Nitrogen budget and critical load determination at a Sahelian grazed grassland site

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Abstract Based on rare and original in situ measurements together with published data, we estimate a complete nitrogen (N) budget for a semi-arid Sahelian grazed grassland located in Dahra (Senegal) in 2014 and 2017. Nitrogen inputs include biological fixation, dry and wet atmospheric deposition, and input from livestock manure. Nitrogen outputs include nitric oxide (NO) and nitrous oxide (N_2O) emissions from soils, NO and ammonia (NH_3) emissions from biomass burning, NH_3 volatilization from manure, ingestion from livestock grazing, uptake by trees and soil leaching. Nitrogen inputs ranged between 11.7 ± 0.5 and 34.4 ± 0.5 kg N ha^{-1} yr^{-1} for low and high estimates respectively, and N outputs ranged between 16.4 ± 1.5 and 45.7 ± 1.5 kg N ha^{-1} yr^{-1} for low and high estimates respectively, on average for both years. Nitrogen depletion was estimated between 4.7 ± 2.0 and 11.3 ± 2.0 kg N ha^{-1} yr^{-1}, which involves N mining from soils. The budget is dominated by the impact of livestock through grazing (63% of the outputs), NH_3 volatilization (15% of the outputs), manure (68% of the inputs) and atmospheric deposition (19% of the inputs). The N critical load (Steady State Mass Balance method) 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 N. The assessment of the critical load in Sahelian landscapes depends heavily on livestock participation to the ecosystem equilibrium.

Keywords Nitrogen budget · Fluxes · Soil leaching · Critical loads · Sahel · Africa

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Introduction

In the Frontiers 2018/2019 report of the United Nations Environment Programme (Sutton et al. 2019),

nitrogen (N) was reported as an "Emerging Issue of Environmental Concern". Humans are substantially altering the global N cycle (Rockström et al. 2009), causing multiple forms of pollution via, for example, fertilizer inputs and biomass burning, making N a key pollutant to tackle at both the local and global scale. Further, 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). Nitrogen 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). N deposition has increased in many natural or semi natural temperate ecosystems (such as grassland, heathland, peatland, forest, and arctic/montane ecosystems), which are considered sensitive and/or vulnerable to N input, since historically N availability has been low (Dise et al. 2011; Johansen et al. 2019; Kosonen et al. 2019).

Sub-Saharan Africa (SSA) is a region with limited N available for plant growth (Sutton et al. 2019), where soil mining or depletion (due to N budget imbalance) and loss of soil fertility are crucial issues (Lassaletta et al. 2014). Optimizing N Use Efficiency (NUE) while minimizing N losses is a key challenge to be faced, especially in SSA (Masso et al. 2017a, b). 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 input from fertilizers to African croplands has increased, leading to a decrease in crop-based N use efficiency and an increase in N losses to the environment (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, N content in soils is very low. In these systems, when N outputs exceed N inputs, N is depleted from the soil and threatens the long-term sustainability of soil fertility (Martinez-Feria et al. 2018). The low soil N content makes these semi-natural ecosystems vulnerable to increases in N deposition, i.e. a small increase in N input can result in a significant change

in contrast to systems with high soil N content (where a small increase in N input will have no effect). Changes in N deposition are likely to influence biodiversity (vegetation species composition and richness). More specifically, atmospheric N deposition is considered as the most important N input to ecosystems in SSA, in comparison to low inputs from N_2 fixation and anthropogenic N supply. Until now, the changes in vegetation diversity and water drainage observed in pastoral areas have most likely resulted from climate variation than from land use changes (Mougin et al. 2009). Meanwhile, the impact of N management and N deposition in semi-natural ecosystems in the Sahel is largely unknown. Nitrogen 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) have been poorly assessed, and ecosystems may be disturbed by an excess of N sooner than initially thought since SSA ecosystems have often been considered unresponsive to N deposition (Bobbink et al. 2010; Bauters et al. 2018).

The N critical load approach has primarily been used to describe the vulnerability of natural ecosystems, or part of a given ecosystem, to the atmospheric deposition of N (De Vries et al. 2007). As it is impossible to get "true" values for each species, when the critical load methodology was initially developed, the choice was to focus on the protection of the first 0.3 m of soil, because this was estimated to include most of the rooting depth, particularly for microbial communities, grass and herbs. Nitrogen 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). Nitrogen critical loads are usually system-dependent and are estimated or calculated using a combination of field observations and dose–response relationship parameterizations. Different approaches have been used to determine critical loads (Posch et al. 2004): empirical approaches (based on expertise), steady-state mass balance models and more recently dynamic integrated soil-vegetation models, which have been adapted or up-scaled in temperate regions (Bobbink and Roelofs 1995; Posch and

Vries de 1999; Belyazid et al. 2011, Probst et al. 2015). Investigations on N critical loads in Africa are scarce, especially for non-cultivated and agro-pastoral ecosystems, as highlighted in Bobbink et al. (2010). In SSA, the estimation of N critical loads has been limited by the availability of input data for the models. Data fluxes concerning the N budget were gathered by Penning de Vries and Djit-eye (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 would increase from 5.9 Tg (10^{12}) N yr^{-1} in 1992 to 6.5 Tg N yr^{-1} in 2015. They also conjectured 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. Further, the important role of livestock in the regulation of soil N in SSA (Powell 2014) has not been taken into account.

In this study, we first calculate the complete soil N budget based on a Steady State Mass Balance approach (SSMB) for a site in a semi-natural eco-system (Dahra, Ferlo region, Senegal), for the years 2014 and 2017 (due to rain collection availability for these two years only). Second, we use the similar mass balance approach (SSMB, adapted from Posch et al. 2004) to assess the NCL, with the same terms as for the N budget, except in the outputs, where the leaching flux was calculated based on a critical con-centration limit. This flux represented the maximum acceptable flux (critical N leaching) without risk for the ecosystem (as stated in the SSMB method). We then calculate the CL exceedance by comparing the NCL to N inputs, to evaluate the risk of the ecosys-tem regarding the N critical load, and discuss the 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 limited soil drainage, we

hypothesize that the N budget is unbalanced with out-puts exceeding inputs and that N deposition does not exceed the critical load.

This study provides new data on detailed N budg-ets and N critical loads. It introduces new field data on N leaching and N atmospheric deposition together with data on livestock grazing for a semi-natural semi-arid pastoral ecosystem in Senegal (Dahra, Ferlo region), where experimental and modeling data are far less available than for cropping systems.

The budget and critical load estimates in this study are calculated at the scale of the field (1 km^2), while taking into account the influence of atmospheric pro-cesses at the regional scale through atmospheric N deposition. Indeed, the regional atmospheric compo-sition (taking into account regional sources of influ-ence, e.g. regional biomass burning) is an important driver of atmospheric N deposition.

Materials and methods

Field site characteristics

Measurements were performed at the Dahra field sta-tion, which is part of the Centre de Recherches Zoot-echniques (CRZ), in the Sahelian region of Ferlo, Senegal (15°24 N, 15° 25 W), and is managed by the Institut Sénégalais de Recherches Agricoles. The climatology of Dahra is described in Laouali et al. (2021).

The Sahel is under the influence of the West Afri-can 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 concen-trated 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 com-position in Africa (INDAAF, https://indaaf.obs-mip.fr/) program. The annual rainfall was 332 mm in 2014 and 395 mm in 2017. The annual rainfall in the Sahel from 1950 until today shows a sharp decrease for the period 1950—1984 (with devastating droughts in 1972 and 1983–1984), then an increase from 1984 to 2000, followed by a rather stable period from 2000 until now (Lebel and 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 Fig. SM2. The mean annual air temperature at 2 m was 28.6 °C in 2014, and 29.0 °C in 2017.

This site is a semi-arid open tree savanna used as grazed grassland. On the site, animal density is high all year round compared to wet savanna ecosystems (75 heads km^{-2} including sheep, goats and cattle, FAO 2009) 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. (2015a) and Delon et al. (2017, 2019). 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. 2015b). The soil is sandy with 89% sand and 6.3% clay (the rest being silt) in the upper soil profile (0–0.05 m) and it can be classified as tropical ferruginous soils (Audry et al. 1965). Soil pH ranges from 6.2 to 7.4.

