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Carbon balance of an intensively managed pasture: methodology, evaluation and impacts of weather conditions and grazing strategy.

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Abstract

Grasslands are an important component of the global terrestrial carbon (C) balance and are widely used for grazing around the world. Because of their C sequestration potential, grasslands are often seen as an important way to mitigate CH_4 and N_2O emissions associated with cattle production systems. However, nowadays, grassland C sequestration potential is still highly uncertain because C sequestration processes are highly affected by soil types and weather conditions. Grazing also affects the C cycle in grasslands through plants consumption, cattle respiration, natural fertilization through excreta, and soil compaction. In addition, other management practices such as fertilization, biomass harvesting and manure spreading can also have an influence on grasslands C sequestration potential.

In southern Belgium, cattle production is an important component of the agricultural sector with grassland covering around 45% of the utilized agricultural lands. The Belgian Blue is a famous breed that is widely used in Wallonia for beef production. In this context, because of the importance of pastures in Walloon production systems, this work focuses on computing a complete C budget of a grazed permanent grassland in relation with weather variability, grazing and management. The studied pasture is a permanent pasture, part of a commercial farm located in Dorinne and grazed by Belgian blue for more than 50 years. The productivity of the pasture is enhanced using mineral and organic fertilizers following usual management practices of the region.

The main objectives of this work were to build a robust methodology to build a complete C budget at the pasture scale and to quantify the soil C content variations and assess its uncertainties. To do so eddy covariance (EC) CO_2 flux measurements were carried out during 5 years in addition to non CO_2 C flux measurements in order to obtain a complete C budget at the pasture scale. We present the 5-years measurements based C budget and its uncertainties. The results showed that, despite the high stocking rate, the old age of the pasture and weather conditions variations, the site acted as a relatively stable CO_2 sink (net ecosystem exchange, NEE) that was further enhanced by lateral organic C fluxes as C imports (organic fertilization, feed complements) were higher than C exports in form of meat. As result, on average over 5 years, the site acted as a net C sink with net C sequestration rate of -100 ± 50 g C m^{-2} yr^{-1}. To go further, this C sequestration rate was compared to the CH_4 emissions of the cattle which were estimated from EC measurements in a separate work as well as to IPCC tier 1 N_2O emission estimates. The results showed that around 70% of the emitted CH_4 and N_2O were offset by C sequestration. However, this greenhouse gas budget was only computed at the pasture scale and does not account for other emissions at the barn or in the crop. Hints are given to extend this greenhouse gas budget to the entire farm in further work.

We also investigated the impact of rotational (RG) and continuous grazing (CG) on NEE dynamics and annual values, by measuring CO_2 fluxes using eddy covariance in two adjacent pastures during a complete grazing season. The results showed that NEE dynamics were greatly impacted by the grazing method.

Following grazing events on the RG parcel, net CO_2 uptake on the RG parcel was greatly reduced compared to the CG parcel. During the following rest periods, this phenomenon progressively shifted towards a higher assimilation. This behavior was attributed to sharp biomass changes in the RG treatment and therefore sharp changes in plant photosynthetic capacity. In terms of annual NEE values, no significant difference between the two treatments was observed.

During this work, we also highlighted two important methodological issues. The first one was associated with the high frequency loss correction of the eddy covariance CO_2 fluxes. We showed that the choice of the cospectrum used to implement this correction had great influence on NEE estimates which was an important component of the C budget. We compared two approaches to do this high frequency loss correction based on either local (sensible heat) cospectra or well-known Kansas cospectra models. We found that the local cospectrum differed from the Kansas cospectrum shape leading to very different correction factors. Night fluxes measured by eddy covariance were found to be in good agreement with chamber based ecosystem respiration estimates when corrected with local cospectra and to be overestimated when corrected with Kansas cospectra. The resulting error acts as a selective systematic error on annual NEE that was as high as 71-150 g C m^{-2} yr^{-1}.

The second methodological issue highlighted concerns the way animal respiration is accounted for in annual C budgets. Using EC for flux measurements, cow respiration may or may not be added to soil and vegetation exchanges depending on their location respective to the area influencing the measurements (called footprint). It is often hypothesized that, over a year, cattle are, on average, distributed evenly over the field so that their respiration is measured in a representative way by the EC flux tower. We tested this hypothesis by comparing daily cow respiration rate per livestock unit estimated by postulating a homogeneous cow repartition over the whole pasture with three other estimates based on animal localization data, animal scale carbon budget and confinement experiments. The study showed that cow respiration was under estimated with a bias of around 60 g C m^{-2} yr^{-1} because of low cow presence in the footprint especially during the night. Consequently, we propose to compute separately cow-free NEE and cow's respiration. For the former we propose a method based on cattle presence detection using CH_4 fluxes, elimination of data with cattle and gap filling on the basis of data without cattle. For the latter we presented and discussed three independent methods (animal localization with GPS, animal scale carbon budget, confinement experiments) to estimate the cattle respiration rate.

Résumé

Les prairies sont une composante importante du bilan terrestre mondial de carbone (C) et sont utilisées comme pâturage dans le monde entier. En raison de leur potentiel de séquestration du carbone, les prairies sont souvent considérées comme un moyen important d'atténuer les émissions de CH_4 et de N_2O associées aux systèmes de production bovin. Cependant, de nos jours, le potentiel de piégeage du carbone dans les prairies est encore très incertain. De plus, les bilans C des prairies sont très variables en fonction des conditions pédoclimatiques et de leur gestion. En effet, les processus annuels de séquestration du carbone sont fortement influencés par des variables météorologiques telles que la température, les précipitations et le rayonnement. On sait aussi que le pâturage affecte directement les échanges de CO_2 dans les prairies via la consommation de plantes, la respiration du bétail, la fertilisation naturelle et le compactage du sol. De plus, d'autres pratiques de gestion comme la fertilisation minérale, la récolte et la fertilisation organique peuvent aussi avoir une influence sur le bilan carbone des prairies.

Dans le sud de la Belgique, la production bovine est une composante importante du secteur agricole avec des prairies couvrant environ 45% des terres agricoles utilisées. La vache Blanc Bleu Belge est une race réputée et largement utilisée en Wallonie pour la production de viande bovine. Au vu de l'importance des pâturages dans les systèmes de production wallons, ce travail se concentre sur le bilan carbone d’une prairie à l'échelle de l’écosystème prairial en relation avec les conditions météorologiques, le pâturage et la gestion. La pâture étudiée est unee prairie permanente au sein d'une ferme commerciale située à Dorinne et pâturée par des vaches Blanc Bleu Belges depuis plus de 50 ans. La prairie est fertilisée avec des engrais minéraux et organiques selon les pratiques de gestion habituelles de la région.

L'objectif principal de ce travail était d'élaborer une méthodologie robuste pour établir un bilan C complet à l'échelle de la prairie afin de quantifier les variations du stock de C du sol et d'évaluer ses incertitudes. Pour ce faire, des mesures de flux de CO_2 par covariance de turbulence ont été effectuées sur une durée de 5 ans combinées à des mesures flux latéraux de carbone dans le but d'obtenir un bilan C complet à l'échelle de la prairie. Nous présentons le bilan carbone basé sur 5 années de mesure ainsi que ses incertitudes. Les résultats ont montré que, malgré la charge en bétail élevée, l’âge de la prairie et les variations météorologiques, le site a agi comme un puits de CO_2 (échange net de l’écosystème) relativement stable avec un taux de séquestration de carbone moyen de -100 ± 50 g C m^{-2} ans^{-1}. Pour aller plus loin, ce chiffre a été comparé aux émissions de CH_4 des animaux mesurées sur le site et aux émissions de N_2O estimées via la méthode tier 1 du proposée par le GIEC. Environ 70% des émissions de CH_4 et N_2O sont compensées par la séquestration de C. Ce bilan gaz à effet de serre ne tient toutefois pas en compte les émissions à l’étable et celles associées aux cultures servant à nourrir les animaux.

Nous avons également étudié l'impact du pâturage rotatif (RG) et du pâturage continu (CG) sur la dynamique et les valeurs annuelles de NEE, en mesurant les flux

de CO_2 par covariance turbulence dans deux pâturages adjacents et ce, pendant une saison complète de pâturage. Les résultats ont montré que la dynamique de la NEE est fortement influencée par le mode de pâturage. Après chaque confinement sur la parcelle RG, l'absorption nette de CO_2 était considérablement réduite par rapport à la parcelle CG. Ensuite, au cours des périodes de récupération suivantes, l’échange a progressivement basculé vers une assimilation plus élevée. Ce comportement a été attribué aux changements soudains de la biomasse aérienne dans le traitement RG qui ont causé d’importantes modifications de la capacité photosynthétique de la plante. En ce qui concerne les valeurs annuelles de la NEE, aucune différence significative n'a été observée entre les deux traitements.

Pendant ce travail, nous avons également indentifié deux points méthodologiques importants. Le premier est associé à la correction en fréquence des flux de CO_2 mesurés par covariance de turbulence. Nous avons montré que le choix du cospectre utilisé pour mettre en œuvre cette correction avait une grande influence sur les estimations annuelles de la production nette de CO_2 de l'écosystème (PNE) qui constitue une composante importante du bilan carbone. Nous avons comparé deux approches pour faire cette correction basées soit sur un cospectre local (chaleur sensible) soit sur le cospectre de Kansas. Nous avons constaté que la forme du cospectre local différait de la forme du cospectre de Kansas, entraînant des facteurs de correction très différents. Les flux nocturnes mesurés par covariance de turbulence se sont révélés en accord avec les mesures de respiration réalisées à l’aide de chambre lorsqu'elles sont corrigées avec un cospectre local et surestimées lorsqu'ils sont corrigés avec le cospectre de Kansas. L'erreur qui en résulte agit comme une erreur systématique sélective et a un impact important sur les flux annuels de carbone qui atteignaient 71-150 g C m^{-2} an^{-1}.

La seconde question méthologique concerne la manière d’inclure la respiration du bétail dans le bilan. Dans les prairies pâturées, la respiration totale de l'écosystème correspond à la somme de la respiration du sol et de la végétation et de la respiration des animaux. Il est souvent fait l'hypothèse que, sur une année, les bovins sont en moyenne répartis uniformément sur le terrain, de sorte que leur respiration est mesurée de manière représentative par la tour à flux. Nous avons testé cette hypothèse en comparant le taux de respiration quotidien des vaches par unité de bétail, estimé en postulant une répartition homogène des vaches sur l'ensemble du de la pâture, avec trois autres estimations basées sur des données de localisation du bétail, le bilan carbone à l'échelle d’une bête et des expériences de confinement. L'étude a montré que la respiration annuelle des vaches était sous-estimée avec un biais d'environ 60 g C m^{-2} an^{-1} en raison de la faible présence des vaches dans le footprint de la tour, surtout pendant la nuit. Par conséquent, nous préconisons de calculer séparément la PNE et la respiration des animaux. Pour la première citée, nous proposons une méthode basée sur la détection de la présence du bétail à l'aide de flux de CH_4, l'élimination des données en présence le bétail et le comblement des données manquantes. Pour la seconde, nous présentons et discutons trois méthodes indépendantes (localisation des animaux à l'aide de GPS, bilan carbone à l'échelle de l’animal, expériences de confinement) pour estimer la respiration du bétail.

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NEE: net CO2 ecosystem exchange

TER: total ecosystem respiration

GPP: Gross primary productivity

C: carbon

EC: eddy covariance

FP: footprint

GHG: Greenhouse gas

LU: livestock unit

CG: continuous grazing

RG: rotational grazing

UAL: utilized agricultural land

SOC: soil organic carbon

NBP: net biome productivity

SOM: soil organic matter

R_{cows}: total cow respiration

E_{cow}: daily respiration rate of one livestock unit

SWC: soil water content

Ts: soil temperature

Ta: air temperature

PPFD: photosynthetic photon flux density

HM: herbage mass

DMI: dry matter intake

ANPP: above ground net primary productivity

DTO : Dorinne Terrestrial Observatory

Chapter 1

Introduction

1. Cattle and greenhouse gas emissions

According to the Food and Agriculture Organization (FAO), livestock production is responsible for around 14.5% of worldwide anthropogenic greenhouse gas emissions (Gerber et al., 2013). Around 44% of those emissions are in form of CH_4 while the remaining are distributed between N_2O (29%) and CO_2 (27%). This emission includes emissions associated with feed production (fertilization including production of fertilizers, energy use, land use change…), non-feed production (farm equipment, farm buildings), direct livestock emissions and other post farm activities such as transport to slaughter (for more details see table 1 of Gerber et al., 2013). It however does not include changes in carbon stocks from land use constant management. More specifically cattle dominate the livestock production sector's emissions by contributing to around 65% of the sector's emissions among which 41% is associated with the production of meat (from both beef and dairy cattle) and the remaining with the production of milk.

Nowadays, CH_4 emissions through enteric fermentation represent around 44% of cattle associated emissions (Figure 1-1). Emissions associated to feed production, including pasture management and fertilization represents around 36% of cattle emissions while remaining emissions are associated to land use changes and fossil fuel consumption in the feed supply chain. If those estimates are of course affected by uncertainties, there is no discussion that cattle are major contributors to anthropogenic GHG emissions.

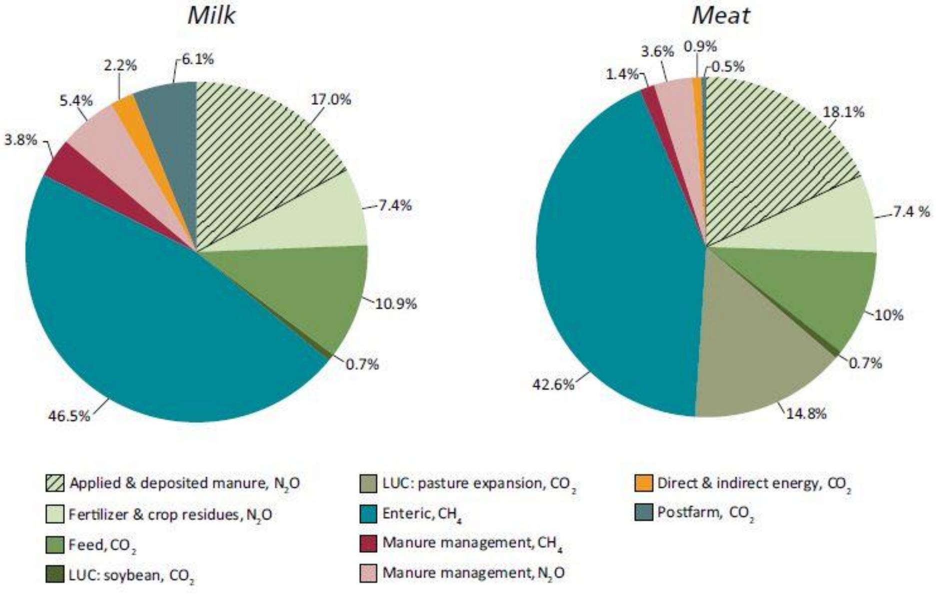

Figure 1-1: Emissions of greenhouse gases associated to the production of meat and milk for cows. Figure taken form Gerber et al., (2013).

Because of these emissions, ruminants are often poorly considered in scientific literature as well by popular media (Garnett et al., 2017). However, an increasing number of studies highlight the fact that the lands grazed by the ruminants are very important carbon stocks and can act as important carbon sinks thereby offsetting a considerable part of the emissions. Improving the management of these grasslands is often seen as a way to mitigate livestock related emissions (Pellerin et al., 2013).

Nonetheless, the emission estimates are highly uncertain (Herrero et al., 2011) for various reasons. Actually, there is a huge diversity in cattle production systems with very different climate, landscape, management, animal feed consumption and animal breed that could result in very different greenhouse gas emissions (Lesschen et al., 2011). Very often, the lack of data and understanding of biological and biophysical processes associated to a specific production system is problematic when trying to find strategies to account its associated emissions (Garnett et al., 2017).

2. Classification of livestock and cattle production systems

In order to narrow things down, livestock systems are often classified into three main categories: mixed crop-livestock systems, landless systems and grazing systems (Garnett et al., 2017; Serré and Steinfeld, 1996):

Mixed crop-livestock systems: From more than 10% to 90% of the dry matter intake of the animals comes from grass. This system is the most frequent in cattle meat and milk production. It is however important to highlight that this category is not well defined in term of animal diet with, for example, grass proportion that can vary from 11% to 90% of the animal feed.

Landless systems: Less than 10% of dry matter fed to animals is farm-produced and annual stocking rates are above 10 livestock units per hectare and per year. These systems are often referred as industrialized or confined systems. This system is of course not specific to ruminants.

Grazing systems: Specific to ruminants, in these systems, more than 90% of dry matter comes from rangelands, pastures and annual forages with average annual stocking rate lower than 10 livestock units per hectare and per year. Less than 10% comes from feed supplements.

Even within those categories, huge variations can still be observed. For example, mixed systems can vary from all farm or locally produced feeds to systems where feeds comes from more distant locations. In the same way, the use of fertilizers is also not well defined. A grazing system could consist of mostly pastures with very low stocking rates and almost no mineral fertilizers while others could be more intensive. Variations in pasture grazing management also exist. Pastures can be continuously grazed with low stocking densities but long grazing periods or rotationally grazed by alternating periods with high stocking densities and rest periods.

Overall, all these variations, in term of management, used grassland surfaces, animal breed but also pedoclimatic conditions, could affect biological and biophysical processes resulting in variable GHG emissions.

3. GHG exchanges associated with cattle production

A typical livestock production farm exchanges all three main GHG that can be separated between several components (Figure 1-2). Only the most important items in term of GHG exchanges are described below.

- Machines emit CO_2 through fuel consumption.
- The stables emit all three GHG mainly because of cattle respiration (CO_2), enteric fermentation (CH_4) and manure decomposition (CH_4 and N_2O).
- The crop field exchanges CO_2 mostly in form of photosynthesis and ecosystem respiration (Moors et al., 2010) and emits N_20 because of crop fertilization (Bouwman et al., 2002). At the field scale, the field also loses C in form of non CO_2 carbon when the crop is harvested and gains C when fertilized with organic fertilizers (Smith et al., 2010). Depending on inputs and outputs, the soil can act as a carbon source or sink (Ceschia et al., 2010a; Ciais et al., 2010).
- The pasture also exchanges CO_2 mostly in form of photosynthesis as well as soil, plants and cattle respiration. At the field scale, the pasture also loses C in form of non CO_2 carbon when the grass is harvested or when milk and meat is exported and gains C when fertilized with organic fertilizers. Depending on inputs and outputs, the soil can act as a carbon source or sink (Soussana et al., 2007, 2010). Cattles emit CH_4 because of enteric fermentation. N_2O is emitted because of grassland fertilization as well as animal droppings induced emissions (Brown et al., 2001; Flechard et al., 2007).
- Manure emits both CH_4 and N_2O (Petersen et al., 2013).

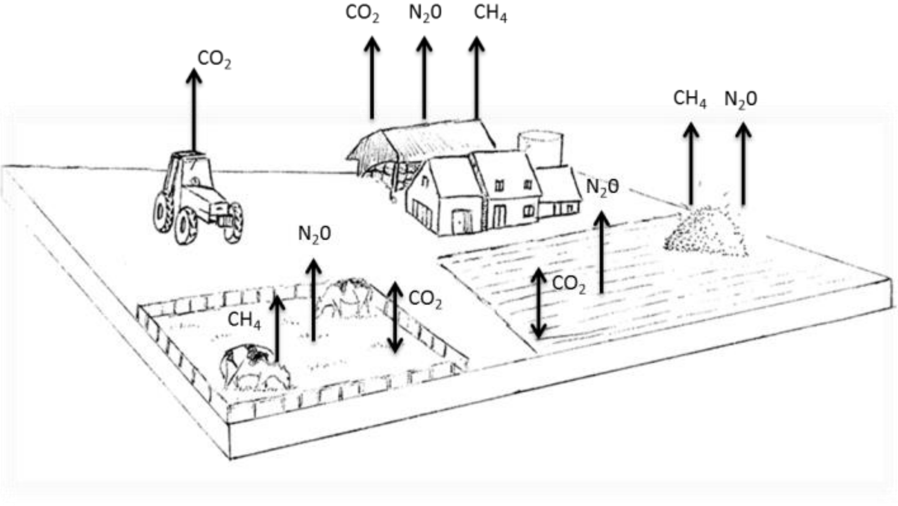

Figure 1-2: Schematic view of the greenhouse gases involved at the farm scale.

The weight of each component in the budget varies according to the farm production system. In landless systems, most of the emissions are located in the stables. Carbon storage in ecosystems is of course totally out of the equation. The GHG budget of a landless exploitation will mostly depend on the breed transformation efficiency and diet (Harper et al., 1999) and the management of manures (Mathot et al., 2012; Petersen et al., 2013). In addition, cattle associated GHG emissions also include off farm emissions linked to the production and the transport of the food given to the animals.

At the opposite, in grazing systems, an important part of the emissions occurs at the pasture. The soil carbon storage (or emission) can become an important component of the GHG budget (Soussana et al., 2010). Lots of variations can still be observed as grazing systems can vary from very extensive pastures with almost no fertilization and very low stocking rates to more intensive ones. These variations can of course affect the GHG budget of the farm in several ways (Allard et al., 2007). An extensive system would use more land and probably induce more CH_4 emissions per unit of production because of low forage quality (Beauchemin et al., 2008). On the other side, the more intensive system would rely more on mineral fertilizers which are energy consuming for their production (Ramírez and Worrell, 2006) and can be associated with N_2O emissions. Both systems could also result in differences in term of soil carbon storage that will be introduced at section 1.6.

Mixed crop-livestock farms are more or less situated between the two other systems. The importance of carbon storage in the total GHG budget depends, in addition to many other variables, on the proportion of grasslands area. The total livestock GHG balance will also depend on the proportion of locally cultivated and imported feed as well as the origin of the latter. As mixed crop-livestock systems are very diverse in many terms, so is the relative importance of the different terms in their GHG budget.

When looking at C storage in grasslands, the role of the grazing lands carbon sink/source activity of the grassland can vary from non-existent in landless system to potentially very important in grazing and mixed systems. Therefore, studies, assessing carbon and GHG of well-defined production systems are therefore much needed in order to better take into account local specificities such as the C absorption potential in grasslands which can greatly vary depending on pedoclimatic conditions and management (Soussana et al., 2007). This is especially important in regions where grasslands are an important component of livestock production systems.

4. Regional context: cattle production in Wallonia

In 2015, GHG emissions from the agricultural sector were estimated to 8.5% (9897 kt CO_2éq) of the total GHG emissions in Belgium at the national level (Belgium's greenhouse gas inventory, 2017). This emission does not include soil carbon stocks variations in agricultural lands which are accounted for in Land Use, Land-Use Change, and Forestry category (LULUCF). It includes enteric fermentation, which was the main source of emissions (46.4%), manure management related emissions (20%), N_2O emissions from soil (32.3%) the remaining being associated with liming activities and urea application. On the other hand, in Wallonia, grassland carbon storage was estimated to 480 kt CO_2 (Agence Wallonne de l'Air et du Climat, personal communication).

In Wallonia, the agricultural sector is highly dominated by cattle breeding and grasslands cover an important part of the utilized agricultural land (UAL). In 2014, more than 69% of the Walloon farmers were possessing bovines among which, 80% were typical Belgian Blue breed suckler cow holders. This represents a total population of 1 150 000 heads with 275 000 suckler cows and 210 000 milk cows (StatBel, 2017). Permanent grasslands cover around 43% (304 400 ha in 2017) of the total region's UAL while temporary grasslands cover around 5% of UAL (StatBel, 2017). These grasslands are mainly used as pastures to produce fresh grass and grass silage to feed the cattle. These numbers highlight the importance of the cattle sector in Wallonia as well as the role of grasslands in production systems. In typical Walloon mixed beef crop-livestock system, around 50% of the feed ratio is in form of fresh grass and 30% in form of grass silage (around 30%) (Cellule d'information viandes, 2017). The rest of the feed consists mainly of maize silage, concentrates, beet pulps and spelt.

5. Carbon sequestration in grasslands

In its narrow sense, grassland can be defined as a ground covered by grasses, with little or no trees. However, unlike forests, grasslands are variously defined (Dixon et al., 2014) which makes quantifying the world grassland cover difficult. According to FAO (Suttie et al., 2005), grasslands covers around 40.5% of worldwide ice-free land area and are therefore one of the most important ecosystem on earth. Among those grasslands, around 2600 Mha (which correspond to around 20% of earth’s land area) are estimated to be grazed by domestic animals (Henderson et al., 2015). More specifically, pastures, are defined as land devoted to the production of introduced or indigenous forage for harvest primarily by grazing (Allen et al., 2011).

Grasslands (as defined by the FAO) are important soil carbon stores that can be estimated to 343 Pg C which is nearly 50% more than worldwide in forests soils (Conant et al., 2017). Carbon can be stored either in its organic form (soil organic carbon, SOC) or in its mineral form when soil minerals contain carbon. Even if, on a long timescale, inorganic carbon can vary in response to management and climate, changes in SOC are much faster in response to management and is what is referred too when talking about carbon sequestration(Post and Kwon, 2000).

To store carbon, plants firstly assimilate CO_2 to build plant tissues and grow (figure 1-3). Then, while most of the absorbed CO_2 is re-emitted through plant autotrophic respiration, some of it is accumulated in their roots and plant litter (Dignac et al., 2017). Some of the assimilated carbon is further re-emitted through soil heterotrophic respiration. If undisturbed, and depending on climate, soil management, soil microbial composition, fertilization, and many other variables, some organic matter can accumulate in the soil (SOM).

Plants bring organic matter to the soil through their plant foliage litter and through their roots turnover but also by rhizodeposition (Jones et al., 2004). Rhizodeposition is a process by which living roots release C compounds directly to the surrounding soil (Jones et al., 2004). In grazed grasslands, organic matter is also brought to the soil in form of animal excretions. A part of this unprotected organic matter is then stabilized into more stable forms following mainly three mechanisms (Jones and Donnelly, 2004; Six et al., 2002). The first one (1) is the stabilization of the SOM through chemical bindings between soil minerals (clay and silt) and SOM (chemical stabilization). The second (2) is the biochemical stabilization caused by the complexing processes between substrates such as lignin and polyphenols and soil particles. Last (3), the SOM is physically protected by soil aggregates that act as barriers against microbes and enzymes. SOM can be protected from decomposition when positioned in pores too small for bacteria and fungi or when inside larger aggregates in anaerobic conditions. In grasslands, the formation of aggregates is favored by worms that mix digested plant residues to soil particles. When compared to arable crops, carbon sequestration is favored in grasslands because more organic carbon is returned to the soil and because a greater part of this carbon is chemically and physically stabilized (Soussana et al., 2004; Soussana and Lemaire, 2014). When stabilized in the deep soil, its residence time may be long (from 1 to >1000 years, (Fontaine et al., 2007)).

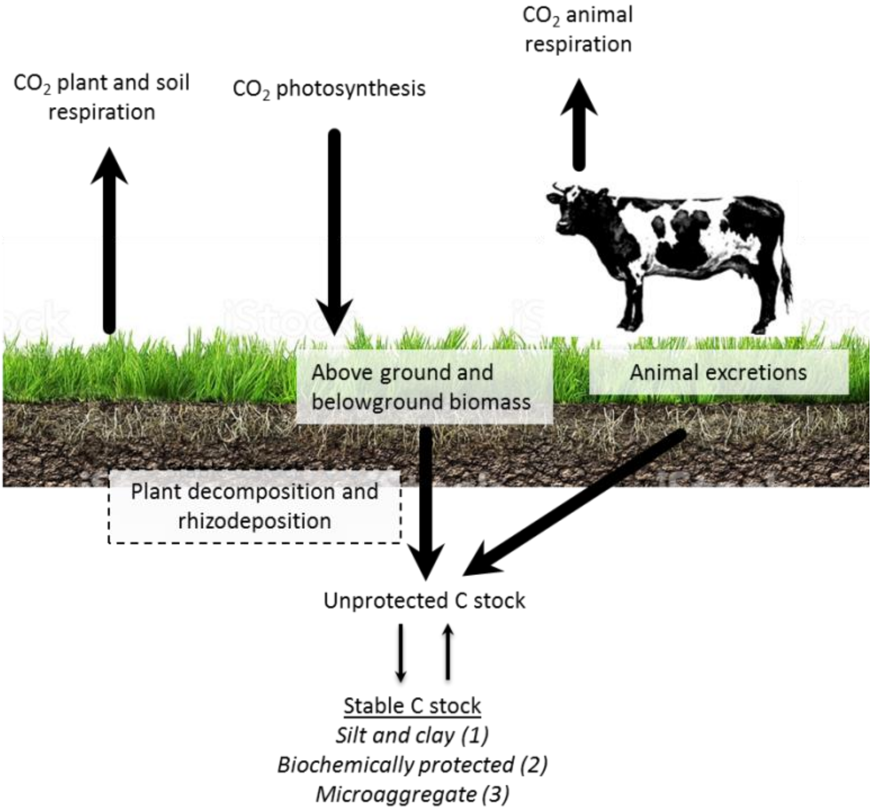

Figure 1-3: Schematic representation of the carbon sequestration process in grassland.

Through these mechanisms, grassland can therefore constitute a net carbon sink towards the atmosphere (Conant et al., 2001; Soussana et al., 2007). This carbon sequestration in grasslands soils could, at least partially, compensate cattle CH_4 emissions as well as N_2O emissions resulting from fertilization and excreta (Hörtnagl et al., 2018; Soussana et al., 2010). However, it is important to emphasize that this carbon storage is fragile and time limited and so is its potential in term GHG emissions mitigation. Indeed, after a few decades, if management and conditions are stable and favorable, carbon stocks will reach an equilibrium (Smith, 2014; Stewart et al., 2007). Added to that, soil carbon sequestration is reversible (Soussana and Lemaire, 2014) so that C can eventually be re-emitted during a particular climatic event (Iii et al., 2008; Reichstein et al., 2013) or other soil disturbance such as pasture renovation (Drewer et al., 2017; Merbold et al., 2014; Rutledge et al., 2014) or land use change from pasture to crop or plantation (Freibauer et al., 2004; Guo and Gifford, 2002; Soussana et al., 2004). Practices like ploughing mixes soil layers and break soil aggregates tend to accelerate SOM decomposition (Conant et al., 2007) .

6. Measuring soil carbon sequestration

Measuring the carbon storage of a pasture require to either measure all the fluxes of carbon entering and leaving the pasture and compute a complete carbon balance or to directly measure changes in soil organic carbon stocks.

6.1. Carbon balance approach

The carbon balance approach consists in measuring all the C fluxes entering and leaving an ecosystem for a certain period. When summing all these fluxes together, the imbalance of the carbon budget (net biome productivity, NBP) corresponds to the soil carbon sink or source activity depending on the sign of the imbalance (Jérôme et al., 2013; Soussana et al., 2007). In this work, fluxes were computed from the atmospheric point of view. As a result, a C flux leaving the ecosystem is accounted positively as a C emission while a flux entering the ecosystem is accounted negatively.

The main fluxes involved in the carbon balance of a grazed pasture are fully described at chapter 3. Here is a brief description (Figure 1-4):

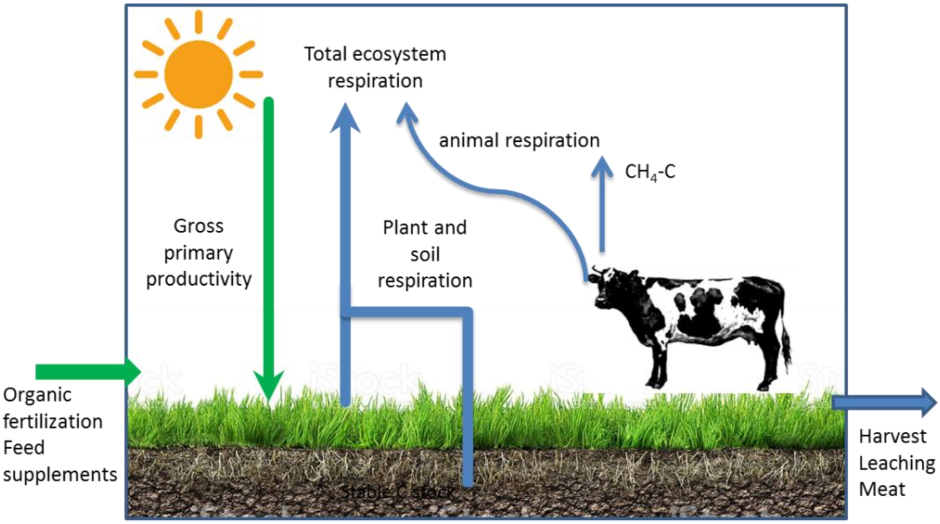

Figure 1-4 : Illustration of the main carbon fluxes involved in the carbon balance of a grazed pasture.

- Gross primary productivity (GPP), which refers the rate at which carbon (CO_2) is fixed through photosynthesis, and total ecosystem respiration (TER), including the cattle respiration. The difference between GPP and TER is the net ecosystem exchange (NEE). NEE is usually measured at the ecosystem scale using the eddy covariance technique described at chapter 2.
- Carbon export through CH_4 emission, mostly emitted by the cows through enteric fermentation.

- Other non CO_2 carbon exports such as the carbon exported through harvest, meat production and leaching.
- Other non CO_2 carbon imports such as feed supplements and manure.

This flux measurement approach allows studying the seasonal, annual and inter-annual variations of C flux dynamics and budgets (Klumpp et al., 2011; Peichl et al., 2011) as well as studying the impact of several management practices and weather variations on the carbon balance with relatively short term experiments (among others : Allard et al., 2007; Merbold et al., 2014; Rutledge et al., 2017a, 2017b). However, flux measurements can potentially be affected by systematic and random errors (Baldocchi, 2003; Richardson et al., 2006) that can significantly affect the overall NBP calculation when summed up. Great care should therefore be taken in order to avoid or correct these errors. In addition, if flux measurements can easily be repeated over time, these cannot be easily spatially repeated. Finally, because of usual inter-annual variability, several years of data are needed in order to provide reliable carbon storage (or emission) estimates. As synthesized in a review (Jérôme et al., 2013), the C balance approach showed that grasslands generally acted as C sinks but with highly variable intensities depending on management and pedoclimatic conditions.

6.2. Soil carbon stock variations

Changes in SOC can also be directly measured in the soil. Because of intrinsic SOC spatial variability, direct SOC measurements requires extensive soil sampling to a depth of 100 cm divided in different layers (Arrouays et al., 2018; Skinner and Dell, 2015). In addition, as the temporal variability is low compared to the SOC spatial variability, the measurements time scale must be relatively long (around a decade) in order to observe significant SOC changes overtime (Goidts and van Wesemael, 2007). It's therefore not easy to understand mechanisms that affect carbon sequestration on shorter timescales. Compared to the carbon balance approach, this technique allows spatially repeated sampling that can be used to better understand C sequestration spatial variability. These kinds of measurements are also potentially less affected by biases and systematic errors when compared to the flux approach. The main source of uncertainty associated may result from bulk density estimates especially in agricultural soils where bulk density may vary because of ploughing (Goidts and van Wesemael, 2007; Wendt and Hauser, 2013). This problem is of course reduced in permanent grasslands.

7. Impact of grazing on soil carbon sequestration

Grazing animals impact the C cycle and the soil carbon both directly and indirectly (Jérôme et al., 2014) animals remove C from the pasture when eating biomass. Most of this C is directly reemitted in form of CO_2 respiration. Only the non-digestible C is returned to the soil as dung. Therefore, increasing the stocking rate of a pasture may lead to increased C losses. When overgrazing is reached, plants are not able to recover from defoliation quickly enough leading to their death. Grazing also impacts plant carbon uptake by reducing their gross primary productivity through defoliation

(Jérôme et al., 2014). On the other hand, at low rate, grazing can stimulate CO_2 uptake by removing plant materials before seed rise. Indeed, as grasses regrow after being eaten, the plant will continue to uptake some CO_2 from the atmosphere. This regrowth of course depends on nutrients, water and N availability and weather conditions.

In certain conditions, grazing animals can also help carbon sequestration by improving nutrient cycling. Indeed, at moderate grazing intensity, excretal returns favor nutrient cycling and increase primary production (De Mazancourt et al., 1998; Soussana and Lemaire, 2014). However, under too high stocking rates, herbivores uncouple C and N cycles leading to environmental problems such as nitrate leaching, N_2O and ammonia emissions (Soussana and Lemaire, 2014). Indeed, when eating biomass, a small amount of the nitrogen contained in the plants is used to produce meat and milk while most of the ingested C is emitted in form of CH_4 or exported as milk. A major parts of N returns to the soil in form of dung (25-40%) and urine (60-85%) (Oenema et al., 1997). N is therefore not returned evenly on the pasture locally decreasing the C:N ratio of organic matter so that the risk of N leaching, ammonia and N_2O emissions are increased under high stocking density.

In term of grazing management, options for the farmers to promote carbon sequestration are, among others to adapt the average stocking rate and grazing schedules or increased grassland species diversity (Rutledge et al., 2017a). Generalizing an ideal stocking rate is probably an impossible task as the optimal stocking rate depends on many variables such as climate, vegetation and soils. Concerning the changes in grazing timings, there is still nowadays no scientific evidence that the use of improved grazing management such as rotational grazing can favor carbon sequestration (Garnett et al., 2017). As a result, local studies are much needed in order to carefully take into accounts local specificities when trying to assess the C sequestration potential of a grazed pasture.

8. Objectives of the project

In order to better include these local specificities when evaluating GHG emissions of the sector, a study assessing the GHG and carbon budget of a representative pasture based Belgian Blue breed farm was much needed. In this context, the project "Establishment of the carbon balance of a Walloon farm practicing the suckling system: effects of climate and grazing management" was carried out. Regarding the importance of pastures in Walloon production systems; this project focuses on carbon and its role in the GHG balance **at the pasture scale** with the following main objectives (Jérôme, 2014):

Objective 1: Establishing a robust methodology to build a complete carbon budget of a grazed pasture and to assess the associated uncertainties.

Objective 2: Building a complete C and CH_4 budget

Objective 3: Testing management strategies that could refine the GHG budget of Walloon production systems thereby contributing to improve these systems durability.

To fulfill these objectives, long term eddy covariance CO_2 fluxes (see chapter 2), biomass and other related C fluxes measurements started in 2010 in an intensively managed pasture grazed by Belgian Blue cows. Farming activities (fertilization, animal stocking rates) on the pasture were carefully monitored and additional micro-meteorological data were also acquired in order to identify the drivers responsible for CO_2 flux variations. Jointly, eddy covariance CH_4 fluxes were measured at the same site (Dumortier et al., 2017). These measurements were carried in order to build a complete C and CH_4 budget of the studied pasture and study the impact of weather variations, grazing (Jérôme et al., 2014) and management activities on the C balance of the pasture.

9. Description of the studied farm

The studied farm is a commercial farm located in the village of Dorinne in the Condroz region. The farm is mixed-crop livestock farm with 45 ha of permanent grasslands and 100 ha of crop fields. Permanent grasslands are mostly used for grazing with occasional harvesting to produce winter forage. Among these cultivated lands, around 7 ha are used for maize production, 2 ha for alfalfa, 2 ha of ray-grass and 12 ha of forage winter cover. All these cited productions are used to feed the cattle. The herd is constituted of 235 Belgian Blue heads with 95 calving per year. During the grazing season, the cows are mostly fed by grazing, additional feeds being given only when grass production is not sufficient. During winter, the animals are fed following the food ration described at table 1-1.

To fulfill the objectives of the project, we selected a pasture of 4.2 ha that is mostly used for grazing during the grazing season. The pasture is (almost) continuously grazed by Belgian Blue cows from March to mid-November with varying starting and ending dates depending on weather conditions. On average, the annual stocking rate was around 2.3 livestock units (LU) per ha and per year with relatively small between-year variations. The pasture is fertilized with an average N fertilization rate of 120 kg N ha^{-1} yr^{-1}. A more complete description of the site and its management will be provided at chapter 3.

Table 1-1 : food rations of the cows during the winter season. The data were provided by the farmer and correspond to the average food ration of the cows during winter 2017-2018.

Food	Proportion of dry matter
Wheat straw	15,3%
protiwanze MF	8,0%
Maize silage	36,3%
Beet pulps	4,3%
Grass silage	12,8%
Winter cover forage	22,3%

10. Objectives of the thesis

In the scope of this project, this work aims at fulfilling the following objectives:

Objective 1: Establishing a robust methodology to build the complete carbon budget of a grazed pasture and to assess its associated uncertainties.

Objective 2: Building a complete carbon budget of this reference pasture to determine if the pasture act as a carbon source or sink and to quantify it.

Objective 3: Testing the impact of grazing strategies on the CO_2 fluxes dynamics and on the C sink or source intensity of the pasture.

To fulfill these objectives, this study was divided into different chapters:

Chapter 2: *The eddy covariance method.*

In this chapter, a general overview of the eddy covariance methodology extensively used in this work is given.

Chapter 3: *Establishment of Carbon the carbon balance of a grazed pasture*

Article 1 : Gourlez de la Motte, L., Jérôme, E., Mamadou, O., Beckers, Y., Bodson, B., Heinesch, B., Aubinet, M., 2016. *Carbon balance of an intensively grazed permanent grassland in southern Belgium.* Agricultural and Forest Meteorology 228–229, 370–383.

In this chapter, the carbon balance of the studied pasture is established by combining eddy covariance CO_2 fluxes measurements with other organic C fluxes during five complete years. The methodology used to obtain all the different terms of the budget is fully described. The paper aims at answering the following questions: **is an intensively managed grassland grazed by Belgian Blue cattle with a high stocking rate a C sink or source**? What is its strength? How do grazing interact with the C budget? What are the main sources of uncertainties when computing the C budget?

