

Extensive estimate of nitrogen budget and critical loads at the scale of a semi-arid tropical grazed site (Dahra, Senegal).

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Abstract

Based on rare and original *in situ* measurements together with published data, we estimate a complete N budget for a semi-arid Sahelian grazed grassland ecosystem (or open tree savanna) located in Dahra (Senegal) for the years 2014 and 2017. Nitrogen (N) budgets in Africa have been estimated primarily for croplands, and very little data are available for grassland ecosystems. N inputs include biological N fixation, dry and wet N atmospheric deposition, and N input from livestock manure. N outputs include NO and N₂O emissions from soils, NO and NH₃ emissions from domestic fires and biomass burning, NH₃ volatilization from manure, N ingestion from livestock grazing, N uptake by trees and N soil leaching. Upper and lower bounds were estimated, depending on the

method used to quantify the contribution of livestock to the N budget. N inputs ranged between 11.7 ± 0.5 and 34.4 ± 0.5 kg N ha⁻¹ yr⁻¹ and N outputs ranged between 16.8 ± 1.5 and 46.1 ± 1.5 kg N ha⁻¹ yr⁻¹ on average for both years. N depletion (inputs minus outputs) was consequently estimated to be between -5.1 ± 2.0 and -11.1 ± 2.0 kg N ha⁻¹ yr⁻¹. The budget is dominated by the impact of livestock through grazing (62% of the outputs), NH₃ volatilization (15% of the outputs), manure (68% of the inputs) and atmospheric deposition (19% of the inputs). The nitrogen critical load (estimated by the Steady State Mass Balance approach) ranged from 16.7 ± 0.8 to 47.5 ± 1.7 kg N ha⁻¹ yr⁻¹, showing that the grazed grassland of Dahra was not yet threatened by an excess of nitrogen. The main uncertainties are related to the assessment of the critical load and to the quantification of parameters linked to livestock.

Keywords: Nitrogen, Africa, Sahel, leaching, N budget, critical loads

1 Introduction

Humanity has transgressed the boundaries of climate change, rate of biodiversity loss and changes to the global nitrogen (N) cycle (Rockström et al., 2009). In the Frontiers 2018/2019 report of the United Nations Environment Programme (Sutton et al., 2019), N was reported as an “Emerging Issue of Environmental Concern”. Humans are massively altering the global N cycle, causing multiple forms of pollution and impacts with for example fertilizer inputs and biomass burning, making N a key pollutant to tackle, at both the local and global scale. Atmospheric N deposition is recognized as one of the most important threats to biodiversity at the global scale (Sala et al., 2000; Dise et al., 2011). Atmospheric N deposition has increased in many temperate natural or semi natural ecosystems, considered as sensitive and/or vulnerable to N input, where historically N availability has been low (Rogora et al., 2006; Dise et al., 2011; Johansen et al., 2019; Kosonen et al., 2019). N deposition is expected to have a large global impact on soil biogeochemistry (Gaudio et al., 2015), and the third largest global impact on biodiversity by the year 2100 (Rizzetto et al., 2016), after land use change and climate change (Chapin et al., 2000).

Sub-Saharan Africa (SSA) is a region with too little N available (Sutton et al., 2019), where optimizing N use efficiency while minimizing N losses is a key challenge to be faced (Masso et al., 2017). From 1960 to 2009, Sahelian croplands showed high N use efficiency (> 60%) thanks to a proportion of N fixation larger than N fertilizer input (Lassaletta et al., 2014). More recently, total N in crop production has increased due to an increase in N input from fertilizers to African croplands, leading to a decrease in N use efficiency and an increase in N losses (Elrys et al., 2019). Furthermore, this trend is supposed to become even more pronounced in the near future: to achieve food self-sufficiency by 2050, inorganic N fertilizer input in African croplands is expected to increase and would inevitably involve large N losses to the environment (Elrys et al., 2020).

In SSA in general, and particularly in the Sahel, soils have a very low N content. This characteristic makes these semi-natural ecosystems vulnerable to increases in N deposition due to large scale anthropogenic activity. Until now, the changes in vegetation diversity and water drainage observed in pastoral areas has most likely resulted from climate variation than from land use changes (Mougin et al., 2009). However, increasing N deposition is predicted to significantly impact biodiversity (species composition, species richness) in the 21st century in semi natural ecosystems in SSA (Bobbink et al., 2010, Midolo et al., 2019). Meanwhile, the impact of N management and N deposition in semi-natural ecosystems in the Sahel is largely unknown. N critical loads, (“the quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge”; Nilsson and Grennfelt, 1988) are poorly assessed, and ecosystems may be disturbed by an excess of N sooner than initially thought (Bobbink et al., 2010, Bauters et al., 2018).

The N critical load approach has primarily been used to describe the vulnerability of an ecosystem (or a part of a given ecosystem, *e.g.* the first 30cm of the soil) to the atmospheric deposition of N (De Vries et al., 2007). N critical loads have been assessed at the regional scale in temperate (usually forested) regions in Europe (Downing et al., 1993; Probst and Leguedois, 2008; De Vries et al., 2010), and in the United States (Pardo et al., 2011) as well as at the global scale (Bouwman et al., 2002a). N

critical loads are calculated using a combination of field observations, empirical data or dose-response relationship parameterizations. Different approaches have been used to model critical loads. For example, steady-state empirical models and dynamic integrated soil-vegetation models have been adapted or upscaled in temperate regions (Bobbink and Roelofs, 1995; Posch and Vries et al., 1999; Belyazid et al., 2011, Probst et al., 2015). In SSA, the estimation of N critical loads has been limited by the availability of input data for the models. For instance, empirical data concerning the N budget were gathered by Penning de Vries & Djiteye (1982) and Breman and de Ridder (1991) in Sahelian pastoral landscapes. However, since that period such exhaustive studies have not been conducted in semi-natural grasslands. At the scale of Western Africa, Bouwman et al. (2002a) calculated that N deposition has increased from 5.9 Tg (10^{12}) N yr⁻¹ in 1992 to 6.5Tg N yr⁻¹ in 2015. They also conjecture that African water bodies may be affected by N eutrophication, with an analysis of N critical loads completely based on assumptions, and particularly uncertain in dry tropical ecosystems, including savannas. Investigations on N critical loads in Africa are scarce, especially for non-cultivated and agro-pastoral ecosystems, as highlighted in Bobbink et al. (2010). Further, livestock plays an important role in the regulation of soil N in Sub Saharan Africa (Powell et al., 2014). This study brings new data on detailed N budget and N critical loads. It uses new field data on N leaching and N atmospheric deposition together with data on livestock grazing for a semi-natural semi-arid ecosystem in Senegal (Dahra, Ferlo region), where experimental and modeling data are far less available than for cropping systems.

