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TOPICAL REVIEW

Direct nitrous oxide (N_2O) fluxes from soils under different land use in Brazil—a critical review

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Abstract

Brazil typifies the land use changes happening in South America, where natural vegetation is continuously converted into agriculturally used lands, such as cattle pastures and croplands. Such changes in land use are always associated with changes in the soil nutrient cycles and result in altered greenhouse gas fluxes from the soil to the atmosphere. In this study, we analyzed literature values to extract patterns of direct nitrous oxide (N_2O) emissions from soils of different ecosystems in Brazil. Fluxes from natural ecosystems exhibited a wide range: whereas median annual flux rates were highest in Amazonian and Atlantic rainforests (2.42 and 0.88 kg N ha^{-1}), emissions from cerrado soils were close to zero. The decrease in emissions from pastures with increasing time after conversion was associated with pasture degradation. We found comparatively low N_2O-N fluxes from croplands (-0.07 to 4.26 kg N ha^{-1} yr^{-1}, median 0.80 kg N ha^{-1} yr^{-1}) and a low response to N fertilization. Contrary to the assumptions, soil parameters, such as pH, C_{org}, and clay content emerged as poor predictors for N_2O fluxes. This could be a result of the formation of micro-aggregates, which strongly affect the hydraulic properties of the soil, and consequently define nitrification and denitrification potentials. Since data from croplands mainly derived from areas that had been under natural cerrado vegetation before, it could explain the low emissions under agriculture. Measurements must be more frequent and regionally spread in order to enable sound national estimates.

1. Introduction

The Food and Agriculture Organization of the United Nations (FAO 2014) has ranked Brazil as the third largest emitting country of greenhouse gases (GHG) from agriculture for the year 2012. One reason is the significant increase in area used for agriculture in recent years (Cederberg *et al* 2009). However, GHG budgeting at national scales is full of uncertainties, particularly for such large countries, and too little is known about the processes that affect such estimations. Literature reviews are one viable first step towards improvements (e.g. Jungkunst *et al* 2006), which can eventually lead to further extrapolations to the national scale (e.g. Brocks *et al* 2014). For the agricultural sector nitrous oxide (N_2O) is one of the most important GHG. It is emitted from soils by natural processes, which can be enhanced and potentially reduced by anthropogenic activities, such as fertilization and land use changes. The principle underlying prerequisite for N_2O emission is the availability of nitrogen. Nitrogen in soils is increased by fertilization, which usually leads to increased N_2O emissions. The Intergovernmental Panel on Climate Change (IPCC 2006) assumes this relationship between N fertilization and N_2O emissions to be linear and defines an emission factor (EF) of 1% (1 kg of every 100 kg of applied N fertilizer is lost as N_2O-N). At the same time, the IPCC strives to improve this approximation to more detailed region-specific approaches. Based on data from temperate climates (e.g. Boeckx and van Cleemput 2001), as well as global scale data (Shcherbak *et al* 2014), a single emission factor is imprecise. Shcherbak *et al* (2014) rather proposed a nonlinear relationship at the global scale. These insights lead to the conclusion that the relationship between fertilizer input and N_2O emissions must vary according to environmental settings like climate and soil conditions (Jungkunst *et al* 2006). Considerably less data exists for the tropics compared with temperate regions (Shcherbak 2014). However, data available from tropical areas indicate that the emission factor used by the IPCC overestimates measured fluxes (Madari *et al* 2007, Jantalia *et al* 2008, Alves *et al* 2010, Cruvinel *et al* 2011, Alvarez *et al* 2012, Carmo *et al* 2013, Carvalho *et al* 2013, Lessa *et al* 2014).

Estimations for national N_2O inventories are challenging, because soils are diffusive emitters and direct N_2O fluxes show extremely high temporal and spatial heterogeneities (e.g. Groffman *et al* 2009). The largest emissions of N_2O mainly result from denitrification under hypoxic conditions (Davidson *et al* 2000); especially during changes between well aerated (WFPS at 40–60%) and wet (WFPS $\geqslant$ 80%) conditions (Vor *et al* 2003). Consequently, when soil moisture increases during wet seasons, N_2O emissions commonly increase as well (e.g. Luizão *et al* 1989). In irrigation experiments by Vasconcelos *et al* (2004) and Carvalho *et al* (2013), N_2O fluxes increased after irrigation during the dry season in Brazil.

Luizão *et al* (1989) and Sotta *et al* (2008) reported that finer textured soils have a higher N availability. Additionally, Matson *et al* (1990) and Sotta *et al* (2008) measured higher N_2O losses from clay soil compared to sand soil. Since tropical soils of Brazil are commonly rich in clay and experience regular changes in moisture through seasonal rainfall patterns, Brazilian soils should emit higher amounts of N_2O than temperate soils. However, reported measurements show fairly low emission levels. Stable micro-aggregates, which form due to adhesion of fine soil particles and (iron-)oxides, are known to create a coarser structure. This leads to better drainage and more oxic conditions than would be expected by measured clay contents. When compared with other predominant soils at the global scale, the general role of clay content as an indicator for N_2O fluxes can be questionable.

Besides accounting for N_2O emission from specific land use types, N_2O dynamics during actual land use changes should be accounted for, particularly with respect to the rapid land use change happening in Brazil. The expansion of cattle ranching is suggested to be the main driver of recent deforestation in the Legal Amazon (e.g. Barona *et al* 2010). However, Morton *et al* (2006) reported an increasing trend of cropland deforestation (direct conversion of forest to cropland) between 2001 and 2004 in the state of Mato Grosso.