Chapter 4: *Eddy covariance high frequency loss correction*

Article 2 : Mamadou, O., Gourlez de la Motte, L., De Ligne, A., Heinesch, B., Aubinet, M., 2016. *Sensitivity of the annual net ecosystem exchange to the* *cospectral model used for high frequency loss corrections at a grazed grassland site.* Agricultural and Forest Meteorology 228–229, 360–369.

In this chapter, we highlighted a key methodological issue associated with the use of the eddy covariance technique for measuring CO_2 fluxes that was discovered when writing chapter 3. This article focuses on the impact of the reference cospectrum used to correct high frequency losses on annual NEE estimates which are critical for the establishment of the C budget.

Chapter 5: *Impact of grazing timing on grassland net ecosystem exchange.*

Article 3 : Gourlez de la Motte, L., Mamadou, O., Beckers, Y., Bodson, B., Heinesch, B., Aubinet, M., 2018. *Rotational and continuous grazing does not affect the total net ecosystem exchange of a pasture grazed by cattle but modifies CO2 exchange dynamics.* Agriculture, Ecosystems & Environment 253, 157–165.

This chapter focuses on different impacts of grazing animals on the net ecosystem exchange of the pasture and, especially on the impact of grazing strategies by comparing the CO_2 fluxes measured in a continuously grazed pasture to the CO_2 fluxes measured in a rotationally grazed one.

Chapter 6: *Impact of cattle respiration on annual NEE*

Article 4: Gourlez de la Motte, L., Dumortier, P., Beckers, Y., Bodson, B., Heinesch, B., Aubinet, M., 2019. Herd position habits can bias net CO2 ecosystem exchange estimates in free range grazed pastures. Agricultural and Forest Meteorology 268, 156–168.

This chapter focuses on how cattle location habits can impact annual NEE estimates through their respiration. In this article, multiple strategies are proposed in order to compute unbiased NEE that include cattle respiration in a representative way.

Chapter 7: *General discussion, conclusion and perspectives.*

11. Personal contribution to the research presented in this manuscript

This manuscript is the result of a team work that started in 2010 when measurements started at the Dorinne Terrestrial Observatory. My work was primarily focused on analyzing, computing and communicating the different results of this work. The technical maintenance of the site was successively done by Henry Chopin, Frederic Wilmus, Gino Mancini, Alain Debacq and Alwin Naiken. Biomass measurements were successively done by Jean Christophe Pector and Melissa Lhonneux. The data acquisition and quality control was successively done by Elisabeth Jérôme, Ossenatou Mamadou, Pauline Buysse and myself. All the presented research has been written in collaboration with the co-authors of each chapter.

The writing process of chapter 3 (Article 1) was started by Elisabeth Jérôme who wrote a first version based on the data collected from 2010-2013 during her Ph.D thesis (Jérôme, 2014). After the discovery of an important methodological issue fully described in the next chapter, we decided to rewrite the paper as some conclusions in the former version were wrong. The new version (based on the former one) was written by myself. It was also decided to add two supplementary years of data (2014-2015) to strengthen the paper.

Chapter 4 (Article 2) has been published by Ossenatou Mamadou who wrote the paper. I participated in the data analysis in order to assess the impact of the used reference cospectrum on the computation of the C budget. I also participated in the redaction process by rereading and commenting the intermediate versions.

In chapter 5 (article 3), I was responsible for analyzing the data and writing the paper.

For chapter 6 (Article 4), Pierre Dumortier was responsible for the GPS data acquisition and analyses. He did all the necessary computation linked to the GPS campaigns and participated in the redaction process. For my part, I analyzed the data and wrote the paper.

Chapter 2

The eddy covariance method

1. General principle and main interest

The eddy covariance technique is a micrometeorological method that allows vertical gas exchanges between a surface (typically an ecosystem) and the atmosphere be captured at a high resolution time rate. This method, central to the project, was extensively used to continuously measure the vertical CO_2 (and other gases) flux between the studied grassland and the atmosphere. A complete updated description of the eddy covariance method has been given by Aubinet et al., (2012b). Here is an overview of the method.

Over an ecosystem, when turbulence is sufficiently developed, gas transport is mainly due to turbulence, gas diffusion being considered negligible when compared to the turbulent transport. The eddy covariance theory is based on the mass conservation equation of a scalar:

$$\frac{\partial(\frac{1}{V_m}\chi_s)}{\partial t}+u\frac{\partial(\frac{1}{V_m}\chi_s)}{\partial x}+v\frac{\partial(\frac{1}{V_m}\chi_s)}{\partial y}+w\frac{\partial(\frac{1}{V_m}\chi_s)}{\partial z}=S_s \qquad (2.1)$$

where V_m is the dry air molar volume χ_s the molar dry mixing ratio (mole mole^{-1}) of a given scalar (*here* CO_2), which correspond to the ratio of the scalar number of moles to those of dry air, *u*, *v* and *w* are the three components of wind speed (m s^{-1}) according to the *x*, *y* and *z* axis and S_s is the absorption or production rate of the scalar (mole m^{-3} s^{-1}).

Each variable can be decomposed (assuming stationary conditions) between a temporal mean (marked by an overbar) and a fluctuation (marked by a prime) $w=\overline{w}+w'$, by the so-called Reynolds decomposition. If we apply this decomposition to *u*, *v*, *w* and χ_s to equation 2.1, average for a given period and integrate on a control volume of area 2*L*x2*L* and height h_m:

$$\frac{1}{4L^2}\int_{-L}^{L}\int_{-L}^{L}\int_{0}^{h_m}\left[\underbrace{\overline{\frac{1}{V_m}}\,\overline{\frac{\partial\chi_s}{\partial t}}}_{I}+\underbrace{\bar{u}\overline{\frac{1}{V_m}}\,\overline{\frac{\partial\chi_s}{\partial x}}+\bar{v}\overline{\frac{1}{V_m}}\,\overline{\frac{\partial\chi_s}{\partial y}}+\bar{w}\overline{\frac{1}{V_m}}\,\overline{\frac{\partial\chi_s}{\partial z}}}_{II}+\underbrace{\frac{\partial\overline{\frac{1}{V_m}}\,\overline{u'\chi'_s}}{\partial x}+\frac{\partial\overline{\frac{1}{V_m}}\,\overline{v'\chi'_s}}{\partial y}}_{III}+\underbrace{\frac{\partial\overline{\frac{1}{V_m}}\,\overline{w'\chi'_s}}{\partial z}}_{IV}\right]\mathrm{dzdxdy}$$

$$=\frac{1}{4L^2}\int_{-L}^{L}\int_{-L}^{L}\int_{0}^{h_m}\underbrace{\overline{S_s}\,\mathrm{dzdxdy}}_{V} \qquad (2.2)$$

where term *I* correspond to scalar storage in the volume control, term *II* corresponds to the advection (horizontal and vertical), term *III* and *IV* are the transport by turbulence and term *V* the source/sink strength. If we further make the following assumptions:

- the surface is flat and homogeneous (horizontal gradients nullify)
- constant dry air molar volume
- conditions are stationary

- the mixing ratios and turbulent fluxes are representative of the whole volume

then horizontal integration is unnecessary and equation 2.2 can be simplified as :

$$\underbrace{\int_0^{h_m} \overline{\frac{1}{V_m}}\,\overline{\frac{\partial \chi_s}{\partial t}}\,dz}_{I} + \underbrace{\int_0^{h_m} \overline{\frac{1}{V_m}}\,\overline{w}\,\overline{\frac{\partial \chi_s}{\partial z}}\,dz}_{IIa} + \underbrace{\int_0^{h_m} \overline{\frac{1}{V_m}}\,\overline{u}\,\overline{\frac{\partial \chi_s}{\partial x}} + \int_0^{h_m} \overline{\frac{1}{V_m}}\,\overline{w}\,\overline{\frac{\partial \chi_s}{\partial z}}}_{IIb} + \underbrace{\overline{\frac{1}{V_m}}\,\overline{w'\chi'_s}\Big|_{h_m}}_{III} = \underset{IV}{F_s} \tag{2.3}$$

where Fs is the net ecosystem exchange for the component s,term *I* corresponds to the change of storage of the scalar between the soil and the measurement height (often called the storage term), term *IIa* represents the vertical advection term resulting from dry air density temporal variations in the air below h_m, term *IIb* represents the horizontal advection and term *III* is the turbulent transport term and term *IV* is the source/sink of the scalar in the control volume.

Most of the time, term *IIa* and *IIb* can be considered negligible. As a result, the source/sink F_S can be described as the sum of the covariance of vertical wind speed and dry mixing ratio of the studied gas at height h_m and the change of storage of this scalar below h_m:

$$\underbrace{\int_0^{h_m} \overline{\frac{1}{V_m}}\,\overline{\frac{\partial \chi_s}{\partial t}}\,dz}_{I} + \underbrace{\overline{\frac{1}{V_m}}\,\overline{w'\chi'_s}\Big|_{h_m}}_{III} = \underset{IV}{F_s} \tag{2.4}$$

Usually, the covariance is calculated on a 30-min time basis from high frequency (usually 10-20 Hz) wind and mixing ratio measurements. For low height masts, similar to those used over grasslands and croplands, the storage term is often computed from a single concentration measured at h_m. If the measured scalar is CO_2, this scalar flux (F_{CO2}) corresponds to the net CO_2 exchange (net CO_2 ecosystem exchange, NEE).

If the above assumptions are most often well respected during the day, the simplifications made in equation 2.4 cannot be applied at night, during low turbulent conditions. In these conditions, the horizontal (equation 2.2) and vertical advection terms are not negligible. Because the advection terms cannot be measured with a single tower, a specific filtering of night data is needed as described in section 3 of this chapter.

Measuring gas fluxes on 30-min time resolution allows a faithful capture of the temporal variability of the studied flux. The missing data can also be filled in order to get seasonal to annual complete time series that can be summed to obtain gas fluxes over longer periods and compute annual/seasonal gas budgets (Falge et al., 2001; Moffat et al., 2007). Finally, these high time resolution measurements allow studying the impact of specific periods or events on the annual flux such as droughts (e.g. Ciais et al., 2005; Hussain et al., 2011; Jongen et al., 2011), biomass harvests (Jones et al., 2017; Wohlfahrt et al., 2008) and others. This method has been widely used to measure CO_2 fluxes in all kind of ecosystems (e.g. Baldocchi et al., 2017) but also other gases such as CH_4 (e.g. Nicolini et al., 2013), N_2O (e.g.

Jones et al., 2011; Kroon et al., 2010; Nicolini et al., 2013; Lognoul et al., in press) and volatile organic compounds (e.g. Laffineur et al., 2013; Bachy et al., 2016).

2. Eddy covariance and supporting instrumentation

Measuring GHG fluxes exchanged between an ecosystem and the atmosphere by eddy covariance requires the use of a fast three dimensional anemometer and a fast gas analyzer (CO_2, CH_4, N_2O…). The anemometer is positioned on a mast above the canopy level (Figure 2.1). The air is pumped at the height of the anemometer in an analyzer and the dry mixing ratio of the gas is measured. Both wind speed and gas mixing ratio are measured at high frequency (generally 10 or 20 Hz).

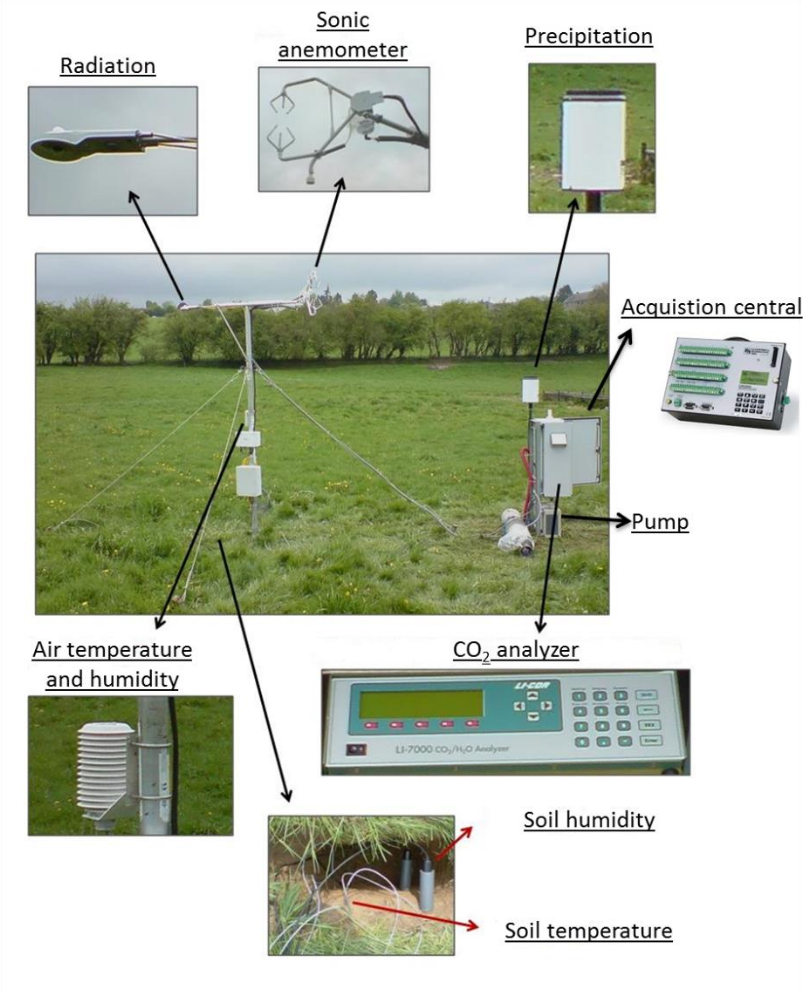

Figure 2-1 : General overview of the eddy covariance flux tower and other meteorological sensors installed at the Dorinne Terrestrial Observatory

This eddy covariance system is associated with a micro meteorological station measuring several variables such as air and soil temperatures, air and soil humidity, precipitations, atmospheric pressure and radiation. All the data are registered by a data data logger. These are frequently collected (around once a week) on site using memory cards. A more complete description of the instrument set up used in this study is given in chapter 3.

3. Flux computation, corrections and quality control

3.1. Data acquisition, quality control and corrections

Measuring GHG exchanges between an ecosystem and the atmosphere requires a relatively complex and specific treatment (Aubinet et al., 2012b, 2000) described at figure 2.2.

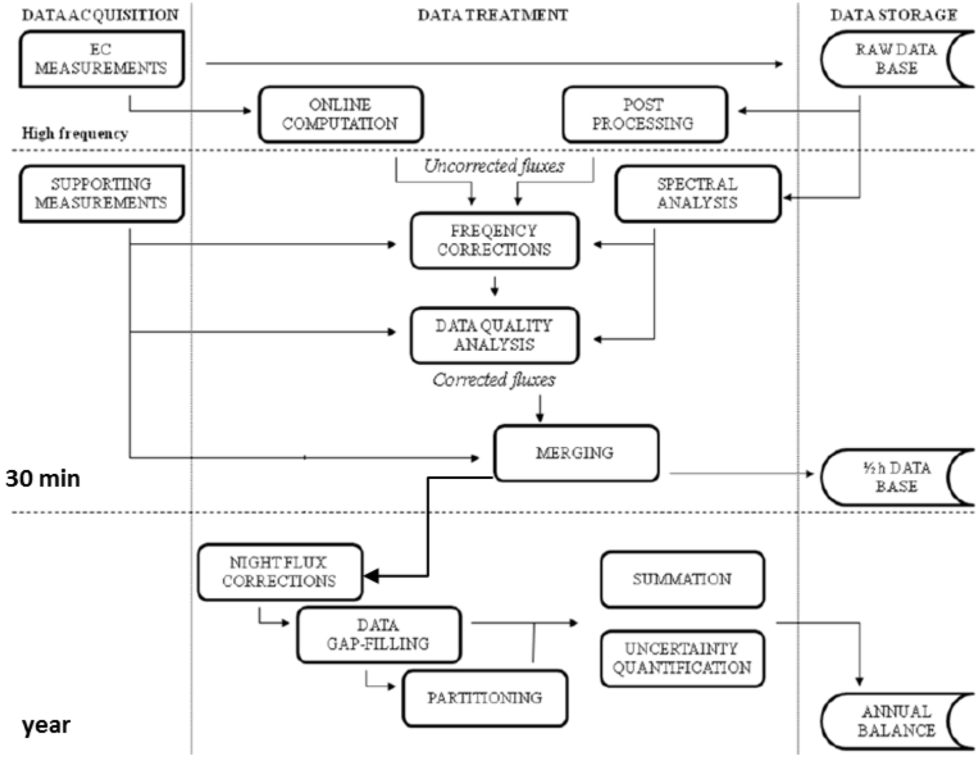

Figure 2-2 : Schematic representation of the eddy covariance data acquisition procedure. Created by Elisabeth Jérôme, adapted from Aubinet et al., (2000).

First, high frequency data must be collected and the covariance must be computed from raw data (Rebmann et al., 2012). All the computations have been done using the EDDYSOFT software package (EDDY Software, Jena, Germany, Kolle and Rebmann, 2007). To do so, the time lag between the acquisition of wind speed and gas concentration must be computed. This delay is mostly due to differences in electronic signal treatment, separation between sensors and air travel through tubes in closed-path analyzers. Coordinate rotation must be applied in order to align the z axis perpendicular to the streamlines.

Once computed, these “uncorrected” fluxes must be corrected for high frequency losses and the so-called WPL density fluctuation correction (Webb et al., 1980). The correction procedures are fully described in Foken et al., (2012b). High frequency losses are mainly due to inadequate sensor frequency response, sensor separation and air transport through the tube in closed path analyzers. Because this high frequency correction can significantly affect the results, a specific methodological work was carried for this correction. The procedure used to do this frequency correction will be fully described at chapter 4.

Once fluxes are computed and corrected, flux data quality must be controlled in order to keep only reliable data. The procedure is also completely described in Foken et al., (2012b). Briefly, out of plausible range data must be removed using despiking algorithms (Vickers and Mahrt, 1997). Then, unsteady state data must be filtered as they do not meet the stationarity assumption needed to obtain relevant EC fluxes and the the turbulent flux is no longer representative of the scalar source/sink (Foken and Wichura, 1996).

It is well known that NEE computed following equation 2.3 under estimate the actual CO_2 source/sink during low turbulence conditions (Aubinet et al., 2012a). During these conditions, there is strong evidence that eddy covariance measurements are affected by systematic errors mostly due to stable atmospheric conditions, extended footprint (cfr. Section 3.3), and the advection terms being non negligible in equation 2.2 as well as nonstationary conditions (Aubinet, 2008). Data under low turbulence must therefore be discarded. In this case this was done by filtering data under low friction velocity, u*, a measure of the intensity of turbulence. After the application of all these filters, around 20-60% of the data are rejected (Papale et al., 2006).

3.2. CO_2 flux data gap filling

After these operations, gaps in flux data time series must be filled in order to compute annual sums (Falge et al., 2001; Moffat et al., 2007). In this work, CO_2 fluxes data were filled using the time-moving look up table algorithm developed by Reichstein et al., (2005). The algorithm fills the data with fluxes averaged in similar environmental conditions. Meteorological variables used by the algorithm are the air temperature (T_{air}), the vapor pressure deficit (VPD) and the global radiation (R_g). In this work, The gap filling was made using the REddyProc online tool (Reichstein et al., 2005).

3.3. *Footprint concept*

Ideally, eddy covariance measurements must be made over perfectly homogeneous terrain to avoid horizontal advection with a terrain that outreach the source of area of the measurement (footprint, Rannik et al.,(2012)). Roughly, the footprint (FP) can be defined as the area seen by the flux tower for a given period and reflects the contribution of each pixel (sub area) in the area to the measured flux. In case of inhomogeneous terrains, the measured flux depends on the strength of each sources/sinks as well as on their respective contribution to the footprint (Rannik et al., 2012). The relationship between the source/sink distribution and the vertical flux over an area measured at the origin of a coordinate system with no contribution from downwind can be computed as (Leclerc and Thurtell, 1990):

$$F(0,0) = \int_{-\infty}^{\infty}\int_{0}^{\infty} \phi(x,y,z)\, S(x,y,0)\, dx dy \qquad (2.3)$$

where F is the measured flux measured at height z, S the surface source/sink and ϕ the footprint weighting function (m^{-2}). Depending on the location of a source/sink from the eddy flux tower and micrometeorological conditions, the contribution of a given sink/source will vary as illustrated at figure 1-3.

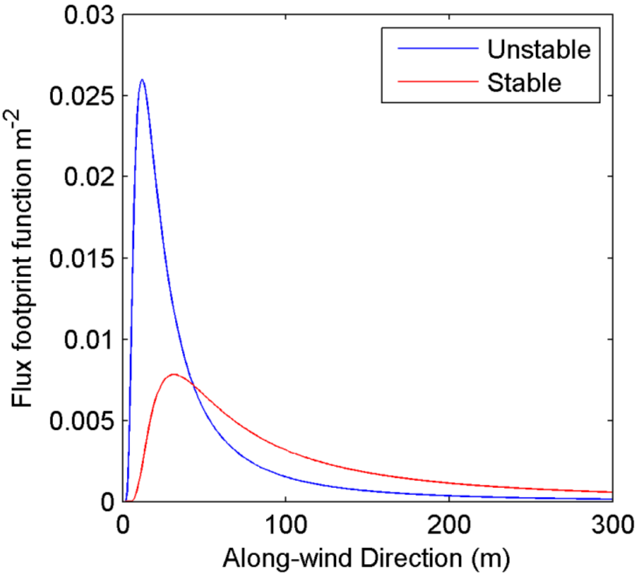

Figure 2-3: Illustration of crosswind integrated flux footprint function in stable and unstable conditions for an observation height of 2.5m using the Korman and Meixner (2001) model.

Over perfectly homogeneous terrain, evaluating the footprint function is not needed as sources/sinks are spread evenly on the field. In practice, this condition is rarely met so that footprint evaluation is often necessary for flux interpretation. In sites such as grazed pastures, cows act as moving sources of CO_2 (through their respiration) and CH_4 (enteric fermentation) that are usually not spread evenly in the field. Therefore, to evaluate their contribution to the measured CO_2 flux or simply compute the flux emitted by each animal, information about their location and strength of their contribution to the footprint is needed (Felber et al., 2015, 2016b). In addition, in our study, the footprint area was sometimes larger than the studied pasture so that a correction factor based on the footprint model had to be implemented for fluxes coming from the studied parcel only (see chapter 6). Different models allow computing this footprint function in order to weight the contribution of the different sink/sources. The FP function can be derived either from analytical (Kormann and Meixner, 2001), Lagrangian dispersion stochastic approaches (Kljun et al., 2015, 2002) or large eddy simulations (Leclerc et al., 1997). In this work, we used the analytical model developed by Korman and Meixner (2001) as this model seemed to be the best choice to reproduce the emission by an artificial source (Dumortier et al., 2019).

3.4. Partitioning of NEE

As previously introduced (figure 1-4), the NEE is the sum of the gross primary productivity (GPP) and the total ecosystem respiration (TER):

$$\mathrm{NEE}=\mathrm{TER}-\mathrm{GPP} \tag{2.4}$$

In grazed sites, the total ecosystem respiration is the sum of the respiration of the soil and vegetation and the respiration from grazing animals. Obtaining GPP and TER is very important to better understand the process that affects NEE. To do this partitioning, we used the fact that, at night, GPP is zero so that NEE is equal to TER. The TER at night was modelled as an exponential response to temperature and this relationship was used in daytime to obtain TER and further on GPP. In this work, we used the procedure developed Reichstein et al., (2005) and available to the research community as the REddyProc online tool.

In a grazed pasture, the measured NEE (also referred as NEE_{tot}) is the sum of the NEE of the soil and vegetation (NEE_{past}) and the respiration of the cows (R_{cows}) (Felber et al., 2016b):

$$\mathrm{NEE}_{tot}=\mathrm{NEE}_{past}+R_{cows} \tag{2.5}$$

If we hypothesize that, on a yearly basis, cattle are spread homogeneously on the field (homogeneous cow distribution hypothesis), we can assume that R_{cows} is included in a representative way in NEE_{tot}. Annual NEE_{tot} measurements are in this case considered representative of the whole ecosystem. This hypothesis was explicitly used from chapter 3 to 5 to characterize the net CO_2 exchange of the ecosystem on a yearly basis. Later on, this important hypothesis was tested when sufficient data about cow location (namely GPS data and CH_4 fluxes) in the footprint were available. The method and the results of this investigation are fully detailed in chapter 6. Eventual consequences are also fully discussed in the final discussion and conclusions section.

Chapter 3

Carbon balance of an intensively grazed permanent grassland in southern Belgium

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Abstract

Grasslands are an important component of the global carbon balance, but their carbon storage potential is still highly uncertain. In particular, the impact of weather variability and management practices on grassland carbon budgets need to be assessed. This study investigated the carbon balance of an intensively managed permanent grassland and its uncertainties by drawing together 5 years of eddy covariance measurements and other organic carbon exchanges estimates. The results showed that, despite the high stocking rate and the old age of the pasture, the site acted as a relatively stable carbon sink from year to year, with a 5-year average net biome productivity of -161 [-134 -180] g C m^{-2} yr^{-1}. Lateral organic carbon fluxes were found to increase the carbon sink because of high carbon imports (organic fertilization, feed complements) and low carbon exports in form of meat compared to dairy pastures. The cattle stocking density was adapted to grass production, which itself depends on weather conditions and photosynthesizing area, in order to maintain a steady meat production. This resulted in a coupling between grazing management and weather conditions. As a consequence, both weather and grazing impacts on net ecosystem exchange were difficult to distinguish. Indeed, no correlation was found between weather variables anomalies and net ecosystem exchange anomalies. This coupling could also partly explain the low C budget inter-annual variability. The findings in this study are in agreement with those reported by other studies that have shown that well-managed grasslands could act as a carbon sinks.

1. Introduction

Grasslands cover 40% of the Earth's ice-free land surface (Steinfeld et al., 2006) and are characterized by soils with a high soil carbon (C) content (Conant et al., 2001). They therefore constitute an important component of the global C balance (IPCC, 2007). Studies assessing the C balance under grasslands are relevant because grassland C sequestration can play an important role in mitigating the total greenhouse gas emissions from livestock production systems (Lal, 2004; Soussana et al., 2010). There is a strong need, therefore, to accurately evaluate grassland C sequestration (Herrero et al., 2011).

Grassland C sequestration can be determined directly by measuring changes in soil organic carbon (SOC) stocks or indirectly by measuring the balance of C fluxes at the system boundaries. Contrary to studies based on SOC change measurements (Goidts and van Wesemael, 2007; Lettens et al., 2005a, 2005b; Meersmans et al., 2009), studies assessing the total C grassland budget by combining eddy covariance measurements with measurements of other C fluxes enable investigations to be made of seasonal, annual and inter-annual C flux dynamics and budgets (Byrne et al., 2007; Gilmanov et al., 2010; Klumpp et al., 2011; Mudge et al., 2011; Peichl et al., 2012, 2011; Soussana et al., 2010; Zeeman et al., 2010). They also enable the impact of specific management practices or weather conditions to be analyzed (Aires et al., 2008; Allard et al., 2007; Ammann et al., 2007; P. Ciais et al., 2010; Harper et al., 2005; Heimann and Reichstein, 2008; Hussain et al., 2011; Jaksic et al., 2006; Jongen et al., 2011; Klumpp et al., 2011; Peichl et al., 2012; Suyker et al., 2003; Teuling et al., 2010).

The results of these studies reveal strong site-to-site variability because of differences in pedoclimatic conditions and management practices: they report increases as losses or no change in soil C balances (Soussana et al., 2010). Grassland C balance and the impact of environmental conditions and management practices on this balance are still not well understood (Mudge et al., 2011; Soussana et al., 2010). Grazing is known to directly affect the carbon dioxide (CO_2) net ecosystem exchange (NEE) via livestock respiration and indirectly via biomass consumption, natural fertilization trough excreta and soil compaction (Jérôme et al., 2014). A high stocking rate could impact the carbon budget by either reducing growth primary productivity (GPP) through defoliation (Jérôme et al., 2014) but also by stimulating GPP by removing less productive plant material before withering. The land use and the management prior to the study could also affect the carbon budget. Indeed, interventions such as ploughing, reseeding, land use change from a crop field to a grassland and improved management could still increase the CO_2 accumulation many years later before reaching an eventual equilibrium (Smith, 2014).

The main objective of this research was to assess the total C balance of a grazed grassland located in Wallonia (southern Belgium) by measuring all C fluxes exchanged at the system boundaries, using the eddy covariance method, direct measurements made in the field, estimates by the farmer and literature data when no measurements were available. The study site has been a permanent grassland since it was used for grazing (probably more than a century). It has been intensively managed with high stocking rates (around 2 Livestock units (LU) per hectare per year) and the application of mineral and organic fertilization for more than 40 years.

This paper also attempts to answer a few specific questions: (i) is a grassland established for more than a century and intensively managed for more than 40 years with a stocking rate exceeding 2 LU per hectare a C sink or a source? (ii) How do management practices and weather conditions affect the C budget? (iii)What are the main sources of uncertainties and how robust is the methodology used to establish the C budget? The research covered 5 years of measurements, providing an opportunity to assess the grassland C budget on monthly and annual scales, evaluate its uncertainties and identify some drivers linked with weather or grassland management

2. Material and method:

2.1. Carbon balance of the pasture

The net balance of C fluxes exchanged at the system boundaries, commonly known as net biome productivity (NBP, g C m^{-2} yr^{-1}), was defined by Soussana et al. (2010) for temperate grazed grassland as (Figure 2-1):

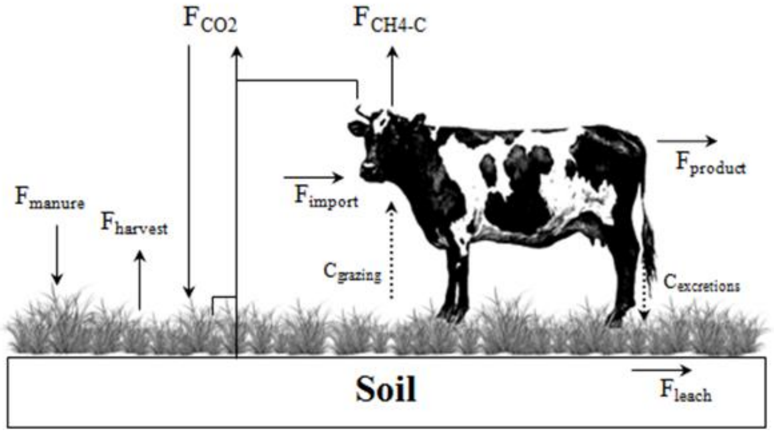

Figure 3-1 : Carbon (C) cycle of the grazing animal. Solid arrows represent C components of the net biome productivity (see Equation 1). Dashed arrows represent internal C fluxes.

$$NBP = F_{CO2} + F_{CH4} + F_{manure} + F_{import} + F_{harvest} + F_{product} + F_{leach} \tag{3.1}$$

where F_{CO2} is the net ecosystem carbon dioxide (CO_2) exchange, corresponding to the difference between gross CO_2 uptake via photosynthesis (gross primary productivity, GPP) and CO_2 loss via respiration (total ecosystem respiration, TER, including cattle respiration); F_{CH4} is the C lost through methane (CH_4) emissions by grazing cattle (the CH_4 fluxes from the soil were considered as negligible as their magnitude was only 2.5% of the cattle fluxes according to Dumortier et al., (2017)); F_{manure} and F_{import} are the lateral organic C fluxes imported into the system through manure and/or slurry application and supplementary feed, respectively; $F_{harvest}$ and $F_{product}$ are the lateral organic C fluxes exported from the system through mowing and animal products (meat), respectively and F_{leach} represents organic and/or inorganic C losses through leaching. Throughout this paper, we adopt the micrometeorological convention that fluxes from the ecosystem are positive and that fluxes to the ecosystem are negative. A negative NBP therefore corresponds to C uptake.

2.2. Site description

The research was carried out at the Dorinne terrestrial observatory (DTO) (50° 18’ 44’’ N; 4° 58’ 07’’ E). Dorinne is 18 km south/south-east of Namur, in the Condroz region in Belgium. The Condroz region is characterized by a succession of depressions and crests with soils suitable for arable land use (mainly cereals and sugar beet) and pastures for cattle breeding (Goidts and van Wesemael, 2007). The climate is temperate oceanic. The mean annual air temperature is 10°C, the annual precipitation is 847 mm and the main wind directions are south-west (IRM, 2011) and north-east. The field is bordered on the south-west by a cultivated field and by pastures on the north-east. The research site is a permanent grassland covering 4.22 ha and dominated by a large colluvial depression exposed south-west/north-east. This depression is situated on a loamy plateau with a calcareous and/or clay substrate. The altitude varies from 240 m (north-east) to 272 m (south). So far as we know, the field has never been cultivated and has been permanent grassland since it started being used for grazing (probably for more than a century). It has been intensively used for cattle grazing, with the application of organic (cattle slurry and manure) and inorganic fertilizers, for about 40 years. The grassland species composition is: 66% grasses, 16% legumes and 18% other species. The dominant species are perennial ryegrass (*Lolium perenne* L.) and white clover (*Trifolium repens* L.). There has been no renovation of the grass vegetation (ploughing – resowing) for more than 50 years. Flux measurements have been taken since spring 2010. The data given in this study cover 5 full years of measurements from 12 May 2010, when the eddy covariance measurements began, to 12 May 2015.

2.3. Grassland management

The field was intensively managed and grazed during the growing season by Belgian Blue cattle (heifers, suckler cows, breeding bulls, calves). The rotation between stocking (periods with cattle) and recovery periods without cattle (rest periods) depended on herbage growth and its consumption by cattle. In this context, weather conditions limited the grazing pressure, which was adjusted when necessary. Feed (corn silage, hay and a mixture of straw and ProtiWanze®, a by-product of bio-ethanol production) was distributed when necessary to supplement grass shortage (drought or beginning/end of the grazing season). Fertilizers, including mineral and organic fertilizers, were applied at various times to the field throughout the growing season (Table 2-1). The reference unit used for calculating LU is the grazing equivalent of one 600 kg liveweight (LW) adult dairy cow producing 3,000 kg of milk annually, without additional concentrated feed (Eurostat, 2013). Breeding bulls and suckler cows correspond to 1 LU, and heifers and calves to 0.6 and 0.4 LU, respectively.

Table 3-1: List of management activities at the Dorinne Terrestrial Observatory. Weighing values are presented with a 95% confidence interval.

Before the start of the experiment		
10-Mar-10	fertilization: compost (t FM ha^{-1})	11.0
25-Mar-10	fertilization: 10/8/4 + selenstar® (Se) (t ha^{-1})	0.6
2010		
3-Jun - 6-Jun-10	cut-harvest (t DM ha^{-1})	2.7
10-Jun-10	fertilization: 24/0/0 + selenstar® (Se) (t ha^{-1})	0.2
20-Jun - 11-Jul-10	supplements: corn silage/mixture (t FM ha^{-1})	0.9
Jul-10	scattering of livestock droppings	
31-Jul - 21-Aug-10	supplements: mixture (t FM ha^{-1})	1.1
5-Aug-10	heifers weighing (kg animal^{-1})	436 ± 13
7-Sep - 22-Nov-10	supplements: mixture (t FM ha^{-1})	3.5
Sep-10	scattering of livestock droppings	
Total fertlization for 2010 (kg N ha^{-1})		**164**
2011		
26-Jan-11	heifers weighing (kg animal^{-1})	549 ± 20
20-Feb-11	fertilization: compost (t FM ha^{-1})	12.0
9-Mar-11	fertilization: 18/5/5 + Mg (t ha^{-1})	0.4
22-Mar-11	liming: magnesian lime (t ha^{-1})	1.5
9-Apr - 23-Apr-11	supplements: mixture (t FM ha^{-1})	0.4
13-May-11	fertilization: 10/8/4 + selenstar® (Se) (t ha^{-1})	0.3
3-Nov - 2-Dec-11	supplements: hay (t FM ha^{-1})	0.3
Total fertlization for 2011 (kg N ha^{-1})		**162**
2012		
19-Mar-12	fertilization: 10/8/4 + selenstar® (Se) (t ha^{-1})	0.4
24-Mar - 2-Apr-12	supplements: mixture (t FM ha^{-1})	0.3
30-May-12	fertilization: n27 (t ha^{-1})	0.2
13-Jul-12	fertilization: n27 (t ha^{-1})	0.2
31-Oct-12 - 14-Nov-12	supplements: hay (t FM ha^{-1})	0.1
Total fertlization for 2012 (kg N ha^{-1})		**148**
2013		
3-Apr-13	fertilization: 10/8/4 (t ha^{-1})	0.4
3-Apr-13	scattering of livestock droppings	
13-Jul-13	fertilization: n27 (t ha^{-1})	0.2
10-Sep-13	scattering of livestock droppings	
Total fertlization for 2013 (kg N ha^{-1})		**94**
2015		
11-Mar-15	fertilization: 10/8/4 (t ha^{-1})	0.3
15-Mar-15	scattering of livestock droppings	
Total fertlization for 2015 (kg N ha^{-1})		**30**

2.4. CO_2 flux measurements

The CO_2 flux was measured using the eddy covariance technique. This involved using a three-dimensional sonic anemometer (CSAT3, Campbell Scientific Ltd, UK) coupled with a fast CO_2-H_2O non-dispersive infrared gas analyzer (IRGA) (LI-7000, LI-COR Inc., Lincoln, NE, USA) to measure fluxes of CO_2, latent heat, sensible heat and momentum. The system was installed on a mast at a height of 2.6 m above ground in the middle of the field and was surrounded by a secured enclosure. Air was sucked into the IRGA through a tube (6.4 m long; inner diameter 4 mm) by a pump (NO22 AN18, KNF Neuberger, D) with a 12 l min^{-1} flow. Data were sampled at a rate of 10 Hz. Zero and span calibrations were performed for CO_2 and H_2O about once a month. Pure nitrogen (Alphagaz 1, Air Liquide, Liege, Belgium) was used for the zero and 350 µmol mol^{-1} mixture (Chrystal mixture, Air Liquide, Liege, Belgium) for the span.

F_{CO2} was computed half-hourly as the sum of the turbulent flux measured by the eddy covariance system and of the storage term (Foken et al., 2012a). Flux computation was performed using the EDDYSOFT software package (EDDY Software, Jena, Germany, Kolle and Rebmann, 2007) and the 10 Hz time series data. All the computation and correction procedures used were the standard procedures defined within the context of the EUROFLUX – CARBOEUROFLUX – CarboEurope IP networks (Aubinet et al., 2012b, 2000). Double rotation was applied to wind velocity in order to align the streamwise velocity component with the direction of the mean velocity vector (Rebmann et al., 2012). Fluxes were corrected for high frequency losses following an original procedure based on the sensible heat cospectra. The complete procedure has been described by (Mamadou et al., 2016).

The turbulent fluxes were scrutinized using a stationary test with a selection criterion of 30% according to (Foken et al., 2012b; Foken and Wichura, 1996). Data were separated between night and day using a photosynthetic photon flux density (PPFD) criterion, with a threshold of 5 µmol m^{-2} s^{-1}. In order to avoid night CO_2 flux underestimation, CO_2 fluxes measured under low nighttime turbulence conditions were filtered (Aubinet et al., 2012a; Goulden et al., 1996). A critical threshold of u^* was determined at the point where the relationship between u* and the bin averaged temperature normalized nighttime F_{CO2} flattens. A value of 0.13 m s^{-1} was found and measurements with u^* below this value were systematically discarded.

Net ecosystem exchange (NEE) gaps were filled using the online REddyProc gapfilling and flux partitioning tool (Reichstein et al., 2005). The reference temperatures used to fill the gaps was the soil temperature at a depth of 2 cm. NEE partitioning into GPP and TER was also calculated using the same tool and same reference temperature.

Measurement footprint was calculated using an analytical model following Kormann and Meixner (2001). On average, during instable conditions, 77% of the footprint area was covered by the measured pasture. During stable conditions, this footprint area is much larger. However, most of the fluxes measured during stable conditions were discarded by the u* filtering (Dumortier et al., 2017).

In order to investigate inter-annual variability, flux (NEE, GPP, TER) and weather variable anomalies (temperatures, radiation, soil humidity…) were computed as follows: first, monthly and annual sums (for fluxes and precipitation) or averages (for other weather variables) were calculated. For each variable, a 5-year average was computed and anomalies for a given year were calculated as the difference between the variable (monthly/annual sum or average) for the considered year and its 5-year average.

2.5. Meteorology

Supporting measurements included air temperature and relative humidity (RHT2nl02, Delta-T Devices Ltd, Cambridge, UK), soil temperature (Pt 1000) at depths of 2, 5, 10, 25 and 50 cm and soil moisture (ThetaProbe, Delta-T Devices Ltd, Cambridge, UK) at depths of 5, 25 and 50 cm, gross and net radiation (CNR4, Kipp & Zonen, Delft, The Netherlands), rainfall (tipping bucket rain gauge, 52203, R.M. Young Company, Michigan, USA) and atmospheric pressure (144S BARO, SensorTechnics, Puchheim, Germany). Meteorological data were sampled at a rate of 0.1 Hz and averaged (summed for precipitation) every 30 min. Data were recorded on a data logger (CR3000, Campbell Scientific Ltd, UK). Raw eddy covariance data, sampled at 10 Hz, and half-hourly meteorological data were then stored on a 2 GB compact flash card. Growing degree days (GDD) were computed in order to evaluate the impact of winter temperatures on NEE. GDD was calculated as the sum of daily mean air temperatures above 0°C from 1 January (Theau and Zerourou, 2008) to 31 March.

