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Time-dependent climate impact of beef production – can carbon sequestration in soil offset enteric methane emissions?

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ABSTRACT

The time-dependent climate impact of beef production, including changes in soil organic carbon, was examined in this study. A hypothetical suckler cow system located in south-east Sweden was analysed using a time-dependent life cycle assessment method in which yearly fluxes of greenhouse gases were considered and the climate impact in terms of temperature response over time was calculated. The climate impact expressed as carbon dioxide equivalents, *i.e.* global warming potential in a 100-year time perspective, was also calculated. The Introductory Carbon Balance Model was used for modelling yearly soil organic carbon changes from land use. The results showed an average carbon sequestration rate of 0.2 Mg C ha^{-1} and yr^{-1}, so carbon sequestration could potentially counteract 15–22% of emissions arising from beef production (enteric fermentation, feed production and manure management), depending on system boundaries and production intensity. The temperature response, which showed a high initial increase due to methane emissions from enteric fermentation, started to level off after around 50 years due to the short atmospheric lifetime of methane. However, sustained production and associated methane emissions would maintain the temperature response and contribute to climate damage. A forage-grain beef system resulted in a lower climate impact than a forage-only beef system (due to higher slaughter age), even though more carbon was sequestered in the forage-only system.

1. Introduction

Ruminants are unique in their ability to utilise energy from roughages to produce highly valued foods in the form of meat and milk. Hence, ruminants can graze pastureland that is unsuitable for growing crops for human consumption, thus contributing to food security without competing for cropland (Van Zanten et al., 2018). In addition, roughage feed production on cropland contributes to soil carbon sequestration (Börjesson et al., 2018; Jarvis et al., 2017; Tidåker et al., 2014). However, enteric fermentation in ruminants gives rise to major methane (CH_4) emissions, which is why ruminant production is considered a major contributing factor to climate change (Reisinger and Clark, 2018). Life cycle assessments (LCAs) have demonstrated a particularly high climate impact of beef and other types of ruminant meat in comparison with meat from monogastric animals (Clune et al., 2017; Gerber et al., 2013). Grain-fed, high-input/high-output beef systems often show lower climate impact than grass-based, low-input systems, as animals with a higher growth rate reach slaughter weight faster and emit less CH_4 per kg beef produced (Clark and Tilman, 2017). However, if sequestration of carbon in soils is accounted for, extensive systems can show more favourable results, since growing grass leys or using grassland for livestock diets often leads to higher sequestration rates than annual cropping (Trydeman Knudsen et al., 2019; Alemu et al., 2017). There is no universal method for including soil organic carbon (SOC) in LCA (Goglio et al., 2015). Previous studies on beef systems that have included SOC have used fixed values based on literature data (*e.g.* Pelletier et al., 2010), relied on measured data during a limited period of time (*e.g.* Stanley et al. (2018) used data from soil measurements on-site over four years) or used modelled soil carbon fluxes (Alemu et al., 2017). Moreover, few studies on ruminant production including changes in SOC have been conducted and there is a need for further analysis of the influence of SOC in LCAs of beef systems.

Most studies on the climate impact of livestock products use global warming potential (GWP) as a metric to weight different greenhouse gases (GHGs). GWP expresses the climate impact of GHGs, most importantly carbon dioxide (CO_2), CH_4 and nitrous oxide (N_2O), in terms of CO_2-equivalents (CO_2-eq), normally during a 100-year time perspective (GWP_{100}), based on the cumulative radiative forcing of the

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different gases (Myhre et al., 2013b). Calculation of GWP necessitates an arbitrary choice of time horizon over which the climate impact is assessed, which can heavily impact comparisons between systems in which the CO_2 to CH_4 ratio differs, *e.g.* intensive versus extensive beef systems, as a shorter time horizon gives more weight to the short-lived CH_4. Use of GWP to assess the impact of short-lived GHGs such as CH_4 (which has an average atmospheric lifetime of 12.4 years) has been criticised for failing to capture its impact on temperature, since CH_4 does not accumulate in the atmosphere, unlike more long-lived GHGs (Lynch, 2019; Reisinger and Clark, 2018; Allen et al., 2016). Other metrics suggested for better displaying the impact of short-lived GHGs include *e.g.* the GWP* metric, where the climate impact of CH_4 is adjusted to better correspond to the temperature response of CH_4 over time (Allen et al., 2018). However, GWP* does not easily lend itself to product-based assessment like calculating the climate impact of different food products, as a decision has to be made on whether CH_4 emissions for the system under study belong to a pool of emissions that is increasing, decreasing or constant.

Another option for handling the climate impact of different GHGs is to calculate the temperature response of these gases. The climate metric referred to as Absolute Global Temperature change Potential (AGTP) by the IPCC considers the timing of GHG fluxes and displays the temperature response over time (Myhre et al., 2013a). The AGTP metric has been used in combination with SOC modelling in previous LCAs, referred to as time-dependent LCA, of *e.g.* bioenergy systems (Hammar et al., 2017; Ericsson et al., 2013). Using AGTP overcomes the arbitrary choice of time horizon and takes into account when in time emissions or sequestration take place.

The aim of this study was to assess the climate impact of beef production from a life cycle perspective, including soil carbon changes and applying a time-dependent climate metric to account for timing of emissions, sequestration and the characteristics of different greenhouse gases. Soil organic carbon changes were modelled with the Introductory Carbon Balance Model (ICBM), in order to account for carbon sequestration over time (Kätterer and Andrén, 2001). Sensitivity of the model to different parameters, including initial SOC content, was evaluated. In addition to GWP, a time-dependent climate metric was applied for assessing the climate impact in terms of temperature response over time. This adds to the knowledge base as regards the potential of soil carbon sequestration to counteract other GHGs from beef production and the importance of changes over time.

A hypothetical farm with a suckler cow system located in Uppsala County in south-east Sweden was selected as the case study. Two different scenarios were investigated, one in which the animals were fed only forage and one in which they were fed forage and grain.

