the soil and high DOC concentrations in soil moisture and groundwater, as was observed in this and other studies in central Amazon (McClain and Richey, 1996; McClain et al., 1997). Litter fragment decomposition contributes significantly to CO₂ production in the soil, accounting for about 27% of soil emission (Wang et al., 1999), which is low at this site. Su (2005) reported that litter decomposition rates decrease over time due to labile compounds leaving the litter being attacked quickly. This process leaves a residue of less readily decomposable substances (Su, 2005), with potentially high C:N ratios (McClain et al., 1997). High nitrogen content in litter is considered a factor affecting the speed of decomposition by micro-organisms. Pate and Layzell (1990) shows that cellulose degradation is also a nitrogen-

limited process and will increase with the nitrogen content of litter. Vitousek and Sanford (1986) compared foliar and fine litterfall nutrients of various rainforests and showed that nitrogen and phosphorus appear to cycle less in *campina* forest than in other lowland forests.

3.3. Groundwater and soil chemistry

Average of chemical analyses of the groundwater obtained from piezometers in control and drainage plots are given in Table 1, whereas time series of pH and DOC concentrations in groundwater for all plots are shown also in Figure 4. DOC levels were consistently highest in groundwater below the *campina* forest, at 20-25 mg L⁻¹, and lowest in the *campinarana* plot, varying seasonally between 2 and 15 mg L⁻¹. The pH was invariably low at piezometer measurements in situ, ranging between 3.5 and 4.7, with a single minimum of 3.21 being observed in the drainage plot at the start of the drainage experiment in both piezometers. In both drainage and control plots the EC was positively correlated with the DOC concentration being R²=0.90 and 0.84, respectively, whereas the pH correlated negatively with DOC concentration R²=0.26 and 0.60, respectively (Figure 4). Concentrations of K⁺ and NO₃⁻ remained invariably low.

Table 1. Average of chemical analysis of groundwater in the drainage and control plots.

Control - piezometer, 1.62-2.12 m depth							Drainage - piezometer, 0.75-1.25 m depth					
Period	pH	DOC (mg.L ⁻¹)	DIC (mg.L ⁻¹)	NO ₃ ⁻¹ (mg.L ⁻¹)	K (mg.L ⁻¹)	EC (μS.cm ⁻¹)	pH	DOC (mg.L ⁻¹)	DIC (mg.L ⁻¹)	NO ₃ ⁻¹ (mg.L ⁻¹)	K (mg.L ⁻¹)	EC (μS.cm ⁻¹)
02/23/2007	3.63	-	-	-	0.00	24.2	4.27	3.90	2.25	0.01	0.03	17.0
After drainage installation (Mar/1/2007)												
04/21/2007	-	-	-	-	-	-	-	19.87	0.76	0.01	0.35	-
06/01/2007	-	-	-	-	-	-	4.21	14.97	4.32	0.01	0.34	25.1
08/01/2007	-	-	-	-	-	-	4.46	13.54	3.68	0.06	0.22	21.0
09/14/2007	-	-	-	-	-	-	4.35	13.64	0.75	0.05	0.40	23.2
10/19/2007	4.29	16.38	8.96	-	0.23	32.9	4.70	4.63	9.17	0.08	0.61	13.5
11/16/2007	4.30	10.14	3.77	0.03	0.26	18.4	4.60	2.26	3.86	0.06	0.41	13.7
12/13/2007	4.00	16.85	7.56	0.03	0.20	29.2	4.30	2.85	8.44	0.04	0.3	13.5
01/15/2008	3.95	23.03	7.71	0.02	0.21	35.8	4.16	13.43	5.38	0.13	0.25	22.4
02/21/2008	4.05	22.56	6.73	-	0.09	35.5	4.16	14.4	4.24	0.04	0.2	25.2
08/02/2008	4.06	18.40	6.42	0.01	0.97	34.0	4.58	8.79	6.07	0.17	0.46	21.0

The temporal variation in DOC levels observed in the *campina* plot is slighter than those in the control, drainage and *campinarana* plots. The fluctuations in the DOC concentrations of valley groundwater are presumably caused by seasonal variations in the rate of upwelling of deeper (plateau) groundwater, in which DOC concentrations are invariably low (<5 mg L⁻¹) according to Waterloo et al. (2006). Groundwater in the control plot had the highest DOC concentrations, whereas below the drainage and *campinarana* plots were consistently lower by about 7 mg L⁻¹. The EC in groundwater varied between 13 and 36 μS cm⁻¹ in the control and drainage plots. These differences could not be attributed to the impact of drainage as the initial pH and EC measurements suggest that DOC concentrations were already higher in the control plot than in the drainage plot at the start of the drainage experiment (Figure 4).