2.6. Biomass measurements

2.6.1. Herbage mass

Herbage mass in the field (HM) was deduced from herbage height (h) measurements with a rising plate meter. The mean canopy height was determined manually by measuring the center height of a light-weight plate of 0.25 m² dropped onto the canopy at 60 points in the field. This estimation was then converted into HM using allometric relationships fitted on to direct sampling measurements. Samples were taken from the field (nine sample surveys, providing about 20-25 samples per survey) and from three secured enclosures (weekly measurements, see Section 2.6.2) during the stocking periods between 12 May 2010 and 11 May 2012. The samples were mowed at a height of 0.05 m using battery-powered hand clippers and a quadrat (0.5 × 0.5 m). They were then dried at 60°C in a forced-air oven until constant weight was achieved. A relationship between grass height difference before and after the cut and harvested dry matter was established:

$$\mathrm{HM} = -2.4 \times \mathrm{h}^2 + 203.7 \times \mathrm{h} \; (\mathrm{R}^2 = 0.77; \mathrm{n} = 381) \qquad (3.2)$$

where n is the number of samples.

2.6.2. Grass growth under grazing

Three secured enclosures from which animals were excluded were installed in the field to assess grass growth under grazing over a period (R_i). Each enclosure consisted of five strips (0.5 × 2 m). By successively cutting the strips, grazing was simulated and the HM accumulation under grazing was deduced from the canopy height measurements. Measurements were conducted over 5 weeks during the stocking cycle. On week 1, strip 1 was mowed and each week thereafter strip 1 and, successively, strips 2 to 5 were mowed. A weekly HM accumulation was obtained from the difference between average initial and final grass height of each strip and equation 2 for each secured enclosure. R_i was calculated as the average HM accumulation for the three secured enclosures over a given period.

2.7. Organic carbon exports and imports influencing net biome productivity

F_{CH4} was estimated as a constant fraction of the ingested dry matter (dry matter intake, DMI) by cattle during grazing using the dimensionless methane conversion factor Y_m, which is the methane emitted per kg of DMI. We assumed a typical Y_m value of 6% (Lassey, 2007). The DMI corresponded to the sum of the HM intake by cattle during grazing and the dry matter of supplementary feed imported. F_{manure} and F_{import} were calculated by multiplying the imported mass by its dry matter fraction and its dry matter C content (Table 2-2). $F_{harvest}$ was estimated by multiplying the HM difference in the field before and after the cut with the grass C content (Table 2-2). $F_{product}$ was estimated by multiplying the daily cattle LW gain for a growing animal, fixed at 0.647 kg LW animal^{-1} day^{-1} based on *in situ* measurements conducted in Year 1, with a concentration factor of 0.165 ± 0.002 kg C (kg LW)$^{-1}$ for Belgian Blue (Mathot et al., 2012). As it was not possible to measure F_{leach} at DTO, it was fixed at 7 ± 7 g C m^{-2} yr^{-1}, based on the work of Schulze et al. (2009).

Table 3-2 : Dry matter fraction (% DM) and dry matter C content (% C) used to calculate the net biome productivity (NBP) components linked to management practices.

NBP components	**Sample taken *in situ***	**% DM**	**Origin**	**% C**	**Origin**
F_{manure}	Compost	21	**Drying:** 60°C in a forced-air oven until constant weight was achieved	36	**Grinding:** Cyclotec - 1 mm screen **Laboratory:** Forest Ecology and Ecophysiology Unit, Institut National de la Recherche Agronomique - INRA) (UMR 1137 INRA-UHP). **Method:** Dumas, 1831. **Analyzer:** Elemental analyzer (NCS2500, CE instrument Thermo Quest, Italy).
F_{import}	Corn silage	44	Data provided by the farmer	40	
	Straw + ProtiWanze®	45		42	
	Hay	85		42	
$F_{harvest}$	Grass	-	Difference in grass height before and after harvest converted to herbage mass dry matter using equation 2	42	

C content analyses of samples taken *in situ* (herbage, complementary feed, compost) were conducted by the Forest Ecology and Ecophysiology Unit at the Institut National de la Recherche Agronomique (INRA) (UMR 1137 INRA-UHP) using the Dumas method (Dumas, 1831). After drying and grinding (Cyclotec – 1 mm screen), the samples were analyzed using an elemental analyzer (NCS2500, CE instrument Thermo Quest, Italy).

2.8. Other carbon fluxes

In order to analyze in detail all the C fluxes exchanged in this grassland and specifically those linked to grazing, we established the C cycle of the animals. It sought to estimate the components described in the sections below (Figure 2-1).

2.8.1. Cattle forage mass consumption and above-ground net primary productivity

For a period of interest (stocking or rest period), HM in the field was measured at the beginning ($HM_{i,beg}$) and end ($HM_{i,end}$) of the period, following the procedure described in Section 2.6.1. During grazing periods, the grass growth under grazing R_i was deduced from secured enclosure measurements, following the procedure described in Section 2.6.2.

From these measurements, we deduced the C intake through HM consumption by cattle during grazing ($C_{grazing,i}$) as (Macoon et al., 2003):

$$C_{grazing,i} = C_{content} \times \left(HM_{i,beg} - HM_{i,end} + R_i \right) \tag{3.3}$$

where $C_{content}$ is the grass C content obtained from laboratory measurements.

We also deduced the above-ground net primary productivity ($ANPP_i$). It was computed as:

$$ANPP_i = C_{grazing,i} + C_{content} \times \left(HM_{t+1} - HM_t \right) \tag{3.4}$$

where $\left(HM_{t+1} - HM_t \right)$, accounted only when positive, is the un-grazed biomass (biomass refusal because of excretions, trampling…) and $C_{grazing\ i}$ was zero during rest period. Annual $C_{grazing}$ and ANPP were obtained by summing $C_{grazing,i}$ and $ANPP_i$ for all periods of interest.

2.8.2. Livestock carbon dioxide losses at grazing

Livestock CO_2 emissions ($F_{CO2,livestock}$) were estimated from the C intake measurements. As most of the C ingested was digestible and therefore respired shortly after intake, we obtained:

$$F_{CO2,\ livestock} = \left(OMD \times C_{intake} \right) - F_{CH4} - F_{product} \tag{3.5}$$

where OMD (%) is organic matter digestibility and C_{intake} is the sum of $C_{grazing}$ and F_{import}.

In the same way, livestock C excreted ($C_{excretions}$) was estimated as:

$$C_{excretions} = NOMD \times C_{intake} \tag{3.6}$$

where NOMD (%) is non-organic matter digestibility.

OMD and NOMD values were obtained from the near infrared reflectance spectrometry analyses (NIRS system monochromator 5000-1100 to 2498 nm wavelength by 2 nm steps; Decruyenaere et al., (2009) of samples taken *in situ* (herbage, supplementary feed). After the samples were dried and ground (Cyclotec – 1 mm screen), analyses were conducted at the Walloon agricultural research center (CRA-W).

2.9. Uncertainty assessments

Eddy covariance fluxes are affected by uncertainties due to the presence of both random and systematic errors (Baldocchi, 2003; Hollinger and Richardson, 2005; Richardson et al., 2006). Systematic errors are due mainly to the underestimation of night fluxes measured during low turbulent conditions (Ammann et al., 2007; Rutledge et al., 2015) and to high frequency losses. In both cases, a correction procedure was applied, as described in Section 2.4. As these procedures are themselves not exact, however, residual uncertainties remain, mainly because of the choice of the correction parameters (u* threshold for night flux correction, cut-off frequency for high frequency correction).

In order to assess the overall uncertainty of our measurements, we considered four main sources of uncertainty: the random error affecting both measured fluxes and filled data (σ_r) resulting from the random character of turbulence and affecting not only measurements but also gap filled data; an additional systematic error resulting from the procedure used to fill the data (σ_{gf}; i.e., two errors associated with the gap filling) and remaining uncertainties after the application of the night flux (u* threshold chosen to filter the nighttime data [σ_{u*}]); and frequency corrections (cut-off frequency used for the spectral correction [σ_{f_0}]).

Estimation of the random uncertainty (σ_r)

The term σ_r combines the random error that affects both measured and filled data. This was calculated adapting a procedure described by Dragoni et al.(2007). The procedure follows three steps. First, the random error for the measured half-hourly flux (ε_m) was computed using the successive days approach developed by Hollinger and Richardson (2005). In this approach, ε_m is estimated as the absolute difference between two valid successive day fluxes at the same hour and during similar weather conditions (maximum PPFD range of 75μmolm^{-2}s^{-1}, maximum T_s range of 3°, maximum horizontal wind velocity range of 1ms^{-1}). The standard deviation of this error, $\sigma(\varepsilon_m)$, was then computed for flux classes (same number of observations) and a relationship between $\sigma(\varepsilon_m)$ and flux magnitude was established (Richardson et al., 2006).

This gave at DTO:

$$\sigma(\varepsilon_m) = -0.11 \times F_{CO2} + 1.47 \quad \text{for } F_{CO2} \le 0 \quad (R^2 = 0.90) \tag{3.7a}$$

$$\sigma(\varepsilon_m) = \quad 0.30 \times F_{CO2} + 0.08 \quad \text{for } F_{CO2} > 0 \quad (R^2 = 0.97) \tag{3.7b}$$

In the second step, a similar approach was used for the filled data. All valid half-hourly data were marked as artificial gaps and filled using the online REddyProc gapfilling tool (Reichstein et al., 2005). This gave a measured value (F_{CO_2}) and a modelled value (M) for each non-missing NEE value. The standard deviation of the residue ($\sigma(\varepsilon_{gf})$) was calculated as $\sigma(F_{CO2} - M)$ for pre-made flux classes with a same number of observations. A relationship between $\sigma(\varepsilon_{gf})$ and the flux magnitude was then established. This gave:

$$\sigma\left(\varepsilon_{gf}\right) = -0.075 \times F_{CO2} + 1.86 \quad \text{for } F_{CO2} \leq 0 \quad (R^2 = 0.87) \tag{3.8a}$$

$$\sigma\left(\varepsilon_{gf}\right) = \quad 0.15 \times F_{CO2} + 0.9 \quad \text{for } F_{CO2} > 0 \quad (R^2 = 0.71) \tag{3.8b}$$

Finally, in the third step, a Monte Carlo simulation was used to estimate the annual random uncertainty. A random error (ε_s) was generated for each half-hourly NEE value assuming a double exponential distribution (Hollinger and Richardson, 2005) with a zero mean, a standard deviation of $\sigma(\varepsilon_m)$ for measured values and $\sigma(\varepsilon_{gf})$ for filled values. Simulated NEE_s values were then calculated as $NEE_s = NEE + \varepsilon_s$ and the annual NEE calculated as the sum of NEE_s. This process was repeated 100 times and σ_r was calculated as the standard deviation of the 100 annual NEE_s values.

Estimation of the gap filling uncertainty (σ_{gf} *)*

As described above, the gap filling procedure led to a random error that is included in the σ_r term. Another non-random source of uncertainty linked to this procedure was identified, however. The preceding approach supposes that the mean residual gap filling residue ($\hat{\varepsilon}_{gf}$) is zero in each flux class. This was, however, not the case (Figure 2-2), as we found that it differed from zero for high absolute fluxes. This would mean that the gap filling procedure underestimates high fluxes both at night and during the day. In order to test the potential influence on annual sums, we conducted another Monte Carlo simulation, but this time used distributions with the corresponding $\hat{\varepsilon}_{gf}$ as means for filled data. σ_{gf} was then calculated as the difference between the average of the 100 annual NEEs and the actual NEE value for each year.

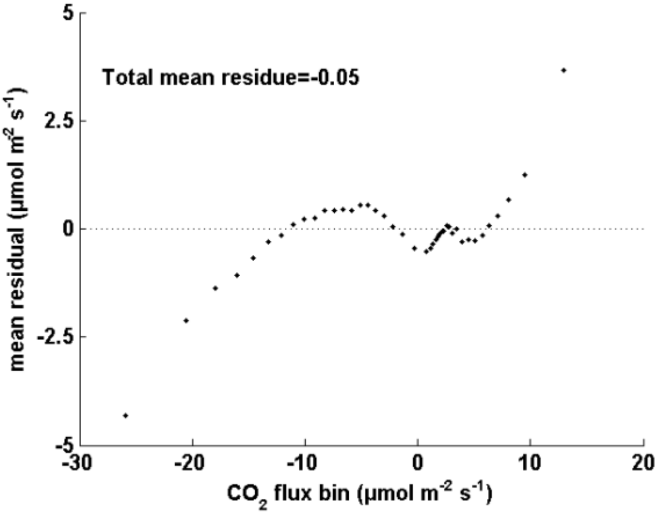

Figure 3-2: Relationship between the flux magnitude and the mean residuals for flux classes. Residual values are calculated as the difference between the measured flux and the flux calculated by the gap filling procedure. All values are given in μmolm^{-2}s^{-1}.

Estimation of the u threshold uncertainty (σ_{u*})*

In order to estimate σ_{u*}, annual NEE was calculated by filtering the nighttime data using plausible u* thresholds around 0.13 (0.08-0.18) and filling the data. σ_{u*} was then calculated as half the difference between the annual NEE values calculated using those thresholds (Rutledge et al., 2015).

Estimation of the cut-off frequency uncertainty (σ_{f_0})

In order to estimate σ_{f_0}, the standard deviation of the cut-off frequency distribution (0.05 Hz) was calculated. New linear regressions of the correction factor as a function of the wind velocity were established for two new cut-off frequencies 0.37 ± 0.05 Hz for stable and unstable conditions. The fluxes were then corrected using the regression parameters and an annual NEE was calculated for both cut-off frequencies. σ_{f_0} was then calculated as half the difference between those values.

Estimation of the total NEE uncertainty (σ_{NEE})

These sources of NEE uncertainties were combined following the random error propagation rules. σ_{gf} was added as a positive one-sided uncertainty. For the 5-year average uncertainty, σ_{gf} , σ_{u*} and σ_{f_0} were simply averaged while σ_r was averaged following the random error propagation rule.

Estimation of the total NBP uncertainty (σ_{NBP})

In order to estimate uncertainties for C flux other than NEE, we considered that errors associated with data obtained from the farmer amounted to 10% (Ammann et al., 2007) and then randomly cumulated this error with uncertainties associated with laboratory measurements.

By assuming the independence and normality of the different error sources, NBP standard deviation (σ_{NBP}) was calculated by squaring each error term, totaling the resulting values and then taking the square root of the sum (Mudge et al., 2011).

3. Results

3.1. Meteorological conditions and management practices

Both air and soil temperatures and PPFD followed a typical seasonal pattern that did not really differ from one year to another. The highest temperature values (around 17°C) were observed during summer in July and August (Figure 2-3 a and b), whereas the highest PPFD values (around 450 µmol m^{-2} s^{-1}) were observed from May to July (Figure 2-3 c). Precipitation was widespread throughout the year. (Figure 2-3 f). The soil water filled pore (WFP) space at 5cm, calculated as the ratio of SWC and SWC at saturation, dropped to 32% in May 2011. Low precipitation, high vapor pressure deficit (VPD) values (Figure 2-3 e) and high temperatures occurred during the same period, suggesting a drought event. The summer of 2013 was also a dry period, with less than average precipitation in July and August, leading to low WFP (38%). At the end of March, GDD was 531°C day in 2014 (highest value), 426°C day in 2011, 410°C day in 2012 and 194°C day in 2013 (lowest value). The low GDD in 2013 is a result a prolonged snow period and colder temperatures until mid-April.

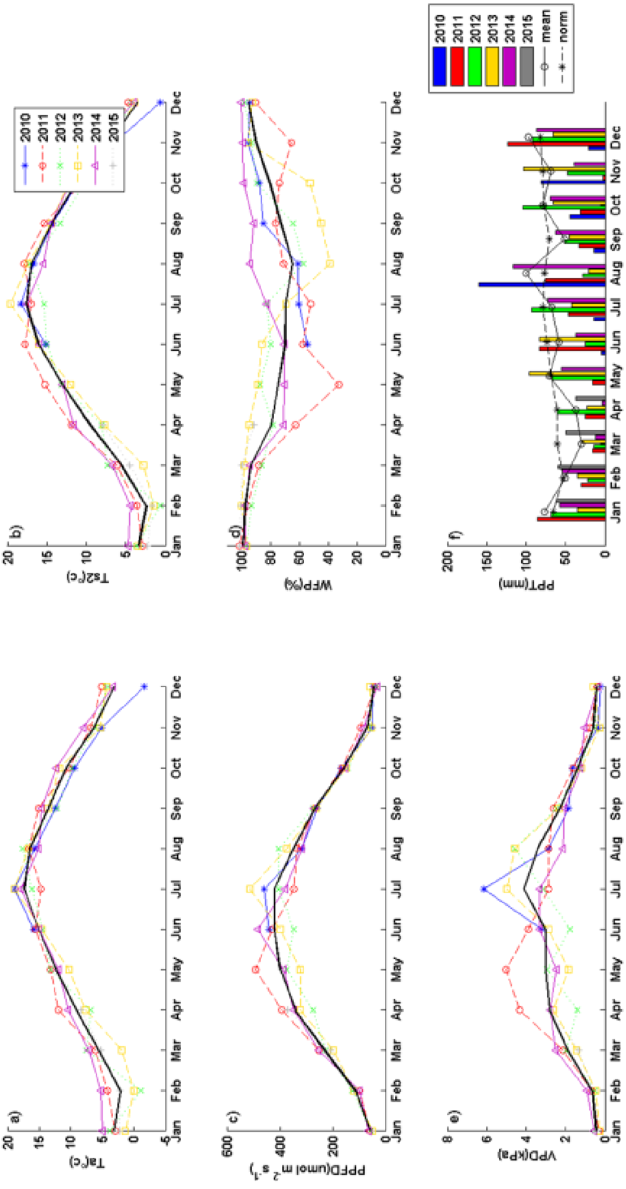

Figure 3-3: Monthly means of a) air temperature (T_A), b) soil temperature at a depth of 2 cm (T_{S2}), c) photosynthetic photon flux density (PPFD), d) soil water filled pore (WFP) space at a depth of 5 cm, e) vapor pressure deficit (VPD), and f) monthly precipitation totals (PPT). Circles connected by a continuous line indicate the 5-year averages of monthly total precipitation. Stars connected by an unbroken lines represent the last 30-year local normal precipitation averages for the Institut Royal Météorologique’s Ciney station, 15 km south-east of the study site.

Over all 5 years of the study, annual averages were within a narrow range for the main environmental variables: air temperature T_A = 9.6°C ((9.0-10.3), soil temperature at a depth of 2 cm T_{S2} = 10.1°C (9.6-10.9), PPFD = 239 µmol m^{-2} s^{-1} (214–249), VPD = 2.00 kPa (1.72-2.29), WFP = 0.82% (72–89) and precipitation PPT = 628 mm (508–672) (Table 2-3a). The annual averaged air temperatures and cumulated precipitation were significantly lower than the 30-year local normal averages (10°C and 847 mm, respectively, reported by the Institut Royal Météorologique's Ciney station, 15 km south-east of the site).

Grazing started on different dates, depending on grass availability and technical constraints. It began as early as 24 March in 2012 and as late as 3 May in 2014 because of delay in the experimental set up installation (Figure 2-4). In 2010, it began only on 12 June, but was preceded by a harvest on 3 June 2010. In 2013, it started only on 25 April because of low temperatures. The average stocking rate was the lowest in 2010 because a considerable amount of biomass had been harvested in June 2010 and was therefore not available for cattle. On average, cattle grazed for 160 days yr^{-1} (from 134 to 202 days yr^{-1}) and the average stocking density during stocking periods was 5.3 LU ha^{-1} (from 7.5 to 2.2 LU ha^{-1} with four one – day confinement periods around 10-12 LU ha^{-1}). The annual average stocking rate, including stocking and rest periods, was therefore 2.3 LU $ha^{-1}year^{-1}$.

The average grass height in the field varied from 4 to 10 cm during the grazing season and reached a minimum value of 3cm in end November (Figure 2-4). Every year, the stocking density was always lower at the end of the grazing season when biomass availability was the lowest and the highest from May to mid-September when grass availability was the highest. Rest periods occurred generally when grass height went down to 5cm or below with a notable exception in 2014, when a permanent grazing was organized for experimental purpose. Overall, 19 rotations between rest and stocking periods were observed during grazing seasons from 2010 to 2013. These adaptations of the stocking density and the grazing duration to grass availability, following usual management practices, induces indirectly some link between grazing management and weather conditions as the latter control, at least partly, grass growth.

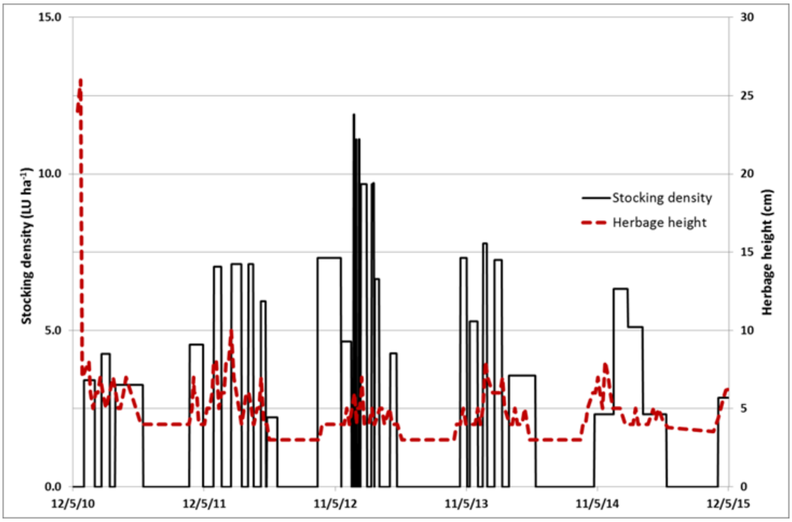

Figure 3-4: Cattle stocking rate (LU/ha) throughout the study period and herbage height. A stocking rate of zero designates rest periods.

3.2. Monthly dynamics of NEE, TER and GPP

The 5-year average of monthly TER and GPP values both followed a seasonal cycle, being minimal in winter and maximal in summer, but not at the same time: GPP reached its maximum value (about 310 gC m^{-2} month^{-1}) between April and June, whereas TER reached it (about 280 gC m^{-2} month^{-1}) between June and August (Figure 2-5). As a result, the monthly 5-year average NEE showed a continuous CO_2 uptake during spring and early summer (March- July), reached its maximum uptake in April, fell to zero around mid-summer (August) and moved to continuous CO_2 emission in autumn and winter. This shift from a CO_2 sink to a source occurred earlier than observed in other temperate ecosystems, such as forests (Aubinet et al., 2002; Falge et al., 2002), probably as the result of grazing that limits vegetation photosynthesizing area and, as a consequence, the GPP.

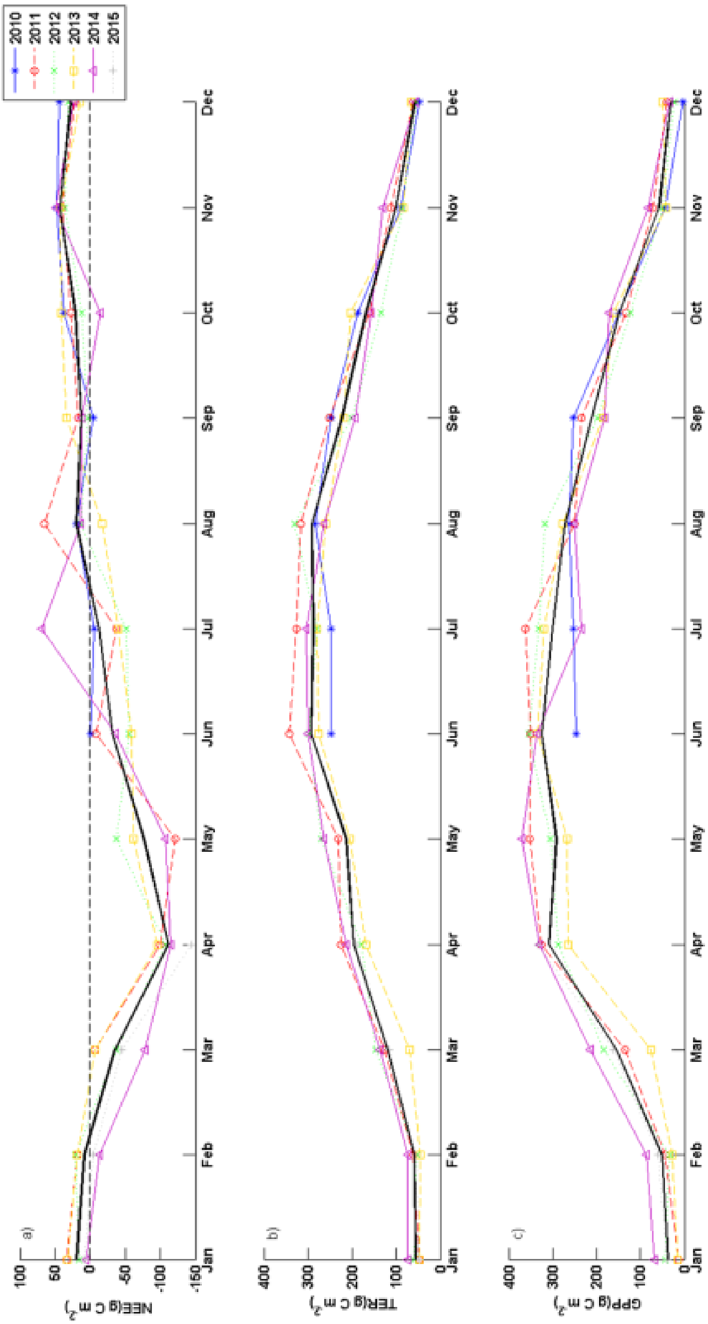

Figure 3-5: Monthly totals of the a) net ecosystem exchange (NEE), b) total ecosystem respiration (TER) and c) gross primary productivity (GPP). The dark black continuous line indicates the 5-year average for each month.

A highly significant linear relationship was found between monthly TER and GPP (pvalue<0.001, R^2=0.84), (Figure 2-6, a). The slope of the regression was 0.72. This dependence should be treated with caution however, because self-correlation between TER and GPP could also derive from the partitioning method used to compute these fluxes (Vickers et al., 2009).

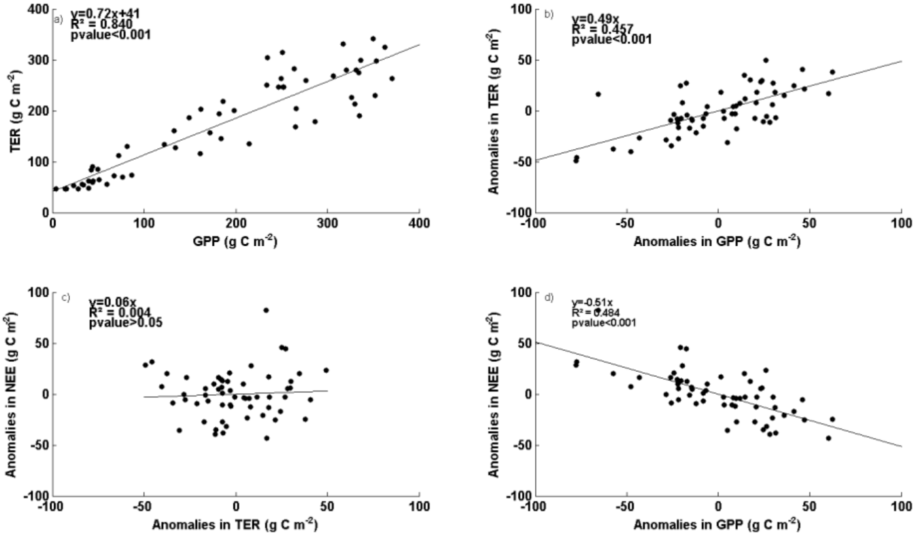

Figure 3-6 : Correlation between a) monthly total ecosystem respiration (TER) and monthly gross primary productivity (GPP), b) anomalies in monthly total ecosystem respiration (TER) and anomalies in monthly gross primary productivity (GPP), c) anomalies in monthly total ecosystem respiration (TER) and anomalies in net ecosystem exchange (NEE) and d) anomalies in monthly gross primary productivity (GPP) and anomalies in net ecosystem exchange (NEE).

In order to assess the impact of meteorological conditions on the C budget inter-annual variability, flux (GPP, TER and NEE) and weather variable (Ts, VPD, WFP, PPFD and precipitation), various anomalies were also investigated. A significant relationship was found between TER and GPP anomalies (pvalue<0.001, R^2=0.42), (Figure 2-6, b). The slope of the regression was 0.48 (pvalue<0.001). NEE anomalies were correlated with GPP anomalies (pvalue<0.001, R^2=0.43) but not with TER anomalies (pvalue>0.05, Figure 2-6, c and d). Monthly GPP and TER anomalies were also both correlated with Ts anomalies (pvalue<0.001, data not shown), but no such relationship was found for NEE. Here again, we cannot exclude the dependence partly resulting from the partitioning method used to compute TER and GPP. No other significant relationship was found between monthly CO_2 flux component anomalies (GPP, NEE, TER) and other meteorological variables.

3.3. *Carbon budget of the pasture*

The 5-year C budget reveals that the pasture behaved each year as a significant C sink (Table 2-3c). The 5-year average annual NBP was -161 [-134 -180] g C m^{-2} yr^{-1} (values in brackets indicate error bounds). This observation is in agreement with most European studies of C fluxes in grasslands, which have found that grasslands generally act as a net C sink (Allard et al., 2007; Ammann et al., 2007; Jaksic et al., 2006; Mudge et al., 2011; Peichl et al., 2011; Rutledge et al., 2015; Zeeman et al., 2010). Let’s note however, that such agreement was not a priori obvious, in view of the high management intensity and the old age of the pasture. The site has indeed been a grassland for probably more than a century and the average annual stocking rate of 2.3 LU ha^{-1} was more than twice the rate observed for most other intensively grazed European grasslands studied (1 LU ha^{-1} in Klumpp et al. (2011), from 0.12 to 1.32 LU ha^{-1} in Soussana et al. (2007)).

Table 3-3 : Annual and 5-year averages for the 5 years of measurements made at the Dorinne Terrestrial Observatory. The 5-year averages are calculated from 12 May 2010 to 12 May 2015. Annual values are given only for the complete years (2011-2014). Consequently, the average given in fifth column *is not* the average of the four first columns. All fluxes and uncertainties were rounded to the unity. An uncertainty of zero means that it is < 0.5. a) Weather variables: air temperature (T_A), soil temperature at a depth of 2 cm (T_{S2}), photosynthetic photon flux density (PPFD), vapor pressure deficit (VPD), soil water filled pore (WFP) space at a depth of 5 cm and yearly cumulated precipitation (PPT). b) Information on grazing conditions: number of grazing days and average stocking rate (SR). c) Carbon fluxes included in the net biome productivity (NBP) budget (see equation 1): total ecosystem respiration (TER); gross primary productivity (GPP); net ecosystem exchange (F_{CO2}); C lost through methane emissions by cattle (F_{CH4}); C imported through manure applications (F_{manure}) and through supplementary feed (F_{import}); C exported through harvest ($F_{harvest}$) and as meat ($F_{product}$); organic and/or inorganic C lost through leaching (F_{leach}). d) Other carbon fluxes of interest: above-ground net primary productivity (ANPP), C intake through grass consumption by cattle ($C_{grazing}$), C intake by cattle (sum of $C_{grazing}$ and F_{import}), livestock CO_2 emissions ($F_{CO2,livestock}$) and livestock C excreted ($C_{excretions}$).

(a) Environmental variables	**2011**	**2012**	**2013**	**2014**	**5-year mean**
T_A (°C)	10.3	9.3	9.0	9.4	9.6
T_S (°C)	10.9	9.6	9.9	9.7	10.1
PPFD (µmol m^{-2} s^{-1})	249	228	236	214	239
WFP %	72	84	80	89	82
SWC (m^3 m^{-3})	0.36	0.42	0.37	0.31	0.37
PPT (mm)	568	672	644	508	628
(b) Management					
Total of grazing days	134	157	161	202	160
Average SR (livestock unit ha^{-1})	2.1	2.8	2.3	2.6	2.3
(c) NBP components (g C m^{-2} y^{-1}					
TER	2260	2091	1921	2164	2085
GPP	2313	2250	2024	2357	2226
NEE	-52 [-25 -64]	-159 [-140 -176]	-102 [-85 -111]	-193 [-158 -218]	-141 [-115 -158]
F_{CH4-C}	14 ± 1	12 ± 1	8 ± 1	10 ± 1	12 ± 1
F_{manure}	-111 ± 18	0 ± 0	0 ± 0	0 ± 0	-22 ± 4
F_{import}	-18 ± 1	-11 ± 1	0 ± 0	0 ± 0	-26 ± 2
$F_{harvest}$	0 ± 0	0 ± 0	0 ± 0	0 ± 0	8 ± 1
$F_{product}$	9 ± 0	4 ± 0	0 ± 0	0 ± 0	3 ± 0
F_{leach}	7 ± 7	7 ± 7	7 ± 7	7 ± 7	7 ± 7
NBP	**-160 [-127 -183]**	**-147 [-127 -156]**	**-87 [-69 -98]**	**-176 [-141 -200]**	**-161 [-134 -180]**
(d) Others C fluxes (g C m^{-2} y^{-1})					
ANPP	392	385	249	365	355
$C_{grazing}$	372	323	230	286	312
$F_{CO2,livestock}$	273	232	161	204	234
$C_{excretions}$	102	86	61	72	87

Looking to the carbon budget (Table 2-3 c), it appears that the main terms were, in order, NEE, F_{import} and F_{manure} (Table 2-3c). NEE ranged from -193g C m^{-2} in 2014 to -52 g C m^{-2} in 2011. The high 5 years average F_{import} value is mainly due to the importation in 2010 of an important C amount (about -100 g C m^{-2}) as a feed supplement (Table 2-1). This feed was imported to compensate for the harvest in June that year (40 g C m^{-2}) and the low precipitation from May to July (Figure 2-3 f), which could have limited grass regrowth. These fluxes affected the 5-year mean budget, but did not appear in the yearly budgets because they occurred in the incomplete year, 2010. Except this contribution, feed supplements remained low compared with NEE. No feed supplements were imported into the field in 2013 and 2014 because the farmer adjusted the stocking rate such that grass regrowth was enough to feed the cattle.

F_{manure} corresponds with the C imported into the field through organic fertilization. It was the most important part of the NBP budget in 2011. As organic fertilization occurred only once during the study period its impact on the average budget was finally small. This is representative of the real management of the pasture as, according to the farmer, organic fertilization frequency is not higher than once every 5 years.

3.4. Inter-annual variability of the carbon budget

Apart from 2013, when it dropped to -87 g C m^{-2}, the NBP did not vary significantly from year to year, remaining at about -161 g C m^{-2}, which indicates a relatively stable annual C budget. These budgets, however, were obtained under contrasting weather conditions and, on a monthly scale, some differences in NEE were notable.

In 2011, a peak emission (NEE anomaly ≈+50 g C m^{-2}) was observed in August (Figure 2-5), however, an important amount of C had also been imported through organic fertilization (Table 2-3c) in February in the same year. These two events impacted the annual NBP in opposed ways and compensated each other. In 2014, the first half of the year (February to June) was characterized by an above-average CO_2 uptake (Figure 2-5a), due to mild winter conditions. However, later in July, an emission peak occurred (NEE anomaly ≈+80 g C m^{-2}), due to below-average GPP (Figure 2-5c). Here again, these events, although significant at monthly scale did not affect the annual NBP due to mutual compensation. Finally, in 2013, the beginning of the year was characterized by prolonged cold and snowy conditions, which induced below-average GPP, TER and NEE values, which probably partly explains the lower NEE for this year.

4. Discussion

The effects of weather and management practices on the C budget are not always easy to discern. A major reason for this is that weather and management might be inter-related by several processes. Indeed, as suggested at section 3.1, a link between grazing management and grass availability and hence, meteorological conditions might exist. Therefore, in order to facilitate the discussion, the effects of climate and management that have been clearly identified will be discussed first separately, after which their combined effects will be assessed when possible.

4.1. *Weather impact*

The absence of relationship between NEE anomalies and weather variables anomalies (Section 3.2) suggests that, apart from the possible response of TER and GPP to temperature, the inter-annual variability of monthly fluxes could not be explained by any overall response to weather conditions. However, despite this absence of relationship, some weather effects were identified for specific periods without cattle.

The relationship between GPP and GDD was found to be similar for three successive years, from 2012 and 2014 (Figure 2-7). As a result, the inter-annual differences between cumulated GPP at the end of March were explained by the GDD. In particular, the high GPP in spring 2014 (375 g C m^{-2}) was explained by the high GDD (about 550°C day) resulting from mild winter conditions, whereas the low GPP in spring 2013 (125 g C m^{-2}) was explained by a lower GDD (around 190°C day) indicating colder winter and spring. This resulted in differences in GPP, TER and NEE of, respectively 250, 120 and 130 g C m^{-2} between those years. In 2011, however, the GPP increase with GDD was delayed and slower (cumulative GPP around 100 g C m^{-2} for 300° day and around 180 g C m^{-2} in 2012 and 2014) probably because of the high temperatures (Figure 2-3a) and low radiation (Figure 2-3c) in February. This led to an early increase in GDD associated with low PPFD, leading to a low GPP/GDD ratio.

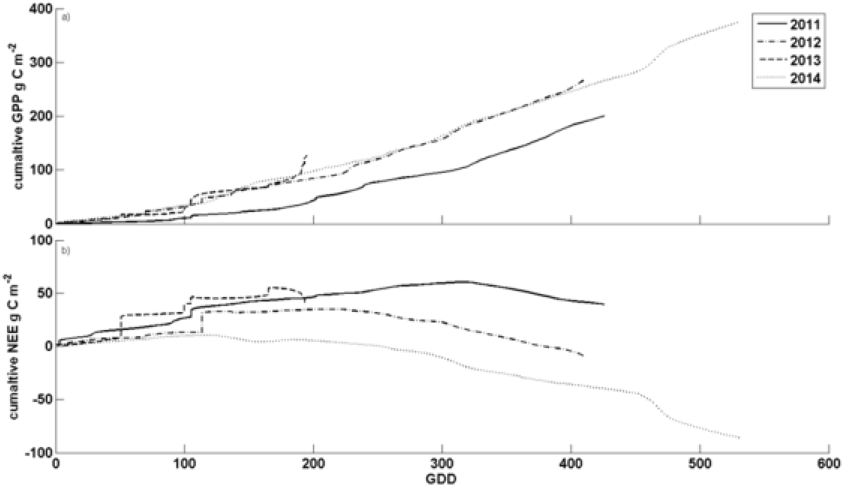

Figure 3-7 : Evolution of (a) the cumulated gross primary productivity (GPP) and (b) the evolution of the cumulated net ecosystem exchange (NEE) in relationship to the cumulated growing degree days (GDD) from 1 January to 31 March.

The high TER values observed in 2011 could have resulted from either high temperatures or the organic fertilization and liming in February that year. In order to identify the most probable cause, the normalized respiration at 10°C (R_{10}) was calculated for each year by fitting an exponential relationship onto the valid night fluxes (Lloyd and Taylor, 1994). As no significant difference between R_{10} values in 2011 and in the other years was found we concluded that high TER observed in 2011 resulted more probably from the high temperatures (Figure 2-3 and b) rather than from an increase in emission due to organic fertilization.

4.2. Management impact

The 5-year averaged GPP and TER values reached 2226 and 2085 g C m^{-2} yr^{-1}, respectively, and were larger than all the values obtained by a multi-site analysis (Gilmanov et al., 2007) of 19 European grasslands (maximum values: 1874 and 1621 g C m^{-2} yr^{-1} for GPP and TER, respectively). They were closer to the values observed in an intensive grassland study by Mudge et al. (2011) (2194 and 1999 g C m^{-2} yr^{-1} for GPP and TER, respectively), but lower than those reported by Zeeman et al. (2010) (2647 and 2583 g C m^{-2} yr^{-1} for GPP and TER, respectively). These high values are probably due to a high biomass production, itself resulting from intensive management and fertilization (120 kg N ha^{-1} yr^{-1} on average, Table 2-1). This was confirmed by the annual ANPP values (Table 2-3 d) that reached 355 g C m^{-2} on average, which is higher than the average production in Wallonia permanent cut grasslands (on average, $\cong$ 250 g C m^{-2} for the 2008-2010 period; (SPW, 2010)). In comparison, Klumpp et al. (2011) reported a much lower values of 95 g C m^{-2} ANPP and about 1650 g C m^{-2} TER and GPP.

These results suggest that, even in presence of a very high grazing pressure, high C assimilation could probably be maintained at the DTO thanks to intensive nitrogen fertilization and natural fertilization through excreta. Similar results were found by Allard et al. (2007), who showed that an intensively grassland could maintain a C sink activity over time while an extensively managed one could not.

The lateral fluxes resulting from C import or export as manure, feed supplement, harvest or meat production had clear effects on C balance. On average, lateral organic C fluxes increased the C sink magnitude. This observation differs from the findings reported in other studies (Allard et al., 2007; Ammann et al., 2007; Byrne et al., 2007; Jaksic et al., 2006; Mudge et al., 2011; Peichl et al., 2011; Rutledge et al., 2015; Zeeman et al., 2010) and is because C imports through organic fertilization and feed supplements exceeded C exports. Indeed, C exports were much lower than in those studies as only one harvest occurred during the 5 years and C exports through meat ($F_{product}$) were much lower than C exports in form of milk in dairy pastures (Byrne et al., 2007; Jaksic et al., 2006; Mudge et al., 2011; Rutledge et al., 2015; Zeeman et al., 2010). C exports through meat were low mainly because the field was most of time occupied by fully grown cattle.

Land use and management prior to the study are suspected to affect the carbon assimilation of a pasture for about a century before reaching equilibrium (Smith, 2014). As the pasture was intensively managed for more than 40 years, we can argue that this hypothetic equilibrium was not reached yet a DTO. This observation is therefore in agreement with the assumption made by Smith (2014) that equilibrium should only occur after several decades (and at least more than 40 years) under continuous management.

