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Contribution of soil organic carbon variations to the life cycle analysis of a farm: case study in Belgium and methodological recommendations

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Travail de fin d'études présenté en vue de l’obtention du diplôme de master bioingénieur en sciences agronomiques.

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Contribution de la variation du carbone organique du sol à l'analyse du cycle de vie d'une ferme:

étude de cas en Belgique et recommandations méthodologiques

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Abstract

Agricultural systems are facing numerous environmental challenges. Increasing soil organic carbon (SOC) content has been suggested as a means to both mitigate and adapt to some of these challenges. Methods assessing environmental impact such as life-cycle assessment (LCA) can provide estimations of the potential of such levers. Yet, there is currently no consensus as to whether and how to account for SOC variation in agricultural LCA. In this study, an LCA of a crop-livestock farm located in southern Wallonia, Belgium was performed out of 16 environmental impact categories. The impacts induced by the entire production of the farm in 2021 as well as the relative contribution of the different farm's processes have therefore been quantified. Furthermore, a range of the farm's SOC variation was estimated using data from soil sampling campaigns as well as from two stations of the *Integrated Carbon Observation System* (ICOS) network, one of these stations being located at the studied farm. The results show that the climate impact of the farm's animal production amounts to 10,2 kg of CO_2-equivalent per kg of liveweight and the associated SOC variation ranges between increasing this footprint by 2% and offsetting it by -22%. This study also highlights the challenges faced with integrating SOC variation into agricultural LCAs.

Résumé

Les systèmes agricoles sont confrontés à de nombreux défis environnementaux. L'augmentation de la teneur en carbone organique des sols (COS) a été suggérée comme un moyen d'atténuer et de s'adapter à certains de ces défis. Les méthodes d'évaluation des impacts environnementaux telles que les analyses en cycle de vie (ACV) peuvent fournir des estimations du potentiel de ces leviers. Cependant, une absence de consensus persiste actuellement sur la question de savoir si et comment prendre en compte les variations du COS dans les ACV agricoles. Dans cette étude, une l'ACV d'une exploitation en polyculture-élevage située dans le sud de la Wallonie, en Belgique a été réalisée sur 16 catégories d'impacts environnementaux. Les impacts induits par l'ensemble de la production de l'exploitation en 2021 et la contribution relative des différents processus de l'exploitation ont ainsi pu être quantifiés. En outre, une gamme de variation du COS de l'exploitation a été estimée à l'aide de données provenant de campagnes d'échantillonnage du sol ainsi que de deux stations du réseau *Integrated Carbon Observation System* (ICOS), dont l'une d'entre étant située sur l'exploitation agricole étudiée. Les résultats montrent que l'impact climatique de la production animale de l'exploitation s'élève à 10,2 kg d'équivalent CO_2 par kg de poids vif et la gamme de variation de COS associée varie entre une augmentation de 2% et une compensation de -22% de ce chiffre. Cette étude met également en évidence les défis posés par l'intégration de la variation du COS dans les ACV agricoles.

Table of content

List of acronym

- BE: Belgium
- CW: Carcass Weight
- CO_2e: CO_2 equivalent
- DM: Dry matter
- DMI: Dry matter intake
- EF: Emissions Factors
- FM: Fresh matter
- FU: Functional Unit
- GHG: Greenhouse gases
- GWP: Greenhouse Warming Potential
- ICOS: Integrated Carbon Observation System
- IPCC: Intergovernmental Panel on Climate Change
- LCA: Life cycle Assessment
- LU: Livestock Unit
- LUC: Land Use Changes
- LMC: Land Management Change
- LW: Live Weight
- LWG: Live Weight Gain
- NIR: National Inventory Report
- SC: Soil Carbon
- SIC: Soil Inorganic Carbon
- SOC: Soil Organic Carbon
- SOM: Soil Organic Matter

Introduction

Context

Agricultural systems are facing numerous environmental challenges. On the one hand, they contribute to the crossing of planetary boundaries such as climate change, biosphere integrity and biogeochemical flows (Wang-Erlandsson *et al.,* 2022). The conversion of land for agriculture as well as the intensification of farming practices both lead to these boundaries overrun (Campbell *et al.,* 2017).

On the other hand, agricultural systems are threatened by these environmental constraints. In particular, the erosion of biodiversity, climate change and resource depletion seriously impair agricultural sustainability (FAO, 2021). For instance, agricultural systems are highly dependent on fossil fuels whose non-renewable supplies are under high pressures (IEA, 2018, Shift Project, 2021). Other resources such as phosphate rock, whose fertiliser role for agriculture is vital, are also critically threatened (Miller *et al.,* 2023).

Soil carbon (SC), is yet another environmental indicator of critical importance for agriculture. It can further be separated between soil inorganic carbon (SIC) and soil organic carbon (SOC). In particular, SOC is essential to soil health and fertility (Lal, 2016), although the exact effect of increased SOC on agricultural yield has been found to be inconsistent (Moinet *et al.,* 2023). Even though exact quantities are difficult to measure, human activities have caused massive losses of SOC (Ruddiman, 2005) through mechanisms that are further described in this document.

In order to quantify the above mentioned environmental impacts, methodological tools have been developed such as Life-Cycle Assessment (LCA). The aim of an LCA is to assess the environmental impacts of a product throughout its entire life-cycle, from the extraction of primary resources, the manufacturing and use processes, to the disposal and/or recycling of the product. The principles and frameworks required to perform such assessments are defined by the international standards ISO 14 040 and 14 044 (Saling, 2006).

LCA are performed on a large number of environmental impact categories simultaneously in order to avoid the "*burdens shifting*" effect. Indeed, whenever only one environmental impact category is assessed, pathways for mitigation risk to shift the burdens to other impact categories. For instance, when climate change is the only impact category assessed, mitigation measures can shift the burden toward other impact categories such as metal depletion potential (Algunaibet and Guillén-Gosálbez, 2019). The systemic view enabled by LCA is especially important in this context as agriculture sustainability is characterised by a high level of transdisciplinary (Francis *et al.,* 2008).

In order to both mitigate and adapt to climate change and to the loss of SOC, the international "*4 parts per 1 000*" initiative, subtitled "*Soils for Food Security and Climate",* was launched in 2015 during the COP21 (ADEME, 2015). The initiative's title refers to the so-called capacity of the world's soils to offset the increasing atmospheric CO_2 concentration if their carbon content were to increase by "*0,4 %*" or "*4 parts per 1 000*". Although there is a clear scientific consensus that bringing back carbon into the soils can lead to numerous agronomic benefits, the climate change mitigation potential suggested by this initiative has been subject to a large number of debates and controversies (Baveye *et al.,* 2018).

In this context, environmental tools such as life-cycle assessments could be sought after in an effort to assess the potential of such initiatives. Yet, while LCAs have extensively been performed on agricultural products (Alhashim *et al.,* 2021), there is currently no consensus as to whether and how to account for soil carbon variation in agricultural LCA (Goglio *et al.,* 2015).

Objectives

At the scale of Wallonia, Southern Belgium, agricultural SOC has been the subject of extensive monitoring through successive soil sampling campaigns across the entire region. More recently, three eddy covariance flux towers, continuously monitoring ecosystem exchanges, were installed in Wallonia. These flux towers allow for a complete ecosystem carbon balance to be performed using a method described further in this document. Two of these towers are settled on agricultural sites: one in a cropland located in Lonzée in the Loamy agricultural region, and the other one in a permanent grassland of Dorinne, in the Condroz agricultural region. These flux towers are part of a larger European network of carbon observation systems called *Integrated Carbon Observation System (ICOS)* (ICOS, 2023).

Lonzée and Dorinne's ICOS stations are managed by the faculty of Gembloux Agro Bio-Tech who strongly collaborates with the farmers managing the associated lands. Several studies have been published in this context (Gourlez 2019, Lognoul 2020, Dumortier *et al.,* 2021). Amongst them, Gourlez (2019) have estimated that between 2010 and 2015, Dorinne's ICOS site was acting as a carbon sink sequestering -1 000 ± 500 kg C/ha/yr. Furthermore, Gourlez (2019) established a greenhouse gases (GHG) budget of Dorinne's ICOS site, consisting of 4,2 hectares of cattle grazed permanent grassland, and thus estimated that the carbon sequestration occurring at this site offsetted 65% of the site's GHG emissions. One prospect of the work of Gourlez (2019) is to extend this GHG budget from this 4,2 hectare of pasture scale to an entire farm scale GHG budget of Dorinne's farm.

In this context, the objectives of this master thesis are to 1) perform a Life-Cycle Assessment of Dorinne's farm (referred to as "the studied farm" in this document) 2) characterise the SOC variation associated with the studied farm and 3) elaborate on the contribution of this SOC variation to the studied farm's climate change impact.

Literature review

Soil organic carbon variation

Estimation methods

Observations of SOC variation are obtained through two main kinds of estimation methods: soil sampling analysis, and carbon balance approach.

Soil sampling analysis

The IPCC (2006) provides guidelines to measure SOC stocks. These recommend to perform the measurements at a default depth of 30 cm and to exclude both organic carbon in surface residues and changes in inorganic carbon. Stocks are then computed by multiplying the SOC content, defined as the mass of carbon per mass of soil (expressed in [kg C / kg soil] or in [%C]) by the depth increment (default is 30 cm), bulk density, and the proportion of rock fragment (i.e., > 2mm fragments) in the depth increment as shown by equation 1 below retrieved from Chartin *et al.* (2022):

(Eq 1.) $$SOCst.30 = SOC.30 * d * BD * (1 - RM)$$

With $SOC_{st.30}$: SOC stocks in the 0-30 cm layer [tonne of C / hectare], $SOC_{.30}$: SOC content in the 0-30cm layer [%C], d: soil depth (cm; 30 cm here), BD: Bulk Density [g/cm^3] and RM: Rock fragment content by mass.

The SOC content, the bulk density and the proportion of rock fragments can be determined by laboratory analysis on soil samples. In particular, bulk density is defined by Buckman and Brady (1960) as "*the mass of the many particles of the material divided by the bulk volume*." The bulk volume being further defined as "*the total volume the particles occupy, including the particle's own volume, interparticle void volume, and the particles' internal pore volume.*" (Buckman and Brady, 1960)

The monitoring of $SOC_{st.30}$ trends through time allows estimations of the annual variation of $SOC_{st.30}$ in agricultural soils. In order to ease the reading of this document, this annual variation of $SOC_{st.30}$ is simply referred to as *SOC variation* and expressed in [kg C / ha / year] in the rest of the document. It is also important to note that multiplying this value by $\frac{44}{12}$ allows to express SOC variation in [kg CO_2 / ha / year] which allows to compare SOC variation to other sources of CO_2 emissions. The ratio $\frac{44}{12}$ refers to the ratio of molar mass of CO_2 (=44 [g / mol]) and of the molar mass of C (=12 [g / mol]).

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 order to compute a complete carbon balance using this method, the following equation 2 is used:

(Eq 2.) $$NBP = NEE + Cimp + Cexp$$

With *NEE* corresponding to the difference between the rate at which carbon (CO_2) is fixed through photosynthesis, and released through total ecosystem respiration. It is usually measured using the eddy covariance technique.

Cexp refers to the non-CO_2 exports such as the carbon exported through harvest, leaching, CH_4 emission and meat production (in the case of grazed pasture).
Cimp refers to the non-CO_2 carbon imports such as manure and feed supplement (in the case of grazed pastures).

Classification and prospects

Soil sampling analysis and the carbon balance approach result in *Observations* that show a high level of certainty but a low level of applicability as shown in figure 1 below.

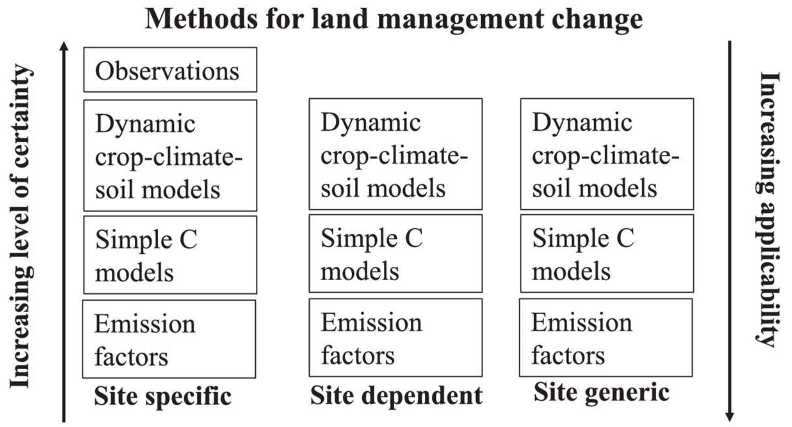


Figure 1: Summary of methods to account for soil C change due to land management change in agricultural LCAs. Source: Goglio *et al.*, 2015.

The IPCC have hierarchised the methods in figure 1 as “Tier 1” methods for those with the highest applicability and lowest certainty, “Tier 3” for the opposite extreme and “Tier 2” for the intermediate methods.

Tier 3 methods correspond to the *Observations* obtained using the two methods described above (soil sampling and carbon balance). Tier 1 methods consist in the use of emission factors that are issued from the current literature state. The use of soil carbon models corresponds to Tier 2 methods, RothC, C-TOOL, DAYCENT are commonly used for soil carbon modelling.

At the European scale, a large number of soil sampling campaigns are available as well as the continuous flow of data from the ICOS system previously mentioned. Recently, the European Union has been working on developing tools which could combine these SOC observations with remote sensing data and with the use of soil carbon models to achieve real-time SOC monitoring across the entire European territory (European Commission, 2023). This type of approach is developing in the rest of the world as well (CSIRO, 2023).

Estimates

In Wallonia, SOC variation is being monitored through both soil sampling and carbon balance approaches. Table 1 below, retrieves some of the studies published on this matter, as well as the SOC variation values associated. In this table, as well as in this entire document, the following convention is followed: positive values correspond to source activity (loss of SOC), negative values correspond to sink activities (SOC sequestration).

Reference	Methods	Time span	Agricultural region	Crop lands [kg C / ha / yr]	Grasslands [kg C / ha / yr]
Buysse *et al.* (2017)	Carbon balance	2004-2016	Loamy	825 ± 540	NA
Gourlez *et al.* (2016)		2010-2015	Condroz	NA	-1 000 ± 500
Chartin *et al.* (2022)	Soil sampling	2005-2019	Wallonia	12,5	220
			Loamy	NS	887,5
			Condroz	-176	67,5
Chartin *et al.* (2019)		1949-2014	Wallonia	66	-302
			Loamy	81	-570
			Condroz	42	-360
Goidts and van Wesemael (2007)		1955-2005	Wallonia	110	-420
			Loamy	332	-396
			Condroz	110	-478

Table 1: SOC variation estimation of croplands remaining croplands and grasslands remaining grasslands in Wallonia.
NA: Not applicable. NS: Non-Significative difference.

Table 1 shows that Wallon cropland saw their SOC content lower over time. More precisely, a soil sampling campaign which occurred in Wallonia between 2015 and 2019, found that 90% of Walloon crop lands showed a SOC content lower than 20 g of C / kg of soil (Chartin *et al.,* 2022). Soils with SOC contents below the 20 g of C / kg of soil threshold present an increased risk of structural instability, with potential consequences in terms of sensitivity to erosion (Shi *et al.,* 2020). Therefore, it is fundamental to monitor SOC content overtime in order to assess these issues.