Tropical forests have high rates of biological turnover and decomposition. High soil moisture and N availability increase these soils emissions of N_2O (Davidson *et al* 2000). Breuer *et al* (2000) estimated a N_2O budget of 3.55 Tg N_2O-N yr^{-1} from tropical rainforest soils. In contrast to rainforests, reported emissions from soils under natural cerrado vegetation (forest to treed grassland ecosystems) were usually very low (Davidson *et al* 2001) or even negative (e.g. Carvalho *et al* 2013). Consequently, knowing the natural ecosystem present before the land use change may be as important in estimating N_2O emissions as knowing the current agricultural land use.

Here, we focus on the regional scale within Brazil in order to improve estimates for atmospheric N_2O increase. Considering single studies without a systematic scientific compilation is neither sufficient to identify regional measurement gaps nor to identify underlying key processes. The value of understanding specific soil and management properties to indicate N_2O fluxes not only lies in better approximations, but also feeds process-based models that are eventually needed for scenario calculations to derive mitigation strategies. A systematic review additionally can help to derive research strategies and to set the basis for regional and temporal N_2O measurement recommendations, based on revealed relationships with environmental parameters. The improved process understanding enables better national estimations.

To provide this we used reported emissions of N_2O-N from soils under different land use. Specifically, we aimed to (1) compare reported N_2O-N fluxes from different land use types and define average annual emissions, (2) evaluate if specific soil and management properties can serve as an indicator of N_2O fluxes, and (3) determine knowledge gaps for improvements of future national N_2O inventory and process understanding.

2. Materials and methods

2.1. Data collection and calculation

We searched English literature for N_2O-N flux data from soils under different land use across Brazil using online databases (Web of Science, Science Direct, Scielo (Brazilian)) and search engines (Google Scholar) between March 2014 and January 2015. Search queries initially included the keywords 'N_2O' AND 'soil' AND 'Brazil', which resulted in large numbers of studies (e.g. 73 studies in the Web of Science). We further specified the search by adding the keyword 'conversion'. Additional specification of the single land use types (AND 'rainforest'/'pasture'/'cropland') or geography (AND 'Amazon'/'Southern Brazil') did not result in additional studies. According to the guidelines of Aiassa *et al* (2015) on how to proceed on systematic reviews, we made use of personal contacts and contacted research groups in Brazil (EMBRAPA) to improve and expand our search strategies towards Portuguese studies that might have been missed using the Scielo database and due to linguistic difficulties. No time frame was set in terms of the age of the studies

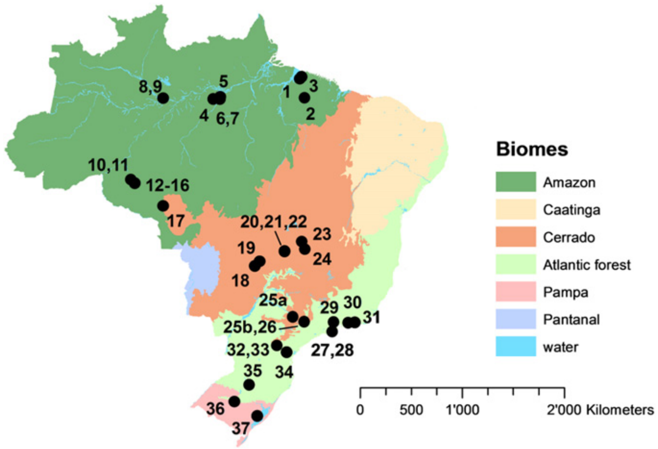

Figure 1. Sites with available N_2O-N flux data in different biomes in Brazil (IBGE 2013). Numbers are according to table A1 in the appendix.

—the aim was to gather a good geographic coverage of Brazil. Only tabular values were analyzed; we did not extract data from graphs. Data sets were divided into three categories: (1) data from natural landscape units (Amazon forest, Atlantic forest, and cerrado), (2) land that was converted to pastures, and (3) land under agricultural management and fertilizer application. Since the third category contained long-term (one year or more), as well as short-term experiments (weeks or months), studies within this category were again divided according to the duration of the measurements. Short-term experiments usually presented as cumulative fluxes over the specific time period rather than annual emissions. Nevertheless, we treated short-term experimental data as long-term data if authors extrapolated to annual values (e.g. Metay *et al* 2007). Similarly, cumulative data resulting from different crops within a rotation (e.g. corn/bean rotation, Cruvinel *et al* 2011) were extrapolated to annual values, if the whole rotation cycle covered one year. Reported units varied among the studies, thus we converted data sets to identical units (e.g. kg N ha^{-1} for N_2O fluxes and fertilizer inputs, or g kg^{-1} for C_{org}). If N_2O fluxes were not given in annual emissions, but mean daily values were given, reported data were projected to one year. For studies which distinguished between dry and wet seasons, the length of the specific period was used in the extrapolation. Soil types were classified according to the World Reference Base (IUSS 2014). Soil texture, usually expressed as clay content, carbon content, and pH have been shown to influence N_2O-N emissions. Thus, we looked at correlations with available soil properties, as well as with the amount of applied fertilizer. The latter did not include studies including soybean because legumes are treated differently by the IPCC. Forests and cerrados were not included in the regression analysis because information on the specific soil properties were derived from the mineral soil, but not from the overlying humus layer. We differentiated between fertilized and unfertilized plots (usually pastures and croplands). Except for correlations with the amount of applied fertilizer, we only regarded fertilizer-induced emissions (N_2O-N/added N). Data from pastures were ordered according to the time since establishment, as pasture ages turned out to be a meaningful factor in forest areas converted to pastures (e.g. Wick *et al* 2005, Neill *et al* 2005).

