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D2.2 Concept paper on GHG observations in livestock systems (from mixed crop to extensive) and approaches to quantify net impacts of smallholders on GHG emissions highlighting hotspots of GHG emissions in such systems

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Outline

Executive Summary

This concept paper is anticipated to be submitted to *Environmental Research Letters* in early 2019, a peer-reviewed scientific journal. The focus of this concept paper is primarily on pastoral systems, which are the major livestock production system (LPS) in arid and semiarid regions of (sub-Saharan Africa) SSA containing the largest animal numbers, and often the only way of agriculture due to the harsh climatic conditions. Simultaneously, environmental research carried out in SSA derived during the past five years predominantly focused on smallholder mixed-crop livestock systems whereas less research was carried out on more extensive and often pastoral LPS. Consequently, a literature review on nitrogen (N) cycling – a core nutrient - in pastoral systems in SSA was carried out to summarize the currently available information on N inputs and losses in pastoral systems, with a specific emphasis on N emission hotspots. Our results not only highlight that there is a large knowledge gap on N fluxes within and from those LPS (as a result of limited experimental information available), but also show that specific components such as cattle enclosures (bomas, kraals) are of high importance in relation to N emissions, such as ammonia (NH_3) and/or nitrous oxide (N2O). This review further presents a N balance for a hypothetical pastoral system of approximately 10000 ha including N flux contribution to the N balance based on currently available knowledge. Our calculations confirm that animal congregation areas (i.e. bomas and piospheres) are N emissions hotspots, and that bomas contribute to more than 30% of the N losses via NH_3 volatilization. We further discuss consequences of intensification of pastoral systems and highlight areas of future research to improve N balance estimates. Better N estimates can be used as indicator of land degradation following LPS intensification.

1 Introduction

In developing countries, keeping livestock can substantially contribute to improve food security, livelihoods and wellbeing by providing food and generating income from food and non-food products and services (e.g. draft power for transport and tillage) (Thornton, (2010); Stroebel *et al.*, (2008)). Population growth and rising income, as well as urbanisation in developing countries, is causing an increase in the demand for protein rich food, and consequently livestock products (Swanepoel *et al.*, 2008). This is the case for Africa, the only continent with a positive population growth trend expected for the next 80 years (FAO 2007). The increase in demand for livestock products in Africa offers the opportunity for poor livestock farmers to generate additional income (Thornton, 2010). Conversely, intensification of livestock production and agriculture is hampered by climate change, while simultaneously contributing to climate change and entailing negative environmental consequences such as land degradation, air pollution, water depletion and detrimental effects on biodiversity (Steinfeld *et al.*, 2006). The most important environmental impacts due to livestock are linked to changes in nitrogen (N) cycling at the ecosystem and regional scales, as well as to environmental N losses, such as enhanced greenhouse gas (GHG) emissions including nitrous oxide (N_2O) and other atmospheric pollutants (i.e. ammonia (NH_3) and nitric oxide (NO)) or leaching of N to ground and surface waters. Documented consequences of livestock driven environmental N losses are degradation of water and air quality, and losses in soil fertility. Ammonia volatilization can cause eutrophication of terrestrial and aquatic ecosystems, and acidification of soils when it is returned to ecosystems via dry deposition (Fowler *et al.*, 2013b, Hou et al. 2015). Ammonia emitted to the atmosphere can also present a risk for human health when inhaled, when NH_3 is combined with nitric acid to form particulate matter (PM) (Anderson *et al.*, 2003). N_2O is a GHG, with a 100-year global warming potential 298 times that of carbon dioxide (CO_2) on a per mass basis (Butterbach-Bahl *et al.*, 2013), as well as the dominant ozone-depleting substance (Ravishankara *et al.*, 2009). In addition to N_2O losses via denitrification, molecular dinitrogen (N_2) is ultimately produced through this process. The release of this non-greenhouse gas in large quantities reduces the N content in the soil, and therefore negatively affects soil fertility. Other N losses that may be driven by the presence of livestock include nitric oxide (NO), which is emitted to the atmosphere by soils and lightning and converted into oxides of nitrogen (e.g. nitrogen dioxide, NO_2). The consequent deposition of the nitrogen oxides (NOx) NO and NO_2 further contributes to acidification and eutrophication of ecosystems and can lead to indirect emissions of N_2O.

Thus, concepts for intensification of livestock production systems (LPS) should go hand in hand with appropriate N cycling monitoring (e.g. farm inventories to identify major N pools and flows) (Bosch *et al.*, 1998), and N management (e.g. improvements in manure collection and storage to avoid nutrient losses). As a result, evidence based mitigation and adaptation strategies can be developed to reduce known negative impacts of livestock systems on the environment, without affecting the additional income for livestock keepers, and consequently improving food and nutrition security (Thornton *et al.*, 2009).

Knowledge on N cycling and environmental N losses from LPS in Africa is scarce, even though approximately 25 % of the global cattle population can be found on the continent (Robinson *et al.*, 2011). In Africa, pastoralism is one of the dominant LPS, occupying at least 40 % of the continent's land mass (IRIN, 2007). The pastoral systems are primarily located in arid and semi-arid areas where climate is less suitable for crop agriculture. As such, livestock herding is often the primary, if not the only source of income for the rural poor local society (Barrett *et al.*, 2003). For instance, 23 % (around 300 million people) of the world's poor live in SSA of which 60 % are known to depend on livestock for their livelihoods (Nelson, 2009). In more detail, 25 million pastoralists in SSA are estimated to depend on livestock as their primary source of income (MacCarthy, 2000).

The magnitude of nutrient flows and predominance of N pathways for pastoral livestock systems in SSA remains uncertain, as little research has focused on N cycling and environmental N losses in such systems. Abundance and spatial distribution of N in pastoral systems are strongly affected by management practices and climatic conditions. As such, N dynamics in pastoral systems in SSA, as compared to sedentary or ranching systems in developed countries, are expected to be different. Consequently, this report aims to identify and summarize existing knowledge on N cycling in pastoral LPS in arid and semiarid SSA, as well as estimating the contribution of N transformation, translocation and loss pathways to the total N budget of a virtual farm. Based on our review, we then identify gaps and constraints that might be impeding a realistic environmental impact and nitrogen management assessment of these livestock systems in SSA. We furthermore aim at identifying areas for future research to close currently existing knowledge gaps.

1.1. Characterization of pastoral livestock systems in SSA

Pastoralism in SSA is a traditional LPS present in arid and semiarid savannas. It often represents the only viable way of agriculture and in cases where small crops are also present, it is the main way of agriculture (i.e. agropastoralism). According to the global LPS classification by Seré & Steinfeld, (1996), the geographic distribution of pastoralism is represented by “arid and semiarid tropics and subtropics grassland-based systems” (LGA), and agropastoralism, where livestock herding remains as the main activity, by “arid and semiarid tropics mixed rainfed systems” (MGA). Pastoralism in SSA is characterized by a set of unique management practices, where daily and seasonal mobility is a key feature. Livestock can affect nutrient distribution through their daily movement patterns via feeding and excretion. In their daily movements, cattle are grazed during the day in the savanna and taken for watering usually in the morning and in the evening, before being grouped overnight in cattle enclosures (*boma* in Swahili; *kraal* in Afrikaans). Nutrients are translocated by cattle grazing and manure excretion in the savanna and in congregation areas. Cattle gathering around waterholes during the day and at enclosures over night leads to large deposits of manure in these spots, changing the nutrient distribution of the savanna soils (Tolsma, 2018). The animal congregation areas around waterholes causes the alteration of nutrient concentrations and the spatial distribution of vegetation (Andrew, 1988) (Young, 1995) (Muchiru *et al.*, 2009) (Augustine, 2003). This impact pattern is called “piosphere” (Andrew, 1988). Within the savannah, cattle trampling on cattle pathways leads to soil compaction and reductions in soil fertility, as pores size and, thus water infiltration are reduced, factors which negatively affect both water and soil nutrient storage (Hamza & Anderson, 2005). Secondly, and due to high variability of rainfall in arid and semiarid savannas, seasonal mobility of herds to allow grazing of fresh pasture in different regions or to reach markets is common practice. Key management practices and features of pastoral systems in SSA and consequent effects on nitrogen dynamics can be summarized as (1) enclosure of cattle/sheep/goat herds in bomas at night, leading to an accumulation of nutrient via manure in them, (2) daily mobility of animals leading to a re-distribution of nutrients through grazing and deposition of faeces across the landscape (3) daily watering of animals, leading to an accumulation of livestock manure and nutrients around watering points, and (4) herds seasonal movements resulting in N losses from the system. Taking into account the individual management practices it is crucial to identify potential hotspots of N inputs/losses that might lead to develop an environmental impact assessment providing options to minimize N losses.

1.2. Nitrogen dynamics in pastoral livestock systems in SSA

Pastoral livestock systems in SSA consist of three distinct areas, i.e. rangeland, occupying >96-98% of the landscape area, animal enclosures (i.e. bomas) and watering points such as wells or rivers (Figure 1). These areas show fundamental differences with regard to N inputs and losses and are

differently affected by translocation of nutrients along atmospheric and hydrological pathways and due to livestock feed intake and excretion of faeces.

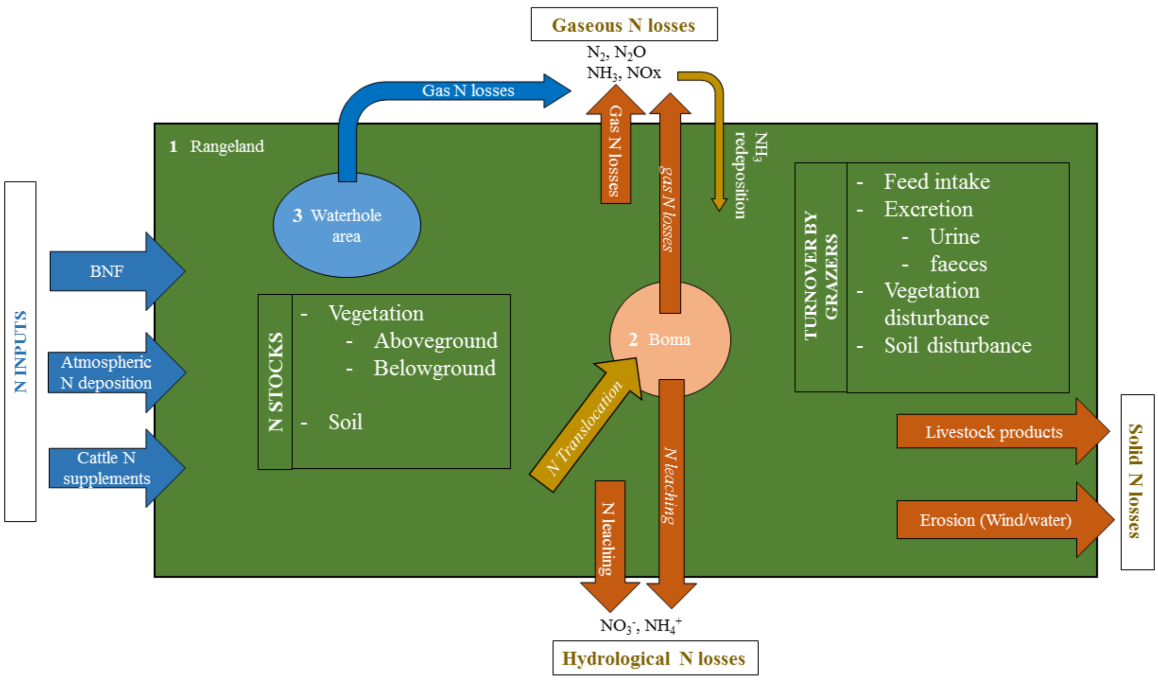

Figure 1. Simplified scheme of nitrogen dynamics in pastoral systems (green) in SSA. Inputs and gaseous N losses from the system with blue and brown arrows respectively.