In this regard, every country has to monitor their domestic GHG emissions, including SOC variation, and to provide a yearly report of their entire GHG inventory to the IPCC from the year 1990 onwards. The IPCC (2006) provides guidelines on the methodology to properly perform this reporting. The SOC variation is reported for land use change (LUC) (e.g. : grasslands converted to croplands) as well as for land use (LU) (e.g. : grassland remaining grassland) (IPCC (2006), volume 4, ch.4-9).

Table 2 below indicates the SOC variation values reported in the National GHG Inventory Reports (NIR) of Belgium (BE) from 2018 until 2023 (BE-NIR). Values used for Flanders have remained the same for this period of time. However, in Wallonia, a recent update from Chartin et al. (2022) has been taken into account for the last two Belgians reports (2022 and 2023).

[kg C / ha / yr]	Wallonia		Flanders	
	Crop lands	Permanent grasslands	Crop lands	Permanent grasslands
BE-NIR22 and BE-NIR23	2004 - 2021= 12,5 1990 - 2004= 66	2004 - 2021 = 220 1990-2004 = 0	16	19
BE-NIR18 until NIR21	66	-302		
Reference	Chartin *et al.* (2019 et 2022)		Meersmans *et al.* (2010)	

Table 2: Belgium's NIR evolution on SOC variation of croplands remaining croplands and grasslands remaining grasslands.

Many studies have estimated grasslands SOC variation in the rest of the world as well. Hence, as part of a meta-analysis, Garnet *et al.,* (2017), found a large spectrum of values amongst studies, ranging from -30 to -1 040 kg C / ha / yr. Strikingly, Jones and Donnelly (2004) found an even wider spectrum in yet another meta-analysis, with SOC variation from temperate grassland reaching more than -8 000 kg C / ha / yr. Multiple drivers can explain such a wide range of values, these SOC variation drivers are the focus of the next section below.

Drivers

Figure 2 below shows that SOC dynamics is influenced by many factors which are reviewed in this section.

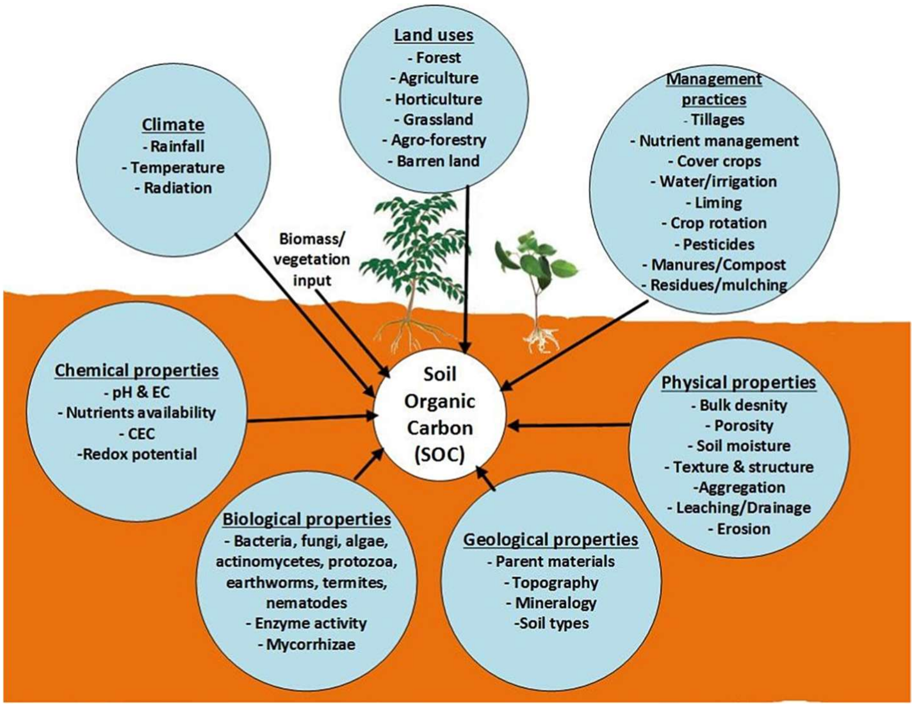


Figure 2: Schematic diagram of the factors influencing organic carbon dynamics in soil. (Ramesh *et al.* 2019).

Land Use Changes (LUC)

Pellerin et al., (2019) showed that SOC content amounts for 81 tonnes of C / ha for forest lands, 84,6 tonnes of C / ha for permanent grasslands and 51,6 tonnes of C / ha for crop fields. The conversion of crop lands to permanent grasslands or forest induces SOC gain, whereas the opposite conversion as well as artificialization of soils lead to SOC loss. For instance, when converting degraded cropland to permanent grassland, Machmuller *et al.,* (2015) reported mean sequestration values as high as -8 000 ± 850 kg C/ha/yr in the top 30 cm of soil for 7 years.

Land Management Changes (LMC)

SOC variation can also occur under constant land use. Goidts and Van Wesemael, (2007) showed that crop lands in Wallonia saw their SOC content decrease from 46,2 tonnes of C / ha in 1955 to 40,6 tonnes of C / ha in 2005. They highlighted agricultural management as the predominant factors involved in this SOC loss.

It is established that fertilisation plays a key role in carbon sequestration of grassland (Soussana et Lemaire, 2014). Pellerin *et al.,* (2019) found that a moderate increase in the N-fertilisation of French grasslands could induce -176 ± 63 kg C / ha / yr of additional SOC sequestration. However, the moderate increase in N-fertilisation leads to additional N_2O emissions, this aspect is discussed further in this document.

Animal management also plays a key role in grassland SOC sequestration through factors such as animal density (stocking rates), plant harvest methods (cut or grazed) and grazing methods (Jérôme *et al.,* 2014). For instance, converting grasslands under continuous grazing to rotational grazing, also known as adaptive multi-paddock (AMP), could lead to additional carbon sequestration in the United States Southern Great Plains according to Wang *et al.,* (2015). Furthermore, grasslands under neither cutting nor grazing management could sequester more carbon than managed ones (Fitter *et al.,* 1997). Indeed, high biomass removals have been found to limit the carbon sequestration potential of mature temperate grasslands (Skinner, 2008).

Changing agricultural management could therefore reverse this trend. Pellerin *et al.,* (2019) showed that 86% of France's SOC sequestration potential lies in croplands, which is mainly due to the fact that current SOC content of France's croplands is very low. They identified several possible land management changes (LMC) that could achieve this potential, with the main ones being: the extension of intermediate crops (36% of total potential), intra-parcel agroforestry (20% of total potential) and the insertion and extension of the time of presence of temporary grasslands (13% of the total potential). Pellerin *et al.,* (2019) also revealed that if all these LMC were implemented at the French territory scale the induced additional sequestration would amount to 8,15 Millions of tonnes of C / year (or 29,9 Millions of tonnes of CO_2/ year). This value would stand for 6,5% of France's national GHG emissions and 39% of France's agricultural GHG emissions (energy use and land use changes excluded).

<u>Other drivers</u>

Besides LUC and LMC, chemical, biological, geological and physical soil properties can influence SOC variation (Ramesh *et al.,* 2019). Climate can also influence SOC dynamics, for instance, increasing temperatures, resulting from climate change, will, among other things, stimulate microbial activity, which is expected to decrease the amount of soil organic matter (Crowther *et al.,* 2016).

<u>Agricultural LCA</u>

A large number of agricultural LCAs have been performed; an extensive literature review of these has been done by Alhashim *et al.,* (2021). This section focuses on bovine meat and wheat LCAs as these are the main production of the studied farm.

In the previous section on SOC variation, climate change was the main environmental impact category mentioned. In this section, further environmental impact categories are included. Life cycle impact assessments are conducted in regards to three main general categories of environmental impacts: resource use, human health and ecological consequences (Saling, 2006). Each of these three general categories include a large number of sub-categories such as greenhouse warming potential (GWP), terrestrial acidification potential (TAP), human toxicity potential (HTP) which are discussed in this section.

It is important to note that no consensus exists amongst the scientific community as to how to account for soil carbon variation in agricultural LCA (Goglio *et al.,* 2015). It is up to the author of a given LCA to state whether he assumed the associated agricultural soils to have reached carbon equilibrium (ΔSOC = 0) or if he accounted for a SOC variation in his study. Concerning the studies reviewed in this section, the current document precises for each of these whether SOC variation is taken into account or not.

Bovine meat production

In terms of greenhouse gases warming potential (GWP), world's emissions from the livestock sector amount to 7.1 Gigatons of CO_2-equivalent per year, representing 14,5% of total anthropogenic emissions. Cattle emissions contribute to 65% of the livestock sector (Gerber *et al.,* 2013). In Belgium, emissions from the livestock sector amount to 13,9 Megatons of CO_2-equivalent per year, representing 12% of the national emissions. Dairy and bovine meat respectively contribute to 33% and 23% of the livestock sector (Riera *et al.,* 2019).

Furthermore, as part of a meta-analysis on the global food chain including 38 700 farms worldwide, Poore and Nemecek (2018) estimated the carbon footprint of beef meat (from beef herds) to amount to 60 kg of CO_2e per kg of beef meat on average. The impacts were attributed as follows: 27% to land use change, 66,5% to farm emissions (including enteric fermentation), 2,5% to animal feed, 2,5% to processing and less than 1,5% to transport, retail and packaging combined (Poore and Nemecek, 2018). This study also showed that the carbon footprint of beef meat from dairy herds is three times lower on average, as the emissions are partially allocated towards dairy production.

Beyond these global values, it is important to highlight that environmental impacts of beef production vary largely among the different production systems (de Vries and de Boer, 2010, National Trust, 2012) as shown on figure 3 below.

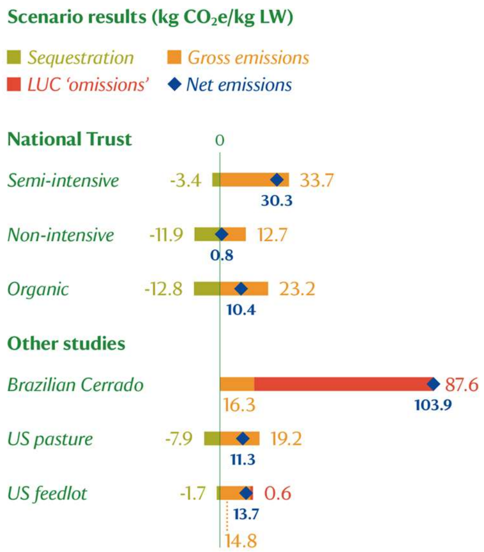


Figure 3: Greenhouse gas emission intensities of different ruminant production systems. Source: National Trust (2012). *CO_2e : CO_2 equivalent, LW: Live Weight*

Figure 3 distinctly separates the contribution of the LCA (in orange) and of SOC variations (in green or red). It also shows the values when both of these categories are simultaneously taken into account (in blue). Concerning the latter, a wide spectrum of values ranging from 0,8 kg CO_2e / kg LW to up to 103,9 kg CO_2e / kg LW are retrieved in figure 3, depending on the production systems. Therefore, it is important to narrow down the environmental impacts to the specific case of the Walloon Belgian Blue production system, which is the one considered in this study. The Belgian Blue is a specialised beef breed typical of the Walloon system.

In this regard, LCAs have been performed on the Walloon Belgian Blue production system. Table 3 retrieves the results of one of these studies (Kokemohr *et al.,* 2022). It is important to mention that this study does not consider SOC variation (ΔSOC = 0).

Functional Unit	GWP (kg CO_2eq)	FEP (g Peq)	MEP (g Neq)	PMFP (g PMeq)	TAP (kg SO_2eq)	FDP (MJ)
Kg of LW	22,6	4,75	34,0	51,0	0,28	14

Table 3: LCA of a Belgian Blue beef production system (Kokemohr *et al.,* 2022)
GWP: Greenhouse warming potential, FEP: Freshwater eutrophication potential, MEP: Marinewater eutrophication potential, PMFP: Particulate matter formation potential, TAP: Terrestrial acidification potential, FDP: Fossil fuel depletion potential.
[1]: Fattening phase only.

Rieria *et al.,* (2023) assessed environmental impacts of 128 specialised Walloon beef farms and found an average carbon footprint of 7,6 kg CO_2e / kg LW for the types of farm that are similar to the one studied in this paper. Mathot *et al.,* (2016) also estimated the carbon footprint of Belgian Blue beef production, but for the fattening phase only, and found an average of 8,5 kg CO_2e / kg of LW. Both of these two studies did not account for SOC variation (ΔSOC = 0).

Wheat production

Table 4 below shows the results of the life cycle impact assessment of 1 hectare and 1 kg of fresh matter (FM) of a typical Walloon conventional wheat (entire plant) production (Van Stappen *et al.,* 2015). This study did not account for SOC variation (ΔSOC = 0).

FU	GWP (kg CO_2eq)	EUP (10^{-3} g Peq.)	TAP (10^{-3} g SO_2eq)	AEP (10^{-3} CTUe)	HTP (10^{-3} CTUh)	CED (10^{-3} MJ)
1 kg FM	0,349	0,002	0,015	0,007	0,150	0,002
1 ha	2 991	19,4	141,7	59,4	1 283	163,6

Table 4: LCAs of a Walloon conventional wheat production (Van Stappen et al., 2015).
GWP: Greenhouse warming potential, EUP: Eutrophication potential, TAP: Terrestrial acidification potential, AEP: Aquatic ecotoxicity potential, HTP: Human toxicity potential, CED: Cumulative energy demand.

Contribution of SOC variation to cattle farming LCA

This section reviews studies that have estimated the contribution of SOC variation to agricultural LCA. Although some researchers consider soil organic matter dynamics as a separate impact category that can be used to assess soil quality change (Brandao et al., 2011), this section, as well as the rest of this document, focus on the contribution of SOC variation to the climate change impact category exclusively.

Chang *et al.,* (2021) showed that, at a worldwide scale, climate warming from managed grasslands reduces the climate mitigation potential of carbon sinks in sparsely grazed and natural grasslands as shown in figure 4 below.