In this study, we calculate the N budget and N critical load assessment for a site in a semi-natural ecosystem (Dahra, Ferlo region, Senegal). We calculate the N budget in a detailed manner rarely done, taking into account inputs and outputs of N compounds, for the years 2014 and 2017. The N critical load (NCL) is calculated using the Steady-State Mass Balance (SSMB) method. We compare the NCL to N inputs, to check if the ecosystem N load exceeds the N critical load, and discuss uncertainties in the parameters of the budget. This study uses both published data and new original data from recent field campaigns, collected in Dahra within the framework of the “*Cycle de l’Azote*”

entre la Surface et l'Atmosphère en Afrique (CASAQUE)" program. The objectives of the CASAQUE program were to improve our understanding of the N cycle in Africa, by analyzing N transformation mechanisms in the soil and their link with soil-atmosphere exchanges of reactive N compounds and greenhouse gases.

Due to the low soil N content and the limited soil drainage, we hypothesize the N budget to be imbalanced and N deposition not to exceed the critical loads.

2. Material and methods

2.1 Field site characteristics

Measurements were performed at the Dahra field station, which is part of the Centre de Recherches Zootechniques (CRZ), in the Sahelian region of Ferlo, Senegal (15°24 N, 15° 25 W), and is managed by the Institut Sénégalais de Recherches Agricoles (ISRA). This site is a semi-arid open tree savanna used as grazed grassland. On the site, animal density is high all year round and grazing occurs permanently (Assouma et al., 2017). The livestock is primarily cows, sheep, and goats, with few donkeys and camels. No crops are cultivated at this site previously described in Tagesson et al. (2015b) and Delon et al. (2017, 2019). The climatology of Dahra is described in Laouali et al. (2021).

The Sahel is under the influence of the West African Monsoon (cool wet southwesterly wind) and the Harmattan (hot dry northeasterly wind) depending on the season. The monsoon season extends from mid-July to mid-October and rainfall events are concentrated in the core of the monsoon season in August. Daily rainfall was measured with a manual rain gauge installed at the Dahra site as part of the International Network to study Deposition and Atmospheric composition in Africa (INDAAF, <https://indaaf.obs-mip.fr/>) program. The annual rainfall was 332 mm in 2014 and 395 mm in 2017. From a historical point of view, the interannual variability of rainfall in the Sahel from 1950 until today shows a sharp decrease until 1984 (with devastating droughts in 1972 and 1983-1984), then an increase until 2000, followed by a rather stable period from 2000 until now (Lebel & Ali,

2009, Biasutti et al., 2019). The annual rainfall at Dahra follows the same trend, as reported in Laouali et al. (2021), with 2017 being slightly higher and 2014 slightly lower than the 1959-2017 mean (358 mm). The geographical location of Dahra, together with monthly and daily rainfall for 2014 and 2017 are represented in figure 1; rainfall events that led to efficient soil leaching are highlighted in red (3 in 2014 – 06/08, 17/08, 15/10, and 2 in 2017 – 28/06, 26/08). The mean annual air temperature at 2 m was 28.6 °C in 2014, and 29.0 °C in 2017, with an annual average of 28.8°C calculated over the period 2012-2017.

Tree cover is approximately 3% (Rasmussen et al., 2011) at the Dahra site. The most abundant tree species are *Balanites aegyptiaca* (D.), *Acacia senegal* (L.) and *Acacia tortilis* (F.). The herbaceous vegetation is dominated by annual C4 grasses with species like *Dactyloctenium aegyptium*(L.), *Aristida adscensionis* (L.), *Cenchrus biflorus* (R.), and *Eragrostis tremula* (L.) (Tagesson et al., 2015a). The soil is sandy, with 89% of sands and 6.3% of clays (the rest being silts) in the upper soil profile (0-5cm) and it can be classified as tropical ferruginous soils (Audry et al., 1965). Soil pH ranges from 6.16 to 7.43.

2.2 Budget and critical load calculation (SSMB model)

The Steady State Mass Balance model (SSMB) was used according to Hettelingh et al. (1991). The equation used for France by Party et al. (2001) was adapted to the present study to include, as far as possible, all components of the nutrient N budget specific to the region of study. The nutrient N budget (in kg N ha⁻¹ yr⁻¹) was calculated at the Dahra site according to equations 1, 2 and 3:

$$N_budget = N_inputs - N_outputs \quad (eq. 1)$$

N_budget may be considered as a N surplus if positive, and a N depletion if negative. Usually, N surplus for croplands is calculated as the difference between N inputs and N harvested (Lassaletta et al., 2014). In the present case of a grassland ecosystem, an analogy is made between N outputs and N harvested.

$$N_inputs = DD + WD + BNF + MN \quad (\text{eq. 2})$$

where DD is dry deposition, WD is wet deposition, BNF is Biological N Fixation and MN is animal manure, as defined in section 2.3.1. All terms are in $\text{kg N ha}^{-1} \text{yr}^{-1}$.

N inputs by synthetic fertilizers are not taken into account because synthetic fertilizers are not added to grasslands in Africa (Elrys et al., 2019).

$$N_outputs = F_{N_2O} + F_{NO} + Vol_{NH_3} + DF + BB + N_{leach} + N_{tree_uptake} + N_{animal_ingestion} \quad (\text{eq. 3})$$

where F_{N_2O} is N_2O emissions, F_{NO} is NO biogenic emissions, Vol_{NH_3} is NH_3 volatilization from animal excreta, DF is NO and NH_3 emissions from domestic fires, BB is NO and NH_3 emissions from biomass burning, N_{leach} is N soil leaching, N_{tree_uptake} is N uptake by trees for growth and $N_{animal_ingestion}$ is N ingested by animals via grazing. All terms defined in section 2.3.2, and are measured in $\text{kg N ha}^{-1} \text{yr}^{-1}$.

N use efficiency (NUE) is usually defined for cropland systems as the ratio between N harvested and N inputs (Lassaletta et al., 2014). In the present case of a grassland system,

$$NUE = (N_outputs / N_inputs) \times 100 \quad (\text{eq. 4})$$

The critical load (CL) was calculated following equation 5:

$$CL = F_{N_2O} + F_{NO} + Vol_{NH_3} + DF + BB + N_{leach_crit} + N_{tree_uptake} + N_{animal_ingestion} \quad (\text{eq. 5})$$

where all terms are described in Eq. 3, except N_{leach} replaced by $N_{leach_crit} = [N]_{crit} \times Q$ is the N critical leaching flux in $\text{kg N ha}^{-1} \text{yr}^{-1}$, with $[N]_{crit}$ the critical nutrient N concentration (in kg N L^{-1}) above which the ecosystem in Dahra may encounter a change in biodiversity, and Q the soil drainage (in mm yr^{-1}) defined in section 2.3.2.

$[N]_{crit}$ data are not available in the literature, neither for the Sahelian region, nor for the African continent. Consequently, we set, $[N]_{crit} = 3\text{-}5 \text{ mgN L}^{-1}$, the critical concentration in temperate forest ecosystems, listed by the National Institute for Public Health and the Environment (RIVM, Netherlands).