Overall, nitrogen inputs into the system occur through biological nitrogen fixation (BNF), wet and dry deposition to the land surface and via supplementary feed if provided. Nitrogen losses from pastoral systems are mainly either in gaseous form as ammonia volatilization, N_2O emission (as a potent greenhouse gas) or emission of other N gases such as nitrogen oxides (NOx) and dinitrogen (N_2), as well as along hydrological pathways due to leaching or surface run-off of organic and inorganic nitrogen (i.e. NO_3^-, NO_2^- and NH_4^+) compounds (Carran & Clough, 1996). The gaseous N losses as well as leaching occurs in all three areas with varying magnitude (Figure 1). Congregation areas are expected to have higher gaseous N losses, specifically N_2O and NH_3 emissions, than the surrounding landscape, as a result of nutrient concentration effects. Enhanced cation exchange capacity with ammonium and potassium and higher urease activity due to manure accumulation, (Sheppard & Bittman, 2011), as well as a pH increase (Whalen *et al.*, 2000) and lower C:N ratio are expected in congregation areas. Other N losses occur through export of animal products such as meat and milk.

2 Methodology

2.1. Search protocol and selection criteria

To characterize and quantify N dynamics and environmental N losses of pastoral systems in SSA an extensive literature research was carried out using available databases (Scopus, Web of Science) and search engines (Google Scholar). Main keywords from Table 1 were combined with N flow specific keywords to search studies on N cycling in pastoral systems in SSA.

Table 1. Keywords used in literature search grouped in main keywords and N flow specific keywords.

Main keywords	Nitrogen inputs and outputs/export	Soil and vegetation nitrogen stocks	N emissions from bomas and waterholes
nitrogen	BNF	stocks	urine
livestock	deposition	soil	faeces
africa	volatilization	vegetation	excreta
pastoral	denitrification	biomass	intake
savanna	leaching	plant	bomas
	ammonia		cattle
	nitrous oxide		enclosures

	nitric oxide		feedlots
			ammonia
			waterholes
			piospheres

Data retrieved from the search were classified into four categories for simplification of analysis: (1) soil nitrogen (N) stocks, (2) plant N stocks, (3) nitrogen inputs and export/losses into and from the landscape and (4) nitrogen emissions from bomas and piospheres. We extracted data from global databases to calculate N dynamics for pastoral systems in SSA when not available in studies on pastoral systems in SSA. When data was not available in literature or global databases, or when we considered that the figures were not appropriate to include in this study (e.g. weak or missing description of the experiment or methods, uncommon management or climatic conditions), N dynamics values were estimated following a best guess approach (Figure 2).

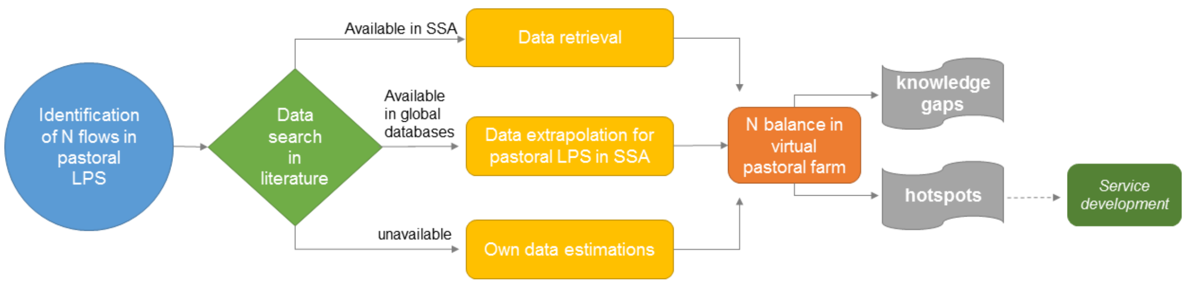

Figure 2. Flow diagram depicting methodology followed in this study.

Data units reported varied amongst studies, thus we unified units to kg N ha^{-1} yr^{-1} for N fluxes and kg N ha^{-1} for soil and vegetation N stocks.

2.2. Data collection and calculation

To visualize estimated N flows and N losses of pastoral systems we used the approach of a virtual farm (Table 2), with features of livestock management, vegetation, soils and climate being representative for parts of Eastern Africa, and here specifically Kenya, although these features can be found as well in the large semi-arid areas in West-/East and South-Africa. There are some differences (e.g. environmental conditions) between pastoral systems across arid and semiarid savannas in SSA that could affect N dynamics, such as rainfall patterns, although in general arid savannas receive less than 650 mm of rainfall every year ((Lulla, 1987); Huntley & Walker, (2012)). For instance, in East African savannas, rainfalls follow a bimodal pattern, with one long rainy season from March to May and a short one from October to November (McCown & Jones, 1992), whereas Southern and West African savannas receive unimodal rainfall patterns, with a short wet season from October to January in Southern Africa (Meixner *et al.*, 1997a) and from June to September in West Africa (Schlecht & Hiernaux, 2004).

Table 2. Data to calculate input/output N balances of a hypothetical pastoral farm in Kenya.

Variables	Units	Value	Reference
Farm area	ha	10000	(Ng'ethe, 1993)
Stocking rate	TLU[a] km^{-2}	17.6	GLiPHA
Number of cattle/boma	TLU	100	-
Boma area	ha	0.1	-
Time in boma/day	d	0.5	Rufino et al 2006, Augustine et al 2003
Number of bomas	#	18	-
Number of waterholes	#	10	-
Time in waterhole/day	d	0.1	-
Waterhole area	ha	0.5	-
Piosphere area[b]	ha	5.7	Smet & Ward, (2006)
Dry season duration	days	240	-
Wet season duration	days	120	-
N excretion dry season	gN $TLU^{-1}d^{-1}$	90 ±10	Schlecht *et al.*, (1995)
N excretion wet season	gN $TLU^{-1}d^{-1}$	142 ±13	Schlecht *et al.*, (1995)
N intake dry season	gN $TLU^{-1}d^{-1}$	90 ± 12	Schlecht *et al.*, (1995)

N intake wet season	gN $TLU^{-1}d^{-1}$	161 ± 14	Schlecht *et al.*, (1995)
N excretion lost as N_2O/NH_3	%	2 (N_2O)/50 (NH_3)	Delon et al 2010, Davidson 2009
Deposited manure N lost as NO_3^- leaching	%	6%	(Eghball *et al.*, 1997)

[a]TLU refers to Tropical Livestock Unit of 250 kg live weight (Schlecht & Hiernaux, 2005), equivalent to 1.4 cattle heads.

[b] The piosphere area was calculated as the area which is 100 m away from the the waterhole

The calculation of input/output N balances of the virtual farm was done stepwise:

1. Retrieved or estimated N fluxes for savanna soils.
2. Calculated N fluxes for potential "hotspots" (i.e. bomas and waterholes).
3. Calculated the contribution of each N flow taking into account boma and waterhole density.

2.2.1. Soil and vegetation nitrogen stocks and tree density

Soil N stocks for 0-10, 10-30, 30-50 and 50-100 cm depths were calculated by extracting data from the Africa soil profiles database (Leenaars, 2013) available at the International Soil Reference and Information Centre (ISRIC) website (http:// www.isric.org). Soil profiles were selected by overlapping GPS coordinates with pastoral systems areas in arid and semi-arid SSA (Figure 3). Firstly, we determined regions in SSA where pastoral systems are represented following Seré & Steinfeld (1996) LPS classification (i.e. LGA and MGA) with QGIS software (QGIS, V. 2.18.13, 2017) by using the "GIS map Global livestock production systems v.5" from the FAO Geo Network website (http://www.fao.org/geonetwork). We then mapped soil types (FAO/UNESCO Soil Map of the World) to assign a soil type to each profile, as soil profiles were not classified under the same system in the Africa soil profiles database. For the soil profiles selected, we calculated nitrogen stocks for 0-10, 10-30, 30-50 and 50-100 cm depths according to Ellertl & Bettany, (1995) formula (1):

$$\boldsymbol{M_{nitrogen} = conc * \rho_b * T * 10000m^2ha^{-1} * 0.001Mgkg^{-1}} \quad (1)$$

Where $M_{nitrogen}$ is the mass of N per unit area (Mg N ha^{-1}), *conc* is the N concentration (kg Mg^{-1}), ρ_b is the bulk density (Mg m^{-3}), T is the thickness of soil layer (m).

Vegetation N stock values for our virtual farm were calculated using tree density calculated by performing a random forest classification from Sentinel imagery in QGis (QGIS, V. 2.18.13, 2017), and multiplying by tree N stock from (Bernhard-Reversat & Poupon, 1980) (Table 4). Grass N stocks were calculated from two studies measuring biomass and N content in biomass developed in SSA savannas ((Wang *et al.*, (2012), Knox *et al.*, (2011)).

2.2.2. Nitrogen inputs and export/losses

Atmospheric N deposition data were extracted from the global maps of atmospheric nitrogen deposition, 1860, 1993 and 2050 database (Dentener, 2006). In this database, N deposition model estimates are calculated with a resolution of 5 degrees longitude by 3.75 degrees latitude. We projected model estimates across the study region (i.e. LGA, MGA) with QGIS (2017), resulting in 54 values. We estimated NH_3 and N_2O emissions from the savanna soils related to number of cattle heads following Delon *et al.* (2010) methodology. The amount of N excreted by cattle (gN head^{-1} day^{-1}) is calculated from Schlecht *et al.* (1995) for rainy and dry seasons. This number was divided by two to calculate the amount of N excreted by cattle while grazing for 12 hours during the day. This number is then multiplied by cattle stocking rate (TLU km^{-2}) in our region using the GLiPHA database (Global Livestock Production and Health Atlas) (Table 2). A 50% and 2% loss rates were applied to the calculations of N input by cattle excretion for NH_3 and N_2O respectively. Annual emissions were calculated by multiplying daily emissions in dry and wet seasons by days in the respective seasons. The same methodology was applied for NH_3 and N_2O emissions from the bomas and piospheres using the corresponding stocking density (Table 2) and applying 40% and 10% to the amount of N excreted due to the time spent in bomas and piospheres, respectively.

$$F_{NH3/N2O}= N_{ex}*t_f*\rho*L_r*d*1000/100 \quad (2)$$

Where $F_{NH3/N2O}$ is the NH_3 or N_2O flux in kg N ha^{-1} yr^{-1}, Nex is the amount of N excreted by cattle in gN head^{-1} day^{-1} in dry or wet season, tf is the percentage of time spent in savanna, bomas or piospheres, ρ is the stocking rate in savanna, bomas or piospheres in TLU ha^{-1}, Lr is the percentage N loss rate for NH_3 or N_2O, and d is days in dry or wet season. Data for other N inputs and losses were extracted from literature and classified into N flows (Table 5).