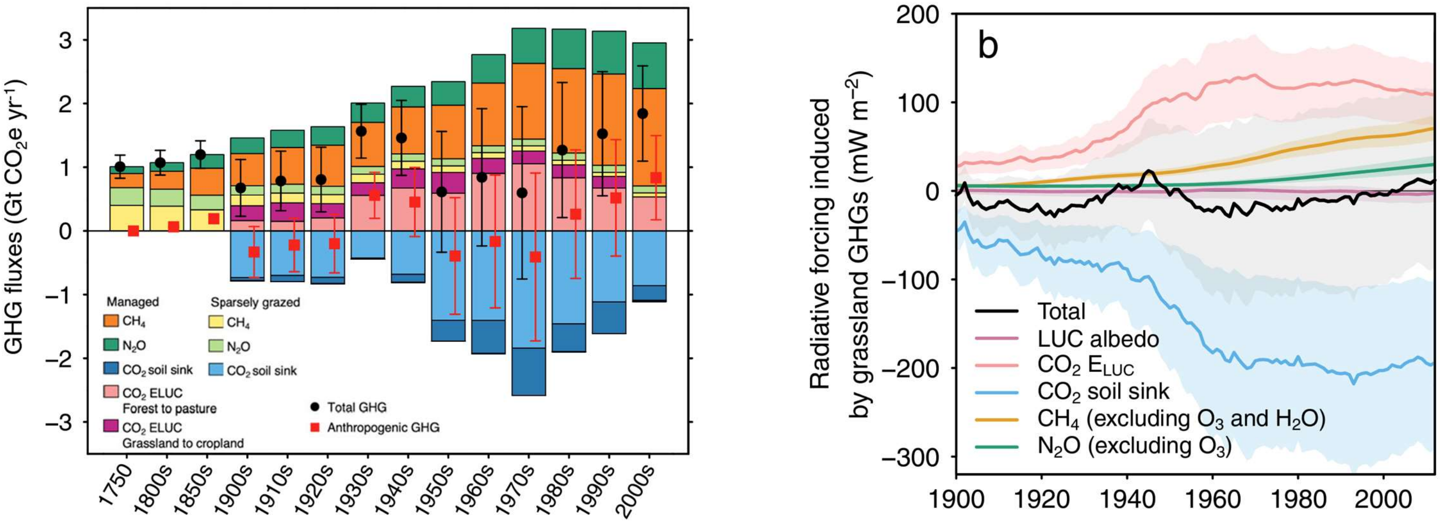


Figure 4: GHG and radiative forcing balances from worldwide grassland.
Source: Chang *et al.,* (2021)

Garnet *et al.,* (2017) also studied the issue at global scale, but with more recent values, and found similar estimates; with world grasslands SOC sequestration potential ranging from -0.3 to -0.8 Gt CO_2 / year thus offsetting 20 - 60% of emissions from grazing systems, 4 - 11% of total livestock emissions, and between 0.6 and 1.6% of total annual greenhouse gas emissions.

Beyond these global scale estimates lies a large number of studies who provide such kinds of estimates but at farm scale. Table 5 below retrieves some of these studies which have estimated the contribution of SOC variation to the climate impact of bovine meat production. It is important to mention that these studies have all estimated SOC variation separately from the climate impact. In other words, the column *"Climate impact"* of table 5 does not account for SOC variation. In table 5, SOC variation is retrieved in its own *"SOC variation"* column and the contribution of the latter on the former is shown in the column labelled *"Offset in %".* Furthermore, it should be noted that the livestock unit (LU) is a reference unit which facilitates aggregation of livestock from various species and ages. The stocking rate refers to the number LU on a given amount of land over a certain period of time.

Study reference (Country)	**Climate impact** (kg CO_2e / kg LW / yr)	**SOC variation** (kg C / ha / yr)	**Stocking rate** (LU/ ha/ yr)	**Offset in %**
Hammar *et al.*, 2022 (Sweden)	11,34	-200	1,02	16 %
Reyes-Palomo *et al.*, 2022 (Spain)	20	-974	0,51	68 %
Stanley *et al.*, 2018 (USA)	5,77	-3 590	2,7	169 %
Beauchemin *et al.*, 2011 (Canada)	13,04	-649	0,06	536 %

Table 5: Studies on the contribution of SOC variation to the climate impact of bovine meat production.

Table 5 demonstrates that grasslands can offset beef-related GWP by a wide spectrum of values ranging from 16% to 536%. The study that found the highest offset (i.e. 536% by Beauchemin *et al.* 2011) is not the one with the highest carbon sequestration rate. In the case of Beauchemin *et al.* (2011), the very low stocking rates (i.e. 0,06), which is common to this type of beef production in Canada, can explain this large offset. The significance of this "land effect" has also been demonstrated in a meta-analysis by Poore and Nemecek (2018) who found that improved pasture management can sequester carbon, but it offsets life-cycle ruminant emissions by a maximum of 22%, with greater sequestration requiring more land.

Materials and methods

Description of the studied farm

The studied farm is located in Dorinne, in the agricultural region of Condroz, in Wallonia, southern Belgium. The farm contains a barn for a herd of Belgian Blue cattle and two other buildings used as sheds for agricultural machinery, materials, crops, feed and manure management. The farm consists of 121 ha of lands scattered around these buildings and 30 ha of lands in Tourinne, located in the Loamy agricultural region, 47 km away from Dorinne. The 2021 crop distribution of these 151 ha of lands is detailed in table 6 below and can be visualised in a map of Dorinne's lands and a map of Tourinne's lands, respectively figure S1 and S2 of the supplementary materials, at the end of this document.

Localisation	Type of crops	Area (ha)
Dorinne (Condroz region)	Permanent grasslands	53,33
	Temporary grasslands	2,17
	Maize silage	7,79
	Winter rapeseed	10,44
	Sugar beetroot	7,34
	Forage beetroot	0,96
	Winter wheat	14,16
	Winter spelt	4,84
	Winter triticale	1,85
	Winter barley	18,04
	Subtotal	120,92
Tourinne (Loamy region)	Sugar beetroot	1,39
	Carrots	6
	Potatoes	4
	Peas	6,13
	Winter wheat	12,75
	Subtotal	30,27
	Total	151,19

Table 6: Farm's crop distribution 2021.

The herd consists of 233 Belgian Blue heads with 95 calvings per year. The 53,33 ha of permanent grasslands are both used for grazing the animals and for harvesting grass.

The farmer never proceeds to monocropping on the lands he manages. The type of crop rotation in Dorinne and Tourinne are described in table 7 below:

	Year 1	Year 2	Year 3
Dorinne (3-year rotation)	Maize, Sugar beetroot or Rapeseed	Winter wheat	Winter barley
Tourinne (2-year rotation)	Sugar beetroot, Carrots, Potatoes, Pea or Linseed	Winter wheat	/

Table 7: Studied farm's crop rotation system (source: farmer)

In Dorinne, 2,17 ha of temporary grasslands lasting 3 to 4 years are also incorporated to the rotation.

Life Cycle Assessment

The LCA is carried out according to the ISO standards 14040/44:2006 (ISO, 2006a; ISO, 2006b), which include the following steps: goal and scope definition, life cycle inventory (LCI) analysis and life cycle impact assessment (LCIA).

It is important to note that the studied farm's associated SOC variation is voluntary not taken into account in this current section about the farm's LCA as it is the main focus of a separate section further in this document (cf. section called *"Contribution of SOC variation to the LCA"*).

Goals and scope definitions

Goals

The aim of this LCA is to estimate the environmental impacts of the total production of the studied farm in the year of 2021.

Functional unit

The ISO standards 14040/44:2006 defines *"Functional Unit"* as a *"quantifiable function of a product and the reference basis for system modelling in environmental assessment."* (ISO, 2006a; ISO, 2006b).

Different types of functional units can be used for crop-livestock LCA. The environmental impacts can be expressed per area of land (ha), per quantity of product (kg), per livestock unit (LU) or at the farm level.

According to Haas *et al.,* (2000), the functional unit that is the most adapted to crop-livestock farms is the farm level functional unit. Therefore, the functional unit selected for this LCA is the total production of the studied farm in the year of 2021.

It is important to specify here that 4 other functional units are assessed further in this study (cf. section called *"Functional unit analysis)*.

Scope

Figure 5 below indicates the system boundaries of the LCA. The boundaries stop at the farm's gates. Therefore, every step of the value chain that happens after the production, such as processing and commercialisation is not taken into account.
Background processes gather all the steps needed to produce the external inputs used by the farm such as fertilisers, pesticides or feed supplements. Foreground processes concern the activities happening on the farm site which can lead to emissions such as fertilisers and pesticides use or manure management.

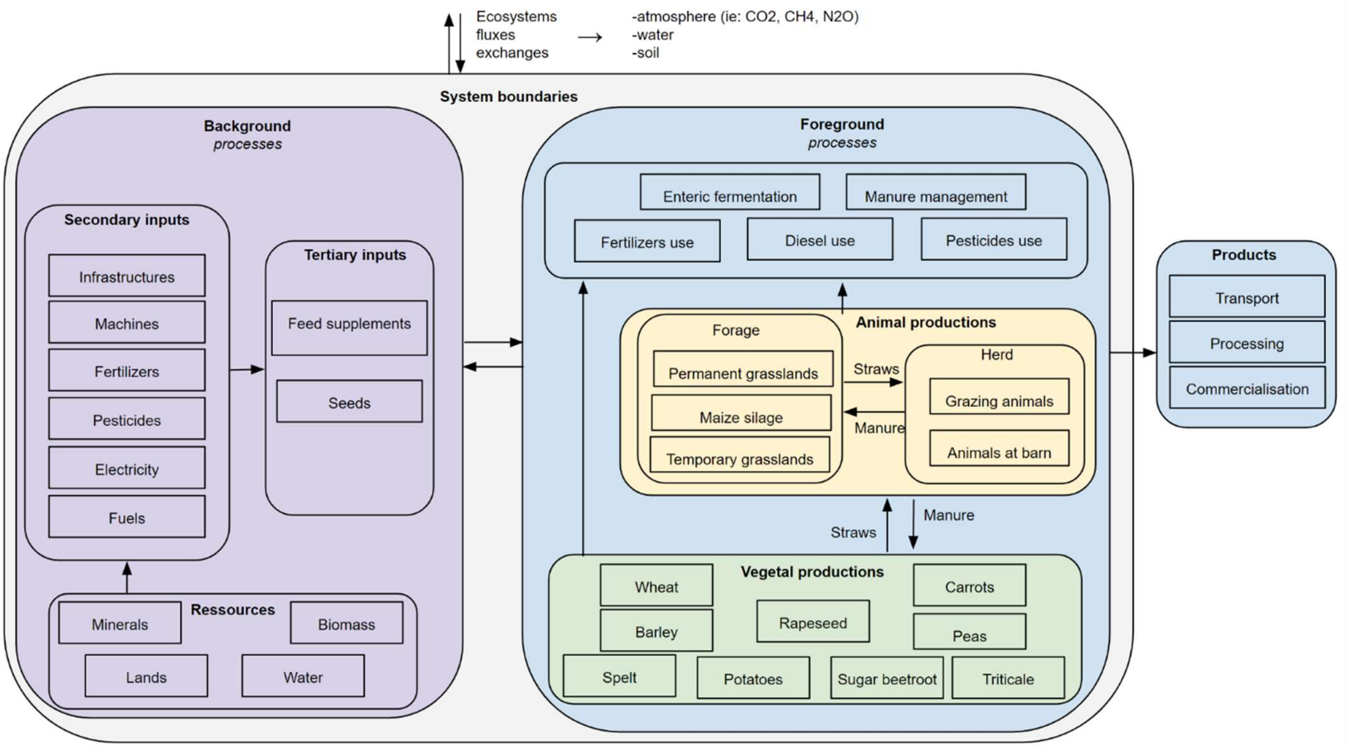


Figure 5: LCA system boundaries

Inventory analysis

This section details the procedure applied to build the life cycle inventory of the studied farm. First of all, every farm's inputs (e.g. materials, energy) and outputs (e.g. yields) were collected from the farmer throughout several meetings and exchanges between the months of June and November 2023. These data are issued from the farmer's accounting records which gather all the bills paid by the farmer as well as the invoices sent to his client. The accounting records from the year 2021 was the most complete and accurate record available to the farmer, therefore it was the year chosen for this analysis.

The life cycle inventory was computed in the LCA software *SimaPro 9.4.0.3.* Built on 30 years of LCA though leadership, research and policy contribution, this LCA software is widely used across the scientific community, businesses as well as for educational purposes. A large diversity of LCA databases can be uploaded to this software. For instance, the database *Agribalyse 3.1.1* is a French database on the agriculture and food sector made by *ADEME (Agence de la transition écologique)* and based on the current literature about agricultural LCA. In this study, some processes of the *Agribalyse* database were used. The exhaustive list of these processes can be found in table S4 from the supplementary materials. The way these processes were associated with the studied farm's datasets in order to build the life cycle inventory is described in the following section.

It is important to note that the studied farm's associated SOC variation is voluntary not taken into account in this LCA inventory analysis as it is the main focus of a separate section further in this document (cf. section called *"Contribution of SOC variation to the LCA"*).

Background processes

Table S1 (supplementary materials) retrieves the list of the farm's background processes.

The quantity and the types of mineral fertilisers used were given by the farmer (cf. Table S1). Corresponding *Agribalyse* processes of mineral fertilisers production were integrated into the inventory. These processes include the entire environmental footprint induced by the production of mineral fertilisers from the extraction of primary materials to the delivery of the final manufactured product at regional storehouses.

As mentioned before, organic fertilisers are also applied on the farm fields in quantity and types communicated by the farmer (cf. Table S1). It consists of the manure from the farm's herd as well as external inputs in the form of compost and pig slurry. The emissions induced by the application of these fertilisers are retrieved in table S2 and detailed in the foreground section below. However, impacts from the production of these organic inputs are entirely attributed to the associated animal rearing and vegetal production that led to the production of these organic fertilisers.

The production of pesticides induces environmental impacts which are part of the background processes. The list of pesticides bought was given by the farmer. However, because of time constraints, the exact composition of each of these pesticides couldn't be computed in the inventory. Nonetheless, *Agribalyse* contains processes which retrieve the average technical itinerary of pesticides application, as well as their detailed composition, for most of the conventional crop production types. Consequently, more than 40 types of pesticides products are integrated in the inventory model. Listing all of them would clutter up this document. Therefore, in order to illustrate how these were computed, the herbicide product *Roundup* is retrieved in table S1 as an example. The rest of the 40+ pesticides products of the inventory can be consulted in the *Agribalyse* database in the respective processes used (Table S4).

The farmer communicated the quantity of diesel consumed for the farm's operation such as tillage, seeding, fertilising, pesticides application, harvesting, which amounts to 24 200 litres (table S1) . The combustion of these fuels on the farm's site is referred to as "Diesel use" and is part of foreground processes, hence the method to account for these emissions are described in the foreground section below. In terms of background processes, environmental impact induced by the production, maintenance and use of machines such as tractors are accounted for as "Infrastructure and materials". More precisely, the farmer's operations are reported as "hours of machines use" (cf. Table S1). Every hour of machine use corresponds to a specific *Agribalyse* process, such as "1 hour of harvesting wheat" or "1 hour of tillage for potato production". Each of these processes include the corresponding amortisation of the environmental impacts induced by the production and maintenance of the specific machines and materials used (e.g. sowing with a classic seeder and harrow, or harvesting with a combine harvester), as well as by the abrasion of tyres, the lubricants oils consumption, the shed and land use needed to park the machines and materials. It also includes the quantity of seeds used for the seeding operations and all the required freight that is needed to convey all of these materials.

The infrastructure and materials section also include the environmental impacts induced by the infrastructure and materials used for animal rearing such as the amortisation of the construction of the animal housing (the barn, feed silos) as well as these buildings' electricity consumption. It is worth mentioning that unlike diesel use, electricity use on farm does not lead to any foreground emissions, environmental impacts associated with electricity use are induced by electricity production and are therefore considered as background processes exclusively. Like the other background processes, these inputs were communicated by the farmers (cf Table S1), and computed in the inventory using existing *Agribalyse* processes (cf Table S4).

In terms of feed consumption, the farm's herd diet is detailed in table 8 and was communicated by the farmer. When the feed is grown on the farm (maize silage and grasslands) the environmental impacts are distributed between the background and foreground processes as described in this section. However, the feed supplement which is bought outside the farm is considered as a separate background process, whose types and quantity were given by the farmer (table 8) and integrated in the inventory using *Agribalyse* processes (table S4).

