Contrasted seasonal balances in a Sahelian pastoral ecosystem result in a neutral annual carbon balance

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Abstract

This paper explores an original approach in which greenhouse gas (GHG) emissions and carbon (C) accumulation are assessed monthly and at landscape scale to account for the highly seasonal monsoon climate and the mobility of pastoral herds that characterize West African pastoral ecosystems.

The study was conducted in northern Senegal, in the service area of the Widou Thiengoly borehole, a circular zone of 706 km² centered on the borehole. The C balance was calculated

using an ecosystem approach, i.e. taking all main sources of GHG emissions and C sinks of the ecosystem, not only anthropogenic sources, into account.

The annual C balance of the pastoral ecosystem was -0.04±0.01 tC-eq.ha^{-1}.year^{-1}, showing that total GHG emissions were mitigated by C accumulation in trees, soil and livestock. The C balance varied considerably with the seasons, with a positive monthly balance in the wet season, from July to October (+0.58 tC-eq.ha^{-1}.month^{-1}) and a negative monthly balance in the cold dry season from November to February and the hot dry season from March to June (-0.57 and -0.05 tC-eq.ha^{-1}.month^{-1} respectively).

Care should be taken when generalizing these results, which were obtained in a dry year, because of strong inter-annual variations in rainfall.

Keywords: Ecosystem functioning, GHG emissions, C sequestration, Landscape, Senegal

1. Introduction

The contribution of the world's livestock sector to global anthropogenic greenhouse gas (GHG) emissions is estimated to be 14.5% (Gerber et al., 2013). The environmental impact of intensive versus extensive livestock systems is a matter of debate in the scientific community. In sub-Saharan Africa, extensive pastoral systems are assumed to be responsible for the highest rates of GHG emissions per unit of animal product despite their modest contribution to global GHG emissions (Gerber et al., 2013). The main reasons are the low productivity of the herds and the high methanogenic potential of forage ingested by grazing livestock especially during the long dry season. Pastoral rangelands account for more than 25% of terrestrial ecosystems and about 40% of Africa's land area, where semi-arid rangelands dominate (Herrero et al., 2016; Tagesson et al., 2015a). The semi-arid tropics in western and central Africa are characterized by contrasted seasonal conditions with rainfall limited to a

short summer season, resulting in highly seasonal forage availability and quality. In response, pastoralism, the dominant economic activity, is based on seasonal livestock mobility (Turner et al., 2014). Rainfall drives rangeland ecosystem functioning (i) by stimulating the growth of herbaceous vegetation providing highly digestible green forage that insures the annual growth and the reproduction of grazing animals (Chirat et al., 2014), and (ii) by activating the soil biological activity responsible for the decomposition and mineralization of organic matter, and hence the recycling of N, P and other nutrients (Delon et al., 2015). N and P availability to plants is the main factor that limits ecosystem production (Penning de Vries and Djiteye, 1982). Consequently, rangeland production is particularly vulnerable to seasonal variations in rainfall amounts and distribution patterns (Nassef et al., 2009).

Pastoralism is an extensive livestock production system with sophisticated herd management based on selective grazing of communal rangelands. Herd management is particularly difficult in the dry season when drinking water and forage availability and quality are limited (McGahey et al., 2014). In pastoral ecosystems, livestock, mostly ruminants, play a major role in speeding up C, N, P and K recycling. Livestock digest grazed fodder largely composed of cellulose that would otherwise degrade slowly (Petersen et al., 2013). The nondigested fraction is excreted as N rich urine and feces that represent a source of rich and easily degradable organic material for meso-fauna and micro-organisms to mineralize, providing the soil with nutrients, especially the limiting N and P, as well as stable soil organic C, which enhances soil biochemical fertility (Whitmore, 2001). The C and N cycles both strongly influence GHG emissions and C sequestration (Soussana et al., 2004). It is thus likely that the strong seasonality of rainfall distribution and of the resulting grazing resource are responsible for the marked seasonal variations in GHG emissions and C sequestration.

The aim of this study was to produce a dynamic seasonal representation of the C balance by assessing key stocks and fluxes of the C and N cycles in a Sahel pastoral ecosystem on a

monthly basis, whereas the C balances of agricultural systems are usually assessed on an annual basis and seasonal variations are not taken into account. In the present study, stocks and fluxes were assessed through monthly *in situ* observations and measurements made over a full year in a single territory, the Widou Thiengoly borehole service area (WTSA). The WTSA is located in the Ferlo, a Sahelian pastoral region in northern Senegal. The sampling strategy was also designed to account for landscape heterogeneity. Such field data are new in the Sahel. This explains why the literature on the C balance of pastoral systems in sub-Saharan Africa is poor. What is more, most of the C balances available for tropical agro-ecosystems use default emission factors provided by IPCC (see for instance Doran-Browne et al. (2016)), with large uncertainties on these default emission factors due to the lack of *in situ* measurements in tropical systems (Rosenstock et al., 2013). Most *in situ* observation-based C balances of African ecosystems use data provided by the regional network of flux towers (Valentini et al., 2014). These studies are based on temporally intense but spatially limited measurements that do not account for the marked spatial heterogeneity of GHG emissions induced by livestock mobility (Assouma et al., 2017). Another originality of this study is the nature of GHG sources taken into account in the assessment. The C balance in agriculture is usually calculated using two different approaches, life cycle analysis (LCA) and GHG inventories, both of which only consider anthropogenic sources of GHG emissions. LCA is used to assess the environmental impact of a product by calculating all the GHG emissions that occur throughout the life cycle of a product, and the assessment is made at the supply chain level (Gerber et al., 2013). GHG inventories are used to assess the environmental impact of human activities within a system and are performed at the country, territory or farm level (Rakotovao et al., 2017). In this study, we designed an original approach, called the "ecosystem C balance", that accounts for all the main sources of GHG emissions and C accumulation in the ecosystem, not only anthropogenic sources. We developed this approach

because some studies demonstrated the ability of permanent grasslands under temperate (Soussana et al., 2010) or tropical conditions (Doran-Browne et al., 2016) to mitigate anthropogenic GHG emissions and we wanted to go further by taking into account all main sources of GHG emissions and C sinks of the ecosystem, to find out if the ecosystem is in equilibrium or not. The ecosystem approach is particularly suitable when pastoral ecosystems are extensive, low-input-systems, mainly based on natural resources, close to natural ecosystems. The ecosystem C balance makes it possible to assimilate the complexity of the ecosystem and to understand the interactions between the components of the ecosystem (animal-soil-vegetation and atmosphere) to explain the C balance and its seasonal and between-year dynamics.

2. Material and methods

2.1 The Sahelian pastoral system at Widou Thiengoly

The Ferlo region (70,000 km^2) is located between latitudes 15° and 16° 30' N and longitudes 13°30' and 16° W and includes a large fraction of the Senegalese pastoral zone. The climate of the region is semi-arid monsoon of Sahelian type characterized by the alternation of two seasons: a 4-month wet season (July to October) and an 8-month dry season (November to June). The dry season is subdivided into two 4-month seasons, the first being the cold dry season from November to February and the second being the hot dry season from March to June. The main economy of families in the Ferlo region is pastoral husbandry, characterized by shared access to communal grazing lands, grazing resources (herbaceous plants plus browsing the foliage, twigs and fruits of woody plants) and drinking water resources. Prior to the 1950s, the Ferlo was only grazed during the rainy and early dry seasons by Fulani transhumance herders. In the dry season, due to lack of accessible water, the pastoralists had

to move either northwards to the floodplain rangelands of the Senegal River valley or southwards to the Sérère croplands or the sub-humid savannas located further south and east. From the 1950s onwards, deep boreholes with mechanical pumps to tap the Maestrichian water table were drilled at regular intervals every 20 to 30 km. Since then, transhumant pastoralists have settled in the Ferlo, in family camps often set up in the vicinity of temporary ponds but nevertheless scattered throughout the service area of the boreholes. Our case study was the service area (WTSA) of the Widou Thiengoly borehole (15° 59' N, 15° 19' W), a circular territory centered on the borehole with a radius of usually 15 km, i.e. half the mean distance between neighboring boreholes. The WTSA extends over 706 km², hosts 354 pastoral camps and a population of 4,800 inhabitants. The total livestock population of the WTSA ranged from 15 274 to 33 095 TLU in August and December, respectively. The average livestock population was composed of cattle (54% of total TLU), sheep (27%), goats (8%), horses (3%) and donkeys (8%). The WTSA was subdivided into six landscape units based on land use, the type of vegetation and soils (Assouma et al., 2017). Landsat image TM 204-049 dated November 3, 2010 and field observations (GPS points to mark the boundaries of the units and identify particular points) were used to map the six landscape units:

- Grazing lands (635.45 km², 89.9% of the WTSA area), sparsely wooded savannas,
- Settlements (44.46 km², 6.3%), the 354 settlements include the pastoralists dwellings, night corrals and the day resting spots for their livestock,
- Forest plantations (6.23 km², 0.9%), include the five most recent forest plantations established since 2005 by the Senegalese government as part of the Great Green Wall project. These forest are fenced and are only accessible to livestock three years after planting,
- Natural ponds (19.34 km², 2.7%), 67 ponds located in the low-lying areas of inter-dune depressions that harbor a string of shallow ponds during the wet season and are

the main sources of water for the herds in the wet season and in the early months of the dry season,

- The borehole (0.78 km², or 0.1%) is the only source that can be used water the herds when the ponds have dried up, and is visited once a day or every two days by most animals (some small ruminants are watered in the camp with water in tanks transported by donkey carts from the borehole) and pastoralist families for at least six months of the year,
- Enclosures (0.24 km², 0.03%), six small fenced experimental plots that were set up as part of a formal research project in 1981, to assess the long term effect of conservation without grazing or burning.

