Soil solution nitrogen losses during clearing of lowland Amazon forest for pasture

Christopher Neill[1,3], Marisa C. Piccolo[2], Carlos C. Cerri[2], Paul A. Steudler[1] & Jerry M. Melillo[1]

[1]*The Ecosystems Center, Marine Biological Laboratory, Woods Hole, MA, 02543, USA.* [2]*Laboratório de Biogeoquímica Ambiental, Centro de Energia Nuclear na Agricultura, Piracicaba, SP, Brazil.* [3]*Corresponding author**

Abstract

Losses of nitrogen (N) often follow severe disturbance of forest ecosystems. In tropical forests, losses of N associated with the disturbance of clearing may be particularly important because rates of soil N cycling are high and forest clearing now occurs on a large scale. We measured soil solution inorganic N concentrations and fluxes for 1 year in an intact forest in the Brazilian Amazon state of Rondônia and in an adjacent 3-ha forest plot that was cleared for pasture by cutting, burning and planting pasture grass and in established cattle pastures on the same soils that were 5 and 22 years old. The cleared forest had higher soil solution NO_3^- concentrations than the intact forest, but the difference between the cleared and control forests declined with time after the start of the first post-clearing rainy season. Established pastures had much lower solution NH_4^+ and NO_3^- concentrations than forest or cleared forest. Estimated annual dissolved inorganic solution N fluxes to below 1 m during the first year after clearing were 2.5 kg ha^{-1} in forest and 24.4 kg ha^{-1} in newly cleared forest compared with only 0.5–1.2 kg ha^{-1} in established pastures. The solution fluxes from cleared forest during the first year after clearing were approximately 7 times greater than gaseous N oxide ($N_2O + NO$) losses estimated for the same time. These results were consistent with the characterization of moist tropical forests on weathered soils as N-rich and likely to respond to disturbances that elevate soil N availability with increased loss to both soil solution and the atmosphere. These results also suggest that the relative increase in N oxide loss is substantially less than the increase solution inorganic N loss.

Introduction

The disturbance of forest cutting can disrupt soil N cycles and lead to losses of N in soil solution and streamflow (Likens et al., 1969; Qualls et al., 2000; Swank, 1988). Several lines of evidence suggest that post-disturbance losses of N in soil solution from rainforests on weathered tropical Ultisols and Oxisols may be high. Rates of N

cycling in these soils are high (Neill et al., 1997b; Reiners et al., 1994; Verchot et al., 1999) and elevated concentrations of N in foliage (Jordan, 1985; Vitousek, 1984), lack of plant response to N fertilization (Herbert and Fownes, 1995) and soil and foliage $\delta^{15}N$ values (Martinelli et al., 1999) all suggest a relatively open N cycle. In many rainforest soils, most mineralized N is rapidly nitrified (Neill et al., 1995; Vitousek and Matson, 1988), microbial N immobilization is generally low (Vitousek and Matson, 1988) and N additions result in increased N oxide emissions

* FAX No: + 508-457-1548.
  E-mail: cneill@mbl.edu

(Erickson et al., 2001; Hall and Matson, 2003). Because nitrification plays an important role in N retention in forest ecosystems (Vitousek et al., 1982), the high percentage of mineralized N nitrified in these tropical forests suggests a high potential for NO_3^- loss in soil solution and stream water when these forests are disturbed. This pattern is consistent with a limited number of studies that report large N losses from soils in the months following the clearing and burning of tropical forests (Matson et al., 1987; Uhl and Jordan, 1984) and with reports of elevated concentrations of inorganic N in stream water from small watersheds undergoing deforestation (Williams and Melack, 1997).

The potential for N losses from disturbed tropical forest soils in the form of N oxide gases is also high. Emissions of both nitrous oxide (N_2O) and nitric oxide (NO) from tropical forest are significant and a general relationship between N oxide emissions and rates of soil N cycling suggests that disturbances that elevate soil N availability will increase N oxide emissions from soils (Davidson et al., 2000). Increased fluxes of N_2O are reported from tropical forests after hurricanes, cutting or fertilization (Erickson and Ayala, 2004; Erickson et al., 2001; Keller et al., 1993; Melillo et al., 2001; Steudler et al., 1991; Steudler et al., 2002). The immediate post-disturbance dynamics of N cycles and losses in tropical forests are not well known, although it is now well established that rates of soil N cycling and gaseous N losses from established pastures are clearly lower than from the original forest (Markewitz et al., 2004; Melillo et al., 2001; Neill et al., 1995; Verchot et al., 1999).

Currently, understanding changes to N cycles following the disturbance caused by clearing of tropical forests is important because tropical forests in many parts of the world are being transformed rapidly to other land uses. This is the case in the Brazilian Amazon Basin, which covers approximately 4 million km^2 of forest and contains approximately one-third of the world's remaining tropical moist forest (Matthews et al., 2000). The Amazon Basin currently experiences deforestation at rates of 15,000–29,000 $km^2 \, y^{-1}$ and pastures now represent the largest single use of converted forest (INPE, 2005). Changes to soil solution and streamwater chemistry caused by this land-use conversion have the potential to alter material export in streams and aquatic

productivity (Neill et al., 2001). They also have the potential to influence N oxide budgets at the global scale (Davidson et al., 2001).