2.2. Statistical analyses

We used the linear regression method to analyze the relationships between soil properties and N_2O-N emissions. The influence of the applied fertilizer was additionally adapted by a nonlinear model, following the suggestions of Shcherback *et al* (2014). Relationships were regarded as being statistically significant for a *p* value of (or smaller than) 0.05. The regression analyses and creation of graphics were conducted using the R software (version 2.15.0).

3. Results

In total, 37 study sites were analyzed based on land use, soil properties, management, and fertilization (table A1).

The geographical locations of the sites (figure 1) divided Brazil into regional land use types: studies conducted in the northern states (e.g. Pará, Rondônia) mainly dealt with N_2O-N emissions from rainforest and cattle pastures. These land use types represented the fundamental land use change (deforestation) in the Amazon region. In contrast, studies from the central and southern states focused on conversion of cerrado area to croplands and the influence of different crop rotations, management, and fertilization strategies.

Annual N_2O-N emissions collected in this study (tables 1 to 3) were differentiated according to the land use type. Table 1 summarizes data from forest (Amazon and Atlantic forest), cerrado, and pasture sites. Table 2 summarizes data from experiments on cropland.

Table 1. Annual N_2O emissions with minimum, median, and maximum value from forest, cerrado, and pasture soils (references are according to table A1).

	Annual N_2O emissions [kg N ha^{-1}]			
Biome	Min	Median	Max	Reference
Forest	0.38	2.29	16.20	1, 2, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14, 27, 28, 31
Amazon Rainforest	0.38	2.42	16.20	1, 2, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
Atlantic Rainforest	0.44	0.88	3.42	27, 28, 31
Cerrado	−0.09	0.14	1.19	2, 17, 18, 19, 23, 24
Pasture [age]				
⩽10	1.32	2.52	10.16	9, 10, 12, 14
>10	−0.27	0.90	3.62	2, 10, 11, 12, 14, 18, 19, 23, 28, 29

Table 3 gives an overview of N_2O-N fluxes from short-term experiments on grassland, pasture and cropland.

3.1. Soil under natural vegetation

In general, N_2O emissions from forest soils were higher than emissions from pasture sites (e.g. Verchot *et al* 1999, Steudler *et al* 2002, Wick *et al* 2005, Carmo *et al* 2012). Reported N_2O-N fluxes from forest soils were positive without exception, but varied from 0.38 up to 16.20 kg N ha^{-1} yr^{-1}. Rainforests (Amazonas and Atlantic rainforest) differ considerably from cerrado: the highest emission (16.20 kg N_2O-N ha^{-1} yr^{-1}) was reported from a forest site in the Amazon, whereas the maximum emission from Atlantic forest was much lower (3.42 kg N_2O-N ha^{-1} yr^{-1}). In contrast, emissions from cerrado sites were exceptionally low (median: 0.14 N_2O-N ha^{-1} yr^{-1} with a maximum of 1.19 N_2O-N ha^{-1} yr^{-1}), and often below the detection limit (<0.6 ng cm^{-2} h^{-1}).

Data presented in table 1 includes studies conducted in primary or moderately altered forests. Studies from Verchot *et al* (1999), Vasconcelos *et al* (2004), and Coutinho *et al* (2010) present data from secondary forests of 12, 20, and 34 years after reforestation. Here, annual N_2O-N emissions amount to 0.35, 0.94, and 0.88 kg ha^{-1}, respectively. These results are lower than the median annual emission from primary forests reported in this study and are in the range of annual emissions from pastures.

3.2. Pasture soil

N_2O-N fluxes from pasture soils varied widely, too, but emissions from pastures younger than 10 years were significantly ($p < 0.05$) higher than from older pastures (table 1, figure 2). Thus, we differentiated data from pastures of 10 years and younger from those older than 10 years. Neill *et al* (2005) modeled the behavior of annual N_2O emissions from forests and pasture sites of different ages as an exponential function. When we fitted an exponential function to our data ($y = 0.65 + 4.15{*}\exp(-0.10\,x)$, with y = flux of N_2O-N [kg ha^{-1} yr^{-1}] and x = pasture age), we found a similar decrease in N_2O-N emissions with pasture age (figure 2).

For short-term experiments on pastures under additional fertilization and soil management (table 3), highest emissions were reported from pastures under urine application and tillage management (5.87 and 2.23 kg N ha^{-1}). These studies were not included in the nonlinear regression (figure 2).

3.3. Cropland soil

Highest annual emissions occurred from crop-pasture rotations (4.26 kg N ha^{-1} yr^{-1}) and cropland under tillage treatment (2.42 kg N ha^{-1} yr^{-1}). The overall median from cropland soils was 0.80 kg N ha^{-1} (table 2).

We could use only 6 studies (15, 16, 22, 26, 34, and 36 in table A1) for calculating an emission factor (EF), because these allowed for subtraction of background emissions. Of these, only study 22 (Santos *et al* 2008) presented long-term measurements focusing on corn and bean cultivation. EFs were 0.24% for corn and 0.13% for bean. EFs from short-term experiments (table 3) ranged from 0.13 to 5.14%, with a median of 0.38%.