2.2.3. Contribution of landscape and congregation areas N flows to the total system N cycle

The contribution of each N flow within the system to the total budget is necessary to understand the importance of each flow and to identify hotspots and thus the resilience of the system to factors that could modify these flows (e.g. climatic conditions, droughts, climate change, land use change). We calculated the contribution of each N flow for our virtual farm (in %) by multiplying N fluxes by the source area.

3 Discussion

Several publications that studied N cycling in tropical savannas are available (Coetsee *et al.*, (2012) in South Africa, (Holt *et al.*, (2018) in Australia and Bustamante *et al.*, (2006) in America). Most of those only present partial N balances or studied only some of the N flows and factors affecting them, e.g. the study by Delon *et al.* (2009) reported a partial N budget for West African savanna using simulated and calculated inventories as well as in-situ measurements and showed that the dominant source of N losses occurred via NH_3 volatilization. However, potential N losses hotspots such as bomas were not considered and only atmospheric N flows are taken into account for the budget calculation. Similarly, other N inputs and loss, ie. via biological nitrogen fixation (BNF) were no and other N losses, via leaching or animal products export were not considered. During our review, we were unable to find studies estimating the full nitrogen balance, including livestock-related flows (e.g. N emissions from bomas and piospheres), for pastoral livestock systems in SSA.

3.1. Soil Nitrogen stocks

Soil N stocks play a key role in calculations of N flows in livestock systems. So far only a limited number of studies on nutrient balances in Africa link N flows to soil N stocks (Cobo *et al.*, 2010). Furthermore, most studies reporting soil N stocks in savannas analysed the soil top layers only (e.g. 10, 30 cm) (Bernhard-Reversat, 1982)(Scholes & Andreae, 2000) (Cech *et al.*, 2010) (Lesschen *et al.*, 2007) justifying this sampling depth with the fact that SOM is usually accumulated in top layers in semiarid savannas. Here, we present soil N stock estimates in pastoral regions in SSA from 178 profiles (figure 3), for four soil layers up to 1 m depth (i.e. 0-10, 10-30, 30-50 and 50-100 cm) (Table 3). Our analysis shows that 41% of the soil N content for 1 m depth is located in the 50-100 cm layer, while approx. 15% of the soil N stocks can be found in the first 10 cm. The average N content found up to 1 m depth soils for our study region were 5.0 ± 3.6 Mg ha^{-1}.

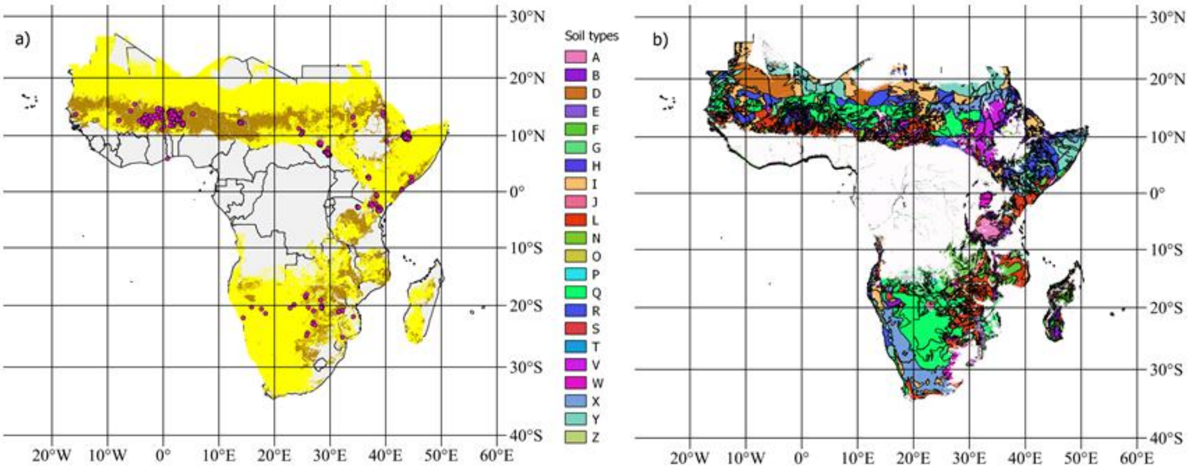

Figure 3. a) Geographic distribution of pastoralism (and agropastoralism) following the livestock production systems classification by Sere and Steinfeld (1996). The yellow region represents the arid and semiarid tropics and subtropics grassland-based systems (LGA) and the brown region represents the arid and semiarid tropics and subtropics mixed rainfed systems (MGA). Soil profiles selected from the Africa soil profiles database (Leenaars, 2013) are represented by pink dots. b) Geographic distribution of FAO soil types in pastoral systems in SSA

Table 3. Summary statistics of soil N stocks at 0-10 cm, 10-30 cm, 30-50 cm and 50-100 cm

Depth (cm)	Mean N stocks (Mg N ha^{-1})	±sd	Mean N concentrations (g N kg^{-1})	±sd
0-10	0.8	0.7	0.6	0.7
10-30	1.2	1.0	0.4	0.4
30-50	1.0	0.7	0.3	0.2
50-100	2.1	1.5	0.2	0.2
0-100	**5.0**	**3.6**		

The N content decreases with depth, following a logarithmic regression with a correlation coefficient R=0.99 (Figure 4). For soil N stocks from 0-100 cm, we found that Regosols (R) is the soil type in pastoral systems with the lowest N stock (Figure 5).

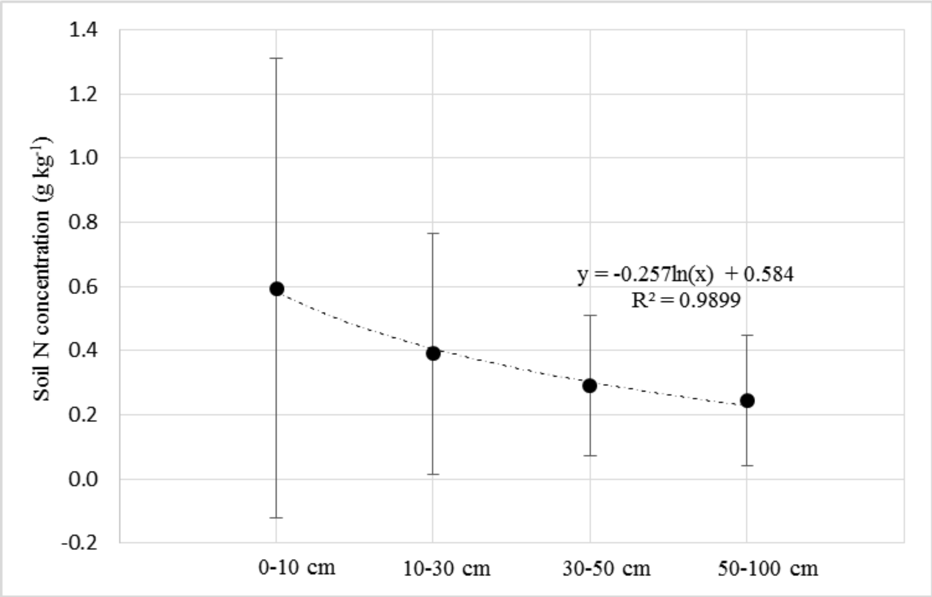

Figure 4. Vertical distribution of soil N concentrations in 0-10, 10-30, 30-50 and 50-10 cm layers

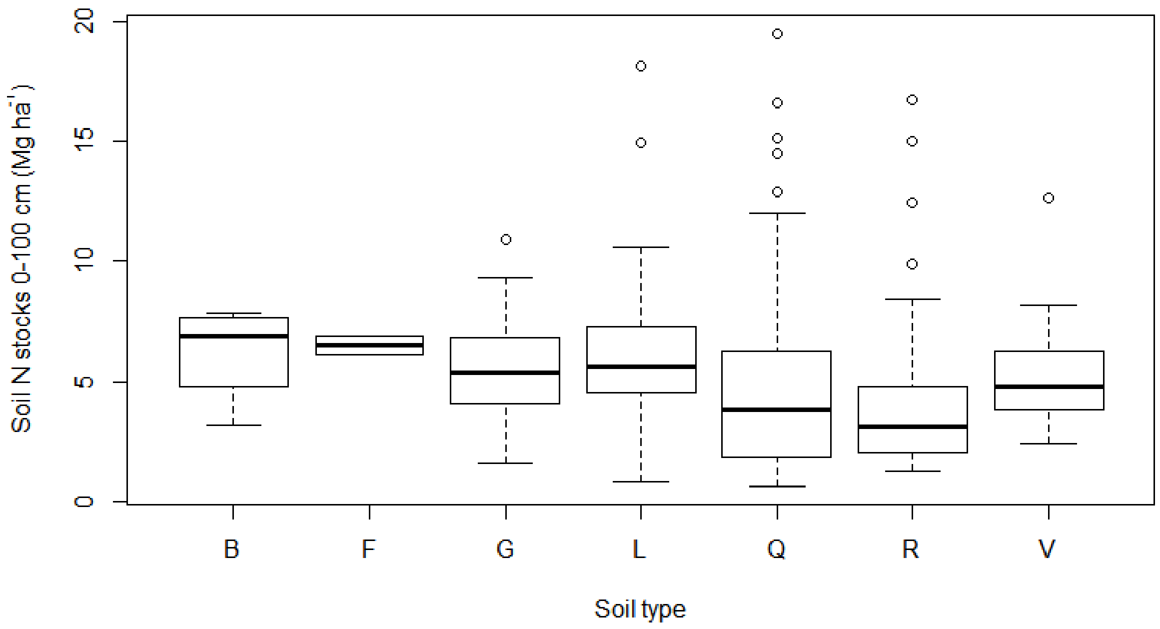

Figure 5. Boxplots of N stocks in 0-100 cm (grey) for the soil types found in pastoral systems in SSA. FAO-Unesco soil type: B=Cambisols (N=4), F= Ferrasols(N=4), G= Gleysols (N=16), L= Luvisols (N=15), Q=Arenosols (N=68), R=Regosols (N=53), V=Vertisols (N=20)

In pastoral systems, the tree-grass patterns affect soil organic matter (SOM) distribution and thus N availability. For instance, it is expected to find higher N availability (i.e. double N content under *Acacia* and *Balanites* compared to the surrounding savanna, (Bernhard-Reversat, 1982)) in areas of bush encroachment, due to the capacity of most *Acacia* species to fix N symbiotically (Cech *et al.*, 2010). This suggests that besides soil type and texture also the vegetation spatial distribution should be considered when estimating soil N stocks. For our hypothetical farm in Kenya, soil N stocks were calculated for Arenosols (Q) (predominant savanna soil in East Africa, Figure 3). For the savanna areas without presence of trees, the value calculated from the Africa soil database was used, i.e. 4858 ± 4013 kg N ha^{-1}. Tree density was used to calculate soil N stocks underneath the trees, assuming that this number doubles in the 3 m^2 area underneath the tree (Bernhard-Reversat, 1982).