Budget and critical load calculation

We calculated the soil nutrient budget using a steady sate mass balance approach (SSMB), with the objective of taking into account as many inputs and outputs as possible. The soil nutrient N budget (in kg N ha^{-1} yr^{-1}) was calculated at the Dahra site according to Eqs. 1, 2 and 3. When high and low estimates are available for one of the terms of the budget, it involves high and low estimates of the resulting budget.

$$N\_budget = N\_inputs - N\_outputs \qquad (1)$$

This soil-based N_budget, as defined in Martinez-Feria et al. (2018), may be considered as a N surplus if positive, and as a N depletion if negative.

$$N\_inputs = DD + WD + BNF + MN \qquad (2)$$

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

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

$$N\_outputs = F_{N2O} + F_{NO} + Vol_{NH3} + BB \\ + N_{leach} + N_{tree\_uptake} + N_{animal\_ingestion} \qquad (3)$$

where F_{N2O} is nitrous oxide (N$_2$O) flux, F_{NO} is nitric oxide (NO) biogenic flux, Vol_{NH3} is ammonia (NH$_3$) volatilization from animal excreta, 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 are defined in "Outputs" section, and expressed in kg N ha^{-1} 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, we used the soil-based NUE approach (Congreves et al. 2021; Martinez-Feria et al. 2018):

$$NUE = (N\_outputs/N\_inputs) \times 100 \qquad (4)$$

For Critical Load calculation, we used the SSMB equation developed for France by Party et al. (2001) and adapted from Posch et al. (2004) to include, as far as possible, all components of the nutrient N budget specific to the area of study. The term related to N immobilization in the original NCL equation (Party et al. 2001) was neglected because of the rapid N turnover in Sahelian soils (immobilization is balanced by mineralization, Penning de Vries et al. 1982). The term of denitrification originally considered in Party et al. (2001) was included in the N$_2$O emission term (F_{N2O}). The critical load methodology was initially developed to protect the first top 0.3 m of the soil, where most of the rooting depth especially for microbial communities, grass and herbs is supposed to concentrate. The critical load (CL) was thus calculated following Eq. 5:

$$CL = F_{N2O} + F_{NO} + Vol_{NH3} + BB + N_{leach\_crit} \\ + N_{tree\_uptake} + N_{animal\_ingestion} \qquad (5)$$

where all terms are described in Eq. 3, except $N_{leach\_crit} = [N]_{crit} \times Q$ which is the N critical leaching flux in kg N ha^{-1} 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}, or L m^{-2} yr^{-1}) defined in "Outputs" section.

$[N]_{crit}$ data are not available in the literature, neither for the Sahelian region, nor for the African continent. Consequently, we set $[N]_{crit} = 3–5$ mg N L^{-1}, the critical concentration in temperate forest ecosystems, to avoid changes in vegetation for grass and herbs (recommended in the Critical Load Mapping Manual by Posch et al. 2004). The sensitivity of the SSMB method to the value of the critical concentration was already mentioned by several authors like Reinds et al. (2008). Without any data in the study area, we choose to use the values recommended for soil solutions to protect grass to herbs from any vegetation changes.

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

Inputs

Biological N Fixation (BNF) represents the introduction of new reactive N (Nr) into the system (Bobbink et al. 2010). At the global scale, Galloway et al. (2004) estimated BNF in terrestrial ecosystems to be 128 Tg N yr^{-1} (40–100 Tg N yr^{-1}), whereas Vitousek et al. (2013) updated this value to a lower estimate of 58 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 16.3–44.0 kg N ha^{-1} yr^{-1}, compared to estimates of 9–34 kg N ha^{-1} yr^{-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}. This last range of values was used in our N budget calculation.

Wet and dry deposition fluxes were determined at the Dahra site with 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 (NH_3, NO_2 and HNO_3) 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 N input from 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 and Hiernaux (2004) and Hiernaux and Diawara (2014), it was considered that 20% of N contained in the vegetation is 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 estimate of N released by livestock to the system, in kg N animal^{-1} yr^{-1}, from Bouwman and Van der Hoek (1997), Mosier et al. (1998) and Schlecht and Hiernaux (2004), multiplied 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 number of individuals found on the site. These two methods provide low and high estimates of N input from animal manure. Details of the calculations for the two methods are given in Table 1.

Table 1 Estimation of N uptake by animals, DM = Dry Matter, MW = Metabolic Weight. Ig = 0.075 kg DM kg MW^{-1}

	Bovine	Ovine	Caprine	Total	ReferenceS
Method 1					
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		Hiernaux, personnal 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 and 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 and Hiernaux (2004), Mosier et al. (1998), Bouwman and 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**	

Bold in table indicate total values of the column

Outputs

Nitrous oxide emissions for 2014 and 2017 were taken from Bigaignon et al. (2020) who simulated N$_2$O fluxes from 2012 to 2017 at the Dahra site using the Sahelian Transpiration Evaporation Productivity model (STEP, Mougin et al. 1995; Jarlan et al. 2005; Delon et al. 2015), coupled to GENeral DEComposition model (GENDEC, Moorhead and Reynolds 1991) and an N$_2$O 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$_2$O 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$_{NH3}$) from animal excreta is proportional to the N input by organic fertilization (MN term detailed in Inputs section). In West Africa, despite some uncertainties associated with livestock numbers, we can assume that thirty percent (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 et al. 1997, 2002b).

Nitric oxide and NH$_3$ emissions from biomass burning (BB) were not calculated directly for the region of Dahra. These emissions at the yearly scale were extracted from Delon et al. (2010) and Galy and Delon (2014), who considered similar semi-arid ecosystems in Mali and Niger.

Soil N leaching was measured in situ in Dahra with two Teflon zero-tension lysimeter plates inserted in the soil at two depths (0.15 and 0.3 m, Fig. SM1b). These measurements were specifically conducted during the 2014 and 2017 wet seasons. The water collected in the lysimeter plates was canalized to two inert plastic bottles, and subsequently analyzed at Laero in Toulouse. The method used to measure soil N leaching and the equation to calculate the flux are detailed in Supplementary Material (Equation S3). The zero-tension methodology was chosen, compared to tension lysimeters, because it is more adapted to the purpose of our study in term of mass balance, i.e. collecting the soil water freely leached down the profile rather than the water sorbed to soil particles (Abdulkareem et al. 2015).

The reliability of the amount of soil drainage measured within the lysimeter plates was tested 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 evaporated by the model or transpired by plants (evapotranspiration) several days after the rain event (but before the following rain event), plus the water drained from layer 1 (0–0.02 m) to layer 2 (0.02–0.3 m).

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_{tree_uptake}) in kg N ha^{-1} yr^{-1} was calculated using a simple model proposed for the purpose of this study and presented in Eq. 6:

$$N_{tree\_uptake} = [N]_{trunk} \times tree\_growth \times D_{trunk} \quad (6)$$

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

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

Results

Dry deposition 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 inputs and outputs were mean values extracted from literature and were thus kept constant for the two years of study.

Inputs

Nitrogen dry and wet atmospheric deposition (DD + WD) reported in this study (4.4 ± 0.5 kg N ha^{-1} yr^{-1}, Figs. 1 and 2, Table 2) is comparable to previous studies in similar semi-arid ecosystems, with DD fluxes ranging from 1.6 to 3.8 kgN ha^{-1} yr^{-1} (Adon et al. 2013), and WD fluxes ranging between 2 and 3.4 kg N ha^{-1} yr^{-1} (Galy and Delon 2014). Nitrogen 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 kg N ha^{-1} yr^{-1}, Elrys et al. 2020).

As seen before, biological fixation ranges between 1 and 5 kgN ha^{-1} yr^{-1}.

The contribution of N input from manure in Dahra (Figs. 1 and 2, 15.6 ± 13.2 kg N ha^{-1} yr^{-1}) is almost twice as large as the sum of manure and inorganic fertilizer input in Western African croplands for the same year (3.45 and 5.08 kg N ha^{-1} yr^{-1} respectively) as estimated in Elrys et al. (2020).