Except for pH ($R^2 = 0.21$, $p = 0.06$), correlations with N_2O-N/added N were significant (figures 3(a)–(c) and table 4), but not important. Clay contents covered a range from 13 to 86%, and the linear regression implied only a slightly increasing trend for fertilized plots (0.0009) and a slightly decreasing trend (−0.003) for unfertilized plots with higher clay contents. N_2O-N/added N slightly decreased with increasing pH (−0.03) or carbon content (−0.06). On unfertilized plots, emissions increased with increasing pH (0.41), but emissions decreased with increasing carbon content (−1.38).

N_2O-N fluxes increased with applied N fertilizer (figure 3(d)). In their global meta-analysis, Shcherbak *et al* (2014) found that a nonlinear model better described the relationship between N_2O fluxes and fertilization than a linear model. In contrast, we found that a nonlinear model of the Brazilian data did not result in a better description of the relationship between emissions and fertilization compared with a linear model ($R^2 = 0.20^*$ for both linear and nonlinear model). Our nonlinear model ($y = 0.93 + 1.98 \cdot x - 0.15 \cdot x^2$, with y = N_2O-N flux [kg ha^{-1} yr^{-1}] and x = applied fertilizer [kg N ha^{-1}]) does not compare well with that of Shcherbak *et al* (2014) (figure 3(d)). Besides the different intercept, which is caused by reported N_2O-N fluxes at low and even zero fertilization, our model has a lower slope, due to the low emissions. Table 4 shows more detailed results of the regressions.

Table 2. Land use, treatment, N application, and annual N_2O fluxes from soils under agricultural use in Brazil. Site numbers are according to table A1.

Site no.	Land use	Treatment	Duration [days]	Applied N [kg N ha^{-1}]	Annual N_2O [kg N ha^{-1}]
		improved fallow plot			
3	cropland (agroforestry)	*inga edulis*	~365	0	0.71
		acacia mangium		0	0.88
		control		0	0.82
		conventional tillage:			
17	cropland	rice (1 year)		7[a]	0.85
		rice (2 years)		15	0.63
19	cropland	crop succession		92	0.57
		crop-pasture rotation		222	2.00
20	cropland	disc harrowing (15 cm)	~182	114	0.04[b]
		direct seeding		114	0.01[b]
22	cropland	corn		0	0.35
			365	80	0.54
		beans		0	0.20
				80	0.30
		MF (NPK)		60	
		MF (NK) + filter cake		122	
	a) cropland (*plant cane*)	MF (NP) + vinasse	~365	87	
		MF (N), + vinasse + filter cake		149	
		MF		120	
25	b) cropland (*ratoon cane*)	MF + vinasse		142	
		7 Mg trash, MF		120	
		7 Mg trash, MF + vinasse		142	
		14 Mg trash, MF	~242	120	
		14 Mg trash, MF + vinasse		142	
		21 Mg trash, MF		120	
		21 Mg trash, MF + vinasse		142	
32	cropland	tillage	365	165	2.42
		no-tillage		165	1.26
33	pasture/cropland	integrated crop-livestock		225	4.26
		(corn/grazed annual-ryegrass)	365	(+excreta)	
	cropland	continuous crop			
		(annual-ryegrass)		165	1.26
		native vegetation		0	0.65
		no tillage:			
		sorghum/wheat (year 1 and 2)[b]	365	195/253	0.65
35	cropland	corn/wheat (year 1 and 2)[c]		162/94	0.71
		conventional tillage:			
		sorghum/wheat (year 1 and 2)[b]	365	171/253	0.71
		corn/wheat (year 1 and 2)[c]		141/78	0.80
		pigeon pea + corn		367.6	1.32
		lablab + corn		167.5	1.12
37	cropland	vetch + corn	347	144.8	0.81
		black oat + corn		98.3	−0.07
		black oat + vetch/corn + cow pea		231.7	1.32
median					0.80

MF: Mineral fertilizer

[a] From Carvalho et al (2007).

[b] Reported as annual emissions.

[c] Soybean/vetch (year 1), sorghum/wheat (year 2).

[d] Maize/wheat (year 1), soybean/vetch (year 2).

4. Discussion

4.1. Importance of land use

4.1.1. Natural vegetation

The different emissions from cerrado and rainforest reveal the high variability of natural N_2O fluxes. While emissions from Brazilian rainforest sites were generally high (2.29 kg N_2O-N ha^{-1}), but still within the range of emissions reported for temperate (−0.1 to 4.9 N_2O-N ha^{-1} yr^{-1}, Jungkunst et al 2004) and Australian tropical forests (1.15 to 5.36 N_2O-N ha^{-1} yr^{-1}, Breuer et al 2000), N_2O fluxes from cerrado soils were often close to zero, below detection limits (e.g. Pinto et al 2002, Varella et al 2004) or negative

Table 3. Land use, treatment, N application, and cumulative N_2O fluxes from soils under agricultural use in Brazil. Site numbers are according to table 5.

Site no.	Land use	Treatment	Duration [days]	Applied N [kg N ha^{-1}]	Cum. N_2O [kg N ha^{-1}]
		control		0	0.07
15	pasture	tillage		42	0.23
		no-till, co-planting rice	~182	33	1.10
		no-till, co-planting soybean		0	1.10
		control		0	0.07
16	pasture	tillage	180	42	2.23
		no-tillage		33	1.62
		urine application	94/37	396[a]/683[b]	
21	pasture	dung application	94/37	188[a]/346[b]	
		corn	173	155.3	0.20
24	cropland	irrigated bean	135	102.7	0.20
		soybean	153	21.2	0.10
		cotton	258	0	0.10
26	grassland	control		0	0.02
		urine application	30	860	1.69
		vegetation cycles			
30	grassland	V1	162	80	0.44
		V2	178	100	0.57
		V3	149	80	1.47
		control		0	0.04
34	pasture	urine application	90	2200	5.87
		dung application		1110	1.43
		fertilization with pig slurry			
36	cropland	no-tillage	28	191	0.40
		conventional tillage		191	0.51

[a] Dry season.
[b] Wet season.