3.2 Vegetation N stocks

Vegetation N stocks are one of the main N pools in pastoral systems and therefore necessary to include in N budget studies. However, similarly to soil N stocks, accurate estimates are difficult to derive due to the different vegetation densities across the arid and semi-arid regions and within this livestock system category. Typical vegetation in SSA pastoral systems is composed by an herbaceous layer (e.g. *Themeda sp.*), and dispersed trees (e.g. *Acacia sp.*) and shrubs (e.g. *Euphorbia sp.*). In this study, we calculated the tree cover of our virtual pastoral farm to be able to estimate the total biomass stocks. We took grass biomass values (i.e. 2980 kg N ha^{-1}) from a study monitoring vegetation responses to different nutrient treatments in Kalahari, Namibia developed by (Wang *et al.*, 2012) (Table 4). The N content in grass biomass (i.e. 0.53% DM) was taken from the study developed in Kruger National Park by Knox *et al.* (2011) where the authors quantified the forage quality using remote sensing. Our calculations of tree (i.e. *Acacia*) N stocks were made from the study on N cycling in a soil-tree system in a Senegalese savanna, by Bernhard-Reversat & Poupon, (1980). The authors also estimated herbaceous N stocks, resulting in 47 kg N ha^{-1}, much higher than our calculations of ~16 kg N ha^{-1}, probably because our biomass values were from grazed pasture samples. Our total vegetation N stocks were compared with results from a study in a savanna in Nylsvley in South Africa (Scholes and Walker, 1993) (Table 4).

Table 4. Soil and vegetation N stocks for pastoral systems in SSA

Ecosystem Compartments	Stock (kg N ha^{-1})	References
Total N in arenosols (0-100 cm) (kg N ha^{-1})	4858±4013	calculated from (Leenaars, 2013)
Total soil N underneath trees (kg N ha^{-1})	NA	calculated from (Bernhard-Reversat, 1982)
Total N in our virtual farm (kg N ha^{-1})	NA	-
Grass biomass (kg ha^{-1})	2980[a]	(Wang *et al.*, 2012)
Grass N content (% DM)	0.53	(Knox *et al.*, 2011)
Grass N stock in our virtual farm (kg N ha^{-1})	15.8	-
Grass N stock (kg N ha^{-1})	47	(Bernhard-Reversat & Poupon, 1980)
Tree N stocks (kg N $tree^{-1}$)	0.44	(Bernhard-Reversat & Poupon, 1980)

Total biomass N in our virtual farm (kg N ha^{-1})	NA	-
N biomass(kg N ha-1)	322.3	Scholes & Walker (1993)
Total N stock (soil + vegetation)	NA	-
Net N mineralization	9.4±8.0	(Augustine & McNaughton, 2004)

NA – values not available at the time of the report

[a] value for grazed pasture

3.3. Nitrogen inputs and exports in the surrounding savanna

3.3.1. Nitrogen inputs

3.3.1.1 Biological nitrogen fixation (BNF)

Tropical savannas have been identified as hotspots for BNF. N-fixing trees, predominantly from the family *Fabaceae* (e.g. *Acacia, Mimosa*) are found in tropical savannas, where temperatures are within the optimum (26°C) of nitrogenase activity (Houlton *et al.*, 2008). However, tree density, and subsequently BNF rates vary widely between savanna regions. Water availability in the soil is also a requisite for BNF to occur, therefore crucial in semiarid environments such as pastoral systems in SSA (Gray *et al.*, 2013). Cleveland *et al.*, (1999) estimated BNF fluxes of 16.4, 30.2 and 44 kg N ha^{-1} yr^{-1} for tropical savanna ecosystems with 5%, 15% and 25% tree cover, respectively. Symbiotic N-fixation is higher than non-symbiotic N-fixation in most cases. Chen *et al.* (2010) estimated through satellite imagery and a chemical transport model that the BNF rate for African savannas is on average 18.6 kg N ha^{-1} yr^{-1}. Houlton *et al.* (1999) estimated through a model-based analysis a BNF range of 20-60 kg N ha^{-1} yr^{-1} for tropical savannas globally, and Robertson & Rosswall, (1986) report a total estimated BNF of 30 N ha^{-1} yr^{-1} for a grazed west African savanna, with symbiotic N fixation contributing 1/3 to total ecosystem BNF. Ndoye *et al.*, (1995) measured, at the end of a five month-experiment, BNF per tree fixed by different Acacia species using the ^{15}N isotope dilution method, finding that they fixed on average 0.48 g N plant^{-1} during the five months after plantation. In this study, for our virtual pastoral system with 5-10% tree density (i.e. *Acacia sp*), we assumed a BNF flux of 16 ± 8 kg N

ha^{-1} yr (Cleveland *et al.,* (1999). The uncertainty of 50% for BNF was estimated by Fowler *et al.*, (2013). Other authors report much smaller BNF values, e.g. 0.4 - 5 kg N ha^{-1} yr^{-1} for Sahelian rangelands (Krul *et al.,* 1982). These low values represent around 1/6 of the BNF estimate we are using here. However, this low estimate seems to be due to much lower density of N fixing species in the Sahelian region in the study of Krul *et al.* (1982) than in our virtual farm in Kenya, or predominance of low N fixing *Acacia* species.

3.3.1.2. Atmospheric N deposition

Our calculations of atmospheric N deposition in savannas across SSA resulted in an estimate of 3.1 ± 2.4 kg N ha^{-1} yr^{-1} (Table 5). Deposition of reduced N compounds (NHx) resulted higher than oxidised N compounds (NOy) with averages of 1.72 ± 1.51 and 1.43 ± 1.09 kg N ha^{-1} yr^{-1}, respectively. These results are similar to those from other studies in SSA savannas. For instance, Bate (1981) reported a range of 2 - 5 kg N ha^{-1} yr^{-1} of wet N deposition in a Burkea savanna in Nigeria. Augustine (2003) measured 4.3 kg N ha^{-1} yr^{-1} of wet deposition in a Kenyan semi-arid savanna while Ruess & McNaughton, (1988) calculated a range of 4 - 11 kg N ha^{-1} yr^{-1} of wet deposition in the Serengeti National Park in Tanzania. Delon *et al.* (2012) measured total N deposition flux of 7.40 ± 1.0 kg N ha^{-1} yr^{-1} in a dry savanna in West Africa. The values for N deposition in West African savannas could be higher than other savannas in SSA due to the effect of the Harmattan wind on dry deposition (Lesschen *et al.*, 2007). Due to difficulties when measuring dry deposition, most of the studies only report wet deposition. From the few studies found where dry as well as wet deposition were measured or estimated, we can assume that wet and dry N deposition contributes around the same proportion to the total atmospheric deposition. However, this seems to be only the case for areas where there are no high concentrations of atmospheric NH_3 as e.g. in direct vicinity of bomas or to other anthropogenic activities. For instance, Scholes et al (2003) estimated a deposition flux of 21.6 kg N ha^{-1} yr^{-1} in Kruger National park, South Africa, situated downwind of coal burning sources resulting in much higher fluxes than the calculations for our virtual farm in Kenya.

Table 5. N fluxes in savanna soil and congregation areas (bomas and piospheres) in pastoral systems in SSA

Flux	N flux value (kg N ha^{-1} yr^{-1})	Region	Source
N flows in savanna soils			
BNF	**NA**	Virtual farm in Kenyan savanna	Ndoye et al (1995) *
	16.4, 30.2 and 44[a]	Tropical savanna ecosystem	Cleveland et al (1999)
	18.6	African savanna	Chen et al (2010)
	20-60	Tropical savanna ecosystem	Houlton et al (1999)
	30	Savanna in West Africa	Robertson&Roswall (1986)
N deposition	**3.1(2.4)**	Virtual farm in Kenyan savanna	Dentener, F.J. (2006)
	2-5	Savanna in Nigeria	Bate and Gunton (1981)
	4.3	Savanna in Kenya	Augustine (2003)
	4-11	Savanna in South Africa	Ruess&McNaughton (1988)
	7.4(1.0)	Savanna in West Africa	Delon et al (2012)
Ammonia (NH_3) emissions	**1.7(0.8)**	Virtual farm in Kenyan savanna	Schlecht et al., (1995) ;Delon et al., (2010) *
	-0.03-0.41	Savanna in Senegal	Delon *et al.*, (2017)
	8.4(3.8), 12.5(5.9)[b]	Savanna in West Africa	Delon et al (2010)
Nitrous oxide (N_2O) emissions	**0.25(0.15)**	Virtual farm in Kenyan savanna	Rees *et al.*, (2006) Scholes *et al.*, (1997)
	0.25 - 0.5	Savanna in Zimbabwe	Rees *et al.*, (2006)
	0.03 – 0.27	Savanna in South Africa	Scholes *et al.*, (1997)
	0.52, 0.67[c]	Savanna in Burkina Faso	Brümmer *et al.*, (2008)
Dinitrogen (N_2) emissions	**0.5(0.3)[d]**	Virtual farm in Kenyan savanna	Schlesinger (2009)
	0.99	Savanna in Zimbabwe	Rees *et al.*, (2006)
Nitrogen oxides	**0.5(0.1)**	Virtual farm in Kenyan savanna	Meixner et al (1997b)*

(NOx) emissions			
	1.5-1.6	Savanna in West Africa	Galy-Lacaux&Delon (2014)
	1.4(0.3)	Savanna in West Africa	Delon et al (2012)
	1.5	Savanna in South Africa	Otter et al (1999)
	1.5	Savanna in South Africa	Scholes et al (1997)
	0.1 - 2.0; 1.8 - 10.7[e]	Savanna in South Africa	Levine et al (1996a)
	0.1; 1.4[f]	Savanna in Zimbabwe	Meixner et al (1997b)
	2	Virtual farm in Kenyan savanna	Rees *et al.*, (2006)
N leaching	2-3	Savanna in Zimbabwe	Rees *et al.*, (2006)
	5.1[g]	Savanna in West Africa	Robertson & Rosswall, (1986b)
N milk offtake	**0.1**	Virtual farm in Kenyan savanna	Nicholson, (1984), Semenye & de Leeuw, (1986) and Rufino et al., (2006) *
		N flows in bomas and piospheres	
Ammonia (NH_3) volatilization from bomas	7728(3864)	Virtual farm in Kenyan savanna	Schlecht et al., (1995) ;Delon et al., (2010) *
Nitrous oxide (N_2O) emissions from bomas	309(185)	Virtual farm in Kenyan savanna	Schlecht et al., (1995) and Davidson (2009) *
Dinitrogen (N_2) emissions from bomas	927(556)	Virtual farm in Kenyan savanna	(Jarvis&Pain, 1994) *
NO3- leaching from bomas	1022	Virtual farm in Kenyan savanna	Eghball & Power, (1994) *
Ammonia (NH_3) volatilization from piospheres	680(340)	Virtual farm in Kenyan savanna	Schlecht et al., (1995) ;Delon et al., (2010) *
Nitrous oxide (N_2O) emissions from piospheres	27(16)	Virtual farm in Kenyan savanna	Schlecht et al., (1995) and Davidson (2009) *