We have been studying how the clearing of lowland Amazon rainforest for pasture influences soil N cycles, N oxide emissions and stream water chemistry in a region of central Rondônia, Brazil. In this paper, we report how forest clearing for pasture influences concentrations of inorganic N in soil solution by comparing intact forest, an adjacent 3-ha forest plot that was experimentally cleared by cutting, burning and planting to pasture grass, and two established cattle pastures of 5 and 22 years old. We used solution concentrations to estimate solution inorganic N fluxes to below 1 m at each site. We then used previously-reported N_2O and NO fluxes measured or estimated from the same sites during the same time (Garcia-Montiel et al., 2001; Melillo et al., 2001) to provide a comprehensive picture of how both solution and gaseous N fluxes through a time-course of forest disturbance and pasture establishment.

Methods

Study area

Studies were conducted in 1994–1995 at Fazenda (Ranch) Nova Vida at 472 km of highway BR-364 (10°13′ S, 62°19′ W) in central Rondônia. We selected four areas for study that were in locations of similar elevation (~150 m) on similar soil (Kandiudult, red–yellow podzolic latosol in the Brazilian classification) and with similar previous forest vegetation. The areas were: (1) a 5-ha area of intact forest, (2) a 3-ha plot of forest, directly adjacent to the intact forest, that was converted to pasture by cutting and burning in a manipulation experiment, (3) an established pasture of 5 years old that was converted from forest by cutting and burning in 1989, and (4) an established pasture of 22 years old that was converted from forest by cutting and burning in the same manner in 1972. The forest and the 5- and 22-year-old pastures were sites included in a very well-documented chronosequence that examines changes to soil properties with forest clearing and pasture ageing (Moraes et al., 1996, Neill et al., 1997a). A number of previous studies of

soil carbon and nutrient stocks and responses of biogeochemical cycles have been conducted at these sites (Moraes et al., 1996; Neill et al., 1995; Neill et al., 1996, 1999).

The forest at this site is classified as open tropical forest with a large number of palms (Projeto RADAMBRASIL, 1978; Pires and Prance, 1986). The site selected for experimental forest clearing was chosen because it contained soils and vegetation similar to the original forest. Biomass of the slash in the cleared site was 307 Mg ha^{-1} (Graça et al., 1999). Changes to extractable soil N concentrations and N cycling rates before and after clearing are reported in Melillo et al. (2001). The experimental clearing was done by cutting trees and brush with chainsaws in June, burning the dried slash in September and planting the pasture grass Brachiaria brizantha [Hochst] Stapf. in January (Melillo et al., 2001). This sequence of cutting and burning was similar to the sequence used to clear other large areas of forest for pastures at Nova Vida and is typical to the way the forest is cleared on large ranches over much of the Amazon (Serrão et al., 1996). The pastures received no fertilizer or other chemical amendments. Both established pastures were within 3 km of each other and received similar management and grazing pressure of about 1.5 animals ha^{-1} since 1972. In this experiment, we sought to simulate typical management practices. Our results represent the combined influences of cutting, burning and establishment of pasture grasses on solution N concentrations and fluxes. Sampling in the cleared plot began before clearing in May 1994.

Soils of the study area were Kandiudults with clay contents that ranged from 18–22% at 0–0.05 m to 39–44% at 0.4–0.5 m (Moraes et al., 1996). They are weathered, low in plant available P (Garcia-Montiel et al., 2000), but have higher base cation concentrations than do more highly weathered oxisols (Moraes et al., 1995; Richter and Babbar, 1991). In general, these and similar Ultisols cover approximately 22% of the Amazon Basin (Moraes et al., 1995). Vegetation in both pastures was Brachiaria brizantha, the most widely planted pasture grass species in the Brazilian Amazon. The climate at Nova Vida is humid tropical, with a dry season from May to September. Precipitation in the year in which we made our measurements was 1993 mm and slightly below the annual average of 2200 mm (Bastos

and Diniz, 1982). Annual mean daily temperature in central Rondônia is 25.6 °C, mean daily temperature for the warmest and coolest months varies less than 5 °C and mean relative humidity is 89% (Bastos and Diniz, 1982).

Field and laboratory methodology

Soil solution was collected from tension lysimeters (Soil Moisture Equipment, Santa Barbara, California, USA) placed in the forest, cleared forest and 5- and 22-year-old pastures. Lysimeters were installed in July in a line 10 m apart, with one lysimeter at 0.3 m and one at 1.0 m at each point. They were flushed thoroughly before installation and were also flushed by collecting and discarding water for 5 consecutive days before sampling. Water samples for analysis were collected from September 1994 to April 1995. Samples were collected by placing a vacuum of 30 kPa on the lysimeter and collecting water 24 h later. Sampling was carried out approximately every 7 days. Solution pH was measured in the field laboratory with a Metrohm 632 pH-meter. Water samples were filtered through 0.22 micron Millipore cellulose acetate filters. One portion of the filtrate was preserved with 1 ml L^{-1} of 6 M sulfuric acid for analysis of NH_4^+, the other portion was preserved with 1 ml L^{-1} of 6 M phosphoric acid for analysis of NO_3^-. Samples were stored in polyethylene bottles and refrigerated. Ammonium was measured colorimetrically by the phenol-hypochlorite method and NO_3^- was measured colorimetrically following Cd reduction.

