



HAL
open science

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

Mohamed-Habibou Assouma, Pierre Hiernaux, Philippe Lecomte, Alexandre Ickowicz, Martial Bernoux, Jonathan Vayssieres

► To cite this version:

Mohamed-Habibou Assouma, Pierre Hiernaux, Philippe Lecomte, Alexandre Ickowicz, Martial Bernoux, et al.. Contrasted seasonal balances in a Sahelian pastoral ecosystem result in a neutral annual carbon balance. *Journal of Arid Environments*, 2019, 162, pp.62-73. 10.1016/j.jaridenv.2018.11.013 . hal-02618241

HAL Id: hal-02618241

<https://hal.inrae.fr/hal-02618241>

Submitted on 21 Oct 2021

HAL is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.



Distributed under a Creative Commons Attribution - NonCommercial 4.0 International License

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

3

4 Mohamed Habibou Assouma^{1,2,3*}, Pierre Hiernaux⁴, Philippe Lecomte^{1,2,3}, Alexandre
5 Ickowicz^{2,5}, Martial Bernoux⁶, Jonathan Vayssières^{1,2,3}

6

7 ¹ CIRAD, Umr SELMET, Dakar, Senegal

8 ² SELMET, Univ Montpellier, CIRAD, INRA, Montpellier SupAgro, Montpellier, France

9 ³ PPZS, Pastoral Systems and Dry Lands, Dakar, Senegal

10 ⁴ Pastoc, Pastoralisme Conseil, Caylus, France

11 ⁵ CIRAD, Umr SELMET, Montpellier, France

12 ⁶ IRD, Umr Eco&Sols – F-34060, Montpellier, France

13

14 ***Corresponding author:** Mohamed Habibou Assouma. Email:

15 habibou.assouma@gmail.com

16 **Present address:** Dp PPZS, ISRA-LNERV, BP 2057 Dakar Hann, Senegal

17

18 **Abstract**

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

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

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

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

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

35

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

37

38 **1. Introduction**

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

50 short summer season, resulting in highly seasonal forage availability and quality. In response,
51 pastoralism, the dominant economic activity, is based on seasonal livestock mobility (Turner
52 et al., 2014). Rainfall drives rangeland ecosystem functioning (i) by stimulating the growth of
53 herbaceous vegetation providing highly digestible green forage that insures the annual growth
54 and the reproduction of grazing animals (Chirat et al., 2014), and (ii) by activating the soil
55 biological activity responsible for the decomposition and mineralization of organic matter,
56 and hence the recycling of N, P and other nutrients (Delon et al., 2015). N and P availability
57 to plants is the main factor that limits ecosystem production (Penning de Vries and Djiteye,
58 1982). Consequently, rangeland production is particularly vulnerable to seasonal variations in
59 rainfall amounts and distribution patterns (Nassef et al., 2009).

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

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

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

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

110

111 **2. Material and methods**

112

113 **2.1 The Sahelian pastoral system at Widou Thiengoly**

114 The Ferlo region (70,000 km²) is located between latitudes 15° and 16° 30' N and longitudes
115 13°30' and 16° W and includes a large fraction of the Senegalese pastoral zone. The climate
116 of the region is semi-arid monsoon of Sahelian type characterized by the alternation of two
117 seasons: a 4-month wet season (July to October) and an 8-month dry season (November to
118 June). The dry season is subdivided into two 4-month seasons, the first being the cold dry
119 season from November to February and the second being the hot dry season from March to
120 June. The main economy of families in the Ferlo region is pastoral husbandry, characterized
121 by shared access to communal grazing lands, grazing resources (herbaceous plants plus
122 browsing the foliage, twigs and fruits of woody plants) and drinking water resources. Prior to
123 the 1950s, the Ferlo was only grazed during the rainy and early dry seasons by Fulani
124 transhumance herders. In the dry season, due to lack of accessible water, the pastoralists had

125 to move either northwards to the floodplain rangelands of the Senegal River valley or
126 southwards to the Sérère croplands or the sub-humid savannas located further south and east.
127 From the 1950s onwards, deep boreholes with mechanical pumps to tap the Maestrichian
128 water table were drilled at regular intervals every 20 to 30 km. Since then, transhumant
129 pastoralists have settled in the Ferlo, in family camps often set up in the vicinity of temporary
130 ponds but nevertheless scattered throughout the service area of the boreholes. Our case study
131 was the service area (WTSA) of the Widou Thiengoly borehole (15° 59' N, 15° 19' W), a
132 circular territory centered on the borehole with a radius of usually 15 km, i.e. half the mean
133 distance between neighboring boreholes. The WTSA extends over 706 km², hosts 354
134 pastoral camps and a population of 4,800 inhabitants. The total livestock population of the
135 WTSA ranged from 15 274 to 33 095 TLU in August and December, respectively. The
136 average livestock population was composed of cattle (54% of total TLU), sheep (27%), goats
137 (8%), horses (3%) and donkeys (8%). The WTSA was subdivided into six landscape units
138 based on land use, the type of vegetation and soils ([Assouma et al., 2017](#)). Landsat image TM
139 204-049 dated November 3, 2010 and field observations (GPS points to mark the boundaries
140 of the units and identify particular points) were used to map the six landscape units:

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

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

152 - The borehole (0.78 km², or 0.1%) is the only source that can be used water the herds
153 when the ponds have dried up, and is visited once a day or every two days by most
154 animals (some small ruminants are watered in the camp with water in tanks
155 transported by donkey carts from the borehole) and pastoralist families for at least six
156 months of the year,

157 - Enclosures (0.24 km², 0.03%), six small fenced experimental plots that were set up as
158 part of a formal research project in 1981, to assess the long term effect of conservation
159 without grazing or burning.

160

161 **2.2 Conceptual model and the components of the C balance**

162 The main C and N stocks and flows between stocks in the pastoral ecosystem and GHG fluxes
163 to the atmosphere (CO₂, CH₄, N₂O) were used in the conceptual model (Figure 2).

164 The conceptual model was designed using data in the literature to identify GHG emissions
165 and the sources and sinks of C that have to be taken into account in the C balance. According
166 to Gerber et al. (2013), extensive livestock systems in sub-Saharan Africa are responsible for
167 high rates of CH₄ emissions due to enteric fermentation. Termites are one of the main natural
168 sources of CH₄ in tropical savannas through the digestion of cellulose by the micro-organisms
169 living in their guts (Jamali et al., 2011). The deposition of livestock excreta on rangeland soils
170 increases N₂O emissions from the soil (Pelster et al., 2016). In the Sahelian pastoral
171 ecosystem, Assouma et al. (2017) also pointed to CH₄ emissions from standing water in ponds
172 and in the vicinity of the borehole due to organic matter directly excreted by livestock during
173 watering or resting and drained by runoff from the vicinity. Bush fires are also known to be a
174 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 gCH₄/kg BW) were estimated using the general equation adapted for
200 tropical conditions developed by Archimède et al. (2011):

201
$$\text{ME} = 0.082 + 0.028 \times \text{DOMI}$$

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

204
$$\text{DOMI} = \text{MSvi} \times 89\% \text{ OM} \times \text{OMd}$$

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

206

207 *2.2.2 Soil and surface water emissions*

208 The method used to measure GHG emissions from soil and surface water is described in full
209 in (Assouma et al., 2017). The static chamber method (40*20*20 cm) was used for GHG
210 emissions from soils. Two chambers were set up in each measuring site to obtain replicates
211 and to insure the measurements were representative. For each flux measurement, the gas was
212 sampled at 30 min intervals, i.e. at T₀, T₃₀, T₆₀ and T₉₀. For GHG emissions, samples of water
213 were taken from both natural ponds and ponds in the vicinity of the borehole and immediately
214 sterilized with mercury chloride (0.01 ml/vial) to halt biological activity. To measure
215 emissions from both soil and surface water, the air samples were analyzed using a SRI 8610C
216 Gas Chromatograph.

217

218 *2.2.3 Other sources of GHG emissions*

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

221 Enteric methane emissions by termites were estimated from coefficients per unit area adjusted
222 to the density of termite mounds in a tropical semi-arid context (Traoré et al., 2008).