2.3 Data sources for the budget calculation: equipment, sampling and analytical measures

2.3.1 Inputs

Biological N Fixation (BNF) represents the introduction of new reactive N (Nr) into the system (Bobbink et al., 2010). At the global scale, Vitousek et al. (2013) estimated BNF in terrestrial ecosystems to be 58 Tg N yr⁻¹ (40-100 Tg N yr⁻¹), whereas Galloway et al. (2004) gave a larger estimate of 128 Tg N yr⁻¹. Fowler et al. (2015) highlighted the particularly important role of tropical ecosystems to terrestrial BNF, and Galloway et al. (2004) proposed a regional repartition of BNF, with 25.9 Tg N yr⁻¹ of BNF occurring in Africa. A more specific estimation of BNF in tropical and wet savannas was given by Cleveland et al. (1999), with a range of 16.3 to 44.0 kg N ha⁻¹yr⁻¹, compared to estimations of 9 to 34 kg N ha⁻¹yr⁻¹ in arid shrublands. The most specific estimate for Sahelian rangelands was reported by Penning de Vries and Djiteye (1982) and ranges between 1 and 5 kg N ha⁻¹ yr⁻¹. In the following, these values were used in our N budget calculation.

Wet and dry deposition fluxes were determined thanks to measurements performed by the long term International INDAAF program, part of the International Global Atmospheric Chemistry / DEposition of Biogeochemically Important Trace Species program (IGAC/DEBITS), of the Aerosol, Clouds and TRace gases InfraStructure – France (ACTRIS-FR), and of the World Meteorological Organization/Global Atmospheric Watch (WMO/GAW) program.

Open field rainwater was collected for wet only deposition during the wet seasons in 2014 and 2017, with an automatic precipitation collector specially designed for the INDAAF network (Galy-Lacaux et al., 2009). The methodology is outlined in the Supplementary Material, and the results used in this budget are comprehensively detailed in Laouali et al. (2021).

Dry deposition fluxes of N compounds (NH₃, NO₂ and HNO₃) were calculated using the inferential method with measured concentrations and modeled deposition velocities (resistance analogy, Wesely et al., 1989). Dry deposition fluxes in kg N ha⁻¹ yr⁻¹ for the years 2014 and 2017 were taken

from Laouali et al. (2021). The calculation and measurement procedures are detailed in the Supplementary Material.

The quantity of animal manure was calculated using two methods. The first method was based on the quantity of ingested herb mass by livestock. In this case, following Schlecht et al. (1998) and Hiernaux & Diawara (2014), it was considered that 20% of N contained in the vegetation was retained by the animals for growth and energy, and the remaining 80% is released into the system through faeces and urine deposition. The second method was based on the multiplication of the N released by livestock to the system, in $\text{kg N animal}^{-1} \text{ yr}^{-1}$, adapted from Bouwman & Van der Hoek (1997), Mosier et al. (1998) and Schlecht and Hiernaux (1998), by the animal population in the region of Dahra (Food and Agriculture Organization, FAO, Global Livestock Production and Health Atlas GLiPHA (2009, <http://kids.fao.org/glipha/>). The methodology is fully detailed in Delon et al. (2010). The livestock considered are cows (bovine), sheep (ovine) and goats (caprine). Donkeys and camels were not considered due to the small numbers of individuals found on the site. Details of the calculations for the two methods are given in Table 1.

2.3.2 Outputs

N_2O emissions for 2014 and 2017 were taken from Bigaignon et al. (2020) who simulated N_2O fluxes from 2012 to 2017 at the Dahra site using the Sahelian Transpiration Evaporation Productivity model (STEP, Mougouin et al., 1995; Jarlan et al., 2005; Delon et al., 2015), coupled to GENERAL DECOMPOSITION model (GENDEC, Moorhead et al., 1991) and an N_2O emission model (Li et al., 2000). These simulations were compared to measurements made in Dahra in July and November 2013, at the beginning and at the end of the wet season (Delon et al., 2017), and in September 2017 in the core of the wet season (Bigaignon et al., 2020).

Biogenic NO emissions for 2014 and 2017 were obtained from the same simulation used for the N_2O flux calculation above (unpublished data) using the NOFlux module coupled to STEP-GENDEC

models (as described in Delon et al., 2019). These coupled models were validated thanks to in situ NO flux measurements made at the Dahra site in 2012 and 2013 (Delon et al., 2019).

Ammonia volatilization (Vol_{NH_3}) from animal excreta is the percentage of the N input by organic fertilization (MN term detailed in Inputs section). Thirty per cent (30 %) of the MN input is released to the atmosphere in the form of NH_3 by volatilization, the rest being assimilated in the soil (Bouwman and Van Der Hoek (1997), Bouwman et al. (1997) and Bouwman et al. (2002b)).

NO and NH_3 emissions by domestic fires and biomass burning (DF and BB) were not calculated directly for the region of Dahra. Domestic fire emissions come from wood and charcoal considered as the main fuels for domestic consumption. Estimations for NO and NH_3 emissions from domestic fires and biomass burning at the yearly scale were extracted from Delon et al. (2010) and Galy and Delon (2014), who consider similar semi arid ecosystems in Mali and Niger.

Soil N leaching was measured in situ in Dahra for the first time during the 2014 and 2017 wet seasons. The method used to measure soil N leaching and the equation to calculate the flux are detailed in Supplementary Material (Equation S3).

The reliability of the amount of soil drainage measured within the lysimeter plates was checked by using the STEP model for the years 2014 and 2017. The model calculates the amount of water drained and/or evaporated and/or absorbed by vegetation (detailed in Supplementary Material). The configuration used in this model for the hydrologic module is detailed in the Supplementary Material. Only the drainage from layer 1 to layer 2 was considered as relevant for the comparison with the collected soil solution. The soil solution collected in the lysimeter plates (before absorption, evaporation and drainage to the lower layer) should correspond to the amount of water simulated by the model as evapotranspiration several days after the rain event (but before the following rain event), plus the water drained from layer 1 (0-2 cm) to layer 2 (2 – 30 cm).

N uptake by grass was neglected since grass will decompose at the end of the wet season and bring N back into the soil. N uptake by trees ($N_{\text{tree_uptake}}$) in $\text{kg N ha}^{-1} \text{ yr}^{-1}$ was calculated according to a simple model proposed for the purpose of this study and presented in equation 6:

$$N_{\text{tree_uptake}} = [N]_{\text{trunk}} \times \text{tree_growth} \times D_{\text{trunk}} \quad (\text{eq. 6})$$

Where $[N]_{\text{trunk}} = 0.48\%$ is the N concentration in tree trunks in the Sahel region (Bernhard-Reversat & Poupon, 1980), $\text{tree_growth} = 0.16 - 0.24 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$, is the volumetric tree growth by year in the Sahel region (Clement, 1982), and $D_{\text{trunk}} = 900 \text{ kg m}^{-3}$ is the mean density of trunks.

Each animal has a different ingestion capacity and a different population density in the Dahra region. N ingestion by animals ($N_{\text{animal_ingestion}}$) depends on the ingestion rate $I_g = 0.075 \text{ kgDM kgMW}^{-1} \text{ day}^{-1}$ where MW is the metabolic weight of each animal, calculated from the real weight (RW): $\text{MW} = (\text{RW})^{3/4}$ (Diawara et al., 2020), and DM is dry matter. Calculations are detailed in Table 1.