We determined if deposition of N contained in the dead biomass or ash produced by cutting and burning increased soil N by measuring soil N stocks to 0.1 m during 6-time periods: 2 months before cutting, 1 day before burning, 2 days after burning, and 5, 8 and 17 months after burning. Soils were collected with steel cylinders of known volume from the walls of 5 small pits located 10 m apart in a line adjacent to the lysimeters. Soils were weighed wet, one subsample was dried at 105 °C for determination of dry bulk density and another subsample was dried at 60 °C for determination of total N on a Perkin Elmer 2400 elemental analyzer. Errors associated with N stocks were calculated based on the variances of density and N concentration (Neill et al., 1997a).

We used a simple monthly soil water balance developed for central Rondônia (Scerne et al., 2000) based on Thornthwaite and Mather (1955) to estimate water exceedence and solution fluxes for the period of the soil solution collections in 1994 and 1995. This model was driven with aggregated daily precipitation and temperature data for the period from Nova Vida. We modified this model for pasture by assuming a rooting depth for pasture plants of 1.5 m. Solution inorganic N fluxes were calculated for each site as the product of total water flux and mean monthly NH_4^+ and NO_3^- concentrations at 1.0 m depth, summed over all the months with positive downward water flux.

To compare concentrations of NH_4^+, NO_3^- and pH in the four sites (forest, cleared forest, 5-year-old pasture, 22-year-old pasture) we performed a repeated measures analysis of variance (GLM procedure of SAS). Data were log $+1$ transformed data where necessary to meet the assumption that data were normally distributed. Each lysimeter was considered as a separate subject in this analysis. Separate analyses were conducted for 0.3 and 1.0 m depths. We tested for differences between 0.3 and 1.0 m depths within each plot by calculating a mean concentration across all collection dates for each lysimeter and performing t-tests ($n = 5$). We interpreted the biogeochemical importance of these comparisons based on the magnitude of differences between the plots and our previous evidence that these plots minimized pre-existing differences caused by vegetation, topography, soils and historical land management (Moraes et al., 1996; Neill et al., 1997a). To compare concentrations of NH_4^+ and NO_3^- with concentrations of dissolved organic N (DON), we used measurements of these solutes in the 5-year-old pasture made during a later time period (1998–2001). We calculated a mean concentration for each lysimeter, then compared means ($n = 5$) using t-tests. Separate analyses were done for 0.3 and 1.0 m depths.

We calculated fluxes of N_2O and NO from the forest, cleared forest and established pastures from our previously published studies. We used our estimates of N_2O emissions measured in the forest and in the pastures across wet and dry seasons in 1992–1993 (Melillo et al., 2001) and 1998–1999 (Garcia-Montiel et al., 2001). Nitrous oxide was measured from the forest and cleared forest during the same period during 1994–1995 when measurements of solution fluxes were made (Melillo et al., 2001). Nitric oxide was measured in the forest and the established pastures during 1998 and 1999 (Garcia-Montiel et al., 2001). Nitric oxide emissions were not measured in the cleared forest and NO fluxes from forest at Nova Vida cannot be predicted reliably from soil moisture, soil NH_4^+ or NO pools or net N mineralization or net nitrification rates (Garcia-Montiel et al., 2001). Fluxes of NO, however, increased in response to increased NH_4^+ availability (Steudler et al., 2002). Because soil extractable NH_4^+ increased for a period of 6 months after the forest cutting (Melillo et al., 2001), we used the mean NO fluxes measured in response to NH_4^+ fertilization (52.9 μg N m^{-2} h^{-1}) to estimate NO fluxes during the first 6 months after clearing. This was our best estimate of NO fluxes under conditions of elevated soil NH_4^+ pools, even though soil NH_4^+ pools after fertilization never reached the high concentrations recorded after forest cutting and burning (Melillo et al., 2001; Steudler et al., 2002). After the first year, we assumed NO fluxes from the cleared forest were equal to the mean NO fluxes measured during the rainy season (1.4 μg N m^{-2} h^{-1}) (Steudler et al., 2002).