223 Emissions in the wet and dry seasons were distinguished (Jamali et al., 2013).

224 To estimate emissions due to the fuel consumed by the motor pump, the quantity of diesel oil
225 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 CO₂
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

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

$$255 \quad D = (10^4) * 4 * (4 * n - 1) / (\pi * \sum dist^2)$$

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

$$259 \quad Mleaf = oo * Cir^{ex}$$

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

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

266

267 - *Estimation of annual growth of wood mass*

268 The annual increment in wood mass was calculated from the mean annual increment of the
269 stem circumference estimated from dendrometric measurements carried out in June 2009 and
270 June 2015, i.e. after an interval of six years. Wood production during the six year period was
271 estimated for each tree measured by the difference in wood mass in 2009 and in 2015, both
272 assessed using the allometric equations proposed by Henry et al. (2011) as a function of the
273 stem circumference. Annual wood production in WTSA was obtained by multiplying the
274 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 CO₂ and CH₄ 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

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

$$302 \quad \text{Buried mass}_m = \text{aboveground mass}_{m-1} - (\text{aboveground mass}_m + \text{ingested mass}_m)$$

303 where m is the month considered and $m-1$ is the preceding month.

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

320 The root turnover of the herbaceous plants was estimated at 53% of net root production (Gill
321 and Jackson, 2000). The same coefficient was used to estimate rhizodeposition (Jones et al.,
322 2009). Both processes only occur when the roots are alive, i.e. during the wet season in
323 herbaceous plants. During the dry season, a constant rate of decomposition was applied to the
324 herbaceous root mass that remained at the end of the wet season.

325 Woody plants recycle leaves, a fraction of the branches that fall and roots through turnover.
326 All the leaves produced in one year were assumed to be recycled over the course of a
327 complete year whatever the phenological regime among the six types identified (Hiernaux et
328 al., 1994). For each phenological type, the breakdown of the leaves over the 12 months was
329 set at a monthly time step based on the monthly coefficients in Hiernaux et al. (1999).
330 The mass of the branches returned to the soil was assessed at 5.4% of annual wood mass
331 production (Marion et al., 2015). The woody plant root turnover and rhizodeposition were
332 estimated as a fraction (56% for both C flows) of the annual increment of the root mass, i.e.
333 net root production (Gill and Jackson, 2000).

334

335 - *C outputs*

336 CO₂ and CH₄ gas emissions were measured at each site using static gas chambers as described
337 in section 2.3.2. The measured gas fluxes were converted into C equivalent using the molar
338 mass of each element.

339

340 **2.4 C balance calculated at three temporal scales: month, season and year**

341 **2.4.1 Sampling strategy designed to account for seasonal dynamics**

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

351 Observations and measurements were made between July 2014 and June 2015 ([table 1](#)).
352 [Table 1](#) lists the frequency and the number of on-field observations and measurements made
353 per flux and stock type. The sources of GHG emissions (enteric CH₄, fuel combustion, bush
354 fires) were assessed monthly except for GHG fluxes from soil and surface water, which were
355 measured once a month in the wet season (July to October 2014), once in the cold dry season
356 (January 2015) and once in the hot dry season (May 2015).
357 To measure C accumulation in livestock, a monthly demographic survey was conducted of 40
358 herds (i.e. 11.3% of the WTSA herds) and a monthly survey of animal weight in three herds
359 (one small, one medium and one large). C accumulation in trees and shrubs was measured
360 through tree inventories taken at the 15 sites in the WTSA and dendrometric measurements
361 taken at an interval of six years (one in June 2009 and one in June 2015) on 24 marked geo-
362 referenced trees. Concerning C accumulation in soils, C input and output flows were
363 estimated monthly (deposition of feces, deposition of litter from herbaceous vegetation during
364 the wet season) or once every two months (deposition of litter from herbaceous vegetation
365 during the dry season), except for deposition of litter from trees and shrubs, which was
366 estimated annually and then interpolated per month ([section 2.4.2](#)).

367

368 *2.4.2 Distribution of C accumulation and GHG emissions per month*

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

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

379 For CH₄ emissions from termites, the seasonal emission factors were considered to be the
380 same in each month of each season.

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

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

389

390 *2.4.3 Carbon balance per month, season and over the year*

391 To establish the C balance of the ecosystem as a whole, the emissions were accounted for
392 positively (return of GHG to atmosphere) and the C stock variations were accounted for
393 negatively (capture of CO₂ from the atmosphere). All the emissions of the three GHG were
394 converted into eq-CO₂ using the global warming potentials proposed by the [IPCC \(2013\)](#),
395 which are 1, 34 and 296 for CO₂, CH₄ and N₂O respectively. The variations in mass in animal
396 bodies, and plants were converted into C accumulation (expressed as C equivalent) using the
397 conversion factors listed in [Assouma \(2016\)](#), and then converted into CO₂ equivalent to
398 establish the C balance. The conversion factor from C equivalent to CO₂ equivalent is 3.67.

399 The balance, the sum of emissions and the sum of variations in C stocks are expressed in CO₂
400 equivalent per unit area to facilitate comparison with other types of ecosystems. For the

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

404

405 **3. Results**

406

407 **3.1 GHG emissions at the pastoral ecosystem level**

408 Most of the GHG emissions occurred during the wet season and, in July and October, were
409 higher than the annual average ([figure 4](#)). Emissions were lower than annual average
410 throughout the dry season, however in the cold dry season, emissions were a little higher than
411 in the hot dry season. The relative contribution of most of the pools varied over the year, with
412 a major contribution to the variation of (i) N₂O emissions from soil related to the deposition
413 of animal excreta on the ground ($24.6 \cdot 10^{-2}$, $5.4 \cdot 10^{-2}$ and $7.9 \cdot 10^{-2}$ tCO₂-eq.ha⁻¹.Month⁻¹ in the
414 wet season, cold dry season, and hot dry season, respectively) and (ii) enteric CH₄ emissions
415 from ruminants ($3.8 \cdot 10^{-2}$, $3.5 \cdot 10^{-2}$ and $3.4 \cdot 10^{-2}$ tCO₂-eq.ha⁻¹.Month⁻¹ in the wet season, cold
416 dry season, and hot dry season, respectively). CH₄ emissions from surface water and from wet
417 soil were particularly dependent on the season ($4.4 \cdot 10^{-2}$, $0.6 \cdot 10^{-2}$ and $0.3 \cdot 10^{-2}$ tCO₂-eq.ha⁻¹.
418 Month⁻¹ during the wet season, cold dry season and hot dry season, respectively).

419 The relative contribution of the different GHG sources to annual GHG emissions from the
420 ecosystem ([figure 5](#)) ranked nitrous oxide (N₂O) first (59% of total emissions), methane CH₄
421 second (41%) and carbon dioxide CO₂ third (<1%). The three main GHG fluxes were
422 emissions from the soil mainly due to the deposition of animal excreta on the ground (66%),
423 emissions from the ponds (20%) and enteric methane from ruminants (11%). Enteric
424 fermentation from termite mounds was responsible for 3% of total GHG emissions.

425

426 3.2 Carbon accumulation in trees, soil and livestock at the pastoral ecosystem level

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

437 C mainly accumulated in woody plants and soil in the pastoral ecosystem, and accounted for
438 69% and 31% of the area's annual C sequestration potential, respectively (Figure 7).
439 Livestock only contributed about 1% of total C accumulation. Overall, the ecosystem
440 sequestered $0.75 \text{ tC} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$ distributed as follows: $0.52 \text{ tC} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$ in trees, $0.23 \text{ tC} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$ in the soil and $0.007 \text{ tC} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$ in livestock.

442

443 3.3 Variability of the GHG balance at the level of the pastoral ecosystem

444 The balances between GHG emissions and C accumulation over WTSA highlighted contrasts
445 between months (figure 8). In the cold dry season, the monthly balances varied between –
446 0.84 in November and $-0.32 \text{ tCO}_2\text{-eq} \cdot \text{ha}^{-1} \cdot \text{Month}^{-1}$ in February. In the hot dry season, the
447 monthly budget varied between -0.07 in March and $-0.02 \text{ Ggeq-CO}_2 \cdot \text{month}^{-1}$ in June. In the
448 wet season, the monthly budget varied between $+0.11$ in October and $+1.03 \text{ tCO}_2\text{-eq} \cdot \text{ha}^{-1} \cdot \text{Month}^{-1}$
449 in September. At the WTSA scale, the C balance varied between seasons: it was
450 positive in the wet season ($+0.58 \text{ tCO}_2\text{-eq} \cdot \text{ha}^{-1} \cdot \text{Month}^{-1}$) and negative in the cool dry season (-

451 0.57 tCO₂-eq.ha⁻¹.Month⁻¹) and in the hot dry season (-0.05 tCO₂-eq.ha⁻¹. Month⁻¹). A
452 seasonal or monthly negative balance means that C emissions more than offset C
453 accumulation.