Total inputs vary between 11.7 (low estimate, average 2014–2017) and 34.4 kg N ha^{-1} yr^{-1} (high estimate, average 2014–2017) with an average of 23.1 ± 16.0 kg N ha^{-1} yr^{-1}. The highest estimate of the present study is similar to the 35.4 kg N ha^{-1} yr^{-1} estimated for West African croplands in 2016 by
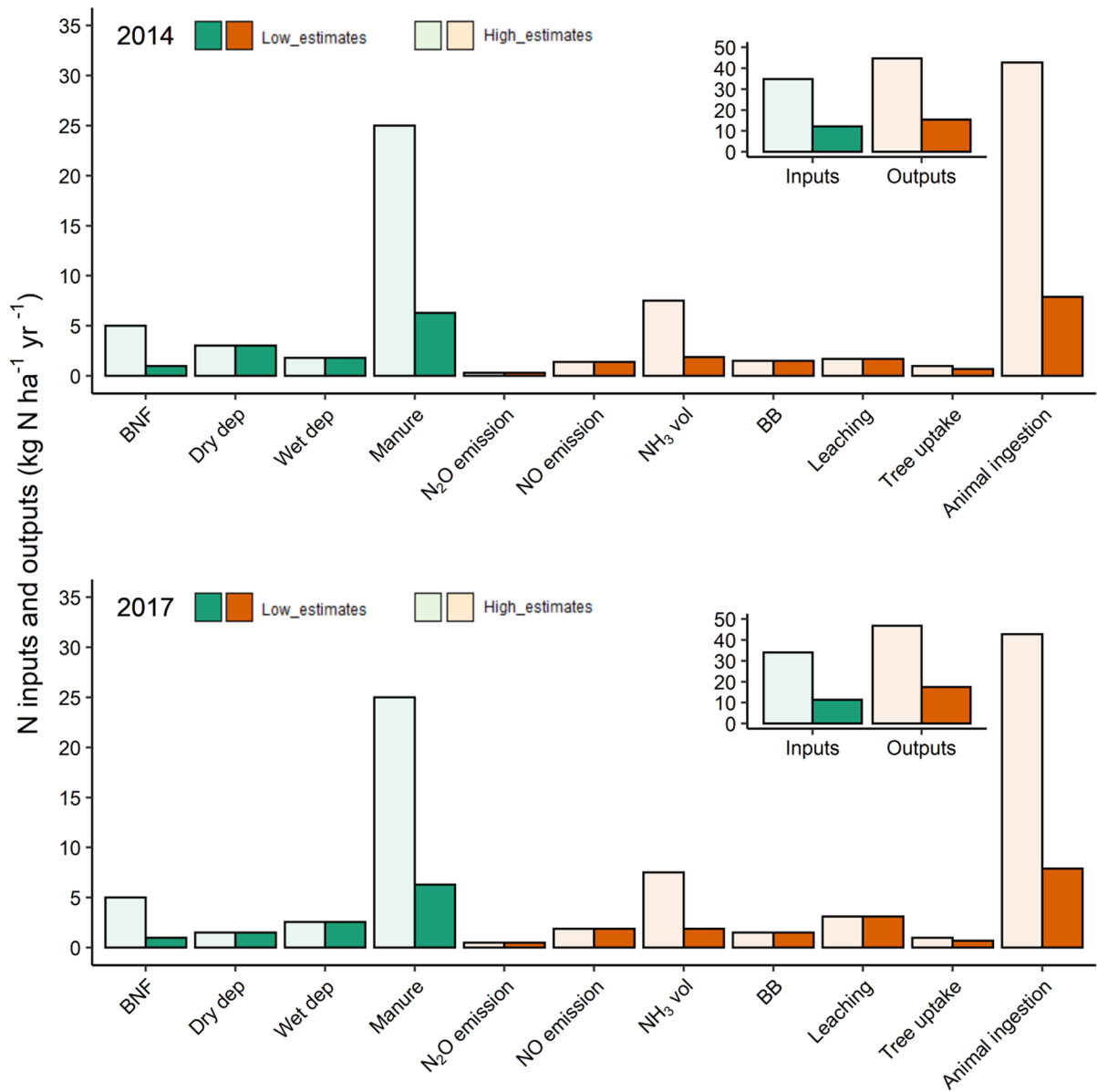
Fig. 1 N inputs (in green) and N outputs (in orange) in kg N ha^{-1} yr^{-1} at the Dahra site. Bright colors refer to low estimates; pale colors refer to high estimates (see text for details).

BNF = Biological Nitrogen Fixation; NH$_3$ vol = NH$_3$ volatilization; BB = Biomass Burning

considering atmospheric deposition, N fixation, animal manure and inorganic fertilizer input (Elrys et al. 2020).

Outputs

N biogenic emissions from soils have rarely been included in N budget evaluations. Although their contribution is minor (1.6 and 0.4 kg N ha^{-1} yr^{-1} representing 5 and 1% of the outputs for NO and N$_2$O respectively), their evaluation is important for refining the budget quantification. The contribution of anthropogenic emissions in Dahra was also weak (1.5 kg N ha^{-1} yr^{-1} for biomass burning,
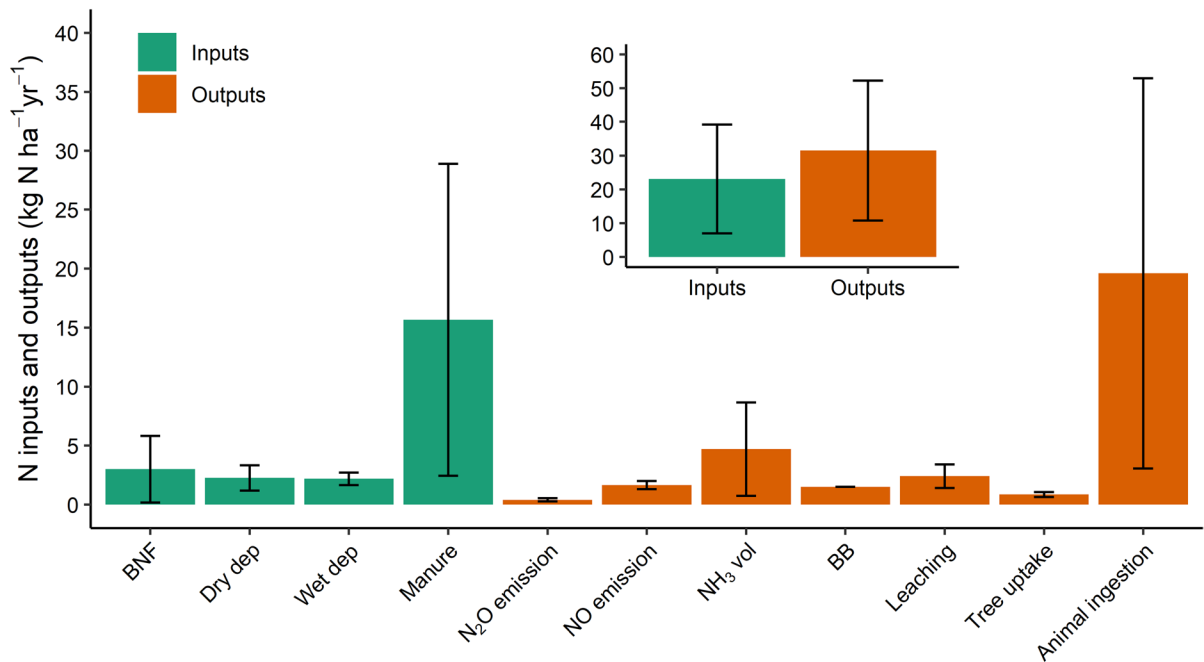
Fig. 2 N inputs (in green) and N outputs (in orange) in kg N ha^{-1} yr^{-1} at the Dahra site (average 2014–2017). BNF = Biological Nitrogen Fixation; NH$_3$ vol = NH$_3$ volatilization; BB = Biomass Burning

representing 5% of the outputs), since the site has a low vegetation density and is located in a remote area far from neighboring villages.

Nitrogen uptake by trees ranges from 0.7 to 1 kg N ha^{-1} yr^{-1} (Figs. 1 and 2, Table 2) and represents 3% of the outputs (Fig. 3).

The contribution of NH$_3$ volatilization from animal manure is 15% of the outputs (1.9–7.5 N ha^{-1} yr^{-1}).