2. Methods

A life cycle perspective was applied for assessing the climate impact of beef production. Two types of climate metric were used, GWP and AGTP, and the results were expressed per unit 'farm and year', *i.e.* the yearly GHG fluxes were presented per unit total output per year (24 Mg bone-free meat and 180 Mg spring barley). As changes to soil carbon are slow, and there are delays in the temperature response due to the inertia of the climate system, we modelled the system over 100 years. Some results were also expressed per 'kg bone-free meat', for comparison with previous studies. When the functional unit 'farm and year' was applied, no allocation between the meat and surplus barley produced was used, since both products were included in the functional unit. When the functional unit 'kg bone-free meat' was applied, system expansion was used for the surplus barley (based on values for spring barley grain from Moberg et al. (2019)). It was assumed that all feed production and grazing took place on cropland, *i.e.* that no permanent grassland was included in the production system.

2.1. System description

The hypothetical suckler cow system studied was assumed to be located in Uppsala County (59°N, 17°E). The system modelled was designed to represent real farms in Sweden in term of diets, feed properties, livestock system characteristics, crop production, manure management etc. The climate impact assessment included processes from cradle to farm-gate, *i.e.* emissions of CO_2, CH_4 and N_2O from animal husbandry, enteric fermentation and field operations, and land use emissions including SOC changes (Fig. 1).

Continuous fallow was assumed as the reference land use to assess the net land use effect, *i.e.* the difference between fallow land and the land use in the beef production system. The SOC modelling for land use was based on the most common soil type of fallow land in the study region, and the median value for initial soil carbon content was used (which was varied in a sensitivity analysis).

Two suckler cow scenarios were considered (forage-fed and forage + grain-fed beef cattle production), where the same amount of bone-free beef meat was produced each year in both scenarios. All livestock were assumed to be fed exclusively on grass in the forage scenario (silage and grazing), while the young bulls in the forage-grain scenario were assumed to be fed silage and barley. All feed was produced on the farm. Mineral supplements likely to be used on a farm like this were not accounted for, as minerals are a minor feed ingredient. In the forage scenario, steers were raised for slaughter at 30 months, while in the forage-grain scenario young bulls were raised for slaughter at 15 months. Animals were kept in houses with a slurry handling system and all slurry was used to fertilise the crops.

2.2. Livestock production

The number of cattle in each scenario was calculated based on a steady-state farm with 100 suckler cows slaughtered at age 91 months (Fig. 2). Calving was assumed to take place in March, with a 98% survival rate and an even distribution between heifer and bull calves. The slaughtered suckler cows were replaced each year by heifers aged 24 months, which gave a replacement rate of 17% of the suckler cows, while the remaining heifers were slaughtered. Bulls for breeding were bought at age 14 months and slaughtered at age 44 months.

A heavy-weight cattle breed was assumed and slaughter weight data were retrieved from Swedish statistics. The average growth rate for the livestock was calculated based on live weight at slaughter, initial weight and slaughter age. Live weight at slaughter was calculated based on carcass weight and a slaughter ratio of 53% (*i.e.* live weight to carcass weight) (Greppa Näringen, 2013). In both scenarios, 70% of the carcass weight was assumed to be bone-free meat.

Average growth rate and nutritional requirement data from Spörndly (2003) were used for calculating the energy requirement of the livestock (Table 2).

The young bulls in the forage-grain scenario were raised indoors and did not graze (except for the first period spend with the suckler cows, *i.e.* up to six months old). Their energy requirement was met by silage (70%) and barley (30%) (Table 1).

A total of 250 animals [49 heifers (10–21 months old), 98 steers (10–21 and 22–30 months old), 100 suckler cows, three bulls] and 152 animals [49 heifers (10–21 months old), 100 suckler cows, three bulls] grazed per season in the forage and forage-grain scenario, respectively, excluding calves. A grazing period of seven months was assumed for all grazing cattle except calves, which were separated from the cows after five months of grazing, and the last grazing period for steers, which were slaughtered after five months of grazing.

Methane emissions from enteric fermentation were calculated based on the gross energy in the feed consumed during 1 year as:

$$EF = \frac{GE \cdot Y_m \cdot 365}{55.65} \tag{1}$$

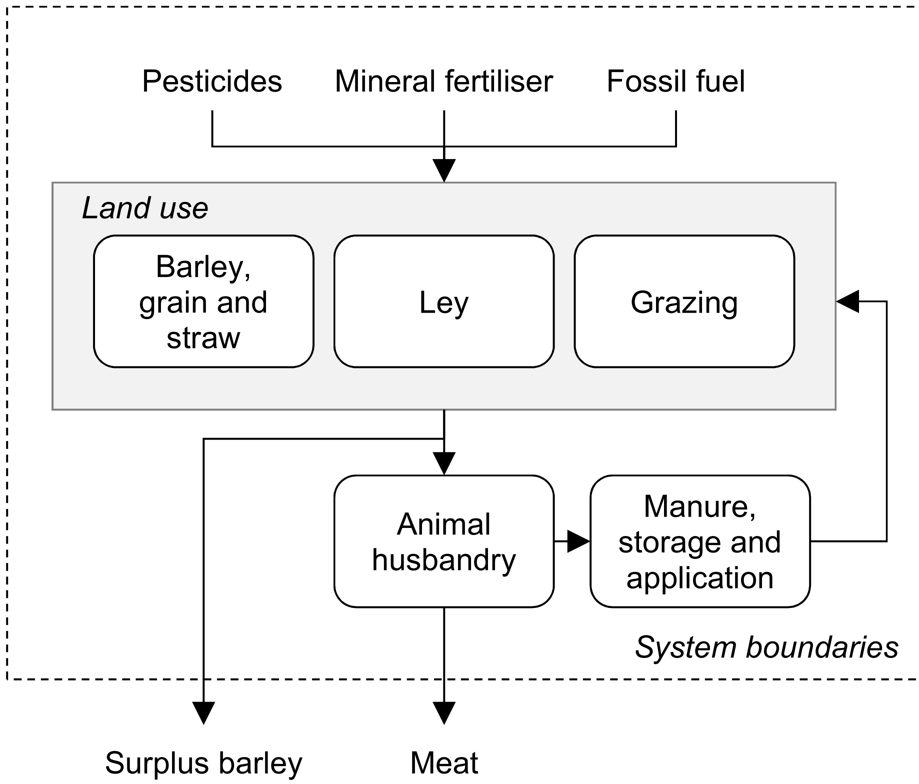

Fig. 1. System boundaries applied in life cycle assessment of Swedish beef production.