2.2 Conceptual model and the components of the C balance

The main C and N stocks and flows between stocks in the pastoral ecosystem and GHG fluxes to the atmosphere (CO_2, CH_4, N_2O) were used in the conceptual model (Figure 2).

The conceptual model was designed using data in the literature to identify GHG emissions and the sources and sinks of C that have to be taken into account in the C balance. According to Gerber et al. (2013), extensive livestock systems in sub-Sahara Africa are responsible for high rates of CH_4 emissions due to enteric fermentation. Termites are one of the main natural sources of CH_4 in tropical savannas through the digestion of cellulose by the micro-organisms living in their guts (Jamali et al., 2011).The deposition of livestock excreta on rangeland soils increases N_2O emissions from the soil (Pelster et al., 2016). In the Sahelian pastoral ecosystem, Assouma et al. (2017) also pointed to CH_4 emissions from standing water in ponds and in the vicinity of the borehole due to organic matter directly excreted by livestock during watering or resting and drained by runoff from the vicinity. Bush fires are also known to be a source of GHG emissions in pastoral ecosystems, especially in tropical and subtropical Africa

175 (Devineau et al., 2010). The main C sinks usually considered in the C balance of agro-
176 ecosystems are vegetation and soils (Soussana et al., 2004). However, the herbaceous mass is
177 not considered as a C sink because the herbaceous vegetation of Sahelian rangelands only
178 comprises annuals (Hiernaux and Le Houerou, 2006), the herbage is thus transient and does
179 not insure long term accumulation of C. Only woody plants contribute to year to year C
180 accumulation in wood and roots. The leaves, flowers, fruits, part of twigs and branches and
181 roots of woody plants are transient, like those of herbaceous plants, and are not considered as
182 C sinks. However, all these transient vegetation components, and the excrement deposited by
183 the grazing livestock contribute to the building up of soil organic matter, an important C sink
184 under tropical grasslands (Doran-Browne et al., 2016).
185 A third sink in a pastoral ecosystem are the bodies of the animals. Indeed a pastoral system in
186 sub-Sahara Africa is based on a core herd of reproductive adults that have strong cultural and
187 socio-economic functions (Turner et al., 2014). This dimension of social capital accumulation
188 is consistent with considering the herd as a C sink in the ecological approach use in the
189 present study.
190
191 **2.2 Measurement and estimation of GHG fluxes**
192
193 ***2.2.1 Enteric methane***
194 The monthly feed intakes and their dietary digestibility for cattle, sheep and goats were
195 predicted using fecal near infrared spectroscopy (NIRS) (Assouma et al., 2018). The feces
196 excreted were collected in fecal bags fitted on the animals and were randomly but
197 systematically sampled along the livestock grazing paths. The predicted monthly intake and
198 digestibility were then used to estimate the methane enteric emissions. The derived methane
199 emissions (ME in gCH_4/kg BW) were estimated using the general equation adapted for
200 tropical conditions developed by Archimède et al. (2011):

$$ME = 0.082 + 0.028 \times DOMI$$

where DOMI is total digestible organic matter intake (in g/kg BW), which was calculated as follows:

$$DOMI = MSvi \times 89\% \ OM \times OMd$$

Default emission factors were used for non-ruminants (horses and donkeys) (IPCC, 2006).

2.2.2 Soil and surface water emissions

The method used to measure GHG emissions from soil and surface water is described in full in (Assouma et al., 2017). The static chamber method (40*20*20 cm) was used for GHG emissions from soils. Two chambers were set up in each measuring site to obtain replicates and to insure the measurements were representative. For each flux measurement, the gas was sampled at 30 min intervals, i.e. at T_0, T_{30}, T_{60} and T_{90}. For GHG emissions, samples of water were taken from both natural ponds and ponds in the vicinity of the borehole and immediately sterilized with mercury chloride (0.01 ml/vial) to halt biological activity. To measure emissions from both soil and surface water, the air samples were analyzed using a SRI 8610C Gas Chromatograph.

2.2.3 Other sources of GHG emissions

As other sources of GHG emissions are considerably smaller, they were not measured on-field, but estimated using emission factors in the literature.

Enteric methane emissions by termites were estimated from coefficients per unit area adjusted to the density of termite mounds in a tropical semi-arid context (Traoré et al., 2008). Emissions in the wet and dry seasons were distinguished (Jamali et al., 2013).

To estimate emissions due to the fuel consumed by the motor pump, the quantity of diesel oil consumed each month was obtained from the Borehole Management Committee, which keeps

226 a management logbook. The quantity consumed was then converted into the quantity of CO_2
227 emitted monthly using the emission factor proposed by (IPCC, 2006).
228 Due to low rainfall in our study year, standing herbaceous biomass was limited, and according
229 to data provided by the local fire management service and in situ observations, no bush fires
230 occurred in the WTSA during the study period.

231

232 **2.3 Estimation of C accumulation in animals, plants and soil**

233

234 ***2.3.1 Variations in the C stock in livestock***

235 Variations in the **C stock in animals** were estimated by keeping track of changes in the
236 livestock population through surveys of the herd and monitoring of 40 sampled herds (i.e.
237 11.3% of the WTSA herds), which provided information on the changes in the composition of
238 the herd according to species, gender and age classes. Herds were monitored to quantify
239 incoming fluxes (purchases, births, loans and returns from transhumance) and outgoing fluxes
240 (deaths, sales, gifts, loans and transhumance departures). In addition, monthly variations in
241 animal live weight were estimated from barometric measurements in three selected herds
242 (Njoya et al., 1997).

243

244 ***2.3.2 Variations in C stock in wood***

245

246 - *Inventory of woody plants and of their mass in the WTSA*

247 The "point-centered quarter" (PCQ) described by Clark and Evans (1954), was used to assess
248 the density of trees (height > 4 m) and of shrubs (height < 4 m) at each site. At 50-m intervals
249 along a 500 m transect, the distance to the nearest tree and shrubs was measured in each of the

four quarters delineated by the transect line and the perpendicular at that point. The mean density (D) of the number of trees and shrubs per ha and the standard error of that mean were calculated using Pollard's algorithm (Pollard (1971), from the sums of squares of the distances measured for the four individuals of the n measuring points using the following formula:

$$\mathbf{D = (10^4) * 4 * (4*n-1)/(\pi * \Sigma dist^2)}$$

Leaf mass (Mleaf) was estimated using a specific allometric function that calculates the leaf mass from the circumference of each trunk (Cir, m). The following power type function was used:

$$\mathbf{Mleaf = oo * Cir^{ex}}$$

The oo and ex coefficients were taken from the overview by Henry et al. (2011) for each species.

Like leaf mass, wood volume was calculated using another power type function of the circumference and the wood mass was derived from the wood volume by multiplying it by specific wood volume densities for each species. The total mass of woody plants at a given site was then obtained by multiplying the leaf and wood mass by the tree and shrub densities.

- *Estimation of annual growth of wood mass*

The annual increment in wood mass was calculated from the mean annual increment of the stem circumference estimated from dendrometric measurements carried out in June 2009 and June 2015, i.e. after an interval of six years. Wood production during the six year period was estimated for each tree measured by the difference in wood mass in 2009 and in 2015, both assessed using the allometric equations proposed by Henry et al. (2011) as a function of the stem circumference. Annual wood production in WTSA was obtained by multiplying the mean annual wood production of the 24 woody plant species by the tree density of the

275 different species observed in the WTSA as a whole. Root mass was estimated as a fraction
276 (38%) of aboveground wood mass, as proposed by Woomer et al. (2004). This coefficient was
277 used to estimate annual root production.
278
279 ***2.3.3 Soil C accumulation***
280
281 - *Variations in C stored in the soil*
282 The soil compartment was considered as a "black box" in which the different processes
283 (mineralization, priming effect, etc.) driving soil organic matter dynamics were not
284 represented and the different forms of C stocks in the soil, more or less stable, were also not
285 distinguished. The accumulation of C in the soil was calculated as the difference between C
286 inputs and C outputs on a monthly basis using the C fluxes described in figure 3. C inputs are
287 organic matter deposited on the soil in the form of livestock excreta, litter from trees, shrubs
288 and herbaceous plants. Root turnover and rhizodeposition are two other sources of C inputs,
289 both of similar magnitude (Jones et al., 2009), that were taken into account in the calculation
290 of soil C accumulation. C outputs are CO_2 and CH_4 emissions from the soil to the atmosphere.
291 Losses due to runoff and leaching are limited under these limited rainfall conditions (Kindler
292 et al., 2011). They were not taken into account in this study.
293
294 - *C inputs*
295 Fecal deposition on the soil was assessed based on the daily fodder intake by animal
296 multiplied by the diet digestibility and by the number of animals of each species present in the
297 WTSA (Assouma et al., 2018).
298 The quantity of herbaceous litter buried in the soil was estimated by the difference between
299 the total herbaceous aboveground mass in the previous month and the sum of the

aboveground herbaceous mass and the herbaceous mass ingested that month using the following formula:

$$\textbf{Buried mass}_{m} = \textbf{aboveground mass}_{m-1} - (\textbf{aboveground mass}_{m} + \textbf{ingested mass}_{m})$$

where $_{m}$ is the month considered and $_{m-1}$ is the preceding month.