Dinitrogen (N_2) emissions from piospheres	81(49)	Virtual farm in Kenyan savanna	(Jarvis&Pain, 1994) *
Leaching, runoff and erosion from piospheres	365	Virtual farm in Kenyan savanna	Eghball & Power, (1994) *

[a] estimations for 5%, 15% and 30% of tree cover

[b] volatilization estimates with N loss rates of 30% and 50%

[c] results for years 2005 and 2006

[d] N_2O:(N_2+N_2O) ration

[e] results for dry and wet seasons

[f] results for dry and wet seasons

[g] N lost as leaching, runoff and erosion

* own calculations

3.3.2. Nitrogen losses

3.3.2.1. Ammonia (NH_3)

In pastoral systems in SSA, NH_3 volatilization due to bacterial decomposition of urea in livestock manure and emissions from natural soils are the two major sources of NH_3 (Adon *et al.*, 2010). Very few studies that estimated NH_3 emissions from pastoral systems have been developed. However, grazing in semi-natural ecosystems (i.e. savanna), related to cattle stocking rates, has been identified as the major source of NH_3 emissions (Delon et al, 2010), elevating the rate of NH_3 volatilization up to an order of magnitude compared to undisturbed ecosystems where there is no presence of livestock (Bowden, 1986). Delon *et al.*, (2017) measured soil NH_3 exchange with manual closed dynamic chambers from a pastoral system in a Senegalese semiarid savanna. The resulting NH_3 flux was close to 0, with fluxes fluctuating between emission and deposition from -0.03 to 0.41 kg N ha^{-1} yr^{-1}. Delon et al (2010) estimated NH_3 emissions from livestock manure in West African dry savanna related to the number of heads of cattle. The total N input from animal was estimated with the amount of N released per cow multiplied by cattle density in each region. Loss rates of N excretion that is lost as NH3 is been reported as 10% to 36%, depending on animal waste management. Due to favourable conditions in the Sahel for NH_3 volatilization (i.e. high temperatures, low soil moisture and bare soils), Delon et al (2010) applied loss rates of 30% and 50%, to the input of N by animal manure leading to NH_3 volatilization of 8.4±3.8 and 12.5±5.9 kg N ha^{-1} yr^{-1} respectively. Our calculations (with 50% of excretion N as NH_3 volatilization) with an uncertainty of 50% resulted in 1.7±0.8 kg N ha^{-1} yr^{-1} (Table 5). Our results are lower than results reported in other studies probably because we estimated a lower ratio of NH_3 volatilization per TLU. For instance, Galy-Lacaux & Delon (2014) and Delon et al (2010) used a ratio of NH_3 volatilization per TLU of 57.5 and 96 g N-NH_3 TLU^{-1} d^{-1} (depending on the loss rate, 30% or 50%), whereas for our estimations we calculated emissions of 22.5 and 35.5 g N-NH_3 TLU^{-1} d^{-1} for dry and wet seasons respectively. These values result from taking daily N excretion (i.e. 90 and 146 gr TLU^{-1}) from (Schlecht *et al.*, 1995), assuming that cattle excrete half of this value while grazing and with 50% of the excreted N in the savanna lost as NH_3.

3.3.2.2. Nitrous oxide (N_2O)

In pastoral systems, soil N_2O emissions are enhanced by the presence of animals. Urine patches are the main source of N_2O in grazing systems (Gerber, 2014) as the urine increases N availability in the soil system and, thus, the microbial formation of N_2O via nitrification and denitrification pathways. Furthermore, trampling increases soil compaction and lowers soil O_2 availability (Liu *et al.*, 2007), a major factor for promoting denitrification. Thus, trampling has been found to affect the magnitude of N_2O emissions from soils, which may even double (Oenema *et al.*, 1997). However, the overall contribution of N_2O to the N budget in semi-arid African savannas with low soil humidity can be considered negligible (Delon *et al.*, 2012) as biogenic soil N_2O emissions are often below the detection limit (Levine *et al.*, 1996a) and, thus, quantitatively of little importance for the overall N budget of a pastoral system. Rees *et al.*, (2006) highlighted that only a little proportion of nitrogen losses occurs via N_2O emissions in the miombo savanna in Zimbabwe, reporting fluxes of 0.25 - 0.5 kg N ha^{-1} yr^{-1} from soils containing 3000 kg N ha^{-1} yr^{-1} in the top 60 cm. Furthermore, Scholes *et al.*, (1997) measured N_2O and NO emissions from a South African savanna, estimating N_2O fluxes of 0.03 – 0.27 kg N ha^{-1} yr^{-1}, and, thus one magnitude lower as NO emissions. Similarly, low N_2O fluxes were reported by Brümmer *et al.*, (2008) in a savanna in Burkina Faso, with annual means of 0.52 and 0.67 kg N ha^{-1} yr^{-1} for 2005 and 2006, respectively. Mean annual N_2O fluxes measured by Brümmer *et al.*,

(2008) are slightly higher than in the few other studies on N_2O emissions from savanna soils, probably due to high soil WPFS during the measurement periods. Overall, the current knowledge indicates that N_2O emissions from grazed savanna soils are quantitatively unimportant for N budgeting, which is also true for other soil systems worldwide, as N_2O is only a side or by product of nitrification/ denitrification. For our study, we took a flux in between the N_2O fluxes reported in the Zimbabwean miombo and South African savanna publications, 0.25 ± 0.15 kg N yr^{-1} ha^{-1}, with 60% uncertainty.

3.3.2.3. Dinitrogen (N_2)

Fluxes of dinitrogen (N_2) gas due to denitrification are only significant in soils that can retain anaerobic conditions for extended periods of time. As most African savannas soils are too sandy to remain anaerobic one might assume that N_2 emissions are low (Scholes *et al.*, 1997) and negligible (Delon *et al.,* 2009). We could only find one study reporting N_2 emissions from a miombo savanna in Zimbabwe, with estimates being based on modelling (0.99 kg N ha^{-1} yr^{-1} via N_2 emissions; Rees *et al.*, 2006). Schlesinger (2009) estimated that the N_2O: (N_2+N_2O) ration for N gas emissions from soils under natural vegetation or recovering vegetation is approx. 0.5, which means that in average N_2 emissions might be twice as high as N_2O fluxes. However, this remains speculation. Nevertheless, we used a value of 0.5±0.3 kg N yr^{-1} ha^{-1}, i.e. twice as high as N_2O fluxes, for the savanna soils of our virtual farm.

4.3.2.3. Nitrogen oxides (NO_x)

Galy-Lacaux & Delon, (2014) found that biogenic NO emissions in savanna soils account for 17% of the total N emissions, excluding N_2 fluxes due to denitrification, in dry savanna ecosystems. They estimated mean soil biogenic NO emissions being in a rather narrow range of 1.5-1.6 kg N ha^{-1} yr^{-1} for three sites in the West African savanna. Similar magnitudes of soil NO emissions were reported in a study at the same research sites: an annual mean of 1.4±0.3 kg N ha^{-1} yr^{-1} was estimated using a model by Delon *et al.* (2012). A mean value of ~1.5 kg N ha^{-1} yr^{-1} was given in two different studies undertaken in the Nylsvley Nature Reserve in South Africa, one using a model to estimate soil NO emissions (Otter *et al.*, 1999), while the other study was based on measuring NO fluxes with chambers (Scholes *et al.*, (1997). Soil NO fluxes were found to differ considerably between hydrological seasons. Levine *et al.*, (1996a) reported a range of

0.1 - 2.0 kg N ha^{-1} yr^{-1} for dry season, and 1.8 - 10.7 kg N ha^{-1} yr^{-1} for the wet season in a study in Kruger National Park in South Africa, where the rainy season was simulated by applying water prior to each measurement. Measured soil NO emissions at a savanna site in Zimbabwe, using a dynamic chamber approach, showed mean emissions of 0.1 kg N ha^{-1} yr^{-1} in the dry season and 1.4 kg N ha^{-1} yr^{-1} in the wet season (Meixner *et al.*, 1997b). This suggests that NO fluxes in the wet season might be elevated up by one magnitude compared to the dry season. Nitrogen oxide emission values for wet and dry seasons from Meixner et al., (1997b) were used in our study, resulting in 0.52 ± 0.1 kg N ha^{-1} yr^{-1}.

4.3.2.4 Hydrological nitrogen losses and erosion

In semiarid regions in SSA nitrate leaching is likely confined to the wet seasons, as only during this period significant rainfall (>30-50 mm/ month) is usually observed. Rees *et al.*, (2006) reported that 2 - 3 kg N ha^{-1} yr^{-1} are lost via leaching from a savanna woodland in Zimbabwe. Livestock urine has a high impact on NO_3^- leaching in pastoral systems (Di & Cameron, 2002), as high peaks of NO_3^- leaching occur under urine patches (Silva *et al.*, 1999). Although many N cycling studies mention leaching and erosion as N loss pathway, we could only find one study reporting on nitrate leaching. Moreover, we could not find studies reporting N losses through erosion in semiarid regions in SSA. Robertson & Rosswall, (1986b) estimated the combination of erosion, runoff and leaching in West Africa of 5.1 kg N ha^{-1} yr^{-1}. For our virtual farm in Kenya we assumed that 2 kg N ha^{-1} yr^{-1} were lost as N leaching (Rees *et al.*, 2006).

4.3.2.5 Nitrogen in animal products

The milk yield estimated for boran (i.e. *bos indicus*) cattle under pastoral conditions in Kenya is 518 kg TLU^{-1} yr^{-1} for the 7 months lactation period (Nicholson, 1984). The milk offtake (i.e. milk not consumed by the calf) for human consumption is estimated to be 20-25% from the total milk yield (Semenye & de Leeuw, 1986), resulting in 104-130 kg TLU^{-1} (lactation period)$^{-1}$ or 0.5-0.6 kg TLU^{-1} d^{-1} during the lactation period. There has been very little research in N partitioning into milk for African cattle breeds, especially for pastoral systems. We estimated the N exported as milk from our virtual farm using a N partition weight ratio of 5 g N kg $milk^{-1}$ calculated from Rufino *et al.*, (2006) for dairy cows in mixed cropped systems in Africa. Assuming 20-25% of milk offtake, we estimated that N losses through milk production is 2.5-3 gr N $TLU^{-1}d^{-1}$ during the lactation period, or 0.5 - 0.6 kg N TLU^{-1} yr^{-1}, and considering cattle density of 17.6 TLU km^{-2}

in our virtual farm, losses of N via export of animal products resulted in 0.1 kg N ha^{-1} yr^{-1}. This very low value was expected due to the extremely low milk production and cattle density in pastoral systems compared to other livestock systems in SSA.