<u>Foreground processes</u>

<u>Emissions to the air</u>

The application of nitrogen fertilisers leads to several types of emissions to the air. The inputs of N-fertilisers (table S1) are associated with emission factors from the IPCC (2006) to account for these emissions which are retrieved in table S2. First of all, every kg of N-fertilisers applied leads to direct N_2O emissions according to the emission factor EF_1 = 0,01 kg N_2O-N[1] / kg N-fertilisers applied retrieved from the IPCC (2006 vol.4 ch.11, table 11.1). This kg of N-fertilisers also leads to further indirect N_2O emissions through two different pathways: 1) nitrogen volatilization and deposition leading to further N_2O emissions and 2) nitrogen leaching to

[1] The common notation "one kg of N_2O-N" corresponds to one kg of N in the N_2O molecule with N accounting for 63,65% of the N_2O molecule mass.

groundwater and runoff to surface water leading to further N_2O emissions. These pathways can be visualised in figure S3 of the supplementary materials.

Hence, every kg of N-fertilisers applied partially volatilized in the forms of NH_3 and NO_x according to the emissions factors $Frac_{GASF}$ = 0,10 kg NH_3-N+NO_x-N / kg N-fertilisers applied for mineral fertilisers an $Frac_{GASM}$ = 0,20 kg NH_3-N+NO_x-N / kg N-fertilisers applied for organic fertilisers. These emissions of NH_3 and NO_x further lead to N_2O emissions according to the emission factor EF_4 = 0,01 kg N_2O-N / kg NH_3-N+NO_x-N retrieved from the IPCC (2006 vol.4 ch.11, table 11.3). It is important to note that NH_3 and NO_x emissions not only indirectly contribute to the climate change indicators through N_2O emissions but also directly contribute to other impact categories such as acidification. The exact amount of NH_3 and NO_x emissions is reported in table S2.

The second pathway through which fertiliser application indirectly emits N_2O to the air is through the nitrogen leaching to groundwater and runoff to surface water. These emissions are computed in the inventory using the following emissions factors $Frac_{Leach}$ = 0,3 kg N leaching-runoff / kg N applied and EF_5 = 0,0075 kg N_2O-N/kg N leaching-runoff (IPCC, 2006 vol.4 ch.11, table 11.3).

As shown by figure S3, the mineralization of SOM and crop residues both contributes to direct and indirect N_2O emissions. In regards to the mineralization of SOM, the associated N_2O emissions are taken into account in a separate section of this document (cf. section called *"Contribution of SOC variation to the LCA"*). As for crop residues, not enough data could be collected from the farmer about this aspect. Data about crop residues per types of crops were taken from *Agribalyse* processes (table S4) and associated with the IPCC emissions factors mentioned above to compute the N_2O emissions induced by crop residues (table S2).

In regards to the methane emissions induced by the herd enteric fermentation, it is important to mention that Dumortier *et al.,* (2021) have estimated the individual methane emissions of a part of the farm's herd grazing at Dorinne's ICOS site by combining eddy covariance measurements with geolocation of the cattle and a footprint model. Estimated emissions were 0,220 ± 0,035 kg CH4/LU/day. This estimate specifically relates to the part of the farm's herd which was grazing at the 4,2 ha ICOS site during the study time and cannot be generalised to the entire herd of 2021. The detailed herd characteristic and dry matter intake (DMI) of 2021 is detailed in table 8 which is based on data given by the farmer. This DMI was associated with a default raw energy content of 18.45 MJ/kg DMI, a methane energy content factor of 55.65 MJ/kg methane, a default methane conversion factor of 6.5% to perform a tier 2 IPCC emissions estimate (IPCC, 2006) as shown by equation 3 below:

$$\text{(Eq.3)} \qquad Enteric\ fermentation\ (kg\ CH4)\ = \frac{Dry\ matter\ intake\ (kg) * 18{,}45\ (MJ/kg) * 0{,}065}{55{,}65\ (MJ\ /\ kg\ CH4)}$$

The result of this calculation is given in table S2 and further detailed in table 8.

Excretions of the farm's herd are of two types: 1) dung and urine deposited by grazing animals and 2) manure collected at the barn.

The first type is considered as organic fertilisers use as mentioned above. It induces direct N_2O emissions according to the following emissions factor $EF_{3PRP,CPP}$ = 0,02 kg N_2O-N / kg N (IPCC, 2006, vol.4 ch.11, table 11.1), as well as indirect N_2O emissions in the same two pathways described above for organic N-fertilisers. The amount of N_2O, NH_3, and NO_x emission induced by these pasture faeces is retrieved in table S2.

In regards to manure collected at the barn, it is entirely and exclusively applied to crop fields in Dorinne at the rate of 35 tons of manure / hectare every 3 years. It is applied every summer on the lands where winter barley has been harvested and is directly followed by rapeseed or by a cover crop (phacelia, vetches) leading to the main crop (maize, sugar beetroot) (cf. table 7). The estimation method of the emissions induced by the application of these organic fertilisers has already been detailed above. However, the storage of this manure also induced emissions. Indeed, while the farm herd's produce manure all year long, it is only applied once a year on the fields, the rest of the time the manure is stored at the barn or next to the fields, which leads to additional emissions to the air in the forms of CH_4, N_2O (direct and indirect), NH_3, and NO_x which are all estimated using emissions factors from the IPCC guidelines (IPCC 2006, vol.4 ch.10) and retrieved in table S2. These emissions induced by the storage of manure are referred to as manure management in this inventory.

The combustion of diesel by the farm's tractors induced the emissions of many types of substances to the air. Hence, the 24 200 liters consumed on farm were computed in a *Agribalyse* process which accounts for 15 of these substances. Among them, CO_2, CH_4, N_2O directly contributes to the climate change impact category, NH_3 contributes to that category through indirect pathways (as explained previously) but also contributes, along with SO_2, to acidification. Non-methane volatile organic compounds (NMVOC) are also emitted which contributes to photochemical ozone formation. This list is non-exhaustive, and only emissions contributing to the climate change impact category in CO_2 -equivalent are retrieved in table S2. Nonetheless, by referring to the *Agribalyse* processes used in this inventory (table S4), one can consult the entire set of the 15 substances emitted to the air by diesel combustion in tractors and their respective effect on all of the 16 impact categories of this LCA.

As mentioned above, more than 40 pesticide products are integrated in the inventory. The use of these products leads to substances being spread into the environment. Listing all of these substances would clutter up this document. Therefore, in order to illustrate how these were computed, the substance glyphosate released by the use of the product *Roundup* is retrieved in table S2 as an example. The quantity and types of substances emitted in the environment by the use of these pesticides were estimated in the *Agribalyse* processes using the *OLCA-Pest* model, further information about this model can be found in Nemecek *et al.,* (2022). In regards to emissions to the air, the farm's use of *Roundup* leads to the emissions of 1,65 kg of glyphosate to the air (table S2). The rest of the substances spread by the other 40+ pesticide products of the inventory can be consulted in the *Agribalyse* database in the respective processes used (Table S4).

Emissions to the water

The quantity of nitrate (NO_3^-) leaching to groundwater amounts to 17,9 tons of NO_3^- (table S2). This amount was estimated using *Agribalyse* processes (table S4) which associate a quantity of NO_3^- leaching to the quantity of nitrogen inputs applied to the soil (mentioned above and retrieved in table S1). To do so, these processes use a model called *NO_3 - COMIFER 2001*. Likewise, the quantity of phosphate (PO_4^{3-}) leaching to groundwater amounts to 139 kg of PO_4^{3-} . Moreover, 89,2 kg of P also flows toward surface water as a result of erosion of soil particles (table S2). These fluxes of phosphate and phosphorus are estimated in the *Agribalyse* processes using the *P leaching - SALCA-P* and *P erosion - SALCA-P* models respectively.

The use of pesticides also leads to emissions of substances to surface water. As mentioned before the entire list of substances is not listed here, but glyphosate from *Roundup* use is shown as an example. In regard to emissions toward surface water, the farm's use of roundup is estimated to lead to the emissions of 143 mg of glyphosate to surface water (table S2). As previously mentioned, the *OCLA-Pest* model was used to perform these estimations.

It is also important to mention that leaching to groundwater of heavy metal during the time of cultivation as well as their flows toward surface water due to erosion are also taken into account in the *Agribalyse* processes used (table S4) using the *Heavy metals leaching - SALCA-SM* and *Heavy metals flows - SALCA-SM* models respectively.

Emissions to the soil

The use of pesticides also leads to emissions of substances to the soil. As mentioned before the entire list of substances is not listed here, but glyphosate from *Roundup* use is shown as an example. In regard to emissions toward the soil, the farm's use of roundup is estimated to lead to the emissions of 15,1 kg of glyphosate to the soil (table S2). As previously mentioned, the *OCLA-Pest* model was used to perform these estimations.

The heavy metals soil flow balance between input and output are taken into account in the *Agribalyse* processes used (table S4) using the *Heavy metals Soil - SALCA-SM* model.

Animal category	Head number	Days grazing	Dorinne's production [kg DM]				Supplement [kg DM]						Feed intake [kg DM]		Enteric fermentation CH4	
			Grazed grass	Grass silage	Corn silage	Alfalfa	Beet Pulp	DDGS (Protiwanze)	Concentrates	CMV	Salt block	Urea	Total	Total per head per day	kg CH4 / year	kg CH4 / LU / day
Bull >2 yr	2	61	1 452	4 268	1 228	0	1 736	1 736	0	219	219	73	10 931	15,0	236	0,32
Steer 1-2yr	15,5	0	0		10 297	35 359	5 436	5 436	28 853	641	641	128	86 793	15,3	1 870	0,33
Steer 0,5-1 yr	20	0	0	37 011	10 658	0	4 655	4 655	0	541	677	68	58 264	8,0	1 256	0,17
Male calf <0,5y	12,5	0	0	9 399	2 738	0	1 352	1 352	0	242	483	48	15 614	3,4	336	0,07
Suckling cow > 2y	80,5	212	186 019	79 195	22 786	0	47	10 739	14	1 478	1 847	493	302 618	10,3	6 521	0,22
Heifer >2y	10	195	19 890		2 941	0	1 314	1 314	0	81	161	32	25 732	7,0	555	0,15
Heifer 1-2y	40	156	45 552	36 032	10 366	0	4 882	4 882	4 180	669	836	167	107 566	7,4	2 318	0,15
Heifer 0,5-1y	33,5	0	0	36 071	10 393	0	5 933	5 933	10 383	742	1 483	148	71 087	5,8	1 532	0,13
Female calf <0,5y	19	0	0	17 615	5 063	0	2 616	2 616	31	380	761	77	29 157	4,2	628	0,09
Total	233		252 913	219 591	76 469	35 359	27 971	38 663	43 461	4 992	7 108	1 234	707 761		15 252	
LU mean															65,46	0,18

Table 8: Herd characteristics, feed intake in [kg DM], enteric fermentation methane. Source: Farmer's data in 2021.

Impact assessment

As previously mentioned in the literature review, life cycle assessments include a large number of environmental impact categories. In order to perform the life cycle impact assessment, different methods exist, which can vary in the choice of selected impact categories and/or in the specific ways to compute these.

Over the last two decades, the European Commission has led a process of standardisation and promotion of the procedure to apply to product environmental assessment. Consequently, a European Environmental Footprint method (EF) emerged and is regularly updated to follow the evolving scientific literature (Commission Recommendation (EU) 2021/2279).

In this study, the European EF method is used to perform the impact assessment. The exact reference of the method applied on the LCA software SimaPro is: *Environmental Footprint 3.1 (adapted for SimaPro substances)*. This method operates with 16 environmental impact categories which are all applied in this LCA. It is important to note that the foreground data concerning the "*Water use"* impact category was not available, therefore only the data from background processes is retrieved in this impact category.

These 16 impact categories are referred to as *midpoint* impact categories. The results contained in these 16 *midpoint* categories can further be expressed as 3 broader categories of environmental impacts namely: resource use, human health and ecological consequences (Saling, 2006), which are referred to as *endpoint* impact categories. Finally, the results entailed in these 3 *endpoint* categories can further be aggregated into one single environmental score. While this aggregation of results can provide benefits in terms of communication for instance, every step of the aggregation is associated with a loss of results characterization. In this study, it was decided to limit the analysis to the 16 *midpoint* impact categories in order to minimise this loss of results characterization.

Contribution of SOC variation to the LCA

As mentioned before, the studied farm's associated SOC variation is not taken into account in the LCA inventory analysis, it is the main focus of this section.

As mentioned previously, this study assesses the contribution of SOC variation to one LCA impact category exclusively: the climate change impact category. In order to do so, a sensitivity analysis is performed in this section, as suggested in such cases by similar studies (Buratti *et al.,* 2017). Practically, three scenarios are investigated, each of them indicating a potential SOC contribution to the LCA. In order to make the reading easier, these scenarios are referred to as 1) Carbon balance 2) Intermediate and 3) Soil sampling, and are detailed in this section.

SOC variation estimations for the entire 151 ha of the studied farm and in 2021 specifically are evidently not available. This approach aims at gathering the closest, temporally and spatially, available observations for the studied farm in order to provide a range of possible values. Therefore, the objective of this procedure is not to provide exact quantitative data, but to produce a qualitative analysis which is subject to high uncertainties.

Carbon balance

As mentioned previously, an important feature of this farm is to host an ICOS station on a parcel of its permanent grasslands. More precisely, this eddy covariance flux tower is implemented on an area covering 4,2 ha of the permanent grasslands of the farm. Gourlez de la Motte*,* (2019) analysed the data provided by this flux tower between 2010 and 2015 using the carbon balance approach (cf. section *Estimation methods* of this document) and found that the pasture was acting as a carbon sink, sequestering -1 000 ± 500 kg C/ha/yr. This scenario assumes that this sequestering rate may be applied to the entire 53 ha of Dorinne's permanent grasslands.

A similar study was conducted in a crop field located in Lonzée, in the loamy agricultural region of Wallonia, 40 km away from the studied farm. An eddy covariance flux tower is also implemented. Buysse *et al.* (2017) applied the carbon balance approach between 2004 and 2016 and found that the crop field was acting as a source, losing 825 ± 540 kg C/ha/yr to the atmosphere. This scenario assumes that this rate of carbon loss may be applied to the farm's entire 108,09 ha of cropland use.

Intermediate

Concerning permanent grasslands, the farmer's expertise was sought after. He provided a detailed description of the different parcels of its 53,33 ha of permanent grasslands and their respective management (cf. Table 9 below). Following his suggestions, the sequestration rate of -1 000 ± 500 kg C / ha / yr found at the 4,2 ha ICOS site (Gourlez*,* 2019) was extended to the rest of the permanent grasslands parcels proportionally to their respective associated stocking rate (cf. Table 9) and according to the following equation 4 below:

(Eq 4.) $$\Delta SOC\#p. = -1\,000 * \frac{Stocking\ rate\#p.}{2,3}$$

With $\Delta SOC_{\#p.}$: SOC variation of a parcel number, and Stocking $rate_{\#p.}$a: stocking rate of that same parcel number.

This kind of correlation is suggested by Gourlez (2019), and also established by Allard *et al.,* (2007), who showed that an intensively managed grassland could maintain a C sink activity over time while an extensively managed one could not. Similar findings were also observed by Ammann *et al.* (2020). In order to accordingly plot each parcel's SOC variation, the *unreported* parcel SOC was considered having reached equilibrium (=0). This is an oversimplified hypothesis as this parcel might be gaining or losing carbon in reality.