The litter buried during the late wet season, i.e. in August and September, was disregarded as the rate of senescence is less than 10% during the growth of annual herbaceous plants. To account for the spatial heterogeneity of the herbaceous mass at each site, the herbaceous layer was divided into four strata based on the apparent bulk of the herbaceous layer: zero in bare soil patches, low, medium or high in vegetated patches (Dardel et al., 2014). The frequency of each stratum was assessed visually and the herbaceous layer was classified at one meter intervals along a one-meter-wide strip along a 500 m long transect (Hiernaux et al., 2009). Total and green vegetation cover (visually estimated as % cover), standing and litter mass (destructive cutting, with harvest, air drying and weighing) were assessed in three 1 × 1 m plots randomly sampled in each stratum along the transects. The total aboveground herbaceous mass of the site was then computed by weighting the mean mass per stratum by the frequency of the stratum along the 500 m transect. The herbaceous root mass was assessed using a tube 7 cm in diameter inserted to a depth of 30 cm in the soil of each quadrat. All the excavated soil was sieved to 2 mm to remove the roots. Subsamples of straw, litter and roots were dried in an oven at 65 °C for 72 hours and then weighed again to determine the dry matter content.

The root turnover of the herbaceous plants was estimated at 53% of net root production (Gill and Jackson, 2000). The same coefficient was used to estimate rhizodeposition (Jones et al., 2009). Both processes only occur when the roots are alive, i.e. during the wet season in herbaceous plants. During the dry season, a constant rate of decomposition was applied to the herbaceous root mass that remained at the end of the wet season.

Woody plants recycle leaves, a fraction of the branches that fall and roots through turnover. All the leaves produced in one year were assumed to be recycled over the course of a complete year whatever the phenological regime among the six types identified (Hiernaux et al., 1994). For each phenological type, the breakdown of the leaves over the 12 months was set at a monthly time step based on the monthly coefficients in Hiernaux et al. (1999).

The mass of the branches returned to the soil was assessed at 5.4% of annual wood mass production (Marion et al., 2015). The woody plant root turnover and rhizodeposition were estimated as a fraction (56% for both C flows) of the annual increment of the root mass, i.e. net root production (Gill and Jackson, 2000).

- *C ouputs*

CO_2 and CH_4 gas emissions were measured at each site using static gas chambers as described in section 2.3.2. The measured gas fluxes were converted into C equivalent using the molar mass of each element.

2.4 C balance calculated at three temporal scales: month, season and year

2.4.1 Sampling strategy designed to account for seasonal dynamics

In order to establish the C balance of the ecosystem over a full year, all the observations and measurements were made in the WTSA, except for C accumulation in trees and shrubs which was assessed in an additional study site where historical data were available. The additional Ferlo site is located near the village of Dier Biran (15°21'N, 15°28'W) close to Dahra (Figure 1). All observations and measurements in the WTSA were repeated at 13 sites selected to account for the spatial heterogeneity of the WTSA landscape (five sites in the grazing lands, two sites in the vicinity of the borehole, settlements and ponds and one site in enclosures and forest plantations). Two more sites, one in an enclosure and one in a forest plantation in the WTSA were added to the 13 sites for a survey of vegetation including grass, trees and shrubs.

Observations and measurements were made between July 2014 and June 2015 (table 1). Table 1 lists the frequency and the number of on-field observations and measurements made per flux and stock type. The sources of GHG emissions (enteric CH_4, fuel combustion, bush fires) were assessed monthly except for GHG fluxes from soil and surface water, which were measured once a month in the wet season (July to October 2014), once in the cold dry season (January 2015) and once in the hot dry season (May 2015).

To measure C accumulation in livestock, a monthly demographic survey was conducted of 40 herds (i.e. 11.3% of the WTSA herds) and a monthly survey of animal weight in three herds (one small, one medium and one large). C accumulation in trees and shrubs was measured through tree inventories taken at the 15 sites in the WTSA and dendometric measurements taken at an interval of six years (one in June 2009 and one in June 2015) on 24 marked geo-referenced trees. Concerning C accumulation in soils, C input and output flows were estimated monthly (deposition of feces, deposition of litter from herbaceous vegetation during the wet season) or once every two months (deposition of litter from herbaceous vegetation during the dry season), except for deposition of litter from trees and shrubs, which was estimated annually and then interpolated per month (section 2.4.2).

2.4.2 Distribution of C accumulation and GHG emissions per month

The different components of the C balance (GHG fluxes and variations in C stocks) were distributed monthly to establish the monthly C balances. The components established from data monitored monthly (emissions of enteric methane, emissions from the motor pump and C accumulation in livestock, see section 2.4.1 for sampling strategy) did not require interpolation, whereas components established from data monitored on a bimonthly, seasonal or annual basis (soil and water GHG emissions, enteric methane from termites and C accumulation in soil and woody plants) did require interpolation.

For soil and water GHG emissions, the measurements made in January were considered to be representative of emissions during the cold-dry season, and those made in May, of the hot-dry season.

For CH_4 emissions from termites, the seasonal emission factors were considered to be the same in each month of each season.

For C accumulation in woody plants, the annual production of aboveground and belowground wood biomass was distributed over the 12 months as a proportion of the monthly production of foliage by each woody species to obtain a monthly estimation of C accumulation in wood.

For C accumulation in soil, the annual root turnover and rhizodeposition of woody plants was also distributed monthly according to the production of leaf mass. There was no reason to distribute the estimated annual fall of dead branches as a function of leaf mass and, given the lack of field observations, a simple uniform distribution over the 12 months of the year was made.

2.4.3 Carbon balance per month, season and over the year

To establish the C balance of the ecosystem as a whole, the emissions were accounted for positively (return of GHG to atmosphere) and the C stock variations were accounted for negatively (capture of CO_2 from the atmosphere). All the emissions of the three GHG were converted into eq-CO_2 using the global warming potentials proposed by the IPCC (2013), which are 1, 34 and 296 for CO_2, CH_4 and N_2O respectively. The variations in mass in animal bodies, and plants were converted into C accumulation (expressed as C equivalent) using the conversion factors listed in Assouma (2016), and then converted into CO_2 equivalent to establish the C balance. The conversion factor from C equivalent to CO_2 equivalent is 3.67.

The balance, the sum of emissions and the sum of variations in C stocks are expressed in CO_2 equivalent per unit area to facilitate comparison with other types of ecosystems. For the

description of the variability of the C balance over time, GHG emissions and variations in C stock were detailed over the 12 months, and then aggregated over the three seasons described in section 2.1 (wet season, cold dry season and hot dry season) and over the whole year.

3. Results

3.1 GHG emissions at the pastoral ecosystem level

Most of the GHG emissions occurred during the wet season and, in July and October, were higher than the annual average (figure 4). Emissions were lower than annual average throughout the dry season, however in the cold dry season, emissions were a little higher than in the hot dry season. The relative contribution of most of the pools varied over the year, with a major contribution to the variation of (i) N_2O emissions from soil related to the deposition of animal excreta on the ground ($24.6.10^{-2}$, $5.4.10^{-2}$ and $7.9.10^{-2}$ tCO_2-eq.ha^{-1}.$Month^{-1}$ in the wet season, cold dry season, and hot dry season, respectively) and (ii) enteric CH_4 emissions from ruminants ($3.8.10^{-2}$, $3.5.10^{-2}$ and $3.4.10^{-2}$ tCO_2-eq.ha^{-1}.$Month^{-1}$ in the wet season, cold dry season, and hot dry season, respectively). CH_4 emissions from surface water and from wet soil were particularly dependent on the season ($4.4.10^{-2}$, $0.6.10^{-2}$ and $0.3.10^{-2}$ tCO_2-eq.ha^{-1}.$Month^{-1}$ during the wet season, cold dry season and hot dry season, respectively).

The relative contribution of the different GHG sources to annual GHG emissions from the ecosystem (figure 5) ranked nitrous oxide (N_2O) first (59% of total emissions), methane CH_4 second (41%) and carbon dioxide CO_2 third (<1%). The three main GHG fluxes were emissions from the soil mainly due to the deposition of animal excreta on the ground (66%), emissions from the ponds (20%) and enteric methane from ruminants (11%). Enteric fermentation from termite mounds was responsible for 3% of total GHG emissions.

3.2 Carbon accumulation in trees, soil and livestock at the pastoral ecosystem level

Monthly variations in the C stock in each pool and at the scale of the whole pastoral ecosystem (figure 6) remained relatively stable in the hot dry season, varied in the cold dry season and were even more variable in the wet season. Most C accumulation in the soil occurred during the dry season, particularly in the cold dry season. However, most C accumulated in the wood and in the roots of trees and shrubs in the wet season. Over the full year and at ecosystem scale, the soil behaved as a C sink. Variations in C stocks were negative in the wet season because of high soil GHG emissions. Conversely, the cold and hot dry seasons were periods of C accumulation in the soil. These results highlight the seasonal variations in C stocks, the wet season being characterized by the concomitant GHG release from soil and C accumulation in plants.

C mainly accumulated in woody plants and soil in the pastoral ecosystem, and accounted for 69% and 31% of the area's annual C sequestration potential, respectively (Figure 7). Livestock only contributed about 1% of total C accumulation. Overall, the ecosystem sequestrated 0.75 tc $ha^{-1}.year^{-1}$ distributed as follows: 0.52 $tC.ha^{-1}.year^{-1}$ in trees, 0.23 $tC.ha^{-1}.year^{-1}$ in the soil and 0.007 $tC.ha^{-1}.year^{-1}$ in livestock.