Results

Solution NH_4^+ concentrations at 0.3 m generally declined with time since start of the rainy season (Figure 1). Mean solution NH_4^+ concentrations at 0.3 m over the entire rainy season were significantly greater in the forest than in the cleared forest and greater in the forest and cleared forest than in either pasture (Table 1). Solution NO_3^- concentrations at 0.3 m were more variable but were also higher in the first half of the rainy season (Figure 1). Mean solution NO_3^- concentrations at 0.3 m over the entire rainy season were significantly greater in the cleared forest than in the forest and greater in the forest and cleared forest than in either pasture (Table 1). Nitrate dominated total dissolved inorganic N (DIN) concentrations in both forest and cleared forest. The seasonal patterns of NH_4^+ and NO_3^- concentrations at 1.0 m were generally similar to those at 0.3 m (Figure 2). Concentrations of
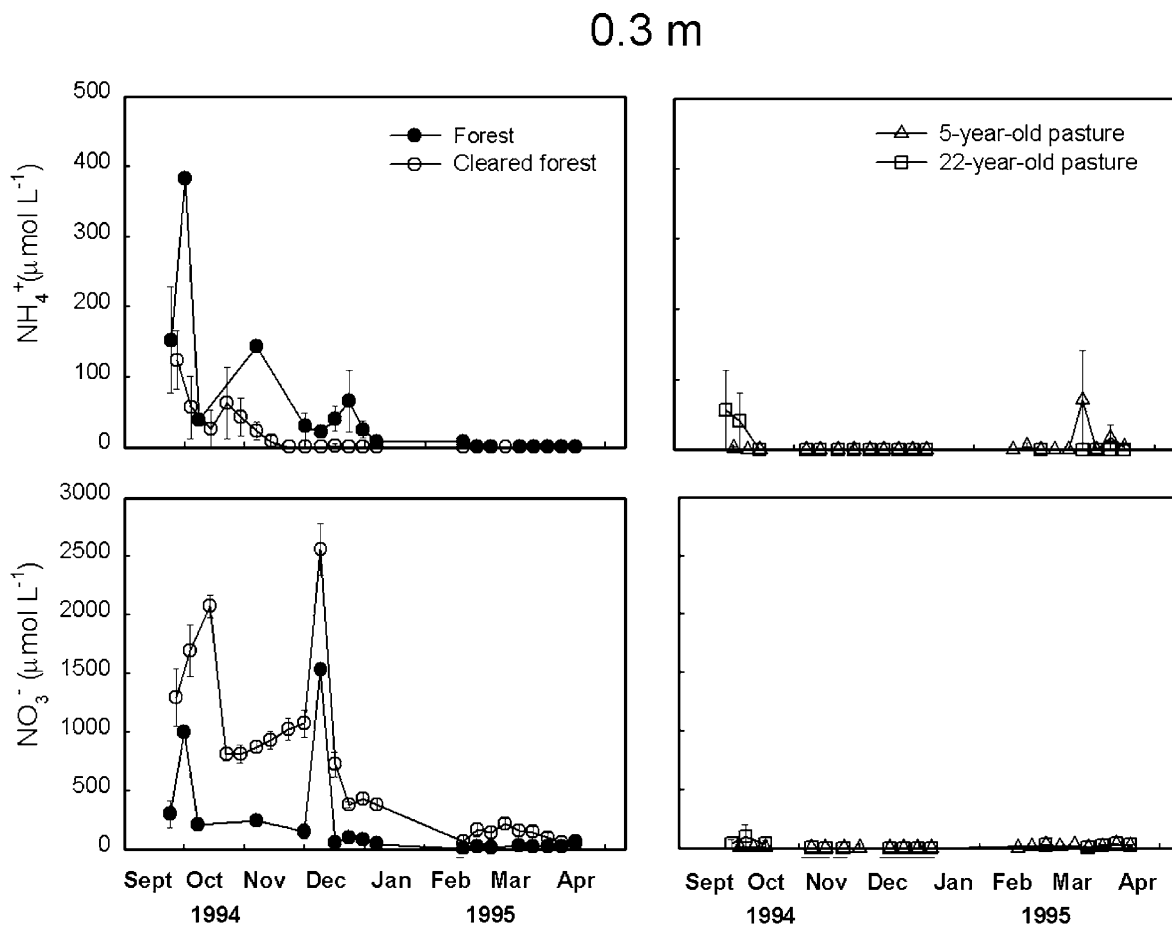
0.3 m

Figure 1. Soil solution concentrations of NH_4^+ and NO_3^- in lysimeters at 0.3 m in forest, forest that was cleared by cutting, burning and planting to pasture, and established pastures 5- and 22-years-old. Values are means ± standard error.

NH_4^+ were lower at 1.0 m than at 0.3 m in the forest but not in the cleared forest or in the pastures (Table 1). Concentrations of NO_3^- were lower at 1.0 m than at 0.3 m in both the forest and cleared forest but not the pastures (Table 1).

Soil solution pH ranged from 5.5 to 6.2 and showed no seasonal pattern or significant difference among sites or depths (Table 1). Soil N was similar in the cleared forest and in the intact forest and did not change significantly following forest clearing (Figure 3).

Annual solution fluxes of NH_4^+ to below 1.0 m from the forest and cleared forest were zero (Table 2). In contrast, annual solution fluxes of NO_3^- to below 1.0 m were 2.5 kg N ha^{-1} y^{-1} in forest and nearly 10-fold higher, 24.4 kg N ha^{-1} y^{-1}, in the cleared forest (Table 2). Annual solution

fluxes of NH_4^+ fluxes ranged from 0.1 to 0.8 kg N ha^{-1} y^{-1} in the pastures (Table 2). Annual solution fluxes of NO_3^- were 0.4 kg N ha^{-1} y^{-1} from both pastures (Table 2).

Discussion

Tropical forest N cycling and response to disturbance

The high concentrations of NO_3^- in soil solution of the rainforest at Nova Vida were consistent with the high soil extractable NO_3^- pools, high rates of soil net mineralization and near-complete nitrification of mineralized N previously measured at this site (Melillo et al., 2001). This

Table 1. Overall mean soil solution N concentrations and pH in lysimeters from 0.3 to 1.0 m collected for one entire rainy season (23 September to 13 April) from sites representing four land uses at Nova Vida, Rondônia