Parcel number	Area (ha)	Management			SOC variation (kg C/ha/yr)
		Number of cut(s)	**N fertilisation** (kg N / ha)	**Stocking rate** (LU/ha/yr)	
5	4,2 (ICOS site)	0	80	2,3	-1 000
	7,235	0	80	2,3	-1 000
	3,235	1	100	1,52	-661
7+10+1+4+2	6,53	0	80	1,78	-774
F608+12+9	10,24	0	80	1,38	-600
11	3,62	1	100	1,29	-561
3+14+15	2,19	0	80	1	-435
8	6,51	2	125	0,66	-287
6+13	1,76	3	150	0	0
Unreported	8	1	0	0	0
Total	53				
Mean					-553,5

Table 9 Permanent grassland detailed management 2021: source A. Paquet.

Concerning croplands, in the context of a SOC stock monitoring network of Walloon agricultural soils (so called “CARBOSOL”), Chartin et al. (2019) estimated trends of SOC variation between two soil sampling campaigns which occurred in 1949-1965 and 2005-2014. The results show that crop lands are acting as sources, losing 42 kgC/ha/yr at the Condroz region level, 81 kgC/ha/yr at the Loamy region level and 66 kg C/ha/yr at the Walloon level. This scenario assumes that these 3 rates of carbon loss may be applied to Dorinne’s 67,59 ha of direct croplands use, Tourinne’s 30,27 ha of direct cropland use and to the farm’s 10,23 ha indirect crop lands use respectively (cf. table 10). The indirect cropland use results from external inputs use: 8,23 ha needed to grow feed supplements and 2 ha needed to grow seeds. These external inputs bought by the farmer are produced in Wallonia, which explains their association to the Walloon level of SOC loss.

Soil sampling

Chartin *et al.,* (2022) have recently updated the SOC stocks and their recent trends in Wallonia. To do so, they analysed a database from *Requasud* (a Walloon agricultural consulting and analysis network) consisting of 86 737 observations (74 418 in cropland and 12 319 in permanent grassland) resulting from soil sampling campaigns which took place between 2004 and 2019 across Wallonia. In the Condroz region, results show that permanent grasslands are acting as sources, losing 67,5 kg C /ha/yr, while crop fields are acting as sinks, sequestering -176 kg C /ha/yr. This scenario assumes that those rates of SOC variation may be applied to Dorinne's 53,33 ha of permanent grasslands and 67,59 ha of direct crop fields use and respectively (cf. table 10).
It is important to note that in this recent update, not all Walloon agricultural regions show statistically significant SOC differences. Furthermore, the authors preconize to only use the statistically significant differences detected to assess SOC variations. In this regard, no statistical differences were found for the croplands of the loamy region where Tourinne is situated. Therefore, this scenario assumes that the previous loss rates of 81 kgC/ha/yr and 66 kg C/ha/yr (Chartin et al. 2019) may be applied to Tourinne's 30,27 ha of direct cropland use and to the farm's 10,23 ha indirect crop lands use respectively (cf. table 10).

Table 10 below summarises the mean SOC variation rate assumed for each of the three scenarios of the sensitivity analysis as described in detail above.

		Soil sampling (kg C/ha/yr)	**Intermediate** (kg C/ha/yr)	**Carbon balance** (kg C/ha/yr)
Croplands	Dorinne (67,59 ha)	-176	42	825 ± 540
	Tourinne (30,27 ha)	81	81	
	Indirect (10,23 ha) - Wallonia mean	66	66	
	Total (108,09 ha)	**-80,52**	**54,78**	**825 ± 540**
Permanent grasslands (53,33 ha)		**67,5**	**-553,5**	**-1 000 ± 500**

Table 10: Mean SOC variation rate assumed for the sensitivity analysis

Associated N_2O emissions

Furthermore, every time a carbon loss is taken into account in the sensitivity analysis above, associated N_2O emissions need to be accounted for as well. Indeed, the IPCC guidelines (IPCC, 2006) requires accounting for the nitrogen mineralisation associated with the loss of soil organic matter (SOM). This mineralised N is regarded as an additional source of N available for conversion to N_2O (Smith and Conen, 2004); just as mineral N released from decomposition of crop residues, for example, becomes a source. In the IPCC guidelines (IPCC, 2006), this source of N is referred to as "*N mineralisation associated with loss of soil organic matter resulting from change of land use or management of mineral soils (F_{SOM})*"

In practice, direct N_2O emissions are accounted using the following equations (IPCC, 2006, vol.4 ch.11):

(Eq.5:)

EQUATION 11.8
N MINERALISED IN MINERAL SOILS AS A RESULT OF LOSS OF SOIL C THROUGH CHANGE IN LAND USE OR MANAGEMENT (TIERS 1 AND 2)

$$F_{SOM} = \sum_{LU}\left[\left(\Delta C_{Mineral,\, LU} \bullet \frac{1}{R}\right) \bullet 1000\right]$$

Where:
F_{SOM} = the net annual amount of N mineralised in mineral soils as a result of loss of soil carbon through change in land use or management, kg N.
$\Delta C_{Mineral,\, LU}$ = average annual loss of soil carbon for each land-use type (LU), tonnes C.
R = C:N ratio of the soil organic matter. A default value of 10 (range from 8 to 15) may be used for situations involving management changes on *Cropland Remaining Cropland*.
LU = land-use and/or management system type
Note: *The inclusion of the term F_{SOM} is a change from the previous 1996 IPCC Guidelines, which did not include the N from mineralisation associated with a loss of soil organic C.*

F_{SOM} is then multiplied by the emission factor EF_1 = 0,01 kg N_2O-N/kgN to account for the N_2O emissions.

In terms of indirect emissions, F_{SOM} is considered not to influence the deposition pathway (cf. figure S3 of supplementary materials). However, F_{SOM} does take part in the leaching and runoff pathway and therefore indirect N_2O emissions need to be accounted for at this level. To do so, emissions factors $Frac_{Leach}$ = 0,3 kg N leaching-runoff / kg N applied and EF_5 = 0,0075 kg N_2O-N/kg N leaching-runoff (IPCC, 2006 vol.4 ch.11, table 11.3) are applied in this study.

Functional unit analysis

As mentioned previously, the main functional unit of this LCA study is the total production of the studied farm in the year of 2021.

Furthermore, four other functional units are analysed in this study as explained in this section below. This is done because of three main reasons.

Firstly, it can be argued that grassland SOC variation may be allocated exclusively to the farm's animal production, in which case animal production needs to be analysed separately.
Secondly, expressing the results of this LCA in terms of the farm's Belgian blue or wheat production allows for these results to be compared to similar production done by other farms. In other words, every farm being different in sizes and types of production, narrowing the results down per quantity and production types enable further comparisons.

Finally, both output-based and area-based functional units are used in this study in order to neither favours intensive nor extensive systems.

Animal production of the farm

The permanent grasslands of this farm are used exclusively for grazing the animals of the farm or to produce grass silage for feeding them. Consequently, the SOC variation occurring in these permanent grasslands can be allocated exclusively to the animal production part of the farm. In order to do so, the environmental impacts of the farm's animal production need to be estimated. In fact, it is already estimated as part of the LCA of this study which concerned the entire production of the farm. Yet, in order for the farm's animal production to be distinctly separated from the rest of the farm's production, a new functional unit is defined here: the animal production of the studied farm in the year of 2021.

Consequently, new system boundaries need to be defined as it has been previously done with figure 5. The animal production of the studied farm's entailed every input associated with this production such as every feed inputs whether they are produced on the farm (grasslands, maize silage) or externally bought (feed supplement). Although manure and straw exchanges occur between the farm's animal and vegetal production, every inputs associated with vegetal productions which are not directly linked to the animal production are not taken into account. Figure 6 below details precisely the system boundaries of this new functional unit.

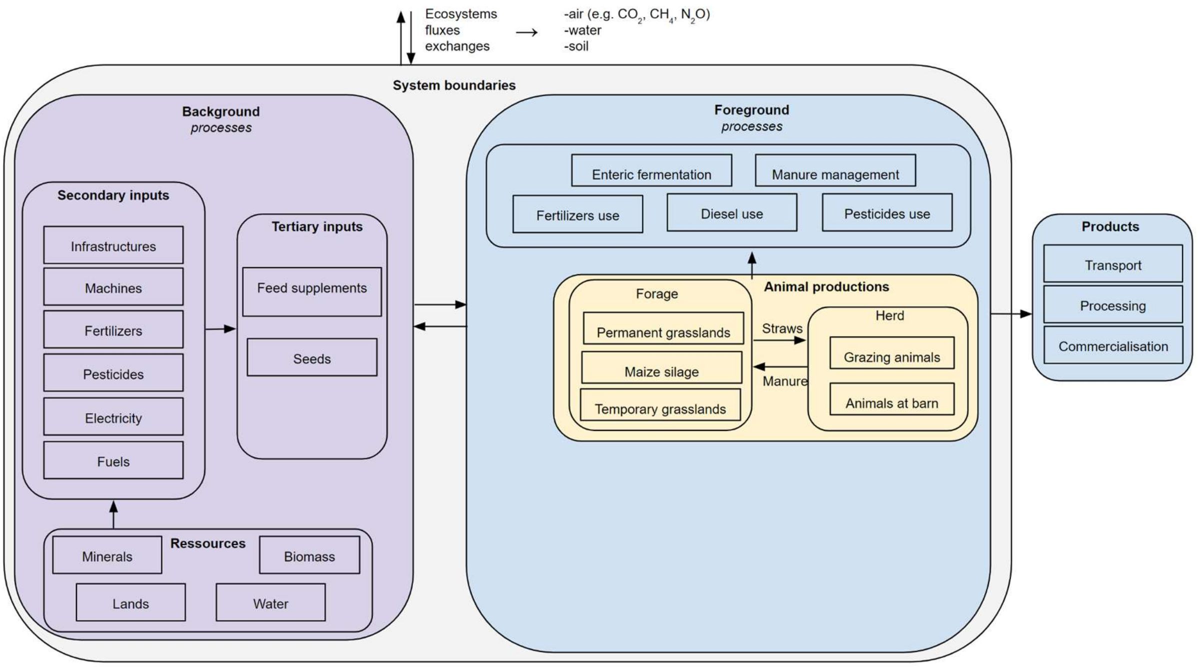


Figure 6: System boundaries for the farm's animal production

Wheat production of the farm

In order to allow for the results of this LCA to be compared to similar production done by other farms, a new functional unit is defined in this section where the farm's wheat production is distinctly separated from the rest of the farm's production. In practice, this third functional unit is defined as the wheat production of the studied farm in the year of 2021.

Consequently, new system boundaries need to be defined. Figure 7 below retrieves the system boundaries for the farm's wheat production. It can be noted that all the processes associated with the animal production and the other crop productions are not taken into account anymore. For instance, the background process *feed supplement* and the foreground processes *enteric fermentation* and *manure management* are entirely withdrawn. However, background processes such as fertilisers production or seed production are kept but only the quantity of inputs associated with wheat production is retained. The stratification of these input data per crop types was made possible by the data communicated by the farmer.

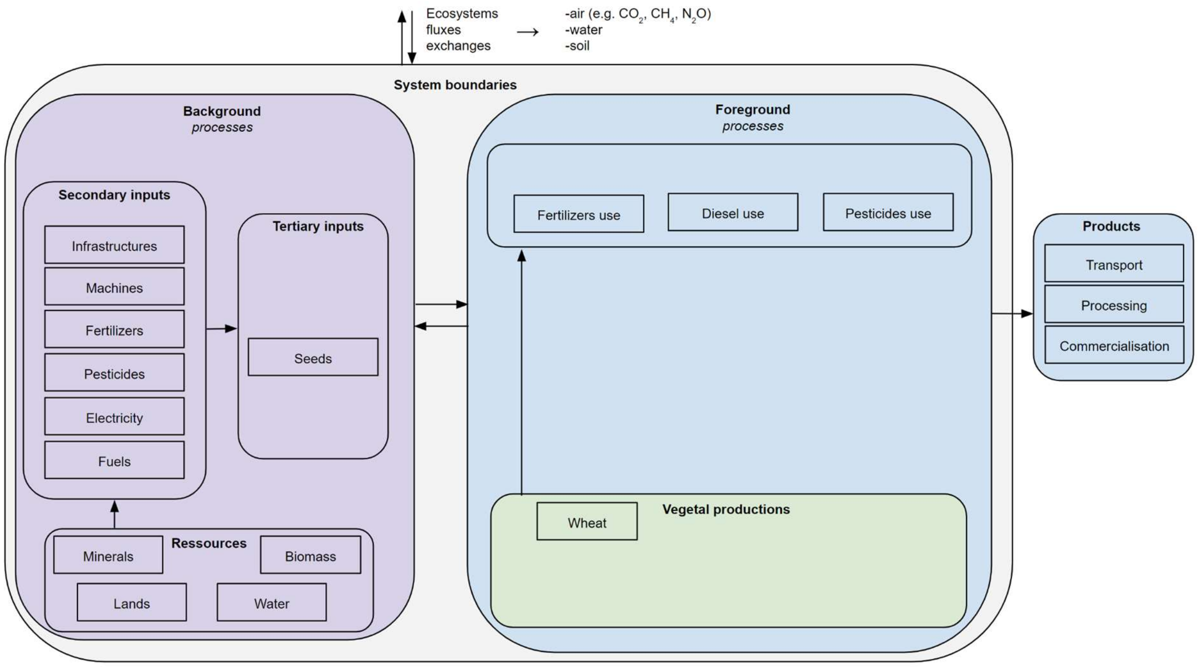


Figure 7: System boundaries for the farm's wheat production

It is important to mention that emissions induced by organic fertilisers use for wheat production were computed in a different way than at the farm scale. As previously mentioned, organic fertilisers are always applied at a specific time of the 3-year rotation (cf. table 7) which is usually after a barley crop. As a result, the quantity of organic fertilisers applied every year at farm scale is constant, and the resulting environmental impacts are constant as well.

Conversely, the allocation of these impacts at the scale of 1 year of wheat production is differently computed. Indeed, the organic fertilisers, applied every 3 years, have a fertilising effect on every one of the 3 successive annual crops (e.g. maize - wheat - barley). Therefore, the environmental impacts induced should be allocated on all of the 3 successive crops. The fertilising effect being non-linear, this allocation cannot simply consist in dividing the impacts by 3. Therefore, in order to accurately allocate the impacts induced by organic fertilisers use to the farm's wheat production, an *Agribalyse* process was used which estimated this allocation effect using a model called *Succession cropping : Cross fertiliser and organic fertiliser allocation*.

The above mention about how emissions induced by organic fertilisers use was allocated to wheat production is also the case for maize silage which is part of the farm's animal production functional unit.

Land used per FU

The SOC variation associated to each of the functional units used in this study is a function of 1) each of the FU's respective land used (cf. table 11 below) and 2) of the rate of SOC variation per ha of land use types (cf. table 10 above). The 9,96 hectares of direct land used for the animal production of the farm is obtained by summing the 7,79 hectares of maize silage and the 2,17 hectares of temporary grasslands. The rest of the surface comes from the farm's crop distribution (table 6).

Functional Unit (FU) Land use [ha]			Total productions of the farm	Animal production of the farm	Wheat production of the farm
Croplands	Direct	Dorinne	67,59	9,96	14,16
		Tourinne	30,27	0	12,75
	Indirect	Feed supplement	8,23	8,23	0
		Seeds	2	0,20	0,55
Permanent grasslands			53,33	53,33	0
		Total	161,42	71,72	27,46

Table 11: FU associated land used considered for SOC variation

Area-based and output-based functional units

Lebacq *et al* (2013) and Van stappen *et al.,* (2015) showed that the choice of FU can have a significant impact on the results of an agricultural LCA. Highly productive systems are favoured by the choice of an output-based functional units while extensive systems are advantaged by an area-based functional unit. In order to overcome this issue, the environmental impacts of the animal and wheat production of the farm are both expressed using an output-based and area-based functional unit. The land used by these productions can be found in table 11 above and their respective yields are retrieved in table S3 of the supplementary materials.