3.3 Variability of the GHG balance at the level of the pastoral ecosystem

The balances between GHG emissions and C accumulation over WTSA highlighted contrasts between months (figure 8). In the cold dry season, the monthly balances varied between – 0.84 in November and – 0.32 tCO_2-$eq.ha^{-1}.Month^{-1}$ in February. In the hot dry season, the monthly budget varied between – 0.07 in March and – 0.02 Ggeq-$CO_2.month^{-1}$ in June. In the wet season, the monthly budget varied between +0.11 in October and +1.03 tCO_2-$eq.ha^{-1}.Month^{-1}$ in September. At the WTSA scale, the C balance varied between seasons: kit was positive in the wet season (+0.58 tCO_2-$eq.ha^{-1}.Month^{-1}$) and negative in the cool dry season (-

0.57 tCO_2-eq.ha^{-1}.$Month^{-1}$) and in the hot dry season (-0.05 tCO_2-eq.ha^{-1}. $Month^{-1}$). A seasonal or monthly negative balance means that C emissions more than offset C accumulation.

Overall, all the landscape units of the pastoral ecosystem emitted on average +2.601 teq-CO_2.ha^{-1}.$year^{-1}$, and sequestered -2.78 teq-CO_2.ha^{-1}.$year^{-1}$, on average. That is equivalent to an annual GHG balance of -0.16 teq-CO_2.ha^{-1}.$year^{-1}$. Thus the WTSA pastoral ecosystem sequestrated -0.04 ±0.01 teq-C.ha^{-1}.$year^{-1}$ the 2014-2015 year.

4. Discussion

The aim of this study was to provide a dynamic C balance that accounts for the variability of GHG emissions and C accumulation over time. In this section, we discuss the main factors that explain this variability and underline key sources of uncertainty. The factors that explain the seasonal variations in the C balance for the 2014-2015 period help predict likely inter-annual variability of the C balance for Sahelian pastoral ecosystems.

4.1 Rainfall is the main factor explaining seasonal variations in the C balance

Results showed marked seasonal variations in the C balance of the pastoral ecosystem studied. The positive C balance during the wet season mainly resulted from the high rates of GHG emissions (mainly N_2O from soil and CH_4 from ponds, see figure 5). C accumulation in trees and shrubs during the wet season, the main period of photosynthesis, was not sufficient to balance GHG emissions, and the return of plant mass to the soil was minimal during herbaceous growth and renewal of the leaves of woody plant resulting in a negative C accumulation in the soil (see figure 6). The negative C balances throughout the dry season was mainly the result of a decrease in GHG emissions as the soil and the ponds dried up and the progressive return to the soil of organic matter, mostly produced in the wet season, via the

burying of livestock excreta, herbaceous litter and the leaves of woody plant. Feces and litter mass were larger in the cold dry season, thereby increasing C accumulation in soil and rendering the balance more negative in the cold dry season than in the following hot dry season. These marked seasonal variations in the C balance were driven by (i) the seasonality of the monsoon climate, and (ii) pastoral practices characterized by seasonal herd mobility (also a consequence of the monsoon climate).

Under a Sahelian semi-arid monsoon climate, most photosynthesis and organic matter decomposition occur in the wet season (de Souza Rezende et al., 2016) . The wet season is consequently the most critical period for the C balance, as indicated by the positive C balance and highly variable GHG emissions (sections 2.3 and 2.1). During the dry season the feces deposited by livestock, the transfer of the standing straw to litter activated by livestock trampling and woody plant leaves and twigs falling onto the soil all contribute to the litter that is progressively fragmented and buried in the topsoil over the course of the dry season (Coleman et al., 1989). These inputs of organic matter into the top soil pave the way for the peak biological activity and decomposition during the three to four months of the following wet season. Rainfall fills the ponds, increases soil moisture and activates soil biological activity, causing a rapid increase in GHG emissions from the soil and even greater from the surface water. Indeed, the high inputs of litter and fecal matter to the soil during the preceding dry season, wetting by rain triggered biological activity in the soil, resulting in high N_2O emissions. Notable quantities of CH_4 were also emitted from the surface of ponds due to anaerobic fermentation of organic materials in the water and in the pond sediments. The organic materials originate from feces deposited by livestock either directly in the pond or around the edges while drinking and resting during the wet season (Assouma et al., 2017), but also from the litter, including the leaves of woody plants that accumulated on the pond floor in the dry season or were transported by runoff from the vicinity of the pond.

Livestock were responsible for a major contribution to the seasonality of the C balance, due to transhumance and its direct and indirect effects on GHG emissions and C accumulation in the soil. In the wet season, transhumant herds moved back to the Ferlo rangeland to take advantage of the high feed quality of the new growth triggered by the resumption of rainfall (Assouma et al., 2018). The livestock stocking rate in the WTSA peaked at 0.39TLU/ha in the wet season, versus 0.35 and 0.33TLU/ha in the cold and hot dry season, respectively. The wet season with its higher stocking rate is characterized by the large quantities of green fodder grazed and of excrement deposited on the ground with major concentrations around settlements and ponds that are full of water at that time, explaining why CH_4 enteric emissions and soil N_2O emissions were higher in the wet season. Free grazing leads to *in situ* consumption of about 30% of herbaceous biomass production (Assouma, 2016) and to *in situ* recycling of more than 50% of this ingested biomass via animal excretion (Diawara et al., 2018). This recycling, in association with dung beetles, accelerates the return of carbohydrates to the soil, thus contributing to C sequestration (Herrick and Lal, 1995; Soussana et al., 2010). In addition, the effect of cattle trampling while grazing facilitates the transfer of standing herbaceous mass (e.g. straw) to the litter, and its fragmentation and burying in the soil (Hiernaux et al., 2014). Dung and litter inputs via cattle trampling are two C input flows activated by livestock that explain the positive C accumulation in soil during the dry season.

4.2 Inter-annual variability of rainfall and its consequences for the C balance

In the Sahelian region, rainfall distribution drives the pastoral ecosystem's annual production (Hiernaux et al., 2009), which is also modulated by the pastoral populations' management practices (Sy, 2010). Soil moisture drives vegetation growth which is also limited by the availability of soil nutrients, especially N and P (Penning de Vries and Djiteye, 1982). Soil moisture also drives GHG emissions and hence the C balance (section 4.1.).

527 The present study was carried out in a year with low rainfall (204 mm.year^{-1}) compared with
528 the long-term average in the area (327.9± 132.6 mm.year^{-1} in Widou Thiengoly from 1956 to
529 2015). In addition, the long dry spell that occurred in July 2014, stopped herbaceous growth
530 which did not start again 41 days later with new germination, resulting in poor final yields. In
531 September 2015, the herbaceous yields measured in the WTSA reached 1978.9 kgDM.ha^{-1}
532 (33% higher than in 2014) for 298.8 mm annual rainfall (45% higher than in 2014) with no
533 long interruption in rainfall distribution between July and October. In 2016, annual rainfall
534 totaled 398.2 mm (95% higher than in 2014), but the weighted mean herbaceous yield over
535 WSTA was only 1567.3 kgDM.ha^{-1} because of the irregular distribution of the rainfall events
536 with two extremely large events cumulating 52% of annual rainfall in a single week in
537 August. The marked inter-annual variations in herbage yield in WTSA between 2014 and
538 2016 confirmed that not only variable total annual rainfall but also rainfall distribution causes
539 interannual variability of the functioning and hence of the C balance of pastoral ecosystems.
540 We can hypothesize that an increase in herbaceous production would increase the livestock
541 fodder intake-excretion loop by delaying the livestock departure for dry season transhumance
542 and increase herd growth and hence C accumulation in livestock. The delay in the departure
543 for transhumance would increase enteric CH_4 emission and hence total GHG emissions in the
544 WTSA. We estimated that if no herds had left for transhumance in the dry season in our study
545 period and if livestock nutrition had remained unchanged, total enteric emission would have
546 increased by 55% and its contribution to the total GHG emissions would have increased by
547 16% (assuming that other GHG fluxes remained unchanged). All these processes would
548 converge and increase transfers of litter and animal excrement to the soil with uncertain
549 consequences for C accumulation in soil. More organic matter input in the soil would not only
550 increase C inputs to the soil but also stimulate soil biological activity and hence GHG
551 emissions from the soil and surface water. However, if the increase in grazing pressure due to

livestock mobility management is not sufficient to use up the additional herbaceous biomass, either litter decomposition will slow down or else the remaining standing herbaceous masses will increase the risk of bush fire during the dry season (Nielsen et al., 2003). Bush fire would increase GHG emissions and decrease inputs of organic matter input into the soil, which would drive C accumulation in the soil toward negative values as a proportion of the burned area.

Finally, this section shows that there are many different effects of rainfall patterns on the ecosystem GHG fluxes and C stock variations, some of which are antagonist, with major uncertainties in the final C balance.

4.3 Rain-based processes explain a large proportion of uncertainties in the C balance

Knowledge gaps about the response of this ecosystem to variable rainfall remain that may explain some of the uncertainties in the C balance calculated in this study. These uncertainties are due to: (i) extrapolation over time and interpolation of some GHG fluxes or stock variations based on one-off measurements, (ii) disregarding certain flows (e.g. leaching, run-off), (iii) under-consideration of interspecific variability that may be of importance in extreme years (very dry or very rainy years).