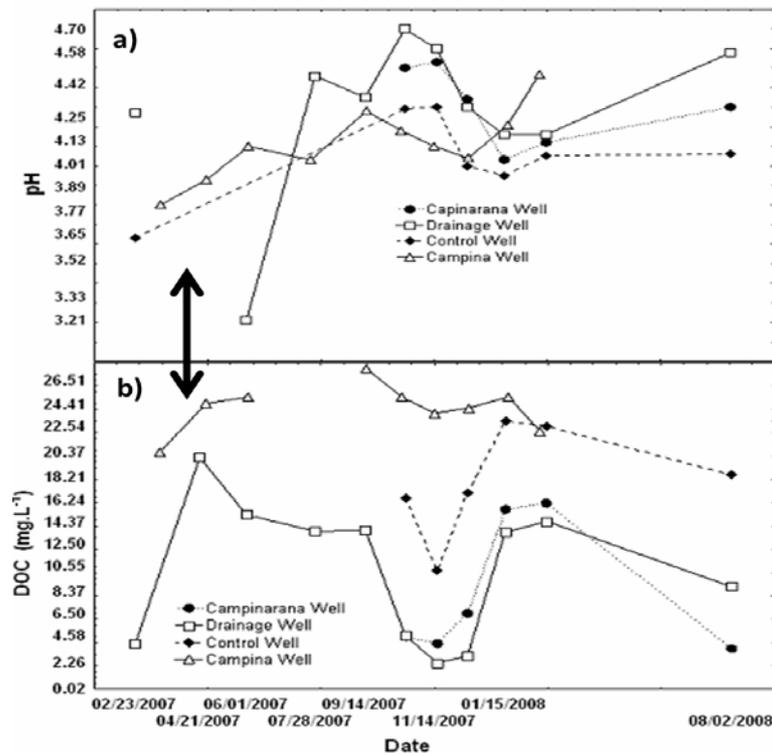


Figure 4. Temporal variation of in pH (a) and DOC (b) concentrations in groundwater (wells samples) in all four research plots. The dual arrow indicates the start period of drainage experiment.

The present Drainage study was observed no significant response on groundwater and soil chemistry conditions, excepting of marked changes in C and N contents of the top 5 cm of the soil in the drainage plot as compared to those in the control plot. The soil acidity, defined as H^+Al , is high in the riparian and *campina* forests. The low pH values observed in soil moisture in the drainage and control plots did not seem to be associated with corresponding increases in DOC concentrations and must therefore be caused by different soil processes, such as the removal of H^+ ions from exchange sites by other cations. Luizão et al. (2007) did observe that H^+ was the dominant ion, rather than Al^{3+} , on the exchange complexes in *campina* forest and suggested that H^+ ion toxicity is a major growth-limiting factor for non-adapted plants in heath forest soils in central Amazonia. Our study indicates that the riparian forest soils may have this low Al (and Fe) sesquioxides content in common with the *campina* soils, which could have implications for the forest succession in a drier climate. The yellowish/orange colour of the sand below *campinarana* forest suggests that these soils do have a higher Al (and Fe) sesquioxides content.

The shallow groundwater, sampled with the ceramic cup samplers, showed less seasonal variation in DOC concentrations and pH than that observed in the deeper piezometer samples (Figure 5). Water extracted from the 0-10 cm soil depth in the drained plot had a comparable DOC concentration as those measured deeper in the soil in the control plot. However, in the drained plot DOC concentrations decreased in the subsoil (below 10 cm) and were consistently lower than those measured at the same depths in the control plot (see Figure 5). Topsoil DOC concentrations also seem to respond to heavy rainfall in both plots. As this was the case from the start of the experiment onwards, it cannot be attributed to the artificial drainage.