In conclusion, the results are expressed in five different functional units namely:

1. The total production of the studied farm in the year of 2021 (the main FU of this study)

2. One kg live-weight (LW) of the farm's animal production in the year of 2021.
3. One hectare of the farm's animal production in the year of 2021.

4. One kg of fresh matter (FM) of the farm's wheat production in the year of 2021.
5. One hectare of the farm's wheat production in the year of 2021.

The total results of the LCA for each of these five FU is retrieved in table 12 of the result section below.

The respective contributions of the processes types and production types to the LCA are calculated for the main FU of this study exclusively (cf. figures 8 and 9 of the results section).

The SOC variation analysis is performed for three following FU: Total farm production, 1 kg LW of the farm's animal production and 1 kg FM of the farm's wheat production (cf. SOC variation sensitivity analysis of the results section).

Results

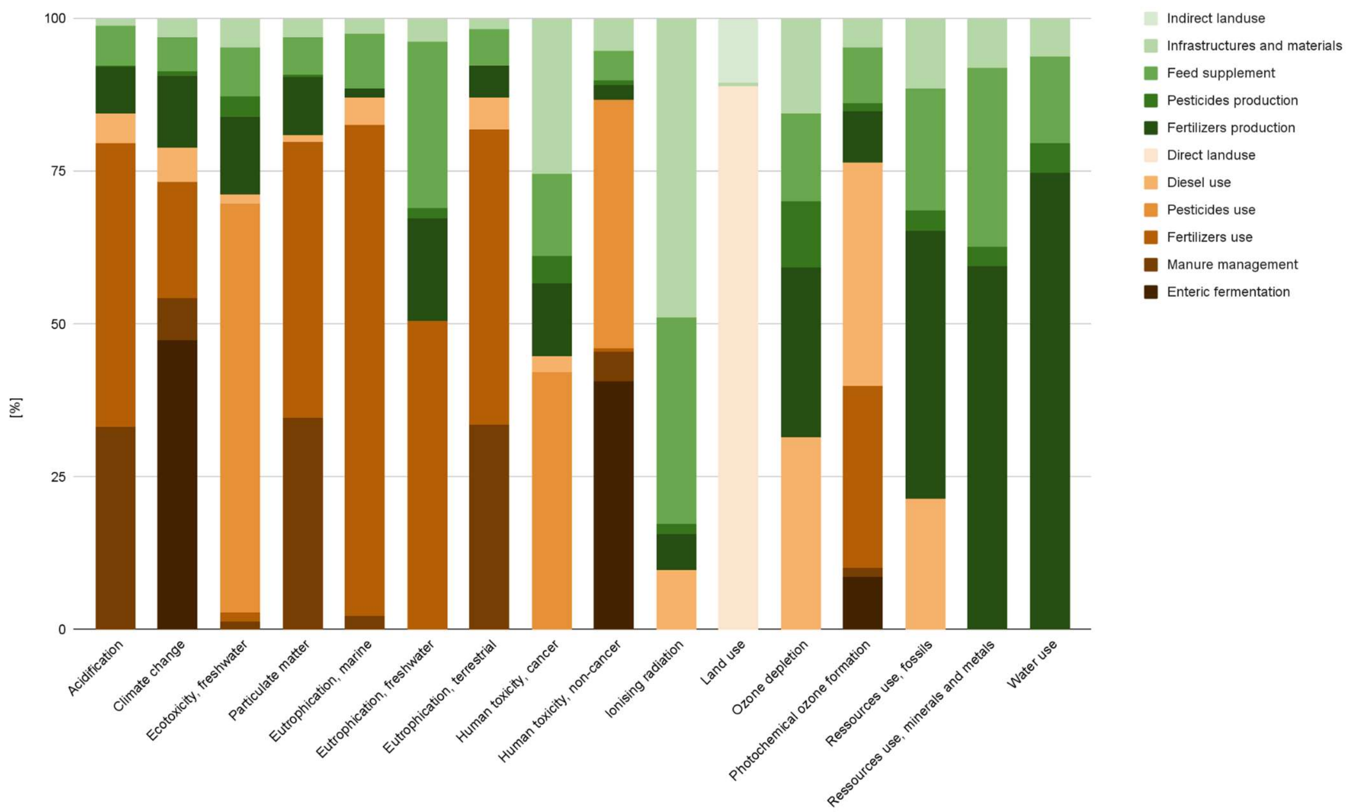
LCA

Figure 8 shows the result of the life cycle impact assessment expressed as the relative contribution (in [%]) of each process type to the 16 impact categories. Foreground processes are shown in figure 8 as shades of brown while background processes are shown as shades of green.

In regards to foreground processes, enteric fermentation contributes to the three following impact categories: climate change (47,3%), human toxicity, non-cancer (40,6%) and photochemical ozone depletion (8,58%). Manure management contributes to more than 30% of acidification, particulate matter and eutrophication, terrestrial. Fertilisers use accounts for more than 45% of the five following impact categories: acidification, particulate matter and the three eutrophication categories. It also significantly contributes to climate change (18,9%) and to photochemical ozone depletion (29,8%). Pesticides use contributes to more than 40% of ecotoxicity, freshwater and to the two human toxicity categories. Finally, diesel use amounts to more than 20% of ozone depletion, photochemical ozone formation and resource use, fossils. As previously mentioned, water used by foreground processes has not been computed. Hence, the water use impact category only entails background processes, with fertilisers production contributing to the most part (74,7%).

In terms of background processes, fertiliser production also highly contributes to resource use (59,4% for minerals and metals, and 43,9% for fossils) and ozone depletion (27,8%). In particular, the Haber-Bosch process, which enables the industrial production of ammonia, the basis of most mineral nitrogen fertilisers, requires high amounts of energy and fossil fuels (natural gas). Unlike pesticides use, pesticides production shows a relatively low contribution to most impact categories. Feed supplements contribute to more than 20% of four impact categories: eutrophication (freshwater), ionising radiation, ressources use (fossils and minerals and metals). Finally, infrastructure and materials significantly contribute to human toxicity, cancer (25,3%). Feed supplement and infrastructure and materials also both highly contribute to ionising radiation (33,7% and 48,9% respectively) which is explained by the fact that electricity use, which is needed for feed transformation and in the farm's buildings, is the main driver of this impact category.

In regards to the land use impact category, figure 8 shows that direct land use contributes to 89%, while indirect land use, which stands for land used to provide external inputs such as feed supplements and seeds, contributes to 10% and the remaining 1% comes from the contribution of infrastructure and materials.



<u>Figure 8: Impact assessments of the entire production of the studied farm in 2021 -per process types</u> S*hades of brown: foreground processes, shades of green: background processes.*

Another way to express the results of the life cycle impact assessment is through the different types of production at the farm as shown in figure 9. In this figure, the different crop productions that do not have a direct link with the farm's animal production are shown in shades of green. On the other hand, the farm's animal production is shown in shades of brown, taking into account the maize silage production, the production of grasslands, the feed supplement bought and the management of the herd.

Figure 9 shows that the farm's animal production contributes to more than 50% of six impact categories: acidification, climate change, particulate matter, eutrophication (terrestrial), human toxicity (non-cancer) and ionising radiation. The highest contribution of grasslands is toward eutrophication, marine (nearly 20%). Maize silage contributes the most to ecotoxicity, freshwater (12,9%). The production of winter wheat highly contributes to human toxicity, cancer (44,6%).

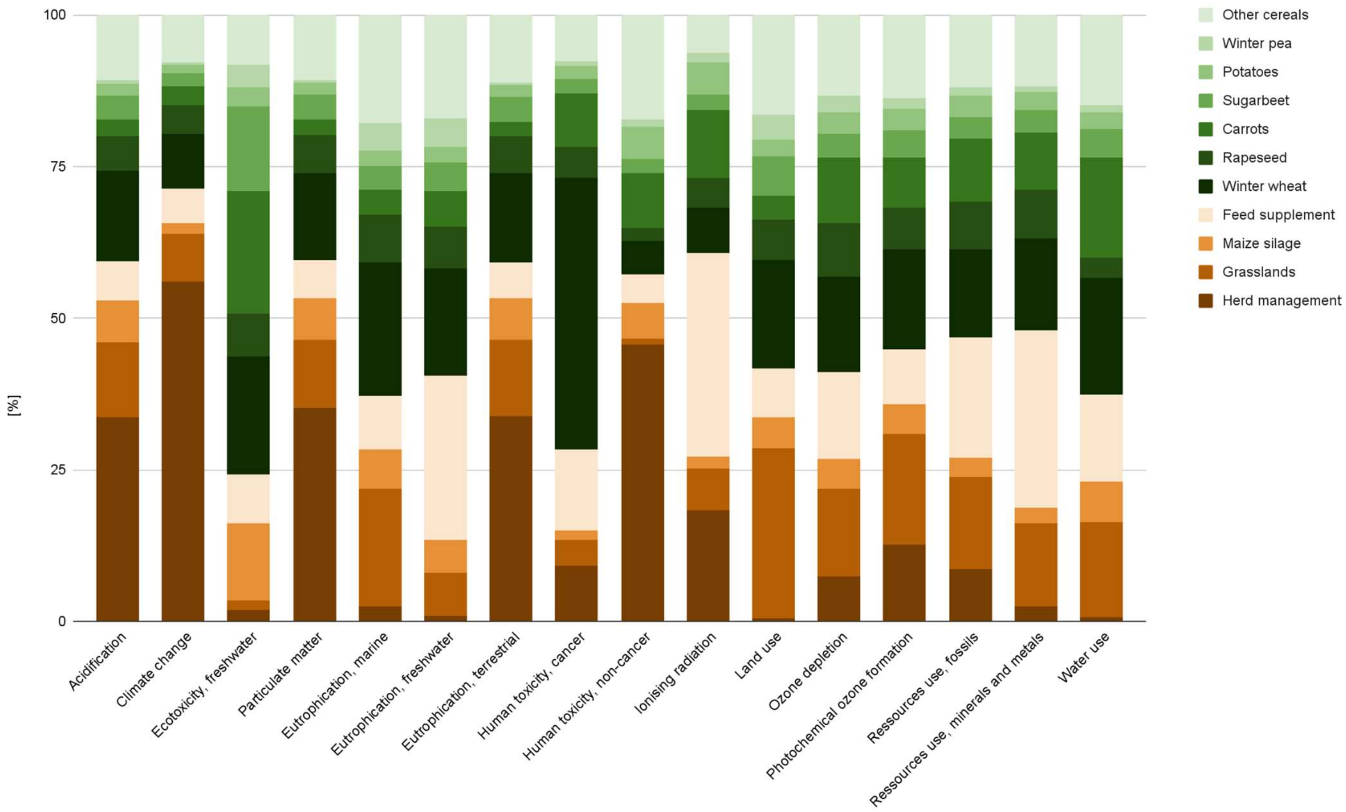


Figure 9: Impact assessments of the entire production of the studied farm in 2021 - per production types. *Shades of brown: animal production, shades of green: crop productions.*

Figures 8 and 9 express the results of the life cycle impact assessments in terms of the relative contribution (in [%]) of each process or production type. The exact value in the respective units of each impact category can be found in table 12 below. Table 12 shows the values for the functional unit of this LCA "Total production of the farm", which correspond to the functional unit used in figure 8 and 9. In addition, table 12 shows the results for 4 other functional units which have been previously defined.

Functional Unit (FU)	Total production of the of the farm	Animal production of the farm		Wheat production of the farm	
Impact categories [unit]		1 kg LW	1 ha	1 kg FM	1 ha
Acidification [mol H^+ eq.]	$1,06*10^4$	0,103	87,8	$7,57*10^{-3}$	58,7
Climate change [kg CO_2 eq.]	$8,70*10^5$	10,2	$8,68*10^3$	0,382	$2,96*10^3$
Ecotoxicity, freshwater [CTUe]	$1,03*10^7$	41,1	$3,51*10^4$	9,63	$7,47*10^4$
Particulate matter [disease inc.]	0,0700	$6,83*10^{-7}$	$5,83*10^{-4}$	$4,84*10^{-8}$	$3,75*10^{-4}$
Eutrophication, marine [kg N eq.]	$4,91*10^3$	0,0300	25,6	$5,17*10^{-3}$	40,1
Eutrophication, freshwater [kg P eq.]	131	$8,71*10^{-4}$	0,744	$1,11*10^{-4}$	0,858
Eutrophication, terrestrial [mol N eq.]	$4,65*10^4$	0,450	385	0,0330	256
Human toxicity, cancer [CTUh]	$9,87 * 10^{-5}$	$4,60 * 10^{-10}$	$3,93 * 10^{-7}$	$2,11 * 10^{-10}$	$1,64 * 10^{-6}$
Human toxicity, non-cancer [CTUh]	$1,83*10^{-3}$	$1,70 * 10^{-8}$	$1,45 * 10^{-5}$	$4,89 * 10^{-10}$	$3,79 * 10^{-6}$
Ionising radiation [kBq U-235 eq.]	$3,14*10^4$	0,313	267	0,0111	86,2
Land use [Pt.]	$7,86 * 10^7$	535	$4,57*10^5$	67,1	$5,20*10^5$
Ozone depletion [kg CFC-11 eq.]	0,0358	$2,41 * 10^{-7}$	$2,06*10^{-4}$	$2,68*10^{-8}$	$2,08*10^{-4}$
Photochemical ozone formation [kg NMVOC eq.]	$1,80*10^3$	0,0132	11,3	$1,41*10^{-3}$	11,0
Ressources use, fossils [MJ]	$3,40*10^6$	26,2	$2,24*10^4$	2,36	$1,83*10^4$
Ressources use, minerals and metals [kg Sb eq.]	3,01	$2,37 * 10^{-5}$	0,0203	$2,16*10^{-6}$	0,0168

Water use [m^3 deprivation]	$1,35*10^5$	0,843	720	0,127	988

Table 12: Results of life-cycle impact assessment per functional unit.

CTUe: Comparative Toxic Unit for ecosystem. disease inc. : disease incidence per kg of $PM_{2,5}$ emitted. CTUh: Comparative Toxic Unit for humans. kBq U-235 eq. : kilobecquerel of Uranium 235 equivalent. Pt: Points → This is a composite indicator measuring impacts on four soil properties (biotic production, erosion resistance, groundwater regeneration and mechanical filtration). NMVOC: Non-Methane Volatile Organic Compounds.

SOC variation sensitivity analysis

The amount of SOC variation associated to each of the functional units used in this study can be calculated using each of the FU's respective land used (cf. table 11 above) and the rate of SOC variation per land use types (cf. table 10). This amount is retrieved in table 13 below and expressed in CO2-equivalent because the loss of soil organic matter leads to N_2O emissions which are accounted for as explained previously in the section: *Materials and methods - Contribution of SOC variation*.