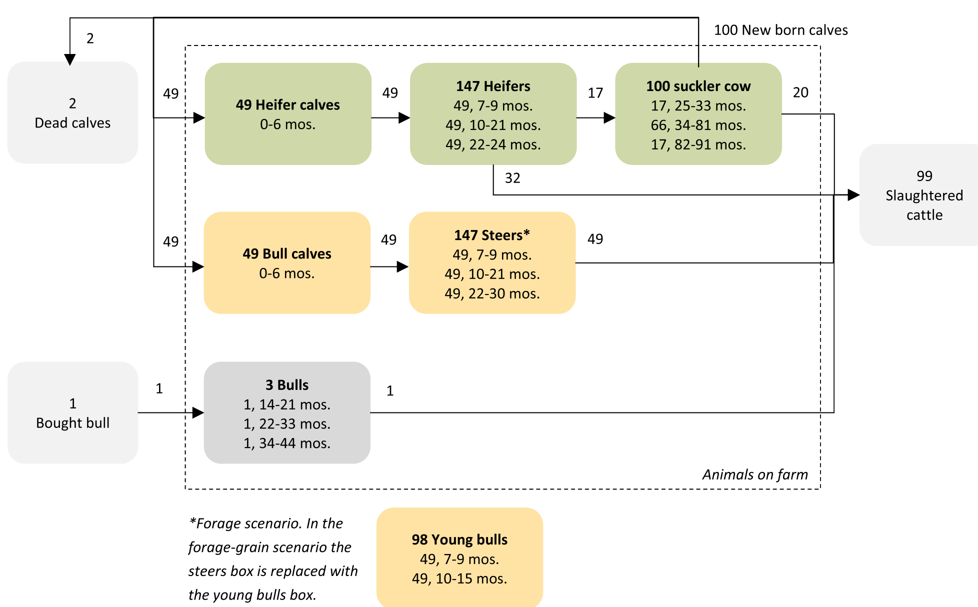

Fig. 2. Flowchart over number of animals on the studied farm during one year. Boxes represent number of animals in the different age groups and arrows represents flows between the different types of livestock and animals going to slaughter (mos. = months).

where EF is the methane emissions factor (kg CH_4 head^{-1} yr^{-1}), GE is gross energy intake (MJ head^{-1} yr^{-1}), Y_m is a methane conversion factor (% of GE) and 55.65 is the energy content in methane (MJ kg^{-1}) (IPCC, 2006). The Y_m factor was calculated based on Bertilsson (2016), which for suckler cows and bulls was:

$$CH_4 = 1.39 \cdot DMI - 0.091 \cdot FA \tag{2}$$

where CH_4 is methane emissions (MJ head^{-1} day^{-1}), DMI is total dry matter intake (kg head^{-1} day^{-1}) and FA is total feed content of fatty acids (g kg^{-1} DM) (Table 2). Considering total GE intake, this resulted in a Y_m value of around 6.4% for suckler cows and 6.5% for bulls. For growing livestock (heifers, young bulls and steers), Y_m was calculated as:

$$Y_m = (-0.046 \cdot ConcP + 7.1379)/100 \tag{3}$$

where $ConcP$ is the concentrate proportion (% of DM) (Bertilsson, 2016). For heifers and steers, this resulted in a Y_m value of 7.1% ($ConcP$ was 0),

Table 1
Characteristics of the beef production systems studied (forage/forage-grain scenario) (DM = dry matter).

	Calf	Heifers	Steers/ young bulls	Suckler cows	Bulls	Total
Initial weight (kg head^{-1})	45	310	310	603	648	–
Live weight at slaughter (kg head^{-1})	–	603	648	796	1132	–
Average growth (g day^{-1} and head^{-1})	145	535	463/ 1236	95	513	–
Slaughter age (months)	–	24	30/15	91	46	–
Carcass weight (kg head^{-1})	–	319[a]	344[a]	422[a]	600[b]	–
Slaughter quantity (head yr^{-1})	–	29	49	20	1	99
Total beef meat produced (kg bone-free meat yr^{-1})	–	7229	11785	4925	420	24359
Average fodder and bedding consumption (kg DM day^{-1})						
Silage	0	6.7	6.4/8.0	9.1	9.9	–
Barley, grain	0	0	0/2.7	0	0	–
Barley, straw	1.4	1.4	1.4	1.4	1.4	–
Grazing	0.3	6.7	6.4/0	9.1	9.9	–
Methane emissions factor (EF) (kg CH_4 head^{-1})						
Average month	0	5	5/6	6	7	–
Total lifetime	0	88	114/57	395	211	–

[a] Växa Sverige (2018).
[b] Assumed.

Table 2
Feed properties (silage is produced from grass-clover ley) (DM = dry matter).

	Gross energy (GE) (MJ kg^{-1} DM)	Digestible energy (MJ kg^{-1} DM)	Fatty acids (FA) (g kg^{-1} DM)
Grazing	18.5[a]	10.1[c]	20[a]
Barley grain	17.3[b]	13.2[d]	27[d]
Ley	18.5[a]	10.6[e]	20[a]

[f] Cederberg and Nilsson (2004).
[a] Assumed.
[b] Heuzé et al. (2015).
[c] Spörndly and Glimskär (2018).
[d] Spörndly (2003).
[e] Strid et al. (2012).

while the Y_m value for young bulls was 6.0% (*ConcP* was about 26). The Y_m value for calves was set to zero.

Methane emissions from enteric fermentation were higher per month for the young bulls in the forage-grain beef scenario, due to their higher growth rate, but since these bulls were slaughtered at a younger age the total emissions over the whole lifespan were lower (Table 1).

2.3. Crop cultivation

Two crop rotations were considered in each scenario; one with ley and spring barley and one with grazing and spring barley (Table 3). Although the livestock in the forage scenario were exclusively grass fed, barley was included in the crop rotation to function as a break crop and to provide straw for bedding. A 30% clover ley, cut three times per year for silage, was assumed. The crop rotations were adapted to fulfil the need for fodder and bedding in the two scenarios.

Table 3
Crop rotations in the forage and forage-grain scenarios.