4.3. Combined weather and management impact

Maintaining a steady meat production and optimizing grass consumption require a careful herd management from the farmer by continuously adapting stocking density to grass availability. As grass regrowth depends on weather conditions and photosynthesizing area, it is logical to conclude that management is achieved in response to weather conditions. As a result, grass height is subjected to small variations all over the season, being maintained in a range of 5-10 cm (Figure 2-4). As a consequence of this link, impacts of climate and management on NEE are difficult to distinguish and sometimes they compensate each other. This could explain why no clear relationship between NEE and weather anomalies was found (Wayne Polley et al., 2008) and, reciprocally, why grazing impact on CO_2 flux dynamics was difficult to discern on both the monthly and seasonal scales (E. Jérôme et al., 2014).

Possible indirect impacts of grazing are the decrease of GPP because of photosynthesizing area reduction following grass consumption but also a decrease of TER via a decrease in autotrophic respiration. The latter is notably supported by the strong coupling observed between GPP and TER. However, an investigation made at DTO by Jérôme et al. (2014) showed that as the impact of grazing intensity on GPP was observed, no such impact was observed on dark respiration suggesting therefore a larger impact of grazing on GPP than on TER. Indeed TER may not only be impacted negatively through defoliation but also positively trough cattle and feces respiration.

A direct impact of grazing is the increase of TER due to cattle respiration to the TER. This effect is not easy to discern as the number of cattle within the footprint varies and is not known (Felber et al., 2016b). To do so, we studied the animal C budget (Figure 2-1, Table 2-3 d). It appeared that around 70 % of total ingested C ($C_{grazing}$ + F_{import}) was lost through cattle respiration ($F_{CO2,livestock}$). Assuming an ideal case where animals are spread evenly over the field at all times so that their respiration signal becomes a constant part of the eddy covariance measurements footprint and considering an average stocking rate of 2.3 LU ha^{-1} yr^{-1}, this represented around 11% of the TER on average.

4.4. Uncertainties

The 5-year average NBP uncertainty was [+27 -19] g C m^{-2} yr^{-1} (Table 2-3c). The main factor influencing NBP uncertainty was NEE, which itself was affected the choice of the u* threshold and the gap filling (Table 2-4). A comparison of the u* corrected and uncorrected fluxes in Table 2-4 suggests that, on average, the night flux underestimation led to an overestimation of the annual sink of about 61 $gCm^{-2}yr^{-1}$. However, an uncertainty results from this correction. An uncertainty of 0.05 m s^{-1}on the u* threshold led to an uncertainty of 17 g C $m^{-2}yr^{-1}$ for annual sums. The random uncertainty, when important on a half-hourly scale, decreases with time because of the partial compensation when summed. As a result, it did not exceed 6 g $Cm^{-2}yr^{-1}$ on an annual scale or 2 g C $m^{-2}yr^{-1}$ on a 5-year scale. The additional uncertainty resulting from the non-annulation of the mean residual error in the gap filling procedure (see Section 2.9), however, led to a systematic flux underestimation estimated to be 19 g C $m^{-2}yr^{-1}$.

Table 3-4 : Annual NEE uncertainty components and correction effects. All the values are given in g C m^{-2} yr^{-1}. The ‘no spectral’ correction value is the NEE value with no spectral correction, but with the u* filter. The ‘no filter’ value is the value with no filter, but with the spectral correction. The corr. values correspond to the annual NEE values with both u* and spectral corrections.

	Spectral correction			U* correction			Gap filling	Random		
Year	No spectral corr	Corr	σ_{f0}	No filter	Corr.	σ_{u*}	σ_{gf}	σ_r	Total Uncertainty	
2011	-64	-52	± 5	-145	-52	± 9	+ 24	± 6	+ 27	- 12
2012	-146	-159	± 5	-259	-159	± 16	+ 8	± 5	+ 19	- 18
2013	-98	-102	± 1	-136	-102	± 7	+ 14	± 5	+ 17	- 9
2014	-177	-193	± 3	-269	-193	± 22	+ 26	± 6	+ 35	- 23
5-year mean	-135	-141	± 2	-202	-141	± 17	+ 19	± 2	+ 26	- 17

Another critical choice was those of the reference cospectrum used for the spectral correction. The use of a local cospectrum (average sensible heat cospectra) was chosen instead of a theoretical cospectrum (Kansas cospectrum, Kaimal et al., 1972). This methodological choice had a major impact on CO_2 fluxes. Therefore, before presenting this budget, a supplementary validation of the correction procedure had been implemented using *in situ* respiration measurements. Fluxes corrected with the local cospectra were found to be in good agreement with the respiration measurements while fluxes corrected with the Kansas cospectra were found overestimated. The details of the procedure and the validation are presented in a paper by (Mamadou et al., 2016). Finally, even by taking all uncertainties into account, the fact that the pasture acts as a significant C sink each year remains a robust finding (Table 2-3 c).

5. Conclusion

This study established and analyzed the total C budget of grassland grazed by Belgian Blue cattle by combining data from CO_2 eddy covariance measurements with other C fluxes and their uncertainties. CO_2 fluxes (NEE) and non CO_2 fluxes in form of manure (F_{manure}) and feed complements ($F_{imports}$) were the main fluxes affecting the C budget, highlighting the need to include them. The results showed that the pasture acted as a relatively stable C sink each year despite the high stocking rate and the old age of the pasture. Both management and weather conditions were found to influence C fluxes. Important C imports through organic fertilization as well as low C exports through meat production helped to maintain a carbon sink. The N fertilization also probably helped to maintain the C sink activity thanks to an improved GPP. However, fertilization could also induce N_2O emissions that could affect the grassland greenhouse gas budgets. These fluxes were not measured. GPP and NEE were affected by low temperatures at the beginning of the year, before the grazing season. Indeed, these weather conditions could have caused a delay in grass growth and GPP that could not always be offset during the rest of the year.

The low inter-annual variability of the C budget and its independence to weather variables anomalies could partially be explained by management practices that adjusted the stocking rate according to grass availability which itself responds to weather conditions. It could also been obtained partly by chance as (i) we didn’t experience really extreme years and (ii) in some years, compensation between events with high and low accumulation occurred. The findings in this study are in agreement with those reported by other studies that have shown that well-managed grasslands could act as a C sinks. Further studies should focus on comparing different grazing management practices in order to better quantify and understand their impact on grassland C storage. Our study also highlighted the need to evaluate the uncertainties linked to flux measurements and to assess the sensitivity of the C budget to methodological choices, such as those linked with spectral correction and the nighttime flux filtering criterion choice, in order to assess how defensible annual C budgets are.

6. Acknowledgments

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Chapter 4

Sensitivity of the annual net ecosystem exchange to the cospectral model used for high frequency loss corrections at a grazed grassland site

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Abstract

Given the increasing use of the eddy covariance technique to estimate CO_2 fluxes, more attention needs to be paid to the measurement method used. Among other procedures, the way high frequency loss corrections are established and, more particularly, the choice of the cospectrum shape that is used to implement the correction appears particularly important in this regard. In this study, we compared three approaches to high frequency loss correction for CO_2 fluxes measured by a closed path eddy covariance system and evaluated their impact on the carbon balance at the Dorinne Terrestrial Observatory (DTO), an intensively grazed grassland site in Belgium. In the first approach, the computation of correction factors was based on the local cospectra, whereas the other two were based on Kansas cospectra models. The correction approaches were validated by comparing the nighttime eddy covariance CO_2 fluxes corrected with each approach and chamber-based total ecosystem respiration estimates. We found that the local cospectra differed from the Kansas cospectra shape, although the site could not be considered as difficult (i.e., fairly flat, homogeneous, low vegetation, sufficient measurement height). The Kansas cospectra have more spectral power at high frequencies than the local cospectra under (un) stable conditions. This difference greatly affected the correction factor, especially for night fluxes. Night fluxes measured by eddy covariance were found to be in good accord with total chamber based ecosystem respiration estimates when corrected with local cospectra and to be overestimated when corrected with Kansas cospectra. As the difference between correction factors was larger in stable than unstable conditions, it acts as a selective systematic error and has an important impact on annual carbon fluxes. On the basis of a 4-year average, at DTO the errors reach 71-150 g C m^{-2} y^{-1} for net ecosystem exchange (NEE), 280-562 g C m^{-2} y^{-1} for total ecosystem respiration (TER) and 209-412 g C m^{-2} y^{-1} for gross primary productivity (GPP), depending on the approach used. We finally encourage site PIs to check the cospectrum shape at their sites and, if necessary, compute frequency correction factors on the basis of local cospectra rather than on Kansas cospectra.

1. Introduction

In the past few decades, measurements of CO_2 fluxes using the eddy covariance (EC) technique have greatly increased around the world (Aubinet et al., 2012b; Baldocchi et al., 2012; Mizoguchi et al., 2008). These data are valuable for testing and improving the land-atmosphere flux parameterizations used in climate models (Bonan et al., 2011; Chang et al., 2013; Kato et al., 2013; Melaas et al., 2013). They are also useful for upscaling exercises (Jung et al., 2011; Papale and Valentini, 2003; Tramontana et al., 2015; Xiao et al., 2012; Yang et al., 2007) and for estimating the annual net ecosystem carbon exchange (Aubinet et al., 2009; Papale et al., 2015). Robust data are needed to prevent biases in the model outputs, as well as for the estimation of the total greenhouse gas (GHG) emissions. However, the (EC) method can fail to represent accurately surface fluxes due to a physical limitation of instrumentation (Massman and Lee, 2002), micrometeorological limitations (Aubinet et al., 2012; chapter 5), footprint heterogeneity or the turbulent nature of the transport process (Richardson et al., 2006b). In addition to other biases, however, EC measurements are known to be affected by high frequency losses, especially when using a closed-path infrared gas analyzer (IRGA) to measure molar concentrations (Aubinet et al., 2012b; Fratini et al., 2012; Ibrom et al., 2007; Mammarella et al., 2009; Runkle et al., 2012). Such losses need to be properly quantified and corrected.

Several methods dealing with high frequency losses in CO_2 measurements have been proposed in the literature. All of them involve computing the correction factor as a ratio between the integral of a reference cospectrum and the integral of the product of this cospectrum with a transfer function (Horst, 1997; Massman, 2000; Moore, 1986). A major difference between methods lies in the procedure used to compute the transfer function based either on *a priori* (theoretical or empirical) (Horst, 1997; Massman and Clement, 2004; Massman, 2000; Moncrieff et al., 1997; Moore, 1986) or *a posteriori* (experimental) methods (Aubinet et al., 2001; De Ligne et al., 2010; Fratini et al., 2012). All these methods have weaknesses and strengths and, although some progress has been made (Fratini et al., 2012), there is still some debate as to which method should be used to correct high frequency loss in EC measurements, particularly for CO_2 fluxes. Herein, the choice of the reference cospectrum used to estimate the correction factor has been investigated, using Kansas cospectra (Eugster and Senn, 1995; Horst, 1997; Mammarella et al., 2009; Massman, 2000; Moore, 1986) or local cospectra (Aubinet et al., 2001; Fratini et al., 2012) being used for this purpose. So far as we know, however, the impact of the reference cospectrum choice on the annual carbon balance has never been discussed.

This question was investigated at the Dorinne Terrestrial Observatory (DTO), an intensively grazed experimental grassland site in Belgium. The impacts of three high frequency loss correction approaches on CO_2 fluxes were compared. In the first, called the ‘local approach’, the local (sensible heat) cospectrum was taken as a reference cospectrum, whereas the other two, the ‘Kansas approaches’, used Kansas parameterization (Kaimal et al., 1972) as reference cospectra. The difference between the two latter approaches is discussed below (section 3.3).

The approaches were evaluated by comparing the nighttime EC fluxes, corrected with each approach, with total ecosystem respiration (TER) estimates obtained from a 4-month chamber measurement campaign at the site. From this, the most realistic approach has been selected. A quantitative evaluation of the impact of the three correction approaches on the annual carbon balance at the DTO was finally made using 4 years of measurements.

2. Material and methods

2.1. Site description

The study site is grazed grassland at Dorinne, 18 km South/South-East (SSE) of Namur, Belgium (50° 18’ 44’’ N; 4° 58’ 07’’ E), covering an area of 4.22 ha. The vegetation of the site is homogenous in terms of the prevailing wind direction. The site is slightly sloping. It is characterized by a colluvial topography with a South-West/North-East (SW/NE) orientation and an altitude that varies from 240 m (NE) to 272 m (S) (1 – 2 % slope). The dominant soils are colluvic regosols type, according to the FAO classification. The grassland species composition is 66% grasses (*Lolium perenne* L., *Holcus lanatus* L., *Poa trivialis* L.), 16% legumes (*Trifolium repens* L.) and 18% of other species (e.g., *Taraxacum* sp. *Ranunculus repens* L.). The mean canopy height was measured manually and varied around 0.1 m, from which we deduced the zero-plane displacement height to be of the order of 0.067 m. A detailed micrometeorological description of the site has been given by Gourlez de la Motte et al. (submitted) and Jérôme et al. (2014).

The EC system, which measured CO_2, sensible heat and water vapor fluxes, consisted of a three-dimensional sonic anemometer (CSAT3, Campbell scientific, Ltd, UK) installed on a mast at 2.62 m above the ground and a closed-path CO_2/H_2O gas analyzer IRGA (LI-7000, LI-COR Inc., Lincoln, NE, USA) maintained in a climate-controlled enclosure. Sample air was drawn from the inlet through a 6.45 m-long polyurethane tube (4 mm inner diameter) into the analyzer by a pump (N022 AN18, KNF, Neuberger, D) at a flow rate of 11 SLPM. Two PTFE filters (Swagelok 2µm and ACRO 50 1 µm, GELMAN, USA) were placed upstream of the inlet and the IRGA, respectively, in order to prevent any pollution of the measurement chamber. Zero and span calibrations were performed for CO_2 about once a month. Pure nitrogen (Alphagaz 1, Air Liquide, Liège, Belgium) was used for the zero and 350 µmol mol^{-1} CO_2 nitrogen mixture (Crystal mixture, Air Liquide, Liège, Belgium) for the span. Data were recorded automatically on a data logger (CR3000, Campbell Scientific Ltd, UK) at a rate of 10 Hz and stored on a 2GB compact flash card.

2.2. Eddy covariance measurements

2.2.1. Data processing and selection

Half-hourly sensible heat and CO_2 fluxes were computed from the 10 Hz time series data, using the EDDYFLUX software (Kolle and Rebmann, 2007) and following the standard procedures defined by Aubinet et al. (2000). A stationarity test was performed after Foken and Wichura (1996). Data for which the stationarity criterion (threshold of 30%) was not satisfied for sensible heat or CO_2 fluxes and data affected by peaks (step change larger than 10µmol mol^{-1} for CO2 concentration and 5m s^{-1} for wind speed components) were rejected from the dataset. Finally, uncorrected CO_2 fluxes lower than -2 µmol m^{-2} s^{-1} were discarded from the dataset to avoid unrealistic correction factors (Hollinger et al., 1999). Nighttime data measured under low turbulence conditions were discarded using a critical u* threshold of 0.13 m s^{-1} (Jérôme et al., 2014). The resulting selection was then segregated into two groups corresponding to stable and unstable stratification.

2.2.2. Correction approaches and calculation of the annual carbon balance

Three frequency correction approaches were compared. They were applied to stable and unstable datasets. During the validation phase, the fluxes corrected using each approach were compared with independent estimates, based on dynamic closed soil chamber measurements (Norman et al., 1992). For this comparison, only eddy covariance data collected during the chamber measurement campaigns were used. This fell between May and August 2015 and the set comprised 1100 half-hourly data for stable conditions and 2020 half-hourly data for unstable conditions. This comparison enabled us to determine the most realistic correction approach. In a second step, the three correction approaches were extended to the whole dataset, between 2011 and 2014, in order to estimate the impact of the selected approach on annual flux estimates. To this end, the corrected annual carbon budgets were computed following the standard procedure of daytime and nighttime data separation, u*-filtering (Jérôme et al., 2014), flux partitioning and gap filling. Especially, the net ecosystem exchange (NEE) was gap-filled as well as decomposed into its components (gross primary productivity (GPP), and total ecosystem respiration (TER)) using the online gap filling and flux partitioning tool (Reichstein et al., 2005). In both cases, the same data selection procedure was applied.

2.2.3. Computation of correction factors

The general procedure followed to compute the correction factors is illustrated on Figure 1.

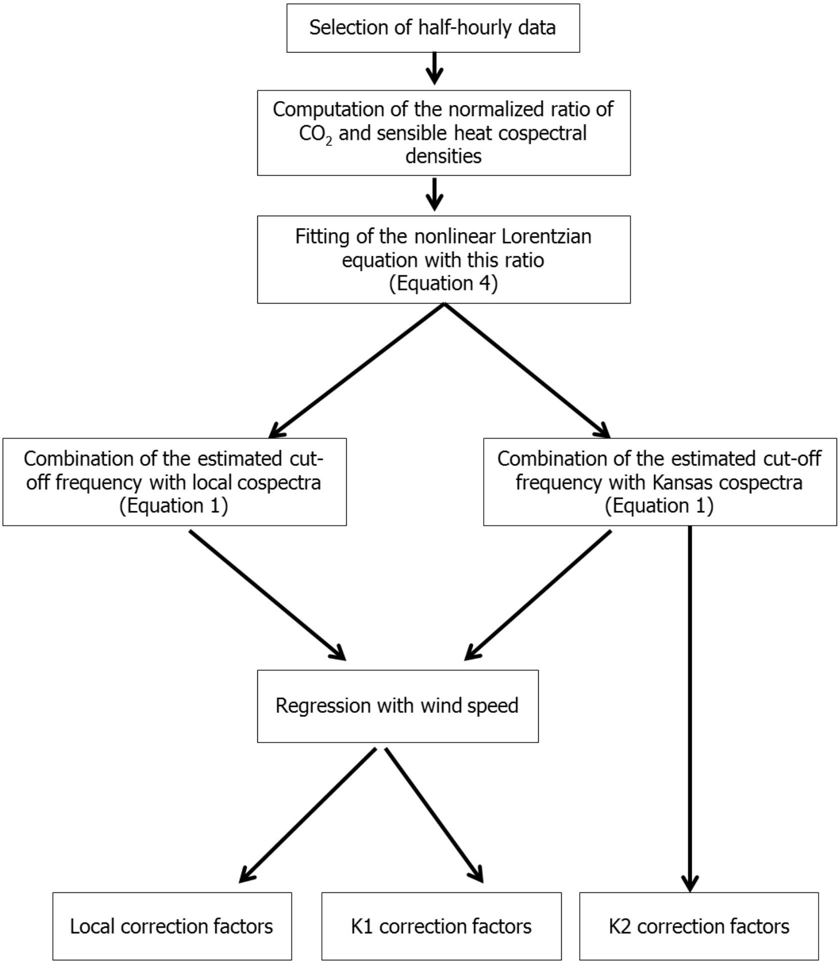

Figure 4-1: Diagram of the three correction approaches developed at the Dorinne Terrestrial Observatory. ‘L’ is the local approach based on the sensible heat cospectrum, K1 and K2 are the Kansas approaches based on the Kansas cospectra.

We computed the frequency correction factor (Φ) as (Moore, 1986):

$$\Phi = \frac{\int_0^\infty C_{ws}(f)\,df}{\int_0^\infty C_{ws}(f)\,\delta(f)df} \tag{4.1}$$

where, $C_{ws}(f)$ is the ‘ideal’ or undamped cospectral density of the vertical wind speed (w) and the scalar dry molar fraction (s), and $\delta(f)$ is the transfer function of the EC system describing the response characteristics of the system and involving high-cut filtering effects (Aubinet et al., 2012b). Implementing (Eq. 1) therefore requires knowledge of the transfer function and undamped cospectral density.

$\delta(f)$ was deduced from the normalized ratio of CO_2 and sensible heat cospectral densities:

$$\delta(f) = N_{FT}\ \frac{C_{wc}(f)df}{C_{wT}(f)df} \tag{4.2}$$

where N_{FT} is the normalization factor computed by assuming scalar similarity, i.e. proportionality of undamped CO_2 and sensible heat cospectra. Consequently, N_{FT} was computed as described in Aubinet et al. (2000):

$$N_{FT} = \frac{\int_{f1}^{f2} C_{wT}(f)df}{\int_{f1}^{f2} C_{wc}(f)df} \tag{4.3}$$

where, f_1 and f_2 are limit frequencies, chosen arbitrary to be low enough for the attenuation be negligible but high enough for the number of points used to estimate the integrals to be sufficient and the uncertainty on the normalization factors to be low (Aubinet et al., 2000).

For each half-hour, these densities were calculated using the EDDYSPEC software (Kolle and Rebmann, 2007) and following the procedure described by Foken et al. (2012b) and De Ligne et al. (2010). A nonlinear Lorentzian equation (Eugster and Senn, 1995) was then fitted on their ratio:

$$\delta(f) = \frac{1}{1+(\frac{f}{fco})^2} \tag{4.4}$$

where, f represents the natural frequency and f_{co} the cut-off frequency of the system for the CO_2. Cut-off frequency (f_{co}) was deduced from the fitting as well as its 95% confidence interval ($\sigma_{f_{co}}$). Only half-hourly data for which ($\sigma_{f_{co}}$) were lower than 0.1Hz were retained for further analysis. This corresponded approximately to 55% of the initial dataset. From this dataset, the modal value in the f_{co} distribution was calculated and kept for the remaining analysis. The same transfer function was used in all approaches.

Undamped cospectral densities could be computed in two ways: local (L) and Kansas (K) approaches. Fundamental to the three approaches is the assumption of cospectral similarity of scalars in the atmospheric boundary layer (Aubinet et al., 2012b; Fratini et al., 2012; Massman and Clement, 2004).

In the local approach, the local (sensible heat) cospectrum was chosen as the undamped cospectrum. The approximation is reasonable because the sensible heat measurements are largely unaffected by cospectral attenuation (cut-off frequency: 0.37±0.05 Hz). The computation was made as above, using a fast Fourier transform algorithm implemented with EDDYSPEC software (Kolle and Rebmann, 2007) on segments of 4,096 data points.

In the Kansas approach, undamped cospectral densities were estimated using the Kansas cospectra models for sensible heat described by Kaimal et al. (1972):

$$\frac{fC_{wT}(f)}{\overline{w\prime T\prime}} = \begin{cases} \frac{11n}{(1+13.3n)^{7/4}} & \text{for } n \leq 1 \\ \frac{4n}{(1+3.8n)^{7/3}} & \text{for } \quad n \geq 1 \end{cases} \tag{4.5}$$

in unstable conditions and,

$$\frac{fC_{wT}(f)}{\overline{w'T'}} = \left\{ \frac{0.81(n/n_0)}{1+1.5(n/n_0)^{2.1}} \right. \qquad (4.6)$$

with $n_0 = 0.23(1 + 6.4\zeta)^{\frac{3}{4}}$

in stable conditions. In Eq. (5) and (6), n is a dimensionless frequency defined as: $n = f(z_m - d)/\bar{u}$; d (m) is the zero plane displacement height; z_m (m) is the measurement height; $\bar{u}$ (m s^{-1}) is the wind speed; $\zeta = (z_m - d)/L_{MO}$ is the stability parameter; L_{MO} (m) is the Monin Obukhov length and $\overline{w'T'}$ is the measured covariance of the vertical wind speed and the air temperature.

The correction factor was then computed every half hour by combining in Eq (1) the transfer function and the reference cospectrum computed every half hour. Depending on the approach, local or Kansas cospectra were used.

In the local approach (L) and in the first Kansas approach (K1), a regression was fitted on the relation between half hourly correction factors and wind speed, separately for stable and unstable conditions. A linear regression was chosen, according to (Aubinet et al., 2001 or Fratini et al., 2012) (see section 3.1, Fig. 4). Non linearities in this response, as predicted by Massman and Clement (2004) and Wohlfahrt et al. (2005) were not taken into account here as they mainly result from a sensor separation effect, which, in the present case, is small compared to the tube attenuation effect. The effective correction factor was then estimated every half hour using this regression and wind speed data. In the second Kansas approach (K2), a correction factor was directly applied to the half hourly data. The difference between the first and the second Kansas approaches was then in their computation procedure. This allowed comparing more easily the local approach and the first Kansas approach; both followed indeed the same procedure for the computation of the correction factors.

2.3. Chamber-based TER estimates

2.3.1. Soil/grass efflux measurements

Eighteen CO_2 efflux measurement campaigns were held between May and August 2015. The measurements were taken between 10 am and 6 pm in four sectors that had been delimited around the EC tower. Three of them were situated SSW of the tower and the fourth was NE of the tower. These are the two dominant wind directions at the site. Some 28 soil collars, 15.5 cm high and 10 cm in diameter, were inserted into the soil (including the present grass) at least 3 days before the chamber soil/grass respiration measurements.

The measurements were taken manually with an EGM-4 IRGA analyzer (PP Systems, Haverhill, MA) connected to an SRC-1 chamber (PP Systems, Haverhill, MA). In total, 450 independent measurements were made. For each measurement there were three repetitions and the CO_2 concentration in the soil chamber was recorded every 4.8 s. One single measurement lasted for 120 s if the maximum change, fixed at 50 ppm, allowed in CO_2 concentration was not reached. It was automatically stopped when the maximum was reached. Finally, soil temperature (H-I 145 T-Shaped Thermometer, HANNA instruments, USA) at a depth of 5 cm and soil moisture (Theta probe ML2X, Delta–T Devices, UK) at a depth of 0-5 cm were measured manually near the soil collars.

The soil/grass respiration R_S (µmol m^{-2} s^{-1}) was calculated based on the increase in CO_2 in the chamber over time (Eq. 7) (Suleau et al., 2011), according to:

$$R_S = \frac{P_{atm} \times V}{R \times T_S \times S} \times \frac{dc}{dt} \tag{4.7}$$

where, R = 8.314 J K^{-1} mol^{-1} is the gas constant; P_{atm} (Pa) is the atmospheric pressure; V (m^3) is the chamber volume; S (m^2) is the soil surface area intercepted by the collar; T_s (K) is the soil temperature; and dc/dt (µmol mol^{-1} s^{-1}) is the rate of CO_2 concentration increase within the chamber.

A quality criterion was applied to the data, with measurements being discarded if the quality of the linear regression was not sufficient ($R^2 < 0.90$). Afterwards, they were averaged per soil collar to capture the variability between repetitions and thus their associated uncertainties

2.3.2. Cattle respiration estimate

As the pasture is grazed, total ecosystem respiration (TER) also includes cattle respiration, which is captured by EC measurements but not by soil/grass chambers. Total chamber based TER estimates (R_{ST}) were thus obtained by summing *in situ* soil/grass respiration and an estimate of cattle respiration based on the analysis performed by Jérôme et al. (2014) at the same site. They estimated the average emission per livestock unit as 2.59 kg C LU^{-1} day^{-1}. On this basis, as during our measurement campaigns the average stocking rate reached 4 LU ha^{-1}, we computed the averaged cattle respiration as 1.02 µmol m^{-2} s^{-1}. It is worth noting, that cattle respiration probably varied during the measurement campaigns because it would also have depended on cattle repartition in the footprint and on daily stocking rate changes. The impact of these variations is however limited as, in average, cattle respiration corresponds to about 12 % of soil/grass respiration.

2.4. Validation of the correction

The three high frequency loss correction approaches were evaluated by comparing the corrected nighttime CO_2 fluxes (R_{SL}, R_{SK1} and R_{SK2}) and the chamber-based TER estimates (R_S).

Nighttime hours were defined as periods when global radiation was lower than 10 W m^{-2}. Eddy fluxes were computed half hourly as the sum of the turbulent flux measured by the EC system and of the storage term (Foken et al., 2012a). In order to avoid the biases (e.g. the decrease of soil/grass respiration when anaerobic conditions prevail and depress aerobic microbial activity) that can occur when soil is saturated by water (Knowles et al., 2015; Luo and Zhou, 2006), data corresponding to soil water content at a depth of 5 cm that were higher than 30% were discarded from both nighttime EC and *in situ* soil CO_2 efflux measurements. These represented 15% and 2% of *in situ* soil CO_2 efflux measurements and nighttime eddy covariance data respectively.

In order to get rid of the temperature dependence of TER, both corrected nighttime EC fluxes (R_{SL}, R_{SK1} and R_{SK2}, ndata = 493) and total chamber based respiration measurements (R_{ST}, ndata = 381) were first binned into soil temperature classes containing 40 elements each. However, as these estimates were not obtained during similar temperature conditions, the comparison could be made only on the overlapping temperature ranges. To this aim, a second data sorting was made, constituting two classes of variable sizes covering the temperature ranges 16±2°C and 20±2 °C. Flux averages (m_i) and standard errors (ε_i) were calculated for each class. The normalized difference (u_{obs}) between the averaged chamber-based TER and averaged nighttime EC fluxes corrected by each approach was calculated for both temperature classes following :

$$u_{obs} = \frac{m_1 - m_2}{\sqrt{\varepsilon_1^2 + \varepsilon_2^2}} \qquad 4.7$$

The difference between the two estimates was considered as significant at $\alpha = 0.05$ if $u_{obs} > 1.96$ (Dagnelie, 2011).

The most realistic approach to high frequency loss correction was selected from this comparison.

The statistical tests, models and figures were made using R software (R version 3.1.2) and Matlab R2014b (Mathworks, Inc., USA) for numerical cospectral analyses.

3. Results and discussion

3.1. *Cospectral analyses and correction factors*

Figure 2 presents the normalized local and Kansas cospectra (Eqs. 5-6) and their averages on the selected half-hourly dataset, separately for stable and unstable conditions. In stable conditions, the local cospectrum reached a -4/3 slope in the inertial range, as expected, but in unstable conditions the Kansas cospectrum have more spectral power than the local one. At lower frequencies, the differences were more apparent: in both stable and unstable conditions, the Kansas cospectrum was larger than the local cospectrum between 0.1 and 1 Hz and smaller than local cospectrum at lower frequencies ($n < 0.1$ Hz).

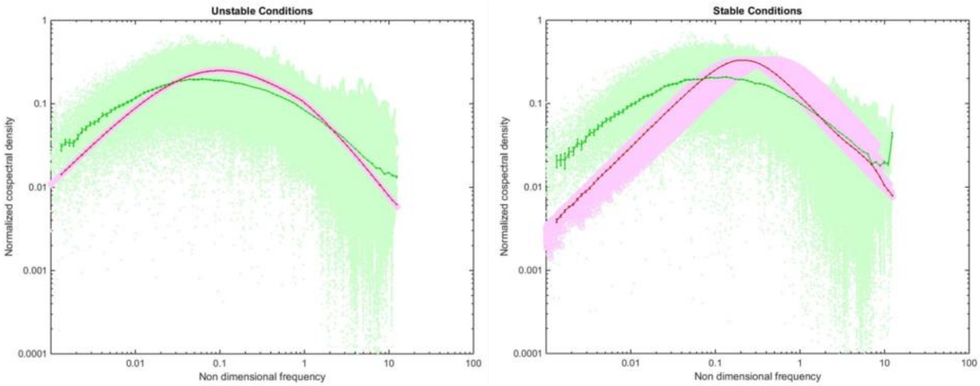

Figure 4-2: Ensemble of normalized Kansas (red dots) and local (sensible heat) cospectra (green dots) during unstable (left) and stable (right) conditions in log-log space. The red and green lines represent their respective averages.

These differences in shape clearly affect correction factors, those calculated using the local cospectrum (Φ_L) being systematically lower than those calculated with the Kansas cospectra (Φ_{K1}, Φ_{K2}). By considering a transfer function for both atmospheric stability conditions with the estimated modal value of the cut-off frequency (0.37 Hz) (Fig. 3), the deviation reached 9% in average between Φ_{K1} and Φ_L and 16% in average between Φ_{K2} and Φ_L.

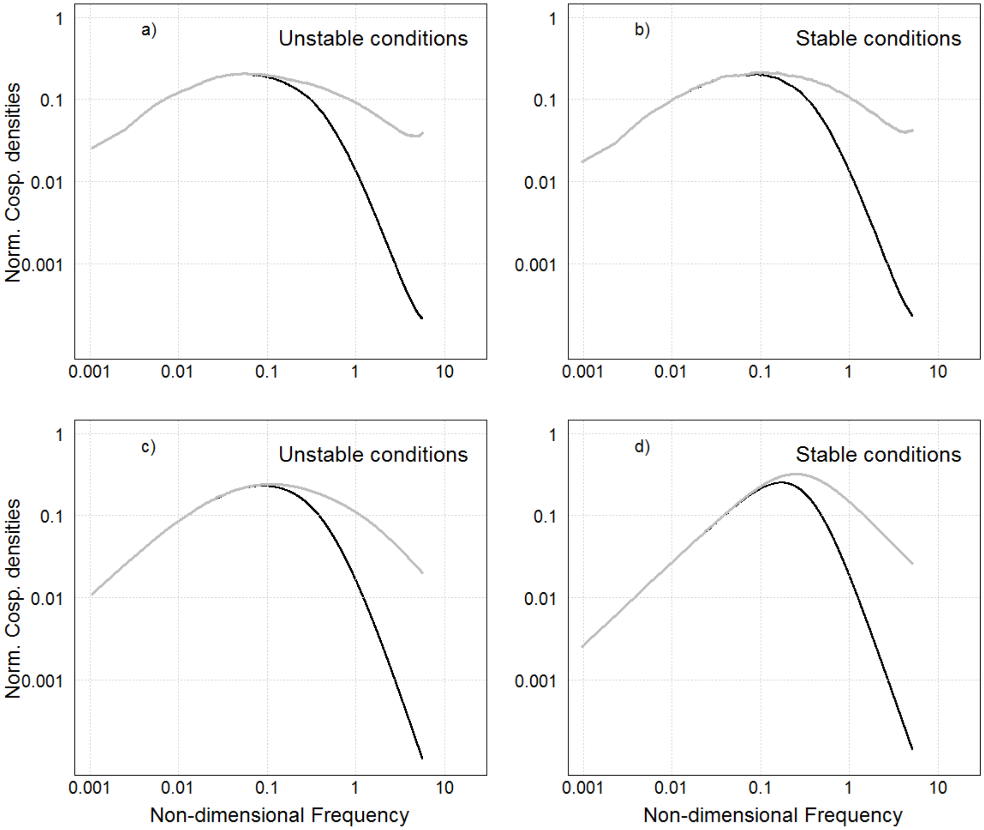

Figure 4-3: Undamped (grey color) and damped (black color) cospectra, considering a Lorentzian transfer functions with a cut-off frequency modal value of 0.37 Hz; a) and b) local cospectra; c) and d) : Kansas cospectra; for unstable (a and c) and stable (b and d) conditions.

In all cases, there was a clear increase in the correction factors with increasing wind speed (Fig. 4), as predicted by theory (Aubinet et al., 2012b). The estimated regression parameters, however, were found to be much higher for the first Kansas approach than for the local approach (Table 1). The difference was particularly large during stable conditions, where the regression slopes differed by a factor greater than two (Fig. 4b, 4d). It was smaller during unstable conditions (Fig. 4a, 4c), but nevertheless reached 37% (Table 1).

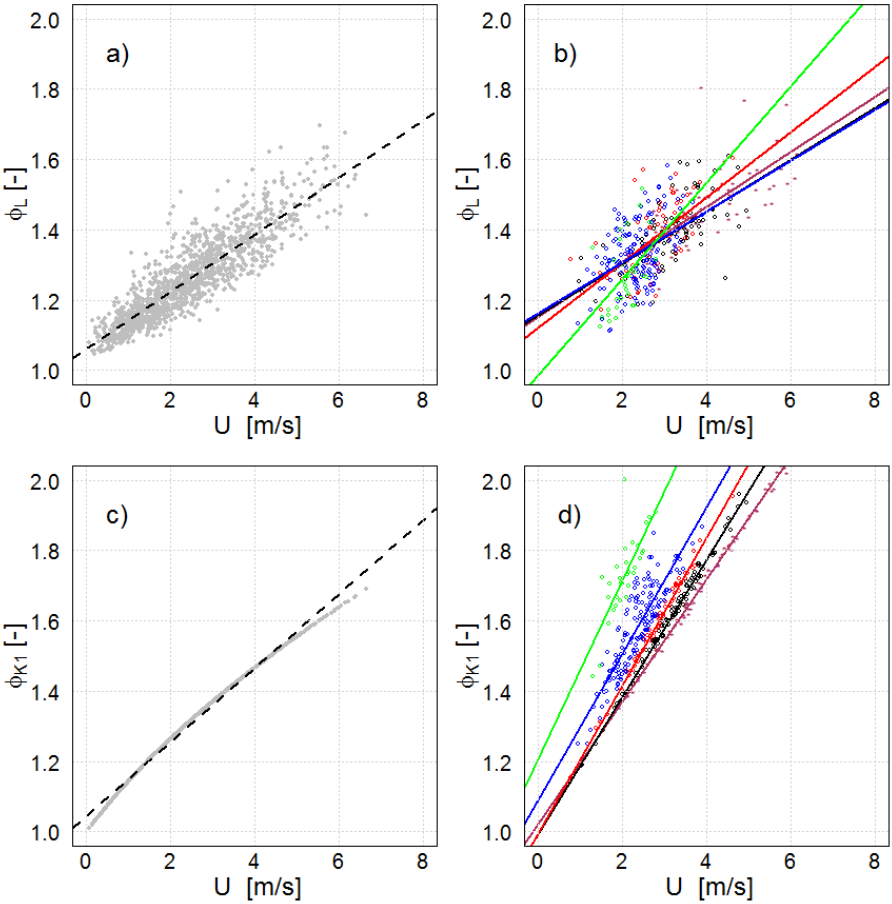

Figure 4-4: Cospectral correction factors for the local and Kansas approaches as a function of wind speed, during unstable conditions (a, c) and for different stable stratifications (b, d) obtained with the cut-off frequency of 0.37 Hz during the investigated period. Black dotted line ($\zeta < 0$) and, maroon ($0 < \zeta < 0.02$), black ($0.02 \leq \zeta < 0.04$), red ($0.04 \leq \zeta < 0.06$), blue ($0.06 \leq \zeta < 0.2$) and green ($\zeta \geq 0.2$) solid lines, represent the linear regressions. The numerical values of the regression parameters and their 95% confidence intervals are given in Table 1.

Table 4-1 Slopes, intercepts and their 95% confidence intervals of the linear regression obtained between the correction factors and wind speed during the investigated period (from May to August 2015), separately for unstable and different stable conditions and for the first Kansas (K1) and the local (L) approaches. ndata represents the number of the half hourly data that was used in the linear regression for each stability condition.

Stability ranges	**Local approach**		**Kansas 1 approach**		**ndata**
	Slope	Intercept	Slope	Intercept	
$\zeta < 0$	0.08 ±0.006	1.044±0.0024	0.11±0.001	1.06±0.0005	1251
$0 < \zeta < 0.02$	0.08±0.065	1.151±0.016	0.17±0.010	1.021±0.002	73
$0.02 \leq \zeta < 0.04$	0.07±0.056	1.158±0.017	0.20±0.011	0.992±0.003	92
$0.04 \leq \zeta < 0.06$	0.09±0.089	1.12±0.031	0.21±0.015	0.99±0.005	61
$0.06 \leq \zeta < 0.2$	0.07±0.063	1.162±0.026	0.21±0.043	1.089±0.018	150
$\zeta \geq 0.2$	0.14±0.194	0.984±0.093	0.25±0.155	1.206±0.074	31

3.2. Evaluation of the correction procedure

The evolutions with temperature of the different estimates of total ecosystem respiration are presented in (Fig. 5). As expected, the temperature ranges did not coincide exactly, due to the difference between nighttime and daytime temperatures. Over the 18 measurement campaigns, the soil temperature at a depth of 5 cm varied between 16 and 27°C in the day and between 9 and 22°C at night. Figure 5 suggests however that, in the common temperature range, the average total chamber – based TER (R_{ST}) was closer to the eddy fluxes corrected using the local approach (R_{SL}) than to the others.

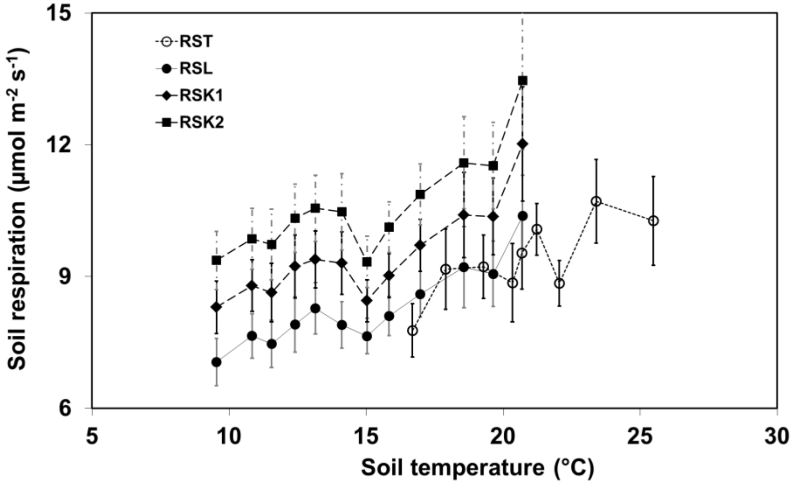

Figure 4-5: Relationship between total ecosystem respiration and soil temperature for the corrected eddy covariance data (closed symbols) and for chamber-based TER estimates (open symbols): R_{SL} (black circles) R_{SK1} (black diamonds) and R_{SK2} (black squares). Each point on the graph is an average of 40 measurements. The errors bars represent the 95% confidence intervals. Only complete data classes are represented in the figure.