Scenarios of sensitivity analysis		**Soil sampling** [kg CO_2 eq.]	**Intermediate** [kg CO_2 eq.]	**Carbon balance** [kg CO_2 eq.]
Functional units (FU)				
Total farm productions	Cropfields ΔSOC	-29 024	32 609	373 827
	Grasslands ΔSOC	15 144	-108 619	-196 240
	Total ΔSOC	**-13 880**	**-76 010**	**177 587**
1 kg LW of animal production	Cropfields ΔSOC	-0,07	0,07	1,03
	Grasslands ΔSOC	0,25	-1,78	-3,21
	Total ΔSOC	**0,18**	**-1,70**	**-2,18**
1 kg FM of wheat production	Cropfields ΔSOC	-0,08	0,02	0,45
	Total ΔSOC	**-0,08**	**0,02**	**0,45**

Table 13: Amount of SOC variation per FU in [kg CO_2 eq.]

The amount of SOC variation (table 13) and the climate change impact (table 12) can be put in perspective to express the contribution of SOC variation to the climate change impact of each of the FU. This contribution is retrieved in table 14 below and expressed in terms of % of the climate change impact of each FU.

Functional units		Scenarios of sensitivity analysis: **Soil sampling** [% of climate change impact]	**Intermediate** [% of climate change impact]	**Carbon balance** [% of climate change impact]
Total farm productions	Cropfields ΔSOC	-3,34	3,75	42,97
	Grasslands ΔSOC	1,74	-12,48	-22,56
	Total ΔSOC	**-1,60**	**-8,74**	**20,41**
1 kg LW of animal production	Cropfields ΔSOC	-0,67	0,73	10,13
	Grasslands ΔSOC	2,44	-17,49	-31,60
	Total ΔSOC	**1,77**	**-16,76**	**-21,47**
1 kg FM of wheat production	Cropfields ΔSOC	-21,79	5,94	116,76
	Total ΔSOC	**-21,79**	**5,94**	**116,76**

Table 14: SOC variation sensitivity analysis

The data from table 14 can be visualised in the forms of three figures which are presented below as figure 10, 11 and 12.

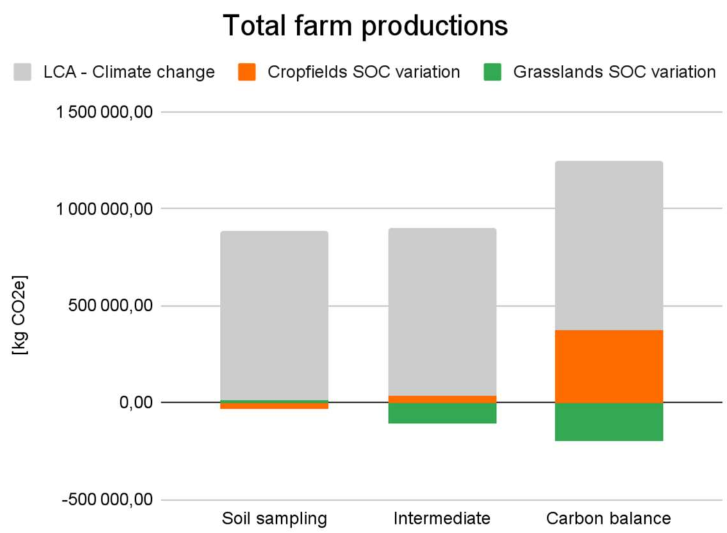


Figure 10: SOC variation sensitivity analysis for the total farm productions

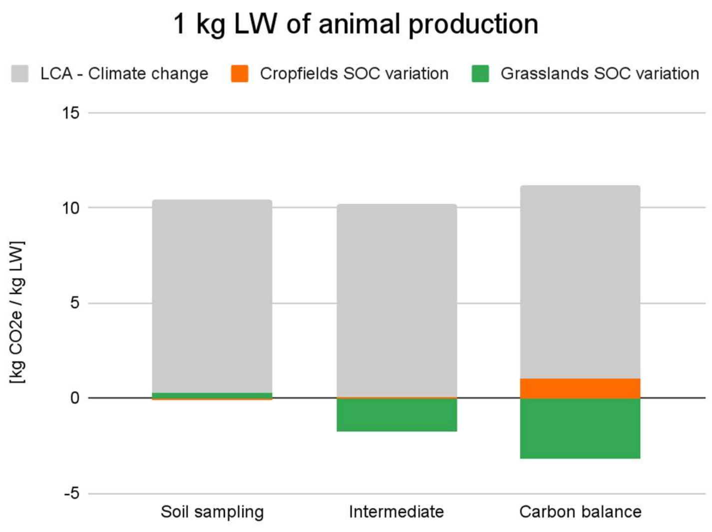


Figure 11: SOC variation sensitivity analysis for 1 kg LW of animal production

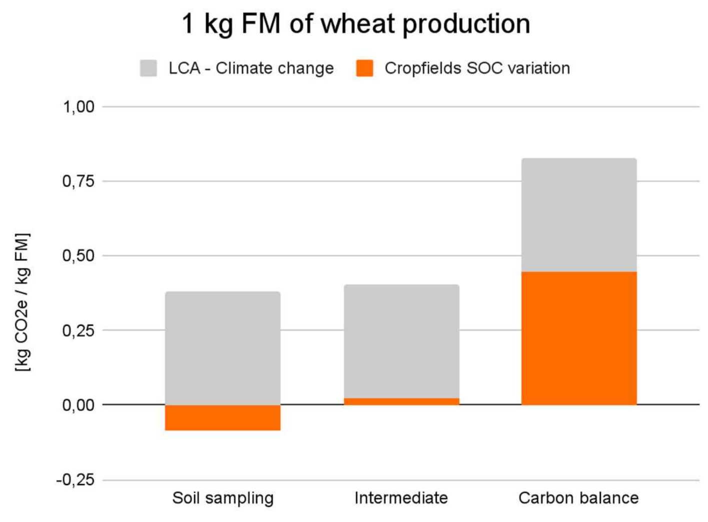


Figure 12: SOC variation sensitivity analysis for 1 kg FM of wheat production at the farm.

Discussion

LCA - Interpretation

The life cycle impact assessments of the studied farm's Belgian Blue productions resulted in a climate change impact of 10,2 kg CO_2eq / kg LW which is similar to the 8,5 kg CO_2eq / kg LW of Mathot *et al.,* (2016) and the 7,6 kg CO_2e / kg LW Rieria *et al.,* (2023), but are twice lower than the 22,6 kg CO_2e / kg LW found by Kokemohr *et al.,* (2022). This could be explained by the fact that Kokemohr *et al.,* (2022) chose a different functional unit, that is "kg of carcass weight". Because of this choice of FU, Kokemohr *et al.,* (2022) differentially expressed the animal output, separating the steer output as main product and considering heifers, culled cows and reproductive bulls output as co-products. Whereas in the LCA of the studied farm, the environmental impacts are allocated to the animal output in "kg of live weight" without differentiating the types of animal output, similarly as how Riera *et al.,* (2023) proceeded.

In comparison to other production systems shown in figure 3, the 10,2 kg CO_2eq / kg LW found in this study are closest to the non-intensive system studied by National Trust (2012). This can seem surprising as the management of Dorinne's animal production is closer to the ones from intensive systems in regards to the high stocking rates and fertilisation rates. One hypothesis which could explain this, is that Dorinne farm's herd management is well optimised by the farmer who carefully monitors reproductive performances, minimising parameters such as age at first calving and length of calving intervals.

Furthermore, when compared to the climate change values from table 5, Dorinne's 10,2 kg CO_2eq / kg LW is close to results found by Hammar *et al.,* 2022 (11,34 kg CO_2eq / kg LW) and by Beauchemin *et al.,* 2011 (13,04 kg CO_2eq / kg LW) but is higher than those found by Stanley *et al.,* 2018 (5,77 kg CO_2eq / kg LW) and lower than those found by Reyes-Palomo *et al.,* 2022 (20 kg CO_2eq / kg LW).

In regards to wheat production, the climate impact of the production of Dorinne's wheat amounts to 0,382 kg CO_2eq / kg FM and 2 960 kg CO_2eq / ha. In comparison, Van Stappen *et al.,* (2015) estimated the climate impact of conventional Walloon wheat production to amount to 0,349 kg CO_2eq / kg FM and 2 991 kg CO_2eq / ha., which consist of similar results. The meta-analyses carried out by Poore and Nemecek, (2018) on farms across the world found higher estimates of 1,4 kg CO_2eq / kg FM of wheat. One hypothesis explaining Van Stappen *et al.,* (2015) and Dorinne's lower carbon footprint of wheat (per kg of product) could be that Walloon conventional wheat yields are amongst the highest in the world.

In terms of relative contribution of processes, pesticides use was found to contribute to more than half of the ecotoxicity impact categories in the LCA of this study. In comparison, Van Stappen *et al.,* (2015) also found that pesticides use was contributing to more than 30% of these impact categories. Likewise, Dorinne's LCA showed that fertiliser use was contributing to more than 45% of eutrophication and acidification impact categories. Van Stappen *et al.,* (2015) also highlight the predominant share of fertiliser use to these impact categories.

As previously mentioned, mineral fertiliser production and use both contribute to a large share of several impact categories assessed in the studied farm LCA. Therefore, the conversion of Dorinne conventional farm to a certified organic farm could be considered in an effort to mitigate environmental impacts. In this regards, Van Stappen *et al.,* (2015) compared the environmental impact of the Walloon conventional and organic systems, investigating both an output-based (1 kg FM) and area-based (1 ha) functional unit. Results of this study showed that due to the nearly twice higher yield of the conventional production, organic wheat has an equivalent or even, in some impact categories, a higher impact than conventional wheat for the 1 kg FM functional unit. Nevertheless, organic production is less impacting than conventional production for most impact categories when it comes to the 1 ha functional unit. The choice of FU leads to complex debates on the role of agriculture production and the use of lands. Moreover, the climate change impacts foster those debates as the food security of certain regions of the world are consequently more threatened than others. Furthermore, historical GHG emissions are highly unequal between countries which raise questions on the responsibility of certain countries toward food security.

Yet, mitigation measures still need to be found to reduce impact for fertilisers. In this regard, great efforts have already been made in Belgium to minimise nitrate lixiviation organised by the Nitrate Directive. Therefore, reducing the quantity of fertilisers applied may not display the highest mitigation potential. However, measures aiming at improving nitrogen use efficiency such as the addition of nitrification inhibitors in fertilisers have been shown to be promising (Uchida and von Rein, 2018).

SOC variation

The soil sampling scenario of this study assumed the SOC variation rate of Dorinne's 68 hectares of croplands may be equal to -176 kg C/ha/yr (cf. Table 10) as suggested by (Chartin et al., 2022). This value stands for a sequestration rate which tackles the established historical trends according to which croplands are losing SOC. Nonetheless, in this study the effect of that assumption was compensated by Tourinne's land use and the indirect land use whose respective SOC variation are both considered positive (i.e. SOC loss) (cf. Table 10). In the end, the soil sampling scenario associates to the studied farm's cropland a SOC variation of -80,52 kg C/ha/yr which is 10 times lower than the SOC variation assumed by the carbon balance scenario (825 ± 540 kg C/ha/yr).

In regards to the farm's animal production, depending on the scenario of sensitivity analysis chosen, the contribution of SOC variation can either slightly increase or decrease the carbon footprint. This outcome highlights the methodological issues faced when accounting for SOC variation into agricultural LCA, which was also brought to light by several other studies such as Goglio *et al.,* (2015). Nevertheless, the sensitivity analysis provides a scope of possible contributions to the farm's animal production carbon footprint ranging from increasing it by 1,77% to decreasing it by -21,47 % (cf. table 12). These results are in line with the meta-analysis from Poore and Nemecek (2018) which found that SOC variation could reduce life-cycle ruminant emissions by a maximum of 22%, with greater sequestration requiring more land.

Increasing the studied farm animal production's carbon footprint (10,2 kg CO_2eq / kg LW) by 1,77% or decreasing it by -21,47% results in a range of carbon footprints comprised between 8,02 kg CO_2eq / kg LW and 10,38 kg kg CO_2eq / kg LW. These values can be compared to the values in blue of figure 3 which are those taking into account SOC variation. In this regards, it is worth mentioning the net emissions of the *Brazilian cerrado* system (103,9 kg CO_2eq / kg LW) exceed by far those of Dorinne's, even with respect to its most emissive scenario (10,38 kg CO_2eq / kg LW).

In light of these results, it is important to note that Gourlez de la Motte, L. (2019) established a GHG budget of Dorinne's 4,2 ha of pasture hosting the ICOS station. This budget, whose scope was restricted to the pasture scale exclusively, resulted in a GHG source, with pasture carbon sequestration offsetting 65% of the total emissions. Therefore, extending this budget to a complete life cycle analysis lowers the offset potential from 65% to a maximum of 22%.

<u>Methodological concerns</u>

Efforts to characterise the contribution of SOC variation to agricultural LCAs lead to several methodological issues which need to be discussed.

First of all, it is crucial that LCAs take into account the qualitative properties of the different types of GHG fluxes before quantitatively comparing them. Although 1 kg of CO_2 emitted from burning fossil fuels is quantitatively identical to 1 kg of CO_2 lost through SOC variation, the inherent qualitative differences of these two fluxes need to be accounted for. For instance, carbon fluxes from SOC variation are temporary; constantly shifting towards an equilibrium which can lead to a well-known "*sink saturation*" effect revealed by numerous studies (Skinner, 2008, Franzluebbers *et al.,* 2012, Smith, 2016, Minasny *et al.,* 2017,). Most importantly, these fluxes are also reversible (Smith, 2012, Godde *et al.,* 2020). Ammann *et al.* (2020) illustrated this characteristic by showing that 6 years of cumulated GHG uptake at a pasture was more than compensated by 3 years of cumulated GHG release following its renovation. This reversibility induces serious concerns at worldwide scale. Resources depletion and climate change impacts threaten our ability to maintain current agricultural yields and exchanges, the growing population could therefore need to convert forests and pastures to croplands in order to sustain itself. These LUC could therefore release the potential SOC sequestered back into the atmosphere. Likewise, any LMC which led to SOC sequestration are not physically guaranteed to be maintained across time. Beyond LUC and LMC, biophysical SOC drivers such as climate change can also reverse potential SOC sequestration. Indeed, Crowther *et al.,* (2016) showed that a business-as-usual climate scenario would drive the loss of 55 ± 50 petagrams of SOC by 2050 which represent around 15 percent of the expected anthropogenic emissions over this period.

Another critical methodological concern associated with assessing the contribution of SOC variation to agricultural LCAs, is to account for all GHG fluxes and not only carbon fluxes. Indeed, while some LMC can lead to additional SOC sequestration they can actually increase GHG emissions in total. For instance, Pellerin *et al.,* (2019) found that a moderate increase in the N-fertilisation of French grasslands could increase SOC sequestration leading to a GHG decrease of -646 kg CO_2e /ha /yr. However, when CO_2 emissions associated with the production of these additional N-fertilisers (+225 kg CO2e /ha /yr) and N_2O emissions from their field applications (+557 kg CO2e /ha /yr) were taken into account, this LMC lead to additional GHG emissions of +145 kg CO_2e /ha /yr. Likewise, based on quantitative data obtained from published meta-analyses, Guenet *et al.,* (2020), found that the climate mitigation induced by increased SOC sequestration is generally overestimated if associated N_2O emissions are not considered but, with the exception of reduced tillage, is never fully offset. Their results suggested that some options such as biochar use may even decrease N_2O emissions.