	Forage scenario		Forage-grain scenario	
	Crop rotation 1	Crop rotation 2	Crop rotation 1	Crop rotation 3
Crop 1	Barley	Barley	Barley	Barley
Crop 2	3-year ley	6-year grazing	3-year ley	3-year grazing

Yield levels assumed for barley grain (of which 180 kg was used for sowing) and harvested ley were based on Swedish statistics and previous studies (Table 4). Harvest index for barley, *i.e.* the ratio between harvested biomass and total aboveground biomass, was used for calculating the amount of available straw (Nilsson and Bernesson, 2009). A harvest rate of 53% of total available straw was used. Less straw was harvested in the forage-grain scenario due to a lower need for bedding, and the remaining straw was returned to the soil (Table 4).

The different lifetimes of the animals resulted in varying number of cattle throughout the year, and thereby varying fodder demand and land use requirement in the two scenarios (Fig. 3).

The forage scenario required more land and the surplus land in the forage-grain scenario was assumed to be under continuous fallow. Both scenarios produced more barley grain (about 180 Mg yr^{-1}) than was consumed by the animals.

Mineral fertiliser and surplus manure (*i.e.* not needed in ley cultivation) was assumed to be applied to the barley fields, while only manure was used for the ley. Emissions from fossil fuel use in machinery, production of mineral fertiliser and use of electricity were calculated based on previous studies (Flysjö et al., 2008) (Appendix 1).

2.4. Manure and fertilisation

2.4.1. Manure production

The manure produced was calculated as volatile solids (VS), *i.e.* organic material:

$$VS = \left(GE \cdot \left(1 - \frac{DE}{100}\right) + (UE \cdot GE)\right) \cdot \left(\frac{1 - ASH}{18.45}\right) \quad (4)$$

where *VS* is organic dry matter (kg VS day^{-1}), *GE* is gross energy (MJ day^{-1}), *DE* is digestibility of the feed (%), *UE•GE* is urinary energy as a fraction of GE (0.04), *ASH* is ash content of manure as a fraction of dry matter feed intake (3% for barley grain, 6% for forage) and 18.45 is a conversion factor for dietary GE per kg of dry matter (MJ kg^{-1}) (IPCC, 2006). An ash content of 7% was used for calculating the volatile solids in straw (Table 5).

Liquid manure was assumed to be applied twice per season to the ley (Table 6). Surplus manure was spread on barley fields, which decreased the amount of mineral fertiliser required (see section *2.4.2*).

2.4.2. Fertilisation rates

Crop nutrient requirements were based on recommendations from the Swedish Board of Agriculture (2017) and fertilisation rates were calculated based on Strid et al. (2012):

Table 4
Crop yield used for fodder or bedding (kg dry matter ha^{-1}) (forage/forage-grain scenario).

	Harvest
Ley year 1–3	8500[a]
Barley, grain	4581[b]
Barley, straw	1695/1251[c]
Grazing	2460[d]

[a] Tidåker et al. (2016).
[b] StatisticsSweden (2018).
[c] Calculated.
[d] 60% utilisation rate of 4100 kg ha^{-1} (Spörndly and Glimskär, 2018).

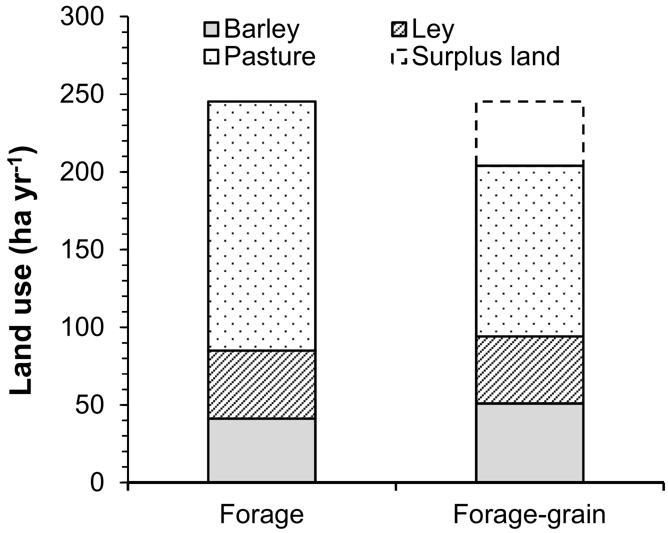

Fig. 3. Land use in the two beef systems studied (forage and forage-grain scenarios).

Table 5
Yearly manure production (Mg volatile solids per yr^{-1} and $farm^{-1}$) in the forage and forage-grain scenarios.

	Forage	Forage-grain
Liquid manure (incl. straw)	191	190
Manure grazing	137	94

Table 6
Manure properties (liquid manure and grazing).

	Manure
Nitrogen content (kg total N Mg^{-1} manure)	4.3[a]
Ammonia (% NH_4^+-N of total N)	50%[a]
Dry matter (DM) (%)	9%[a]
Volatile solids (% of DM)	83%[b]
C/N ratio	5[a]
Phosphorus (kg P Mg^{-1} manure)	0.6[a]
Potassium (kg K Mg^{-1} manure)	3.8[a]

[a] Swedish Board ofAgriculture (2017).
[b] Strid et al. (2012).

$$N_{applied} = n_r - (N_R + N_s) \quad (5)$$

where $N_{applied}$ is added nitrogen in fertiliser, n_r is nutrient requirement, N_R is residual nitrogen from the previous crop and N_S is nitrogen delivered from the soil via mineralisation due to long-term application of manure (Table 7).

Table 7
Nutrient requirements of the barley and ley crops and the fertilisation rate.

	Barley	Ley
Nitrogen requirement (n_r) (kg NH_4^+ ha^{-1} and yr^{-1})	85[a]	130[a]
Nitrogen from soil (N_s) (kg N ha^{-1} and yr^{-1})	20[b]	20[b]
Residual nitrogen (N_r) (kg N ha^{-1} and yr^{-1})	40[a]	0
Added nitrogen (kg NH_4^+ ha^{-1} and yr^{-1})	25	110
Phosphorus (kg P ha^{-1} and yr^{-1})	20[b]	14[b]
Potassium (kg K ha^{-1} and yr^{-1})	30	80

[a] Swedish Board ofAgriculture (2017).
[b] Strid et al. (2012).