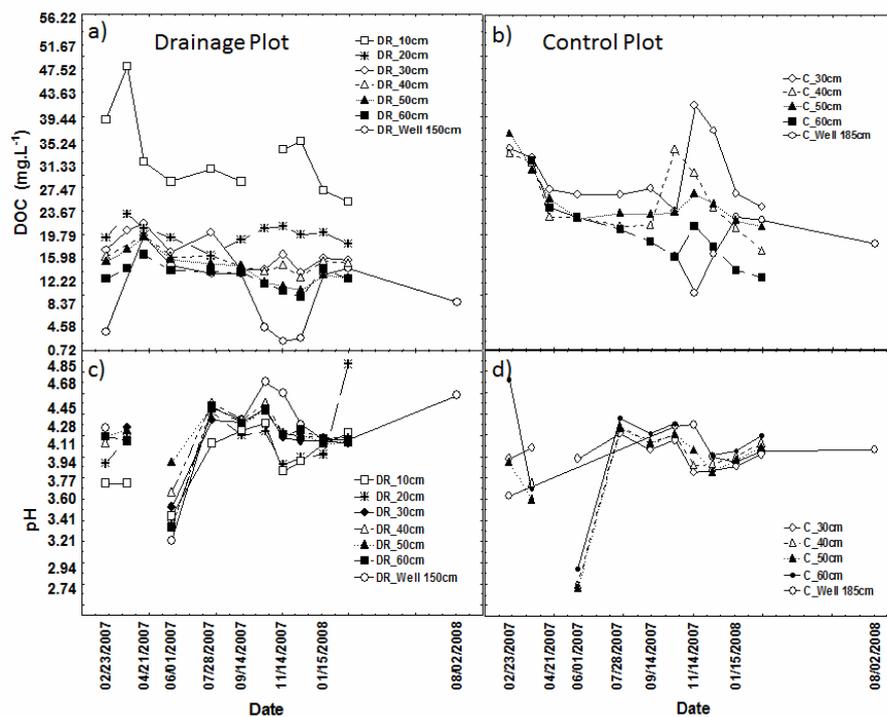


Figure 5. Temporal variation in pH and DOC concentrations from cup sampler (shallow) and piezometer (deep) groundwater in drainage (a and c) and control (b and d) plots.

The forest plots in exfiltration zones showed higher groundwater levels, higher soil nutrient contents and soil respiration rates, and faster litter decomposition rates. In spite of this, DOC levels in the shallow groundwater and stream water at these sites are also high and may reach levels locally that are similar to those presently observed in the *campina* forest groundwater. This suggests that the litter decomposition process that provides DOC in the valley is different from that acting in *terra firme* forests on slopes and plateaus in the central Amazonian landscape, where DOC levels in groundwater remain low (Waterloo et al., 2006). Luizão et al. (2004) and Marques, (2009) showed that soil C and N contents were significantly higher in the clayey topsoil (0-10 cm) on plateau and slope, than in the sandy valley, which they attributed to higher biological activity and higher intensity of mineralization processes at the former sites. Ion concentrations are invariably low in soil moisture and groundwater, as is the pH which varies between 3 and 5.

In spite of the lower DOC concentrations in soil moisture in the drainage plot for most of depths (Figure 5), the pH was comparable to that in the control plot, but showed a somewhat lower seasonal variation never reaching the extremely low values observed in the control plot (pH of about 2.8 in June 2007, Figure 5).

A comparison of the DOC concentrations measured at all sites is given in Figure 6. We have tested (F-test) differences between samples collected at all sites. The DOC concentrations were lowest in the riparian forest plots, whereas the highest values were observed in the *campinarana* forest. However, there was a significant difference between the DOC concentrations in soil moisture at 0.3 m depth in control and drainage plots ($n=16$, $F=13$, $p=0.00002$). The DOC concentration at the surface 0-10 cm layer in the drainage plot was similar to that measured at 0.3 m in the control plot. DOC concentrations in the control (at 0.3 m depth; $n=15$, $F=4.27$, $p=0.05$) and drainage (at 0.2-0.3 m depth; $n=25$, $F=3.8$, $p=0.04$) plots were significantly different from those (10 – 40cm) in the *campinarana* plot. DOC concentrations in the *campinarana* plot showed a large variation and resembled those measured in the *campina* plot more than those in the riparian forest plots. It should be noted

that the *campina* forest soil was often too dry for soil moisture extraction with the ceramic cup samplers and samples could therefore only be collected shortly after high rainfall events. It is because sandy soil in those dry areas needs heavy rain events to fill in the micro/macro soil pores, because the Hydraulic conductivity is bigger than other soils.