3.4 Nitrogen losses from potential N emission hotspots: bomas and piospheres

Chronosequence studies have been used to explore the impact of bomas on soil nitrogen dynamics and vegetation biomass at landscape scale (Augustine *et al.*, 2003). Differences in soil nitrogen inside cattle bomas and its sorroundings in African pastoral systems were investigated in Kenya (Young *et al.*, (1995), Reid & Ellis, (1995), Muchiru *et al.*, (2009), Augustine, (2003) Porensky & Veblen, (2015)) and in South Africa (Valls Fox *et al.*, 2015). The boma chronosequence studies show that abandoned bomas supported vegetation species not found elsewhere, having a positive effect on vegetation diversity and higher soil nutrient levels compared to the surrounding area. For instance, Young (1995) measured the effect of abandoned bomas in Laikipia, Kenya and reported that the tree cover of some species (e.g. *Digitaria sp., Portulaca oleracea*) were up to 24 times higher in the bomas than in the surroundings, and soil nutrients content (i.e. nitrogen, carbon, sodium and calcium) were more than double in the abandoned bomas than surroundings. However, we could not find studies where the N flows in bomas in pastoral systems in SSA were studied to estimate its contribution to the N budget. Some research on N emission hotspots from animal enclosures has been done in comparable arid and semiarid ecosystems. For instance, NO and N_2O emissions and NH_3 air concentrations from sheepfolds in Inner Mongolia, China, were investigated (Liu *et al.*, 2009) (Holst *et al.*, 2007), finding that they are much higher in the enclosures than in the grazed steppe (i.e. three orders of magnitude higher). Thus, it is expected that bomas in pastoral systems in SSA are also hotspots for N gas emissions. The identification of these emission hotspots and their contribution to the N balance in pastoral systems will allow to assess a proper manure management that will derive on reduction of environmental and health impacts as well as fertilization benefits when properly used.

3.4.1. NH_3 volatilization, N_2 and N_2O emissions and NO_3^- leaching in bomas and piospheres

We aimed at estimating the overall contribution of hotspots (i.e. bomas and piospheres) to the total N budget of a pastoral livestock system. Our estimations resulted in 7728 ± 3864 kg N ha^-

$^{1}yr^{-1}$ for NH_3 and 309 ± 185 kg N ha^{-1} yr^{-1} for N_2O emitted from bomas, calculated using equation (2). Only a few authors studied other forms of N emissions (e.g. N_2) from accumulated manure, however all of them show that a significant percentage of N in manure is lost as N_2. Moral *et al.*, (2012) estimated, using the acetylene inhibition technique, that N_2 emissions from manure were more than 5 times higher than the N_2O emissions (5.2% and 1% of the initial N were emitted as N_2 and N_2O, respectively). However, due to the jar incubation method used where aerobiocity could be altered, this value probably differs from values under field conditions. Lee *et al.*, (2011) developed N isotope fractionation experiments to estimate the proportion of manure N lost as non-NH_3 gas, finding that 25% of the N losses were likely in the form of N_2. Jarvis and Pain (1994) estimated N_2 losses from manure as three times greater than those as N_2O from a dairy farm on a temperate grassland. This value was taken to calculate N_2 losses from the bomas in our hypothetical farm as a first approximation, estimated to be 927±556 kg N ha^{-1} yr^{-1}. Data on N runoff and leaching losses from manure management systems are extremely limited. We assumed that 6% of N excreted is lost through leaching during the wet season (Eghball & Power, 1994) in dry environments during the wet season, resulting in NO_3^- losses in bomas of 1022 kg N ha^{-1} yr^{-1} for our hypothetical farm, with N leaching considered 0 during dry periods in dry environments (Eghball & Power, 1994). Estimations of gaseous losses from piospheres in our virtual farm, assuming an affected area of 0.5 around each waterhole, were 680 ± 340 kg N ha^{-1} yr^{-1} as NH_3 volatilization, 27±16 as N_2O emissions and N_2 losses of 81±49 kg N ha^{-1} yr^{-1}. N leaching, erosion and runoff from piospheres resulted in 362 kg N ha^{-1} yr^{-1} (360 kg N ha-1 yr-1 leached from manure deposition as 6% of excreted N in rainy season plus 2 kg N ha-1 yr-1 losses from soils).

3.5 N balance in pastoral systems in SSA

Values from Table 5, were extrapolated to the source areas (i.e. piospheres 5 ha, bomas 1.8 ha and savanna soils 9993 ha) to estimate contributions of each N flux to total N inputs and losses in our hypothetical farm (Table 6).

Table 6. N fluxes, source area and contribution to the total N fluxes

N Flux	N flux values (kg N ha^{-1} yr^{-1})	Area	N inputs/outputs (kg N yr^{-1})	%
BNF	16.4	9993	163885.2	84.1
Deposition	3.1	10000	31000	15.9
N supplements	0	10000	0	0.0
NH_3 savanna	1.7	9993	16988.1	22.9
N_2O savanna	0.25	9993	2498.25	3.4
N_2 savanna	0.5	9993	4996.5	6.7
Nox savanna	0.5	9993	4996.5	6.7
Leaching and runoff savanna	2	9993	19986	26.9
Milk offtake	0.1	10000	1000	1.3
NH3 bomas	7728	1.8	13910.4	18.7
N2O bomas	309	1.8	556.2	0.7
N2 bomas	927	1.8	1668.6	2.2
Leaching bomas	1022	1.8	1839.6	2.5
NH3 piospheres	680	5	3400	4.6
N2O piospheres	27	5	135	0.2
N2 piospheres	81	5	405	0.5
Nox piospheres	0.5	5	2.5	0.0
Leaching piospheres	362	5	1810	2.4

N losses on our hypothetical farm were dominated by biological N fixation, estimated to be more than 80% of the N inputs. Major N losses occur via N leaching contributing to 27% of the N losses from the system, followed by NH_3 volatilization from the savanna (23%) and NH_3 volatilization from the bomas, contributing to 19% of the N losses. Most N balances in livestock systems in SSA in the literature (i.e. mixed cropped systems/agro-pastoral systems) are negative (Cobo *et al.*, 2010), i.e. indicating net losses of N from the system. However, results in this study suggest that mining is not necessarily happening in SSA pastoral systems soils. This difference could be attributed to much higher rates of BNF in savanna (symbiotic and algal fixation) (Robertson & Rosswall, 1986b), which seem to be enough to at least balance N losses from the system, and the absence of crops mining the soils. The N balance could be smaller if wildlife (2 kg N ha^{-1} yr-^{1}, (Ruess & McNaughton, 1988)) and occasional fires N losses (1.5±0.3 kg N ha^{-1} yr^{-1}, (Galy-Lacaux & Delon, 2014)) were included, although we calculated that the resulting N balance reduction is relatively small (20%), and surely not enough to turn the N balance negative for those rates of BNF and cattle density.

For Wildlife, from Ruess & McNaughton: The authors observed, using closed-chamber systems with acid gas traps, that the volatilization process started immediately after urine application and reached a peak after 24 hours. NH_3 emissions continued for another 48 hours before decreasing exponentially. They calculated NH_3 losses via volatilization from wildebeest assuming a density of 800 animals km^{-2} 150 d^{-1}. Peak NH_3 loss rates due to urine deposition were estimated to range from 0.71 ± 0.18 to 4.04 ± 0.4 kg N ha^{-1} yr^{-1}, with lowest volatilization rates on highly grazed areas and highest rates on less heavily grazed grasslands.

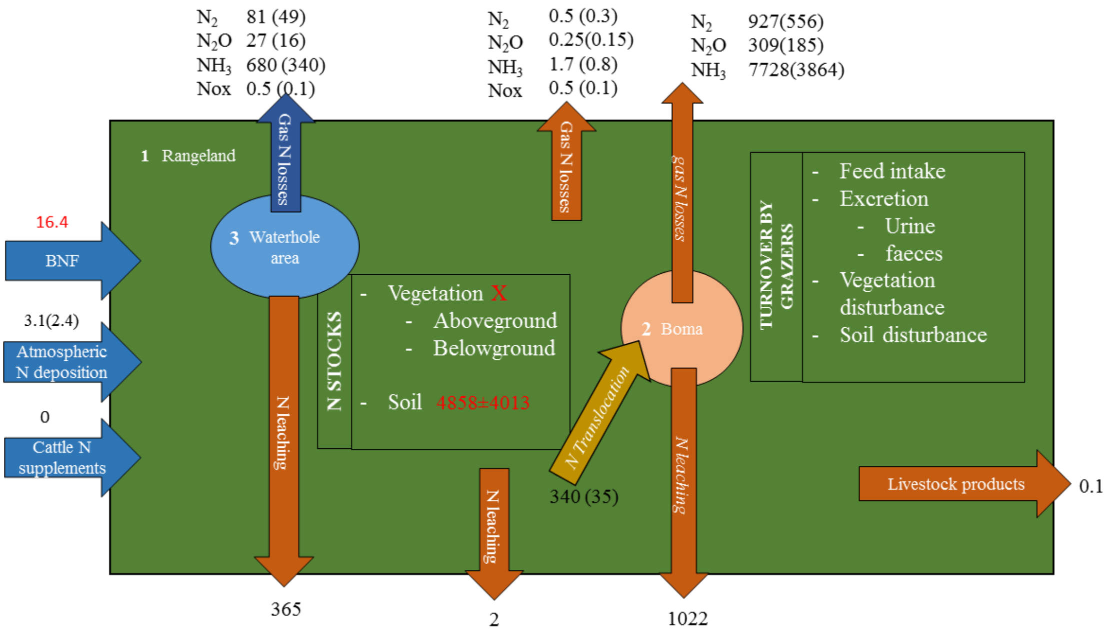

Figure 6. N flows calculated for a pastoral farm of 10000 ha. Fluxes are given in kg ha^{-1} yr^{-1}, N stocks in kg N ha^{-1} and Intake/excretion rates in kg N TLU^{-1} yr^{-1}

3.6. Conclusions

This review presents for the first time a N budget for a typical pastoral livestock systems in sub-Saharan Africa. Our results show a positive N budget, suggesting that not all livestock systems in SSA cause soil N mining at system scale. Bomas, were identified as hotspots for N emissions, with N emissions three orders of magnitude higher than the surrounding landscape for a stocking rates of 17.6 TLU ha^{-1}. Overall NH_3 emissions dominated N losses from pastoral systems sharing 50% of the total N losses. Although our results show that piospheres can be considered N emissions hotspots, as emissions from these areas are one order of magnitude larger than the surrounding landscape, the upscaled N fluxes to the farm area show that the surrounding savanna soils emit larger N fluxes. Another large N loss pathway found is nitrate leaching. In the bomas, N loss via leaching is around 200 times larger than in savanna soils. However, N leaching from savanna soils, represents almost 30% of the total N loss from the system. Biological N fixation was the largest N flux in the system, representing around 85% of the total N input. Despite uncertainties on N fluxes estimations, N inputs through BNF seem to originate positive N balances in pastoral farms.