As regards extrapolation, most of the fluxes and stocks described in this study were estimated from measurements that were one-off measurements made monthly, bi-monthly, or once a year, and extrapolated to the length of the study period, assuming stability over the period (months, seasons or years). However, these measurements may be too far apart. Indeed, GHG fluxes from the soil are highly sensitive to soil moisture. Soil moisture can vary very rapidly during the wet season depending on the rainfall distribution pattern (Delon et al., 2015) and it is consequently risky to extrapolate a one-off measurement to a whole month. This uncertainty could be reduced by using data from flux towers with continuous recording of

578 CO_2 fluxes as described in (Tagesson et al., 2015b) and using modeling approaches (i.e.
579 watering experiments to calculate a regression between soil GHG emissions and soil
580 moisture). Moreover, our estimation of the annual wood mass increment of trees was based on
581 interpolation from the stem circumference increment over a period of six years (section 2.4.2).
582 This six-year period is long compared to the one-year period used for the assessment of C
583 accumulation in other ecosystem compartments (soil and livestock). Nevertheless, the six-
584 year period may conceal the inter-annual variability of tree growth in relation with inter-
585 annual rainfall variability (Takimoto et al., 2008).
586 Disregarding some processes whose impact was considered minor in a dry year, (which was
587 the case in our study year), would need to be reconsidered when assessing some **C fluxes** and
588 stock variations in more rainy years. For instance, the CO_2 and CH_4 emissions from the soil
589 were the C output flows used for the assessment of C accumulation in the soil. C losses
590 through leaching and runoff were disregarded, as both are low when rainfall is limited
591 (Kindler et al., 2011), but could be non-negligible in a more rainy year. It would be worth
592 checking these processes in a future study including more rainy years.
593 Moreover, in the present study, interspecific differences in C sequestration or enteric
594 emissions between woody plants and livestock were not taken fully into account. The annual
595 increment in tree diameter was obtained from measurements made on 24 individual trees
596 belonging to the four main species. Although they were the most representative species in the
597 area, a more exhaustive consideration of specific diversity should be envisaged in a future
598 study. It would be useful to include rainfall-driven processes, like the population dynamics of
599 woody plants (mortality, regeneration, inter-individual interactions) that were not taken into
600 account in the present study but that may have an impact on the variability of C stocks and
601 fluxes between years depending on rainfall. The lack of knowledge about the root mass of
602 woody plants, its turnover and rhizodeposition flows also led to uncertain estimations in this

study, since they were all based on single, not species-specific, fixed rainfall-independent coefficients (see sections 2.4.1). Similarly, the population dynamics of livestock depend on rainfall. Forage availability determines animal mortality and birthrates. Between year changes in these parameters may call for reassessment of the long-term C accumulation in livestock. For instance, high mortality rates were observed in the Sahel during the severe droughts in the 1970s and 1980s. Large uncertainties on these zootechnical parameters persist due to the difficulties encountered by the majority of Sahel countries to establish reliable national statistical information systems (Lesnoff et al., 2012). Changes in these zootechnical parameters in response to rainfall patterns is species-dependent. For instance, small ruminant herds (goats in particular) are more resilient to drought events. Estimated enteric emissions were likewise based on measurements made only in ruminants (Assouma et al., 2018), while default emission factors were used for non-ruminants (donkeys and horses, that account for 11% of the total livestock population in TLU). The proportion of ruminants in the livestock population of the WTSA varies depending on transhumance practices that also vary between years, due to inter-annual rainfall variability. Indeed ruminants (cattle and sheep in particular) move about more in transhumance (Sy, 2010) than non-ruminants. Donkeys and horses are kept in the WTSA as draft animals to transport water from the borehole (donkeys) or people between camps and markets (horses).

The sensitivity of the C balance of this pastoral ecosystem to both changes in rainfall volume and distribution, and herd mobility and composition deserves further research and highlights the interest of establishing observatories to monitor pastoral systems over periods of a few consecutive years.

5. Conclusions and future outlook

This study highlights the marked seasonal variability in the C balance of the Sahelian pastoral ecosystem studied here. The wet season was characterized by positive C balances due to major emissions from the soil (N_2O), surface water and animals (CH_4). Conversely, in the dry season C balances were negative because fluxes were dominated by C accumulation in soils. Soil C inputs (animal excreta, burying of litter in the soil) mainly occurred at the beginning of the dry season, during the cold dry season, when animals had not yet left for transhumance. These soil C inputs paved the way for the high GHG emissions measured during the wet season. These marked seasonal variations in GHG emissions and C sequestration were explained by biotic factors (e.g. the flushes of soil biological activity during the wet season), and by livestock farmers' practices (e.g. seasonality of the presence of livestock, adaptation of the animal stocking rate to available forage) all driven by rainfall patterns (e.g. the seasonality of rainfall and soil moisture).

The seasonal variations in emissions and C sequestration resulted in a slightly negative full C balance over the annual cycle at -0.04±0.01 teq-C.ha^{-1}.year^{-1} for the 2014-2015 cycle. These results support the hypothesis that tropical grassed ecosystems can act as a carbon sink, as demonstrated in better documented ecosystems supporting extensive livestock systems under temperate climates (Soussana et al., 2004). The present study underlines the fact that woody vegetation is a key C sink, more than soil, in Sahelian pastoral ecosystems. Indeed the soil sequestration potential is lower in the Sahel than under temperate conditions because of faster organic matter mineralization in sandy soils (Kalbitz et al., 2000), and in hot climates (Kotir, 2011). This lower potential is also explained by the fact that in sandy soils (with poor clay content), the organic C generated after the decomposition process cannot bind to clay minerals (phyllosilicates), i.e. chemical protection, or be embedded in soil microaggregates, i.e. physical protection (Six et al., 2002).

651 The negative C balance found in this study clashes with the traditional image of African
652 livestock having a major impact on climate change. The negative image was consolidated by
653 sectorial and LCA studies (Gerber et al., 2013) that only took anthropogenic sources of GHG
654 into account. Our results differ from those of other studies because the new methodology
655 tested in this study considers all the ecosystem components (animals, soil and plants) and the
656 interactions between themselves and with the atmosphere. This ecosystem angle revealed key
657 points that need to be taken into account when allocating impacts between livestock and
658 natural processes to improve the accuracy of future LCAs of pastoral livestock products. For
659 instance, a more detailed analysis of organic input flows (feces versus plant material) is
660 needed to allocate N_2O and CH_4 emissions from the soil and water to livestock versus
661 vegetation. The ecosystem approach also revealed numerous indirect mitigation effects of
662 livestock on climate change. Pastoral systems maintain rangelands that have a sequestration
663 potential in soil and trees, and these systems also indirectly avoid GHG emissions by reducing
664 the termite population and the risk of bush fires as the grazing herds ingest the herbaceous
665 mass and bury it in the soil as they move around (section 4.1).
666 The other benefits of this ecosystem approach are a better understanding of the drivers of the
667 C balance that identifies appropriate and effective mitigation options with reference to the
668 seasonal and between year dynamics of the C balance (sections 4.1 and 4.2). For instance,
669 controlling organic matter inputs in natural ponds in the wet season and keeping water
670 flowing in the vicinity of the borehole in the dry season are key options to mitigate CH_4
671 emissions from surface water. Harvesting surplus forage at the beginning of the dry season in
672 more rainy years will both reduce the risk of bush fires and increase livestock productivity.
673 Indeed, storing forage will increase forage availability during the dry season, and hence
674 reduce transhumance with consequently less energy expenditure for animal movements for

675 the same level of enteric methane emissions. More details on mitigation options and their
676 potential adoption are available in Assouma (2016).
677 More largely in terms of research agendas, these new results call for more ecosystem
678 approaches to the C balances in agricultural systems worldwide to fully account for both
679 direct and indirect, and both negative and positive, effects of agricultural activities on climate
680 change. If the mitigation effects of livestock activities are confirmed in other ecosystems, we
681 can imagine that these new elements will be incorporated in IPPC guidelines and
682 recommendations, in the same way as C sequestration in pasture was included in the revised
683 guidelines (IPCC, 2006). Agricultural activities differ from other human activities as they
684 largely depend on natural resources and contribute to numerous ecosystem services. This is
685 particularly true for low-input systems that largely dominate agricultural systems in tropical
686 regions. The first studies on sylvo-pastoral and agro-pastoral systems in West Africa suggest
687 that the share of rangelands in the landscape is a key parameter for C neutrality in these
688 agricultural systems (Vayssières et al., 2017). Generalization of ecosystem C balances for
689 diverse ecosystems along agro-climatic gradients (from arid to sub-humid climate areas) will
690 facilitate the full integration of mobile livestock impacts in LCA studies. Integrating the many
691 impacts livestock have on the diverse ecosystems they cross along their transhumance path is
692 a key methodological issue for LCA studies of pastoral livestock products.
693 Despite the low productivity of pastoral systems, they are already known for their
694 contribution to food security and the socio-economic benefits they have for local population
695 and national economies, in particular in the Sahel. The mitigation potential of pastoral
696 ecosystems revealed by the present study is a complementary argument for the reinforcement
697 of policies supporting pastoral systems. Such policies will need to reinforce the mobility of
698 herds and people as a key to the sustainability of pastoral populations, the ecological
699 equilibrium of rangelands and hence a neutral C balance of pastoral ecosystems. This mobility

is threatened in many SSA countries by human population growth and the resulting urban and crop expansion onto rangelands.

The results of this study provide arguments for including pastoral livestock systems in payment for environmental services programs. Despite the limited C sequestration potential of arid ecosystems occupied by pastoral systems, they account for about 40% of African land stretching over 6,000 km from the Atlantic Ocean to the Red Sea (Le Houerou, 1989). C markets may be an opportunity to pay pastoral populations for the environmental services they provide, thereby increasing their livelihoods (Lipper et al., 2010). These ecosystems support limited human population densities which may increase the efficiency of the mechanism (Dutilly-Diane et al., 2007).