Table 8
Emission factors for production of mineral fertiliser (Börjesson et al., 2010).

	CO_2 (g kg^{-1})	CH_4 (g kg^{-1})	N_2O (g kg^{-1})
Nitrogen	3200	3.1	11.5
Phosphorus	2900	7.2	0.29
Potassium	440	1.1	0.002

2.4.3. Mineral fertiliser

Mineral fertiliser was assumed to be applied to the spring barley and emissions from production of the fertiliser were calculated based on previous studies (Table 8).

2.4.4. Methane emissions from manure storage

Methane loss during storage of manure was calculated as:

$$\text{Storage}_{CH_4} = \text{VS}\cdot\text{B}_\text{O}\cdot 0.67\cdot\text{MCF} \quad (6)$$

where *VS* is volatile solids content in manure, B_O is maximum methane-producing capacity of the manure (set to 0.18 m^3 kg^{-1} VS), 0.67 converts methane from volume to mass and *MCF* is methane conversion factor, which was set to 3.5% for liquid manure (Rodhe et al., 2009) and 0.47% for manure deposited by grazing animals (Gavrilova et al., 2019).

2.4.5. Nitrous oxide emissions

Nitrogen inputs to the soil via plant litter, mineral fertiliser and application of manure lead to direct and indirect N_2O emissions. These were calculated according to Hergoualc'h et al. (2019) as:

$$N2O_{direct} = \text{EF}_\text{N}\cdot(\text{N}_\text{applied})\cdot\frac{44}{28} \quad (7)$$

$$N2O_{indirect} = \text{N}_\text{applied}\cdot(\text{F}_\text{A}\cdot\text{EF}_\text{D} + \text{N}_\text{leached}\cdot\text{EF}_\text{L})\cdot\frac{44}{28} \quad (8)$$

where EF_N is the emissions factor for direct N_2O emissions, $N_{applied}$ is total amount of nitrogen applied, F_A is the fraction of applied nitrogen emitted as ammonia, EF_D is emissions from volatilisation and re-deposition, $N_{leached}$ is nitrogen lost by leaching, EF_L is an emissions factor for N_2O emissions due to nitrogen leaching and the fraction $\frac{44}{28}$ converts nitrogen into N_2O (Table 9).

Nitrogen input from crop residues was calculated based on nitrogen content and residual biomass (Table 10).

2.5. Soil organic carbon changes

The Introductory Carbon Balance Model (ICBM) was used for modelling soil carbon changes. It is designed for agricultural soils and

Table 9
Direct nitrous oxide (N_2O) emissions factor (EF_N), fraction of applied nitrogen emitted as ammonia (F_A), nitrogen lost by leaching ($N_{leached}$), emissions from volatilisation and re-deposition (EF_D) and N_2O emissions due to nitrogen leaching (EF_L).

	EF_N	F_A	$N_{leached}$	EF_D	EF_L
Liquid manure					
House	0.10%	7%[b]	–	1%[a]	–
Storage	0.50%[a]	3%[b]	–	1%[a]	–
Application	0.6%[a]	15%[b]	24%[a]	1%[a]	1.1%[a]
Mineral fertiliser	1.6%[a]	1.20%[c]	24%[a]	1%[a]	1.1%[a]
Plant residues	0.6%[a]	1%[d]	24%[a]	1%	1.1%[a]
Grazing	0.6%[a]	8%[b]	24%[a]	1%[a]	1.1%[a]

[a] Hergoualc'h et al. (2019).
[b] Strid et al. (2012) 15% = spring application.
[c] Ahlgren et al. (2009).
[d] Assumed based on de Ruijter and Huijsmans (2012).

Table 10
Nitrogen content in crop residues (IPCC, 2006).

	Aboveground biomass	Belowground biomass
Fallow	1.5%	1.2%
Barley	0.7%	1.4%
Ley	2.5%	1.6%
Grazing	1.5%	1.2%

can be extended to include several soil pools (Kätterer and Andrén, 2001). A version with three young pools, one for aboveground biomass (Y_a), one for belowground biomass (Y_b) and one for manure (Y_m) (Kröbel et al., 2016), was used in this paper. The relationship between the three young pools and the old pool (O) is described by:

$$O(t)=\left(O_{t-1}-\left(\frac{h_a\cdot k_Y}{(k_O-k_Y)}\cdot\left(Y_{a_{t-1}}+i_{a_{t-1}}\right)+\frac{h_b\cdot k_Y}{(k_O-k_Y)}\cdot\left(Y_{b_{t-1}}+i_{b_{t-1}}\right)+\frac{h_m\cdot k_Y}{(k_O-k_Y)}\cdot\left(Y_{m_{t-1}}+i_{m_{t-1}}\right)\right)\right)\cdot exp^{-k_O\cdot r_e}+\left(\frac{h_a\cdot k_Y}{(k_O-k_Y)}\cdot\left(Y_{a_{t-1}}+i_{a_{t-1}}\right)+\frac{h_b\cdot k_Y}{(k_O-k_Y)}\cdot\left(Y_{b_{t-1}}+i_{b_{t-1}}\right)+\frac{h_m\cdot k_Y}{(k_O-k_Y)}\cdot\left(Y_{m_{t-1}}+i_m\right)\right)\cdot exp^{-k_Y\cdot r_e} \tag{9}$$

where the young pools are described by:

$$Y_{[a,b,m]}(t)=\left(Y_{[a,b,m]_{t-1}}+i_{[a,b,\ m]_{t-1}}\right)\cdot exp^{-k_Y\cdot r_e} \tag{10}$$

and where k_Y and k_O are constants representing the decay rate of the two pools (Andrén et al., 2004; Andrén and Kätterer, 1997). The total SOC content each year is the sum of the pools. The humification parameter h was varied for the three young carbon pools, where the aboveground humification parameter (h_a) was set to 0.13 and the belowground parameter (h_b) calculated as:

$$h_b=h_a\cdot 2.3 \tag{11}$$

The humification parameter for manure (h_m) was set to 0.31 (Kätterer et al., 2008). The carbon input (i) from aboveground and belowground plant material was calculated based on Bolinder et al. (2007). The amount of aboveground crop residues (Y_S) was calculated from yield of the harvested product (Y_P) (Table 11):