In order to make the comparison more substantial, the different TER estimates were gathered in two classes covering the temperature ranges 16±2°C and 20±2°C, respectively, and their averages were compared. Results are given in Table 2. It appears first that, in both temperature classes, the normalized differences (u_{obs}) between chamber-based and eddy covariance TER estimates are all positive, suggesting that eddy covariance estimates are always larger than chamber-based estimates. However, these differences are not significant ($p > 0.05$) for fluxes computed with the local approach while they are highly significant ($p < 0.001$) for the fluxes computed with the Kansas 1 and Kansas 2 approaches. This suggested clearly that both Kansas approaches provided significantly higher estimates than the total ecosystem respiration while the local approach gave more compatible estimates. Some uncertainties still affect the comparison procedure, however. First cattle respiration was estimated as an average. This does not take changes in daily stocking rate and in cattle position in the footprint. As a result, an uncertainty remains on this term. Another issue is the choice of the reference temperature that was used to sort respiration data, which could have influenced the comparison between eddy covariance and chamber based TER estimates. In the present study we chose soil temperature at 5 cm, which appears reasonable as soil contributes largely to TER. Air temperature would have been a good alternative too and this choice would have slightly increased the difference between chamber-based and eddy covariance TER estimates. Indeed, chamber fluxes were collected during the day when air temperatures were higher than soil temperature while eddy covariance fluxes were taken from nocturnal periods when air temperatures were lower than soil temperatures. This would have induced in Figure 5 a shift to the right of chamber-based estimates and a shift to the left of the eddy covariance estimates. This would have increased again the mismatch between Kansas and chamber-based estimates but also resulted in a less good agreement between local and chamber-based estimates. Clearly, more extensive chamber campaigns, including both night and day measurements, would be necessary to refine this comparison.

Table 4-2: Results of the comparison at similar temperature between total chamber-based (R_{ST}) and eddy covariance TER estimates corrected with different approaches (R_{SL}, R_{SK1} and R_{SK2}). ndata represents the number of data in each class. p represents the probability level.

	Temperature class	R_{ST}	R_{SL}	R_{SK1}	R_{SK2}
Mean ± sterr	16	8.12 ±0.35	8.50 ±0.17	9.55 ±0.19	10.67 ±0.21
(ndata)		(59)	(144)	(144)	(144)
	20	9.39 ±0.16	9.96 ±0.31	11.44 ±0.40	12.77 ±0.47
		(202)	(89)	(89)	(89)
			R_{SL}-R_{ST}	R_{SK1}-R_{ST}	R_{SK2}-R_{ST}
Difference	16		0.38	1.43	2.55
	20		0.57	2.04	3.38
u_{obs}	16		0.96	3.55	6.19
			($p=0.33$)	($p < 0.001$)	($p < 0.001$)
	20		1.62	4.76	6.81
			($p = 0.055$)	($p < 0.001$)	($p < 0.001$)

3.3. Impact of the reference cospectrum choice and correction approach on CO_2 fluxes

3.3.1. Half hourly fluxes

The analysis was extended to the daytime EC data and the deviation between the local and Kansas approaches was quantified by comparing corrected CO_2 fluxes with the local approach and both Kansas approaches. The differences were estimated to be 14 and 28% for the nighttime CO_2 fluxes (Figs. 6a, 6b) and 4 and 9% for the daytime CO_2 fluxes (Figs. 6c, 6d). The fact that these differences were larger at night than during the day suggests that the error resulting from an incorrect correction might have acted as a 'selective systematic' error (Moncrieff et al., 1996; Rannik et al., 2004), as in the case of the u*-filtering correction (Moncrieff et al., 1996; Aubinet et al., 2001). Therefore, because the error had a greater impact on the positive fluxes than on the negative fluxes, it would be expected to result in important biases in annual sums.

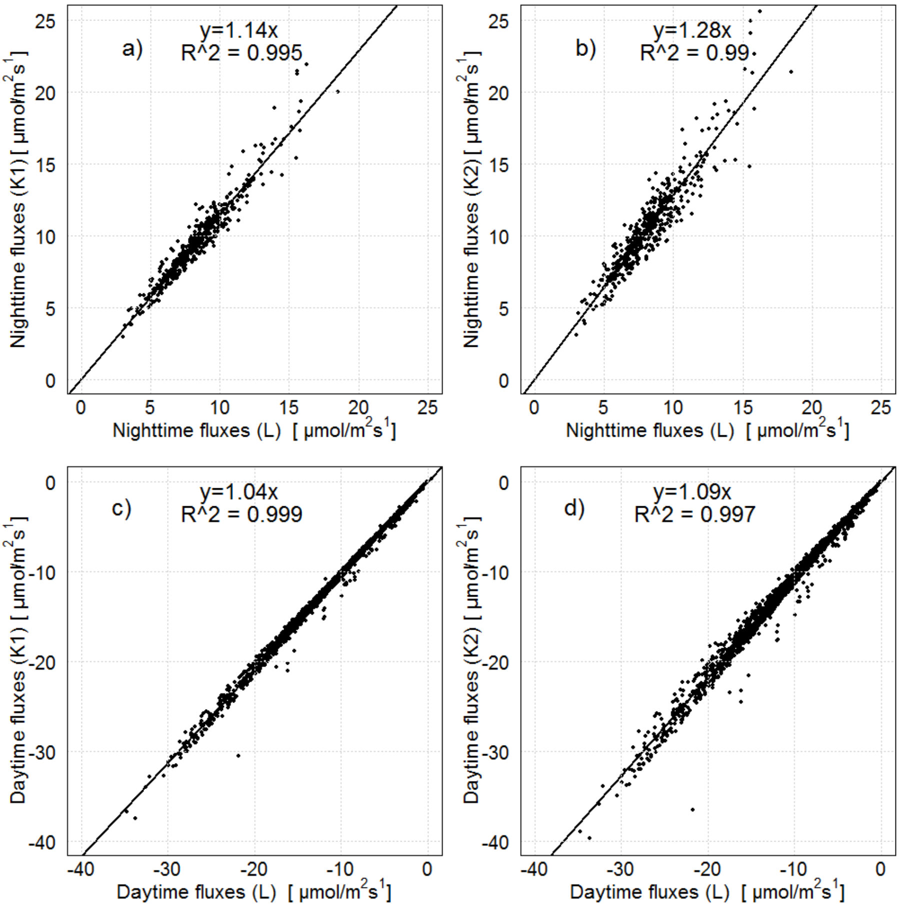

Figure 4-6: Correlation between the CO_2 fluxes corrected by the Kansas and local approaches during (a, b) nighttime and (c, d) daytime; a and c first Kansas approach; b and d second Kansas approach during the investigated period: May to August 2015.

3.3.2. Annual sums

The preceding analysis clearly showed that the choice of the cospectral correction approach was critical at the DTO. In order to evaluate the impact of such choice on the annual fluxes, the correction procedures were extended to the 2011-2014 dataset. Over these 4 years, the average flux difference between L and K1 or L and K2 amounted to 412 and 209 g C m^{-2} y^{-1} for GPP (Fig. 7a), 562 and 280 g C m^{-2} y^{-1} for TER (Fig. 7b) and 150 and 71 g C m^{-2} y^{-1} for NEE (Fig. 7c). The relative differences ranged from 9 to 19% (GPP) and 14 to 27% (TER) between L and K1 and between L and K2, respectively. This shows that the choice of reference cospectrum could significantly affect all fluxes (GPP, TER) and, in this case, change the site from being a net C sink to being a weak net C source. The fact that the approaches based on the Kansas cospectral corrections gave the highest estimates of all annual fluxes could be attributed to the site-specific cospectral shape obtained, whatever the atmospheric stability conditions, because the Kansas correction factors were always larger than those computed with the local cospectra.

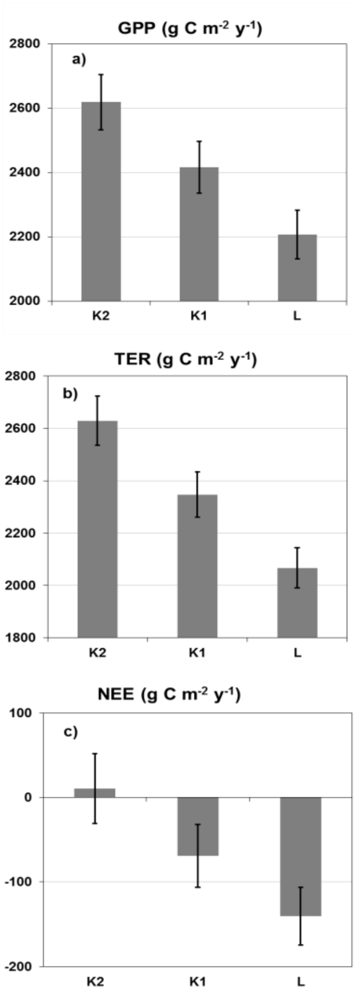

Figure 4-7: The 4-year (2011-2014) average and their standard errors of the annual estimates of (a) gross primary productivity (GPP, g C m^{-2} y^{-1}), (b) total ecosystem respiration (TER, g C m^{-2} y^{-1}) and (c) net ecosystem exchange (NEE, g C m^{-2} y^{-1}) corrected with the local (L), first (K1) and second (K2) Kansas approaches, respectively, at the Dorinne Terrestrial Observatory (DTO).

3.3.3. Shape of the cospectrum

The main reason for these differences was that the local cospectrum differed from the cospectrum shape proposed by Kaimal et al. (1972). Let remark that these differences appear although both site choice and tower design were made taking the standard requirements (Munger et al., 2012) into account. The site is almost flat (a relative slope of 1-2% in the NE direction) and has homogeneous vegetation in terms of prevailing wind direction. The measurement height is 2.62 m, much higher than the vegetation canopy, which rarely exceeds 30 cm in height. Although there are a few sparse trees in the footprint area, they are far way and we considered them unlikely to have created a flow distortion that would have explained the observed differences, particularly in the inertial subrange. The presence of a valley in the North-West direction is expected to be of small importance because it corresponds to infrequent wind occurrences. This suggests that cospectra differing from Kansas shape can be met even at sites that meet recommended quality criteria.

Previous literature provides some examples not only above complex sites (Massman and Clement, 2004; Sakai et al., 2001; Su et al., 2004), where measurements were made in the roughness sublayer in which turbulent flow is known to be affected by the size of roughness elements (Kaimal and Finnigan, 1994) but also at a flat land site (Smedman et al., 2007), at a pasture and a flat paddocks sites (Laubach et McNaughton, 2009 and over a smooth playa (McNaughton et al., 2007). By investigating how these cospectral models fitted *in situ* data at two forested sites, Su et al. (2004) found that their Kansas cospectra differed from those of sensible heat in both stability conditions, and in particular were more sharply peaked in the inertial subrange. During neutral atmospheric conditions and above a flat terrain and a rocky mountain site, Massman and Clement (2004) reported similar results as those obtained by Su et al. (2004). Smedman et al. (2007) however, argued that these differences were determined by the dynamics of the whole boundary layer rather than being simply dependent on the surface boundary conditions. An alternative parameterization of the heat cospectral density have also been proposed by Wohlfahrt et al. (2005) and Massman and Clement (2004). In view of the numerous differences reported between local cospectrum and Kansas cospectra, even at sites that could not be considered as difficult (in the sense of Finnigan, 2008), it can be expected that other sites will experience similar problems. Given the impact of the cospectrum shape on both high frequency correction and annual carbon balance, we therefore recommend that site PIs systematically check the cospectrum shape at their sites and, if necessary, compute frequency correction factors on the basis of local cospectra rather than on Kansas cospectra. Finally, the corrected fluxes should be validated when possible.

4. Conclusion

In this study, we compared three approaches to high frequency loss correction; all based on the Monin-Obukhov similarity, and evaluated their impact on the annual carbon balance at the Dorinne Terrestrial Observatory, an intensively grazed grassland site in Belgium. The CO_2 fluxes were measured using a closed-path eddy covariance system. The results showed that the correction factor based on the local cospectra was more appropriate and gave more realistic estimates of nighttime CO_2 fluxes when compared with total chamber-based TER estimates than the correction factors computed with Kansas cospectra. This is because the shapes of sensible heat cospectra at the DTO were found to differ from the Kansas shapes, having less spectral power at high frequency than the Kansas cospectra. This led to an overestimation of the cospectral correction factor, which averaged over 4 months, of 4-9% in the daytime and 14-28% in the nighttime CO_2 fluxes, depending on the approach. The impact on annual sums is huge. Especially, at the DTO, the choice of Kansas rather than local cospectra reversed the annual carbon balance from being a net C sink to being a weak C source.

As the DTO is not a complex site, we suspect that many sites could be affected by a similar problem and we thus strongly advocate site PIs to apply the spectral correction on the basis of locally established cospectra rather than on Kansas cospectra. Although challenging, a comparison between eddy covariance and chamber-based TER estimates could help validating the correction procedure.

5. Acknowledgments

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Chapter 5

Rotational and continuous grazing does not affect the total net ecosystem exchange of a pasture grazed by cattle but modifies CO_2 exchange dynamics

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Abstract

Grassland carbon budgets are known to be greatly dependent on management. In particular, grazing is known to directly affect CO_2 exchange through consumption by plants, cattle respiration, natural fertilisation through excreta, and soil compaction. This study investigates the impact of two grazing methods on the net ecosystem exchange (NEE) dynamics and carbon balance, by measuring CO_2 fluxes using eddy covariance in two adjacent pastures located in southern Belgium during a complete grazing season. Rotational (RG) grazing consists of an alternation of rest periods and short high stock density grazing periods. Continuous grazing (CG) consists of uninterrupted grazing with variable stocking rates. To our knowledge, this is the first study to assess the impact of these grazing methods on total net ecosystem exchange and CO_2 exchange dynamics using eddy covariance. The results showed that NEE dynamics were greatly impacted by the grazing method. Following grazing events on the RG parcel, net CO_2 uptake on the RG parcel was reduced compared to the CG parcel. During the following rest periods, this phenomenon progressively shifted towards a higher assimilation for the RG treatment. This behaviour was attributed to sharp biomass changes in the RG treatment and therefore sharp changes in plant photosynthetic capacity. We found that differences in gross primary productivity at high radiation were strongly correlated to differences in standing biomass. In terms of carbon budgets, no significant difference was observed between the two treatments, neither in cumulative NEE, or in terms of estimated biomass production. The results of our study suggest that we should not expect major benefits in terms of CO_2 uptake from rotational grazing management when compared to continuous grazing management in intensively managed temperate pastures.

1. Introduction

Livestock total greenhouse gas (GHG) emissions represent 14.5% of all anthropogenic GHG emissions (IPCC, 2014), among which cattle production represents 41% of the sector's emissions (Gerber et al., 2013). Therefore, there is a strong need to find and evaluate levers to mitigate these GHG emissions. During the last decade, several studies suggested that grasslands could act as important carbon (C) sinks (Klumpp et al., 2011; Mudge et al., 2011; Peichl et al., 2011; Rutledge et al., 2015; Soussana et al., 2007, 2010) with a notable site to site variability depending on several factors, such as pedoclimatic conditions and management practices. Maintaining and increasing the C sink activity of grasslands by improving their management has been identified as a lever to reduce the sector's GHG emissions (Pellerin et al., 2013; Soussana and Lemaire, 2013).

Grassland C balance and net ecosystem exchange are known to be greatly impacted by management (Smith, 2014; Soussana and Lemaire, 2013). The annual net carbon dioxide ecosystem exchange (annual NEE) is known to be directly impacted by grazing intensity through cattle respiration and indirectly through biomass consumption, natural fertilisation in the form of excreta, and soil compaction (Felber et al., 2016b, 2016a; Jérôme et al., 2014; Rong et al., 2017). The fertilisation rate also affects grassland carbon balance and carbon dioxide (CO_2) flux dynamics (Allard et al., 2007; Ammann et al., 2007; Klumpp et al., 2011; Skinner, 2013). Several studies assessing CO_2 fluxes and total C balance in rotational grazing (Campbell et al., 2015; Felber et al., 2016b; Mudge et al., 2011; Peichl et al., 2011; Rutledge et al., 2015), continuous grazing systems (Allard et al., 2007; Gourlez de la Motte et al., 2016; Klumpp et al., 2011) or both (Soussana et al., 2007) have been carried out. In those studies, grazing impacts on CO_2 exchanges were not easy to discern as they were blurred by CO2 flux responses to meteorological variables. Studies comparing CO_2 and C exchanges of both grazing methods in similar pedoclimatic conditions are very scarce (Chan et al., 2010; Cowie et al., 2013; Sanderman et al., 2015). These cited studies investigated the impact of rotational and continuous grazing by comparing direct soil organic carbon (SOC) measurements in different pastures. However, the lack of exactly similar management (stocking rates, fertilization etc.), pedoclimatic conditions and inherent SOC random variability between the investigated farms made differences difficult to analyze.

This research investigates the impact of two conventional cattle grazing methods on the CO_2 flux dynamics and its implication for the C balance. The first method, continuous grazing (CG), consists of uninterrupted grazing with variable stocking rates. It favours the ingestion of growing biomass thereby maintaining a relatively low standing biomass on the field during the whole grazing season. When well managed this method maintains a relatively stable grass height in the field by adjusting the stocking density to forage mass. This common system is not labour intensive and is well adapted to humid grasslands where grass production remains steady. The second method, rotational grazing (RG, also known as multi paddock grazing), consists of an alternation of short grazing periods (around 5 days) with high stocking densities and rest periods. During grazing periods, the forage mass accumulated during the preceding rest period is quickly eaten by the cattle leading to a rapid grass height shortening. This grazing system is commonly used in cattle production and has several advantages. First, it is very easy to keep an ungrazed paddock for harvest and therefore reduce forage loss. It is also easier to adapt the rotations to grass growth and maintain high productivity as well as good animal nutrition. It also facilitates operations such as fertilisation after grazing, scattering of livestock droppings, and the harvest of uneaten biomass because of cattle rejections, flowering etc. On the other hand, rotational grazing requires more workforce than continuous grazing, a good soil carrying capacity, and more drinking infrastructure across paddocks.

The main objectives of this study are to assess the impact of these two grazing methods on CO_2 flux dynamics as well as implications for the C balances. For this, a full grazing season (14th April to 17th November) monitoring of CO_2 turbulent fluxes using the eddy covariance (EC) method was performed simultaneously over two adjacent pastures managed according to these two grazing methods.

2. Material and methods

2.1. Site description and grassland management

This research was performed at the Dorinne Terrestrial Observatory (DTO) (50° 18’ 44’’ N; 4° 58’ 07’’ E) in southern Belgium. The mean air temperature is 10 °C and annual precipitation is 847 mm. Briefly, the site consists of two adjacent intensive permanent grasslands both similarly managed by the same farmer before the experiment (Figure 1). The carbon balance and management of one of the pastures has been described in detail in a preceding paper (Gourlez de la Motte et al. (2016)), the second one has been added and fully equipped for the present experiment. Both pastures have been grazed by Belgian Blue cattle and fertilised using organic and mineral fertilisers for more than 40 years. According to the farmer there has been no vegetation restoration for more than 40 years. The grassland species composition is mainly grasses, with legumes and other species. The dominant species are perennial ryegrass (*Lolium perenne* L.) and white clover (*Trifolium repens* L.). The main wind directions are south-west and north-east. The site used for this study is part of a commercial farm so that stocking rates, fertilization rates and other management practices are, as much as possible, representative of the common practices in beef cattle farms around the region.

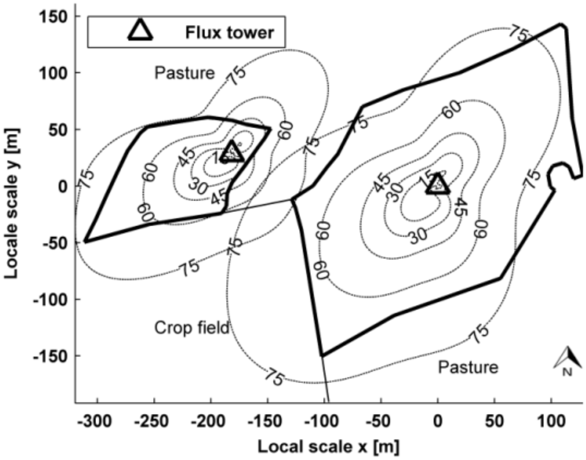

Figure 5-1: Plan of the measurement site with both the rotational grazing parcel (RG) and the continuous grazing parcel (CG). Cumulative footprint contributions for the whole measurement season are illustrated by the dashed lines. Contribution levels are given in the labels for each line.

The continuous grazing treatment (labelled “CG”) was operated on a 4.2 ha pasture. The pasture was fertilised in March 2015 with 7 kg N ha^{-1} just before the beginning of the experiment. The field was continuously grazed from 14th April 2015 to 17th November 2015 (220 days) with a varying stocking rate depending on forage availability and weather conditions (Figure 2). The annual stocking rate was 2.1 LU ha^{-1}.

In order to simulate rotational grazing (labelled “RG”), a plot of 1 ha was delimited within a bigger pasture for the purpose of the experiment (Figure 1). The field was grazed with an alternation of high stocking density periods and rest periods (Figure 2). A total of six grazing periods, each an average of six days with a stocking density of 19.3 LU ha^{-1} were carried out, leading to 36 days of grazing and an average annual stocking rate of 1.9 LU ha^{-1}. The cattle were confined in the parcel when grass height was between 10 and 15 cm. The stocking densities and grazing duration were adapted, so that similar stocking rates were obtained for both treatments with stocking densities and grazing durations in agreement with common practices in the region.

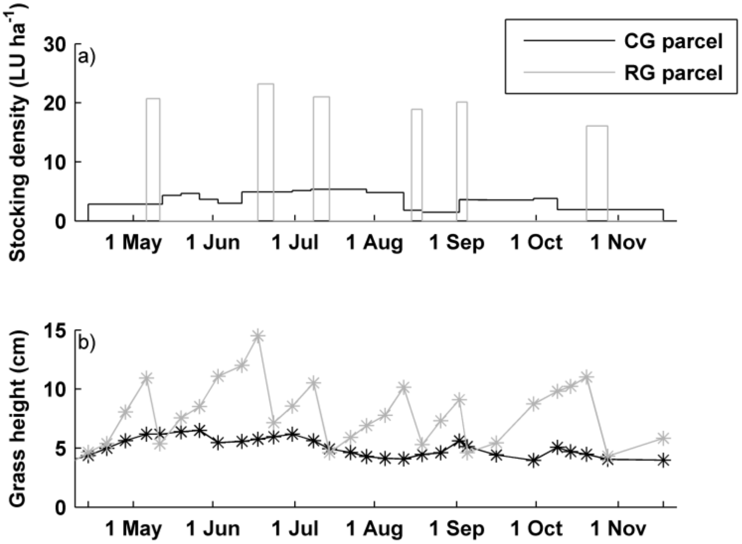

Figure 5-2: Cattle stocking density (a) and herbage height (b) throughout the grazing season in the CG and RG parcels. A stocking density of zero designates rest periods.

Throughout the paper, all variables labelled "RG" concern the rotational grazing treatment and all variables labelled "CG" concern the continuous grazing. Differences between the two treatments are always calculated as RG–CG and labelled using the symbol "Δ". The reference unit used for calculating LU is the grazing equivalent of one 600 kg liveweight (LW) adult dairy cow producing 3000 kg of milk annually, without additional concentrated feed (Eurostat, 2013). Breeding bulls and suckler cows correspond to 1 LU, and heifers and calves to 0.6 and 0.4 LU, respectively.

2.2. Instruments and setup

2.2.1. CO_2 flux measurements

The CO_2 fluxes were measured simultaneously on both fields with two eddy covariance setups each using a three-dimensional sonic anemometer (CSAT3, Campbell Scientific Ltd, UK) coupled with a closed-path CO_2/H_2O gas analyser IRGA (LI-7000, LI-COR Inc., Lincoln, NE, USA). On the CG parcel, the system was installed at 2.6 m height on a mast in the middle of the field. Air was pumped into the analyser through a polyurethane tube (6.45 m long; 4 mm inner diameter) by a pump (NO22 AN18, KNF Neuberger, D) at a 12 l min^{-1}. A more detailed description of the CG set up can be found in (Gourlez de la Motte et al., 2016). The system was identical for the RG parcel and was installed at 1.92 m height on a mast on the border of the parcel. This disposition and height was chosen in order to optimise the footprint under south-west wind direction (Figure 1).

2.2.2. Ancillary measurements

Meteorological sensors were installed on the CG mast and are described in Gourlez de la Motte et al. (2016). Measurements included air temperature and relative humidity (RHT2nl02, Delta-T Devices Ltd, Cambridge, UK), soil temperature and soil moisture (ThetaProbe, Delta-T Devices Ltd, Cambridge, UK), global and net radiation (CNR4, Kipp & Zonen, Delft, The Netherlands), rainfall (tipping bucket rain gauge, 52203, R.M. Young Company, Michigan, USA) and atmospheric pressure (144S BARO, SensorTechnics, Puchheim, Germany).

The herbage height was measured with a rising plate meter of 0.25 m^2 at 60 equidistant points in each field. Measurements on the field were taken once a week during the grazing season in the CG parcel and just before and after cattle confinements in the RG parcel. Previously (Gourlez de la Motte et al., 2016), an allometric relationship was established for the site to convert herbage height to herbage mass (HM). To establish this, direct samples were taken from the field underneath secured enclosures. Then, the relationship between grass height and harvest dry matter (DM) was computed. Samples were clipped from within0.5×0.5m quadrats. DM was obtained by drying the samples at 60°C using a forced air-oven. Biomass carbon content ($C_{content}$) was measured from laboratory measurements using the Dumas method (Dumas, 1831). The analyses were conducted by the Forest and Ecophysiology unit at the Institut National de la Recherche Agronommique (INRA).

Three secured enclosures were also used to obtain grass growth (HM_{gr}) under grazing for the CG treatment.

Cattle C intake through biomass consumption was deduced from biomass measurements for a given period using:

$$C_{intake} = C_{content}\left(HM_{beg} - HM_{end} + HM_{gr}\right) \tag{5.1}$$

where HM_{beg} and HM_{end} are the herbage mass at the beginning and at the end of the period.

2.3. *Eddy flux computation and data processing*

Half hourly CO_2 fluxes were computed following the procedure defined by the EUROFLUX-CARBOEUROFLUX-CarboEurope IP networks (Aubinet et al., 2000, 2012) and were fully described in Gourlez de la Motte et al. (2016). Briefly, CO_2 fluxes were calculated as the sum of the turbulent flux and of the storage term (Foken et al., 2012) using EDDYSOFT software package (EDDY Software, Jena, Germany, Kolle and Rebmann (2007)). A double rotation was applied to wind velocity (Rebmann et al., 2012). Fluxes were corrected for high frequency loss on both masts following the procedure proposed by Mamadou et al., (2016). They were later filtered using a stationarity criterion according to Foken et al. (2012b) and low friction velocity (u*) (Aubinet et al., 2012a). The u* threshold value was 0.13 ms^{-1} for the CG set up and 0.10 ms^{-1} for the RG set up. These thresholds were determined at the u* value where the relationship between u* and bin averaged temperature nighttime NEE flattens.

The complete CG dataset from 14th April to 17th November consists of 10608 30-min flux measurements. After filtering, the data consisted of 5276 30-min flux measurements corresponding to a data coverage of around 50%. Because of the limited RG parcel size, some fluxes had to be discarded when the parcel contribution to the footprint was not sufficient. To do that, we used the footprint evaluation tool proposed by Neftel et al. (2008). This tool calculates the contribution of a delimited surface (φ, in %) to the flux footprint relying on an analytical model (Kormann and Meixner, 2001) for the footprint function evaluation. Cumulative footprint contributions for the whole grazing season are illustrated in Figure 1. The fluxes within the RG data set were automatically discarded when the contribution of the parcel to the footprint was less than 65%. As a result, fluxes measured under north-east wind conditions were automatically discarded. We tried, if possible, to confine the cattle when the parcel was within the measurement footprint. Confinements were advanced or delayed only when weather forecasts indicated a favorable wind direction change within a few days. Otherwise, confinements were done regardless of wind direction. After filtering, the RG data consisted of 3490 30-min fluxes corresponding to a data coverage of 33%.

Missing NEE data were filled following Reichstein et al. (2005). This algorithm fills the gaps using time-moving look up tables with data from time periods with similar environmental conditions. We adapted those look up tables so that data gaps occurring during confinements were not filled using rest periods data and vice versa. Filtering the data with too low footprint contribution and adding this condition should ensure that grass height is relatively steady during the time window used to fill the data in order to limit possible biases (Merbold et al., 2014).

2.4. Instruments validation before the experiment

In order to make sure that both eddy covariance systems measured fluxes identically, an instrument validation was carried out before the start of the experiment during 11-17th June 2014. To do so, both eddy covariance systems were placed next to each other in the CG parcel at the same heights (2.62m). All the needed corrections described above were made and a regression between fluxes measured by both systems was computed. The slope of the regression was not significantly different than one (R2=0.97, no intercept) indicating that both systems effectively measured identical fluxes.

2.5. Regression and data analysis

In order to remove the influence of the most important meteorological variables controlling NEE (radiation and temperature), a function describing NEE response to those variables was fitted on seven days times series and relevant physiological parameters were deduced from these. The objective was to assess how the variation of those parameters was affected by the grazing method. To do so, both data sets were divided into grazing and rest periods according to the RG treatment’s grazing schedule so that a grazing period corresponds to a period when both parcels were grazed, while rest periods correspond to periods when only the CG parcel was grazed. A total of six grazing periods and seven rest periods were identified. Each of the rest periods were divided into seven day windows and a daytime NEE light response curve was fitted for each window. Grazing periods were not divided as their duration was mostly less than seven days. We used a modified *Michaelis Menten* light response curve (Falge et al., 2001; Lasslop et al., 2010) including temperature sensitivity to respiration (Lloyd and Taylor, 1994; Reichstein et al., 2005):

$$\mathrm{NEE_{day}} = -\frac{\alpha \times \mathrm{PPFD} \times \mathrm{G}_{\mathrm{PPFD_{ref}}}}{\alpha \times \mathrm{PPFD} + \mathrm{G}_{\mathrm{PPFD_{ref}}}\left(1 - \frac{\mathrm{PPFD}}{\mathrm{PPFD_{ref}}}\right)} + \mathrm{Rd_{10}} \times \exp\left\{ \mathrm{E_0}\left(\frac{1}{\mathrm{T_{ref}} - \mathrm{T_0}} - \frac{1}{\mathrm{T_s} - \mathrm{T_0}}\right)\right\} \tag{5.2}$$

where G_{ref} is the gross primary productivity at a reference photon flux density ($PPFD_{ref}$). $PPFD_{ref}$ was fixed at 1500 µmol m^{-2} s^{-1} and $G_{PPFDref}$ was therefore named G_{1500} throughout the paper. The traditional *Michaelis Menten* equation was modified in order to obtain G_{1500} instead of gross primary productivity at light saturation because light saturation was not reached at the end of the season. Rd_{10} (µmol m^{-2} s^{-1}) is the dark respiration normalised at reference temperature (T_{ref}) set at 10 °C. The other parameters are α, the quantum light efficiency (µmol CO_2 $µmol^{-1}$ photons), T_0 which was set at -46.02°C (Reichstein et al., 2005) and the respiration sensitivity to temperature E_0. T_s (°C) is the averaged soil temperature at 2 cm for the time window. A fixed long term E_0 value deduced from the annual response of nighttime u*-filtered NEE to soil temperature was used for each regression. The standard errors (ε) of the coefficients were also computed.

In order to compare the regression coefficients, normalised differences (u_{obs}) between two parameters (c) of the same time window were computed as follows:

$$u_{obs} = \frac{c_{RG} - c_{CG}}{\sqrt{\varepsilon_{RG}^2 + \varepsilon_{CG}^2}} \tag{5.3}$$

Differences between two coefficients were considered significant ($\alpha = 0.05$) when $|u_{obs}| > 1.96$.

2.6. Cattle respiration

2.6.1. Estimation of cattle respiration from eddy covariance fluxes

The net ecosystem exchange (NEE) measured by eddy covariance is the sum of cow respiration (R_{cows}) and soil and vegetation net exchange (Felber et al., 2016). The procedure used to estimate R_{cows} is described in Figure 3. First, we selected valid nighttime fluxes in the RG data set. Then, the data set was divided into periods with cows in the field (total ecosystem respiration, TER) and periods without cows (ecosystem respiration, ER) according to the grazing schedule. Then, as ER is sensitive to soil temperature, a two parameter exponential equation (Lloyd and Taylor, 1994) was fitted on the ER data set (see Equation 1) and a modelled ecosystem respiration (ER_m) was computed using this equation. As ER_m is representative of the average respiration response to soil temperature without cows, the average R_{cows} can be computed as:

$$\sum(\mathrm{TER}-\mathrm{ER}_m)/n_{obs}=\mathrm{R}_{cows} \tag{5.4}$$

where n_{obs} is the number of valid TER observations. Then the average estimated respiration for one livestock unit (E_{cow}) was calculated as

$$\hat{E}_{cow}=\frac{R_{cows}A}{n_{LU}} \tag{5.5}$$

where n_{LU} is the average number of livestock units in the field and A the surface of the field.

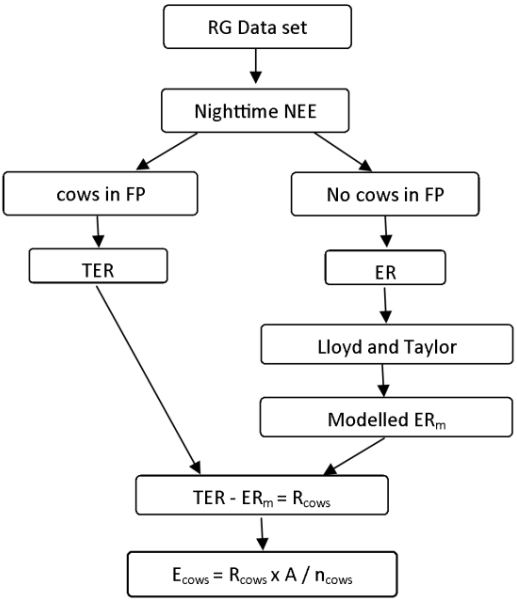

Figure 5-3: Flowchart of the cattle respiration calculation.

2.6.2. Estimation of cattle respiration from ingested biomass

Cattle respiration was also estimated from ingested biomass by assuming that only a fraction of the ingested C is re-emitted in the form of CO_2 as described by Gourlez de la Motte et al., (2016). During a grazing event, cattle respiration was estimated as follows:

$$E_{cow}=\frac{((\mathrm{OMD}\times C_{intake})-F_{CH4-C})}{\hat{n}_{LU}} \tag{5.6}$$

where OMD (%) is the digestible organic matter and C_{intake} the ingested C during grazing. OMD was obtained from near infrared reflectance spectrometry analysis

(Decruyenaere et al., 2009) of samples taken *in situ*. $F_{CH4\text{-}C}$ was estimated as a fraction of ingested DM using a constant methane emission factor fixed at 6% of DM intake (Lassey, 2007).

3. Results and discussion

3.1. Grazing method impact on carbon dioxide flux dynamics

Daily averaged NEE_{CG} and NEE_{RG} showed different patterns during the grazing season. Daily averaged NEE_{CG} showed mostly net CO_2 uptakes from the start of the grazing season until late June and then shifted to mostly net CO_2 emissions for the rest of the year (Figure 4a). This early shift was previously observed at the same site by Gourlez de la Motte et al., (2016) and was attributed to grazing that limits gross primary productivity by limiting photosynthetic capacity. In contrast, NEE_{RG} showed different dynamics (Figure 4b). Considerable CO_2 emission peaks were observed during grazing events predominantly because of cattle respiration followed by a progressive shift towards CO_2 uptake during rest periods. Prolonged CO_2 uptake events were observed in August and October at the end of the rest periods. This led to more pronounced CO_2 emissions on the RG treatment during cattle confinement compared to the CG treatment, and more pronounced CO_2 uptake after several days of recovery (Figure 4c). Similar switches from a source to a sink were previously observed after grazing or cutting because of rapid changes in standing biomass (Nieveen et al., 2005; Peichl et al., 2012; Rogiers et al., 2008; Rutledge et al., 2015; Wohlfahrt et al., 2008). It is noted that a long gap between the 5th and 6th confinements could not be filled because of prolonged north-west wind direction conditions.

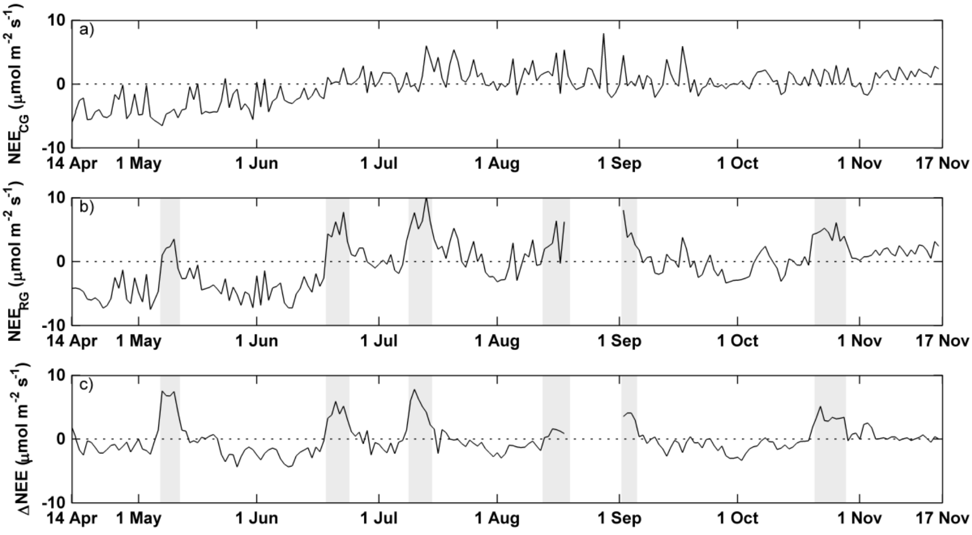

Figure 5-4: Daily means of (a) net ecosystem exchange of the CG parcel (NEE$_{CG}$), (b) net ecosystem exchange of the RG parcel (NEE$_{RG}$) and (c) differences between NEE$_{RG}$ and NEE$_{CG}$ (ΔNEE = NEE$_{RG}$–NEE$_{CG}$). Confinement periods on the RG parcel are colored in grey.

Herbage heights in RG were similar at the beginning of each rest period. However, the height was mostly stable in the CG parcel due to continuous grazing, while grass grew quickly in the RG parcel during rest periods (Figure 2). These differences in standing biomass caused by the grazing method could have impacted gross primary productivity as well as the total ecosystem respiration and therefore NEE dynamics. In order to identify which processes were responsible for the observed differences in NEE dynamics a regression analysis was carried out to compute G$_{1500}$, the gross primary productivity at high radiation, and Rd$_{10}$, the dark respiration normalised at 10 °C.

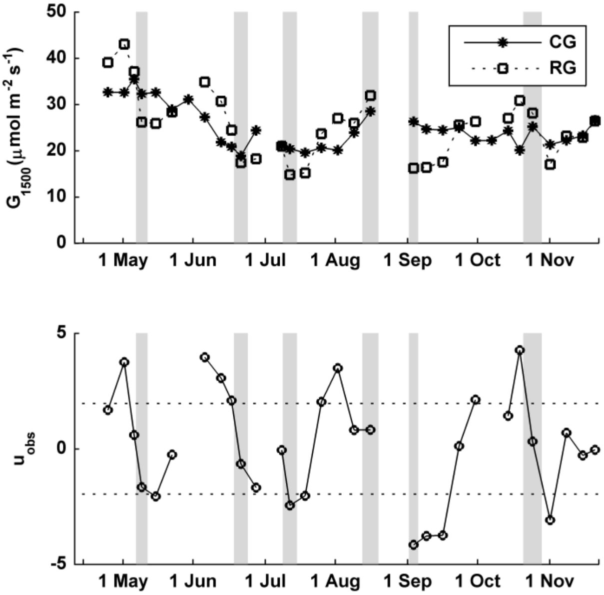

Figure 5-5: Evolution of (a) gross primary productivity at high radiation (G_{1500}) and (b) normalised differences between the two coefficients (u_{obs}). Confinement periods on the RG parcel are coloured grey. Horizontal dashed lines correspond to the 95% level of confidence (±1.96).

During each rest period, notable differences in G_{1500} dynamics could be observed (Figure 5). At the beginning of each rest period, just after the cattle confinement, G_{1500} was lower on the RG parcel. This difference progressively shifted towards a higher G_{1500} at the end of the rest period. This behaviour was less visible for the last rest period at the end of the growing season when grazing intensity was very low on the CG parcel because of low biomass production. As a result, ΔG_{1500} was significantly correlated (p value < 0.05) to the difference of herbage height between the two parcels (confinement periods excluded, Figure 6). This correlation illustrates the influence of grass height on gross primary productivity (GPP) and the plant's photosynthetic capacity. The impact of fast changes in vegetation heights due to rotational grazing on gross primary productivity at high radiation was also observed by Felber et al., (2016b) using a similar approach and by Campbell et al., (2015) using an automated phytomass index analysis (Lohila et al., 2004).

It is also notable that G_{1500} was systematically lower (less assimilation) on the RG parcel following the confinements even when grass heights were similar on both parcels. This may be due to a reduced regrowth rate after intensive grazing. Indeed, following intensive grazing, the ratio of leaf area per plant weight is reduced thereby limiting its regrowth rate (Oesterheld and McNaughton, 1991).

No similar impact of grazing on Rd_{10} dynamics was observed. No significant correlation (p value > 0.05) was found between ΔRd_{10} and the difference of herbage height between the two parcels (Figure 6). These results are in agreement with another investigation made at the same site (Jérôme et al., 2014) that found a decrease in gross primary productivity at light saturation during grazing periods and an increase during rest periods, but no impact of grazing intensity on normalised respiration at 10 °C, probably due to opposing effects of grazing on the total ecosystem respiration. Therefore, in our study, the observed switch from a CO_2 source to a CO_2 sink after a grazing event on the RG parcel was more likely to be due to changes in photosynthetic capacity rather than processes influencing the total ecosystem. Other studies have also shown that changes in NEE after grazing or cutting were more driven by changes in GPP rather than changes in ER (Rogiers et al., 2008; Wohlfahrt et al., 2008).