The main gaps found on N cycling on pastoral systems in SSA were the exclusion of N fluxes for N balance calculations (i.e. partial N balances) and high uncertainties associated to N flux calculations. Livestock-related N flows were often excluded from N balance calculations, and N emission hotspots, such as bomas and waterholes have so far not been considered. N fluxes that present difficulties to be measured, such as N_2, are excluded from N balance studies. Furthermore, studies often do not cover spatial and temporal variations, despite representing crucial aspects of LPS in arid and semiarid SSA. High uncertainties of N balance calculations in pastoral systems in SSA are derived from the climatic conditions in arid and semiarid regions (not comparable to temperate regions, where more research has been done), from stocking rates present as well as tree density variability across arid and semiarid savannas in SSA. To reduce uncertainties, more experimental data on N fluxes are needed to understand the behaviour of the different interrelated compartments integrating these LPS. Particularly in-situ data such as field measurements of NH_3 and other N flows will allow to estimate the contribution of N losses via NH_3 volatilization. N balances, as useful tools for LPS management assessments in Africa, should be prioritized towards currently existing gaps as well as to propose reliable GHG mitigation strategies (ie. soil C sequestration). Furthermore, and due to the potential intensification of these LPS in SSA in the future as a result of increase of livestock products demand, stock rates might increase to the point where N outputs of the system will be much higher than inputs. This might cause large rates of NH_3 emissions and NO_3^- leaching and leading to an important increase in eutrophication of terrestrial and aquatic ecosystems as well as acidification of SSA savanna soils, if LPS management assessments are not implemented. Based on the knowns and unknowns highlighted in this review, researchers are enabled to aim future research towards closing knowledge gaps in N cycling in LPS in SSA, and pastoral systems in particular, to improve N balance estimations as indicators of land degradation for further management assessment.

Literature

Adon M, Galy-Lacaux C, Yoboué V et al. (2010) Long term measurements of sulfur dioxide, nitrogen dioxide, ammonia, nitric acid and ozone in Africa using passive samplers. *Atmospheric Chemistry and Physics*, **10**, 7467–7487.

Anderson N, Strader R, Davidson C (2003) Airborne reduced nitrogen: ammonia emissions from agriculture and other sources. *Environment International*, **29**, 277–286.

Andrew M. (1988) Grazing Impact in Relation Watering Points Livestock. *Trends in Ecology & Evolution*, **3**, 336–339.

Augustine DJ (2003) Long-term, livestock-mediated redistribution of nitrogen and phosphorus in an East African savanna. *Journal of Applied Ecology*, **40**, 137–149.

Augustine DJ, McNaughton SJ, Frank DA (2003) Feedbacks between soil nutrients and large herbivores in a managed savanna ecosystem. *Ecological Applications*, **13**, 1325–1337.

Barrett CB, Chabari F, Bailey D, Little PD, Coppock DL (2003) Livestock pricing in the Northern Kenyan rangelands. *Journal of African Economies*, **12**, 127–155.

Bate GC (1981) Nitrogen cycling in savanna ecosystems. *Ecological Bulletins (Sweden).*

Bernhard-Reversat F (1982) Biogeochemical cycle of nitrogen in a semi-arid savanna. *Oikos*, 321–332.

Bernhard-Reversat F, Poupon H (1980) Nitrogen cycling in a soil tree system in a Sahelian savanna: Example of Acacia senegal. In: *Proceedings of a workshop arranged by the SCOPE/UNEP*, pp. 363–369.

Bosch H Van Den, Jager a De, Vlaming J (1998) Monitoring nutrient ¯ ows and economic performance in African farming systems (NUTMON) II . Tool development. *Agriculture, Ecosystems and Environment*, **71**, 49–62.

Bowden WB (1986) Gaseous nitrogen Emissions from undisturbed terrestrial ecosystems: an assessment of their impacts on local and global nitrogen budgets. *Biogeochemistry*, **2**, 249–279.

Brümmer C, Brüggemann N, Butterbach-Bahl K et al. (2008) Soil-atmosphere exchange of N2O and NO in near-natural savanna and agricultural land in Burkina Faso (W. Africa). *Ecosystems*, **11**, 582–600.

Bustamante MMC, Medina E, Asner GP, Nardoto GB, Garcia-Montiel DC (2006) Nitrogen cycling in tropical and temperate savannas. *Biogeochemistry*, **79**, 209–237.

Butterbach-Bahl K, Baggs EM, Dannenmann M, Kiese R, Zechmeister-Boltenstern S (2013) Nitrous oxide emissions from soils: how well do we understand the processes and their controls? *Philosophical transactions of the Royal Society of London. Series B, Biological sciences*, **368**, 20130122.

Carran RA, Clough T (1996) Environmental impacts of nitrogen in pastoral agriculture. *SPECIAL PUBLICATION-AGRONOMY SOCIETY OF NEW ZEALAND*, 99–102.

Cech PG, Venterink HO, Edwards PJ (2010) N and P Cycling in Tanzanian Humid Savanna: Influence of Herbivores, Fire, and N2-Fixation. *Ecosystems*, **13**, 1079–1096.

Chen Y, Randerson JT, Van der Werf GR, Morton DC, Mu M, Kasibhatlas PS (2010) Nitrogen deposition in tropical forests from savanna and deforestation fires. *Global Change Biology*, **16(7)**, 2024–2038.

Cleveland CC, Townsend AR, Schimel DS et al. (1999) Global patterns of terrestrial biological nitrogen (Nz) fixation in natural ecosystems. **13**, 623–645.

Cobo JG, Dercon G, Cadisch G (2010) Nutrient balances in African land use systems across different spatial scales: A review of approaches, challenges and progress. *Agriculture, Ecosystems and Environment*, **136**, 1–15.

Coetsee C, Jacobs S, Govender N (2012) An overview of nitrogen cycling in a semiarid savanna:

Some implications for management and conservation in a large African park. *Environmental Management*, **49**, 387–402.

D . J . Tolsma W. H. O. E and R. A. V (2018) Nutrients in Soil and Vegetation Around Two Artificial Waterpoints in Eastern Botswana. **24**, 991–1000.

Delon C, Galy-Lacaux C, Boone a. et al. (2009) Atmospheric nitrogen budget in Sahelian dry savannas. *Atmospheric Chemistry and Physics Discussions*, **9**, 14189–14233.

Delon C, Adon M, Liousse C, Serc D (2012) Nitrogen compounds emission and deposition in West African ecosystems : comparison between wet and dry savanna. 385–402.

Delon C, Galy-lacaux C, Serça D et al. (2017) Soil and vegetation-atmosphere exchange of NO , NH 3 , and N 2 O from fi eld measurements in a semi arid grazed ecosystem in Senegal. *Atmospheric Environment*, **156**, 36–51.

Di HJ, Cameron KC (2002) Nitrate leaching in temperate agroecosystems: Sources, factors and mitigating strategies. *Nutrient Cycling in Agroecosystems*, **64**, 237–256.

Eghball B, Power JF (1994) Beef cattle feedlot manure management. *Journal of Soil and Water Conservation*, **49**, 113–122.

Eghball B, Power JF, Gilley JE, Doran JW (1997) Nutrient, carbon, and mass loss during composting of beef cattle feedlot manure. *Journal of environmental quality*, **26**, 189–193.

Ellertl BH, Bettany JR (1995) Calculation of organic matter and nutrients stored in soils under contrasting management regimes.

Fowler D, Coyle M, Skiba U et al. (2013) The global nitrogen cycle in the twenty-first century The global nitrogen cycle in the twenty- first century.

Galy-Lacaux C, Delon C (2014) Nitrogen emission and deposition budget in West and Central Africa. *Environmental Research Letters*, **9**, 125002.

Gray A, Odee D, Cavers S et al. (2013) Does geographic origin dictate ecological strategies in

Acacia senegal (L.) Willd.? Evidence from carbon and nitrogen stable isotopes. *Plant and soil*, **369**, 479–496.

Hamza MA, Anderson WK (2005) Soil compaction in cropping systems A review of the nature , causes and possible solutions. **82**, 121–145.

Holst J, Liu C, Yao Z, Brüggemann N, Zheng X, Han X, Butterbach-Bahl K (2007) Importance of point sources on regional nitrous oxide fluxes in semi-arid steppe of Inner Mongolia, China. *Plant and soil*, **296**, 209–226.

Holt JA, Division RJC (2018) Nutrient Cycling in Australian Savannas Author (s): J . A . Holt and R . J . Coventry Australian Perspectives and Intercontinental Comparisons (Jul . - Sep ., 1990), pp . 427-432 Published by: Wiley Stable URL: https://www.jstor.org/stable/2845373 N. **17**, 427–432.

Houlton BZ, Wang Y-P, Vitousek PM, Field CB (2008) A unifying framework for dinitrogen fixation in the terrestrial biosphere. *Nature*, **454**, 327–30.

Huntley BJ, Walker BH (2012) *Ecology of tropical savannas*, Vol. 42. Springer Science & Business Media.

Knox NM, Skidmore AK, Prins HHT et al. (2011) Dry season mapping of savanna forage quality, using the hyperspectral Carnegie Airborne Observatory sensor. *Remote sensing of environment*, **115**, 1478–1488.

Lee C, Hristov AN, Cassidy T, Heyler K (2011) Nitrogen isotope fractionation and origin of ammonia nitrogen volatilized from cattle manure in simulated storage. *Atmosphere*, **2**, 256–270.

Leenaars JGB (2013) *Africa Soil Profiles Database, Version 1.1. A compilation of georeferenced and standardised legacy soil profile data for Sub-Saharan Africa (with dataset). Africa Soil Information Service (AfSIS) project.* ISRIC-World Soil Information.

Lesschen JP, Stoorvogel JJ, Smaling EMA, Heuvelink GBM, Veldkamp A (2007) A spatially explicit methodology to quantify soil nutrient balances and their uncertainties at the national level. *Nutrient Cycling in Agroecosystems*, **78**, 111–131.

Levine JS, Winstead EL, Parsons DAB et al. (1996a) Biogenic soil emissions of nitric oxide (NO) and nitrous oxide (N2O) from savannas in South Africa: the impact of wetting and burning. *Journal of Geophysical Research: Atmospheres*, **101**, 23689–23697.

Levine JS, Winstead EL, Parsons DAB, Scholes MC, Scholes RJ, Iii WRC, Jr DRC (1996b) Biogenic soil emissions of nitric oxide (NO) and nitrous oxide (NzO) from savannas in South Africa: The impact of wetting and burning. **101**, 689–697.

Liu C, Holst J, Brüggemann N et al. (2007) Winter-grazing reduces methane uptake by soils of a typical semi-arid steppe in Inner Mongolia, China. *Atmospheric Environment*, **41**, 5948–5958.

Liu C, Holst J, Yao Z et al. (2009) Sheepfolds as “hotspots” of nitric oxide (NO) emission in an Inner Mongolian steppe. *Agriculture, ecosystems & environment*, **134**, 136–142.