$$Y_S=Y_P\cdot\frac{(1-HI)}{HI} \tag{12}$$

where HI is harvest index for the specific crop (Unkovich et al., 2010). The HI value for ley included harvest losses, while additional losses from storage (5%; Grovfoderverktyget, 2019) and feeding (4%; Strid et al., 2012) were subtracted from the yield and assumed to be returned to the field. Carbon stored in the harvested product (C_P), aboveground residues (C_S) and belowground residues (C_R) from the different biomass fractions was calculated as:

$$C_P=Y_P\cdot C_C \tag{13}$$

$$C_S=Y_S\cdot C_C \tag{14}$$

$$C_R=Y_P\cdot\frac{1}{(S:R\cdot HI)}\cdot C_C \tag{15}$$

where C_C is the carbon content (set to 45% for all crops) and $S:R$ is shoot:root ratio. The carbon from extra-root material (root exudates and material from root turnover) was also included:

$$C_E=C_R\cdot Y_E \tag{16}$$

where Y_E is extra-root C, which was set to 0.65 according to Bolinder et al. (2007). Total net primary production was then taken as the sum of all four carbon fractions.

Aboveground (i_a) and belowground (i_b) carbon input to the SOC pool was calculated as:

$$i_a=C_P\cdot S_P+C_S\cdot S_S \tag{17}$$

$$i_b=C_R\cdot S_R+C_E\cdot S_E \tag{18}$$

where S is the proportion of carbon in the specific plant fraction returned to the soil each year. Carbon input from ley roots was added in the last year of the ley cultivation, while extra-root input was added each year (Bolinder et al., 2012). The carbon/nitrogen (C/N) ratio for manure (Table 6) was used for calculating the carbon input from manure application (grazing and liquid manure) (Table 12).

The r_e parameter in Equations (9) and (10) describes external factors such as soil temperature and water-holding capacity (Karlsson, 2012). The initial SOC content was set to 77.8 Mg ha^{-1} (2.5% topsoil, 25 cm) and the r_e parameter was set to 0.91 (representing the median value for fallow land with clay soil in the study region) (Hammar et al., 2017). The initial SOC content was varied in a sensitivity analysis, based on the 5th and 95th percentile of the SOC values for fallow fields on clay soils in the study region (Table 13).

2.6. Climate metrics

Two climate metrics were used in this study, GWP and AGTP (referred to as temperature response in the Results section). Global warming potential is the most commonly used climate metric and is thus useful for comparison with other studies. However, it has the disadvantage that it does not consider the timing of GHG fluxes. Therefore it is valuable to use two climate metrics, which can display more information (Levasseur et al., 2016).

Both metrics are based on radiative forcing, which is measured in Wm^{-2} at the top of the troposphere. Greenhouse gases have different radiative efficiencies and remain in the atmosphere for varying time periods, and the radiative forcing caused by each impulse emission thus varies over time. The GWP characterisation factor for GHGs is calculated as:

$$GWP_x(H)=\frac{CRF_x(H)}{CRF_{CO_2}(H)} \tag{19}$$

where H is the time horizon (commonly 100 years) and CRF is the

Table 11
Values used for calculating carbon fluxes in the ICBM model.

	Yield product (Y_P)	Harvest index (HI)	Shoot:root (S:R) ratio[b]
Fallow	3000[a]	–	2.0
Spring barley	4581	0.59	9.5
Ley, year 1–3	8500	0.80	2.0
Grazing	2460	0.60	2.0

[a] Left in the field.
[b] M. Bolinder, personal communication 2019.

Table 12
Carbon input (i) for aboveground (a) and belowground (b) biomass and manure (m) used in the ICBM model (forage/forage-grain scenario) (Mg C ha^{-1}).

	i_a	i_b	i_m
Fallow	1.4	1.1	0
Spring barley	0.7/0.9	0.6	0.04
Ley, year 1	1.3	1.6	1.2
Ley, year 2	1.3	1.6	1.2
Ley, year 3	1.3	3.9	1.2
Grazing	0.7	1.5	0.2

Table 13
Initial soil organic carbon content of fallow land on clay soils in the study region (N = 535).

	Mg ha^{-1}	%
5th percentile	38.5	1.2
50th percentile	77.8	2.5
95th percentile	318.7	10.1

cumulative radiative forcing of an impulse emission of the specific gas *x* compared with an impulse emission of carbon dioxide during the same period. According to the IPCC (AR5 report), the GWP_{100} factor for carbon dioxide, biogenic/fossil CH_4 and N_2O is 1, 34/36 and 298 (including climate-carbon feedbacks), respectively (Myhre et al., 2013b). The GWP was calculated by adding up the yearly emissions during the study period and then multiplying the cumulative GHG fluxes by the GWP_{100} factors. The AGTP of each GHG emission is described by:

$$\mathrm{AGTP_x(H)} = \int_0^H \mathrm{RF_x(t)R_T(H-t)dt} \tag{20}$$

where *RF* is the radiative forcing and R_T is the temperature impulse response function due to a unit change in RF from a pulse emission of the specific greenhouse gas *x* (Fig. 4).

The total temperature response is the sum of the AGTP of all GHG emissions (*E*) during the study time horizon (*H*) (measured in degrees K):

$$\mathrm{Temperature\ \ response\ \ (H)} = \sum_x \int_0^H \mathrm{E_x(t)AGTP_x(H-t)dt} \tag{21}$$

where *t* is the time of emission or uptake and x is the gas (CO_2, CH_4, N_2O).

3. Results

3.1. Soil organic carbon

The crop rotation containing ley resulted in the greatest build-up of SOC content per hectare, since the high biomass productivity (as a result of fertilising) in comparison with grazed land (Table 4) led to a larger carbon input from crop residues (Fig. 5). The long rotation grazing (crop rotation 2) gave a higher SOC content than the short grazing rotation (crop rotation 3) or continued fallow, which resulted in the lowest SOC content, although the difference between the two grazing rotations and fallow was relatively small.