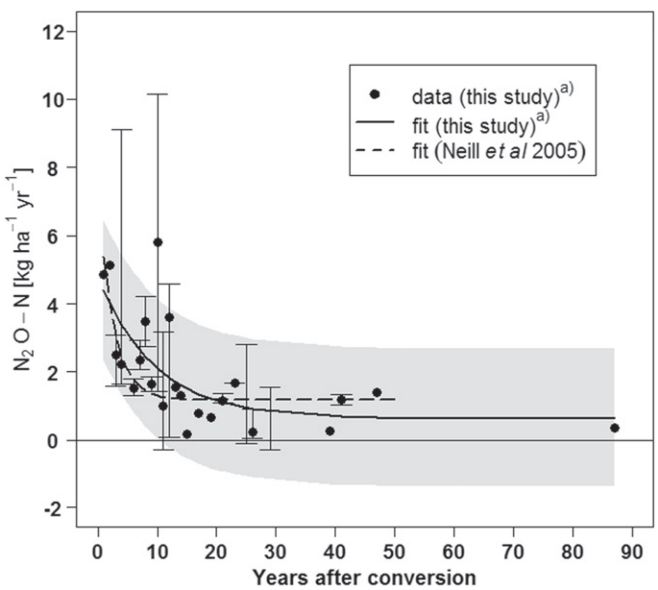

Figure 2. An exponential model of annual N_2O-N flux data ($n = 24$) dependent on years after conversion ($R^2 = 0.48$). Pasture ages varied from 1 to 87 years. The gray shaded area represents the 95% confidence interval. The broken line is the fit of Neill *et al* (2005), based on N_2O-N emissions from pastures of different ages ($n = 9$) in the state of Rondônia ($y = 1.18 + 6.21*\exp(-0.39\,x)$). In their study, pasture ages varied from 1 to 41 years. a) Including data from Neill *et al* (2005).

(Verchot *et al* 1999). Nitrification is a more important source of N_2O emissions in cerrado soils because of the better drainage caused by the coarse soil structure (Pinto *et al* 2002). Consequently, these soils become increasingly important in terms of nitric oxide (NO) emissions. Pinto *et al* (2002) assumed low nitrification