Lulla K (1987) SAVANNA CLIMATESavanna climate. In: *Climatology*, pp. 748–749. Springer US, Boston, MA.

MacCarthy N (2000) *Property rights, risk, and livestock development in Africa*. IFPRI.

McCown R, Jones R (1992) A Search for Strategies for Sustainable Dryland Cropping in Semi-arid Eastern Kenya. *Proceedings*.

Meixner FX, Fickinger T, Marufu L et al. (1997a) Preliminary results on nitric oxide emission from a southern African savanna ecosystem. *Nutrient Cycling in Agroecosystems*, **48**, 123–138.

Meixner FX, Fickinger T, L. Marufu DS, Nathaus FJ, Makina E, Mukurumbira L, Andreae MO (1997b) Preliminary results on nitric oxide emission from a southern African savanna ecosystem. *Nutrient Cycling in Agroecosystems*, **48**, 123–138.

Moral R, Bustamante MA, Chadwick DR, Camp V, Misselbrook TH (2012) N and C transformations in stored cattle farmyard manure, including direct estimates of N2 emission. *Resources, Conservation and Recycling*, **63**, 35–42.

Muchiru AN, Western D, Reid RS (2009) The impact of abandoned pastoral settlements on plant and nutrient succession in an African savanna ecosystem. *Journal of Arid Environments*, **73**, 322–331.

Ndoye I, Gueye M, Danso SKA, Dreyfus B (1995) Nitrogen fixation in Faidherbia albida, Acacia raddiana, Acacia senegal and Acacia seyal estimated using the 15N isotope dilution technique. *Plant and Soil*, **172**, 175–180.

Nelson GC (2009) *Agriculture and climate change: An agenda for negotiation in Copenhagen*, Vol. 16. Intl Food Policy Res Inst.

Ng'ethe JC (1993) Group ranch concept and practice in Kenya with special emphasis on Kajiado District. In: *Future of Livestock Industries in East and Southern Africa. Proceedings of the Workshop Held at the Kadoma ranch Hotel, Zimbabwe, 20–23 July 1992*, pp. 187–200.

Nicholson MJ (1984) Pastoralism and milk production.

Oenema O, Velthof GL, Yamulki S, Jarvis SC (1997) Nitrous oxide emissions from grazed grassland. *Soil Use and Management*, **13**, 288–295.

Otter LB, Yang WX, Scholes MC, Meixner FX (1999) Nitric oxide emissions from a southern African savanna. *Journal of Geophysical Research: Atmospheres*, **104**, 18471–18485.

Pierre J. Gerber BH and HPSM (2014) *Mitigation of greenhouse gas emissions in livestock production*.

Porensky LM, Veblen KE (2015) Generation of ecosystem hotspots using short-term cattle corrals in an African savanna. *Rangeland Ecology and Management*, **68**, 131–141.

Ravishankara AR, Daniel JS, Portmann RW (2009) Nitrous oxide (N2O): the dominant ozone-depleting substance emitted in the 21st century. *science*, **326**, 123–125.

Rees RM, Wuta M, Furley PA, Li C (2006) Nitrous oxide fluxes from savanna (miombo) woodlands in Zimbabwe. *Journal of Biogeography*, **33**, 424–437.

Reid RS, Ellis JE (1995) Impacts of Pastoralists on Woodlands in South Turkana, Kenya: Livestock-Mediated Tree Recruitment. *Ecological Applications*, **5**, 978–992.

Robertson GP, Rosswall T (1986a) Nitrogen in West Africa: the regional cycle. *Ecological Monographs*, **56**, 43–72.

Robertson GP, Rosswall T (1986b) Nitrogen in West Africa: the Regional Cycle. *Ecological Monographs*, **56**, 43–72.

Robinson T, Thornton P, Franceschini G et al. (2011) *Global livestock production systems.* Food and Agriculture Organization of the United Nations (FAO).

Ruess RW, McNaughton SJ (1988) Ammonia volatilization and the effects of large grazing mammals on nutrient loss from East African grasslands. *Oecologia*, **77**, 382–386.

Rufino MC, Rowe EC, Delve RJ, Giller KE (2006) Nitrogen cycling efficiencies through resource-poor African crop-livestock systems. *Agriculture, Ecosystems and Environment*, **112**, 261–282.

Schlecht E, Hiernaux P (2004) Livestock related nutrient budgets within village territories in western Niger.

Schlecht E, Hiernaux P (2005) Beyond adding up inputs and outputs: Process assessment and upscaling in modelling nutrient flows. *Nutrient Cycling in Agroecosystems*, **70**, 303–319.

Schlecht E, Mahler F, Sangaré M, Susenbeth A, Becker K (1995) Quantitative and qualitative estimation of nutrient intake and faecal excretion of zebu cattle grazing natural pasture in semi-arid Mali. In: *International Conference on Livestock and Sustainable Nutrient Cycling in Mixed Farming Systems of Sub-Saharan Africa, Addis Ababa (Ethiopia), 22-26 Nov 1993*. ILCA.

Scholes M, Andreae M (2000) Biogenic and pyrogenic emissions from Africa and their impact on the global atmosphere. *AMBIO: A Journal of the Human Environment*, **29**, 23–29.

Scholes MC, Martin R, Scholes RJ, Parsons D, Winstead E (1997) NO and N2O emissions from savanna soils following the first simulated rains of the season. *Nutrient Cycling in Agroecosystems*, **48**, 115–122.

Semenye PP, de Leeuw PN (1986) Milk for calves or man in a Maasai pastoral system in East Africa.

Seré C, Steinfeld H (1996) World livestock production systems: Current status, issues and trends. *Animal Production and Health Paper*, 82.

Sheppard SC, Bittman S (2011) Farm survey used to guide estimates of nitrogen intake and ammonia emissions for beef cattle, including early season grazing and piosphere effects. *Animal feed science and technology*, **166**, 688–698.

Silva RG, Cameron KC, Di HJ, Hendry T (1999) A lysimeter study of the impact of cow urine, dairy shed euent, and nitrogen fertiliser on nitrate leaching. *Soil Research*, **37**, 357–370.

Smet MÃ, Ward D (2006) Soil quality gradients around water-points under different management systems in a semi-arid savanna , South Africa. **64**, 251–269.

Steinfeld H, Gerber P, Wassenaar TD, Castel V, de Haan C (2006) *Livestock's long shadow: environmental issues and options*. Food & Agriculture Org.

Stroebel A, Swanepoel FJC, Nthakheni ND, Nesamvuni AE, Taylor G (2008) Benefits obtained from cattle by smallholder farmers: A case study of Limpopo Province, South Africa. *Australian Journal of Experimental Agriculture*, **48**, 825–828.

Swanepoel F, Stroebel a., Moyo S (2008) *The role of livestock in developong communities: Enhancing multifunctionality*. 1-24 pp.

Thornton PK (2010) Livestock production: recent trends, future prospects. *Philosophical Transactions of the Royal Society B: Biological Sciences*, **365**, 2853–2867.

Thornton PK, van de Steeg J, Notenbaert A, Herrero M (2009) The impacts of climate change on livestock and livestock systems in developing countries: A review of what we know and what we need to know. *Agricultural Systems*, **101**, 113–127.

Valls Fox H, Bonnet O, Cromsigt JPGM, Fritz H, Shrader AM (2015) Legacy Effects of Different Land-Use Histories Interact with Current Grazing Patterns to Determine Grazing Lawn Soil Properties. *Ecosystems*, **18**, 720–733.

Wang L, Katjiua M, D'Odorico P, Okin GS (2012) The interactive nutrient and water effects on vegetation biomass at two A frican savannah sites with different mean annual precipitation. *African Journal of Ecology*, **50**, 446–454.

Whalen JK, Chang C, Clayton GW, Carefoot JP (2000) Cattle manure amendments can increase the pH of acid soils. *Soil Science Society of America Journal*, **64**, 962–966.

Young TP, Patridge N, Macrae A (1995) Long-Term Glades in Acacia Bushland and Their Edge Effects in Laikipia, Kenya. *Ecological Applications*, **5**, 97–108.

Young P. PN and MA (1995) Long-Term Glades in Acacia Bushland and Their Edge Effects in Laikipia , Kenya Author (s): Truman P . Young , Nathaniel Patridge and Alison Macrae Published by : Wiley Stable URL : http://www.jstor.org/stable/1942055 REFERENCES Linked references are ava. *Ecological applications*, **5**, 97–108.

Zinke, P. J., A. G. Stangenberger, W. M. Post, W. R. Emanuel, and J. S. Olson (1998) Global Organic Soil Carbon and Nitrogen (Zinke et al.). Data set. Available on-line [http://www.daac.ornl.gov] from Oak Ridge National Laboratory Distributed Active Archive Center, Oak Ridge, Tennessee, U.S.A. Previously published in Worldwide Organic Soil Carbon and Nitrogen Data, CDIAC NDP-018, Carbon Dioxide Information and Analysis Center, Oak Ridge National Laboratory, Oak Ridge, Tennessee, U.S.A., 1986

Annex

ACRONYMS

AEZ. Agro-ecological zones

ANPP. Above ground Net Primary Production

ASAL. Arid and semi-arid lands

BNF. Biological Nitrogen Fixation

DON. Dissolved organic matter

GHG. Greenhouse gas

IMAGE. Integrated Model to Assess the Global Environment

LGA. Arid/semi-arid rangeland-based system.

LPS. Livestock Production Systems

LU. Livestock Unit

LUC. Land use change

N. Nitrogen

Nr. Reactive nitrogen

N_2O. Nitrous Oxide

NO. Nitric Oxide

NO_2. Nitrogen dioxide

P. Phosphorous

PM. Particulate matter

SOM. Soil organic matter

SSA. Sub-Saharan Africa

TLS. Traditional Livestock Systems

DEFINITIONS

Manure. Sheldrick et al (2003) defined manure as "part of livestock excreta (i.e. urine and faeces) collected for discretionary use, including on-the-spot excreta produced in the field by livestock feeding on crop residues".

Livestock production system (LPS) defined by Seré & Steinfeld (1996) as "a subset of the farming systems, including cases in which livestock contribute more than 10 % to total farm output in value terms or where intermediate contributions such as animal traction or manure. [3]**Livestock Grassland**

Arid and Semi-arid Tropics and Sub-tropics System (LGA) is the LPS classification by (Seré & Steinfeld, 1996) for the solely livestock systems where >10% of the dry matter fed to animals is farm produced and have stocking rates of an annual average of <10 livestock units (LU) per hectare in arid and semiarid areas.

GIS TOOLS

Software: QGIS v2.18.13. Used to determine soil profiles-livestock systems classification overlapping.

Koppen maps: http://koeppen-geiger.vu-wien.ac.at/shifts.htm

ZINKE database: http://daac.ornl.gov/daacdata/global_soil/ZinkeSoil/data/zinke_soil.txt

ISRIC-WISE database: https://daac.ornl.gov/SOILS/guides/Isric.html

AfSIS database: http://www.isric.org/data/africa-soil-profiles-database-version-01-1