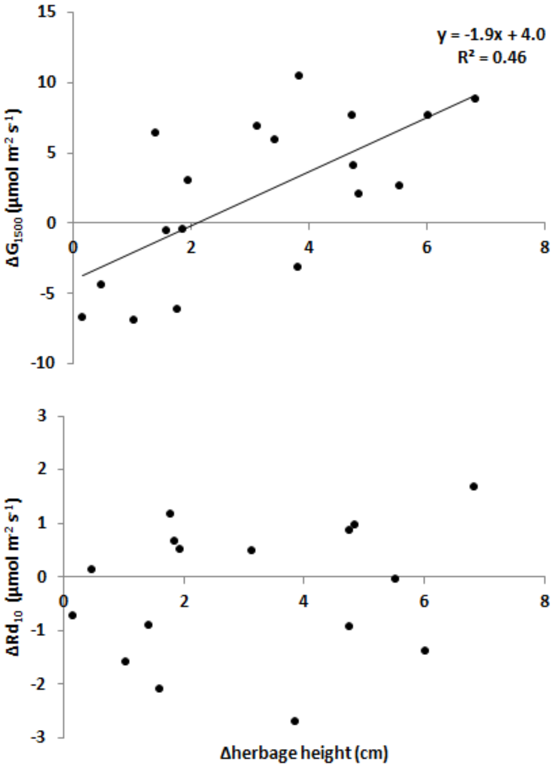

Figure 5-6: Relationship between differences in herbage height and (a) differences in dark respiration normalised at 10 °C (ΔRd_{10}) and (b) differences in gross primary productivity at high radiation (ΔG_{1500}).

During confinements, NEE_{RG} was found to be greatly affected by cattle respiration. Indeed, cattle respiration was estimated at 3.0 ± 0.8 kg C LU^{-1} d^{-1} (see section 3.3). The average stocking density during cattle confinement was 19.3 LU ha^{-1} while it was 3.5 LU ha^{-1} on average in the CG parcel. Therefore, this difference of stocking densities should have led to a contribution of 4.5 ± 1.2 µmol m^{-2} s^{-1} to ΔNEE on average. Nighttime NEE_{RG} was 3.4 µmol m^{-2} s^{-1} higher on average than NEE_{CG} during confinement, which is within the error bound of the estimated cattle respiration. Therefore, it is more likely that differences in total ecosystem respiration during confinements were mostly due to cattle respiration (Felber et al., 2016b; Jérôme et al., 2014). The higher difference observed during the daytime (5.7 µmol m^{-2} s^{-1}) can be explained by the gross primary productivity reduction in the RG parcel because of defoliation during confinement.

Cattle respiration could also have had an impact on measured NEE_{CG} dynamics. However, on short term measurements (daily to monthly), the contribution of moving emissions spots like cattle are highly uncertain and variable because of uneven spatial and temporal cattle distribution within the footprint (Dumortier et al., 2017; Felber et al., 2016b). This probably explains why no clear impact of cattle respiration could be observed on short term NEE_{CG} dynamics.

3.2. Biomass production and consumption

A total production of 6270 kg DM ha^{-1} and from 6470 to 7420 kg DM ha^{-1} were estimated on the CG parcel and the RG parcel respectively leading to a rather small difference between the two treatments. For the RG treatment, the lower value is obtained by considering zero growth during confinements while the higher value is obtained by using the same grass growth as the CG treatment (around 950 kg DM ha^{-1}). Considering a zero growth might underestimate the annual grass production regarding the length of those events (a total 36 days). However, assuming identical growth rate is also unlikely as growth should have been highly constrained once trampled and grazed. We note that similar forage production between rotational and continuous grazing has previously been observed (Briske et al., 2008; Popp et al., 1997) but under very different climatic conditions and farm management than in our study.

3.3. Estimation of cattle respiration

Cattle respiration was estimated from eddy covariance measurements following the procedure described in section 2.4.1. A R_{10} value of 5.6 µmol m^{-2} s^{-1} and a E_0 value of 238.4 K were obtained from the fit of the exponential equation on nighttime fluxes without cows. R_{cows} on the RG parcel was 6.1 ± 1.6 µmol m^{-2} s^{-1} leading to an E_{cow} value of 3.0 ± 0.8 kg C LU^{-1} d^{-1}. Values are presented with their 95% confidence intervals. Cattle respiration was also independently estimated from ingested biomass. A value of 2.5 kg C LU^{-1} d^{-1} was estimated which is within the uncertainty of the estimated value using eddy covariance.

The cattle respiration value estimated from eddy covariance was not significantly different from the value of 2.59 ± 0.58 kg C LU^{-1} d^{-1} obtained from eddy covariance at the same site and from the value obtained from ingested biomass by Jérôme et al. (2014). Jérôme et al. (2014) also used confinement experiments to estimate cattle respiration but followed a different methodology. In that experiment, cattle respiration was estimated by calculating the average difference between fluxes just before and after the confinement experiment under similar environmental conditions. A different method was proposed in this study because the confinement periods were longer (average 6 days vs. 1 day) and the changes in standing biomass were very different leading to non-similar conditions before and after confinement. Other limitations of confinement experiments to estimate cattle respiration were widely discussed by Jérôme et al. (2014). In a similar way, Felber et al. (2016b) estimated a cattle respiration value of 4.6 kg C head^{-1} d^{-1} for dairy cows using either a precise grazing schedule or animal positioning system and eddy covariance.

The total contribution of cattle respiration to NEE (R_{cows}) could also be estimated by upscaling E_{cow} to the entire year using the grazing schedule for both parcels. The total R_{cows} was 230 ± 61 g C m^{-2} yr^{-1} for the CG parcel and 208 ± 55 g C m^{-2} yr^{-1} for the RG. The difference of contribution of cattle respiration to ΔNEE is therefore around 22 g C m^{-2} yr^{-1}. This scaling up assumes spatially homogenous cattle distribution over time so that their respiration signal becomes a constant part of the eddy covariance measurements signal. This hypothesis is more likely to be met for the RG treatment as fluxes are discarded when the measurement footprint is outside the parcel increasing the probability that the herd is in the system footprint (Jérôme et al., 2014). For the CG parcel, this hypothesis is less likely to be met (Felber et al., 2016b) as herds can or cannot contribute to the CO_2 flux depending on wind direction and herd position in the field. However, as suggested by Dumortier et al.(2017) for methane flux measurements at DTO, this hypothesis is more likely to be reached when integrating fluxes over long periods.

3.4. Impact of grazing method on cumulative net ecosystem exchange

In order to assess the impact of the grazing method on the cumulative NEE, the data sets were divided into seven periods. The first period started at the beginning of the grazing season and ended at the beginning of the first confinement (Table 1, Figure 7). Then, each rotation corresponds to the cattle confinement and its resting period until the start of the next rotation. For each rotation cumulative NEE_{RG} increased during confinement leading to a positive difference between cumulative NEE_{RG} and NEE_{CG} (ΔNEE). Then, cumulative ΔNEE stagnated for a few days and eventually started to decrease if the rest period was long enough. After the last confinement, cumulative ΔNEE stagnated because of very limited photosynthetic activity at the end of the grazing season. Cumulative ΔNEE could therefore be negative ($NEE_{RG} < NEE_{CG}$) or positive depending on the length of the rest period and when it occurred in the grazing season. ΔNEE ranged from -30 to 41 g C m^{-2} (Table 1). It was negative for the two rotations with the longest rest periods. For these two rotations, neutrality ($NEE_{RG} = NEE_{CG}$) was obtained after a recovery period of 31 and 27 days. We also noticed that the 4th rotation's recovery period lasted for 28 days leading to a budget close to neutrality (ΔNEE = +8 g C m^{-2}). Although these observations lack replicates, we can argue that the time needed to reach neutrality should be around four weeks depending on weather conditions and stocking densities. It is also important to note that rest period fluxes were similar at the end of the season when grass growth was practically zero.

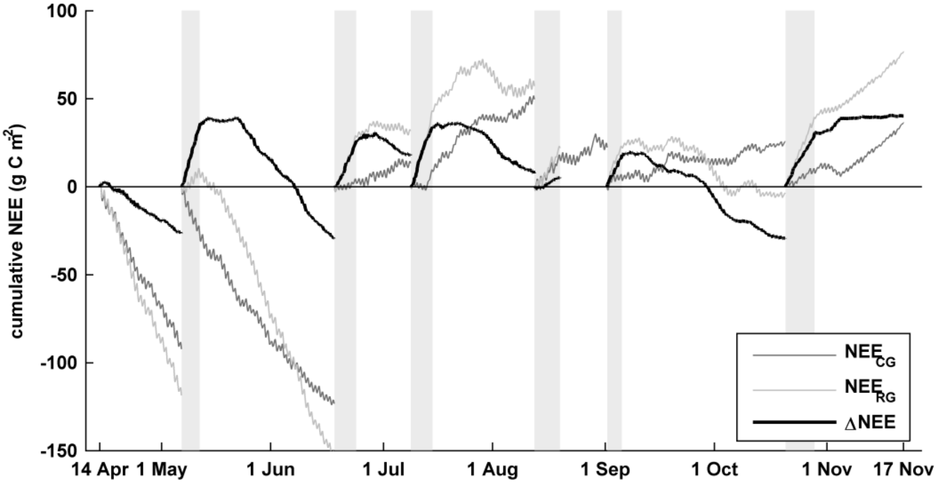

Figure 5-7: Evolution of cumulative NEE and ΔNEE (ΔNEE = NEE_{RG}–NEE_{CG}). The data set is divided into 7 periods indicated by vertical dashed lines. Confinement periods on the RG parcel are coloured grey.

When accounting only for periods that could be completely filled, NEE_{RG} was –88 g C m^{-2} and NEE_{CG} was –74 g C m^{-2} leading to a ΔNEE of –14 g C m^{-2}. Accounting for the difference in cattle respiration due to a difference in stocking densities shifts NEE_{RG} to –66 g C m^{-2} which leads to a cumulative ΔNEE of +8 g C m^{-2}. At DTO, for the CG parcel, an average uncertainty for the annual cumulative NEE of +26 (upper range) and –17 (lower range) g C m^{-2} yr^{-1} were estimated by Gourlez de la Motte (2016). When considering both the lower data coverage and the fast changes in standing biomass in the RG treatment, we can presume that the uncertainty in RG treatment is even greater. Therefore, this very small difference in NEE is not likely to be significant. It may also be noted that no significant difference in terms of annual productivity and ingested biomass was observed between the two treatments. This leads to the conclusion that, in our study, no significant difference in total NEE could be observed between the two grazing methods assuming similar stocking rates. Similar conclusions have been observed using direct soil organic carbon measurements comparing rotationally grazed and continuously grazed grasslands with similar management for at least a decade (Chan et al., 2010; Cowie et al., 2013; Sanderman et al., 2015). It is more likely that grassland carbon budgets depend more on the stocking and fertilisation rates than the grazing method (Allard et al., 2007; Klumpp et al., 2011; Soussana and Lemaire, 2013).

Table 5-1: Starting and ending dates (in year 2015), cumulative net ecosystem exchange for the continuous grazing (NEE$_{CG}$), rotational (NEE$_{RG}$) grazing treatments, difference in net ecosystem exchange between those treatments (ΔNEE= NEE$_{RG}$ – NEE$_{CG}$), stocking densities and grazing durations for each period. The first period starts at the beginning of the grazing season (14th April 2015) and ends at the beginning of the first confinement. Next periods correspond each to a confinement followed by its restoration period. The 5th period marked with * is incomplete because of too low data coverage.

			Continuous grazing		Rotational grazing				
Period n°	Dates	Duration	Stocking density	NEE$_{CG}$ (gCm^{-}	Confinement (days)	Rest periods	Stocking density	NEE (gCm^{-2})	ΔNEE (gCm^{-2})
1	14/04-6/05	23	2,8	-92	--	23	0	-119	-27
2	6/05-17/06	42	3,6	-123	5	37	20,7	-153	-30
3	17/06-8/07	21	4,5	13	6	15	23,3	32	19
4	8/07-11/08	34	5	49	6	28	21,1	57	8
5*	11/08-31/08	20	3,9	17	7	13	18,9	22	5
6	31/08-19/10	49	3,2	26	4	45	20,1	-4	-30
7	19/10-20/11	32	1,9	36	8	24	16,1	77	41
Total/average*		221	3.5*	-74	36	185	19.3*	-88	-14

The absence of significant difference between the two treatments assumes no inherent variability in terms of annual NEE between the two parcels. Although this hypothesis is widely used by other studies using paired eddy covariance measurements to study management impact on CO$_2$ fluxes (eg. Allard et al., 2007; Cowan et al., 2016; Klumpp et al., 2011; Skinner, 2013), Rutledge et al.,(2017a) found that this strong assumption was not always met. Indeed, by measuring C fluxes in each block during one complete year before the experiment, they found significant differences between the blocks that could not easily be attributed to large pre-treatment differences in term of management, soil types and site history. In this experiment, we tried to limit those possible biases as much as possible by choosing two adjacent pastures with very similar soil, site history and management.

3.5. Conclusion

To our knowledge, this study is the first to compare the impact of rotational and continuous grazing in terms of CO$_2$ flux dynamics and C budget measured by eddy covariance. It was carried out in an intensively managed pasture grazed by Belgian Blue suckler cows located in southern Belgium. The results showed that despite CO$_2$ fluxes showing very different dynamics between the two grazing management systems, overall NEE sums were very similar. Although no significant differences in term of cumulative NEE was observed, it is important to emphasise that this result is highly dependent on the stocking rates and the length of the rest periods. Shorter rest periods (with similar stocking densities) on the RG treatment could have led to an overall reduced photosynthetic capacity of the pasture, thereby emphasising the need to maintain a suitable stocking rate. The strong link between gross primary productivity at high radiation and herbage height also highlights the strong need to account for continuous biomass changes when modelling or studying the relationship of NEE to other environmental variables (Campbell et al., 2015; Lohila et al., 2004).

3.6. Acknowledgments

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Chapter 6

Herd position habits can bias net CO_2 ecosystem exchange estimates in free range grazed pastures.

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Abstract

The eddy covariance (EC) technique has been widely used to quantify the net CO_2 ecosystem exchange (NEE) of grasslands, which is an important component of grassland carbon and greenhouse gas budgets. In free range grazed pastures, NEE estimations are supposed to also include cattle respiration. However, cattle respiration measurement by an EC system is challenging as animals act as moving points emitting CO_2 that are more or less captured by the EC tower depending on their presence in the footprint. Often it is supposed that, over the long term, cattle distribution in the pasture is homogeneous so that fluctuations due to moving sources are averaged and NEE estimates are reasonably representative of cattle respiration.

In this study, we test this hypothesis by comparing daily cow respiration rate per livestock unit (LU) estimated by postulating a homogeneous cow repartition over the whole pasture with three other estimates based on animal localization data, animal scale carbon budget and confinement experiments.

We applied these methods to an intensively managed free range grassland and showed that the NEE estimate based on a homogeneous cow repartition was systematically lower than the three other estimates. The bias was about 60 g C m^{-2} yr^{-1}, which corresponded to around 40% of the annual NEE. The sign and the importance of this bias is site specific, as it depends on cow location habits in relation to the footprint of the EC measurements which highlight the importance of testing the hypothesis of homogeneity of cattle distribution on each site.

Consequently, in order to allow estimating the validity of this hypothesis but also to improve inter site comparisons, we advocate to compute separately pasture NEE and grazer's respiration. For the former we propose a method based on cattle presence detection using CH_4 fluxes, elimination of data with cattle and gap filling on the basis of data without cattle. For the second we present and discuss three independent methods (animal localization with GPS, animal scale carbon budget, confinement experiments) to estimate the cattle respiration rate.

1. Introduction

Grasslands cover around 40% of Earth's land area (Steinfeld et al., 2006) and are therefore one of the most important ecosystems on earth. More specifically, pasturelands are dedicated to the production of forage for harvest by grazing, cutting, or both. These lands constitute important carbon (C) stocks estimated at 343 Pg C, which is nearly 50% more than the carbon stored in worldwide forest soils (Conant et al., 2017). They can therefore act as important carbon sinks that can play an important role in mitigating livestock production-related GHG emissions (Hörtnagl et al., 2018; Soussana et al., 2007). There is therefore a strong need to accurately quantify grassland C sequestration.

The most used technique to quantify CO_2 exchanges between grasslands and the atmosphere is the Eddy Covariance (EC) technique (Aubinet et al., 2012b). In addition, by combining net CO_2 ecosystem exchanges (NEE) obtained with this technique with other non-CO_2 carbon export and import measurements, a complete ecosystem carbon budget (net biome productivity, NBP) can be obtained (Soussana et al., 2007). Studies measuring NBP showed that pastures could act as important C sinks that could at least partially offset the CH_4 and N_2O emitted in the pasture, depending on management and pedoclimatic conditions. Study sites were either grazed (Allard et al., 2007; Felber et al., 2016a; Gourlez de la Motte et al., 2016; Klumpp et al., 2011; Nieveen et al., 2005; Rutledge et al., 2015, 2017b, 2017a; Wayne Polley et al., 2008), mown (Ammann et al., 2007; Merbold et al., 2014; Wohlfahrt et al., 2008), or both (Jones et al., 2017; Mudge et al., 2011; Skinner, 2008; Skinner and Dell, 2015; Zeeman et al., 2010).

Flux measurements over grazed pastures are especially challenging. In the presence of cattle, the total net ecosystem exchange (NEE_{tot}) of a pasture can be partitioned between the net ecosystem exchange without grazing animals (NEE_{past}) and the total respiration of the animals on the field (R_{cows}) (Felber et al., 2016b):

$$\mathrm{NEE_{tot}} = \mathrm{NEE_{past}} + \mathrm{R_{cows}} \tag{6.1}$$

which can further be combined with other C exports and C imports to obtain the NBP of a pasture :

$$\mathrm{NBP} = \mathrm{NEE_{tot}} - \mathrm{C_{exports}} + \mathrm{C_{imports}} \tag{6.2}$$

However, as cattle act as moving CO_2 sources their emissions either will or won't be captured by the measuring system, depending on the presence of the cattle in the footprint area. Although R_{cows} is a small flux compared to gross primary productivity (GPP) and the total ecosystem respiration (TER), it can be of the same order of magnitude as NEE_{tot}. Even if its magnitude may vary from site to site, R_{cows} around 200 g C m^{-2} yr^{-1} may be expected in pastures with a high stocking rate (Jérôme et al., 2014). Therefore, an under- or overestimation of this flux could lead to a non-negligible systematic bias in annual NEE_{tot} values and therefore in annual NBP.

Historically, most of the studies on grazed sites assumed (explicitly or not) that, averaged over a grazing season, cattle were spread evenly over the field so that their respiration signals become a part of NEE_{tot} and are correctly estimated by EC. Although most often not verified, this hypothesis was commonly (sometimes implicitly) used for free range grazed pastures where the presence or not of cattle within the footprint at a given time is not easy to assess (Byrne et al., 2007; Gourlez de la Motte et al., 2016; Jaksic et al., 2006; Klumpp et al., 2011; Zeeman et al., 2010).

When the pasture is divided into several paddocks for rotational grazing this hypothesis is not met, but the presence of cattle in the footprint is much easier to assess so that the computation of NEE_{past} is possible by filtering fluxes affected by cattle respiration. In an intensively rotationally grazed site with multiple paddocks, Skinner (2008) advocated that fluxes affected by cattle respiration should be removed as CO_2 fluxes were very erratic in the presence of a high stocking density within the footprint. He proposed to filter out the fluxes from paddocks affected by cattle respiration, compute NEE_{past}, and account for the biomass ingested by the animals as C exports and the animal excretions as C imports, thereby considering cattle to be external to the system. More recently, several studies also identified this problem and adapted their methodology to exclude grazer respiration and thus, compute NEE_{past} (Felber et al., 2016a; Hunt et al., 2016; Rutledge et al., 2017a, 2017b). Kirschbaum et al., (2015) also highlighted the need to filter fluxes in the presence of high stocking density in the footprint in order to obtain good agreement between modelled and measured CO_2 fluxes in a rotationally grazed pasture.

Alternatively, Felber et al. (2016b) used GPS trackers on cows in combination with a footprint model to separate fluxes with and without cattle respiration. Animal positions were then used to estimate a reference respiration rate per animal. In order to verify the hypothesis that NEE_{tot} includes R_{cows} in a representative way, they compared this respiration rate value to the respiration rate calculated considering a homogeneous cattle distribution on the pasture. For their site, a rotationally grazed multi-paddock pasture, they found that on a yearly basis animal respiration was included in NEE_{tot} in a representative way suggesting that there were no correlations between the animal positions and the wind direction. However, this result is site specific and such observations has yet to be verified for continuously grazed pastures (Felber et al., 2016b). In those sites the animals are allowed to move freely in the pasture so that, if cattle are more likely to remain grouped in specific areas of the pasture such as shade areas or near water/feed supplies, which is very probable, NEE_{tot} would be biased in a way and to an extent that depends on the position of these specific areas relative to the footprint.

The aim of the present study is to test different methods to verify if the contribution of grazing animal respiration is adequately represented in the NEE measured in a continuously grazed pasture. The methods were applied at the Dorinne Terrestrial Observatory (DTO), an intensively managed pasture with a high annual stocking rate (>2 livestock units (LU) per hectare). A solution is also proposed to correct cow respiration values if not estimated properly. Conclusions and consequences regarding the computation of the carbon budget of the pasture are also discussed. Advantages and drawbacks of the different methods proposed in the paper are discussed and more general guidelines are provided for researchers who aim to measure consistent NEE and cow respiration rates in grazed pastures.

2. Materials and methods

2.1. Site description and grassland management

The method was tested at the Dorinne Terrestrial Observatory (DTO) (50° 18’ 44’’N; 4° 58’ 07” E) in southern Belgium. The site consists of a 4.2 ha intensively managed permanent pasture grazed by Belgian Blue beef cattle with an average stocking rate of about 2.3 LU ha^{-1} yr^{-1}. Cattle are usually on the field from April to mid-November and are free to graze throughout the whole pasture at all times. The pasture is fertilized with an annual nitrogen fertilization of around 120 kg N ha^{-1} (excluding cow excreta). The main wind directions are South-West and North-East during anticyclonic weather conditions. The locations of the flux tower, water trough, hedges, feeding place, and fences are described in Figure 6-1 and have not changed since the start of the measurements in 2010. The carbon (Gourlez de la Motte et al., 2016) and the methane (Dumortier et al., 2017) budgets of the site have been presented in previous studies. The vegetation is mainly composed of ryegrass (*Lolium perenne L.*) and white clover (*Trifolium repens L.*). The site is a commercial farm with management that is, as much as possible, representative of the common practices on beef cattle farms around the region. Breeding bulls and suckler cows correspond to 1 LU, heifers and calves to 0.6 and 0.4 LU, respectively.

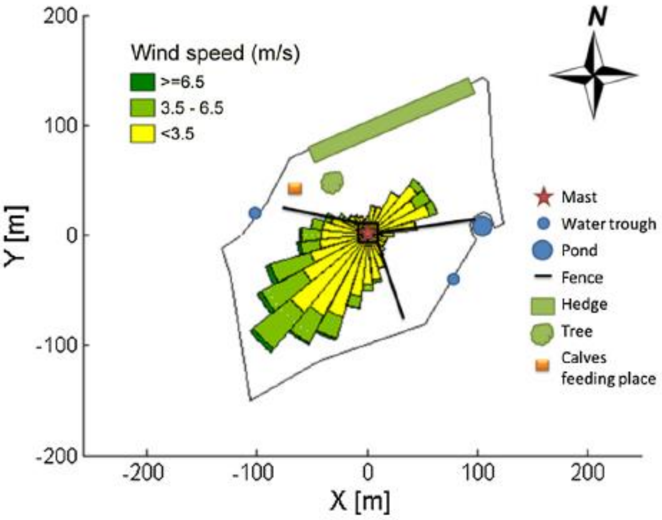

Figure 6-1: Schematic map of the site. During confinements, internal fences were closed and the cattle were confined in the south-west part of the pasture. Figure taken from Dumortier et al., 2017.

2.2. Flux measurements and processing

The CO_2 flux was measured with an eddy covariance setup using a three-dimensional sonic anemometer (CSAT3, Campbell Scientific Ltd, UK) coupled with a closed path CO2/H2O gas analyzer IRGA (LI-7000, LI-COR Inc., Lincoln, NE, USA). The system was installed at a height of 2.6 m in the middle of the field. Air was pumped into the analyzer through a polyurethane tube (6.45 m long; 4 mm inner diameter) by a pump (NO22 AN18, KNF Neuberger, D) with a flow of 12 l min^{-1}. A more detailed description of the CO_2 set up can be found in (Gourlez de la Motte et al., 2016).

The CH_4 flux was measured using the same anemometer on the same mast coupled with a fast CH_4 analyzer (PICARRO G2311-f, PICARRO Inc, USA). Air was pumped into the analyzer using a heated tube (6.85 m long, 6 mm inner diameter). A more detailed description can be found in Dumortier et al. (2017).

Half hourly CO_2 and CH_4 fluxes were computed following the standard procedure defined by the CarboEurope IP network (Aubinet et al., 2012b, 2000). CO_2 fluxes were calculated as the sum of the turbulent flux and of the storage term (Foken et al., 2012) using the EDDYSOFT software package (EDDY Software, Jena, Germany, (Kolle and Rebmann, 2007)). They were corrected for high frequency loss following the procedure proposed by Mamadou et al. (2016). They were later filtered for stationarity using a selection criteria of 30%, according to Foken et al. (2012b). CH_4 fluxes were calculated using the EddyPro® (LI-COR Inc, Lincoln, NE, USE) open source software (Version 6). A double rotation was applied to wind velocity for both fluxes (Rebmann et al., 2012). Both CO_2 and CH_4 fluxes were filtered for low turbulence using a friction velocity (u*) threshold of 0.13 m s^{-1}. This threshold was determined as the u* value where the relationship between u* and the temperature normalized nighttime CO_2 flux flattens. A more detailed description of CO_2 and CH_4 flux computation can be found in Gourlez de la Motte et al. (2016) and Dumortier et al. (2017), respectively. Note that, in this study, the requirement for the CH_4 flux quality is low as the fluxes are only used as a tool to assess the presence or absence of cows in the footprint (binary test).

2.3. Meteorological measurements

Meteorological measurements included air temperature and relative humidity (RHT2nl02, Delta-T Devices Ltd, Cambridge, UK), soil temperature and soil moisture (ThetaProbe, Delta-T Devices Ltd, Cambridge, UK), global and net radiation (CNR4, Kipp & Zonen, Delft, The Netherlands), rainfall (tipping bucket rain gauge, 52203, R.M. Young Company, Michigan, USA), and atmospheric pressure (144S BARO, SensorTechnics, Puchheim, Germany).

2.4. General description of the methodology

A methodology was developed to assess if cow respiration is included in a representative way in annual NEE_{tot} estimates and, if needed, to make the necessary corrections. The main steps of this methodology are:

First (homogeneous approach), average cattle respiration rates per LU were computed postulating a homogeneous cow repartition over the whole pasture on an annual timescale. For this, CH_4 fluxes were used as a tool to detect the presence of cattle in the footprint and filter NEE_{tot} to compute the net ecosystem exchange of the pasture without cow respiration (NEE_{past}) for extensive data sets. Both NEE_{tot} and NEE_{past} data sets were gap filled and total annual R_{cows} values were then computed by subtraction of these two estimates. The average annual cattle respiration rates per LU (E_{cow}) was then deduced by dividing R_{cows} by the average stocking density on the pasture (SD_p).

Secondly, as a tool of comparison, three reference cow respiration rates per LU were computed. The first (GPS approach) consists in localizing the animals with GPS trackers during several measurement campaigns in order to compute the stocking density in the footprint (SD_f) as proposed by Felber et al. (2015, 2016b). The second (confinements approach) consists in constraining the movement of the animals on the pasture by confining them to a small part of the field in the main wind direction and for a short period in order to compare fluxes during this period with fluxes during animal-free periods, just before and after the confinement (Gourlez de la Motte et al., 2018; Jérôme et al., 2014). The third method (animal C budget approach) consists in building a complete carbon budget at the animal scale by estimating the ingested biomass and measuring its carbon content and digestibility (Gourlez de la Motte et al., 2018, 2016).

Finally, the respiration rates obtained considering a homogenous stocking density on the field at the annual scale were compared to reference respiration rates in order to verify if animal respiration was measured in a representative way. A significantly lower value would indicate a lower than average cow presence in the footprint, while a higher value would indicate the opposite. A procedure is also proposed to correct the fluxes in case cow respiration would not be measured in a representative way.

2.5. Stocking density in the footprint and on the pasture

Both the homogeneous and the GPS approaches rely on stocking density estimates. The homogeneous approach (average stocking density, SD_p) rely on the average number of LU on the whole field (n_{avg}), which was carefully monitored by the farmer during the whole grazing season, and corrected (factor φ) to take into account the average pasture contribution to the footprint:

$$SD_p = \frac{n_{avg} \times \varphi}{A} \tag{6.3}$$,

where A is the total pasture area. The average pasture contribution to the footprint φ was computed for every half hour, using an analytical footprint model (Kormann and Meixner, 2001) designated hereafter as the KM model. This correction was necessary as, very often, the footprint area was bigger than the pasture. It supposes there are no cattle in the footprint area outside of the experimental area, which is the case in the main wind direction (SW) where the pasture is bordered by a crop field.

In the other directions, the pasture is surrounded by other pastures where some cows may be present from time to time. As a result, around 80% of the cumulated footprint is coming from the pasture and from the crop. The remaining contribution is coming from pastures that may, sporadically, be polluted by other cows. To take this into account, an uncertainty of 10% was accounted for SD_p.

The second estimate (geolocation-based stocking density, SD_f) is based on geolocation tracking. The individual contribution of each animal was estimated half-hourly using the KM model and was summed as (Felber et al., 2016b):

$$SD_f = \sum_i \sum_j n_{ij} \phi_{ij} \frac{n_{avg}}{n_{detected}} \quad (6.4),$$

where i and j represent the position of each cell on a 2D grid, n_{ij} is the number of animals in the cell ij, ϕ_{ij} is the value of the footprint function in the cell ij (m^{-2}) and $n_{detected}$ the number of LU detected for a specific half hour. For each half hour, the position of some animals was unknown (calves were not tracked and not all geolocation devices were always operational), the calculated SD_f was thus corrected in order to also include undetected or unaccounted animals. The resulting average correction factor ($\frac{n_{avg}}{n_{detected}}$) was of 1.47.

Both SD_p and SD_f depend on the model used to compute the footprint function and its associated uncertainties. The footprint model used in this study was thus carefully selected through an artificial source experiment run by (Dumortier et al., 2019) at the same site.

2.6. Homogeneous approach for E_{cow}

In the homogeneous approach (Figure 6-2), annual R_{cows} were computed using equation 6-1. For the determination of NEE_{past}, CH_4 fluxes were used as a cow detection tool, considering that CH_4 fluxes emitted by the cattle were much higher than those exchanged by the soil and the vegetation (Dumortier et al., 2017). The advantage of this CH_4 flux filtering approach is that it can be used throughout the year, even outside GPS tracking campaigns. Annual CO_2 flux data series were filtered in order to only keep data when net ecosystem exchange was unaffected by cow respiration (NEE_{past}).

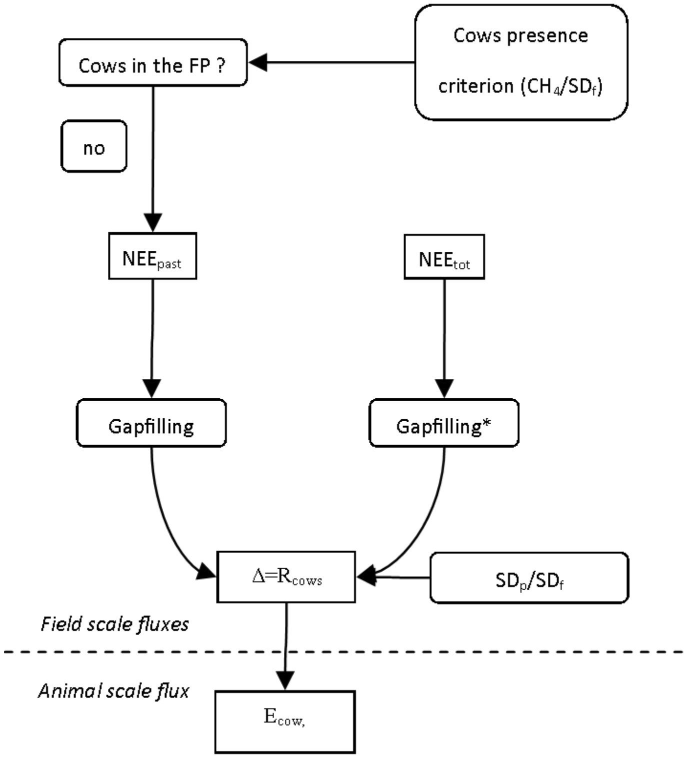

Figure 6-2: Flow chart of the procedure used to estimate cow respiration rates per livestock unit (E_{cow}) using either GPS campaigns or assuming a homogeneous cow repartition in the field (CH_4 approach). Both procedures are similar, differing in their way of assessing the presence of cows in the footprint (FP) and of assessing the stocking density (stocking density in the pasture (SD_P) for the CH_4 filtering approach, or stocking density in the footprint (SD_f) for the GPS method). Gaps in total net ecosystem exchange (NEEtot) were filled only for the CH_4 approach. Gaps in pasture net ecosystem exchange (NEE_{past}) were filled for both approaches. Figure modified after Felber et al., (2016b).

The CH_4 flux threshold used for filtering was calibrated during the GPS tracker campaigns: cows were considered to be absent when SD_f was lower than 2×10^{-5} LU m^{-2}. The CH_4 flux threshold was then fixed in order to keep a maximum of events without cows and a minimum of events with cows. The best compromise (>85% of events without; <10% of events with cows) was obtained for a value of 25 nmol CH_4 m^{-2} s^{-1}.

Missing NEE data were filled for both NEE_{past} and total NEE_{tot} data sets using the online REddyProc gap filling and flux partitioning tool (https://www.bgc-jena.mpg.de/bgi/index.php/Services/REddyProcWeb, (Reichstein et al., 2005)). This algorithm uses time-moving look up tables and finds fluxes measured in similar meteorological conditions to fill the data. Meteorological variables used by the algorithm are the air temperature (T_{air}), the vapor pressure deficit (VPD), and the global radiation (R_g). R_{cows} was then obtained by subtracting filled NEE_{tot} and

NEE_{past} data series, and average monthly/annual respiration rates per LU ($E_{cow,hom}$) were obtained by dividing this result by monthly/annual average SD_p.

The uncertainties on $E_{cow,hom}$, besides those affecting SD_p, are due to uncertainties affecting R_{cows} estimation, which itself depends on NEE_{tot} and NEE_{past} estimates during grazing periods. To be complete, the uncertainties on NEE_{tot} and NEE_{past} were computed for the whole year but were combined only during grazing periods to estimate uncertainties on R_{cows}.

Annual NEE estimates are typically affected by different sources of random and systematic errors:

1) Random errors affecting both the measured fluxes and the gap filling procedure (Dragoni et al., 2007; Richardson et al., 2006).
2) Error associated with the additional gaps in NEE_{past} due to cow presence.
3) A residual uncertainty associated with the choice of the u* threshold used to filter fluxes under low turbulence conditions (Aubinet et al., 2018).
4) A residual uncertainty associated with the choice of the cut-off frequency for the high frequency loss corrections (Gourlez de la Motte et al., 2016; Mamadou et al., 2016).

Each sources of error were computed separately:

(1) The random error on half-hourly fluxes was computed using the successive days approach developed by Hollinger and Richardson, (2007). In this approach, half hourly errors on measured fluxes (ε_m) were computed as the absolute difference between two valid successive day fluxes with similar weather. A regression between bin-averaged NEE (same number of observations per bin) and the standard deviation of the error ($\sigma(\varepsilon_m)$) was established separately for positive and negative flux values for NEE_{tot} (Felber et al., 2016b; Gourlez de la Motte et al., 2016) :

$$\begin{cases} \sigma\left(\varepsilon_m\right) = -0.11 \times \mathrm{NEE} + 1.47 \quad \text{for NEE} \le 0 \quad (R^2 = 0.90) \\ \sigma\left(\varepsilon_m\right) = \quad 0.30 \times \mathrm{NEE} + 0.08 \quad \text{for NEE} > 0 \quad (R^2 = 0.97) \end{cases} \qquad (6.5),$$

and for NEE_{past} :

$$\begin{cases} \sigma\left(\varepsilon_m\right) = -0.1 \times \mathrm{NEE} + 1.02 \quad \text{for NEE} \le 0 \quad (R^2 = 0.84) \\ \sigma\left(\varepsilon_m\right) = \quad 0.21 \times \mathrm{NEE} + 0.22 \quad \text{for NEE} > 0 \quad (R^2 = 0.94) \end{cases} \qquad (6.6)$$

For both data sets, random noise was then added to half-hourly NEE assuming an exponential distribution (Richardson and Hollinger, 2007) with zero mean and a standard deviation $\sigma(\varepsilon_m)$ (Monte Carlo simulation (Dragoni et al., 2007)). Data were then filled and annual NEE values were computed. The operation was repeated 100 times and the random error was computed as 2σ (standard deviation) of the 100 annual NEE values.

(2) The error due to additional gaps in NEE_{past} was estimated using the following procedure. First, missing data in the NEE_{past} data set were filled. Then, gaps initially present in NEE_{past} except those due to cow presence were re-added. Noise was also

added to the gap filled data using equation 6-6. By doing so, we obtain a data set without cow respiration influence but with the same number of gaps as the NEE_{tot} data set. Then, a number of gaps corresponding to the amount of additional gaps due to cow presence in the footprint were randomly added to the data set only during grazing periods. The operation was repeated 100 times and the annual NEE_{past} were computed. The error was computed as 2σ of the 100 annual NEE values.

(3) The uncertainty associated with the choice of the u* threshold was estimated by computing annual NEE values by varying the u* threshold within a plausible range of 0.13 ± 0.5 m s^{-1} (Gourlez de la Motte et al., 2016). The error was computed as 2σ of the computed values.

(4) The uncertainty associated with the choice of the cut-off frequency amounted to only 2 g C m^{-2} yr^{-1} on average at our site and was therefore neglected (Gourlez de la Motte et al., 2016).

The different sources of uncertainties were combined following Gaussian propagation rules to estimate annual uncertainties on NEE_{tot} and NEE_{past}.

Finally the uncertainty on R_{cows} was computed. As R_{cows} is computed as the difference between NEE_{tot} and NEE_{past} which are computed from the same data sets (with additional gaps for NEE_{past}), the last two sources of errors nullify. The error on R_{cows} is therefore the combination of (1) the random error affecting both NEE_{tot} and NEE_{past} during grazing events only and (2) the error due the presence of additional gaps in NEE_{past} (also only during grazing events). The resulting uncertainty on R_{cows} was computed by combining these terms following Gaussian error propagation rules. The magnitude of each error term during grazing periods is computed for both years in Table 6-1. The uncertainty on $E_{cow, hom}$ was computed by adding the relative errors on R_{cows} with the relative error of 10% on SD_p.

Table 6-1: Sources of uncertainties for annual R_{cows} values. Values are provided in g C m^{-2} yr^{-1} but are accounted only during grazing period. Random error (2σ) on NEE_{past} and NEE_{tot} were computed by adding some random noise in the data during grazing periods only. The error due to the additional gaps in NEE_{past} was computed by randomly adding gaps in NEE_{past} data set. The uncertainty or R_{cows} (2σ) was computed by combining the different error terms following Gaussian error propagation.

	Random		Gap filling	
	NEE_{past}	NEE_{tot}	NEE_{past}	R_{cows}
2013	14	12	8	20
2015	17	15	9	24

2.7. Heterogeneous approaches for Ecow

2.7.1. GPS approach

Four cattle geolocalization campaigns were organized (Table 6-2). During each campaign adult cattle positions and behavior were recorded using lab-made geopositioning trackers attached to the cows' necks. The trackers included a GPS module (FASTRAX, UP501), 4 batteries (3.8 V, 2000 mAH) and a communication antenna which allowed distant detection of malfunctions. In order to reach one month of autonomy, the devices only turned on once every 5 minutes, waited for the acquisition of at least 3 satellite signals (which typically took about 30 s), and recorded the position before turning off. Although the devices' autonomy was approximately one month, some batteries had to be replaced during the measurements, leading to some data loss. The GPS module precision was assessed by leaving the device motionless at a known position for 41 days. During this test, 50% of the points were found within 3 m, 76% within 5 m, and 95% within 11 m.

The GPS approach uses a partly similar procedure to the homogeneous approach, differing only by three steps. First, the criterion used to filter the data with the presence of cows and compute NEE_{past} is based on SD_f instead of the CH_4 flux. The filtering used a threshold of $SD_f > 2\times10^{-5}$ LU ha^{-1}. Secondly, only the NEE_{past} data set was gap filled. As result, a valid R_{cows} value is computed to be the difference between a valid NEE_{tot} measurement and a filled NEE_{past}. Finally, the cattle respiration rate per LU ($E_{cow,GPS}$) was deduced as the slope of the linear regression between R_{cows} and SD_f (Felber et al., 2016b). Only the best gap filling quality NEE_{past} values were kept for the regression (time window used by the gap filling routine lower than 15 days and all meteorological variables available (Reichstein et al., 2005)).

Table 6-2: Description of the GPS campaigns.