2.4 Data treatment

Excel software was used for analyzing the data and the simulation results presented in this work.

3. Results and discussion

Overall budget, N depletion and NUE

Figure 5 gives an overall picture of the contribution of each term of the budget (summarized in Table 2). DD fluxes, WD fluxes, N_2O and NO emissions from soils, and leaching, are the only terms with inter-annual variation as they were measured in situ or derived from modeling exercises. All other terms of the budget were mean values extracted from literature and are thus kept constant for the two years of study.

According to low and high N inputs and outputs estimates (Table 2), N depletion ranged from -3.7 to -10.3 $\text{kg N ha}^{-1} \text{ yr}^{-1}$ in 2014 and from -6.5 to -13.1 $\text{kg N ha}^{-1} \text{ yr}^{-1}$ in 2017, with an average of -8.4

$\pm 4.7 \text{ kg N ha}^{-1} \text{ yr}^{-1}$. The reasoning here is proposed with respective sums of inputs and outputs (as reported in Table 2). However, input and output mean values are not significantly different (p -value > 0.2), which highlights the uncertainty in soil depletion estimation. NUE was $130 \pm 1\%$ in 2014, $148 \pm 13\%$ in 2017, and $139 \pm 7\%$ in average for both years. The ecosystem in Dahra suffers from N depletion and confirms the regional character of Sahelian soils and more generally of African soils to be N depleted (Masso et al., 2017b, Elrys et al., 2020). The ratio between outputs and inputs was used as a substitute for N use efficiency in croplands, and showed rates superior to 100% in Dahra, a site considered as representative of the Sahelian region (Assouma et al., 2017, Mougin et al., 2009, Hiernaux et al., 2009). The positive aspect of a very high NUE in grasslands is counterbalanced by the negative aspect of N depleted soils, whereas in West African croplands, NUE is 70% despite low N synthetic fertilizer input (Elrys et al., 2020). If grasslands were to be replaced by croplands, in the context of increasing food demand on the African continent, this would lead to increased use of synthetic fertilizers, which in turn would result in an increase in N emissions and N deposition and a decrease in NUE (Elrys et al., 2020). However, the predominant contribution of livestock completely smoothes out a possible effect of increased fertilizer use. Our study gives an estimation of the N budget in a semi-arid grassland, and is complementary to studies of N budgets in croplands, hence providing critical information required to estimate the impact of changing from grassland to cropland. This study will enable future regional studies to improve N budget estimates in West Africa, if the proportion of grasslands compared to croplands is known.

N atmospheric deposition (DD+WD) reported in this study ($4.4 \pm 0.5 \text{ kg N ha}^{-1} \text{ yr}^{-1}$) is comparable to previous studies in similar semi-arid ecosystems, with DD fluxes ranging from 1.6 to $3.8 \text{ kgN ha}^{-1} \text{ yr}^{-1}$ (Adon et al., 2013), and WD fluxes ranging between 2 and $3.4 \text{ kgN ha}^{-1} \text{ yr}^{-1}$ (Galy and Delon, 2014). N atmospheric deposition in the Dahra grassland is above the value calculated in a Business-As-Usual (BAU) scenario proposed for 2050 on croplands in Western Africa ($3.7 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, Elrys et al., 2020). This suggests that projected N deposition in modelling studies is likely being underestimated, and highlights the importance of the atmospheric deposition term (and its

interannual variation) in N budget estimations. N biogenic emissions from soils have rarely been included in N budget evaluation. Although their contribution remains weak (5% for NO, 1% for N₂O), their evaluation is important for refining the budget quantification. The contribution of anthropogenic emissions in Dahra was also weak (5% for biomass burning and 1% for domestic fires), since the site is located in a remote area far from neighboring villages, and has a low vegetation density.

Livestock contribution and associated uncertainty

In our study, total inputs vary between 11.7 and 34.4 kg N ha⁻¹ yr⁻¹. This is similar to the 35.4 kg N ha⁻¹ yr⁻¹ estimated for West African croplands in 2016 (Elrys et al., 2020). The contribution of N input brought from manure in Dahra (15.6 ± 13.2 kg N ha⁻¹ yr⁻¹) is twice as large as the sum of manure and synthetic fertilizer input in Western African croplands for the same year (3.45 + 5.08 kg N ha⁻¹ yr⁻¹ respectively) estimated in Elrys et al. (2020). Indeed, the main contribution to both N inputs and N outputs in this budget is clearly from livestock (Figure 5), with inputs dominated by manure (68%) and atmospheric deposition (19%), and outputs dominated by N ingestion from grazing (62%) and NH₃ volatilization (15%). Indeed, pastoral activities in Africa have a large impact on ecosystems, as well as being an important economic activity. Livestock products are high value products, and contribute to food security both directly as food source and indirectly through cash income (Herrero et al., 2013). Forty percent of Africa's land area is composed of pastoral rangelands, dominated by semi-arid rangelands (Herrero et al., 2016). Livestock mobility and pastoral ecology are closely connected, and livestock mobility has been recognized as a key adaptation to climate change in tropical dry lands of Africa (Turner and Schlecht, 2019; Audoin et al., 2015). The uncertainty in livestock management is probably the most important factor for the budget calculation in this study. The quantification of livestock in Africa, especially in regions where pastoralism is dominant, is highly uncertain because livestock is associated with people's wealth and is not easily communicated to

national databases, census or surveys. The estimation of livestock populations and their spatial distribution remains therefore uncertain due to the difficulty in scaling up household numbers of livestock (Rufino et al., 2014) to the regional scale. In addition to uncertainty associated to livestock numbers, there is uncertainty on N content in animal excreta. Recent studies have confirmed that synthetic fertilizers are not widely available in SSA (Elrys et al., 2019), enhancing the role of organic fertilizers and increasing the need to quantify the N content in manure. Available N from cattle manure in West Africa was estimated to be 3.45 kg N ha⁻¹ yr⁻¹ for the period 2010-2016 (Elrys et al., 2020) with the highest values found in the Sahel (Rufino et al., 2014), and is likely to increase to 26 kg N ha⁻¹ yr⁻¹ by 2050. Our present estimate of manure input between 6.3 and 25.0 kg N ha⁻¹ yr⁻¹ is similar to future projected values, which highlights the difficulty in evaluating this contribution.

Soil N leaching: influence of hydrological conditions

Despite the small contribution of N leaching to the outputs (8%), this term is the key contributor to the calculation of the NCL. The calculation of the N leaching term requires a thorough analysis of each leaching event, including volumes of collected water samples and measured mineral N concentrations (Table 3). The water volumes collected in bottles A and B resulted from rainfall episodes on the days preceding the leaching event. On August 6, 2014, soil leaching resulted from 27mm and 31mm of rain falling on August 4 and 5 respectively. On August 17 and 18, and October 14, 2014, 51, 12 and 45 mm of precipitation were registered (Fig. 1, Table 4). On June 26 2017, 47mm of precipitation occurred, while 30 mm fell on June 28 late at night, but soil leachate had been collected in the morning. The month of June 2017 represented 22% of the total annual rainfall, with the event on June 26 alone representing 12%. The last soil leaching episode in 2017 resulted from a single precipitation event of 72 mm on August 26. No soil solution was collected in the bottles except for these 3 days in 2014, and 2 days in 2017, meaning that the other precipitation events reported on Fig. 1 were not sufficient to allow effective soil drainage.