Acknowledgements

The Islamic Development Bank (the Merit Scholarship Program for High Technology) and the European Union (the FP7-KBBE-2010 Animal Change project) provided significant financial support.

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Table 1. Schedule of observations per component of the ecosystem with the number of measurement sites per landscape unit at Widou Thiengoly (W) and Dier Biran (D)

Variations in fluxes/Stock	July. 14	Aug. 14	Sept. 14	Oct. 14	Nov. 14	Dec. 14	Jan. 15	Feb. 15	Mar. 15	Apr. 15	May 15	June 15	Location of observations
Enteric fermentation	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	5 steers from 3 herds across all landscape units
GHG fluxes from soil and water	**W**	**W**	**W**	**W**			**W**				**W**		Graz(5), Set(2), Pon(2), Bor(2), Pla(1), Encl(1)
Bush fires	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	Graz(5), Set(2), Pon(2), Bor(2), Pla(2), Encl(2)
Fuel combustion	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	Bor(1)
C accumulation in livestock	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	5 steers and 40 herds across all landscape units
C accumulation in trees and shrubs (stocks)							**W**					**D**	Graz(5), Set(2), Pon(2), Bor(2), Pla(2), Encl(2)
Soil C accumulation (stocks)							**W**						Graz(5), Set(2), Pon(2), Bor(2), Pla(2), Encl(2)
Soil C accumulation (C inputs)	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	**W**	Graz(5), Set(2), Pon(2), Bor(2), Pla(2), Encl(2)
Soil C accumulation (C outputs)	**W**	**W**	**W**	**W**			**W**				**W**		Graz(5), Set(2), Pon(2), Bor(2), Pla(1), Encl(1)

W: Measurements made in the Widou Thiengoly service area**,** **D:** Measurements made in the village of Dier Biran
Graz: Grazing lands, **Set:** Settlements, **Pon:** Ponds, **Bor:** Borehole, **Pla:** Forest plantations, **Encl:** Enclosures

Figure captions

Figure. 1. The pink circle shows the location of the Widou Thiengoly service area (study area). The red star shows the complementary study site Dier Biram in the Ferlo region.

Figure 2. Conceptual model of the functioning of a Sahelian pastoral ecosystem in terms of nitrogen and carbon stocks-fluxes (the thickness of the borders of the boxes for C stocks and of the arrows of the GHG fluxes is proportional to their volume).

Figure 3. Aboveground-belowground and belowground-atmosphere carbon fluxes affecting the accumulation of carbon in the soil in a typical Sahelian pastoral ecosystem.

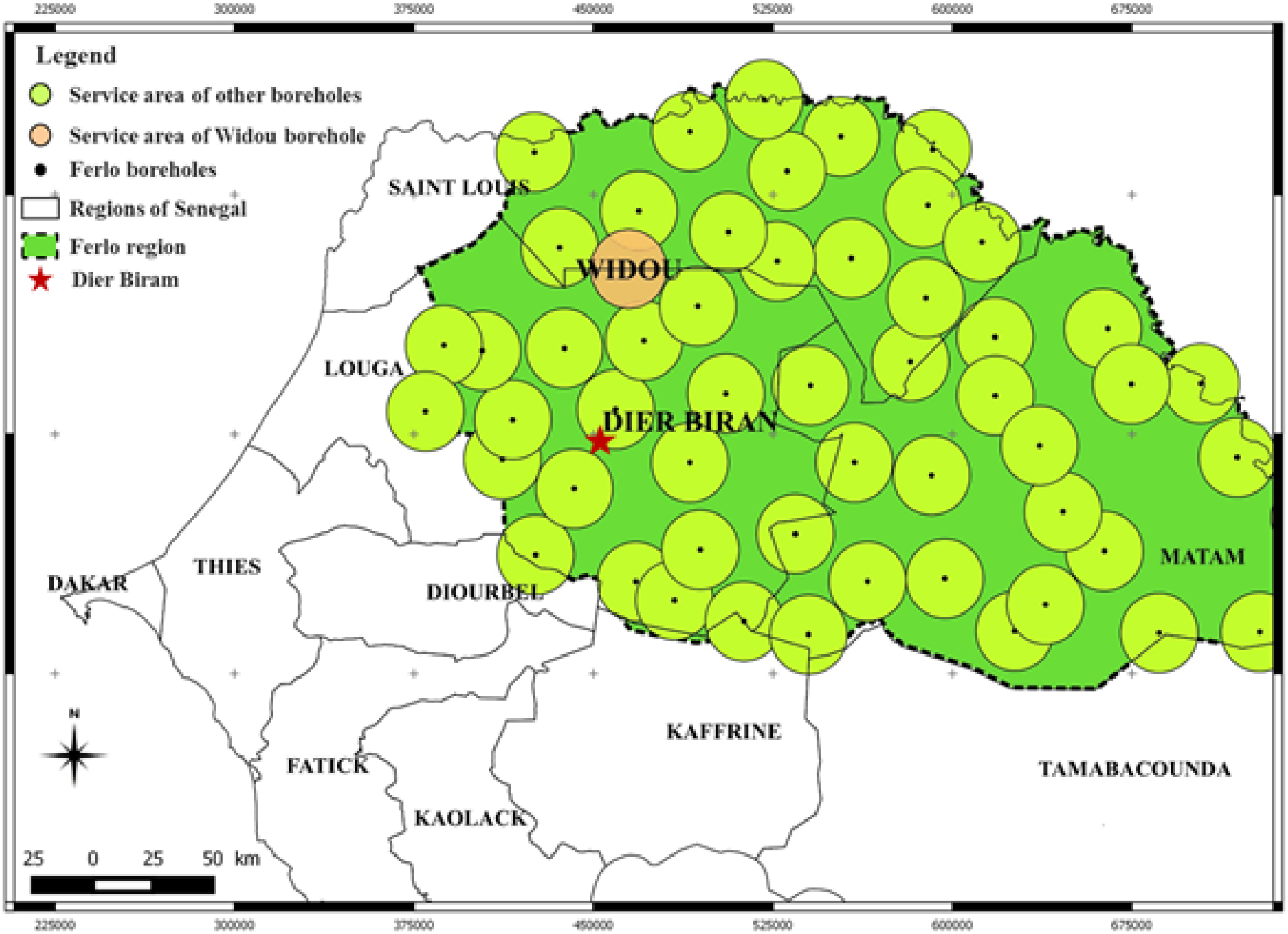
Figure 4. Temporal variability of total GHG emissions at the pastoral ecosystem scale (all GHG sources included)

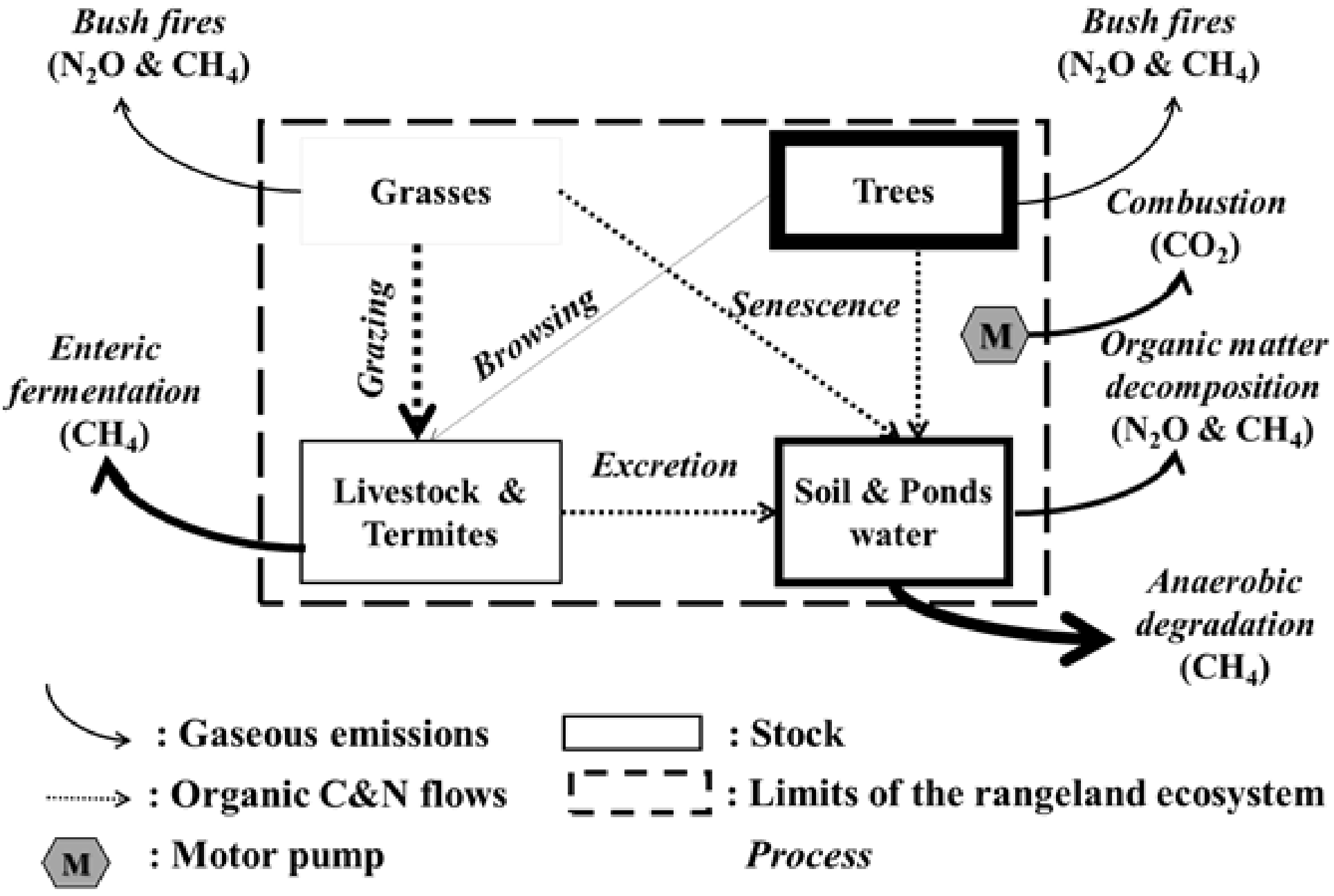
Figure 5. Total GHG emissions at pastoral ecosystem scale ranked according to the category of emission (in CO_2-eq).