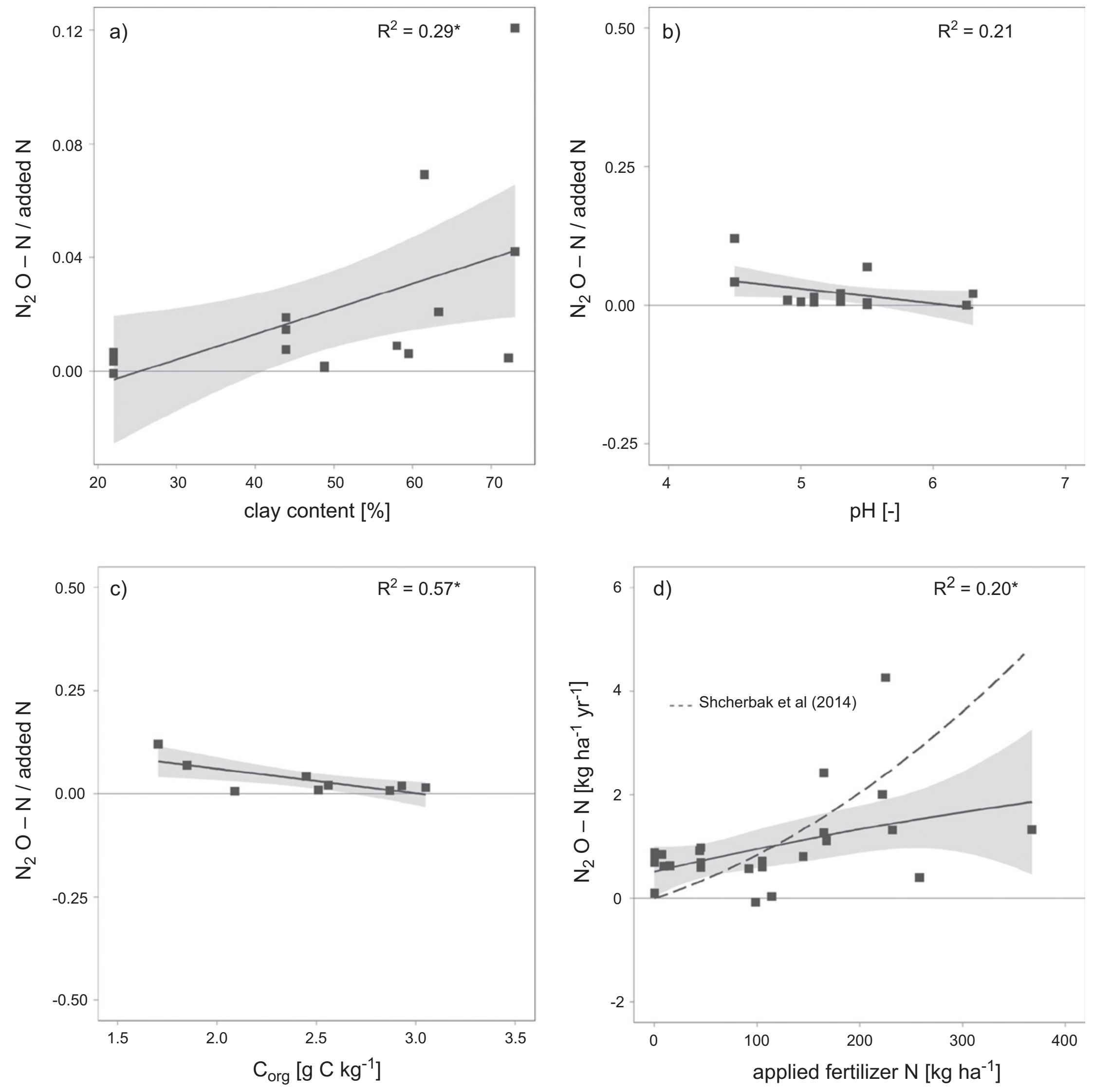

Figure 3. Relationships between N_2O-N fluxes [kg ha^{-1} yr^{-1}] per added N [kg N ha^{-1} yr^{-1}] and (a) soil clay content [%], (b) pH, (c) carbon content [%], and relationship between N_2O-N fluxes and (d) the annual amount of applied fertilizer N [kg ha^{-1} yr^{-1}]. Figure 3 (d) also includes the model of Shcherbak *et al* (2014) (excluding N-fixing crops) (broken line). Only fertilized plots (croplands and pastures) are presented. Gray shaded areas represent the 95% confidence intervals; see table 4 for more precise information on the regressions. $^* = p$-value < 0.05.

Table 4. Intercepts and slopes (including the lower and upper values of 95% confidence interval) for linear regression between soil properties and N_2O-N/added N ratio for fertilized and unfertilized plots, and applied N and N_2O-N fluxes for the linear and nonlinear regression. $^* = p$-value < 0.05.

	Clay content		pH		C_{org}		Applied N	
Regression parameter	fertilized	unfertilized	fertilized	unfertilized	fertilized	unfertilized	linear	nonlinear
intercept	−0.22	1.68	0.16	−1.29	0.18	3.84	0.544	0.93
slope	0.000 09*	−0.003	−0.03	0.41*	−0.06*	−1.38	0.004	1.98*
lower	0.0002	−0.017	−0.06	0.07	−0.10	−9.04	0.0009	0.43
upper	0.0016	0.010	0.002	0.74	−0.02	6.28	0.0068	3.54
slope								−0.15
lower								−1.71
upper								1.41
R^2	0.29*	0.007	0.21	0.51*	0.57*	0.23	0.20*	0.20*

rates and low NO_3^- contents resulted in low N_2O fluxes. However, further attention should be paid to cerrado soils, in order to identify the underlying processes.

The Amazonian forest soils in the north of Brazil showed higher emissions than those of the coastal Atlantic forests in the south-east of the country. Only 3 studies dealt with the Atlantic forest, but 13 were found for the Amazonian rainforest. More studies from the Atlantic forest would help to confirm this difference between the two forest types. Emissions from secondary forests (12, 20, and 34 years after reforestation) ranged between 0.35 and 0.94 kg N_2O-N ha^{-1} yr^{-1} and were lower than from primary forests. This suggests that N cycles in these reforested areas had not completely recovered. Regardless of the high emissions from rainforest soils compared with soils under other land use, precise knowledge concerning emissions during the conversion from rainforest to pasture is missing. This is a key aspect, since some studies report increased N_2O emissions from soil after conversion of forest to pasture (Keller *et al* 1993, Veldkamp *et al* 1999, Davidson *et al* 2001, Melillo *et al* 2001). They explain this event with a temporal increase of N availability. The removal of plants as a sink for nutrients causes very high nutrient availability in soils (Bormann and Likens 1979), which is known to increase N_2O emissions at barren sites (Repo *et al* 2009). In addition to the emissions from soil to atmosphere, soil–water degassing can be an important source for N_2O fluxes directly after forest clear-cutting (Bowden and Bormann 1986).

4.1.2. Pastures

According to Keller and Reiners (1994) and Melillo *et al* (2001), young pastures have increased emissions directly after a clear-cut, followed by decreasing emissions as the pastures age. Decreasing denitrification rates ($N_2O + N_2$) in mid-successional sites compared with primary forest and early successional sites may explain this trend (Robertson and Tiedje 1988).

The duration of higher N_2O emissions after the creation of a pasture varies from 3 months (Elligson *et al* 2000), to over 2 years (Melillo *et al* 2001), to up to 10 years (Keller *et al* 1993). In our review, we found that N_2O emissions from young pastures ($\leqslant$10 years) were significantly higher than from older pastures. According to Davidson *et al* (2001), *Brachiaria spp.* grasses, which were introduced from Africa (Boddey *et al* 2004) and are commonly used for pastures in Amazonia, can be effective sinks for soil N. Quick immobilization of nitrogen that is released after the disturbance of the soil might delay the degradation of the pasture. Subbarao *et al* (2009) found a reduction in N_2O emissions of more than 90% under plots with *Brachiaria* species compared with soybean plots. The *Brachiaria* roots produce and deliver nitrification inhibitors to soil-nitrifier sites (Subbarao *et al* 2009). In grazed *Brachiaria* pastures, intense uptake of nitrogen by grazing animals degrades pastures (Boddey *et al* 2004). The decrease of available N in the litter leads to a reduction in the amount of N available for plant growth. Cerri *et al* (2005) and Hohnwald *et al* (2006) also report that many pastures suffer from degradation (declining fertility and grass productivity, and increasing weed cover) already 4 to 10 years after establishment. Thus, pastures are unsustainable—a point supported by our finding of decreasing N_2O fluxes from pastures about 10 years after conversion.