The outputs are dominated by the N ingestion by animals, which ranges from 7.3 to 31.3 kg N ha^{-1} yr^{-1} (19.6 ± 16.5 kg N ha^{-1} yr^{-1} in average, Figs. 1 and 2) and represents 68% of the outputs (Fig. 3).

N leaching ranges from 1.7 to 3.1 kg N ha^{-1} yr^{-1} in 2014 and 2017 respectively (Figs. 1 and 2) and represents 8% of the outputs (Fig. 3). 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 the bottles connected to the lysimeter plates (Fig. SM1b) resulted from rainfall episodes on the days preceding the leaching event (Fig. SM2, Table 4). Due to the Sahelian climate characteristics, soil drainage was limited. Soil solution was only collected in the bottles

for 3 days in 2014, and 2 days in 2017, meaning that the other precipitation events reported on Fig. SM2 were not sufficient to allow effective soil drainage, i.e. all of the water was absorbed in the rooting soil layer.

Total outputs vary between 16.4 (low estimate, average 2014–2017) and 45.7 kg N ha^{-1} yr^{-1} (high estimate, average 2014–2017) with an average of 31.1 ± 20.7 kg N ha^{-1} yr^{-1}.

Overall budget, N depletion and NUE

An overall picture of the contribution of each term of the budget is displayed in Fig. 3 (summarized in Table 2). According to low and high N input and output estimates (Fig. 1, Table 2), N depletion ranged from 3.3 to 9.9 kg N ha^{-1} yr^{-1} in 2014 and from 6.1 to 12.7 kg N ha^{-1} yr^{-1} in 2017, with an average of 8.0 ± 4.7 kg N ha^{-1} yr^{-1} for both years calculated 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 estimated soil depletion. Soil-based NUE was 128 ± 1% in 2014, 146 ± 12% in 2017, and 136 ± 5% on average for both years.

Table 2 Summary of the different terms of the budget with their value ranges (kg N ha^{-1} yr^{-1})

		2014	2017	Range (2014–2017)
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
	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	**15.4–44.7**	**17.5–46.8**	**16.4–45.7**

Bold in table indicate total values of the column

BNF Biological nitrogen fixation; DD Dry deposition, WD Wet deposition; MN Manure; F_{N2O} = N$_2$O flux from soil; F_{NO} NO flux from soil; Vol_{NH3} NH$_3$ volatilization; BB Biomass burning; $N_{animal\_ingestion}$ = N ingested by animals; N_{leach} = N leaching flux; N_{leach_crit} Critical N leaching flux; N_{tree_uptake} N uptake by trees; Low and high estimates result from the two methods used for the calculation of MN, $N_{animal\_ingestion}$ and Vol_{NH3}, and from range of values available for BNF, N_{leach_crit} and N_{tree_uptake}

Nitrogen critical load

Based on data from Table 2 and N_{leach_crit}, the critical load ranged from 16.9 to 48.3 kg N ha^{-1} yr^{-1} in 2014 and from 15.8 to 46.0 kg N ha^{-1} yr^{-1} in 2017, with an average of 31.7 ± 21.8 kg N ha^{-1} yr^{-1}. It was calculated for the first 0.3 m 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 then livestock feeding.

Discussion

Budget imbalance and possible future evolution

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 soil-based NUE, 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. In West African croplands NUE is 70% despite low N inorganic fertilizer input (Elrys et al. 2020). The depletion of soil N releases N to the environment, increases soil degradation and seriously threatens soil fertility and food security (Masso et al. 2017a, b; Elrys et al. 2021). Furthermore, soil degradation associated to N depletion is detrimental to the productivity of grasslands and may prevent them from carrying out key ecosystem functions, such as grazing (Dlamini et al. 2014). In the case of the present study, re-equilibrating the budget would require either reducing N outputs or increasing N inputs. Despite being a very interesting question, proposing recommendations on how to achieve this is far beyond the scope of this study, as it would involve many socio-economic implications on grassland management in SSA. The results of this study may however provide detailed N budget to fill scientific gaps and held land managers and policy makers to make informed decisions.

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 inorganic fertilizers, which in turn would result in an increase in N emissions and N deposition and a decrease in crop NUE (Elrys et al. 2020). Our study gives an estimation of the soil-based N budget in a semi-arid grazed grassland, and is different from (but complementary to) studies of N budgets in croplands. It provides critical information required to estimate the impact of changing from grassland to cropland. This study will also enable future regional studies to improve N budget estimates in West Africa, if the proportion of grasslands compared to croplands is known.

Livestock contribution and associated uncertainty

The main contribution to both N inputs and N outputs in this budget is obviously from livestock (Fig. 3), with inputs dominated by manure (68%) and outputs dominated by N ingestion from grazing (63%) and NH$_3$ volatilization (15%). Indeed, pastoral activities
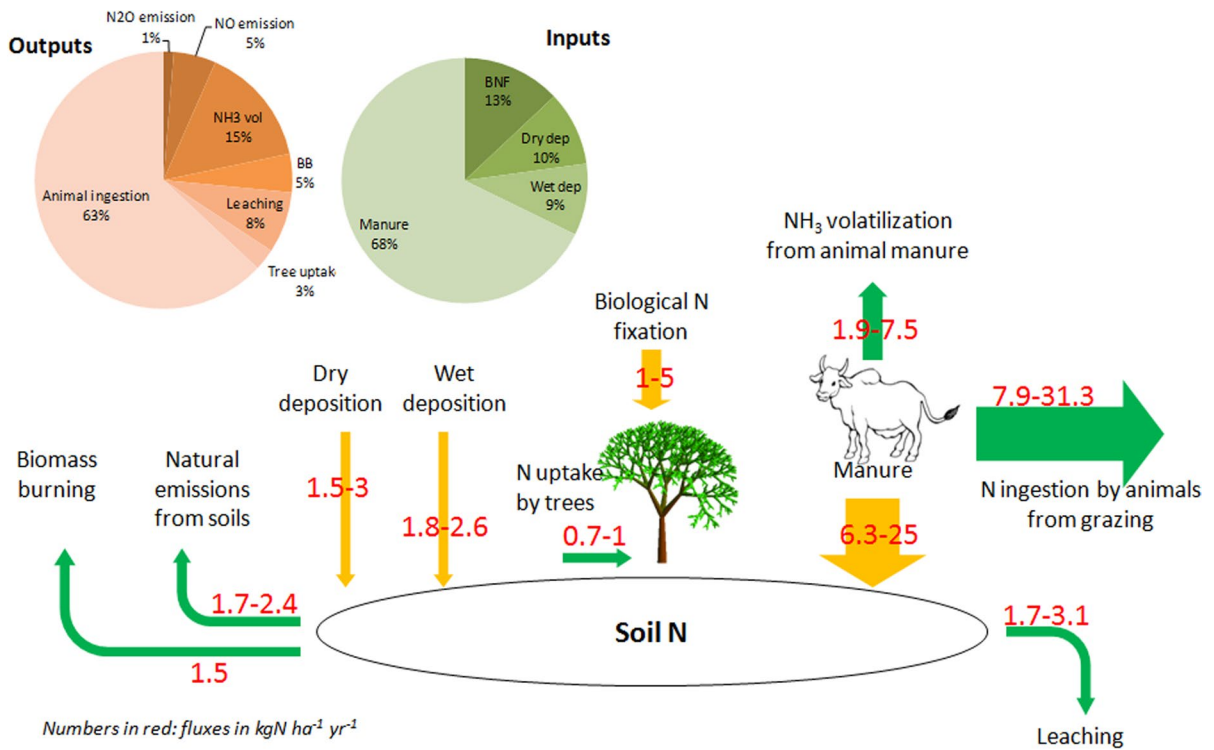
Fig. 3 Schematic representation and % for each contribution of the N cycle at Dahra. Green color is for inputs, orange color for outputs. Small raws = contribution ≤ 10%, medium raws = contribution between 11 and 50%, large raws = contribution ≥ 50%. Red numbers give the range in kg N ha^{-1} yr^{-1} for each input and output

Table 3 NO_3^- and NH_4^+ concentrations and water volume in collected samples in lysimeters

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	11,262	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**	

Bold in table indicate total values of the column

Bottle A corresponds to the lysimeter located at 0.15 m depth and Bottle B corresponds to the lysimeter located at 0.30 m depth in the soil horizon. The surface of the lysimeters is 0.054 m^2. The concentrations of NO_3^- and NH_4^+ were converted from µg L^{-1} to µgN L^{-1} using molar masses of N (14 g mol^{-1}), NO_3^- (62 g mol^{-1}) and NH_4^+ (18 g mol^{-1}). $[N_{min}] = [NO_3^-] + [NH_4^+]$

Table 4 Evapotranspiration and drainage of water from layer 1 (0–0.02 m) to layer 2 (0.02–0.30 m) simulated by STEP, compared to the leaching measured in situ

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

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)

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 number is probably the most important factor for the budget calculation in this study, and we have tried to illustrate this uncertainty by using two different methods of evaluation. 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 the uncertainty associated with livestock numbers, there is an uncertainty on N content in animal excreta. Recent studies have confirmed that inorganic 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 considering diet and fertilizer management. 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 and its range of uncertainty in the N budget.