Period	Time frame	Duration (days)	Number of cows/calves	Main wind direction
n°1 Spring 2014	27 May 2014 - 25 Jun 2014	30	17-19/17-19	N-E
n°2 Spring 2015	14 Apr 2015 - 7 May 2015	24	12/0	S-W
n°3 Summer 2015	14 Aug 2015 - 2 Sep 2015	20	12/10	S-W
n°4 Fall 2015	19 Oct 2015 - 2 Nov 2015	15	8/0	S-E

The uncertainty on $E_{cow,GPS}$ was computed as 2 times the standard error associated to the slope of the regression. This random error on the slope of the regression is the result of errors affecting booth R_{cows} (section 2.6) and SD_f estimates. The random uncertainty associated with the computation of SD_f include three main sources of uncertainties which are the random error on GPS measurements, the fact that the position of some cows (calves and instrument failures) was unknown for certain periods as well as the use of the KM footprint function to weight the animals' contribution. It however does not include uncertainties associated with the choice of the footprint model as stated at section 2.5.

2.7.2. Confinements approach

Confinement experiments specifically designed to estimate the cattle respiration rate per LU were carried out at DTO. The methodology and the results are fully described and discussed in a previous paper (Jérôme et al., 2014). Briefly, the method consists of confining the entire herd for one day on a small part of the pasture located in the main wind direction. By confining the cows in the main wind direction area (Figure 6-1) and by filtering the fluxes according to wind direction, the probability that the cows are in the footprint area is greatly increased. The designated paddock was not grazed the day before or the day after the confinement. Fluxes measured during the confinement periods were then compared to the fluxes measured one day before and after:

$$R_{cows,conf} = \frac{\sum(NEE_i - NEE,_{i\pm24h})}{n_{obs}} \tag{6.7}$$

Where $R_{cows,conf}$ is the average respiration of all the cows in the confinement area, NEE_i is the NEE at a given hour during the confinement, $NEE_{i\pm24h}$ is the NEE at the same hour 24 h before and after the confinement, and n_{obs} the number of valid paired NEE observations. To make sure that these differences were due to cow respiration and not to micrometeorological variability, only data pairs with similar conditions were kept (soil and air temperature within 3°C, wind speed 3 m s^{-1} and photon photosynthetic flux density (PPFD) within 75 µmol m^{-2} s^{-1}). The experiment was repeated four times. The average livestock respiration rate ($E_{cow,conf}$) during the confinement was then obtained by converting the average difference in terms of kg C LU^{-1} d^{-1} by dividing $R_{cows,conf}$ by SD_c (stocking density during confinements), computed using Equation 6-3 considering φ as the average contribution of the confinement area to the footprint, A the confinement area and n_{avg} the number of animals in this area. By doing so, we consider a homogeneous repartition of the cows in the confinement area which is more realistic as cattle are confined in a smaller area that is within the footprint extent, ensuring that cows are contributing to the measured flux. In the present study, the results obtained from this former study were used but note that this latter footprint correction was not implemented in Jérôme et al. (2014) (i.e. φ was considered equal to 1).

The uncertainty on $E_{cow,conf}$ was computed as 2 times standard error of the average $E_{cow,conf}$. Note that, again, this uncertainty estimate does not account for uncertainties associated with the choice of the footprint model.

2.7.3. Animal carbon budget approach

Another possibility to estimate the cow respiration rate per LU is to compute a complete carbon budget at the animal scale when the animal is on the pasture (C fluxes at the barn are not included). This carbon budget was computed from ingested biomass estimates, combined with their C content and digestibility. The methodology and the results are fully described and discussed in a former paper (Gourlez de la Motte et al., 2016). Figure 6-3 describes the C fluxes involved in the C budget of an animal. Briefly, to build this C budget, the C ingested in dry matter

(C_{intake}) was estimated using biomass measurements combined with laboratory dry matter C content measurements.

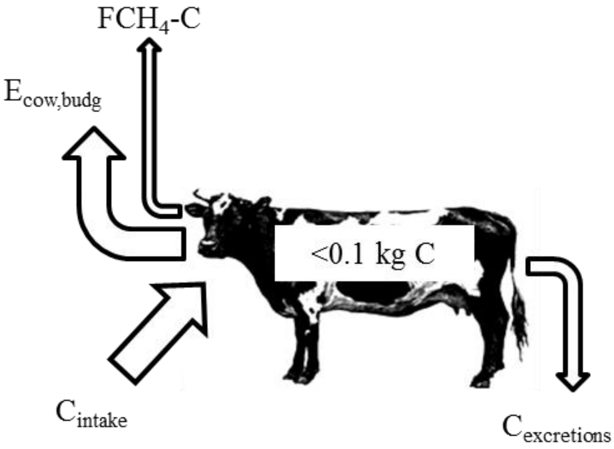

Figure 6-3: Illustration of the fluxes involved in the carbon (C) budget of a cow. $E_{cow,budg}$ corresponds to the respiration of a cow estimated from the carbon budget, FCH_4-C the methane emitted by the cow, $C_{excretions}$ the C lost in excretions, and C_{intake} the C ingested through biomass consumption.

To do so, herbage heights were measured almost once a week during the grazing season using a 0.25 m^2 rising plate herbometer over 60 points covering the whole field. Previously, an allometric equation between the herbage height and the herbage mass (HM, dry matter) was calibrated in order to convert herbage heights into HM (Gourlez de la Motte et al., 2016). For this, samples were directly harvested in the field and protected enclosures with a 0.25 m^2 quadrat. Herbage heights were measured right before and after being sampled. The samples were then dried using a forced-air oven to obtain their dry matter content. A relationship between grass height differences and harvested dry matter content was then established. Biomass C content was determined by laboratory measurements of samples following the dumas method (Dumas, 1831). Three secured enclosures were used to obtain grass growth rates during grazing periods ($HM_{gr,i}$). Cattle C intake through biomass consumption for a given period i was computed as:

$$C_{intake,i} = C_{content,grass}(HM_{beg,i} - HM_{end,i} + HM_{gr,i}) + C_{content,feeds}F_{import,i} \qquad (6.8)$$

where $HM_{beg,i}$ and $HM_{end,i}$ are the herbage mass at the beginning and at the end of the period i (weekly), $C_{content,grass}$ the C content of grass in the field, $C_{content,feeds}$ the C content of feeds supplements and $F_{import,i}$ the dry matter ingested in form of feed supplements. This equation was used on a weekly basis and the annual C_{intake} was computed by summing all the periods. Note that, when $HM_{beg,i}$>$HM_{end,I}$, this biomass is accounted negatively and is therefore considered uneaten.

The C lost by the animal through excretions ($C_{excretions}$) was computed as the fraction of non-digestible ingested carbon. Digestible and non-digestible organic matter contents were obtained by analyzing the biomass samples collected almost every week in the field using near infrared reflectance spectrometry analysis (Decruyenaere et al., 2009). Cow CH_4-C emissions were estimated using a constant fraction of the ingested biomass, which was 6% (Lassey, 2007). The meat production term ($F_{product}$) was estimated from live weight gain measurements but was negligible compared to other fluxes. Finally the CO_2 cow respiration ($E_{cow,budg}$) was computed by closing the C budget of the animal. The results obtained from this former study were directly used in the present paper.

In lack of a suitable method to evaluate the uncertainty associated with this method, no error bound was computed for $E_{cow,budg}$. Note that the main factor influencing $E_{cow,budg}$ uncertainty should be the uncertainty on dry biomass intake which is especially challenging to estimate in continuously grazed pastures.

2.8. Alternative NEE_{tot} determination

As direct NEE_{tot} estimates rely on the homogeneity hypothesis assuming an even distribution of the grazing animals, significant biases may appear if this hypothesis is not met. An alternative annual NEE_{tot} may then be provided by computing NEE_{past} (using CH_4 filter, see section 2.6) and R_{cows} independently and by summing them using equation 6-1. R_{cows} can be obtained by combining the cow respiration rate per LU obtained by one of the three methods detailed above (Section 2.7) with the average stocking density (SD_p). The uncertainty on the up scaled R_{cows} was computed by adding the relative errors on both the concerned E_{cow} and SD_p. The choice of the used respiration rate depends on the available data and the site configuration and is fully discussed in Section 4.

3. Results

3.1. Animal positions on the pasture and footprint area

Cow positions were recorded every 5 minutes during the GPS campaigns. From these position measurements, cow distribution maps were computed for both daytime (global radiation >2.5 W m^{-2}) (Figure 6-4, a) and nighttime (Figure 6-4, b). Typical annual wind roses (year 2015) are presented for these conditions. The maps show that, during the day, cattle visited the whole pasture with a slightly more important presence in the south-west direction. They also tend to cluster near the water trough and near the border with an adjacent pasture in the north-west. During the night, the cows tend to cluster in the north-east part of the pasture near the hedge. Consequently, during the nights, an important part of the pasture (essentially the south-western part), which is under the main wind direction, is not visited at all. Therefore, this observation suggests that the night stocking density in the footprint (SD_f) should be quite low when the wind is blowing from the south-west, which would imply an underestimation of cow respiration during these periods. This statement was confirmed when comparing SD_f to SD_p during the GPS campaigns (Table 6-3). When the wind was coming from the south (campaigns n°2 to 4) SD_f

observed during the nights were much lower than SD_p, while being much closer to SD_p when observed during the day. This behavior was much less visible during campaign n°1 when the wind was mainly blowing from the north-east.

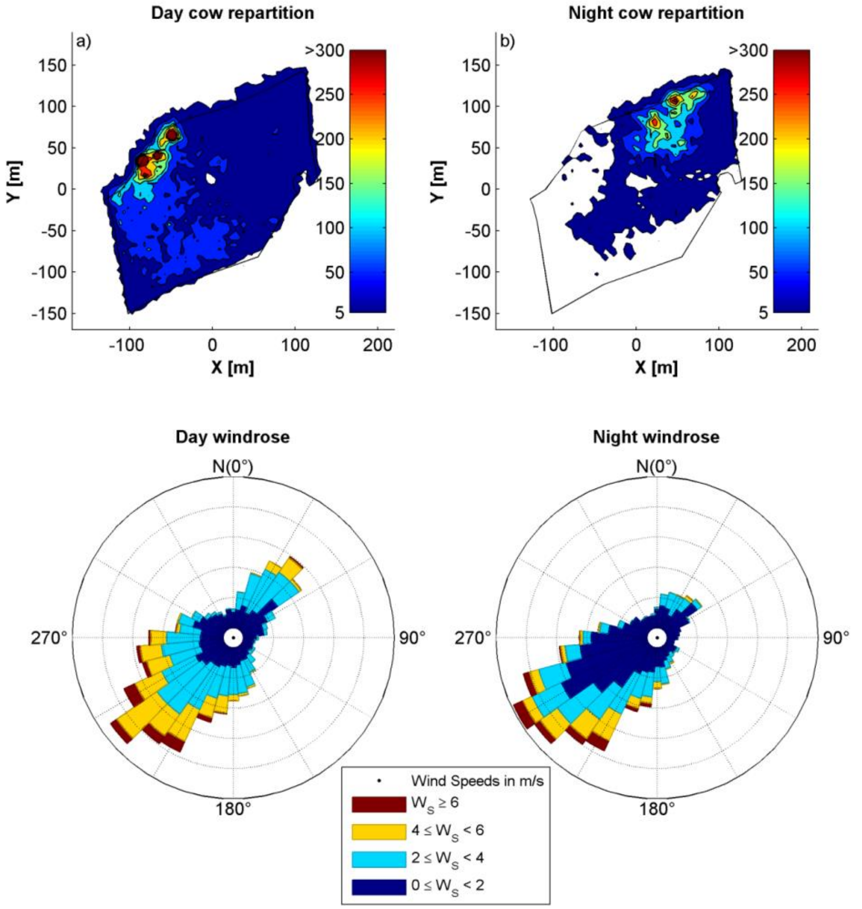

Figure 6-4: Cow distribution maps during the GPS campaigns for both days (a) and nights (b). The same scale is used for both maps. The numeric scale of the color map is given for a comparison purpose. One unit corresponds to the presence of one animal in a pixel of 5×5m^{-2} during 5 minutes. Areas colored in white are areas that are never visited by the herd. The average wind rose for the year 2015 is also presented both during the day (c) and during the night (d). For interpretation of the colors in this figure, the reader is referred to the electronic version of this article.

Table 6-3: Comparison of the average stocking densities on the pasture (SD_p) with the average stocking density in the footprint (SD_f) for the GPS measurement campaigns. The averages calculated are for all data from all campaigns combined.

Campaign n°	Main wind direction	SD_p (LU ha^{-1})	Day SD_f (LU ha^{-1})	Night SD_f (LU ha^{-1})	SD_f (LU ha^{-1})	SD_f/SD_p
1	N-E	4.9	2.7	3.9	3.1	0.64
2	S-W	1.9	1.2	1.1	1.1	0.59
3	S-W	2.7	3.2	1.0	2.3	0.85
4	S-E	1.3	1.4	0.5	0.9	0.70
Average	–	**2.7**	**2.2**	**1.7**	**2.0**	**0.75**

In addition, in regard to the shape of the footprint function (Kormann and Meixner, 2001), the contribution of the animals to the footprint also depends on their distance from the tower. Given the clustering of the cattle, particularly at night, their contribution could be low if clustered far away from the flux tower. This was investigated by comparing the average SD_f to SD_p during the night when the wind was blowing from the north-east (campaign n°1). On average, during these periods, SD_f (6.9 LU ha^{-1}) was higher than SD_p (4.9 LU ha^{-1}). This observation show that, at our site, the low SD_f observed at night were due to low cow presence in the footprint and not that much to their distance from the tower.

On average, SD_f was 25% lower than SD_p during the campaigns. This result however cannot be directly extrapolated to the entire year in terms of cow respiration, as the north-east wind conditions were over represented in the data when compared to yearly wind direction statistics (data not shown).

Nevertheless, the cow distribution maps clearly show that the cows are not evenly distributed on the pasture, especially during the night.

3.2. Cow respiration rate per LU considering a homogeneous cow repartition

3.2.1. Validation of the CH_4 flux filtering approach

In order to validate the CH_4 flux filtering approach, NEE_{past} was computed during GPS tracking campaigns by using both the CH_4 and the cow presence (GPS) criterion. The results show that, after gap filling, very similar NEE_{past} were obtained when using both partitioning methods for each campaign (Table 6-4) with differences in NEE_{past} that varied only from 0 to 4 g C m^{-2}. Identical differences between R_{cows} were observed, as they were computed as the difference between NEE_{tot} (which was the same for both methods) and NEE_{past}.

Table 6-4: Gap filled net ecosystem exchange of the pasture without cow influence (NEE_{past}) using the CH4 cow presence filtering criterion and the GPS criterion for each GPS campaign.

Campaign n°	CH_4 filter NEE_{past} (g C m^{-2})	GPS filter NEE_{past} (g C m^{-2})
1	–68	–68
2	–98	–98
3	23	22
4	17	13

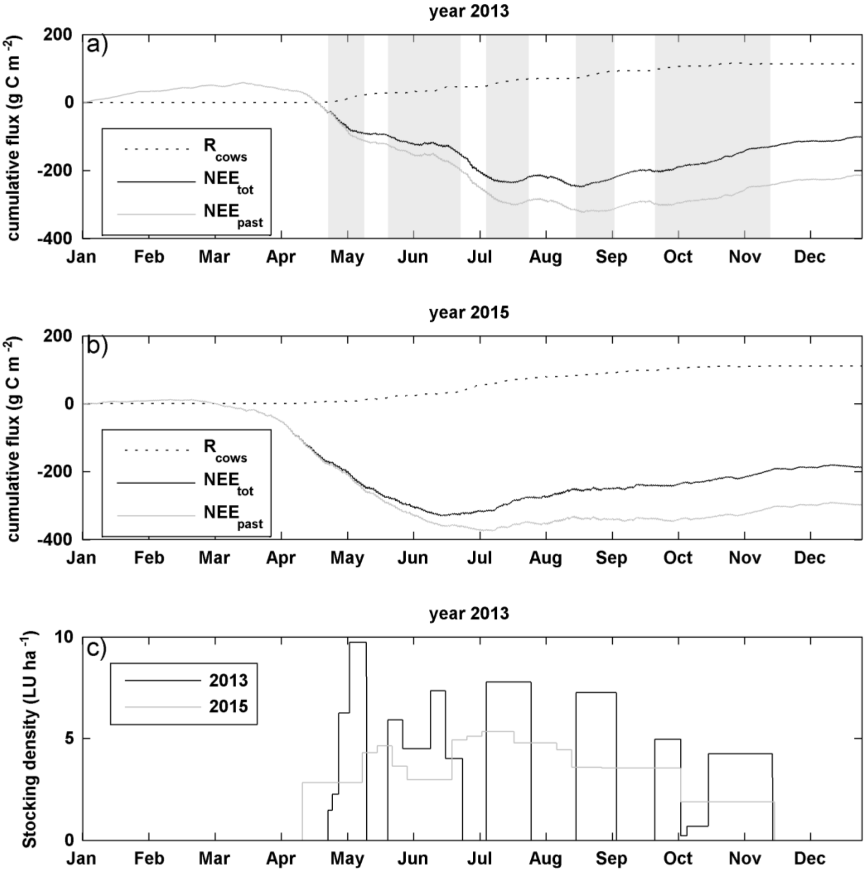

Figure 6-5: Evolution of the gap filled total cow respiration (R_{cows}), the net ecosystem exchange including cow respiration (NEE_{tot}) and the net ecosystem exchange excluding cow respiration NEE_{past} for both 2013 (a) and 2015 (b). Grazing periods are indicated in grey. (c) Evolution of stocking densities on the field for both years.

3.2.2. Discriminating NEE_{tot} into NEE_{past} and R_{cows}

The CH_4 flux filtering approach was then applied to two years of measurements. After filtering, the NEE_{tot} data set consisted of 8579 (49%) and 8432 (48%) valid fluxes (Table 6-5) in 2013 and 2015 respectively, while the NEE_{past} data set consisted of 6911 (39%) and 6325 (36%) valid fluxes. Cumulative NEE_{tot}, NEE_{past}, R_{cows} and stocking densities are shown in Figure 6-5 for 2013 and 2015. The same trend can be observed for both years. At the beginning of the year, NEE_{tot} and NEE_{past} were identical as there were no animals on the pasture. Then, the curves start to deviate from each other because of the animal. At the end of the year, when no animals were on the pasture, the curves evolve again in parallel. The total annual R_{cows} amounted to very similar values of 112 ± 20 and 111 ± 24 g C m^{-2} yr^{-1} in 2013 and 2015 respectively.

Table 6-5: Number of valid net ecosystem exchange measurements, including the cow respiration rate (NEE_{tot}) and excluding it (NEE_{past}), annual gap filled sums of both net ecosystem exchange and the total gap filled annual respiration R_{cows} for both 2013 and 2015. Note that error bar on R_{cows} are not the combination of the error bars on annual NEE_{tot} and NEE_{past} (see section 2.6).

Year	valid NEE_{tot}	valid NEE_{past}	NEE_{tot} (g C m^{-2})	NEE_{past} (g C m^{-2})	R_{cows} (g C m^{-2})
2013	8579	6911	−102 ± 22	−214 ± 24	112±20
2015	8432	6325	−188 ± 31	−299 ± 32	111±24

3.2.3. Cow respiration rate per LU ($E_{cow,hom}$)

Cow respiration rates could be computed monthly and annually from R_{cows} data sets assuming a homogeneous cow distribution on the pasture. The annual SD_p were very similar and amounted to 1.4 and 1.5 LU ha^{-1} in 2013 and 2015 respectively. As a result, the average annual $E_{cow,hom}$ amounted to 2.0 ± 0.6 and 2.0 ± 0.6 kg C LU^{-1} d^{-1} for both years (Figure 6-6, a, Table 6-6) with relatively consistent values every month except in November. During this month, SD_p was very low making R_{cows} difficult to compute. To check if $E_{cow,hom}$ was the same during the day and during the night, $E_{cow,hom}$ was calculated separately from day (Figure 6-6, b) and from night fluxes (Figure 6-6, c). The $E_{cow,hom}$ value was much higher when calculated from daylight fluxes (2.4 and 2.6 kg C LU^{-1} d^{-1} in 2013 and 2015) than from night fluxes (1.4 and 1.0 kg C LU^{-1} d^{-1} in 2013 and 2015), confirming that the cow presence in the footprint is much higher during the day than during the night, as already suggested by the cow repartition maps.

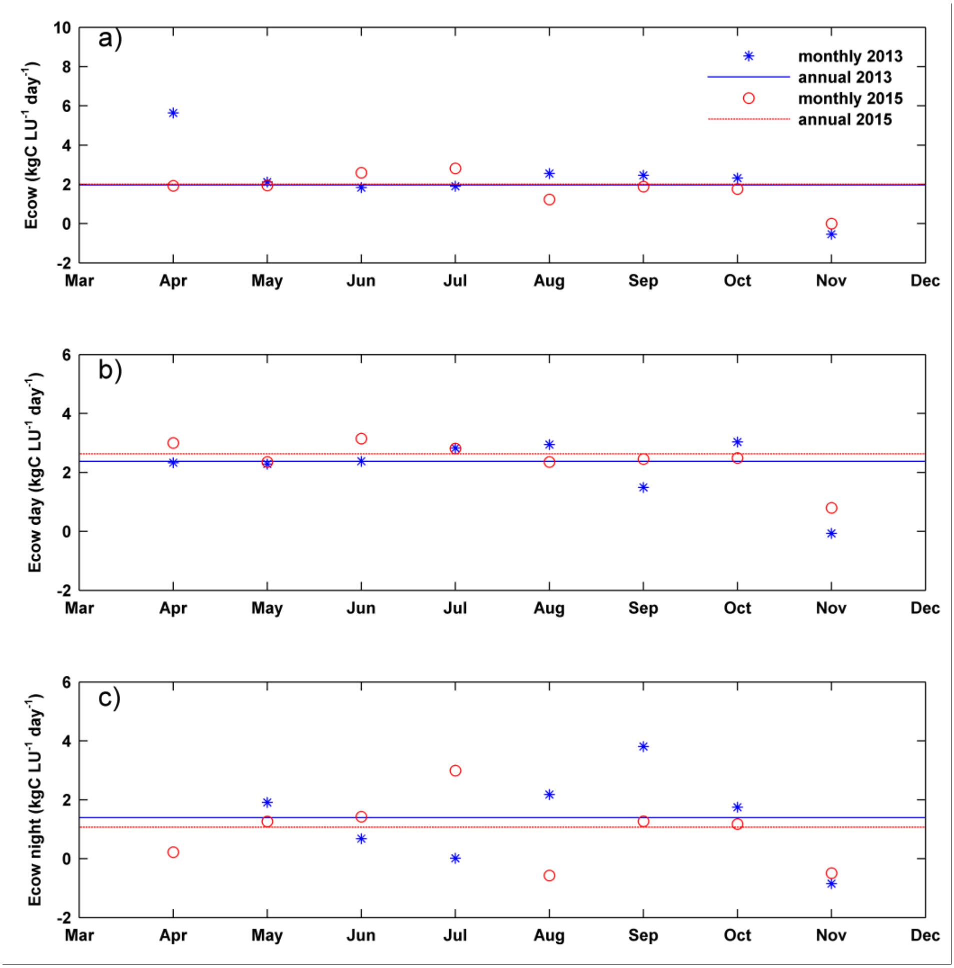

Figure 6-6: Mean cow respiration rates per LU in 2013 and 2015 computed from (a) all the data ($E_{cow,hom}$), (b) daylight data ($E_{cow,hom,day}$, global radiation >2.5 W m^{-2}), and (c) night data ($E_{cow,hom,night}$) considering a homogeneous cow repartition. Average monthly/annual respiration rates per LU were obtained by dividing total annual/monthly cow respiration (R_{cows}) by monthly/annual average SD_p. Annual values are marked by lines while circle markers correspond to the monthly values.

3.3. Cow respiration rate per LU with considering heterogeneous cow repartition

3.3.1. GPS trackers ($E_{cow,GPS}$)

A linear regression between the stocking density in the footprint (SD_f) and the total cow respiration R_{cows} was carried out on a half hourly basis in order to compute $E_{cow,GPS}$ (Figure 6-7). All GPS tracker campaigns were grouped together for a total of 803 data points available for the regression. The slope of the regression was 3160 ± 491 μmol CO_2 LU^{-1} s^{-1} (p value < 0.001, $R^2 = 0.1$) which corresponds to an average $E_{cow,GPS}$ of 3.2 ± 0.5 kg C LU^{-1} d^{-1}. The intercept of the regression was forced to zero as it was not significantly different from zero (p value = 0.96).

The linear regression is affected by important random noise. This uncertainty results in a relatively low R^2 and rather large error bounds on $E_{cow,GPS}$. Such a large dispersion was expected in view of the random error at the half hourly scale when computing R_{cows} as described at section 2.6 as well as in view of the uncertainties associated with the use GPS combined to the KM footprint function to compute SD_f (section 2.7.1).

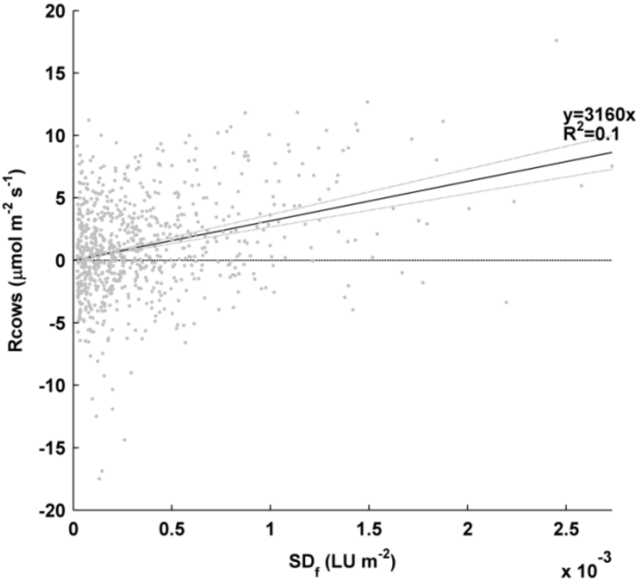

Figure 6-7: Linear regression between the total respiration of the cows in the footprint (R_{cows}) on a half-hourly time scale and the weighted stocking density in the footprint (SD_f). The fitted line (y = 3160x SE = 245, $R^2 = 0.1$) corresponds to a daily cow respiration rate of 3.2 ± 0.5 kg C LU^{-1} d^{-1}. The uncertainty bound is given as 2SE.

3.3.2. Confinement experiments ($E_{cow,conf}$)

A total of 4 confinement experiments were carried out in 2012 as detailed in Jérôme et al. (2014). After applying all selection criteria, 44 pairs of NEE data were available for the analysis. The data from two of the experiments could not be used because of inappropriate wind direction. Before footprint correction, Jérôme et al. (2014) found a cow respiration rate of 2.59 ± 0.58 kg C LU^{-1} d^{-1}. On average the contribution of the confinement area to the footprint was 71% during the experiments. As a result, after the footprint correction, $E_{cow,conf}$ was found to be 3.6 ± 0.8 kg C LU^{-1} d^{-1}, which is within the error bounds of $E_{cow,GPS}$.

3.3.3. Animal scale carbon budget ($E_{cow,budg}$)

The daily carbon budget of an animal on the pasture was computed (Figure 6-8). The results correspond to the average C budget for 5 years (2010-2014) of grazing at DTO. All the results are detailed in Gourlez de la Motte et al. (2016) but with different units (g C m^{-2} yr^{-1}). On average, cows ingested 9.5 kg of dry matter per day (8.9 kg from grazing and 0.6 from feeds). Around 87% of total above ground net primary productivity was eaten by the cows. The measured forage and feeds digestibility amounted to around 70% which corresponded to a daily cow respiration rate $E_{cow,budg}$ of 2.9 kg C LU^{-1} d^{-1}. This value is in the error bounds of both $E_{cow,GPS}$ and $E_{cow, conf}$. However, it's important to note that this budget varied from one year to another. In 2013, the productivity of the pasture was the lowest, so that the estimated C_{intake} of the cattle amounted to only 2.9 kg C LU^{-1} d^{-1} (6.8 kg of dry matter) with a cow respiration rate of only 2.0 kg C LU^{-1} d^{-1}, which is much lower than the 5-year average value. According to the farmer, such a low dry matter intake is not realistic and would have resulted in supplementary feeds given to the cows (which was not the case in 2013). It is therefore very likely that this respiration rate is under-estimated. Contrastingly, the highest C_{intake} was observed in 2011 with value as high as 5.1 kg C LU^{-1} d^{-1} resulting in a respiration rate per LU as high as 3.5 kg C LU^{-1} d^{-1}. These unexpected variations highlight the difficulty to obtain robust C_{intake} estimates in continuously grazed pastures as discussed at section 4.3. For these reasons, only the 5-years averaged $E_{cow,budg}$ value was used as a tool of rough comparison.

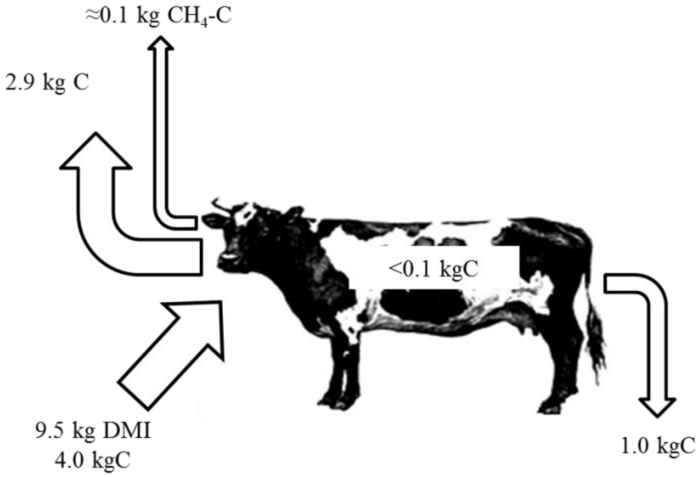

Figure 6-8: Average daily carbon budget of a Belgian Blue beef cow.

3.4. Bias induced by a non-homogeneous cow distribution

As shown in Table 6-6, $E_{cow,hom}$ was significantly (non-overlapping uncertainty bounds) than the cow respiration rate per LU estimated using either the GPS (37%

lower) or the confinement (45% lower). It was also much lower than the value estimated from the carbon budget method (31% lower). This was even more true during the night when $E_{cow,hom}$ was on average 65% lower than during the day. These results suggest a low presence of the cows in the footprint, especially during the night, as illustrated by the cow repartition maps (Figure 6-4). Despite the different methods were applied at different periods (GPS campaigns were carried out in 2014-2015, confinement experiments were carried out in 2012 and $E_{cow,hom}$ were measured in 2010-2014), which could have induced variations in cow respiration rates, we expect these variations to be limited as the herd characteristics and management remained the same during the whole experiment.

Table 6-6: Average footprint contribution of the pasture and stocking density on the pasture (SD_p), daily average cow respiration rates per livestock unit (LU) computed from a) annual gap filled data sets assuming a homogeneous cow repartition on the field from day (global radiation > 2.5 W m^{-2}, $E_{cow,hom,day}$), night ($E_{cow,hom,night}$), and all the data ($E_{cow,hom}$) and b) without assuming this cow repartition and using GPS trackers ($E_{cow,GPS}$), confinement experiments ($E_{cow,conf}$), and the carbon budget of the animal ($E_{cow,\ budg}$). Field scale cow respiration rates are also given when computed from the CH_4 partitioning (R_{cows}) and when upscaled using $E_{cow,GPS}$ ($R_{cows,GPS}$). The footprint is expressed as the percentage of the flux that comes from the field on average for each year according to the KM model.

	2013	2015
Footprint %	68%	69%
SD_p (LU ha^{-1})	1.4	1.5
Animal scale fluxes (kg C LU^{-1} d^{-1})		
a) Homogeneous cow repartition hypothesis		
$E_{cow,hom}$	2.0 ± 0.6	2.0 ± 0.6
$E_{cow,hom,day}$	2.4	2.6
$E_{cow,hom,night}$	1.4	1.0
b) No homogeneous cow repartition hypothesis		
$E_{cow,GPS}$	3.2 ± 0.5	
$E_{cow,conf}$	3.6 ± 0.6	
$E_{cow,budg}$	2.9	
Field scale fluxes (g C m^{-2} yr^{-1})		
$R_{cows,hom}$	112 ± 20	111 ± 28
$R_{cows,GPS}$	164 ± 41	175 ± 44
Bias (absolute value)	52	64

In order to assess the magnitude of the bias due to low cow presence in the footprint during the night, annual reference R_{cows} could be computed by scaling up the obtained reference E_{cow} value to the entire year. This can be done by using the E_{cow} values with one of the three methods previously proposed. For illustration purposes, $E_{cow,GPS}$ was used to quantify and correct the systematic error made at DTO. This method was chosen as it seemed to be the most suitable for free range pastures as discusses at section 4.3. Nevertheless, similar conclusions would have been met using other methods. When scaled up, $R_{cows,GPS}$ amounted to 164 ± 41 and 175 ± 44 g C m^{-2} in 2013 and 2015 respectively (Table 6-6), which suggests a systematic underestimation of R_{cows} and thus an overestimation of NEE_{tot} of 52 and 64 g C m^{-2} yr^{-1} (51% and 34% of NEE_{tot},) in 2013 and 2015. As a result, new NEE_{tot} (computed as NEE_{past} + $R_{cows,GPS}$) values were –50 ± 48 and –122 ± 55 g C m^{-2} yr^{-1} (the error bounds were computed by quadratically adding errors on annual NEE_{past} and $R_{cows,GPS}$).

4. Discussion

4.1. Using methane fluxes as a NEE_{tot} partition tool

The CH_4 flux filtering approach has proven to be a useful tool to partition NEE_{tot} and disentangle the net ecosystem exchange of the soil and the vegetation (NEE_{past}) from the respiration of the cows. The results at DTO showed that similar NEE_{past} values were obtained using this method and the GPS tracker method.

Compared to the GPS method, the main advantage of the CH_4 flux filtering approach is that it can be more easily used routinely, whereas the use of GPS trackers requires specific instrumentation that is not commercially available, and is man-power consuming. The use of the CH_4 flux filtering approach was also supported by Felber et al., (2016b, Figure 13) who found a good correlation between measured CH_4 fluxes and cow respiration in the EC footprint. To do so, CH_4 fluxes must be available, but these are more and more frequently measured at grazed sites (Coates et al., 2018; Dengel et al., 2011; Dumortier et al., 2017; Felber et al., 2015; Jones et al., 2017) thanks to the increasing availability of fast and precise CH_4 sensors. This method can therefore be used on larger data sets as long as CH_4 fluxes are measured (which we advocate).

The method cannot be used to estimate consistent cow respiration rates per LU when the cows are not evenly distributed on the pasture, but is promising as a partitioning tool of NEE_{tot} into NEE_{past} and R_{cows}, which is the first step needed to check if R_{cows} is measured in a representative way and to correct NEE_{tot} estimates if this is not the case. The successful application of the partitioning method in the present study overrules the statement by Felber et al. (2016a) that the computation of NEE_{past} would not be possible for continuously grazed pastures as no sufficient and defined periods without cows in the footprint would be available.

4.2. Biased NEE estimates because of a non-homogeneous cow repartition

The application of the methodology at the DTO site showed that NEE_{tot} estimates based on direct EC measurements were subject to a non-negligible bias of about 60 g C m^{-2} yr^{-1} because of non-homogeneous cow repartition resulting in an underestimation of R_{cows}. This underestimation implies that the carbon sink activity of the pasture was considerably overestimated when using NEE_{tot} values to compute its net biome productivity. The NBP (including cow respiration, equation 6-2) of the pasture was computed for 5 years (2010-2014) in a previously published paper using NEE_{tot} estimates and other non CO_2 carbon fluxes (Gourlez de la Motte et al., 2016). Those results showed that the pasture acted as a C sink every year with an average NBP value of -161 g C m^{-2} yr^{-1} (lowest absolute in 2013: -87 g C m^{-2}, highest absolute value in 2014: -176 g C m^{-2}) and an average annual stocking rate of 2.3 LU ha^{-1}. If we assume that the NBP was affected by the same bias of ≈ 60 g C m^{-2} yr^{-1} (around 37% of NBP) every year because of cow respiration underestimation, the corrected average NBP is reduced (in absolute values) to ≈ -100 g C m^{-2} yr^{-1}. The magnitude and sign of this bias is of course site specific so that, depending on the site configuration, the wind direction, and the gregarious behavior of the animals, it can lead to either positive or negative systematic errors. This must therefore be verified on a case by case basis. It is important to highlight the fact that gregarious behaviors of the animals on free range pastures are expected, at least for cows (Hassoun, 2002) and sheeps (Dumont and Boissy, 2000). The methodology presented in this paper may be used at each site to detect and, if necessary, estimate this bias and correct C budgets accordingly.

4.3. Method to measure a reference cow respiration rate per LU

In this paper, three methods were proposed and tested at DTO to estimate a reference E_{cow} that does not assume a homogeneous cow repartition in the pasture and that can be used as a basis of comparison to check if R_{cows} is measured in a representative way. This respiration rate per LU can also be used to correct R_{cows} if necessary.

The GPS tracker method appeared to be very useful as it provided an improved understanding of animal location habits. The distribution maps have proven to be a useful tool to detect heterogeneous cattle distributions. The use of GPS devices combined with footprint models also provides a more realistic stocking density in the footprint (Felber et al., 2015, 2016b). This footprint function is however also the subject of several uncertainties (Dumortier et al., 2019). Finally, the GPS tracking method has the advantage of not disturbing the behavior of the cows when compared, for example, to confinement experiments.

The confinement method gave consistent results when compared to the other methods. This method is less time consuming than the use of GPS trackers and doesn’t require any specific equipment. This is true especially in intensive rotationally grazed pastures where confinement is expected (Gourlez de la Motte et al., 2018). Confinement in rotational grazing systems can be exploited to compute $E_{cow,conf}$ as shown by Gourlez de la Motte et al. (2018). If the rotations are longer than one day, an adapted procedure is proposed in the cited paper. However, confinement also has several drawbacks. First, very similar weather conditions and wind direction during and after the confinement must be met in order to compare the fluxes from the same area. Secondly, the respiration may also be modified (especially for free range pastures) as confinements may alter the cow's feeding behavior and activity. In addition, confinement experiments are based on the hypothesis of a homogeneous cow repartition. This is more realistic as confinement is exerted in a smaller area that is within the footprint extent, ensuring that cows are contributing to the measured flux. However, it cannot be determined to what extent. This source of uncertainty should however be lowered when replicating confinement experiments and when using daily fluxes as cows tend to spread more evenly during the day. Finally, as stated above, cow contribution cannot be weighted by using a footprint model which may lead to other biases.

The animal carbon budget approach requires an estimation of the C_{intake} of the cows which requires reliable biomass growth measurements as well as forage digestibility measurements for the whole grazing season. These types of measurement are time consuming but are often carried out at grazed EC sites (Gourlez de la Motte et al., 2016; Klumpp et al., 2011; Rutledge et al., 2017b; Skinner, 2008; Skinner and Dell, 2015). Estimating the C_{intake} of cows is especially difficult in continuously grazed sites where grass growth during grazing must be estimated. This was done at the DTO by simulating grazing using protected enclosures. However, it is not easy to ensure that grass growth observed in these protected enclosures is representative of the whole pasture. In short rotation grazing sites, the regrowth can be considered negligible, making the computation of C_{intake} easier and more reliable (Skinner, 2008). Another option to compute C_{intake} is to estimate the energy requirements of the animals for maintenance, activity, and grazing and convert this energy into dry matter intake (and then C_{intake}) (IPCC, 2006a) or, for dairy cows, using equations based on milk yields and the lactation week of the cows, as proposed by Felber et al. (2016a).

5. Conclusions and recommendations

The results of this study highlight the necessity to carefully check if cow respiration is measured in a representative way by the EC system when dealing with grazed pastures. To do so, monitoring the presence and number of cows on the field is highly advised (Figure 6-5, c). For beef cattle, monitoring the presence of the cattle on the field is easier as off pasture times are greatly reduced. For dairy cattle, the task is a bit more difficult as the cows often leave the pasture for milking. These milking periods must therefore be accounted for as well. Measuring the CH_4 fluxes is also highly advisable as it allows the computation of NEE_{past} which is the first step of the proposed methodology and can be used for any kind of pasture (i.e., continuous grazing, rotational grazing, etc.) grazed by ruminants. Finally, estimating a reliable cow respiration rate as a reference is also required. For this last step, three methods are proposed and the choice of the method can differ depending on the available data and the configuration of the site. As a general rule, combining two or three methods is always better as their comparison gives the most defensible results.

For a continuously grazed site, the GPS campaigns are very useful as they allow the habits of the herd to be assessed without disturbing their behavior. However, organizing these campaigns can be time consuming and requires expensive equipment. As an alternative, the use of digital camera combined with an animal detection software have also proven to be a valuable tool to detect the presence of cows in the EC footprint (Baldocchi et al., 2012a). If GPS (or any other localization devices) monitoring is not available, repeated confinement experiments are cheap, relatively easy to implement, and also provide consistent results. Combining these confinement experiments with animal C budget estimates is advised in order to check the consistency of the results. Using only the animal C budget is less advisable as C_{intake} estimates may be uncertain for continuously grazed pastures.