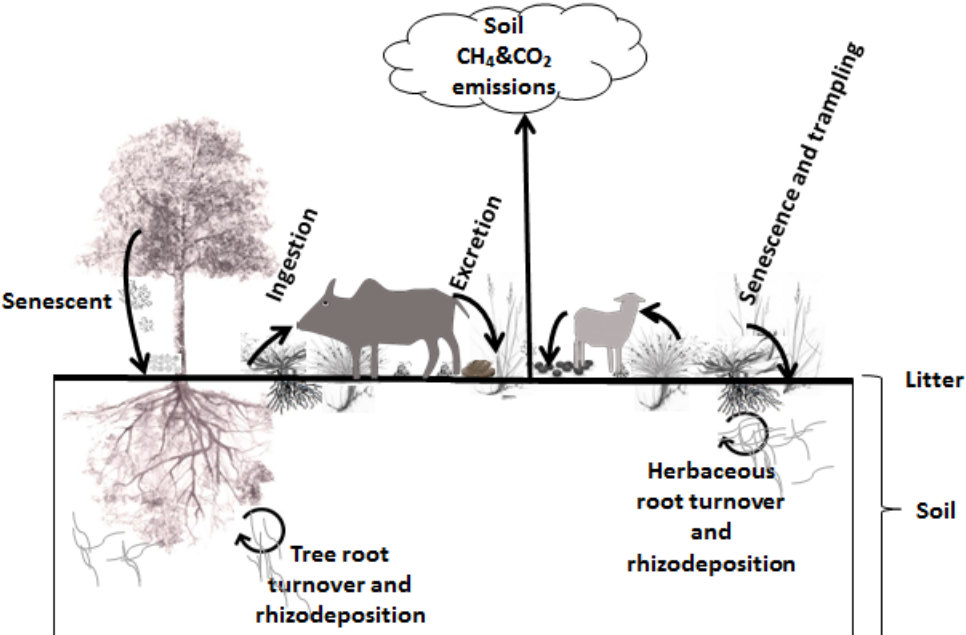
Figure 6. Temporal variability of C stock at pastoral ecosystem scale

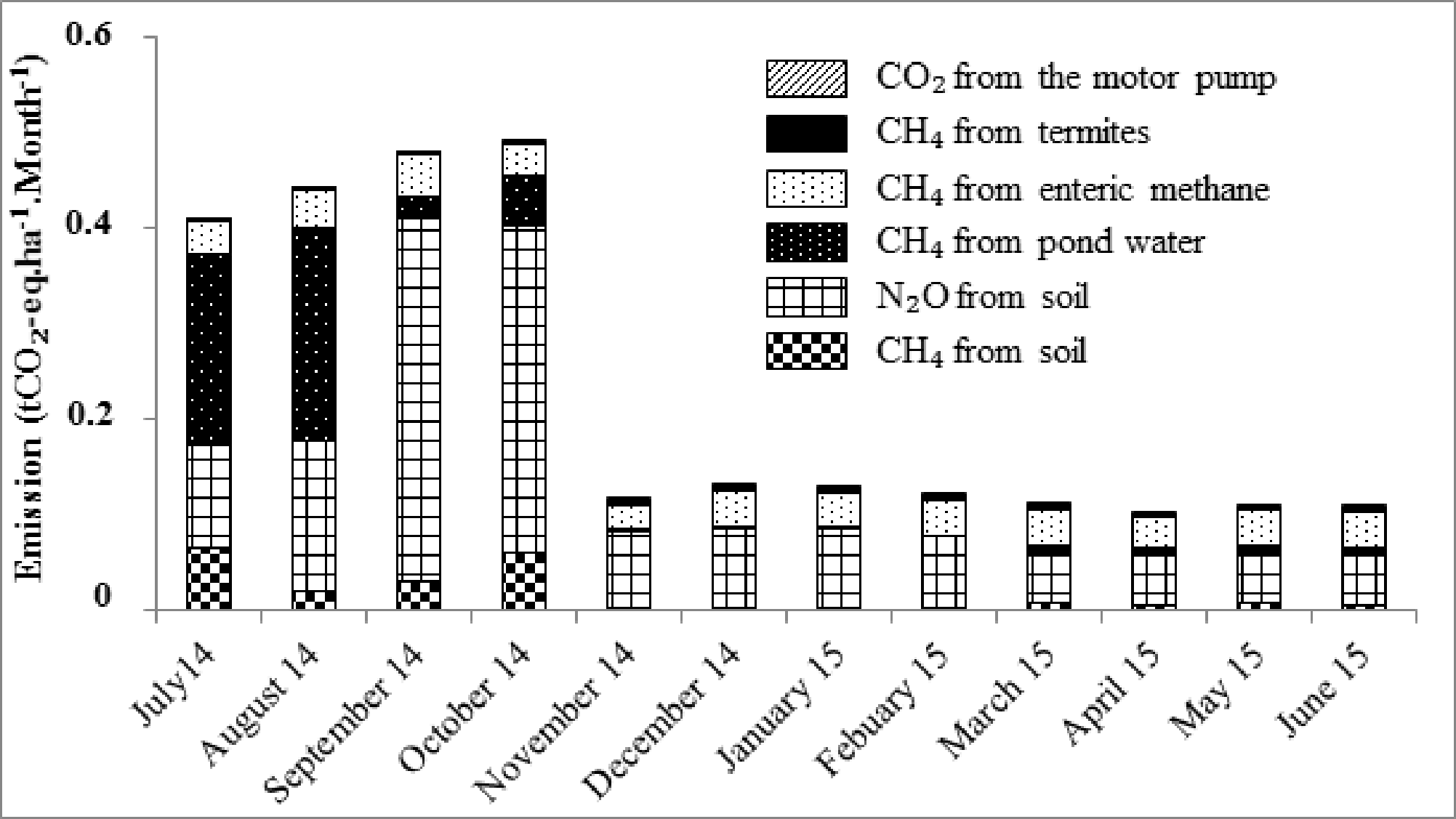
Figure 7. Annual C accumulation in the WTSA pastoral ecosystem per category of C stocks

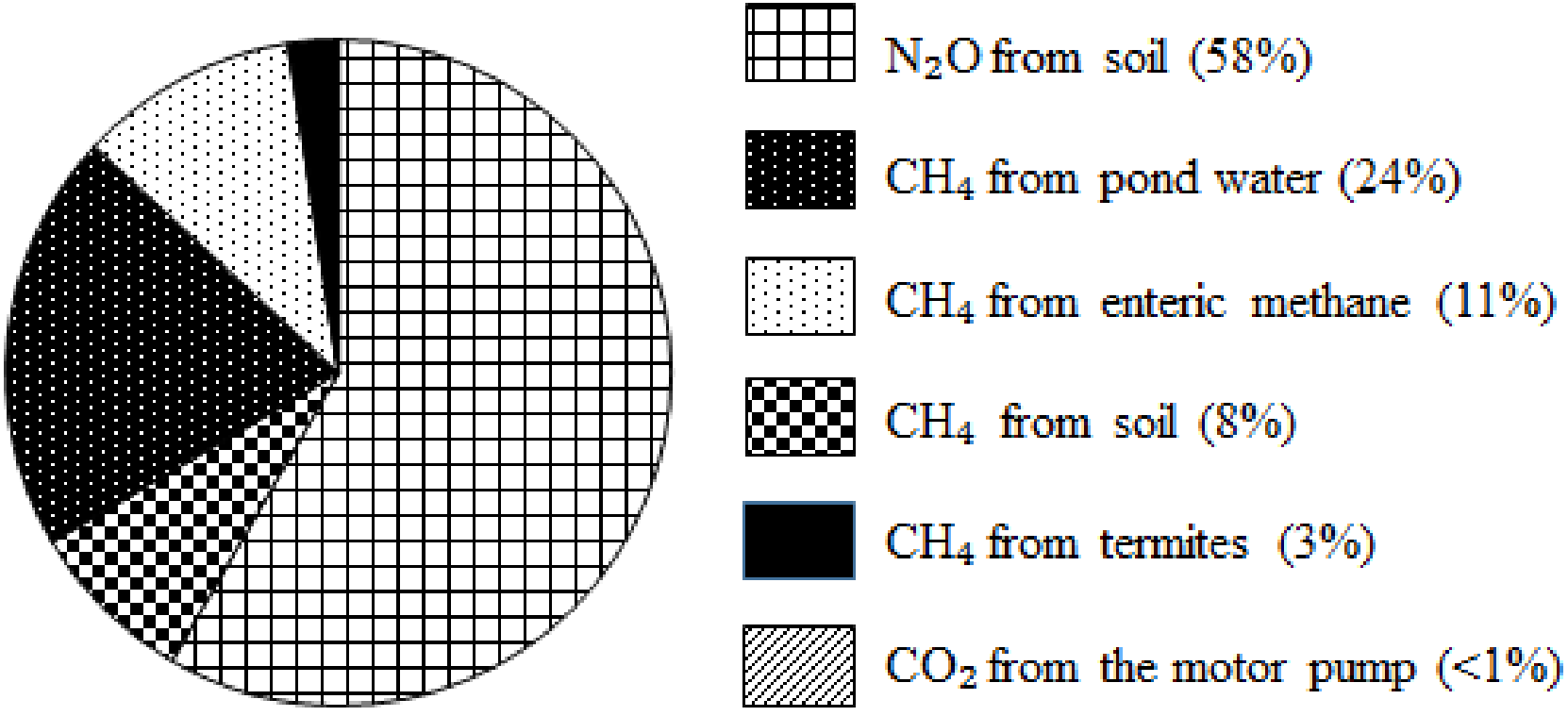
Figure 8. Monthly variations in the C balance at the WTSA ecosystem scale balancing GHG emissions (from soil, water, livestock, termites and motor pump) and C accumulation (in plants, soil and livestock).