Soil N leaching: influence of hydrological conditions

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). Different cumulated precipitation events (166 mm in 2014, 119 mm in 2017) led to soil drainage. Indeed, the soils are very sandy, and 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 run-off 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 47 mm 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 appear robustly linked to the cumulated rain some days before (threshold minimum of 45 mm in our study). Rain data collected at the meteorological station in Dahra showed that the 3 soil leaching events in 2014 occurred when there were violent storms (duration 15 min) followed by squall lines (duration between 1.5 and 2.5 h) 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. More than the annual amount of rainfall, the rain distribution and frequency throughout the wet season, as well as its intensity, significantly influences the water percolation through the soil which is a non linear phenomenon with thresholds.

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

The critical load is estimated for the first 0.3 m of the soil because the vegetal biodiversity depending on N contained in this soil depth is expected to be the most sensitive to excess N deposition (De Vries et al. 2007; Posch et al. 2011). As previously mentioned, the quantification of critical N leaching depends on the correct estimation of drainage and critical concentration in soils. The suitability of the nutrient mass balance approach for Sahelian regions is also a key question. This is particularly pertinent for Sahelian soils where the rainy season is short, resulting in a need to adapt the nutrient mass balance approach (based on the SSMB method) 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). This has also been highlighted as a weakness of applying this method for southern Europe. However, we have adapted it to a tropical semi-arid climate with a single rainy season lasting over 4 months, by adapting the equation developed by Posch et al. (2004). For this adaptation, the terms of the initial NCL equation were modified, which may introduce a new uncertainty. The critical load assessment in semi arid systems as the one considered in this study would need more

research on *e.g.* processes in soil N turnover, linked to livestock impact.

The robustness of the critical load assessment with the SSMB approach also depends on a better knowledge of the critical concentration in these ecosystems. As far as we know, long term experiments allowing for the determination of critical concentrations have never been conducted in SSA, and critical loads have not previously been calibrated for most of the terms of the nutrient 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 (16.9–48.3 kg N ha^{-1} yr^{-1} in 2014, 15.8–46.0 kg N ha^{-1} yr^{-1} in 2017) was 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) with a coarse modeling approach. Discrepancies between the two estimates 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. Livestock impact is also a source of uncertainty in NCL determination, and needs further research. Observations by inhabitants of suffering vegetation, knowing the N deposition load, could help in determining empirical NCL and adjusting modeled NCL from the SSMB method, and 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.

On average, the results showed that N inputs into the system were almost constant for the two years considered 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.

However, the current demographic expansion in Sub-Saharan Africa could change the picture. An increasing population, accompanied by an increase in livestock (and shifting from extensive to intensive animal breeding), would result in land use changes from grassland to cropland, in 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). Additionally, determining a threshold above which N deposition is detrimental to the ecosystem does not tell us how long until the threshold will be crossed in Sahelian ecosystems.

Conclusion

The results obtained here provide an extensive estimate of the N budget and N critical load in an open tree savanna, never previously calculated, and based on rare in situ measurements. The limits of the method (based on a steady state analysis) and the uncertainties linked to the input data, in terms of residual N input and output to and from the system, highlight the need to broaden the investigation to other sites with similar ecosystems. The rare in situ measurements available from this study in the Sahel pastoral region, coupled to literature data emphasize the important role of N brought into the ecosystem by animals and its impact on atmospheric chemistry, showing that livestock participates strongly to the ecosystem equilibrium. 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 unbalanced which favors N depletion, even though the intensity of this depletion still remains uncertain. 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.

We emphasize the need to acquire new data to improve N budget and critical load evaluations. A more thorough estimation of the impact of livestock on N soil stocks (and moreover their evolution), coupled to an adapted modeling hydrological approach of drainage and leaching in semi arid areas, are key to developing a dynamic view of the N budget.

In the future, a dynamic modelling approach for critical loads based on biodiversity criteria, as currently used in northern ecosystems (Posch et al. 2011; Rizzetto 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). Developing long term networks and research infrastructures in remote regions in general and in the Sahel in particular is necessary to provide long term data combining multidisciplinary approaches with experts in atmospheric physics and chemistry, biogeochemistry, microbiology and ecology.

The quantification of the N budget and N critical loads is essential to provide information to stakeholders and policy makers about the current state of soils in the Sahelian region. The results of this study could be used to determine the need of N input to Sahelian soils for plant growth, to get a balanced N budget and to reach animal food security in grasslands, leading to food security for humans as a consequence.

The erratic nature of precipitation in the region (in terms of spatio-temporal distribution during the single rainy season) has a strong impact on N soil leaching.

Data availability The datasets on N leaching generated and analysed during the current study are available from the corresponding author on reasonable request. Other data are published data.

Declarations

References

Abdulkareem JH, Abdulkadir A, Abdu N (2015) A review of different types of lysimeter used in solute transport studies. Int J Plant Soil Sci 8(3):1–14

Adon M, Galy-Lacaux C, Delon C, Yoboue V, Solmon F, Kaptue Tchuente AT (2013) Dry deposition of nitrogen compounds (NO2, HNO3, NH3), sulfur dioxide and ozone in west and central African ecosystems using the inferential method. Atmos Chem Phys 13:11351–11374. https://doi.org/10.5194/acp-1311351-2013

Assouma MH, Serça D, Guérin F, Blanfort V, Lecomte P, Touré I, Ickowicz A, Manlay RJ, Bernoux M, Vayssières J (2017) Livestock induces strong spatial heterogeneity of soil CO2, N2O and CH4 emissions within a semi-arid sylvo-pastoral landscape in West Africa. J Arid Land 9:210–221. https://doi.org/10.1007/s40333-017-0001-y

Audoin E, Vayssières J, Odru M, Masse D, Dorégo S, Delaunay V, Lecomte P (2015) Réintroduire l'élevage pour accroître la durabilité des terroirs villageois d'Afrique de l'ouest. Le cas du bassin arachidier au Sénégal. In: Sultan et al "Les sociétés rurales face aux changements climatiques et environnementaux en Afrique de l'ouest, pp 403–427

Audry P, Bonfils P, Charreau C, Dubois J, Fauck R, Faure J, Gavaud M, Maignien R, Maymard J, Peirera-Barreto S, Turenne JF, Vizier JF (1965) http://sphaera.cartographie.ird.fr/images/telechargement/00702.pdf

Bauters M, Drake TW, Verbeeck H, Bodé S, Hervé-Fernández P, Zito P, Podgorski DC, Boyemba F, Makelele I, Cizungu Ntaboba L, Spencer RGM, Boeckx P (2018) High fire-derived nitrogen deposition on central African forests. PNAS 115(3):549–554. https://doi.org/10.1073/pnas.1714597115

Belyazid S, Kurz D, Braun S, Sverdrup H, Rihm B, Hettelingh JP (2011) A dynamic modelling approach for estimating critical loads of nitrogenbased on plant community changes under a changing climate. Environ Poll 159:789–801. https://doi.org/10.1016/j.envpol.2010.11.005