454 Overall, all the landscape units of the pastoral ecosystem emitted on average +2.601 teq-
455 CO₂.ha⁻¹.year⁻¹, and sequestered -2.78 teq-CO₂.ha⁻¹.year⁻¹, on average. That is equivalent to an
456 annual GHG balance of -0.16 teq-CO₂.ha⁻¹.year⁻¹. Thus the WTSA pastoral ecosystem
457 sequestrated -0.04 ±0.01 teq-C.ha⁻¹.year⁻¹ the 2014-2015 year.

458

459 **4. Discussion**

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

465

466 **4.1 Rainfall is the main factor explaining seasonal variations in the C balance**

467 Results showed marked seasonal variations in the C balance of the pastoral ecosystem
468 studied. The positive C balance during the wet season mainly resulted from the high rates of
469 GHG emissions (mainly N₂O from soil and CH₄ from ponds, see [figure 5](#)). C accumulation in
470 trees and shrubs during the wet season, the main period of photosynthesis, was not sufficient
471 to balance GHG emissions, and the return of plant mass to the soil was minimal during
472 herbaceous growth and renewal of the leaves of woody plant resulting in a negative C
473 accumulation in the soil (see [figure 6](#)). The negative C balances throughout the dry season
474 was mainly the result of a decrease in GHG emissions as the soil and the ponds dried up and
475 the progressive return to the soil of organic matter, mostly produced in the wet season, via the

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

482 Under a Sahelian semi-arid monsoon climate, most photosynthesis and organic matter
483 decomposition occur in the wet season ([de Souza Rezende et al., 2016](#)). The wet season is
484 consequently the most critical period for the C balance, as indicated by the positive C balance
485 and highly variable GHG emissions ([sections 2.3 and 2.1](#)). During the dry season the feces
486 deposited by livestock, the transfer of the standing straw to litter activated by livestock
487 trampling and woody plant leaves and twigs falling onto the soil all contribute to the litter that
488 is progressively fragmented and buried in the topsoil over the course of the dry season
489 ([Coleman et al., 1989](#)). These inputs of organic matter into the top soil pave the way for the
490 peak biological activity and decomposition during the three to four months of the following
491 wet season. Rainfall fills the ponds, increases soil moisture and activates soil biological
492 activity, causing a rapid increase in GHG emissions from the soil and even greater from the
493 surface water. Indeed, the high inputs of litter and fecal matter to the soil during the preceding
494 dry season, wetting by rain triggered biological activity in the soil, resulting in high N₂O
495 emissions. Notable quantities of CH₄ were also emitted from the surface of ponds due to
496 anaerobic fermentation of organic materials in the water and in the pond sediments. The
497 organic materials originate from feces deposited by livestock either directly in the pond or
498 around the edges while drinking and resting during the wet season ([Assouma et al., 2017](#)), but
499 also from the litter, including the leaves of woody plants that accumulated on the pond floor
500 in the dry season or were transported by runoff from the vicinity of the pond.

501 Livestock were responsible for a major contribution to the seasonality of the C balance, due to
502 transhumance and its direct and indirect effects on GHG emissions and C accumulation in the
503 soil. In the wet season, transhumant herds moved back to the Ferlo rangeland to take
504 advantage of the high feed quality of the new growth triggered by the resumption of rainfall
505 (Assouma et al., 2018). The livestock stocking rate in the WTSA peaked at 0.39TLU/ha in the
506 wet season, versus 0.35 and 0.33TLU/ha in the cold and hot dry season, respectively. The wet
507 season with its higher stocking rate is characterized by the large quantities of green fodder
508 grazed and of excrement deposited on the ground with major concentrations around
509 settlements and ponds that are full of water at that time, explaining why CH₄ enteric
510 emissions and soil N₂O emissions were higher in the wet season. Free grazing leads to *in situ*
511 consumption of about 30% of herbaceous biomass production (Assouma, 2016) and to *in situ*
512 recycling of more than 50% of this ingested biomass via animal excretion (Diawara et al.,
513 2018). This recycling, in association with dung beetles, accelerates the return of
514 carbohydrates to the soil, thus contributing to C sequestration (Herrick and Lal, 1995;
515 Soussana et al., 2010). In addition, the effect of cattle trampling while grazing facilitates the
516 transfer of standing herbaceous mass (e.g. straw) to the litter, and its fragmentation and
517 burying in the soil (Hiernaux et al., 2014). Dung and litter inputs via cattle trampling are two
518 C input flows activated by livestock that explain the positive C accumulation in soil during
519 the dry season.

520

521 **4.2 Inter-annual variability of rainfall and its consequences for the C balance**

522 In the Sahelian region, rainfall distribution drives the pastoral ecosystem's annual production
523 (Hiernaux et al., 2009), which is also modulated by the pastoral populations' management
524 practices (Sy, 2010). Soil moisture drives vegetation growth which is also limited by the
525 availability of soil nutrients, especially N and P (Penning de Vries and Djiteye, 1982). Soil
526 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 CH₄ 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

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

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

561

562 **4.3 Rain-based processes explain a large proportion of uncertainties in the C** 563 **balance**

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

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

578 CO₂ 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 CO₂ and CH₄ 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

603 study, since they were all based on single, not species-specific, fixed rainfall-independent
604 coefficients (see [sections 2.4.1](#)). Similarly, the population dynamics of livestock depend on
605 rainfall. Forage availability determines animal mortality and birthrates. Between year changes
606 in these parameters may call for reassessment of the long-term C accumulation in livestock.
607 For instance, high mortality rates were observed in the Sahel during the severe droughts in the
608 1970s and 1980s. Large uncertainties on these zootechnical parameters persist due to the
609 difficulties encountered by the majority of Sahel countries to establish reliable national
610 statistical information systems ([Lesnoff et al., 2012](#)). Changes in these zootechnical
611 parameters in response to rainfall patterns is species-dependent. For instance, small ruminant
612 herds (goats in particular) are more resilient to drought events. Estimated enteric emissions
613 were likewise based on measurements made only in ruminants ([Assouma et al., 2018](#)), while
614 default emission factors were used for non-ruminants (donkeys and horses, that account for
615 11% of the total livestock population in TLU). The proportion of ruminants in the livestock
616 population of the WTSA varies depending on transhumance practices that also vary between
617 years, due to inter-annual rainfall variability. Indeed ruminants (cattle and sheep in particular)
618 move about more in transhumance ([Sy, 2010](#)) than non-ruminants. Donkeys and horses are
619 kept in the WTSA as draft animals to transport water from the borehole (donkeys) or people
620 between camps and markets (horses).

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

625

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 (N₂O), surface water and animals (CH₄). 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 N₂O and CH₄ 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 CH₄
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 IPCC 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

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

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

710

711 **Acknowledgements**

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

715 **References**

- 716 Archimède, H., Eugène, M., Marie Magdeleine, C., Boval, M., Martin, C., Morgavi, D.P.,
717 Lecomte, P., Doreau, M., 2011. Comparison of methane production between C3 and C4
718 grasses and legumes. *Animal Feed Science and Technology* 166–167, 59-64.
- 719 Assouma, M.H., 2016. Approche écosystémique du bilan des gaz à effet de serre d'un
720 territoire sylvo-pastoral sahélien : contribution de l'élevage, AgroParisTech. L'Institut des
721 Sciences et Industries du Vivant et de l'Environnement, AGROPARISTECH, Montpellier,
722 p. 230.
- 723 Assouma, M.H., Lecomte, P., Hiernaux, P., Ickowicz, A., Corniaux, C., Decruyenaere, V.,
724 Diarra, A.R., Vayssières, J., 2018. How to better account for livestock diversity and fodder
725 seasonality in assessing the fodder intake of livestock grazing semi-arid sub-Saharan
726 Africa rangelands. *Livestock Science* 216, 16-23.
- 727 Assouma, M.H., Serça, D., Guérin, F., Blanfort, V., Lecomte, P., Touré, I., Ickowicz, A.,
728 Manlay, R.J., Bernoux, M., Vayssières, J., 2017. Livestock induces strong spatial
729 heterogeneity of soil CO₂, N₂O, CH₄ emissions within a semi-arid sylvo-pastoral
730 landscape in West Africa. *Journal of Arid Land* 9, 210 - 221.
- 731 Chirat, G., Groot, J.C.J., Messad, S., Bocquier, F., Ickowicz, A., 2014. Instantaneous intake
732 rate of free-grazing cattle as affected by herbage characteristics in heterogeneous tropical
733 agro-pastoral landscapes. *Applied Animal Behaviour Science* 157, 48-60.
- 734 Clark, P.J., Evans, F.C., 1954. Distance to Nearest Neighbor as a Measure of Spatial
735 Relationships in Populations. *Ecology* 35, 445-453.
- 736 Coleman, D.C., Oades, J.M., Uehara, G., 1989. Dynamics of soil organic matter in tropical
737 ecosystems, p. 71.
- 738 de Souza Rezende, R., S. Graça, M.A., dos Santos, A.M., Medeiros, A.O., Santos, P.F.,
739 Nunes, Y.R., Gonçalves Júnior, J.F., 2016. Organic Matter Dynamics in a Tropical Gallery
740 Forest in a Grassland Landscape. *Biotropica* 48, 301-310.
- 741 Delon, C., Mougin, E., Serça, D., Grippa, M., Hiernaux, P., Diawara, M., Galy-Lacaux, C.,
742 Kergoat, L., 2015. Modelling the effect of soil moisture and organic matter degradation on
743 biogenic NO emissions from soils in Sahel rangeland (Mali). *Biogeosciences* 12, 3253-
744 3272.
- 745 Devineau, J.-L., Fournier, A., Nignan, S., 2010. Savanna fire regimes assessment with
746 MODIS fire data: Their relationship to land cover and plant species distribution in western
747 Burkina Faso (West Africa). *Journal of Arid Environments* 74, 1092-1101.