		0.3 m	1.0 m	Test for difference between depths ($p <$)
		μmol L^{-1}		
Forest	NH$_4^+$	34 ± 10	8 ± 8	0.013
	NO$_3^-$	116 ± 38	51 ± 15	0.037
	pH	5.5 ± 0.1	5.6 ± 0.1	NS
Cleared forest	NH$_4^+$	16 ± 5	10 ± 3	NS
	NO$_3^-$	745 ± 68	264 ± 28	0.0001
	pH	5.8 ± 0.1	5.8 ± 0.0	NS
5-year-old pasture	NH$_4^+$	3 ± 2	13 ± 4	NS
	NO$_3^-$	9 ± 2	9 ± 1	NS
	pH	6.2 ± 0.7	6.1 ± 0.0	NS
22-year-old pasture	NH$_4^+$	5 ± 4	2 ± 2	NS
	NO$_3^-$	16 ± 6	8 ± 1	NS
	pH	5.9 ± 0.1	5.9 ± 0.0	NS
Test for difference among treatments ($p <$)	NH$_4^+$	0.026 ($N = 237$)	NS ($N = 247$)	
	NO$_3^-$	0.0001 ($N = 237$)	0.0001 ($N = 247$)	
	pH	NS ($N = 302$)	NS ($N = 282$)	

pattern was also similar to high extractable N concentrations in forest soils from other lowland Amazon forests on weathered soils (Neill et al., 1997b; Verchot et al., 1999) and high soil solution DIN concentrations found at other tropical forest sites (Markewitz et al., 2004; McClain et al., 1994; McDowell et al., 1992).

The 10-fold increase in fluxes of NO$_3^-$ to below 1 m during the first year after clearing indicated a pronounced opening up of the soil N cycle in response to this major disturbance. Several observations suggest that a post-disturbance period of elevated soil N availability and elevated solution DIN concentrations was brief and was controlled primarily by the absence of actively growing vegetation. First, we saw a sharp decline of NO$_3^-$ concentrations at 0.3 m depth in January that coincided with grass establishment. Second, this was very similar to the timing of a decline in soil extractable NH$_4^+$ and NO$_3^-$ concentrations after clearing (Melillo et al., 2001). Third, while nitrification consumed nearly all NH$_4^+$ produced by mineralization, rates of net N mineralization and net nitrification were not elevated during the disturbance sequence (Melillo et al., 2001), suggesting that changes to soil

extractable and solution DIN concentrations were not caused by changes to soil organic matter net mineralization or net nitrification rates.

The absence of a change to total N stocks also suggested that transfer of N from vegetation to soils following clearing was not a major control of soil N availability and solution DIN concentrations. Biomass burning consumed 36% of the original biomass at this site (Graça et al., 1999). This was similar to losses of total ecosystem N stock of 22–35% found in similar slash fires in Rondônia (Kauffman et al., 1995). Kauffman et al. (1995) also reported that cutting and burning did not alter soil N stocks in Rondônia.

The importance of plant N demand in regulating soil N pools and N losses has been demonstrated in a variety of forests. Nitrate losses in streamwater from a deciduous temperate forest ecosystem at Hubbard Brook were elevated during 3 years when herbicide application prevented plant regrowth but declined when herbicide was discontinued and secondary forest vegetation became established (Bormann et al., 1974). Steudler et al. (1991) found higher soil NH$_4^+$ and NO$_3^-$ concentrations in Puerto Rican forests
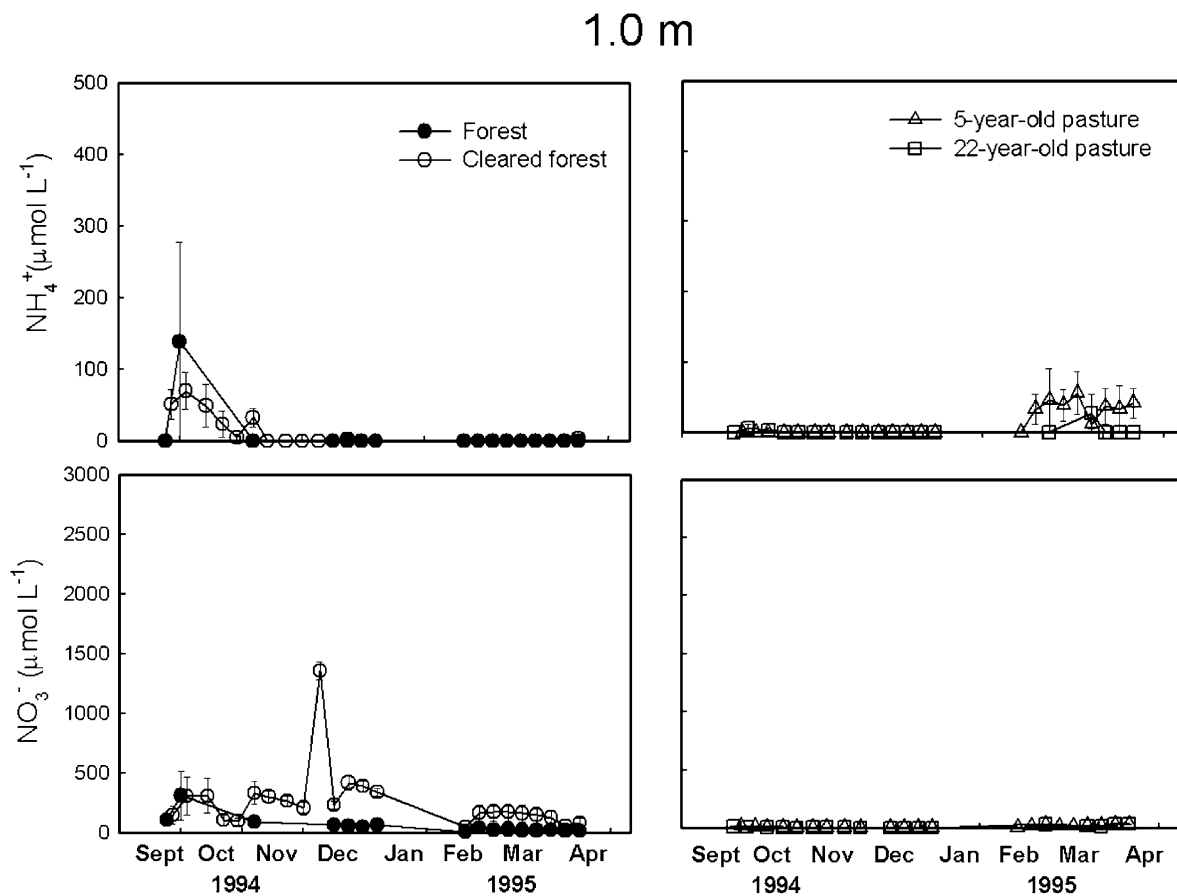
1.0 m