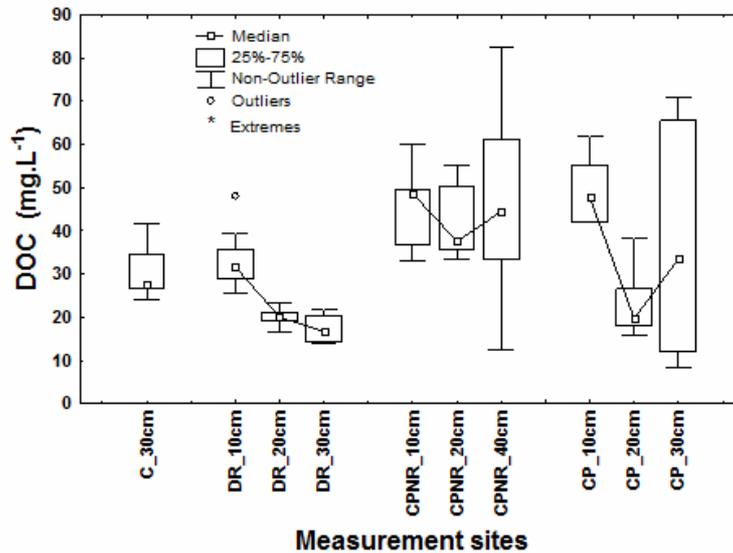


Figure 6. Variation in DOC concentrations in shallow groundwater sampled with the ceramic cup samples at depths between 10 to 30 cm below the soil surface in control (C), drainage (DR), *campinarana* (CPNR) and *campina* (CP) plots.

Table 2 shows the chemical properties of the soils in the drainage, control and *campina* plots. The soil samples collected in the control and drainage plots at the start of the study were inadvertently lost and the results shown in Table 2 are for soil samples collected in July 2008, at the end of the drainage experiment. All samples have low exchangeable cation concentrations, consistent with the sandy substrate and high leaching environment of these valley rain forests, and showed a decline with depth. The poor retention capacity of the valley soil can be due to the virtual absence of clay minerals (<2%) and the low organic matter content (Table 2). As expected %C, %N and exchangeable cation concentrations in control and drainage plots are rather similar. The C:N ratios remained fairly constant with depth down to 0.7 m, averaging to 24 ± 2 over the profile in the drained plot and to 28 ± 3 in the control plot. The exception is the top 0-5 cm soil layer in the drainage plot, which had extremely low %C (0.07%) and %N (0.00%) contents as compared to those observed in the same layer in the control plot (1.38% C and 0.05% N, Table 2). The change in carbon and nitrogen contents were in line with visual observations of a change in top soil colour in drainage plot (from dark grey to white) during the study, whereas no change in soil colour was observed in the control plot. Carbon and nitrogen difference in the first 0.05 m comparing drainage to the control plot after 16 months of treatment was about 10020 kg C ha and 365 kg N ha, respectively, with a measured bulk density of the top soil of 1400 kg m^{-3} . The analyses suggest that drainage did cause a marked decrease in the organic matter content of the top 5 cm of the soil, but not in deeper layers. Exchangeable macro-nutrient cations (Ca, Mg, P and K) remained at similar levels in drainage and control plots throughout the soil profiles (Table 2). The dominant ions on the exchange complexes were, in descending order, $\text{Al}^{\text{H}} > \text{K} > \text{Ca} > \text{Mg} > \text{Fe} > \text{P}$.