4.1.3. Croplands

Except for one study (site no. 3), data from croplands were derived from areas that had been under cerrado vegetation before. This might justify the low emissions from cropland, since cerrado soils appear to be a less considerable source for N_2O fluxes. Although N fertilization increased emissions for short periods of 3 to 7 days after application, the reaction of the soil to N addition at the annual scale was very low. For application rates below 100 kg N ha^{-1}, which are frequently applied, the reaction was negligible. The data collected in this study did not fully agree with the global nonlinear model suggested by Shcherbak *et al* (2014). Their model includes data from 84 locations worldwide, and is consequently designed for a much larger scale than our country-specific analysis. This difference emphasizes that large scale or global relationships may be inappropriate to apply to more regional aspects.

Annual fluxes of N_2O-N from cropland soils in Brazil ranged from −0.07 to 4.26 kg N ha^{-1}, with a median of 0.80 kg N ha^{-1} (table 2). This value is much lower than emissions reported by Roelandt *et al* (2005) from croplands in Canada (2.27 kg N_2O-N ha^{-1} yr^{-1}), Europe (2.47 kg N_2O-N ha^{-1} yr^{-1}), and the United States (3.35 kg N_2O-N ha^{-1} yr^{-1}). Highest emissions (figure 3(d)) occurred from the two cropland areas that were under either conventional tillage (2.42 kg N_2O-N ha^{-1} yr^{-1}; Piva *et al* 2012) or integrated cropping systems (4.26 kg N_2O-N ha^{-1} yr^{-1}; Piva *et al* 2014).

4.2. Importance of soils

4.2.1. Soil texture and structure

Soil texture and structure are highly relevant driving factors for N_2O emissions, mainly as controllers of water balances and nutrient availability. Generally, finer textured soils have a higher N availability (Luizão *et al* 1989) and consequently emit higher amounts of N_2O than sandy soils (Matson *et al* 1990). In a laboratory experiment, N losses from heavily weathered tropical soils were higher in a clay textured soil variation than from a sandy variation (Sotta *et al* 2008). Based on these findings Sotta *et al* (2008) suggest a higher N availability in a clay compared with a sand soil in Amazonian forests. Due to the good drainage of sandy soils, anaerobic conditions are rare and the potential for denitrification is low. In this study, clay proved to be a poor predictor for N_2O emissions from fertilized (slope = 0.0009) and unfertilized plots (slope = −0.003), most likely due to the formation of micro-aggregates and the associated different water retention properties. Tomasella *et al* (2000) mention the rapid decrease in water content between saturation and −100 kPa, and underline that

Brazilian soils behave more like coarse-textured soils. As a result, the water holding capacity does not necessarily increase with increasing clay content, and nitrification is more likely to occur than denitrification. Therefore, tropical soils with high clay contents, formation of micro-aggregates, and high drainage can be expected to emit less N_2O than is reported for temperate soils. Thus, the clay content is not necessarily a reliable indicator for N_2O-N emissions from Brazilian soils.

4.2.2. Soil chemical properties

In this study, neither pH nor C_{org} content seemed to have an influence on N_2O-N fluxes. However, fertilized and unfertilized plots differed. On fertilized plots, linear regression slopes were negative for both pH (−0.03) and C_{org} content (−0.06). For unfertilized plots, pH (slope = 0.41) turned out to be more predictive than C_{org} (slope = −1.38). Although both parameters have been reported to influence denitrification rates (Knowles 1982), the general findings within this study suggest that N_2O fluxes occur from nitrification. Thus, pH and C_{org} are of secondary importance. The contribution of pH and especially C_{org} in the formation of micro-aggregates, however, should be further investigated.

4.3. Knowledge gaps

Considerable data gaps exist for certain biomes. We found no reported N_2O emissions from the Caatinga and Pantanal biomes. Except for one site, data from croplands were derived from areas that had been established in cerrado areas, which were found to have extremely low emitting soils under natural vegetation. This lack of data hinders our ability to explain the low emissions from croplands, even after fertilizer application, and points out the need for measurements from additional land use types.

Since N_2O emissions exhibit short-termed emission peaks caused by environmental changes, high temporally resolved measurements are needed in order to explain mechanisms. Automated measurements enable continuous data acquisition, but the establishment of such studies is restricted to sites with a power supply and, for certain approaches, flat topography. Therefore, to achieve an adequate spatial measurement distribution across a large nation such as Brazil, we still have to rely on manual measurements that also take into consideration environmental (dry/wet cycles) and human induced (land conversion) changes. Biweekly measurements throughout the year, as done by most authors, are no longer suitable for increasing our understanding of biogeochemical processes. Furthermore, exact knowledge of how N_2O-N emissions change during land conversion is missing and desperately needed, since this time frame may likely account for large emission pulses that need to be accounted for in national budgeting.

Improving the existing understanding of the underlying processes, especially during land conversion can only be ensured by consistent monitoring and frequent measurements. Such monitoring data could provide the basis for further model refinement and allow for spatial and temporal extrapolations. The goal would be to develop regional solutions to improve national inventories. At this point, most process-oriented models have been developed for temperate conditions and application to tropical conditions is challenging. The different hydraulic conditions caused by micro-aggregates need to be considered, since adequate description of the soil moisture is a prerequisite for modelling N_2O-N fluxes from soils.