The soil N leaching flux was twice as large in 2017 as in 2014, while the amount of water in drainage was more than twice as large in 2014 as in 2017 (107 mm vs 45 mm). More than the annual amount of rainfall, the rain distribution and frequency throughout the wet season, as well as its intensity, may significantly influence the water percolation through the soil. Different cumulated precipitation events led to soil drainage (166 mm in 2014, 119 mm in 2017). Indeed, at the beginning of the wet season, soils are very dry and therefore short and violent convective events with high amounts of precipitation lead to significant runoff rather than water infiltration. Conversely, at the end of the wet season, soil infiltration is favored during precipitation events, occurring on more humid soils. This explains the large soil drainage on August 26, 2017 with 72 mm of rain in one day (and 2.1 L collected in the bottles), compared to June 27 2017, when there was 47mm of rain but only 0.325 L collected. In 2014, all rain events leading to significant soil leaching occurred in the core of the wet season, when soils had already been moistened. The number of leaching events was rather low (3 in 2014, 2 in 2017), and appear robustly linked to the cumulated rain some days before (threshold of minimum 45mm in our study). Rain data collected at the meteorological station in Dahra showed that the 3 leaching events in 2014 occurred when there were violent storms (duration 15 min) followed by squall lines (duration comprised between 1.5h and 2.5h) as reported in Table 4. In 2017, power breaks during storms prevented us from calculating the length of rain events, but we hypothesize that local storms followed by squall lines is the most probable precursor to soil leaching.

The sum of evapotranspiration and drainage to the layer below calculated by STEP was in the same range as the soil water quantity measured in lysimeter plates (Table 4). However, some discrepancies occurred for 2 events (15/10/2014 and 27/6/2017), which may be explained by a possible deviation of the water flow not collected by the lysimeters. These discrepancies may introduce an error in the leaching calculation, which was estimated to be 15% and 13% in 2014 and 2017 respectively (ratio between the amount of water leached in one day and the total annual leached water).

Critical load estimation and uncertainty

Based on data from Table 2 and $N_{\text{leach_crit}}$, the critical load ranged from 17.3 to 48.7 kg N ha⁻¹ yr⁻¹ in 2014 and from 16.2 to 46.4 kg N ha⁻¹ yr⁻¹ in 2017, with an average of 32.1 ± 21.8 kg N ha⁻¹ yr⁻¹. It was calculated considering the first 30 centimeters of the soil, which is considered as the zone to be safeguarded (from an ecological point of view) because this soil depth has a direct impact on herbal mass growth and livestock feeding. Moreover, vegetal biodiversity depending on N contained in this soil depth would be the most sensitive target impacted by N deposition in excess (De Vries et al., 2007; Posch et al., 2011). As mentioned before, the quantification of N leaching is dependent on the right estimation of both drainage and N concentration in soil solution. This is particularly true for Sahelian soils where the rainy season is short, resulting in a need to adapt of the SSMB approach usually used in temperate ecosystems and generally not recommended for sites without any drainage (Posch et al., 1993) (which is the case during the dry season at the Dahra site). However, we have adapted it to a tropical semi-arid climate with a single rainy season lasting over 4 months. Determining *in situ* N critical concentrations necessitates long term experiments. In tropical African ecosystems, as far as we know, long term experiments allowing for the determination of critical concentrations have never been conducted, and critical loads have not previously been calibrated for most of the terms of the steady-state mass balance approach. Nonetheless, whatever the ecosystem, biodiversity changes when N is added to the ecosystem and the greater the N addition to the soil, the larger the negative impact on biodiversity (Midolo et al., 2019). In the present study, NCL (32.1 ± 21.8 kg N ha⁻¹ yr⁻¹) appears far larger than the estimated critical load value of 12.4 kg N ha⁻¹ yr⁻¹ for semi-arid ecosystems calculated by Bouwman et al. (2002a). Discrepancies between the two estimations may be attributed to local scale measurements vs coarse modeling resolution for deposition estimates, and because of the absence of livestock in the Bouwman et al. (2002a) approach.

On average, the results showed that N inputs into the system were almost constant for the two considered years and did not exceed the estimated N critical load. Given the available data and assumed steady-state conditions, our results indicate that Dahra may be considered as an ecosystem still protected from an excess of N input.

Finally, the steady-state mass balance approach presented in this paper identifies the limits of the method and the uncertainties linked to the input data, in terms of residual N input and output to and from the system. This paper also presents important new field-data collected in a region where field work remains a challenge. We emphasize the need to acquire new data to improve N budget and critical load evaluations. In the future, a dynamic modelling approach for critical loads based on biodiversity criteria, as currently used in northern ecosystems (Posch et al., 2011; Rizetto et al. 2016), would allow the prediction of the health status risk for these tropical ecosystems relative to forecasted N deposition trends. However, this would require complementary input data on remote sites and testing to select the most accurate model to apply (Probst et al., 2015).

4. Conclusion

The impact of human activities on N deposition and biodiversity needs to be assessed. In remote areas such as in the Sahel, local experiments on both N budgets and ecological sampling are scarce. The results obtained here provide an extensive estimate of the N budget and N critical load in an open tree savanna, never previously calculated. The study area can be considered as representative of the Sahel pastoral region, both from an ecological and economic point of view, with livestock participating heavily to the ecosystem equilibrium. Uncertainties associated with the estimated N budget result primarily from uncertainty in the quantity of N brought into the ecosystem by animals, its impact on atmospheric chemistry, and the erratic nature of precipitations in the region (in terms of spatio-temporal distribution during the single rainy season), which have a strong impact on the N soil leaching. As a main conclusion of this study, we can infer that the Dahra open tree savanna site is not yet threatened by an excess of N deposition, but that the N budget is imbalanced and favours N depletion. By generalizing these results to other Sahelian sites with common characteristics, we can conclude that damages due to excess N deposition have not yet been observed.

However, the current demographic expansion in Sub-Saharan Africa could change the picture. An increasing population, accompanied by an increase in livestock, is resulting in land use changes from grassland to cropland, an increase in fertilizer use, and an increase in N emissions. Furthermore, the concept of a “safe operating space” is difficult to quantify, as highlighted by Hillebrand et al. (2020) who proposed that threshold transgressions are rarely detectable because the thresholds are difficult to be detected clearly from empirical data. Further, the assessment of a threshold above which N deposition is detrimental to the ecosystem, and the fact that Sahelian ecosystems may still be considered as protected, do not tell us how long until the threshold will be crossed. A more thorough estimation of the impact of livestock on N soil stocks (and moreover their evolution), coupled to an adapted hydrological approach of drainage and leaching in semi arid areas, seem to be the key components to propose a dynamic view of the N budget. Moreover, several years of additional measurements would be needed to assess the interannual variability of soil leaching. These coupled developments would provide the possibility to monitor and forecast the potential N load and the effect of an excess of N on ecosystem biodiversity.