748 Diawara, M.O., Hiernaux, P., Mougin, E., Grippa, M., Delon, C., Diakit , H.S., 2018. Effets
749 de la p ture sur la dynamique de la v g tation herbac e au Sahel (Gourma, Mali): une
750 approche par mod lisation. *Cahiers Agricultures* 27, 15010.

751 Doran-Browne, N.A., Ive, J., Graham, P., Eckard, R.J., 2016. Carbon-neutral wool farming in
752 south-eastern Australia. *Animal Production Science* 56, 417-422.

753 Dutilly-Diane, C., McCarthy, N., Turkelboom, F., Bruggeman, A., Tiedemann, J., Street, K.,
754 Serra, G., 2007. Could payments for environmental services improve rangeland
755 management in Central Asia, West Asia and North Africa? IFPRI, Washington,  tats-Unis.

756 Gerber, P.J., Steinfeld, H., Henderson, B., Mottet, A., Opio, C., Dijkman, J., Falcucci, A.,
757 Tempio, G., 2013. Tackling climate change through livestock – A global assessment of
758 emissions and mitigation opportunities, Food and Agriculture Organization of the United
759 Nations (FAO), Rome., p. 139.

760 Gill, R.A., Jackson, R.B., 2000. Global patterns of root turnover for terrestrial ecosystems.
761 *New Phytologist* 147, 13-31.

762 Henry, M., Picard, N., Trotta, C., Manlay, R., Valentini, R., Bernoux, M., Saint-Andr , L.,
763 2011. Estimating tree biomass of sub-Saharan African forests: a review of available
764 allometric equations. *Silva Fennica* 45, 477-569.

765 Herrero, M., Henderson, B., Havlik, P., Thornton, P.K., Conant, R.T., Smith, P., Wirsenius,
766 S., Hristov, A.N., Gerber, P., Gill, M., Butterbach-Bahl, K., Valin, H., Garnett, T.,
767 Stehfest, E., 2016. Greenhouse gas mitigation potentials in the livestock sector. *Nature*
768 *Clim. Change* advance online publication.

769 Herrick, J.E., Lal, R., 1995. Soil physical property changes during dung decomposition in a
770 tropical pasture. *Soil Science Society of America Journal* 59, 908-912.

771 Hiernaux, P., Ayantunde, A., Kalilou, A., Mougin, E., G rard, B., Baup, F., Grippa, M.,
772 Djaby, B., 2009. Trends in productivity of crops, fallow and rangelands in Southwest
773 Niger: Impact of land use, management and variable rainfall. *Journal of Hydrology* 375,
774 65-77.

775 Hiernaux, P., Biolders, C.L., Valentin, C., Bationo, A., Fernandez-Rivera, S., 1999. Effects of
776 livestock grazing on physical and chemical properties of sandy soils in Sahelian
777 rangelands. *Journal of Arid Environments* 41, 15.

778 Hiernaux, P., Diawara, M., Gangneron, F., 2014. Quelle accessibilit  aux ressources
779 pastorales du Sahel ? L' levage face aux variations climatiques et aux  volutions des
780 soci t s sah liennes. *Afrique contemporaine* 249, 21-35.

781 Hiernaux, P., Le Houerou, H.N., 2006. Les parcours du Sahel. *S cheresse* 17, 51-71.

782 Hiernaux, P., M.I., C., L., D., P.N., D.L., 1994. Fluctuations saisonnières de la feuillaison des
783 arbres et des buissons sahéliens. Conséquences pour la quantification des ressources
784 fourragères. *Revue d'Elevage et de Medecine Veterinaire des Pays Tropicaux* 17, 9.

785 IPCC, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Prepared by the
786 National Greenhouse Gas Inventories Programme, Report IPCC. IPCC,
787 SWITZERLAND, p. 307.

788 IPCC, 2013. Climate Change 2013: The Physical Science Basis. Contribution of Working
789 Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change
790 in: Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels,
791 Y. Xia, V. Bex and P.M. Midgley (Ed.), Cambridge University Press, Intergovernmental
792 Panel on Climate Change ed. Cambridge, Cambridge, United Kingdom and New York,
793 NY, USA, p. 1535.

794 Jamali, H., Livesley, S.J., Grover, S.P., Dawes, T.Z., Hutley, L.B., Cook, G.D., Arndt, S.K.,
795 2011. The Importance of Termites to the CH₄ Balance of a Tropical Savanna Woodland of
796 Northern Australia. *Ecosystems* 14, 698-709.

797 Jamali, H., Livesley, S.J., Hutley, L.B., Fest, B., Arndt, S.K., 2013. The relationships between
798 termite mound CH₄/CO₂ emissions and internal concentration ratios are species specific.
799 *Biogeosciences* 10, 2229-2240.

800 Jones, D.L., Nguyen, C., Finlay, R.D., 2009. Carbon flow in the rhizosphere: carbon trading
801 at the soil–root interface. *Plant and Soil* 321, 5-33.

802 Kalbitz, K., Solinger, S., Park, J.-H., Michalzik, B., Matzner, E., 2000. Controls on the
803 dynamics of dissolved organic matter in soils: A review. *Soil Science* 165, 277-304.

804 Kindler, R., Siemens, J.A.N., Kaiser, K., Walmsley, D.C., Bernhofer, C., Buchmann, N.,
805 Cellier, P., Eugster, W., Gleixner, G., GrÜNwald, T., Heim, A., Ibrom, A., Jones, S.K.,
806 Jones, M., Klumpp, K., Kutsch, W., Larsen, K.S., Lehuger, S., Loubet, B., McKenzie, R.,
807 Moors, E., Osborne, B., Pilegaard, K.I.M., Reibmann, C., Saunders, M., Schmidt, M.W.I.,
808 Schrumpf, M., Seyfferth, J., Skiba, U.T.E., Soussana, J.-F., Sutton, M.A., Tefs, C.,
809 Vowinckel, B., Zeeman, M.J., Kaupenjohann, M., 2011. Dissolved carbon leaching from
810 soil is a crucial component of the net ecosystem carbon balance. *Global Change Biology*
811 17, 1167-1185.

812 Kotir, J.H., 2011. Climate change and variability in Sub-Saharan Africa: a review of current
813 and future trends and impacts on agriculture and food security. *Environ Dev Sustain* 13,
814 587-605.

815 Le Houerou, H.N., 1989. The grazing land ecosystems of the African Sahel. Springer-Verlag,
816 Berlin ; New York.

817 Lesnoff, M., Corniaux, C., Hiernaux, P., 2012. Sensitivity analysis of the recovery dynamics
818 of a cattle population following drought in the Sahel region. *Ecological Modelling* 232, 28-
819 39.

820 Lipper, L., Dutilly-Diane, C., McCarthy, N., 2010. Supplying Carbon Sequestration From
821 West African Rangelands: Opportunities and Barriers. *Rangeland Ecology & Management*
822 63, 155-166.

823 Marion, P., Veronique, L., Edgar, T., Jeremy, C., MinSheng, K., Vun, K.C., Maria, P.,
824 Robert, M.E., 2015. Deadwood biomass: an underestimated carbon stock in degraded
825 tropical forests? *Environmental Research Letters* 10, 044019.