Figure 2. Soil solution concentrations of NH_4^+ and NO_3^- in lysimeters at 1.0 m in forest, forest that was cleared by cutting, burning and planting to pasture, and established pastures 5- and 22-years-old. Values are means ± standard error.

that had been disturbed by clearing and hurricane blow down compared with undisturbed forest. The clearing of a 75-year-old Costa Rican secondary forest led to higher soil extractable NH_4^+ and NO_3^- concentrations for 6 months, and preventing natural regrowth in subplots of the cleared area led to higher NH_4^+ concentrations, a longer period of elevated soil NO_3^- concentrations and greater leaching losses of inorganic N (Matson et al., 1987). Roberson (1989), in a similar experiment, reported higher soil NO_3^- concentrations and higher potential nitrification rates in plots of 20-year-old lowland Costa Rican forest that were cleared and maintained bare compared with plots that were cleared but where vegetation was allowed to regrow. In the central Amazon, Piccolo et al.

(1994) found higher NO_3^- concentrations in cleared forest than in intact forest during the first wet season following clearing but not during the second wet season when the forest was regrowing.

While evidence suggests that brief post-disturbance elevated solution NO_3^- concentrations are likely to be a general feature of tropical forests on weathered soils, some of the major controls on this pattern are still poorly understood. Uhl and Jordan (1984) reported higher solution NO_3^- concentrations for 3 years following clearing of forest on sandy soils at San Carlos, Venezuela, despite rapid regrowth of vegetation. Soil texture clearly influences rates of net N mineralization, net nitrification and denitrifying enzyme activity (Silver et al., 2000), but how this and other soil
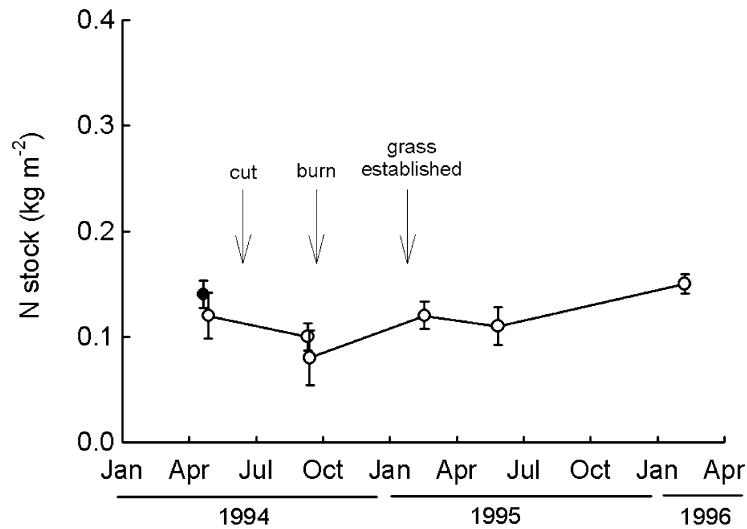
Figure 3. Soil N stocks in the top 10 cm measured before and after forest clearing. Stocks were calculated from soil N concentration and bulk density at each collection date. Open symbols are from the cleared forest, the closed symbol is from the intact forest. Values are means ± 1 standard error.

Table 2. Comparison of solution inorganic N fluxes and N oxides fluxes from Amazon terrestrial ecosystems. Solution fluxes are to below a depth of 1 m. N oxide losses were from Garcia-Montiel et al. (2001) and Melillo et al. (2001). NO fluxes from the cleared forest were estimated from fluxes from intact forest and forest plots fertilized with NH_4^+ (Steudler et al., 2002)

	Water flux mm y^{-1}	NH_4^+	NO_3^-	Total solution DIN flux	NO	N_2O	Total N oxide flux	Total N flux	% N flux as solution	% N flux as gas
		kg N ha^{-1} y^{-1}								
Forest	426	0	2.5	2.5	1.4	2.0	3.4	5.9	42	58
Cleared forest	546	0	24.4	24.4	3.0	4.9	7.9	32.3	76	24
5-year-old pasture	546	0.8	0.4	1.2	0.2	1.7	1.9	3.1	40	60
22-year-old pasture	546	0.1	0.4	0.5	0.2	1.2	1.4	1.9	26	74

characteristics influence post-disturbance N losses remain poorly known.

Nitrification that leads to elevated soil NO_3^- concentrations after disturbance is a source of acidity that can lower soil pH (Likens et al., 1969; Schlesinger, 1991). We measured no appreciable change in solution pH even though the pH of surface soils increased after clearing and burning (Moraes et al., 1996); a common trend as base cation-rich ash produced by forest burning becomes incorporated into the soil (Nye and Greenland, 1964). While ash incorporation after tropical forest clearing typically leads to long-lasting increases in the soil pH on a variety of

soil types (Markewitz et al., 2004; Neill et al., 1997a), the overall effects on soil solution pH were small.