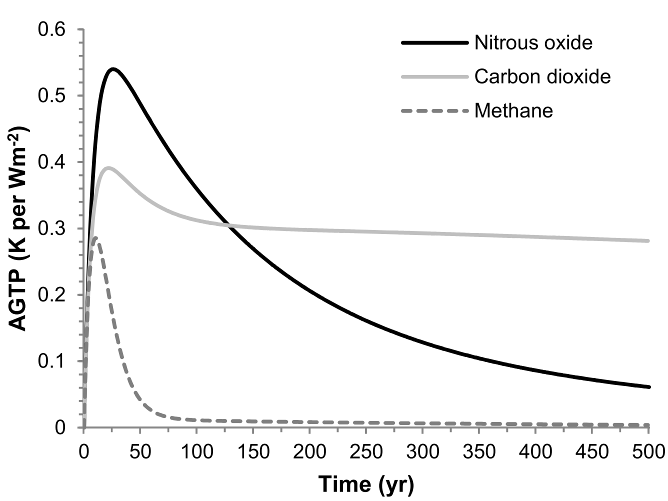

Fig. 4. Absolute Global Temperature change Potential (AGTP) per unit radiative forcing.

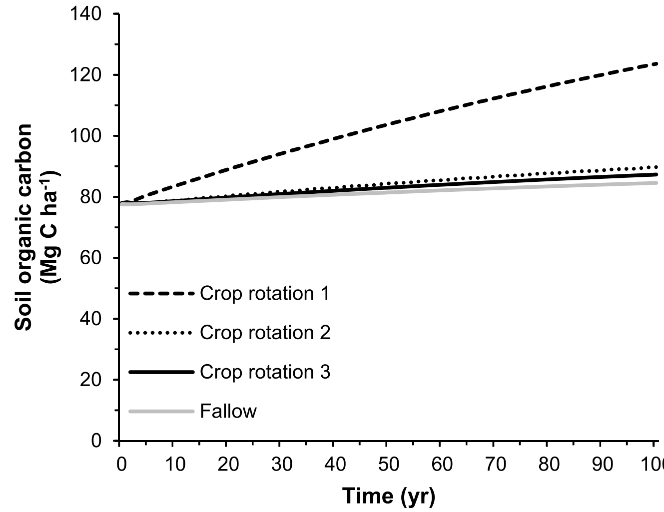

Fig. 5. Soil organic carbon stock per hectare for the different crop rotations in the forage scenario (crop rotation 1 - 1 yr barley, 3-yr ley and crop rotation 2–1 yr barley, 6 yr grazing), the forage-grain scenario (crop rotation 1- 1 yr barley, 3-yr ley and crop rotation 3–1 yr barley, 3 yr grazing) and the reference land use (continued fallow).

The total soil carbon stock over time was higher for the forage beef scenario than the forage-grain scenario (Fig. 6). However, the difference was relatively small when surplus land was included in the forage-grain scenario. The average carbon sequestration rate was around 0.2 Mg C ha^{-1} and yr^{-1} for both scenarios.

3.2. Global warming potential

The forage beef scenario resulted in higher GWP than the forage-grain beef scenario (650 and 570 Mg CO_2-eq farm^{-1} and yr^{-1}, corresponding to 27 and 23 kg CO_2-eq kg^{-1} bone-free beef, respectively) (Fig. 7). The CH_4 emissions from enteric fermentation had the highest warming impact. The carbon sequestration effect was higher in the forage scenario (−180 Mg CO_2 farm^{-1} and yr^{-1}) than in the forage-grain beef scenario (−150 Mg CO_2 farm^{-1} and yr^{-1}), due to more land being occupied by ley, resulting in higher biomass input to soil. The increased soil carbon stocks offset 21–22% of the total GWP in the forage-grain and forage scenario, respectively. On including the surplus land in the forage-grain scenario (continued fallow), the carbon sequestration was −160 Mg CO_2 farm^{-1} and yr^{-1}.

Including the net land use effect, *i.e.* the yearly difference compared with continued fallow, lowered the carbon sequestration potential (to around −120 and −100 Mg CO_2 farm^{-1} and yr^{-1} for the forage and forage-grain scenario, respectively), which offset about 15–16% of the GWP for the two scenarios. Including avoided emissions from cutting the fallow once per year and N_2O soil emissions lowered the total GWP to 640 and 560 Mg CO_2-eq farm^{-1} and yr^{-1} for the forage and forage-grain fed suckler cow systems, respectively.

3.3. Temperature

The temperature response revealed how the climate impact varied over time (Fig. 8). Methane emissions from enteric fermentation had the highest temperature increase initially, but these started to stabilise after around 50 years as a result of the short atmospheric lifetime of methane.

Even though the carbon sequestration effect was largest for the forage scenario, the temperature response was higher than for the more intensive forage-grain fed cattle system due to higher CH_4 emissions from the animals, which lived longer in that forage scenario (Fig. 9).