Beyond the complete GHG budget, the compiled impacts of all physical drivers affecting climate change should be taken into account. For instance, Pellerin *et al.,* (2019) found that extension of cover crops was the main lever for increasing SOC in France. Yet, cover crops also modify the agricultural land's albedo. In this regard, Pique *et al.,* (2023) showed that the mitigation potential of cover crops through albedo effects could reach 91,2 kg CO_2e /ha /yr across Europe. However, once introduced, cropland should be permanently covered by vegetation or straws in order to avoid the feedback loop effect due to soil darkening associated with SOC content increasing which could lead to a loss of 20% of the climate benefit.

Lastly, another critical methodological concern associated with assessing the contribution of SOC variation to agricultural LCAs, is whether the LCA is "*attributional*" or "*consequential*", the LCA of this study is attributional. Attributional LCA (ALCA) attributes a share of the potential environmental impact of the world to a product life cycle, while consequential LCA (CLCA) assesses the environmental consequences of a decision (Schaubroeck *et al.,* 2021). For instance, the decision of lowering animal production and switching from concentrates to grass-based feed would imply that croplands used for producing concentrates could otherwise be used to grow forest back or produce biofuels which would alter the climate change impact. Therefore, in a CLCA, an opportunity cost associated with keeping the cropland for feed production would be accounted for (Garnett, 2009). CLCA can also account for the impacts associated with the decision of preserving permanent grasslands. In this regard, Baudrier *et al.,* (2015) showed that if such a preservation is bound to the conservation of a similar animal herd, then the GHG emissions triggered by this animal herd would exceed the emissions that a conversion to croplands would have induced.

Prospects

In this study, the contribution of SOC variation was assessed to one LCA impact category exclusively: the climate change impact category. Yet, SOC variation could also contribute to other impact categories such as water use or ecotoxicity potential. For instance, by increasing the soil cation-exchange capacity (CEC), SOC variation could contribute to lowering the eutrophication potential (by reducing leaching losses) or to lower the climate change impact (by reducing the quantity of fertilisers applied). In this regard, Brandao *et al.,* (2011) suggested considering soil organic matter dynamics as a separate impact category that can be used to assess soil quality change. An important prospect of this work would therefore be to account for SOC variation contribution to the other environmental impact categories.

Another important prospect of this work would be to perform an uncertainty assessment of the LCA. Indeed, each step of the LCA is associated with uncertainties such as those linked with data collection or the use of emission factors. An uncertainty assessment would characterise each of these uncertainties and their combined effect on the LCA results. A methodological framework already exists for this uncertainty assessment to be made, but time constraints related to this work hindered its application.

Conclusion

The LCA performed in this study allowed the estimation of the studied farm's impacts on 16 environmental impact categories. The relative contribution of the farm's processes and production types to these impact categories were established. The farm's associated SOC variation was characterised according to three possible scenarios based on multiple soil sampling campaigns and flux tower stations from the ICOS network. Results show that the farm's climate change impact amounts to 870 000 kg CO_2e with the farm's SOC variation ranging between increasing this number by 21% and offsetting it by 9% depending on the scenarios selected. Furthermore, 4 other functional units were analysed as part of this study which led to further results characterization. For instance, the farm's animal production induces 10,2 kg of CO_2-equivalent per kg of liveweight produced and the associated SOC variation ranges between increasing this footprint by 2% and offsetting it by -22%. Moreover, methodological challenges associated with estimating the contribution of SOC variation to agricultural LCA were addressed. Finally, performing an uncertainty assessment would strengthen this study and investigating the potential contribution of SOC variation to additional environmental impact categories constitute a significant prospect.

Contributions

Professors B. Heinesch and A. Di Maria introduced me to the subject of this master thesis as well as to Dorinne's farm, its ICOS site and to the farmer Adrien Paquet. I collected data from the farmer throughout multiple meetings and exchanges. Astrid Loriers from the CRA-w instructed the specificities of the DECiDE tool which had previously been used with Adrien Paquet. Pr. Di Maria taught me how the SimaPro software functions and instructed LCA background knowledge essential to this work. I computed the data collected on SimaPro and performed the LCA. Caroline Chartin communicated recently updated data essentials to this work. I established the SOC analysis of this work with the guidance of Pr. Heinesch. I operated the results processing and carried out the redaction of this document.

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Supplementary materials

Categories of processes	Processes	Input quantity [unit]
Fertilisers production		
	Mineral fertilisers as N	19 505 [kg N]
	Mineral fertilisers as P_2O_5	1 452 [kg P_2O_5]
	Mineral fertilisers as K_2O	2 372 [kg K_2O]
	Organic fertilisers	631,4 [tons of manure]
Pesticides production		
	Roundup:	1 199,5 [litres]
	The rest of the 40+ other products can be consulted in the *Agribalyse* database in the respective processes used (Table S4).	
Infrastructures and materials		
	Diesel input	24 200 [litres]
	Machine use [in houres]	761 [houres]
	Barn infrastructures	14 400 [kWh of electricity] 630 [m2] of steel buildings 80 [m2] of manure tank
Feed supplement		
	The quantities and types of feed supplements consumed by the farm's herd are retrieved in table 8.	

Table S1: Life cycle inventory of the studied farm LCA - background processes

Types of emissions	Processes inducing emissions	Emissions quantity [unit]
Emissions to air		
	Fertilisers use	
		443,12 [kg N_2O] 1,617 [tons of NH_3] 1,2486 [tons of NO_x]
	Enteric fermentation	
		15,3 [tons of CH_4]
	Manure management	
		1,82 [tons of CH_4] 6,18 [kg N_2O] 1,16 [tons of NH_3] 0,0434 [tons of NO_x]
	Diesel use	
		65 119 [tons of CO_2 -equivalent] The 12 other substances emitted to the air by this process can be consulted in the *Agribalyse* database in the respective processes used (Table S4).
	Pesticides use	
		1,65 [kg of glyphosate] The rest of the substances emitted to the air by the use of the other 40+ pesticides products can be consulted in the *Agribalyse* database in the respective processes used (Table S4).
Emissions to water		
	Fertilisers use	

		to groundwater: 17,9 [tons of NO_3^-] 139 [kg of PO_4^{3-}] *to surface water:* 89,2 [kg of P]
	Pesticides use	
		143 [mg of glyphosate] The rest of the substances emitted to the water by the use of the other 40+ pesticides products can be consulted in the *Agribalyse* database in the respective processes used (Table S4).
	Heavy metals	
		The types and quantity of heavy metals leaching to groundwater and flowing to surface water can be consulted in the *Agribalyse* processes used (table S4).
Emissions to soil		
	Pesticides use	
		15,1 [kg of glyphosate] The rest of the substances emitted to the soil by the use of the other 40+ pesticides products can be consulted in the *Agribalyse* database in the respective processes used (Table S4).
	Heavy metals	
		The types and quantity of heavy metals soil flow balance between input and output can be consulted in the *Agribalyse* processes used (table S4).

Table S2: Life cycle inventory of the studied farm LCA - foreground processes

Production type	Outputs [unit]	Comments
Winter wheat	10 552 [kg FM / ha]	15% moisture
Winter spelt	7 411 [kg DM / ha]	
Winter triticale	6 467 [kg DM / ha]	
Winter barley	9 463 [kg DM / ha]	
Sugar beetroot	89 520 [kg FM / ha]	88% moisture 17,24% sugar content
Forage beetroot	73 227 [kg FM / ha]	88% moisture
Rapeseed	3 443 [kg DM / ha]	
Potatoes	40 000 [kg DM / ha]	
Carrots	86 986 kg [FM / ha]	87% moisture
Winter pea	3 200 [kg DM / ha]	
Maize silage	18 700 [kg DM / ha]	
Temporary grasslands	13 490 [kg DM / ha]	
Permanent grasslands	13 490 [kg DM / ha]	
Production the Belgian Blue herd	61 079 [kg LW]	

Table S3: Life cycle inventory of the studied farm LCA - farm's output.

LCI stage	***Agribalyse* processes**	
	Name	**Reference**
Permanent grasslands	Grazed grass, permanent meadow, without clover, Northwestern region, on field {FR} U	AGRIBALU000000003106967
	Grass silage, horizontal silo, permanent meadow, without clover, Auvergne, at farm {FR} U	AGRIBALU000000003106934
Temporary grasslands	Grass silage, horizontal silo, temporary meadow, with clover, Northwestern region, at farm {FR} U	AGRIBALU000000003106936
Maize silage	Silage maize, conventional, national average, animal feed, at farm gate, production {FR} U	AGRIBALU000000003113815
Sugar beetroot	Sugar beet roots, conventional, production year 2009, at farm gate {FR} U	AGRIBALU000000003115097

Winter rapeseed	Rapeseed, conventional, 9% moisture, national average, animal feed, at farm gate, production {FR} U	AGRIBALU000000003112460
Carrots	Carrot, conventional, fall, Creances, Lower Normandie, at farm gate {FR} U	AGRIBALU000000003102587
Potatoes	Ware potato, conventional, for industrial use, at farm gate {FR} U	AGRIBALU000000003116701
Pea	Winter pea, conventional, 15% moisture, at farm gate {FR} U	AGRIBALU000000003117318
Winter wheat	Soft wheat grain, conventional, national average, animal feed, at farm gate, production {FR} U	AGRIBALU000000003114120
Winter spelt	Durum wheat grain, conventional, national average, at farm gate {FR} U	AGRIBALU000000003105058
Winter triticale	Triticale grain, conventional, national average, animal feed, at farm gate, production {FR} U	AGRIBALU000000003116013
Winter barley	Winter barley, conventional, malting quality, animal feed, at farm gate {FR} U	AGRIBALU000000003117288
Beet Pulp	Sugar beet pulp dehydrated, animal feed, at plant {FR} U	AGRIBALU000000003115089
DDGS (Protiwanze)	DDGS, dehydrated, from wheat distillation, animal feed, at plant {FR} U	AGRIBALU000000003104690
Concentrates	Cereals mixture (wheat + barley + triticale/pea), organic, animal feed, at farm gate {FR} U	AGRIBALU000000003102767
CMV	Bovine feed, CMV 5-25-5, at farm gate {FR} U	AGRIBALU000000003101740
Salt block	Sodium chloride, animal feed, at retailer gate {FR} U	AGRIBALU000000003113989
Urea	Urea (with 46% N), at plant {RER} - Adapted from WFLDB U	AGRIBALU000000003116234
Manure management	Young suckler bull, conventional, fattening system, more than 1.2 LU per ha, at farm gate {FR} U	AGRIBALU000000003117521

Table S4: List of *Agribalyse* processes used.

Maps

Figure S1: Map of the farm’s land distribution in Dorinne. The yellow squares correspond to the farm’s lands. The farm’s buildings are surrounded by parcels #4,#3, #10 and #14. The ICOS station is located on parcel #5.

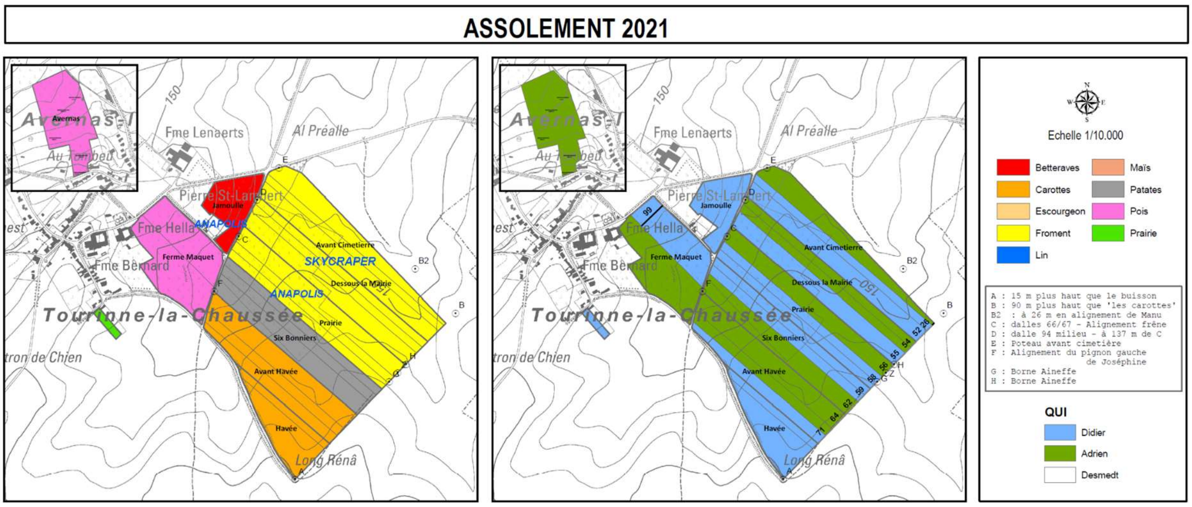

Figure S2: Map of the farm’s land distribution in Tourinne.

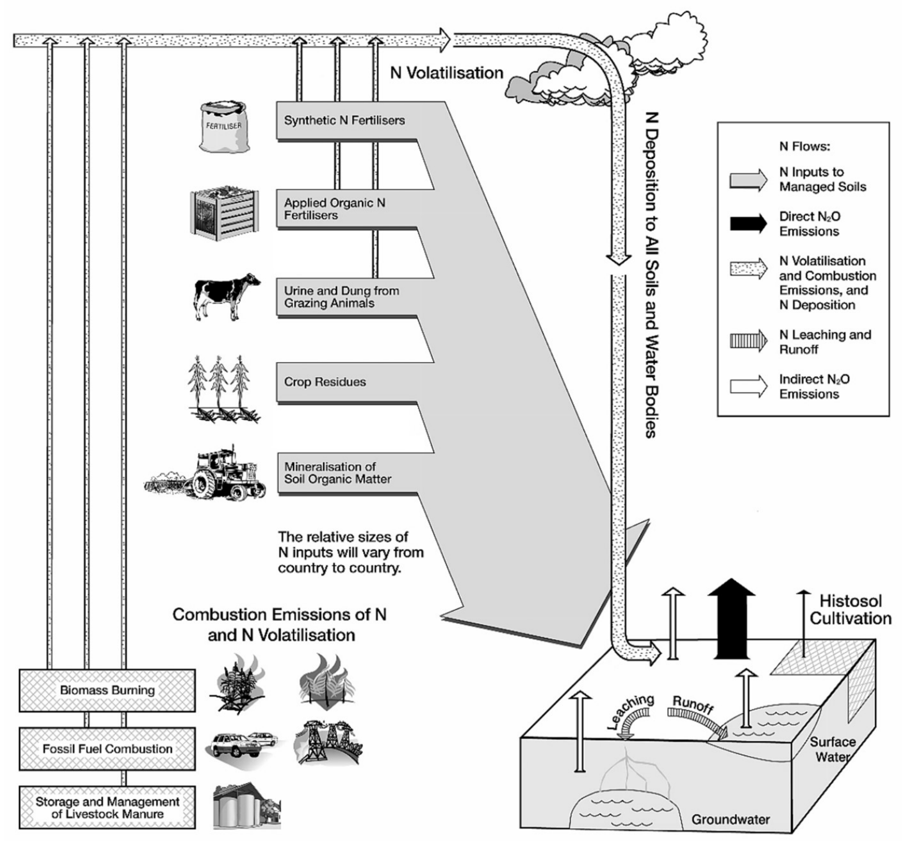

Figure S3: Schematic diagram illustrating the sources and pathways of N that result in direct and indirect N2O emissions from soils and waters (IPCC, 2006, vol.4 ch.11)