Table 2. Soil nitrogen, carbon, exchangeable cations and pH at the different research sites sampled in February 2007 (CP plot) and July 2008 (DR, C plots).

samples	N	C	Ca	Mg	Al + H	K	P	Fe	Zn	Mn	pH _{KCL}	pH _{H2O}
	%	%	mg kg ⁻¹	mg kg ⁻¹	cmol _c kg ⁻¹	mg kg ⁻¹						
DR - 0-5 cm	0.00	0.07	26.50	5.80	0.23	71.00	5.98	11.9	1.50	7.30	-	4.31
DR - 5-10 cm	0.15	3.94	14.50	1.80	0.16	25.80	2.02	9.70	0.50	2.50	-	4.38
DR - 10-20 cm	0.11	2.48	13.50	11.00	0.18	22.10	1.49	12.5	0.50	1.40	-	4.42
DR - 20-30 cm	0.11	2.39	9.00	7.50	0.14	12.70	1.22	10.6	0.40	0.80	-	4.58
DR - 30-40 cm	0.07	1.81	7.00	5.90	0.12	8.80	1.29	7.50	0.20	0.40	-	4.59
DR - 60-70 cm	0.05	1.25	6.50	4.60	0.12	9.50	0.40	10.8	0.30	0.40	-	4.50
C - 0-5 cm	0.05	1.38	19.50	5.10	0.18	46.30	2.25	13.0	0.80	1.40	2.69	4.25
C - 5-10 cm	0.13	3.40	38.00	8.90	0.22	53.80	4.43	16.6	1.20	1.90	2.68	4.29
C - 10-20 cm	0.23	5.94	13.50	1.90	0.19	28.90	2.38	13.6	0.70	1.10	2.83	4.36
C - 20-30 cm	0.11	2.80	14.50	1.30	0.19	21.20	1.65	11.8	0.60	0.90	2.95	4.55
C - 30-40 cm	0.09	2.59	10.00	8.40	0.17	26.60	0.89	12.4	0.50	0.40	2.84	4.55
C - 60-70 cm	0.07	2.22	9.50	6.90	0.11	11.60	0.59	11.8	0.40	0.40	2.83	4.60
CP-0-5 cm	-	-	28.2	2.1	0.55	1.77	1.19	9.75	0.23	0.83	2.83	4.27
CP-5-10 cm	0.04*	0.20*	1	0	0.50	1.65	0.90	11.5	0.15	0.68	3.20	4.67
CP-10-20 cm	0.037*	0.28*	0.5	0	0.15	1.67	0.37	11.8	0.23	0.25	3.46	5.05
CP-20-30 cm	0.03*	0.15*	0.8	0	0.10	0.55	0.35	11.1	0.20	0.28	3.93	5.31
CP-60-70 cm	0.017*	0.08*	0.6	0	0.05	0.37	0.28	9.38	0.10	0.30	4.23	5.37
CP-90-100 cm	0.01*	0.07*	0.7	0	0.05	0.80	0.27	9.25	0.15	0.30	4.20	5.28

DR= Drainage (July 2008)

C=Control (July 2008)

CP=*Campina* forest

* *Campina* data from Luizão (1996)

Exchangeable cation concentrations were lowest in the sandy *campina* soil, which also had the highest soil pH and pH_{KCL} values. Al⁺H were again dominant on the exchange complex, but this was followed by Fe and very low levels of, in descending order, Ca > K > P > Mg. Observations of %C and %N made by Luizão (1996) in *campina* forest soil indicated that %N was similar to that measured in the riparian forest plots, whereas %C was much lower and close to that found in the top soil of the drained plot (Table 2). The soil C:N ratio also remained fairly constant throughout the profile in the *campina* forest, but was significantly lower at 9.4±1.0 (Luizão, 1996).

3.4. Soil CO₂ respiration

Soil CO₂ respiration rates were significantly different between forest types and drainage and control plots (Figure 7). The *campina* forest exhibited very low soil respiration rates at 1.2±0.3 μmol CO₂ m⁻² s⁻¹, whereas *campinarana* forest showed the highest rate at 5.7±1.8 μmol CO₂ m⁻² s⁻¹. Both forest types grow on soil with similar coarse sandy texture and with phreatic levels generally below 0.8 m. These forests do differ in the presence (*campinarana*) or absence (*campina*) of a root mat and continuous litter layer, which may explain the large difference in autotrophic and heterotrophic soil respiration rates. The riparian forest on near-saturated sandy soil (before drainage tube installation) respired at an intermediate rate of 3.6±0.6 μmol CO₂ m⁻² s⁻¹, which is similar to that measured in the tall *terra firme* forest on the plateau.