826 McGahey, D., Davies, J., Hagelberg, N., Ouedraogo, R., 2014. Pastoralism and the Green
827 Economy – a natural nexus? IUCN and UNEP, Nairobi, p. 72.

828 Nassef, M., Anderson, S., Hesse, C., 2009. Pastoralism and climate change. Enabling adaptive
829 capacity. Humanitarian Policy Group. Overseas Development Institute. London. 35pp.

830 Nielsen, T.T., Rasmussen, K., Mbow, C., Touré, A., 2003. The fire regime of Senegal and its
831 determinants. *Geografisk Tidsskrift-Danish Journal of Geography* 103, 43-53.

832 Njoya, A., Bouchel, D., Ngo Tama, A.C., Moussa, C., Martrenchar, A., Letenneur, L., 1997.
833 Systèmes d'élevage et productivité des bovins en milieu paysan, in: Seiny-Boukar Lamine,
834 P.J.-F.F.G. (Ed.), *Atelier d'échange agricultures des savanes du nord-Cameroun : vers un*
835 *développement solidaire des savanes d'Afrique centrale*. CIRAD, CMR, pp. 109-121.

836 Pelster, D.E., Gisore, B., Goopy, J., Korir, D., Koske, J.K., Rufino, M.C., Butterbach-Bahl,
837 K., 2016. Methane and Nitrous Oxide Emissions from Cattle Excreta on an East African
838 Grassland. *Journal of Environmental Quality*, 9.

839 Penning de Vries, F., Djiteye, M., 1982. The productivity of Sahelian rangeland: a study of
840 soils, vegetation and the exploitation of this natural resource. Centre for Agricultural
841 Publishing and Documentation, Wageningen, The Netherlands, p. 547.

842 Petersen, S.O., Blanchard, M., Chadwick, D., Del Prado, A., Edouard, N., Mosquera, J.,
843 Sommer, S.G., 2013. Manure management for greenhouse gas mitigation. *Animal : an*
844 *international journal of animal bioscience* 7, 266-282.

845 Pollard, J.H., 1971. On Distance Estimators of Density in Randomly Distributed Forests.
846 *Biometrics* 27, 991-1002.

847 Rakotovo, N.H., Razafimbelo, T.M., Rakotosamimanana, S., Randrianasolo, Z.,
848 Randriamalala, J.R., Albrecht, A., 2017. Carbon footprint of smallholder farms in Central

849 Madagascar: The integration of agroecological practices. *Journal of Cleaner Production*
850 140, 1165-1175.

851 Rosenstock, T., Rufino, M., Butterbach-Bahl, K., Wollenberg, E., 2013. Toward a protocol
852 for quantifying the greenhouse gas balance and identifying mitigation options in
853 smallholder farming systems. *Environmental Research Letters* 8, 021003.

854 Six, J., Callewaert, P., Lenders, S., De Gryze, S., Morris, S., Gregorich, E., Paul, E., Paustian,
855 K., 2002. Measuring and understanding carbon storage in afforested soils by physical
856 fractionation. *Soil science society of America journal* 66, 1981-1987.

857 Soussana, J.F., Loiseau, P., Vuichard, N., Ceschia, E., Balesdent, J., Chevallier, T., Arrouay,
858 D., 2004. Carbon cycling and sequestration opportunities in temperate grasslands. *Soil use*
859 *and management* 20, 12.

860 Soussana, J.F., Tallec, T., Blanfort, V., 2010. Mitigation the greenhouse gas balance of
861 ruminant production systems through carbon sequestration in grasslands. *Animal : an*
862 *international journal of animal bioscience* 4, 33.

863 Sy, O., 2010. Aménagements Hydrauliques et Vulnérabilité de L'élevage Transhumant au
864 Ferlo (Sénégal). *Sustentabilidade em Debate* 1, 47-60.

865 Tagesson, T., Fensholt, R., Cropley, F., Guiro, I., Horion, S., Ehammer, A., Ardö, J., 2015a.
866 Dynamics in carbon exchange fluxes for a grazed semi-arid savanna ecosystem in West
867 Africa. *Agriculture, Ecosystems & Environment* 205, 15-24.

868 Tagesson, T., Fensholt, R., Guiro, I., Rasmussen, M.O., Huber, S., Mbow, C., Garcia, M.,
869 Horion, S., Sandholt, I., Holm-Rasmussen, B., Götsche, F.M., Ridler, M.-E., Olén, N.,
870 Lundegard Olsen, J., Ehammer, A., Madsen, M., Olesen, F.S., Ardö, J., 2015b. Ecosystem
871 properties of semiarid savanna grassland in West Africa and its relationship with
872 environmental variability. *Global Change Biology* 21, 250-264.

873 Takimoto, A., Nair, V.D., Nair, P.K.R., 2008. Contribution of trees to soil carbon
874 sequestration under agroforestry systems in the West African Sahel. *Agroforest Syst* 76,
875 11-25.

876 Traoré, S., Nygård, R., Guinko, S., Lepage, M., 2008. Impact of *Macrotermes termitaria* as a
877 source of heterogeneity on tree diversity and structure in a Sudanian savannah under
878 controlled grazing and annual prescribed fire (Burkina Faso). *Forest Ecology and*
879 *Management* 255, 2337-2346.

880 Turner, M.D., McPeak, J.G., Ayantunde, A., 2014. The Role of Livestock Mobility in the
881 Livelihood Strategies of Rural Peoples in Semi-Arid West Africa. *Human Ecology* 42,
882 231-247.

883 Valentini, R., Arneth, A., Bombelli, A., Castaldi, S., Cazzolla Gatti, R., Chevallier, F., Ciais,
884 P., Grieco, E., Hartmann, J., Henry, M., Houghton, R.A., Jung, M., Kutsch, W.L., Malhi,
885 Y., Mayorga, E., Merbold, L., Murray-Tortarolo, G., Papale, D., Peylin, P., Poulter, B.,
886 Raymond, P.A., Santini, M., Sitch, S., Vaglio Laurin, G., van der Werf, G.R., Williams,
887 C.A., Scholes, R.J., 2014. A full greenhouse gases budget of Africa: synthesis,
888 uncertainties, and vulnerabilities. *Biogeosciences* 11, 381-407.

889 Vayssières, J., Assouma, M.H., Lecomte, P., Hiernaux, P., Bourgoïn, J., Jankowski, F.,
890 Corniaux, C., Vigne, M., Torquebiau, E., Ickowicz, A., 2017. Livestock at the heart of
891 ‘climate-smart’ landscapes in West Africa, in: Caron, P., Valette, E., Wassenaar, T.,
892 Coppens, D.E.G., Vatché, P. (Eds.), *Living territories to transform the world*, Quae ed,
893 Versailles pp. 111-117.

894 Whitmore, A., 2001. Impact of livestock on soil. *Landbauforschung Volkenrode FAL*
895 *Agricultural Research, Sonderheft* 226, 39-41.

896 Woomer, P.L., Tieszen, L.L., Tappan, G., Touré, A., Sall, M., 2004. Land use change and
897 terrestrial carbon stocks in Senegal. *Journal of Arid Environments* 59, 625-642.

898

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

Variations in fluxes/Stock	July.	Aug.	Sept.	Oct.	Nov.	Dec.	Jan.	Feb.	Mar.	Apr.	May	June	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

901 **Figure captions**

902

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

905

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

909

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

912

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

915

916 **Figure 5.** Total GHG emissions at pastoral ecosystem scale ranked according to the category
917 of emission (in CO₂-eq).