Bernhard-Reversat F, Poupon H (1980) Nitrogen cycling in a soil-tree system in a Sahelian savanna. Example of Acacia senegal. Nitrogen Cycl West Afr Ecosyst 2:363–369. https://doi.org/10.4161/oxim.2.4.8878

Biasutti M (2019) Rainfall trends in the African Sahel: characteristics, processes, and causes. Wires Clim Change 10:e591. https://doi.org/10.1002/wcc.591

Bigaignon L, Delon C, Ndiaye O, Galy-Lacaux C, Serça D, Guérin F, Tallec T, Merbold L, Tagesson T, Fensholt R, André S, Galliau S (2020) Understanding N2O emissions in African ecosystems: assessments from a semi-arid savanna grassland in Senegal and sub-tropical agricultural fields in Kenya. Sustainability 12:8875. https://doi.org/10.3390/su12218875

Bobbink R, Hicks K, Galloway J, Spranger T, Alkemade R, Ashmore M, Bustamante M, Cinderby S, Davidson E, Dentener F, Emmett B, Erisman J-W, Fenn M, Gilliam F, Nordin A, Pardo L, De Vries W (2010) Global assessment of nitrogen deposition effects on terrestrial plant diversity: a synthesis. Ecol Appl 20:30–59. https://doi.org/10.1890/08-1140.1

Bobbink R, Roelofs JGM (1995) Nitrogen critical loads for natural and semi natural ecosystems: the empirical approach. Water Air Soil Poll 85:2413–2418

Bouwman AF, Van Der Hoek KW (1997) Scenarios of animal waste production and fertilizer use and associated ammonia emission for the developping countries. Atmos Environ 31:4095–4102. https://doi.org/10.1016/S1352-2310(97)00288-4

Bouwman AF, Lee DS, Asman WAH, Dentener FJ, Van Der Hoek KW, Olivier JGJ (1997) A global high-resolution emission inventory for ammonia. Global Biogeochem Cycl 11:561–587. https://doi.org/10.1029/97GB02266

Bouwman AF, Van Vuuren DP, Derwent RG, Posch M (2002a) A global analysis of acidification and eutrophication of Terrestrial Ecosystems. Water Air Soil Pollut 141:349–382. https://doi.org/10.1023/A:1021398008726

Bouwman AF, Boumans LJM, Batjes NH (2002b) Modeling global annual N2O and NO emissions from fertilized fields. Global Biogeochem Cycles 16(4):1080. https://doi.org/10.1029/2001GB001812

Breman H and de Ridder N (1991) Manuel sur les pâturages des pays sahéliens, Paris; Wageningen (The Netherlands): Karthala; CTA; CCTA, p 485. https://edepot.wur.nl/136871

Chapin FS III, Zavaleta ES, Eviner VT, Naylor RL, Vitousek PM, Reynolds HL, Hooper DU, Lavorel S, Sala OE, Hobbie SE, Mack MC, Díaz S (2000) Consequences of changing biodiversity. Nature. https://doi.org/10.1038/35012241

Clement J (1982) Estimation des volumes et de la productivité des formations mixtes forestières et graminéennes tropicales, (Estimation of volumes and productivity of heterogeneous formations of tropical woodlands and graminaceae). Bois Et Forêts Des Tropiques. https://doi.org/10.19182/bft1982.198.a19428

Cleveland CC, Townsend AR, Schimel DS, Fisher H, Howarth RW, Hedin LO, Perakis SS, Latty EF, Von Fischer JC, Elseroad A, Wasson MF (1999) Global patterns of terrestrial biological nitrogen (N2) fixation in natural ecosystems. Glob Biogeochem Cycles 13:623–645. https://doi.org/10.1029/1999GB900014

Congreves KA, Otchere O, Ferland D, Farzadfar S, Williams S, Arcand MM (2021) Nitrogen use efficiency definitions of today and tomorrow. Front Plant Sci 12:637108. https://doi.org/10.3389/fpls.2021.637108

De Vries W, Wamelink W, Van Dobben H, Kros H, Reinds GJ, Mol-Dijkstra J, Smart S, Evans C, Rowe E, Belyazid S, Sverdrup H, Van Hinsberg A, Posch M, Hettelingh J-P, Spranger T, Bobbink R (2010) Use of dynamic soil–vegetation models to assess impacts of nitrogen deposition on plant species composition: an overview. Ecol Appl 20(1):60–79. https://doi.org/10.1890/08-1019.1

De Vries W, Kros H, Reinds GJ, Wamelink W, Mol J, van Dobben H, Bobbink R, Emmett B, Smart S, Evans C, Schlutow A, Kraft P, Belyazid S, Sverdrup H, van Hinsberg A, Posch M, Hettelingh JP (2007) Developments in deriving critical limits and modelling critical loads of nitrogen for terrestrial ecosystems in Europe. Report 1382, p 206.

Alterra Wageningen UR, Wageningen, The Netherlands. http://nora.nerc.ac.uk/id/eprint/1779/

Delon C, Galy-Lacaux C, Boone A, Liousse C, Serça D, Adon M, Diop B, Akpo A, Lavenu F, Mougin E, Timouk F (2010) Atmospheric nitrogen budget in Sahelian dry savannas. Atmos Chem Phys 10:2691–2708. https://doi.org/10.5194/acp-10-2691-2010

Delon C, Mougin E, Serça D, Grippa M, Hiernaux P, Diawara M, Galy-Lacaux C, Kergoat L (2015) Modelling the effect of soil moisture and organic matter degradation on biogenic NO emissions from soils in Sahel rangeland (Mali). Biogeosciences 12:3253–3272. https://doi.org/10.5194/bg-12-3253-2015

Delon C, Galy-Lacaux C, Serça D, Loubet B, Camara N, Gardrat E, Saneh I, Fensholt R, Tagesson T, Le Dantec V, Sambou B, Diop C, Mougin E (2017) Soil and vegetation-atmosphere exchange of NO, NH3, and N2O from field measurements in a semi arid grazed ecosystem in Senegal. Atmos Environ 156:36–51. https://doi.org/10.1016/j.atmosenv.2017.02.024

Delon C, Galy-Lacaux C, Serça D, Personne E, Mougin E, Adon M, Le Dantec V, Loubet B, Fensholt R, Tagesson T (2019) Modelling land–atmosphere daily exchanges of NO, NH3, and CO2 in a semi-arid grazed ecosystem in Senegal. Biogeosciences 16:2049–2077. https://doi.org/10.5194/bg-16-2049-2019

Diawara MO, Hiernaux P, Mougin E, Sissoko S, Kergoat L, Diakite HS, Diallo A, Soumaguel N (2020) Joint monitoring of livestock stocking rates and rangeland vegetation of Hombori district in Mali during the 2010–2011 dry season. Int J Biol Chem Sci 14(6):1941–1956

Dise NB, Ashmore M, Belyazid S, Bleeker A, Bobbink R, de Vries W, Erisman JW, Spranger T, Stevens CJ, van den Berg L (2011) Nitrogen as a threat to the biodiversity. In: Sutton MA, Howard CM, Erisman JW, Billen G, Bleeker A, Grennfelt P, Van Grinsven H, Grizzetti B (eds) The European nitrogen assessment. Cambridge University Press. https://doi.org/10.1017/CBO9780511976988.023

Dlamini P, Chivenge P, Manson A, Chaplot V (2014) Land degradation impact on soil organic carbon and nitrogen stocks of sub-tropical humid grasslands in South Africa. Geoderma 235–236:372–381. https://doi.org/10.1016/j.geoderma.2014.07.016

Downing RJ, Hettelingh JP, de Smet PAM (1993) Calculation and mapping of critical loads in Europe: Status Report 1993. RIVM Report No. 259101003, CCE, Bilthoven, The Netherlands

Elrys AS, Abdel-Fattah MK, Raza S, Chen Z, Zhou J (2019) Spatial trends in the nitrogen budget of the African agro-food system over the past five decades. Environ Res Lett 14:124091

Elrys AS, Metwally MS, Raza S, Alnaimy MA, Shaheen SM, Chen Z, Zhou J (2020) How much nitrogen does Africa need to feed itself by 2050? J Environ Manag 268:110488. https://doi.org/10.1016/j.jenvman.2020.110488