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Figures

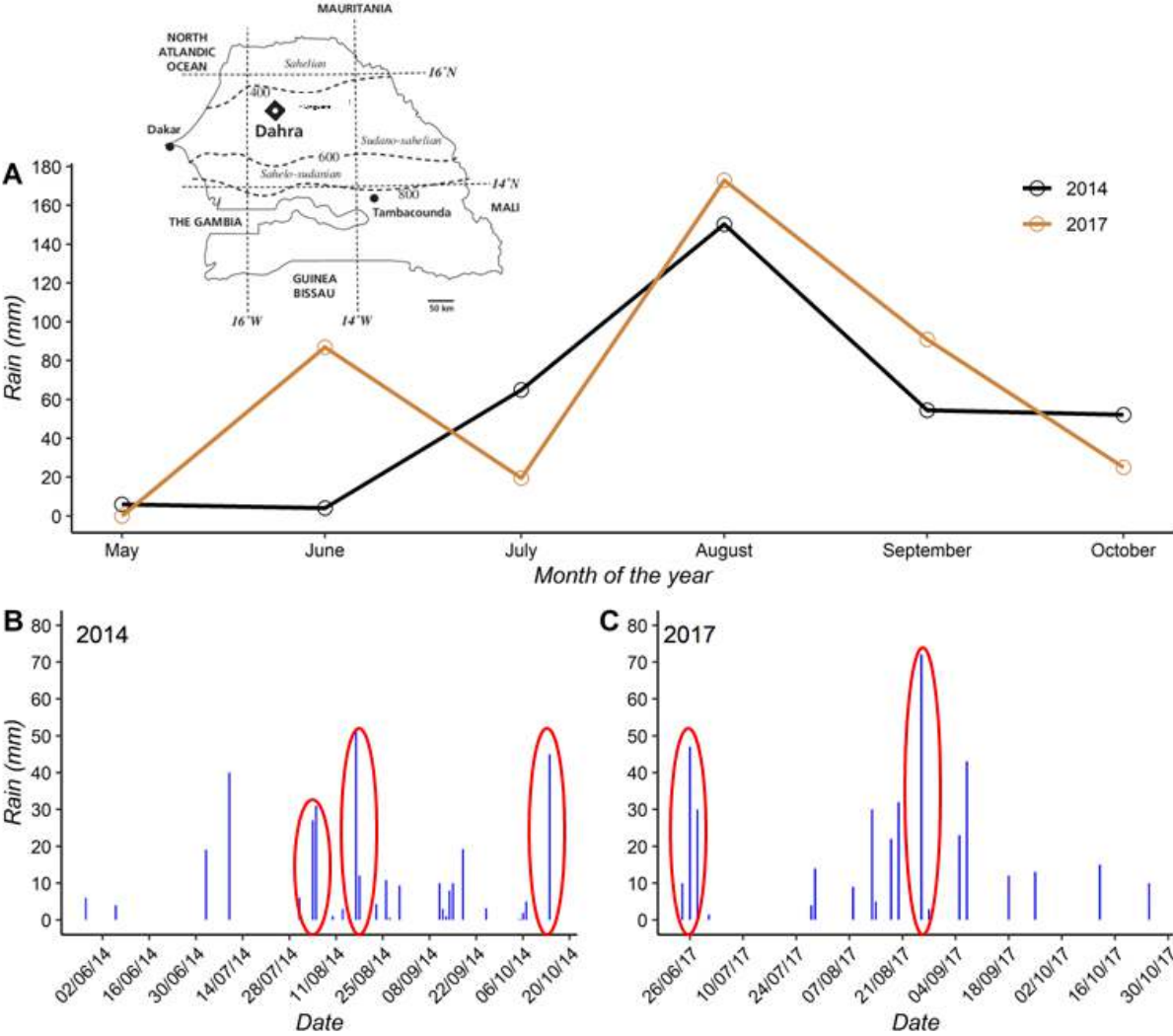


Figure 1: (A) Location of the Dahra site in Senegal and isohyets, adapted from Elberling et al. (2003). Monthly rainfall (in mm) in 2014 and 2017. Rainfall is 0 mm during the other months. (B) Daily rainfall (mm) in 2014, (C) Daily rainfall (mm) in 2017. Red ellipses highlight the rain events that led to effective soil leaching.



Figure 2: (a) Precipitation collector for rainwater chemistry and (b) lysimeter plates connected to bottles for soil N leaching during the installation process and (c) after installation completed.