Solution concentrations of NH_4^+ and NO_3^- at 0.3 m and 1.0 m in the forest and cleared forest declined with time after the onset of the rainy season (Figure 1). Because this pattern occurred in both forest and cleared forest, it is unlikely that this was caused solely by an increase in N demand of planted pasture grasses during this period, which only occurred in the cleared forest. Flushing of high NH_4^+ and NO_3^- concentrations that accumulated during the dry season (when net downward movement of soil solution

is absent) could have contributed to this result. This phenomenon occurs in tropical forests with distinct dry seasons (Garcia-Mendez et al., 1991), and both NH_4^+ and NO_3^- are known to accumulate in the soils at Nova Vida during the dry season (Garcia-Montiel et al., 2003; Neill et al., 1995). While the flushing of solutes that accumulate seasonally has been demonstrated to be an important mechanism that delivers a disproportionately large mass of solutes to streams in some temperate forests (Boyer et al., 1997; Creed et al., 1996), early wet season flushing of accumulated solutes has been suggested but not clearly demonstrated for lowland tropical forests.

We did not measure solution DON concentrations during forest clearing, but DON concentrations measured in later years (1998–2001) from the same lysimeters at 1.0 m in the intact forest showed that DON comprised 33% of total solution N (Neill et al., 2001). If DON concentrations in the forest during the 1994 water year were the same proportion to DIN measured during 1998–2001, annual DON fluxes to 1 m depth would have been approximately 1.2 kg N ha^{-1} compared with a DIN flux of 2.5 kg N ha^{-1}. In the younger pasture, DON made up 81% of total solution N (Neill et al., 2001) and flux would have been a greater proportion of total dissolved N flux, or 3.8 kg N ha^{-1} compared with a DIN flux of 1.2 kg N ha^{-1}. In the forest, this pattern differs from that in some old-growth temperate forests in high rainfall environments where DON overwhelmingly dominates total dissolved N exports (Perakis and Hedin, 2002; Sollins and McCorison, 1981), but it more closely resembles the proportions of DON and DIN in solution reported from old growth temperate forests or temperate forests that experience high N deposition (Campbell et al., 2000; Goodale et al., 2000). In the pasture, DON made up a greater portion of solution N losses because DIN losses were so greatly reduced.

Solution fluxes of DIN of 2.5 kg N ha^{-1} y^{-1} to below 1 m depth estimated for the forest at Nova Vida were generally large compared to equivalent fluxes measured in temperate forests outside of regions of very high N deposition (Currie et al., 1996; Qualls et al., 2000) and they were at the high end of the range reported as streamwater losses from a variety of temperate forest watersheds (Campbell et al., 2000; Goodale

et al., 2000; Vitousek et al., 1979). Even so, the fluxes from the Nova Vida forest represented only 1% of total annual net N m mineralization of 237 kg N ha^{-1} that were measured in this forest in 1993 (Neill et al., 1995). In the first year after clearing, fluxes from the Nova Vida forest rose to 24.4 kg N ha^{-1}, or 12% of an annual net N mineralization of 266 kg N ha^{-1} y^{-1} that was measured in the first year after the burn (Melillo et al., 2001). This was near the middle of a wide range of observed losses in streamwater from clear-cut temperate watersheds (Vitousek et al., 1979).

In our manipulation, we chose to clear a single large 3-ha plot. Our previous work on the adjacent forest and pastures on similar topography, soils and management at Nova Vida suggested that differences between the four sites compared in this study were reasonably attributed to differences in land cover and not other factors. While replicate sites would have been preferable, it would not have been possible to duplicate the effects of clearing and the cattle-effects of trampling, grazing and other factors in small plots.

Longer-term effects of pasture establishment

Our previous work on N pools and transformations in Rondônia indicated that pasture establishment was associated with increases in extractable soil NH_4^+, declines in extractable NO_3^-, and declines in rates of net N mineralization and net nitrification (Neill et al., 1995). These patterns appear to hold across a wide variety of lowland tropical landscapes where deforestation is a major land use change (Dias-Filho et al., 2001; Markewitz et al., 2004; Neill et al., 1997b; Reiners et al., 1994; Verchot et al., 1999). Clearing raised soil DIN pools relative to the forest, but these changes were short lived and the typical pattern of lower rates of N transformation in pastures and lower soil solution NH_4^+ and NO_3^- concentrations in pastures was established likely within 1 year and certainly within 3 years (Melillo et al., 2001; Neill et al., 1995). The effect over many years of this land use transition on N cycling was to reduce N losses from deforested landscapes compared with the original forest.

While solution fluxes are potentially important for understanding regional effects of deforestation on stream water biogeochemistry, quantifying the movement of total dissolved N from soil solution to streamwater will depend on several factors not examined here. We measured solution concentrations at 1.0 m depth, but plant N uptake may occur at deeper depths over large regions of the lowland Amazon where soils are deep and can easily exceed 10 m (Nepstad et al., 1994). Nitrate retention at depth, including by anion exchange, could also be a mechanism that regulates N movement to groundwater and streams (Matson et al., 1987). In addition, a wide variety of flow-paths that connect soil solution to streams are possible in tropical landscapes and some shallow water flows may bypass deeper soils, particularly in pastures and on Ultisols where less permeable B horizons initiate horizontal water movement (Elsenbeer, 2001). Further, we did not measure solution concentrations of DON during the clearing sequence. There are indications from temperate forests that DON concentrations and export respond much less dramatically to disturbance and land use history than NO_3^- concentrations (Goodale et al., 2000; Qualls et al., 2000), but it is currently not known how well this applies to forests on weathered tropical soils.