Elrys AS, Desoky ESM, Alnaimy MA, Zhang H, Zhang JB, Cai ZC, Cheng Y (2021) The food nitrogen footprint for African countries under fertilized and unfertilized farms. J Environ Manag 279:111599

FAO (2009) Food and Agriculture Organization, FAO, Global Livestock Production and Health Atlas GLiPHA. http://kids.fao.org/glipha/

Fowler D, Steadman CE, Stevenson D, Coyle M, Rees RM, Skiba UM, Sutton MA, Cape JN, Dore AJ, Vieno M, Simpson D, Zaehle S, Stocker BD, Rinaldi M, Facchini MC, Flechard CR, Nemitz E, Twigg M, Erisman JW, Butterbach-Bahl K, Galloway JN (2015) Effects of global change during the 21st century on the nitrogen cycle. Atmos Chem Phys 15:13849–13893. https://doi.org/10.5194/acp-15-13849-2015

Galloway JN, Dentener FJ, Capone DG, Boyer EW, Howarth RW, Seitzinger SP, Asner GP, Cleveland CC, Green PA, Holland EA, Karl DM, Michaels AF, Porter JH, Townsend AR, Vörösmarty CJ (2004) Nitrogen cycles: past, present, and future. Biogeochemistry 70(2):153–226

Galy-Lacaux C, Delon C (2014) Nitrogen emission and deposition budget in West and Central Africa. Environ Res Lett 9:125002. https://doi.org/10.1088/1748-9326/9/12/125002

Galy-Lacaux C, Laouali D, Descroix L, Gobron N, Liousse C (2009) Long term precipitation chemistry and wet deposition in a remote dry savanna site in Africa (Niger). Atmos Chem Phys 9:1579–1595. https://doi.org/10.5194/acp-9-1579-2009

Gaudio N, Belyazid S, Gendre X, Mansat A, Rizetto S, Nicolas M, Sverdrup H, Probst A (2015) Combined effect of atmospheric nitrogen deposition and climate change on temperate forest soil biogeochemistry: a modelling approach. Ecol Model 306:24–34. https://doi.org/10.1016/j.ecolmodel.2014.10.002

Herrero M, Grace D, Njuki J, Johnson N, Enahoro D, Silvestri S, Rufino MC (2013) The roles of livestock in developing countries. Animal 7(s1):3–18. https://doi.org/10.1017/S1751731112001954

Herrero M, Addison J, Bedelian C, Carabine E, Havlík P, Henderson B, van de Steeg J, Thornton PK (2016) Climate change and pastoralism: impacts, consequences and adaptation. Rev Sci Tech off Int Epiz 35(2):417–433. https://doi.org/10.20506/rst.35.2.2533

Hiernaux P, Diawara M (2014) Livestock: recyclers that promote the sustainability of smallholder farms. Rural 21(04):9–11

Hiernaux P, Ayantunde A, Kalilou A, Mougin E, Gérard B, Baup F, Grippa M, Djaby B (2009) Trends in productivity of crops, fallow and rangelands in Southwest Niger: impact of land use, management and variable rainfall. J Hydrol 375:65–77. https://doi.org/10.1016/j.jhydrol.2009.01.032

Hillebrand H, Donohue I, Harpole WS, Hodapp D, Kucera M, Lewandowska AM, Merder J, Montoya JM, Freund JA (2020) Thresholds for ecological responses to global change do not emerge from empirical data. Nat Ecol Evol. https://doi.org/10.1038/s41559-020-1256-9

Jarlan L, Mougin E, Mazzega P, Schoenauer M, Tracol Y, Hiernaux P (2005) Using coarse remote sensing radar observations to control the trajectory of a simple sahelian land surface model. Remote Sens Environ 94(2):269–285. https://doi.org/10.1016/j.rse.2004.10.005

Johansen AM, Duncan C, Reddy A, Swain N, Sorey M, Nieber A, Agren J, Lenington M, Bolstad D, Samora B, Lofgren R (2019) Precipitation chemistry and deposition at a

high-elevation site in the Pacific Northwest United States (1989–2015). Atmos Environ 212:221–230. https://doi.org/10.1016/j.atmosenv.2019.05.021

Kosonen Z, Schnyder E, Hiltbrunner E, Thimonier A, Schmitt M, Seitler E, Thöni L (2019) Current atmospheric nitrogen deposition still exceeds critical loads for sensitive, semi-natural ecosystems in Switzerland. Atmos Environ 211:214–225. https://doi.org/10.1016/j.atmosenv.2019.05.005

Laouali D, Delon C, Adon M, Ndiaye O, Saneh I, Gardrat E, Dias-Alves M, Tagesson T, Fensholt R, Galy-Lacaux C (2021) Source contributions in precipitation chemistry and analysis of atmospheric nitrogen deposition in a Sahelian dry savanna site in West Africa. Atmos Res 251:105423. https://doi.org/10.1016/j.atmosres.2020.105423

Lassaletta L, Billen G, Grizzetti B, Anglade J, Garnier J (2014) 50 year trends in nitrogen use efficiency of world cropping systems: the relationship between yield and nitrogen input to cropland Environ. Res Lett 9:105011

Lebel T, Ali A (2009) Recent trends in the Central and Western Sahel rainfall regime (1990–2007). J Hydrol 375:52–64. https://doi.org/10.1016/j.jhydrol.2008.11.030

Li C, Aber J, Stange F, Butterbach-Bahl K, Papen H (2000) A process-oriented model of N_2O and NO emissions from forest soils: 1. Model development. J Geophys Res 105:4369–4384

Martinez-Feria RA, Castellano MJ, Dietzel RN, Helmers MJ, Liebman M, Huber I et al (2018) Linking crop- and soil-based approaches to evaluate system nitrogen-use efficiency and tradeoffs. Agric Ecosyst Environ 256:131–143. https://doi.org/10.1016/j.agee.2018.01.002

Masso C, Baijukya F, Ebanyat P, Bouaziz S, Wendt J, Bekunda M, Vanlauwe B (2017a) (2017a) Dilemma of nitrogen management for future food security in Sub-Saharan Africa? A Review. Soil Res 55:425–434. https://doi.org/10.1071/SR16332

Masso C, Nziguheba G, Mutegi J, Galy-Lacaux C, Wendt J, Butterbach-Bahl K, Wairegi L, and Datta A (2017b) Soil fertility management in Sub Saharan Africa. In: Licht-fouse E (ed) Sustainable agriculture reviews. Springer International Publishing. https://doi.org/10.1007/978-3-319-58679-3_7

Midolo G, Alkemade R, Schipper AM, Benítez-López A, Perring MP, De Vries W (2019) Impacts of nitrogen addition on plant species richness and abundance: a global meta-analysis. Global Ecol Biogeogr 28:398–413. https://doi.org/10.1111/geb.12856

Moorhead DL, Reynolds JF (1991) A general model of litter decomposition in the northern Chihuahuan desert. Ecol Modell 56:197–219

Mosier A, Kroeze C, Nevison C, Oenema O, Seitzinger S, van Cleemput O (1998) Closing the global N2 O budget: nitrous oxide emissions through the agricultural nitrogen cycle. Nutr Cycl Agroecosyst 52:225–248. https://doi.org/10.1023/A:1009740530221

Mougin E, Lo Seen D, Rambal S, Gaston A, Hiernaux P (1995) A regional Sahelian grassland model to be coupled with multispectral satellite data. I. Description and Validation. Remote Sens Environ 52:181–193. https://doi.org/10.1016/0034-4257(94)00126-8

Mougin E, Hiernaux P, Kergoat L, Grippa M, DeRosnay P, Timouk F, LeDantec V, Demarez V, Arjounin M, Lavenu F, Soumaguel N, Ceschia E, Mougenot B, Baup F, Frappart F, Frison PL, Gardelle J, Gruhier C, Jarlan L, Mangiarotti S, Sanou B, Tracol Y, Guichard F, Trichon V, Diarra L, Soumaré A, Koité M, Dembélé F, Lloyd C, Hanan NP, Damesin C, Delon C, Serça D, Galy-Lacaux C, Seghiéri J, Becerra S, Dia H, Gangneron F, Mazzega P (2009) The AMMA-CATCH Gourma observatory site in Mali: relating climatic variations to changes in vegetation, surface hydrology, fluxes and natural resources. J Hydrol. https://doi.org/10.1016/j.Jhydrol.2009.06.045