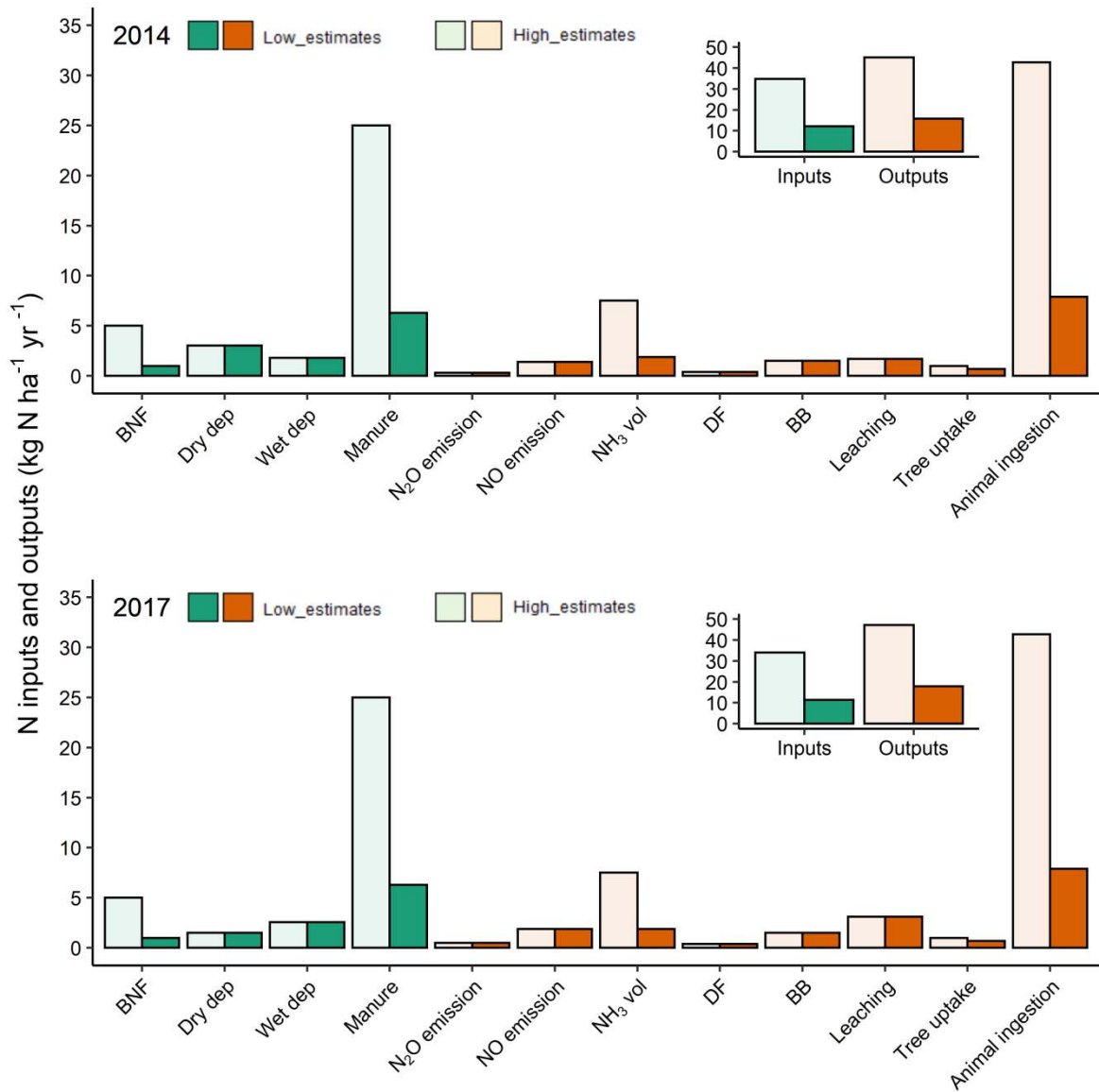


Figure 3: N inputs (in green) and N outputs (in orange) in kg N ha⁻¹ yr⁻¹ at the Dahra site. Bright colors refer to low estimates; pale colors refer to high estimates (see text for details). BNF=Biological Nitrogen Fixation; NH₃ vol=NH₃ volatilization; DF=Domestic Fires; BB=Biomass Burning.

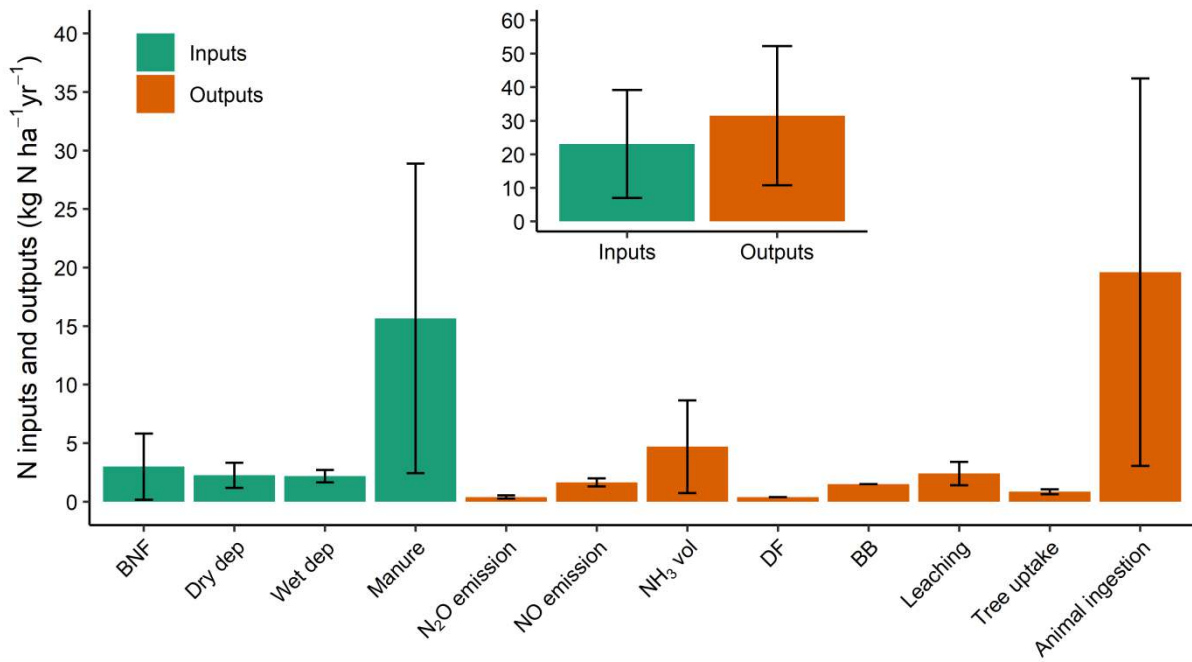


Figure 4: N inputs (in green) and N outputs (in orange) in kg N ha⁻¹ yr⁻¹ at the Dahra site (average 2014-2017). BNF=Biological Nitrogen Fixation; NH₃ vol=NH₃ volatilization; DF=Domestic Fires; BB=Biomass Burning.