Relative magnitudes of solution and gaseous fluxes

We compared solution fluxes with soil fluxes of N_2O measured in the same forest, pastures and clearing sequence (Garcia-Montiel et al., 2001; Melillo et al., 2001; Steudler et al., 2002). This allowed us to construct a time-course of change for both solution inorganic N fluxes and N oxide fluxes from forest through clearing and subsequent 20 years of pasture establishment. These estimates are uncertain for several reasons. First, NO could not be predicted with any certainty from moisture or soil N status during clearing. Estimation of NO based on responses to elevated soil NH_4^+ as occurred during forest clearing, however, places a reasonable upper bound on NO fluxes. In addition, we did not attempt to estimate fluxes of N_2. Our laboratory incubations of soils conducted over the range of moistures likely to be encountered in the field indicate that the N_2 released varies but never exceeds 1.8 times N_2O

fluxes in the forest or about 1.3 times N_2O fluxes in pasture (unpublished data). It is also important to note that gas fluxes calculated here do not necessarily indicate ecosystem N losses. Consumption of NO by reaction with ozone to deposit NO_2 on leaf surfaces substantially reduces the amount emitted above plant canopies (Jacob and Wofsy, 1990; Rummel et al., 2002). This is not important for N_2O, which is stable and likely to be exchanged above the surface boundary layer into the lower troposphere (Prather and Ehhalt, 2001).

Our estimates indicated that N oxide fluxes comprised 58% of total N fluxes in forest and 60–74% of fluxes in established pastures (Table 2). Nitrogen oxide fluxes changed less in response to the disturbance of forest clearing than solution fluxes (Figure 4), approximately doubling from 3.4 to 7.9 kg N ha^{-1}, while solution fluxes increased nearly 10-fold in the first year after clearing (Table 2). Nitrous oxide and NO fluxes made up only 24% of total DIN plus gaseous fluxes in the first year after disturbance.

Overall, we found that forest clearing led to both greater solution DIN losses and greater soil N oxide emissions, but that solution losses were of much greater magnitude and of shorter duration than gaseous losses. Currently, approximately 650,000 km^2 of Amazon forests (INPE, 2005) have been replaced by other land coves, largely pastures that have lower rates of soil N cycling and lower solution and N oxide losses. It is also important to understand that this may change in the near future. Tropical forests of the Amazon are predicted to experience both increased rates of N deposition and continued high rates of disturbance and conversion to other land uses (Fearnside, 2001; Galloway et al., 1998; Matson et al., 1999; Matthews, 1994). Fertilization with N is becoming more common on aging Amazon pastures and fertilized row crops or N-fixing soybeans are replacing pasture in many locations. Understanding the transformations and losses of N that follow increases in soil N availability in tropical forest and pasture will be key to understanding both land-atmosphere and land-water N fluxes from an Amazon landscape that has been made N-impoverished by deforestation but may
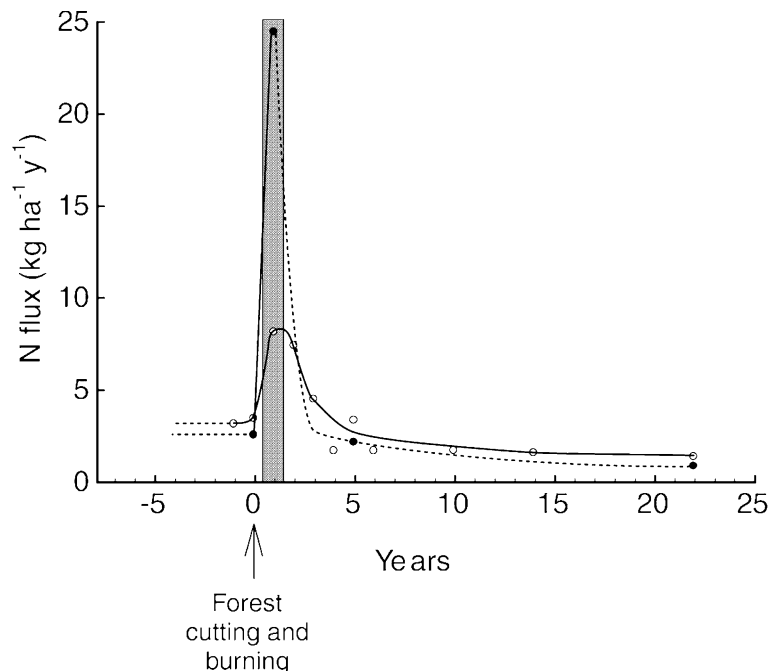
Figure 4. Annual fluxes of N oxide gases (N₂O + NO) and solution DIN to 1 m following forest clearing. Gaseous fluxes are open circles, solution fluxes are dark circles. Dashed lines represent where fluxes are extrapolated. The vertical bar represents the first year after the burning that marked the end of the clearing sequence.

become more N-enriched by other forms of agriculture in the future.

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Section editor: H. Lambers