Nilsson J, Grennfelt P (1988) Critical loads for sulphur and nitrogen. Report from Skokloster Workshop. Skokloster. https://doi.org/10.1007/978-94-009-4003-1_11

Pardo LH, Fenn NE, Goodale CN, Geiser LH, Driscoll CT, Allen EB, Baron JS, Bobbink R, Bowman WD, Clark CM, Emmett B, Gilliam FS, Greaver TL, Hall SJ, Lilleskov EA, Liu L, Lynch JA, Nadelhoffer KJ, Perakis SS, Robin-Abbott MJ, Stoddard JL, Weathers KC, Dennis RL (2011) Effects of nitrogen deposition and empirical nitrogen critical loads for ecoregions of the United States. Ecol Appl 21(8):3049–3082

Party JP, Probst A, Thomas AL, Dambrine E (2001) Calcul et cartographie des charges critiques azotées en France: application de la méthode empirique. Pollut Atmos 172:531–544

Penning de Vries FWT and Djiteye MA (eds) (1982) La productivité des pâturages sahéliens, une étude des sols, des végétations et de l'exploitation de cette ressource naturelle. (The productivity of Sahelian rangelands, a study of soils, vegetations, and exploitation of this natural resource.) Agric. Res. Rep. (Versl. Landbouw. Onderz.) 918, p 525. ISBN 90 220 0806 1

Posch M, de Vries W (1999) Derivation of critical loads by steady-state and dynamic soil models. In: Langan SJ (ed) The impact of nitrogen deposition on natural and semi-natural ecosystems. Dordrecht, The Netherlands, Kluwer, pp 213–234

Posch M, Hettelingh JP, Sverdrup HU, Bull K, de Vries W (1993) Guidelines for the computation and mapping of critical loads and exceedances of sulphur and nitrogen in Europe. In: Downing RJ, Hettelingh JP, De Smet PAM (eds) Calculation and mapping of critical loads in Europe. Status Report 1993. Bilthoven, The Netherlands, Coordination Centre for effects, pp 21–37

Posch M, Aherne J, and Hettelingh JP (2011) Nitrogen critical loads using biodiversity-related critical limits. Environ Pollut 159:2223–2227

Posch M et al. (2004) Mapping Critical Loads, Chap. V, V - 10 - V - 28, In: Spranger T, Lorenz U, Gregor HD (eds) Manual on methodologies and criteria for modelling and mapping critical loads and levels and air pollution effects, risks and trends, 52/02, p 266. ISBN: 0722–186X

Powell JM (2014) Feed and manure use in low-N-input and high-N-input dairy cattle production systems. Environ Res Lett 9:115004. https://doi.org/10.1088/1748-9326/9/11/115004

Probst A, Obeidy C, Gaudio N, Belyazid S, Gégout JC, Alard D, Corcket E, Party JP, Gauquelin T, Mansat A, Nihlgard B, Leguédois S, Sverdrup HU (2015) Evaluation of

plant-responses to atmospheric nitrogen deposition in France using integrated soil-vegetation models. In: de Vries W, Hettelingh JP, Posch M (eds) Critical loads and dynamic risk assessments: nitrogen, acidity and metals in terrestrial and aquatic ecosystems. Springer, Dordrecht, The Netherlands, pp 359–379

Probst A and Leguedois S (2008) National Focal Centre report: France. In: Hettelingh JP, Posch M, Slootweg (eds.) Critical loads, dynamic modelling and impact assessment in Europe of S and N and dynamic modelling. CCE Status report 2008, report No 500090003/2008, p 24. ISBN No. 978-90-6960-211-0, 133-140

Rasmussen MO, Göttsche FM, Diop D, Mbow C, Olesen FS, Fensholt R, Sandholt I (2011) Tree survey andallometric models for tiger bush in northern Senegal and comparison with tree parameters derived from highresolution satellite data. Int J Appl Earth Obs Geoinf 13:517–527. https://doi.org/10.1016/j.jag.2011.01.007

Reinds GJ, Posch M, De Vries W, Slootweg J, Hettelingh JP (2008) Critical loads of sulphur and nitrogen for terrestrial ecosystems in Europe and Northern Asia using different soil chemical criteria. Water Air Soil Pollut 193(1):269–287

Rizzetto S, Belyazid S, Gégout JC, Nicolas M, Alard D, Corcket E, Gaudio N, Sverdrup H, Probst A (2016) Modelling the impact of climate change and atmospheric N deposition on French forests biodiversity. Environ Pollut 213:1016–1027. https://doi.org/10.1016/j.envpol.2015.12.048

Rockström J, Steffen W, Noone K, Persson A, Stuart III Chapin F, Lambin E, Lenton TM, Scheffer M, Folke C, Schellnhuber HJ, Nykvist B, de Wit CA, Hughes T, van der Leeuw S, Rodhe H, Sörlin S, Snyder PK, Costanza R, Svedin U, Falkenmark M, Karlberg L, Corell RW, Fabry VJ, Hansen J, Walker B, Liverman D, Richardson K, Crutzen P, Foley J (2009) Planetary boundaries: exploring the safe operating space for humanity. Ecol Soc 14(2):32. http://www.ecologyandsociety.org/vol14/iss2/art32/

Rufino MC, Brandt P, Herrero M, Butterbach-Bahl K (2014) Reducing uncertainty in nitrogen budgets for African livestock systems. Environ Res Lett 9:105008. https://doi.org/10.1088/1748-9326/9/10/105008

Sala OE, Chapin FS III, Armesto JJ, Berlow E, Bloomfield J, Dirzo R, Huber-Sanwald E, Huenneke LF, Jackson RB, Kinzig A, Leemans R, Lodge DM, Mooney HA, Oesterheld M, Poff NLR, Sykes MT, Walker BH, Walker M, Wall DH (2000) Global biodiversity scenarios for the year 2100. Science 287(5459):1770–1774

Schlecht E, Hiernaux P (2004) Beyond adding up inputs and outputs: process assessment and upscaling in modelling nutrient flows. Nutr Cycl Agroecosyst 70:303–319

Sutton M, Raghuram N, Adhya TK, Baron J, Cox C, De Vries W, Hicks K, Howard C, Ju X, Kanter D, Masso C, Ometto JP, Ramachandran R, Van Grinsven H, Winiwarter W (2019) The Nitrogen Fix: from nitrogen cycle pollution to nitrogen circular economy, UNEP, Frontiers 2018/19 Emerging Issues of Environmental Concern. United Nations Environment Programme, Nairobi. http://hdl.handle.net/20.500.11822/27543

Tagesson T, Fensholt R, Guiro I, Rasmussen MO, Huber S, Mbow C, Garcia M, Horion S, Sandholt I, Holm-Rasmussen B, Göttsche FM, Ridler M-E, Olèn N, Olsen JL, Ehammer A, Madsen M, Olesen FS, Ardö J (2015a) Ecosystem properties of semi-arid savanna grassland in West Africa and its relationship to environmental variability. Glob Change Biol 21:250–264. https://doi.org/10.1111/gcb.12734

Tagesson T, Fensholt R, Cropley F, Guiro I, Horion S, Ehammer A, Ardö J (2015b) Dynamics in carbon exchange fluxes for a grazed semi-arid savanna ecosystem in West Africa. Agr Ecosyst Environ 205:15–24. https://doi.org/10.1016/j.agee.2015.02.017

Turner MD, Schlecht E (2019) Livestock mobility in sub-Saharan Africa: a critical review, in Pastoralism: research. Policy Pract 9:13. https://doi.org/10.1186/s13570-019-0150-z

Vitousek PM, Menge DNL, Reed SC, Cleveland CC (2013) Biological nitrogen fixation: rates, patterns and ecological controls in terrestrial ecosystems. Phil Trans R Soc B 368:20130119. https://doi.org/10.1098/rstb.2013.0119

Wesely ML (1989) Parameterization of surface resistances to gaseous dry deposition in regional-scale numerical models. Atmos Environ 23:1293–1304. https://doi.org/10.1016/0004-6981(89)90153-4