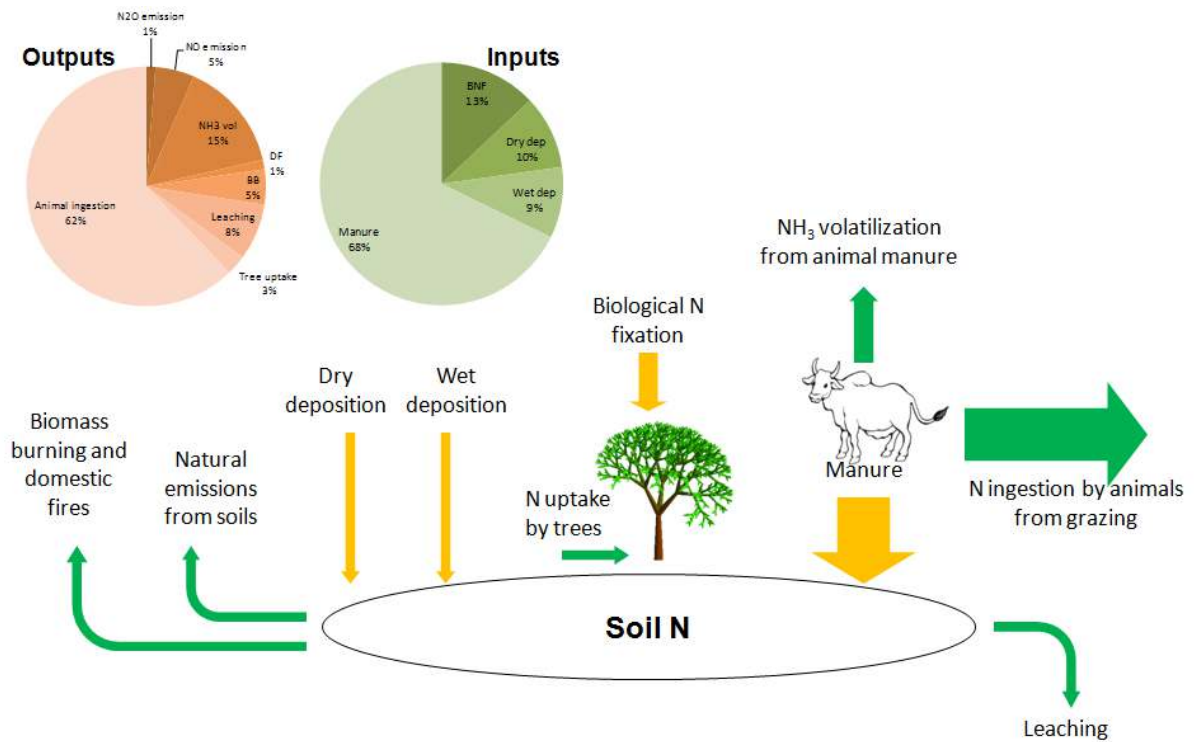


Figure 5: Schematic representation and percentage for each contribution of the N cycle at Dahra. Green color is for inputs, orange color for outputs. Small rows = contribution $\leq 10\%$, medium rows = contribution between 11 and 50%, large rows = contribution $\geq 50\%$.

Table 1: Estimation of N uptake by animals, DM=Dry Matter.

Method 1	Bovine	Ovine	Caprine	Total	Reference
Real weight RW (kg)	250	30	25		
Ingestion per animal (Ig1 in kgDM animal ⁻¹ day ⁻¹)	Ig×(RW) ^{3/4} 4.7	Ig×(RW) ^{3/4} 1.0	Ig×(RW) ^{3/4} 0.8		P. Hiernaux, personal communication
Ingestion per animal (Ig2=Ig1×365 in kgDM animal ⁻¹ yr ⁻¹)	1721	351	306		
Animal density (Da in animal ha ⁻¹)	0.20	0.35	0.20		FAO GLiPHA (2009)
Ingestion per area unit (Ig3=Ig2×Da in kgDM ha ⁻¹ yr ⁻¹)	344	122	61		
N concentration in ingested DM ([N] _{DM} in kg N kgDM ⁻¹)	0.015 – 0.02	0.015 – 0.02	0.015 – 0.02		Penning De Vries & Djitèye, (1982)
Total N ingestion (Ig_{tot1}=[N]_{DM}×Ig3 in kg N ha⁻¹ yr⁻¹)	5.2 – 6.9	1.8 – 2.4	0.9 – 1.2	7.9 – 10.5	
Method 2					
N in animal excreta ([N] _{exc} , in kg N animal ⁻¹ yr ⁻¹)	70	20	20		Schlecht et al., (1998), Mosier et al., (1998), Bouwman & Van der Hoek (1997)
N released to the system via excreta, i.e. 80% of the total ingestion Nreleased=([N] _{exc} ×Da in kg N ha ⁻¹ yr ⁻¹)	14.0	7.0	4.0	25.0	
Total N ingestion (Ig_{tot2}=Nreleased/0.8 in kg N ha⁻¹ yr⁻¹)	17.5	8.8	5.0	31.3	

Table 2: Summary of the different terms of the budget with their value ranges (kg N ha⁻¹ yr⁻¹)

		2014	2017	Range
Inputs	BNF	1 - 5	1 - 5	1 - 5
	DD	3.0 ± 0.1	1.5 ± 0.0	1.5 – 3.0
	WD	1.8 ± 0.2	2.6 ± 0.1	1.8 – 2.6
	MN	6.3 - 25	6.3 - 25	6.3 - 25
	Total inputs	12.1 – 34.8	11.4 – 34.1	11.4 – 34.8
Ouputs	F _{N2O}	0.3	0.5	0.3 - 0.5
	F _{NO}	1.4 ± 1.4	1.9 ± 2.1	1.4 - 1.9
	Vol _{NH3}	1.9 - 7.5	1.9 - 7.5	1.9 - 7.5
	DF	0.4 ± 0.1	0.4 ± 0.1	0.4 ± 0.1
	BB	1.5 ± 0.3	1.5 ± 0.3	1.5 ± 0.3
	N _{animal_ingestion}	7.9 – 31.3	7.9 – 31.3	7.9 – 31.3
	N _{leach}	1.7	3.1	1.7 – 3.1
	N _{leach_crit}	3.2 – 5.3	1.3 – 2.2	1.3 – 5.3
	N _{tree_uptake}	0.7-1.0	0.7-1.0	0.7-1.0
	Total outputs	15.8 – 45.1	17.9 – 47.2	15.8 – 47.2

Table 3: NO₃⁻ and NH₄⁺ concentrations and water volume in collected samples in lysimeters. Bottle A and B correspond to the lysimeters located at 15cm and 30 cm depth in the soil horizon, respectively. The surface of the lysimeters is 540 cm². The concentrations of NO₃⁻ and NH₄⁺ were converted from μg L⁻¹ to μgN L⁻¹ using molar masses of N (14 g mol⁻¹), NO₃⁻ (62 g mol⁻¹) and NH₄⁺ (18 g mol⁻¹). [N_{min}] = [NO₃⁻] + [NH₄⁺].

Date of event	Water volume in bottle A (L)	Water volume in bottle B (L)	[NO ₃ ⁻] in bottle A (μg L ⁻¹)	[NO ₃ ⁻] in bottle B (μg L ⁻¹)	[NH ₄ ⁺] in bottle A (μg L ⁻¹)	[NH ₄ ⁺] in bottle B (μg L ⁻¹)	[N _{min}] in bottle A (kg N ha ⁻¹)	[N _{min}] in bottle B (kg N ha ⁻¹)
06/08/2014	1.50	0.40	60	11262	79	8270	0.021	0.665
18/08/2014	2.48	0.48	3740	1201	717	343	0.644	0.048
15/10/2014	0.35	0.55	6427	4653	982	935	0.144	0.181
Total 2014							0.808	0.894
N_{leach} 2014 (kg N ha⁻¹ yr⁻¹)								1.702
27/06/2017	0.25	0.075	20	6657	3117	249	1.124	0.236
26/08/2017	2.1	0	574	N/A	402	N/A	1.720	N/A
Total 2017							2.844	0.236
N_{leach} 2017 (kg N ha⁻¹ yr⁻¹)								3.080

Table 4: Evapotranspiration and drainage of water from layer 1 (0-2 cm) to layer 2 (2-30cm) simulated by STEP, compared to the leaching measured *in situ*. The rain amount leading to leaching is calculated with 1 or 2 cumulated days of rain (indicated in brackets). The evapotranspiration is calculated several days after the collection (indicated in brackets).

Day of leaching collection	Rain amount leading to leaching, mm (nb of days)	Leaching, mm (measurement)	Evapotranspiration, mm (STEP model) (days after leaching, before next rain)	Drainage layer 1 to layer 2, mm (STEP model)
6/8/2014	58 (2)	35	12 (4)	22
18/8/2014	63 (2)	55	20 (5)	26
15/10/2014	45 (1)	17	12 (4)	26
27/06/2017	47 (1)	6	6 (2)	10
26/08/2017	72 (1)	40	17 (5)	25