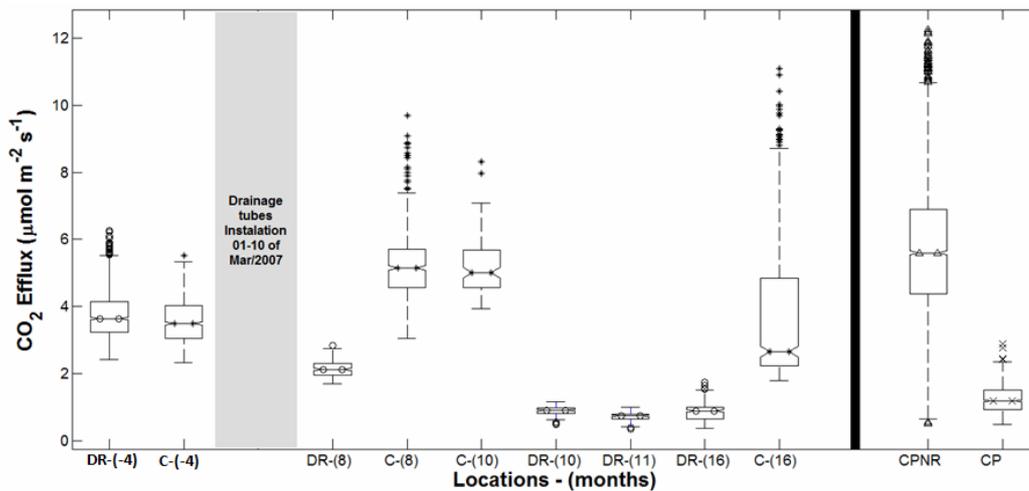


Figure 7. Boxplot of soil CO₂ respiration measurements at the different forest plots: Drainage plot (DR), control plot (C), *campinarana* plot (CPNR) and *campina* plot (CP). The grey rectangle separates measurements done before and after drainage tube installation and the numbers between brackets indicate the month during which measurements were made in 2007, where month 4 represents the start of the drainage experiment. The black vertical bar separates *campinarana* and *campina* measurements from those made in control and drainage plots. Fluxes in both *campinarana* and *campina* forests were measured over periods of several weeks in the wet, as well as in the dry season.

At the start of the drainage experiment soil respiration was measured at two different locations within the drained area between 18 November and 5 December 2006, followed by measurements at a single location in the control plot between 5 and 12 December, 2006, and at a second location between 7 and 9 February, 2007. Measured soil respiration rates were very similar in control and drainage plots at 3.6 ± 0.6 and 3.7 ± 0.7 $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$, respectively (Figure 7). A comparison of the soil respiration rates in the drainage and control plots indicated significant reductions of 42% (2.1 ± 0.2 $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) in the former with respect to the initial value after 8 months of drainage and 76% (0.8 ± 0.2 $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) after 16 months of drainage, whereas soil respiration in the control plot remained at or above the initial value at 5.1 ± 0.8 $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ after 8 months and 3.7 ± 2.8 $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ after 16 months (Figure 7). Another drought experiment also found 20% of soil CO₂ emission decrease due to soil moisture decrease (Meir et al., 2008, 2009). This suggests that the drainage of the riparian forest soil caused a significant decrease in soil CO₂ respiration to the values observed in the *campina* forest plot.

This study is the first to provide information on soil respiration in *campina* forest and on the impact of drainage of sandy valley soil on soil respiration rates. The drainage experiment revealed that the largest changes are likely to occur in the top soil where the carbon and nitrogen contents decreased markedly after artificial lowering of the phreatic level. The reduction in the carbon and nitrogen content in the top soil were matched by a strong reduction in the soil respiration rate as compared to that measured in the same plot before drainage and in the control plot. The observations suggest that changes in topsoil carbon and nitrogen content occur within a few months after a lowering of the phreatic level and that this has a pronounced effect on the soil respiration rate. Such fast response may be caused by a combination of the rapid flushing of fine particulate and colloidal carbon in the soil pores to lower soil layers with infiltrating rainfall (Marques, 2009) and removal of particulate carbon in the form of CO₂ by enhanced decomposition rates in the better aerated top soil after drainage. Impacts of drainage on other nutrients and on those in deeper soil layers were not

observed (Table 2, Figures 5 and 6) and may only become apparent after a more extended period of drainage.