918

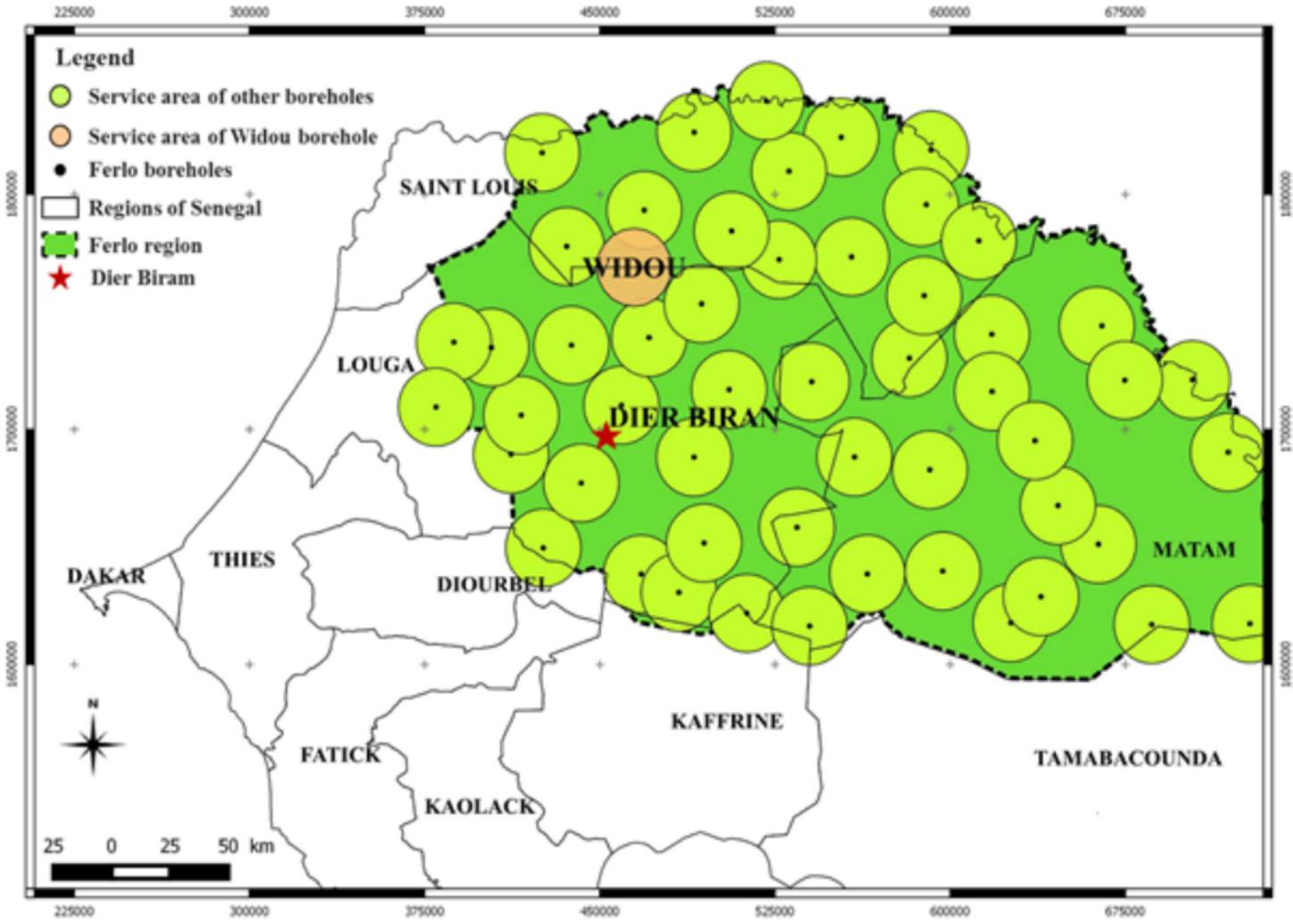
919 **Figure 6.** Temporal variability of C stock at pastoral ecosystem scale