5. Conclusions

This systematic review on N_2O fluxes from Brazilian soils provides a good basis for further estimations and inventories on the national scale and eventually for explaining atmospheric N_2O increases. The land use types differed in direct N_2O fluxes from soils, but emissions were generally low. Systematic regional measurement gaps were identified, of which the Caatinga biome in northeastern Brazil is the most prominent example. Furthermore, land use types were not randomly distributed between biomes. In other words, pastures were studied in rainforest biomes, and croplands in cerrado biomes. Therefore, no predictions can be made on the behavior of N_2O fluxes from croplands in the rainforest biome. Soil parameters, such as pH, C_{org}, and clay content, had proven to be unsuitable as indicators for N_2O fluxes. Oddly enough, N_2O itself was found to be an indicator for the degradation stage of pastures, as emission levels decrease along with the productivity and years since conversion. A kind of tipping point from high to low N_2O emissions was found to be in the range of 10–15 years after forest conversion. N_2O is known to have high event-based emissions, which the current measurement concept did not account for. Future studies must focus on high temporal resolutions in order to promote process understanding. Otherwise, sound national inventories will not be possible.

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Appendix

Table A1. Geographical location, climatic condition, soil type and properties of studies with N_2O-N flux measurements in Brazil.

		Geographical location [°]						
Site no.	Reference	south	west	MAT [°C]	MAP [mm]	Soil type WRB	Clay content [%]	C_{org} [g kg^{-1}]
1	Vasconcelos *et al* (2004)	1.31	47.95	25.5	2539	Ferralsol	20	2.20
2	Verchot *et al* (1999)	2.98	47.51		1850	Ferralsol		2.47–3.02
3	Verchot *et al* (2008)	1.11	47.78	26	2500	Leptosol		
4	Keller *et al* (2005)	3.03	54.95	25	2000	Ferralsol	80	2.50
						Acrisol	38	
5	Davidson *et al* (2004)	2.88	54.95		2000	Ferralsol	60–80	
6	Silver *et al* (2005)	2.64	54.59	25	2000	Ferralsol	60	
						Acrisol	18	
7	Varner *et al* (2003)	2.64	54.59	25	2000	Ferralsol	80	
						Acrisol	38	
8	Livingston *et al* (1988)	3.0	60.0	26	1770	Ferralsol		
						Acrisol		
9	Luizão *et al* (1989)	3.0	60.0		2200	Ferralsol		
10	Melillo *et al* (2001)	10.16	62.81	25.6	2200	Acrisol	19–29	
11	Steudler *et al* (2002)	10.16	62.81	25.5	2200	Nitisol	23–29	
12	Garcia-Montiel *et al* (2001)	10.5	62.5	18.8–25.6	2270	Acrisol	<30	
13	Garcia-Montiel *et al* (2003)	10.5	62.5	18.8–25.6	2270	Acrisol	20–30	
14	Neill *et al* (2005)	10.5	62.5	25.6	2200	Acrisol	13–76	
15	Carmo *et al* (2005)	10.5	62.5		2270	Ferralsol		
16	Passianoto *et al* (2003)	10.5	62.5	25.5	2200	Acrisol		
17	Carvalho *et al* (2009)	12.48	60.0	23.1	2170	Ferralsol	73	1.71–2.77
18	Neto *et al* (2011)	17.78	51.91	23.3	1550	Ferralsol	54–68	1.85–2.90
19	Carvalho *et al* (2014)	17.36	51.48	23	1500–1800	Ferralsol	56–60	2.09–2.89
20	Metay *et al* (2007)	16.48	49.28	22.5	1500	Ferralsol	40	
21	Lessa *et al* (2014)	16.48	49.28			Ferralsol	43	
22	Santos *et al* (2008)	16.48	49.28	22.5	1460	Ferralsol		
23	Varella *et al* (2004)	15.65	47.75		1500	Ferralsol	57–74	2.41
								4.74
24	Cruvinel *et al* (2011)	16.3	47.5			Ferralsol	49–72	
		16.25	47.61			Ferralsol	68–76	
25a								
25b	Carmo *et al* (2013)	22.25	48.56	21	1390	Lixisol	11	
		22.68	48.55			Ferralsol	29	
26	Barneze *et al* (2014)	22.7	47.61			Nitisol		3.03
27	Neto *et al* (2011)	23.56	45.08		3050		35	4.59–9.15
		23.4	45.18		2300		21	
28	Carmo *et al* (2012)	23.31	45.08	19.1–25.5	2500	Acrisol	23	
		23.4	45.25					
29	Coutinho *et al* (2010)	22.73	44.95	20	1500	Ferralsol	30–36	
30	Morais *et al* (2013)	22.76	43.68	24	1300	Acrisol	45	
31	Maddock *et al* (2001)	22.75[a]	43.08[a]	21	1500–2250	Ferralsol	27–29	2.01
								2.03
32	Piva *et al* (2012)	24.78	49.95		1400	Ferralsol	44	2.87
								3.05
33	Piva *et al* (2014)	24.78	49.95		1400	Ferralsol	44	2.87
								2.93
34	Sordi *et al* (2014)	25.38	49.11		1400	Cambisol	44	2.50
35	Jantalia *et al* (2008)	28.25	52.4		1430	Ferralsol	63	
36	Giacomini *et al* (2006)	29.75	53.7			Luvisol		
37	Gomes *et al* (2009)	30.1	51.7	19.4	1440	Acrisol	22	

MAT: mean annual temperature.
MAP: mean annual precipitation.
[a] Coordinates from MMA/IBAMA (2006).

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