The soil respiration rate in the drained plot became as low as that in the *campina* plot, which presumably reflects the low C and N contents in the topsoil after drainage, where most of the soil CO₂ respiration is produced (Hanson et al., 2000). These soil carbon and nitrogen levels were much lower than those in the control plot where soil CO₂ respiration remained constant and at a rate similar to that in the drainage plot before the start of the experiment. Xu et al. (2004) and Lambers et al. (1998) have reported that nitrogen and carbon availability in the soil are important for both plant growth and the production of CO₂. Hence, the reduction of soil CO₂ respiration in the drainage plot may be an indication that large-scale drainage of these sandy valley areas, and associated decrease in the top soil carbon and nitrogen contents, may lead to a change in the carbon cycle towards that of *campina* forest. A lowering of the phreatic level in this zone and corresponding infiltration of rain water could affect soil respiration fluxes rapidly, as was observed in the present study (Figure 7).

Although a modest decrease in the phreatic level may not cause hydrological stress on the vegetation, the almost daily precipitation events (Figure 2b) may eventually cause a further flushing of particulate organic carbon and nutrients from the topsoil to deeper layers in the valley. However, with the low carbon contents in our sandy soils, this is less likely to be very important here. Finally, because the phreatic level is at or very close to the soil surface in riparian forest, degassing of groundwater with high pCO₂ might also contribute to measured soil respiration. Further research is required to discover if degassing of shallow groundwater is an important factor in this area or if biotic processes are dominant. Measuring such individual contributions of biotically, chemically and physically produced CO₂ to the soil respiration flux is extremely difficult, but would provide valuable insight into the importance of the various processes generating CO₂ and would also be important for predicting changes in soil respiration as a consequence of a drier climate.

4. CONCLUSIONS

Forests on sandy soils cover a significant part of the central Amazonian landscape. The present study shows that depending on the drainage status of sandy soils, a large natural variation exists in soil carbon, nitrogen and exchangeable cation concentrations, soil water chemistry and litter decomposition rates. Litter decomposition and soil CO₂ respiration rates are very low in *campina* forests, where the capacity of the coarse sandy soil to retain nutrients is extremely low. The highest soil CO₂ respiration rates within the *terra firme* landscape are found in ecotone *campinarana* forests that form the transition from riparian forests on near-saturated sandy soils to tall *terra firme* forests on slopes and plateaus with clayey soils.

Under present climate conditions the vegetation in the riparian forest is not likely to experience significant periods of droughts. Artificial drainage of the riparian forest plot, lowering the phreatic level by a modest 0.3 m, did not cause apparent changes in the soil nutrient content, chemical composition of soil water or litter decomposition rates within the study period of one year. However, marked reductions were observed in the top soil carbon and nitrogen content, as well as in the soil CO₂ respiration rate, which decreased to the levels observed in *campina* forest. The drainage study therefore indicates that the riparian forest soil is very sensitive to changes in the phreatic level. It remains unclear if slower changes in soil and water chemistry in the subsoil will become apparent within the next few years. This suggests that the characteristics of drained riparian forest soil show a transition towards that of *campina* forest, with its low nutrient content and very low soil respiration rate. However, shifts in vegetation species and a future development of a root mat, such as exists in the *campinarana* forest, may result in increased soil respiration rates. It is as yet uncertain what

the direction of succession of drying riparian forest will be and continued monitoring of the drained and control plots may provide valuable clues in this sense.

The large difference in soil respiration rates between *campina* and *campinarana* forests imply that knowledge about the likelihood of riparian forest to be succeeded by one type or the other is crucial for determining the future carbon budget of *terra firme* rainforest ecosystem experiencing a drier climate.

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