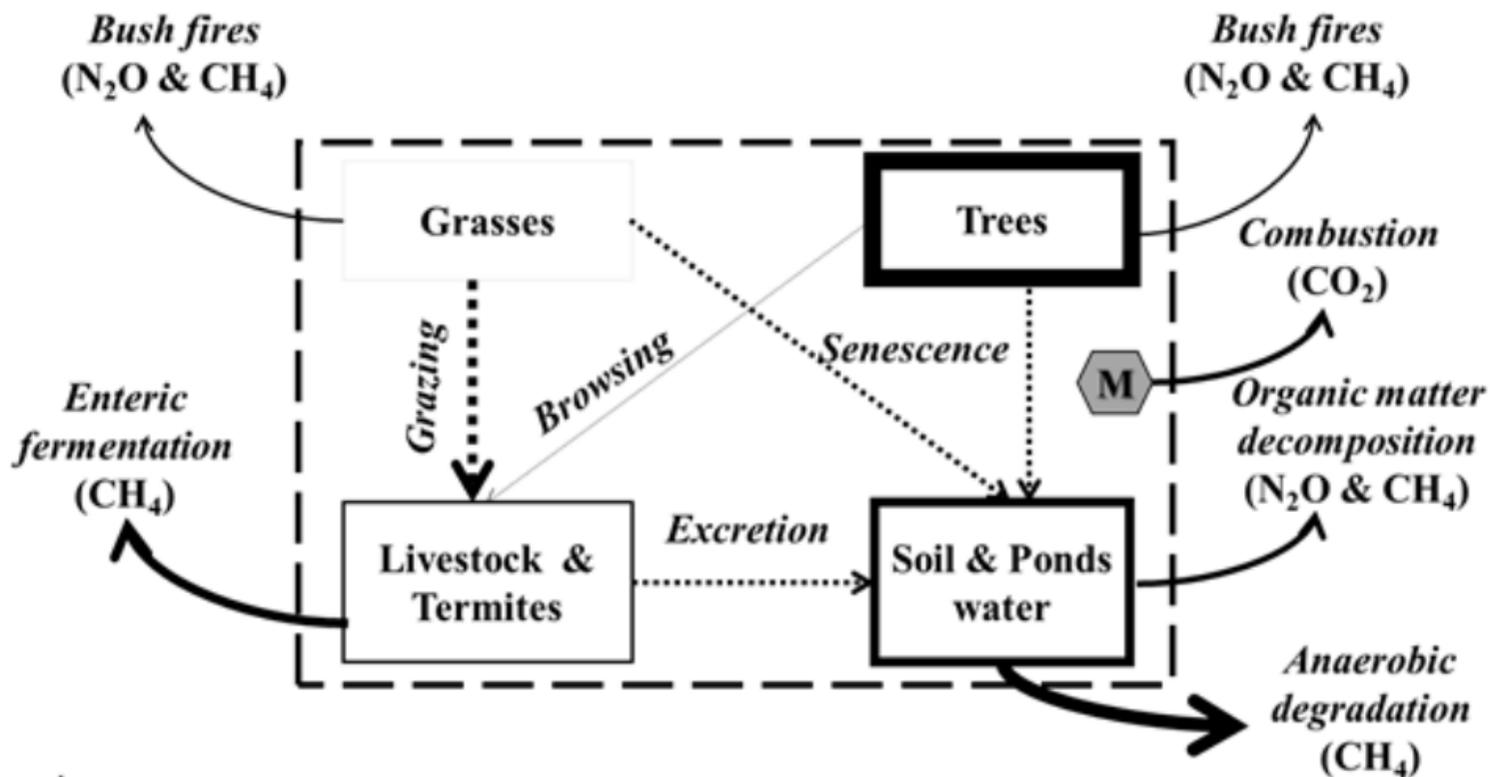
920

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

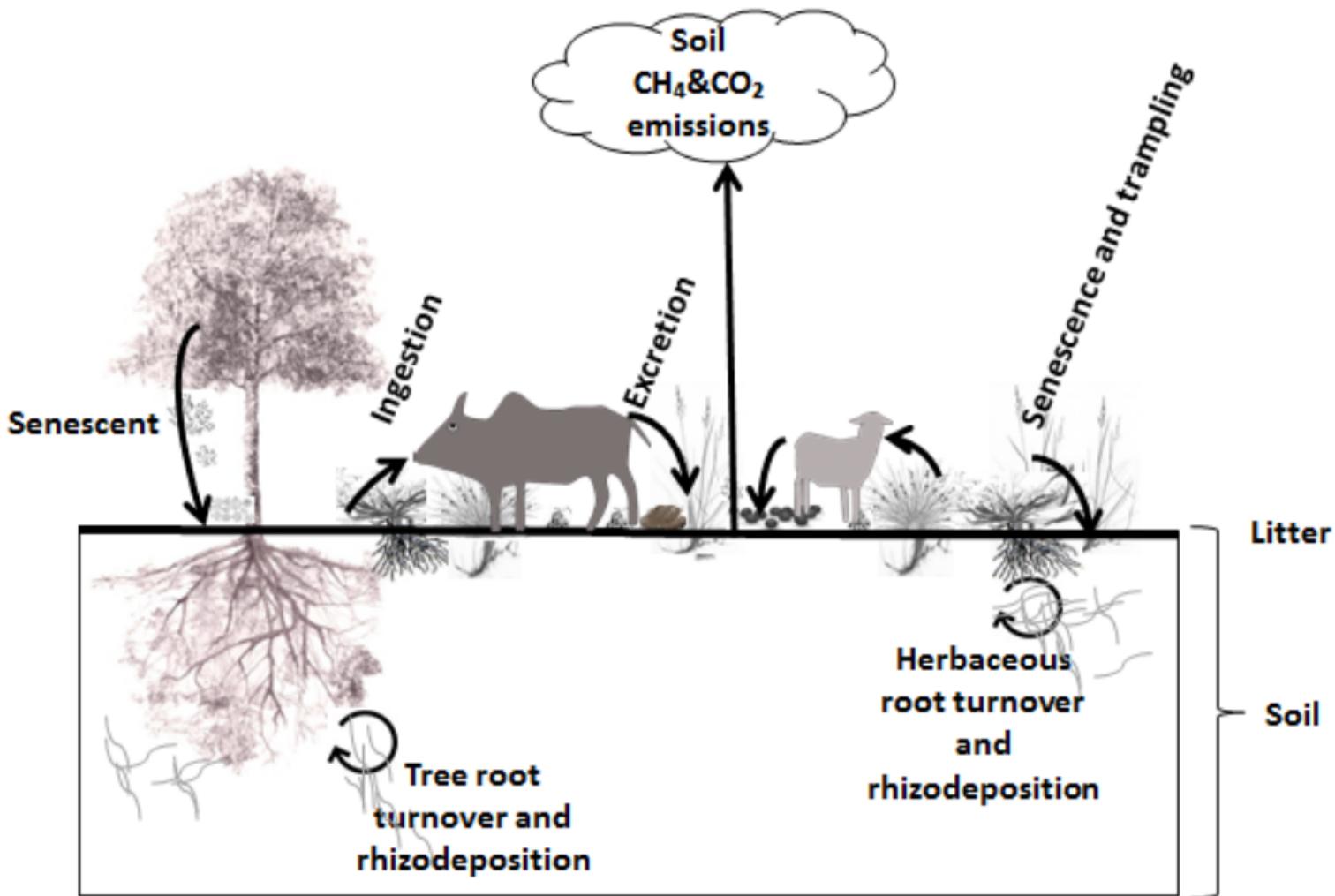
922

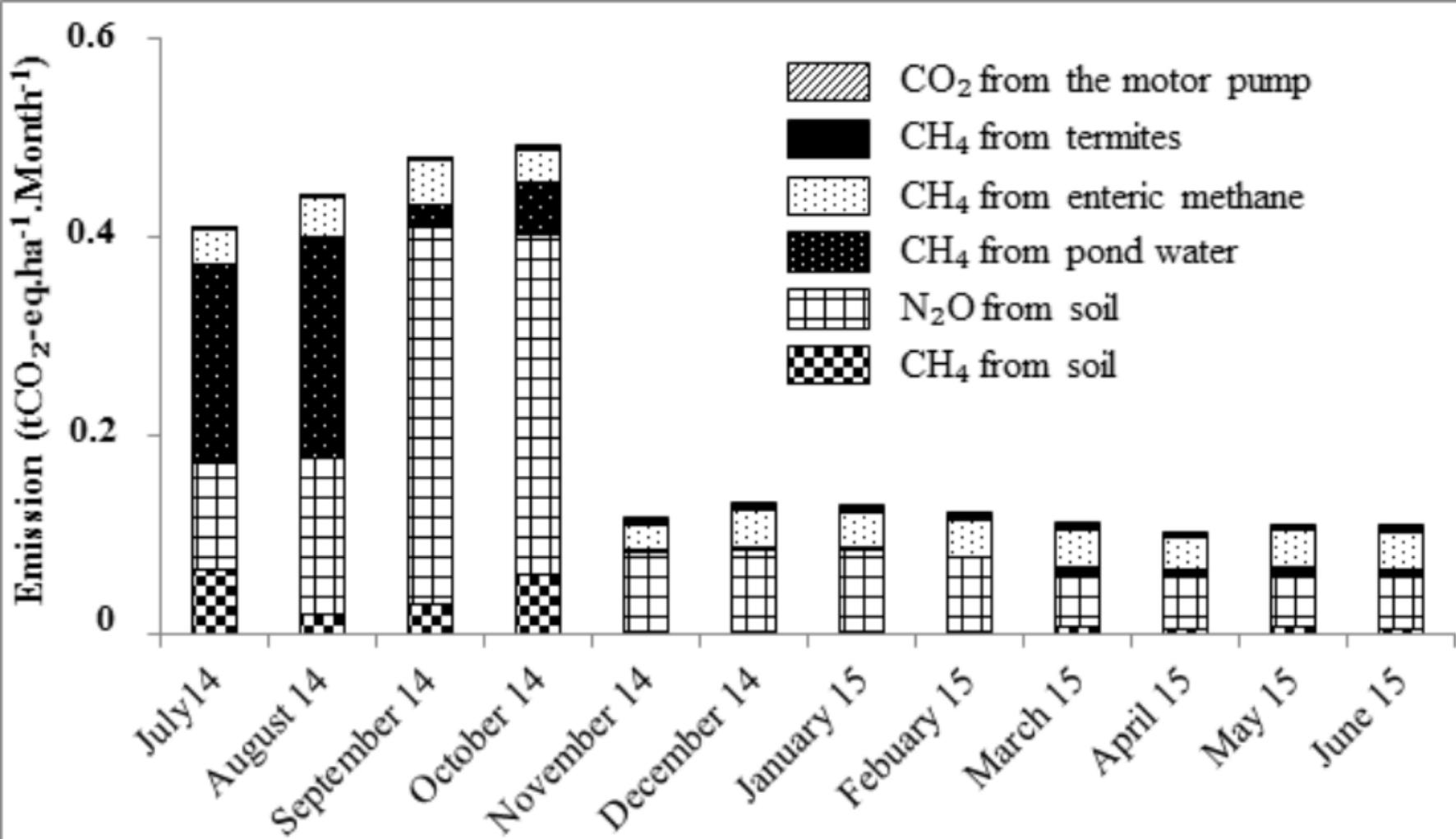
923 **Figure 8.** Monthly variations in the C balance at the WTSA ecosystem scale balancing GHG
924 emissions (from soil, water, livestock, termites and motor pump) and C accumulation (in
925 plants, soil and livestock).

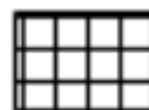
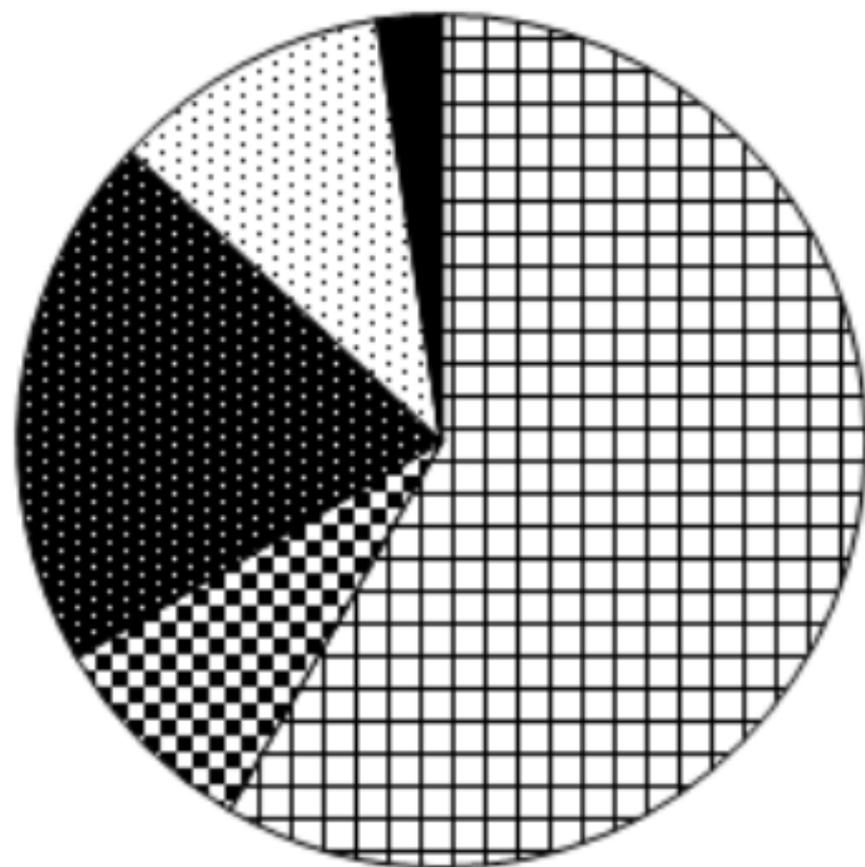