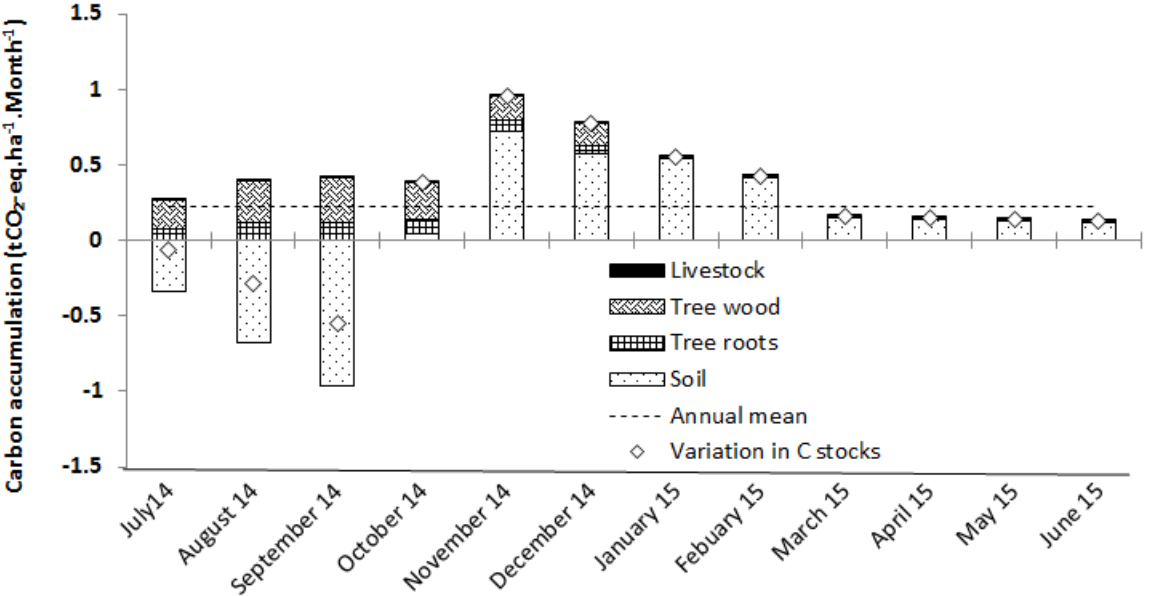

Legend
Service area of other boreholes
Service area of Widou borehole
Ferlo boreholes
Regions of Senegal
Ferlo region
Dier Biram
SAINT LOUIS
WIDOU
LOUGA
DIER BIRAN
THIES
DAKAR
DIOURBEL
MATAM
KAFFRINE
FATICK
TAMABACOUNDA
KAOLACK
N
25 0 25 50 km
225000
300000
375000
450000
525000
600000
675000
1800000
1700000
1600000

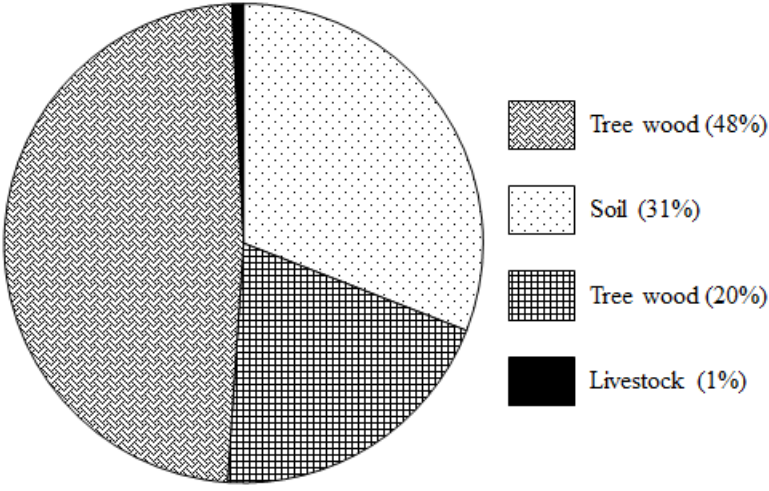

Bush fires
(N_2O & CH_4)
Bush fires
(N_2O & CH_4)
Grasses
Trees
Combustion
(CO_2)
Grazing
Browsing
Senescence
M
Enteric
fermentation
(CH_4)
Organic matter
decomposition
(N_2O & CH_4)
Livestock &
Termites
Excretion
Soil & Ponds
water
Anaerobic
degradation
(CH_4)
: Gaseous emissions
: Stock
: Organic C&N flows
: Limits of the rangeland ecosystem
M
: Motor pump
Process

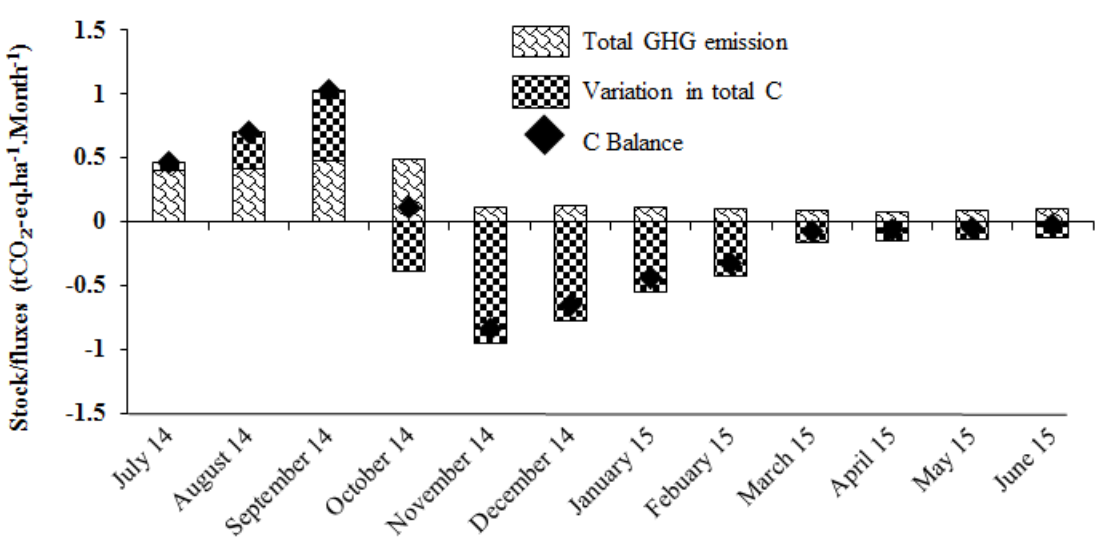

Soil
CH_4&CO_2
emissions
Senescent
Ingestion
Excretion
Senescence and trampling
Litter
Soil
Herbaceous
root turnover
and
rhizodeposition
Tree root
turnover and
rhizodeposition


Emission (tCO2-eq.ha-1.Month-1)
0.6
0.4
0.2
0
CO2 from the motor pump
CH4 from termites
CH4 from enteric methane
CH4 from pond water
N2O from soil
CH4 from soil
July 14
August 14
September 14
October 14
November 14
December 14
January 15
Febuary 15
March 15
April 15
May 15
June 15


N_2O from soil (58%)
CH_4 from pond water (24%)
CH_4 from enteric methane (11%)
CH_4 from soil (8%)
CH_4 from termites (3%)
CO_2 from the motor pump (<1%)


Carbon accumulation (tCO2-eq.ha-1.Month-1)
1.5
1
0.5
0
-0.5
-1
-1.5
Livestock
Tree wood
Tree roots
Soil
Annual mean
Variation in C stocks
July14
August 14
September 14
October 14
November 14
December 14
January 15
Febuary 15
March 15
April 15
May 15
June 15


Tree wood (48%)
Soil (31%)
Tree wood (20%)
Livestock (1%)


Total GHG emission
Variation in total C
C Balance
Stock/fluxes (tCO2-eq.ha-1.Month-1)
1.5
1
0.5
0
-0.5
-1
-1.5
July 14
August 14
September 14
October 14
November 14
December 14
January 15
Febuary 15
March 15
April 15
May 15
June 15