For rotationally grazed sites composed of several paddocks, GPS trackers may be avoided. In these sites, the cows are constrained to a relatively small paddock so that their location is known. Combining a footprint model (or simply wind direction) with a precise grazing schedule allows correct assessment of the presence of cows in the footprint in order to compute NEE_{past}, as shown by Felber et al. (2016b). If available, CH_4 fluxes can still be used as a partitioning tool. For these sites, the confinement method should be preferred as cattle are already expected to be confined (Gourlez de la Motte et al., 2018). Again, it's advisable to combine the confinement experiments with an animal carbon budget in order to constrain the $E_{cow,conf}$ value to obtain more defensible estimates. For rotationally grazed sites, another solution would consist in computing NEE_{past} and excluding the grazers from the ecosystem. When computing NBP, the grazers are therefore considered to be an agent of C export (by grazing) and import (by excretions) (Felber et al., 2016a; Rutledge et al., 2017a, 2017b; Skinner, 2008). This solution requires reliable biomass measurements and/or animal performance data in order to compute C_{intake} and $C_{excretions}$. For this reason, using this solution for continuously grazed sites is less advisable. Note that, if the estimation of E_{cow} and $C_{excretions}$ are estimated from the animal C budget, both methods are equivalent and give the same results.

Finally, the results of this study highlighted how grazers can significantly affect NEE values reported in grazed grassland studies. Therefore, a consistent approach to report CO_2 fluxes derived from eddy covariance in grazed ecosystems is needed in order to allow better NEE inter-site comparisons. In this line of thought, we advocate that, when possible, NEE_{past} and grazers respiration should be computed separately in both continuously and rotationally grazed systems. By excluding grazer's respiration, the reported NEE_{past}, which correspond to the NEE of the vegetation and soil only, would be more comparable to the values reported by other grazed grassland studies as well as those reported by mown meadows. This would also help modelers as it would allow the computation of both fluxes separately (Kirschbaum et al., 2015). In this sense, continuously measuring CH_4 fluxes in grazed ecosystems has proven to be very useful to obtain consistent NEE_{past} values.

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Chapter 7

General discussion, conclusions and perspectives

1. Validity of the flux approach

In this work, soil carbon storage was estimated by building a complete carbon budget (NBP) of the pasture. This approach requires the measurements of several fluxes that may be affected by some random and systematic error. One of the main objectives of this work was to develop a robust methodology to compute a complete carbon budget of grazed pasture and assess uncertainties associated with the method. More specifically, two main sources of systematic error affecting annual NEE and NBP were identified. In both case, we proposed a method to correct the errors.

The first systematic error that we identified was linked to the choice of the cospectral model used for the high frequency loss correction. Indeed, in chapter 4, we showed that the choice of the cospectrum used to correct the data had a major impact on annual NEE values reaching errors as high as 71-150 g C m^{-2} yr^{-1}. By comparing the fluxes measured by eddy covariance with chamber flux measurements, we showed that using a local cospectrum for the high frequency loss correction should be preferred instead of using a generic one (Kansas cospectrum).

The second systematic error highlighted in this work is more specific to grazed sites and is linked to the cow distribution on the field. Indeed, in grazed pastures, cattle act as moving sources of CO_2 that will or will not be measured by the eddy covariance system, depending on their location and wind conditions. In the first part of this work (chapter 3 and 5), in the absence of sufficient additional data on the herd position habits, we explicitly made the hypothesis that, on average over a complete year, the distribution of the cows on the field could be considered as homogeneous so that their annual respiration was deduced from the flux tower data in a representative way. When enough GPS and CH_4 flux data became available, a procedure was developed to verify this hypothesis (chapter 6) and the results highlighted that it was not the case, this hypothesis causing an under estimation of cow contribution to NEE of around 60 g C m^{-2} yr^{-1}.

Once these two sources of error were carefully identified and corrected, residual uncertainties affecting NBP had to be estimated. As a reminder, the NBP of the ecosystem can be computed using (equation 3.1):

$$NBP = NEE_{tot} + F_{CH4} + F_{manure} + F_{import} + F_{harvest} + F_{product} + F_{leach}$$

For a complete description of the different variables, the reader is referred to chapter 3. As mentioned at chapter 6, NEE_{tot} should be computed by separating NEE_{past} and R_{cows}:

$$NEE_{tot} = NEE_{past} + R_{cows}$$

The residual uncertainty on NBP after all the needed correction can be computed by combining the different error terms of each variable of equation 3.1. Note that a first uncertainty analysis was proposed in chapter 3. The estimated error bound must however be adapted as the partitioning of NEE_{past} in its two components results in new sources of residual uncertainties.

For NEE_{past}, the uncertainty was computed by combining (1) a random error term affecting both measured and filled data, (2) an error due to the additional gaps in the data set because of cow presence in the footprint, (3) a residual uncertainty associated with the choice of the u* threshold used to filter fluxes under low turbulence conditions and (4) a residual uncertainty associated with the choice of the cut-off frequency for the high frequency loss corrections. The method used to compute these terms is fully described at chapter 6.

R_{cows} was computed by upscaling the cow respiration rate per LU measured with the GPS approach ($E_{cow,GPS}$) to the entire year using the annual stocking density on the pasture (SD_p). The uncertainty on R_{cows} was computed by combining the uncertainty on $E_{cow,GPS}$ with the uncertainty on SD_p as described in chapter 6.

In the absence of a better approach,, the uncertainty on the other terms of NBP was computed, by considering a 10% error on the other terms of NBP. All the different error terms were combined following Gaussian propagation rules.

In 2013 and 2015, NEE_{tot} could be partitioned in its NEE_{past} and R_{cows} components. By combining both sources of error, the total uncertainty on corrected NEE_{tot} values amounted to 48 and 55 (52 on average) g C m^{-2} yr^{-1} in 2013 and 2015 respectively. The total uncertainty associated with the other components of NBP amounted to 8 g C m^{-2} yr^{-1}. Note that, if this last source of uncertainty may look small, these fluxes only amount to 20 g C m^{-2} yr^{-1} when summed which is much smaller than NEE_{past} and R_{cows}. As a result, the average uncertainty on NBP was mostly due to uncertainties associated to NEE_{past} and R_{cows} computation and amounted to around 52 g C m^{-2} yr^{-1} which can further be rounded to 50 g C m^{-2} yr^{-1} which is in the higher range of the uncertainty bounds proposed by Baldochi, (2003) for annual NEE estimates for all kind of ecosystems. This uncertainty bound was also very similar to the uncertainty on NBP computed for a dairy farm in new Zealand (±54 g C m^{-2} yr^{-1}, Rutledge et al., (2015)) but a bit lower than the uncertainty bound estimated for a dairy farm in Switzerland (± 62 g C m^{-2} yr^{-1} Felber et al., (2016a) mostly because of higher C imports and exports resulting in higher absolute uncertainties for those components.

One of the main remaining questions concerning the uncertainties is their combination especially when trying to compute uncertainties affecting annual NEE estimates. In this work, in the absence of a better approach, we combined all the different sources of errors following Gaussian error propagation rules. By doing so, we consider that all these sources of uncertainties are uncorrelated. Further work is needed in order to better understand and take into account the eventual correlations between the different sources of uncertainties.

2. Corrections regarding cow respiration

In this section, conclusions regarding the carbon budget (chapter 3) and the comparison between rotational and continuous grazing (chapter 5) are revised in regards to the systematic error highlighted in chapter 6.

2.1. Impact on the carbon budget

The C budget of an intensively managed permanent grassland grazed by Belgian Blue cattle was computed for a period of 5 years (2010-2014). The results showed that, despite its old age and its intensive management, the pasture acted a C sink every year (chapter 3, article 1). However, because, on an annual time scale, cow respiration was under-estimated at DTO, the average annual NEE computed in chapter 3 was biased. The corrected NBP value, given in Chapter 5, was -100 ± 50 g C m^{-2} yr^{-1}. Note that despite this important correction, the pasture still acted as a significant C sink and that the analysis made in chapter 3 are still valid. Indeed, CO2 fluxes and (NEE and R_{cows}) are still the main components of the computed NBP and all the considerations made on CO_2 fluxes dynamics as well as weather and management impact on NBP are still valid.

2.2. Impact on the grazing timing experiment

An experiment was designed to better understand the impact of rotational and continuous grazing on the pasture CO_2 exchange dynamics (chapter 5). Again, cattle distribution was explicitly considered homogeneous for the CG treatment. For the RG treatment, as mention in the paper the hypothesis is more likely to be met as fluxes were discarded when the footprint area was outside the confinement parcel thereby increasing the probability that the herd is in the footprint. Nevertheless, as already stated in the paper, all the observations on the impact of grazing on NEE dynamics are valid for the two parcels as, on a small time scale basis, cow respiration only adding noise to the data. However, concerning the cumulative NEE, some re-evaluation of the final numbers must be made.

As the objective of the study was to compare the impact of both managements on the NEE of the vegetation, comparing NEE_{past} of each treatment is more suitable than comparing NEE_{tot}. For the CG treatment, $NEE_{past,CG}$ was computed following the methodology fully described at chapter 6. For the RG treatment, $NEE_{past,RG}$ was computed by removing the fluxes measured during confinement experiments before filling the data.

In 2015, for the period of the study (from 14th April to 17th November), revised $NEE_{past,CG}$ and $NEE_{past,RG}$ amounted to -185 g C m^{-2} yr^{-1} and -235 g C m^{-2} yr^{-1} respectively leading to a significant difference ΔNEE ($NEE_{past,RG}$-$NEE_{past,CG}$) of +50 ± 42 g C m^{-2} yr^{-1} between the two treatments. The uncertainty bound was computed by considering an uncertainty of 30 g C m^{-2} yr^{-1} for both $NEE_{past,CG}$ (table 6-5) and $NEE_{past,RG}$. However, if we take into account the fact that the average stocking rate was slightly lower in the RG treatment (2.1 and 1.9 LU ha^{-1} for the CG and RG treatment respectively), this difference can be considered non-significant when standardizing both NEE to the same stocking rate of 2.0 LU ha^{-1} (normalized ΔNEE=29 ± 42 g C m^{-2} yr^{-1}). On the other hand, the used uncertainty bound estimates is rather conservative. Indeed, this error term includes the error associated with the u* threshold used to filter the data. If we consider that the magnitude of this error term is very similar for both fields so that this error term can be almost neglected when comparing NEE of both fields (Ammann et al., 2007; Rutledge et al., 2017a), the error on ΔNEE is lowered to around 30 g C m^{-2} yr^{-1}. As a result, even with this lower uncertainty bound, the standardized difference between the two treatments is too small to be detected. The NEE standardization by stocking rates is however questionable as increasing the stocking rate can be done by either increasing the stocking density during the confinement, increasing the duration of the rotations or even by adding a new rotation. As each option can probably affect NEE in very different ways, one can not be sure that simple standardization is a reliable way to compare both treatments. In conclusion, the observed difference between the two treatments is relatively small and it is not possible to conclude with confidence that this difference is significant and that it is not due to differences of stocking rates.

3. Paired tower experiments to study management effects on CO_2 fluxes

A paired tower experiment was carried out at DTO to study the impact of grazing strategies on CO_2 fluxes dynamics and annual NEE. These approaches allow isolating one management practice and study its impact on C sequestration processes but also on other gas emissions such as N_2O (Drewer et al., 2017). Similar paired tower experiments have been carried out in other studies to assess the impact of different management practices on CO_2 fluxes and C budgets such as pasture renewal (Drewer et al., 2017; Rutledge et al., 2017b), increased species diversity (Rutledge et al., 2017a), grazing and/or fertilization intensity (Allard et al., 2007; Ammann et al., 2007; Klumpp et al., 2011) as well as mowing and grazing (Senapati et al., 2014) using relatively short term experiments.

In this work, the paired experiment was focused on the impact of a management practice on NEE. In grazed pasture, such studies should be carried out by comparing NEE_{past} instead of NEE_{tot} in order to reduce uncertainties as highlighted in chapter 6 especially when comparing grazed and mowed pastures (Senapati et al., 2014). By doing so, the uncertainty analysis showed that only differences > 30g C m^{-2} could be detected if the experiment is perfectly designed with very similar management (during and before the experiment) except the isolated treatment. If the focus of the study concerns not only NEE but extends to NBP, uncertainties associated with the other fluxes such as cow respiration (but not only) should also be added leading to even higher detection limit. Such high differences in term of NEE and NBP for different treatments were already highlighted in other paired tower studies for pasture renewal (Rutledge et al., 2017b), increased mixed species (Rutledge et al., 2017a) and fertilization intensity (Ammann et al., 2007).

Ideally, paired experiments should start their measurements several years prior to the treatment in order to evaluate possible pre-treatment differences (Rutledge et al., 2017a, 2017b). However, these kinds of measurements are expensive and it is therefore not always possible to do these measurements for such long periods. In such case, it is important to make sure that both soil and management of the two parcels prior to the treatment are as close as possible in order to limit pre-treatment associated uncertainties (Rutledge et al., 2017a). Note that these limitations only concerns conclusions regarding annual NEE and NBP budgets. Indeed, paired tower experiments have proven to be a valuable tool to understand the impact of an isolated treatment (in this case grazing management) on CO_2 flux dynamics.

4. Comparing C budget with direct soil sampling SOC variations measurements

As discussed above, the flux approach can be affected by different systematic errors. Some of them are well identified and can be corrected. However, despite all the efforts, one must not discard the possibility that some unknown systematic errors remain. Without crossed data, it’s very difficult to ensure that SOC variations derived from the C budget approach are not affected by some errors. Therefore, there is a strong need to constrain flux based soil carbon sequestration estimates with direct SOC variations measurements. This was done by Skinner et Dell, (2015) in an intensively managed grassland. The study compared SOC variations based on direct samplings with long term flux based measurements (8 years of data). The results highlighted an important C loss in the deep layers that was not explained by the flux approach thereby highlighting the possibility of potentially higher than expected C leaching and run off. This study also highlighted that sampling as deep as 1m depth was required in order to properly compute SOC variations. To do so, long term flux measurements are required as around 10 years between each sampling are needed in order to faithfully catch SOC variations. In an extensive literature survey, Soussana et al., (2010), highlighted that, when averaging several studies, C storage estimated using the flux approach was not significantly different than C storage rates obtained from SOC variations measurements. However, as previously mentioned, combined studies are still lacking. As a perspective of this work, combining the flux approach with the direct soil sampling approach would have several advantages. This will be done in the future at DTO in the scope of the Integrated Carbon Observation System (ICOS) monitoring program. A first soil sampling is intended in 2019 following ICOS standardized ICOS protocols (Arrouays et al., 2018).

An extensive regional assessment of SOC changes in agricultural soil was carried in southern Belgium for the period 1955-2005 (Goidts and van Wesemael, 2007) with direct soil samplings. The survey showed that grassland in the Condroz region (region where Dorinne is located) gained around 23.9 t C ha^{-1} for the whole period in the 30 first cm which corresponds to a yearly sequestration rate of 48g C m^{-2} $year^{-1}$ which is lower than the observed C sequestration rate at DTO. It’s however difficult to go further in the comparison for several reasons. First, even if the authors of this study tried to sample the territory in homogeneous units, some spatial variation in C sequestration rates within the Condroz region unit is expected. Observing a higher rate in a given spot is therefore not surprising. Secondly, some temporal variations in C sequestration rates are also expected. Last, this study only measured SOC variation in the 30 first cm soil layer although SOC changes can occur in the deepest layers (Skinner and Dell, 2015).

Depending on many variables such as grassland types (temporary vs permanent), climatic conditions (Baldocchi et al., 2017), fertilizer supply (Ammann et al., 2007), manure management (Hirata et al., 2013; Shimizu et al., 2009) and grazing pressure (Allard et al., 2007; Klumpp et al., 2011) as well as past management (Smith, 2014), net carbon sequestration varies both in time and space. To assess these variations, both approaches have their own strengths and weaknesses. Monitoring CO_2 and C fluxes using the flux approach has proven very useful to catch short and long term temporal variations in the C balance and its drivers at the ecosystem scale. Thanks to a high temporal resolution, this method is suitable to assess which mechanisms are responsible for temporal variations. The flux method was also successfully used in this work but also in other studies (Klumpp et al., 2011; Mudge et al., 2011; Rutledge et al., 2015; Soussana et al., 2007) to determine the respective importance of each term of the C budget and which processes were the most responsible for inter-annual variations not only in grasslands but also in other ecosystems (Baldocchi et al., 2017). On the other hand, one of the main weaknesses of the flux approach is that, because of high costs and labor requirements, the method cannot easily be replicated to faithfully catch C sequestration spatial variability. Studying soil sequestration spatial variability can be more easily done using extensive direct samplings measurements (Conant and Paustian, 2002; Goidts and van Wesemael, 2007; Lettens et al., 2005a) combined with modelling (Meersmans et al., 2011). One other drawback of the flux approach is that, only the random components of the uncertainty will decrease with time while, with the soil samplings approach, uncertainties should be reduced when the time between samplings increases.

5. Carbon storage mitigation role in total GHG budget

Three main GHG are exchanged at the pasture scale: CO_2, CH_4 and N_2O. The results of this work showed that the grassland acted as a significant carbon sink thereby removing some CO_2 from the atmosphere. Jointly some CH_4 is emitted mostly through cattle enteric fermentation and some N_2O is emitted after fertilization events as well as by cattle dejections. Therefore, several questions arise: is the pasture system a GHG sink or source (in term of CO_2-eq)? What is the exact mitigation potential of soil C sequestration?

5.1. Pasture scale CH_4 emissions

During the project, CH_4 emissions have been measured at DTO using the eddy covariance technique first without cattle localization (Dumortier et al., 2017). The whole measurement set-up is described in the cited publication. The results showed that most of the CH_4 was emitted by the cattle, the exchanges from the continuum soil/vegetation being negligible. To go further, EC CH_4 measurements were performed during the GPS measurements campaigns described in chapter 6.

The used procedure to compute a CH_4 flux by LU was very similar to the one used to compute CO_2 respiration per LU. Briefly, the stocking density in the footprint (SD_f) was computed by combining cattle localization measurements with an analytical footprint model (Kormann and Meixner, 2001) as described in chapter 6. Then a linear regression between CH_4 fluxes and SD_f was computed, the slope of the regression corresponding to the CH_4 flux per LU and per day. This CH_4 flux per LU could then be up scaled to the entire year using the pasture annual stocking rate. All the work has been carried out by Pierre Dumortier and the results presented in this manuscript are the results of his work.

On average, the cows emitted 160 g CH_4 day^{-1} LU^{-1} at DTO. This value is higher than the value estimated from EC measurements at DTO by considering a homogeneous cow repartition (Dumortier et al., 2017) but is very close to the tier 2 IPCC emission estimate (IPCC 2006, raw energy content of 18.45 MJ kg-1 and default conversion factor 6.5%) that amounted to 164 g C m^{-2} yr^{-1} . When considering an average annual stocking rate of 2.3 LU ha^{-1}, the total annual emission reached 134 kg CH_4 ha^{-1} yr^{-1}. When considering a 100 year global warming potential of 28 (IPCC, 2014), the total emission amounted to 375 g CO_2-eq m^{-2} yr^{-1}.

5.2. Pasture-scale N_2O emissions

N_2O is emitted by the soil during nitrification and denitrification. Nitrification is the process where ammonium is oxidizes into nitrite and nitrate. The reaction is performed by both autotrophic and heterotrophic bacteria located in the soil under aerobic conditions. Denitrification is the reduction of nitrate to nitrite and, at the end of the chain reaction N_2. The reaction is performed by soil heterotrophic bacteria and fungi under anaerobic conditions (Lognoul et al., 2017). In grazed grassland N is brought to the pasture in several ways (Figure 7-1) and in different forms (Liang et al., 2018). Some N is brought to the soil through organic and mineral fertilization (N_{fertil}), atmospheric depositions (N_{dep}), plant residuals (N_{resid}), and in form of cattle excreta (dungs and urines) and by the decomposition of the plants. These sources of N are brought to the soil in different forms that may be the subject to nitrification or denitrification depending among other on their forms and on oxygen availability which may result in some N_2O emissions.

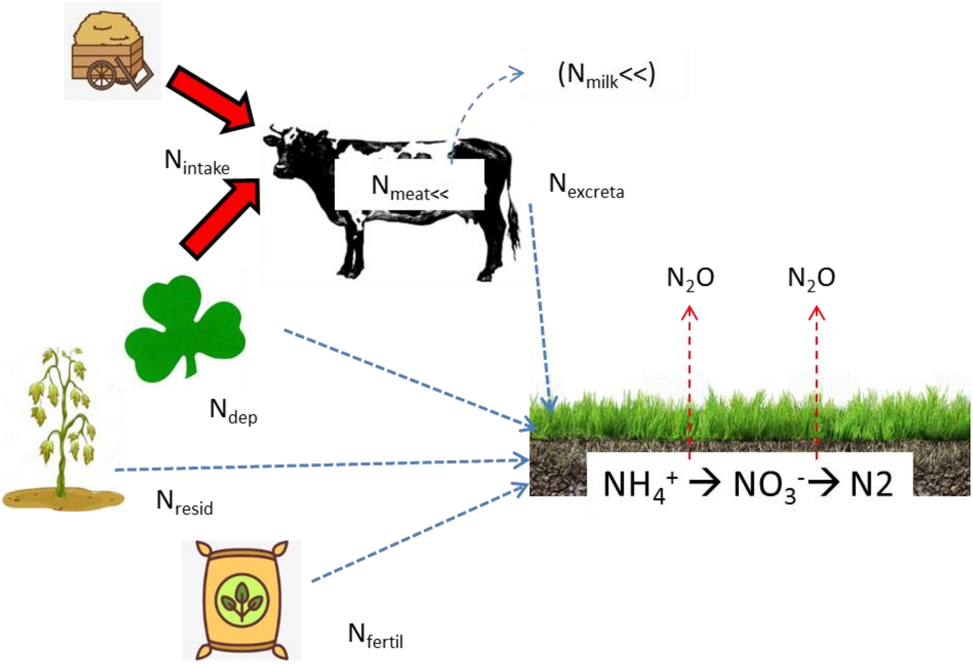

Figure 7-1: Schematic illustration of the N_2O emission process in a grazed pasture.

In the lack of available direct measurements of N_2O fluxes at DTO, N_2O emissions were estimated following tier 1 IPCC methodology and emission factors (IPCC, 2006a):

$$F_{N2O} = (N_{dep} + N_{resid} + N_{fertil}) * f_1 + N_{excreta} * f_2 \qquad (7.1)$$

where F_{N2O} is the flux of N_2O, f_1 and f_2 are the IPCC emission factors with a default value of f_1=0.01 (0.003-0.03) and f_2=0.02 (0.007-0.06). Uncertainty values on the emission factors are given in brackets. N_{fertil} was obtained by averaging data from management inventory for the period 2010-2014. N_{dep} is assumed to be 25 kg N ha^{-1} yr^{-1} which is in the higher range of the N_{dep} estimated by Flechard et al., (2011). N_{resid} is estimated as the proportion of N in uneaten biomass which was estimated to be 15% of cattle N_{intake}. N_{intake} was measured from feed dry matter intake estimates and from dry matter laboratory N content measurements. The dry matter N content was relatively stable among samples with an average value of 3±0.04% (error given as SE, 167 samples). $N_{excreta}$ was considered equal to N_{intake} as N accumulated in meat can be considered negligible.

The annual average emission rate over the period 2010-2014 was estimated to 3.9 (0.7-6.0) kg N_2O-N ha^{-1}, which corresponds to an emission rate of 188 (63-564) g CO_2–eq m^{-2} yr^{-1}, with a major contribution of cattle excreta induced emissions (68%). This value is in the same order of magnitude when compared to N_2O emissions estimated from eddy covariance measurements in an intensively managed and grazed grasslands (3.0 kg N_2O-N ha^{-1}) in Scotland (Jones et al., 2017) but were much lower than those observed in an intensively managed rotationally grazed dairy pasture in new Zealand (6.5 kg N_2O-N ha^{-1}) but with much higher stocking rates and very different climatic conditions (Liang et al., 2018).

This N_2O emission is associated with a large uncertainty on both the emission factors and the estimation of the total N input. Indeed, the estimation of $N_{excreta}$ mostly depends on the estimation of cattle dry matter intake which is difficult to estimate in continuously grazed pastures. In addition, some other terms of the equation are estimated using coefficients from the literature which are also associated with uncertainties. However, all these sources of uncertainties are probably lower than those associated to the emission factors themselves (Brown et al., 2001; Flechard et al., 2007).

One of the perspectives of this work would be to measure N_2O fluxes at the pasture scale in order to better quantify the weight of this flux in the pasture's GHG balance and to better understand its drivers (Voglmeier et al., 2018).

5.3. Pasture scale GHG budget

The GHG budget at the pasture is shown at figure 7-2. The results showed that no term of the budget is negligible when compared to the others, thereby highlighting the need to measure all three GHG when computing pasture scale GHG budget. The CO_2 sink activity of the pasture offset around 70% of the total N_2O and CH_4 emissions. The pasture therefore acted as a GHG source. This result is in agreement with multi-site studies that showed that C sequestration (or C emission in some cases) was an important component of European grasslands GHG balance (P. Ciais et al., 2010; Soussana et al., 2007). In the cited studies, grasslands could either act as GHG sources or sinks depending on management intensity and on other pedoclimatic conditions (Jones et al., 2017).

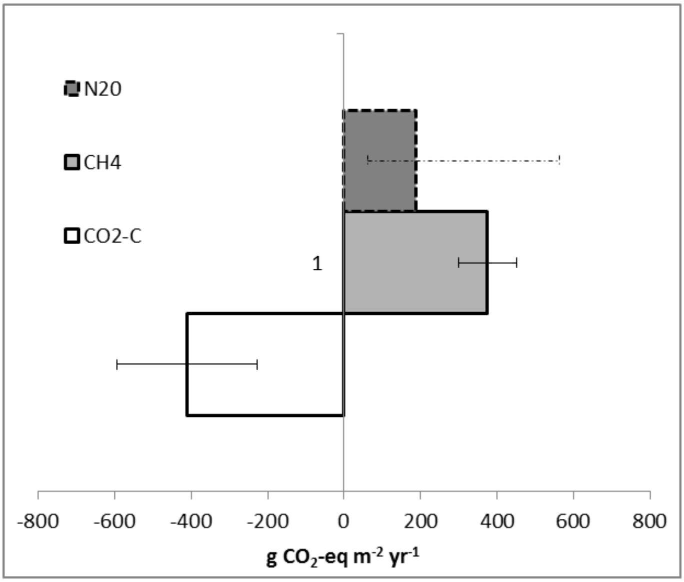

Figure 7-2 : Greenhouse gas budget of the pasture at Dorinne. All the values are given in g CO_2-eq m^{-2} yr^{-1} considering a global warming potential of 28 for CH_4 and 298 for N_2O. The emission of CO_2 amounts to -410 CO_2-eq m^{-2} yr^{-1}, the CH4 emission to 375 CO_2-eq m^{-2} yr^{-1} and the N_2O emission to 188 CO_2-eq m^{-2} yr^{-1}

In a more recent multi-site analysis from 14 managed grassland sites, Hörtnagl et al., (2018) found that every site acted as net GHG sinks, the only exception being a site that acted as a CO_2 and GHG source right after being ploughed for restoration. This study is however misleading because it focuses on CO_2 fluxes only, so that the reported CO_2 sink activity takes only into account the net CO_2 exchange between the pasture and the atmosphere (NEE) without considering other C exports through harvest, CH_4-C emissions or (at least not explicitly), animal CO_2 respiration. An important part of this absorbed CO_2 (evaluated from 91 to 1783 g CO_2 m^{-2} yr^{-1}) is stored in the biomass and will be, in major part, reemitted when eaten and respired. The CO_2 absorption mitigation potential is therefore (largely) overestimated.

Nevertheless, all these studies (including the present work) are only snapshots of the GHG balance at a given period that do not consider the fact that carbon sequestration is time limited and so is its GHG mitigation potential (Johnston et al., 2009; Smith, 2014). On the long term, under similar climate and with constant management, soil C stocks will reach equilibrium and no C sequestration will happen anymore. As an illustration purpose, this can be simulated by postulating that C sequestration rate (NBP) will decrease exponentially from now on and that no C sequestration will further happen after 100 years (Garnett et al., 2017). This time changing sequestration rate can then be compared with the methane emitted by a given stocking rate using a constant emission factor of 160 g CH_4 day^{-1} LU^{-1} measured at DTO (figure 7-3). Note that this simulation is only a schematic diagram to illustrate how C sequestration potential will decrease. It only accounts for CH_4

emissions and considers the same initial NBP for both stocking rates. When adding N_2O emissions, the C sequestration potential would be even lowered. The curve illustrates the limited mitigation potential of C sequestration when compared to CH_4 emissions and that, on the long term, pastures are not likely to act as continuous C sinks.

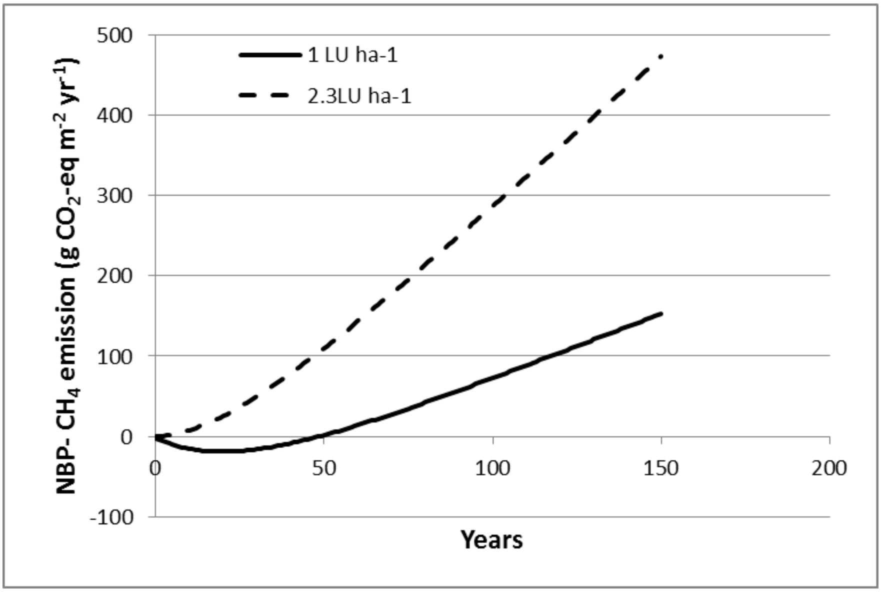

Figure 7-3 : Cumulative net greenhouse gas balance (CO_2-CH_4) in CO_2-eq between net C sequestration (NBP) and CH_4 emission for a stocking rate of 1 LU ha^{-1} yr^{-1} and 2.3 LU ha^{-1} yr^{-1} (stocking rate at DTO). Both curves consider an initial net biome productivity of 367 g CO_2-eq m^{-2} yr^{-1} and an emission rate of 58 kg CH_4 LU^{-1} yr^{-1}. Adapted from Garnet et al., (2017).

5.4. From pasture to farm scale

All the conclusions of this work concerning the C and GHG budget have been taken at the pasture scale which was the main focus of this study. As a result, some GHG fluxes of importance at the farm scale are not accounted for at the pasture scale so that some misinterpretation could be done trying to identify levers to mitigate farm scale GHG emissions. The different C and GHG fluxes involved in C and GHG budgets of mixed crop-livestock farm are illustrated in a schematic view at figure 7-4. We consider a farm that produces all the forage and additional feeds given to the animals on site. Fluxes associated to fuel consumptions, energy for the buildings and emissions associated with transport are not considered. This farm can be divided in three main components: the pastures, the barn and the crops. Note that in a real farm however, some C is very often imported into the farm from other exploitation mostly in form of feeds for the animals.

5.4.1. Carbon budget

The main fluxes involved at the pasture scale were fully described in the present manuscript. In mixed crop-livestock farms, the animals are fed in winter with pasture-produced forages and other feeds from crops. When computing NBP at the pasture scale, we make the assumption that the entire C contained in harvested forage is emitted within a year. This assumption is actually not totally true as some of the harvested C will be further contained in the manure produced at the barn that will further be spread on pastures and crops resulting in an off-site sequestration (Chang et al., 2015; Soussana et al., 2010). The real C sequestration might therefore be under or overestimated depending on fraction of grass based manure that is returned to the pasture. At our site, the real mitigation potential of the pasture might be slightly overestimated as the amount of manure spread on the pasture was slightly higher (14 g C m^{-2} yr^{-1}) than the harvested biomass, meaning that some the manure spread on the site came from other grasslands or crops. Note however, that at the farm scale, these fluxes associated with manure spreading are internal to the system so that C gained in some place should be lost elsewhere. As a result, if manure spreading can help to reduce soil C losses locally, this action is neutral when looking at the whole farm. The same statement can be made for supplemental feeds that are only a C transfer from the crops to the pasture. It’s also important to highlight the fact that, in this manuscript; the studied pasture was mostly used for grazing. Additional mowed grasslands are therefore needed in order to produce the needed forage stocks for winter.

To obtain a complete C budget at the farm scale, changes in SOC in crop fields should also be accounted for (figure 7-4). To do so, the easiest way would be to compute the C budget by considering only the C leaving or entering the farm (NEE_{past}, NEE_{crop}, $F_{product}$ and R_{cows} and CH_4-C at the barn and in the pasture). By doing so, many C fluxes are considered internal so that fewer components need to be estimated. However, this solution would not allow understanding the respective role of each ecosystem in the C budget of the farm which would be a major downside.

Another possibility is to compute the C budget of the farm as the sum of the NBP of the pastures and the NBP of the crops. By doing so, to avoid double accountings, the C lost in form of CO_2 respiration by the animals at the barn should not be accounted as it’s already done when accounting the C contained in the harvested biomass. The C budget of a crop field depends, in addition to the absorbed CO_2, on the amount of C contained in the harvested biomass (Buysse et al., 2017; Ceschia et al., 2010b). This harvested biomass can follow three pathways: a part of this biomass can be given to the animals at the barn or in the pasture as supplemental feeds, the remaining being exported to be further consumed. As a result, if one wants to compute the complete C budget associated with livestock production, only a portion of the NBP is attributable to the animals depending on the proportion of biomass that is effectively given to the animals.

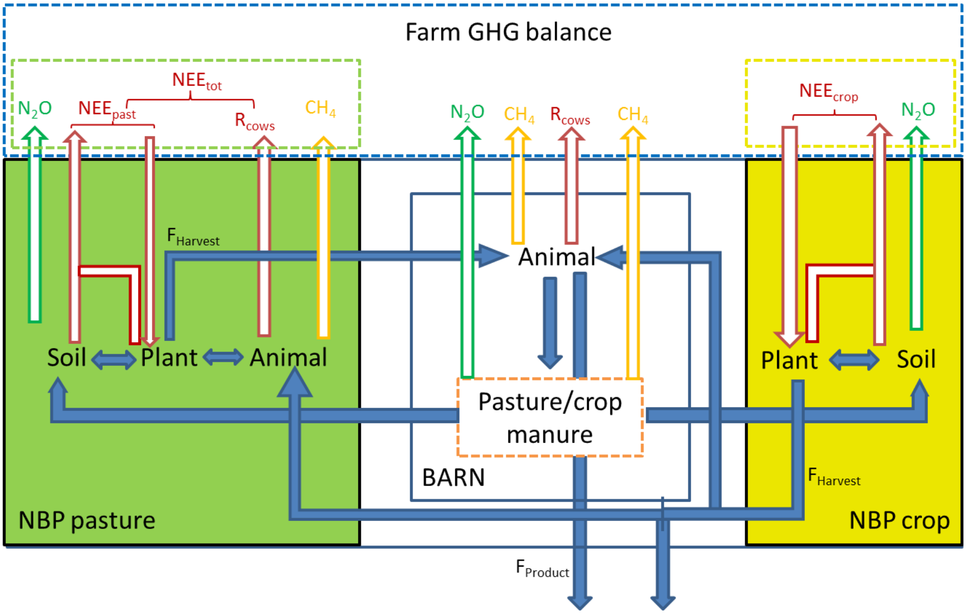

Figure 7-4 : Schematic representation of organic carbon (blue arrows), CO_2 (red arrows), CH_4 (yellow arrows) and N_2O fluxes involved at the farm scale (figure adapted from Chang et al., (2015)).

As perspective of this work, extending the C budget to the whole farm, including the crops needed to produce supplemental feeds as well as mowed meadows used to produce winter forage would allow better understand the relationships between the C cycles of each components and to develop integrated management practices to improve C cycling both in fields and pastures (Lemaire et al., 2014).

5.4.2. Farm scale greenhouse gas budget

Concerning the farm scale GHG budget, figure 7-4 highlights the different sources of GHG that needed to be included. The most direct way to compute the farm GHG budget is to compute only the GHG and C fluxes leaving or entering the farm. These are the CO_2 fluxes (NEE, and R_{cows}) in each components, the CH_4 emissions from animal enteric fermentation, CH_4 and N_2O fluxes form manure (Petersen et al., 2013), N_2O from pastures and crop fields as well as C exports in form of animal and vegetal products. However, as stated above, by doing so, the mitigation potential through soil C sequestration cannot be evaluated. To avoid this, the total GHG budget (NGB_{farm}) at the farm scale can be expressed as follow:

$$NGB_{farm} = \left(NBP_{past} - C\text{–}CH_{4,past} + NBP_{crop}\right) \times \frac{44}{12} + \left(CH_{4,past} + CH_{4,barn}\right) \times GWP_{CH4} + \left(N_2O_{past} + N_2O_{barn} + N_2O_{crop}\ \right) \times GWP_{N2O}$$

where *GWP* is the global warming potential of the concerned gas . Because only a fraction of the harvested crops is given to the animals, only a fraction of the crop related fluxes should be accounted when computing the GHG budget associated with livestock production.

Computing this complete GHG budget would require lots of additional measurements that are labor and cost intensive. However, with the development of fast N_2O gas analyzers, it is now technically possible to measure all the three GHG fluxes (Baldocchi, 2014) both in pastures (Hörtnagl et al., 2018; Merbold et al., 2014) and crop sites using the EC technique. At the barn, CH_4 fluxes from enteric fermentation can be measured using different techniques such as the SF6 method and closed chambers (Hammond et al., 2016) or estimated from feed intakes (IPCC, 2006b). To complete this GHG budget, CH_4 and N_2O emissions from manure during storage should also be computed. This would allow computing a complete GHG budget at the farm scale and, moreover, to evaluate livestock related GHG emissions. The gathered data would also help to build a complete life cycle assessment of cattle production including soil C sequestration which can be an important component. Indeed, when comparing grass-based to confinement dairy system, (O' Brien et al., 2014) showed that grass based systems had the lowest carbon footprint when including carbon sequestration while omitting it resulted in similar carbon footprint.

6. Conclusion and perspectives

In this work, soil C storage in an intensively managed pasture grazed by Belgian Blue heifers was studied by building a complete C budget (NBP) at the pasture scale by combining eddy covariance fluxes to other non CO_2 C imports and exports estimates. A specific attention has been paid to some methodological aspects in order to compute unbiased C budgets. The main findings of the work are the followings:

- In order to avoid biased annual NEE estimates, a local cospectrum should be used instead of a universal one when correcting the CO_2 fluxes for high frequency losses.
- The CO_2 flux exchanged between the pasture and the atmosphere should be partitioned between the exchange of the vegetation (NEE_{past}) and grazing animal's respiration using, for example, CH_4 fluxes. If there is evidence that the cows are not distributed evenly on the pasture on a yearly basis, both components should be computed separately to avoid biased CO_2 flux estimates.
- Once these two major systematic errors were corrected, the remaining uncertainties on NBP was estimated to be ≈50 g C m-2 yr^{-1}.
- The annual NBP amounted to -100 ± 50 g C m^{-2} yr^{-1} which show that, over 5 years, the pasture acted as a significant C sink.
- A paired tower experiment showed that no significant difference in term of NEE should be expected between a rotationally and continuously grazed pasture assuming similar stocking rates.
- The soil C sequestration in the pasture was compared to the cattle CH_4 emission estimated from combined GPS-eddy covariance CH_4 fluxes and to

tier 1 IPCC N_2O emission estimates. The result showed that, in term of GHG exchanges, C sequestration compensated around 65% of the CH_4 and N_2O emissions at the pasture scale.

In addition to this work, the following perspectives were highlighted:

- Concerning the uncertainty assessment, further work is needed in order to better understand and take into account the possible correlations between the different sources of uncertainties.
- Crossing long term soil C sequestration estimated from C budgets with values estimated from direct soil samplings would strengthen our confidence in soil sequestration estimates.
- The impact of management practices on C and GHG budgets was tested in this work by using paired tower experiments.The method appeared promising but only two management practices (continuous vs rotational grazing) were tested. More practices could be investigated thanks to this method. . In particular, the impact of pasture renovation, grazing and fertilization intensity, increased species diversity as well as mowing compared to grazing could be assessed using these experiments. This approach can also be extended to study the impact of these practices on N cycling and N_2O emissions.
- Directly measuring the N_2O emitted in the pasture would allow to improve the GHG budget at the pasture scale and to better quantify the total GHG emissions associated at the pasture scale. Some management practices, such as N fertilization, may increase C sequestration but also increase N_2O emission. Measuring both fluxes (also during paired tower experiments) would allow better quantifying the real benefits or losses in term of GHG budgets.
- Confronting our data to existing biogeochemical mechanisitic ecosystem scale models such as PaSim (Calanca et al., 2007), cenw (Kirschbaum et al., 2015), or others (Del Prado et al., 2013) would allow a better understanding of the effect on the carbon cycle of weather conditions, grazing and management and extreme events, such as drought or heatwaves . It would aslo allow us to check the consistency of our conclusions in regards to state of the art models
- In mixed crop-livestock farms, the animals are fed with feeds coming from crop production, pastures and mowed meadows. In term of C flows, these components are interconnected through manure management and animal nutrition. If lots of studies have been carried out to quantify and understand C cycling in crops and pastures separately, integrated studies that take into account the interconnections between crops, pastures and the barn are highly needed. Doing so would allow better understanding of the C transfer from the crops to the pastures and vice versa.
- The GHG budget was quantified at the pasture scale. Extending this GHG budget to the farm scale, including GHG emitted at the barn and in crop fields is highly needed to quantify the total GHG gas emissions associated

with cattle production. These data could further be used in life cycle assessments to fulfil this last objective.

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