- ↷ : Gaseous emissions
 ⋯→ : Organic C&N flows
 ⬡ : Stock
 - - - : Limits of the rangeland ecosystem
 ⬡ M : Motor pump
 Process



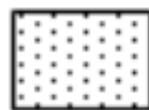




N₂O from soil (58%)



CH₄ from pond water (24%)



CH₄ from enteric methane (11%)



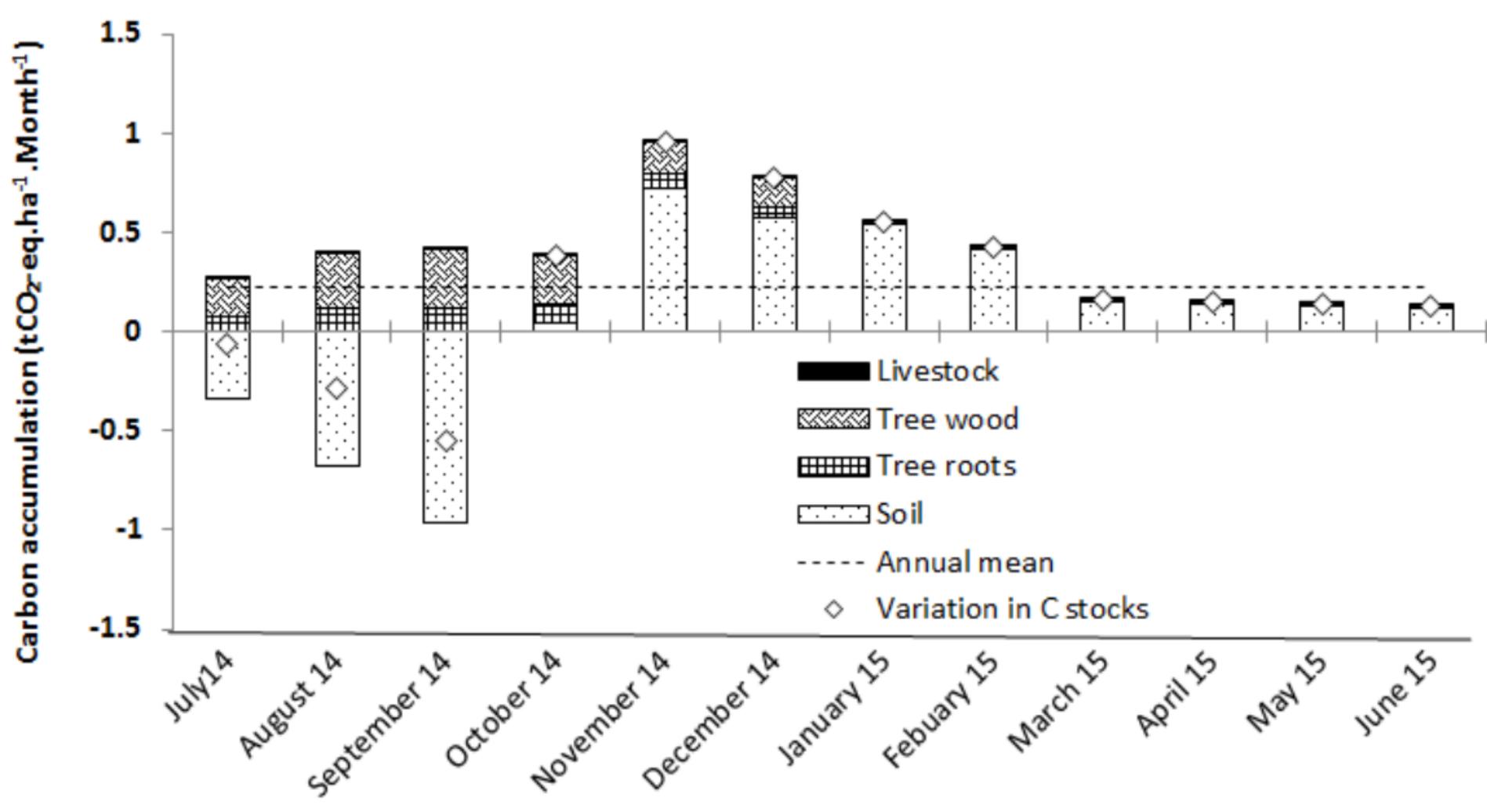
CH₄ from soil (8%)

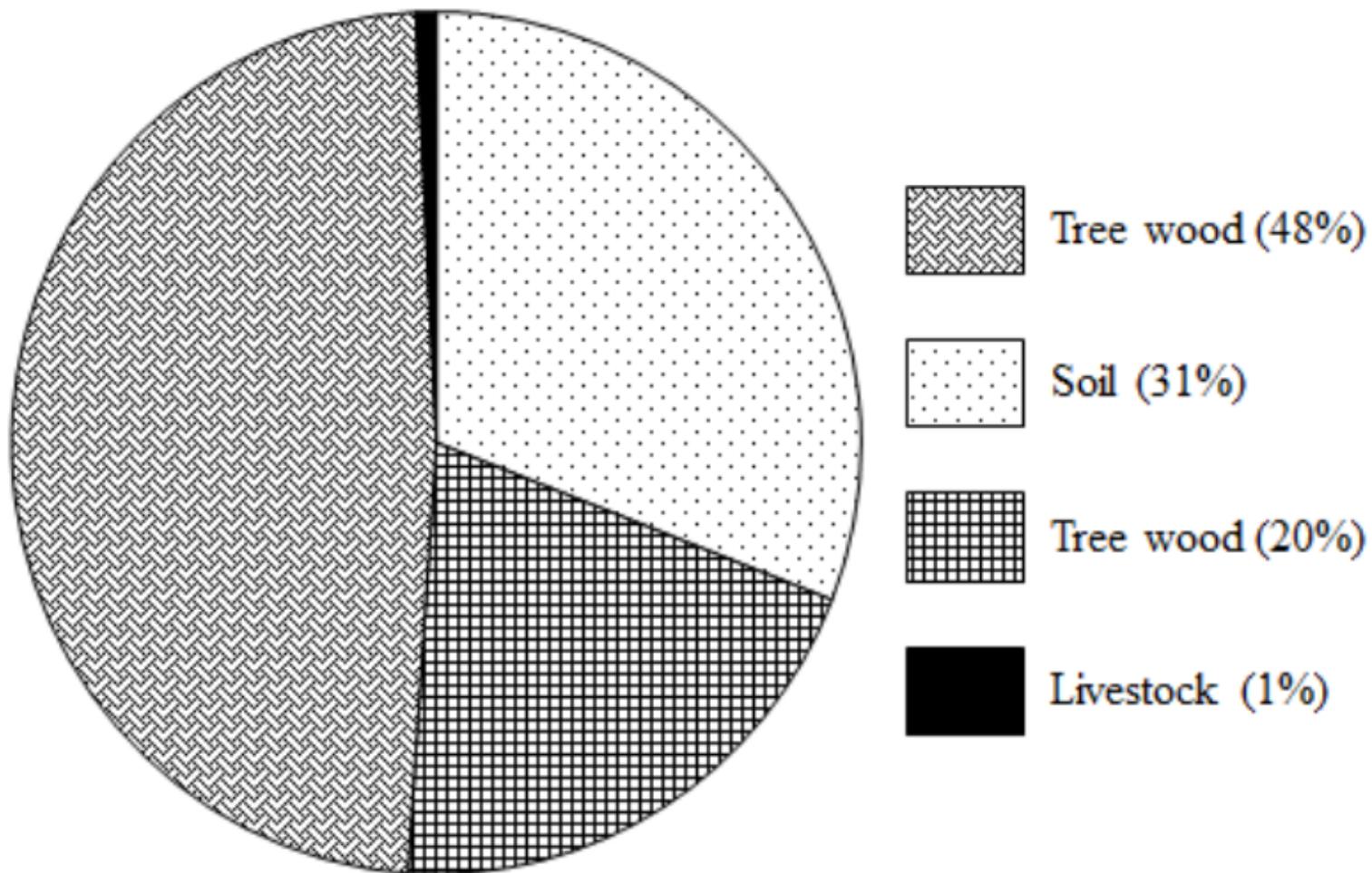


CH₄ from termites (3%)



CO₂ from the motor pump (<1%)





Stock/fluxes ($\text{tCO}_2\text{-eq.ha}^{-1}.\text{Month}^{-1}$)