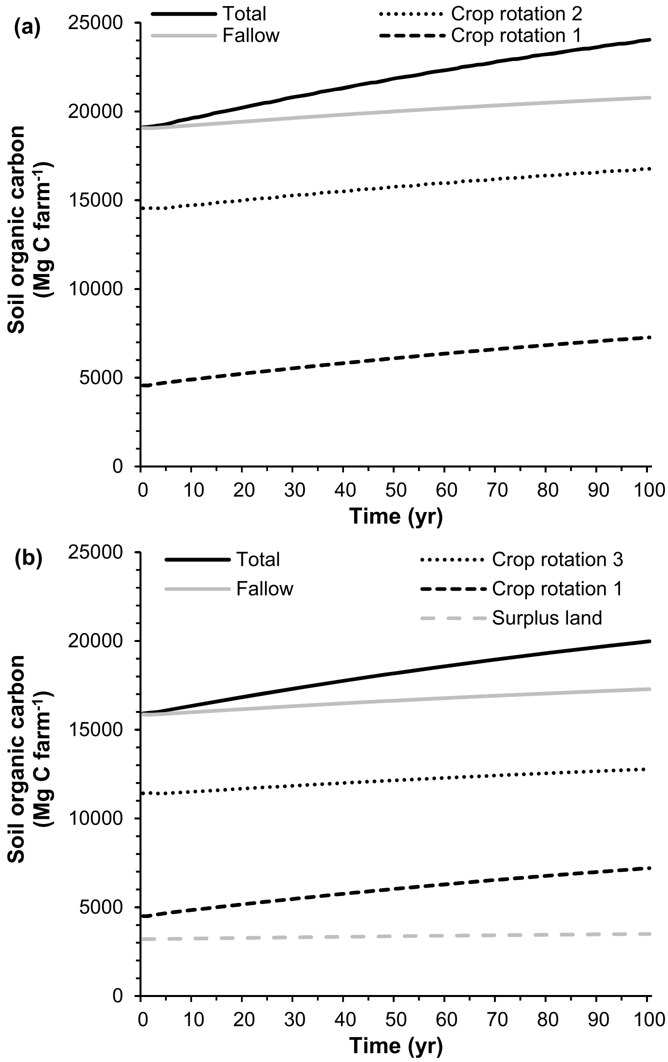

Fig. 6. Soil organic carbon stock in a) the forage beef scenario (total includes crop rotation 1 - 1 yr barley, 3-yr ley and crop rotation 2–1 yr barley, 6 yr grazing, 245 ha) and b) the forage-grain beef scenario (total includes crop rotation 1- 1 yr barley, 3-yr ley and crop rotation 3–1 yr barley, 3 yr grazing, 204 ha, *i.e.* excluding 41 ha surplus land).

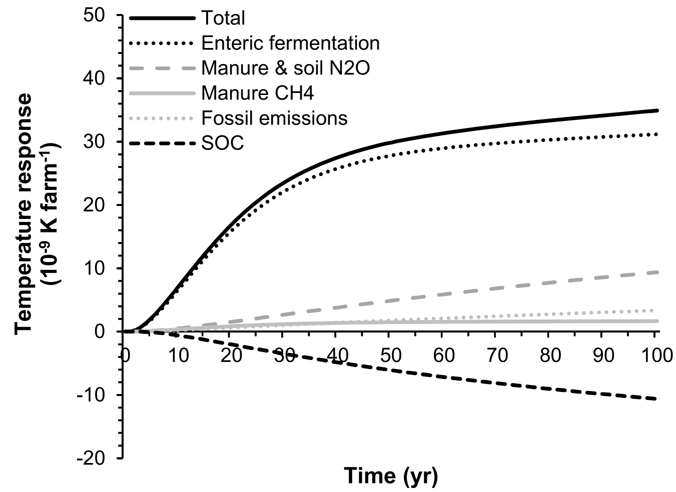

Fig. 8. Temperature response of the forage beef scenario. Manure CH_4 includes methane emissions from manure management (storage and application). Manure and soil N_2O includes nitrous oxide emissions from manure management and plant residues. Soil organic carbon (SOC) excludes net land use effect.

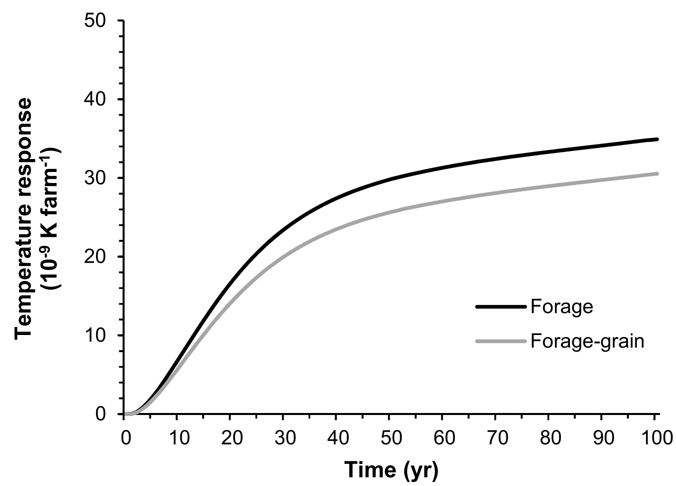

Fig. 9. Temperature response of the forage and forage-grain beef scenarios. Excluding net land use effect and surplus land in forage-grain scenario.

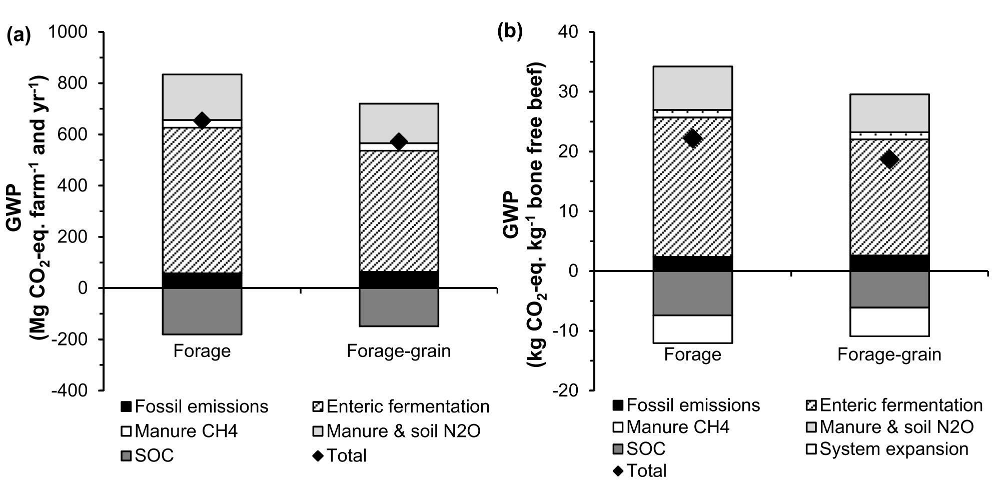

Fig. 7. Global warming potential (GWP) of the forage and forage-grain beef scenarios with the functional unit (a) farm^{-1} and yr^{-1} and (b) kg^{-1} bone-free beef, with system expansion for surplus barley grains. Fossil emissions includes fossil energy use, (field operations and energy use at the farm) and production and application of mineral fertilisers (including soil N_2O emissions). Manure CH_4 includes methane emissions from manure management (storage and application). Manure and soil N_2O includes nitrous oxide emissions from manure storage and application of manure and plant residues. Soil organic carbon (SOC) excludes net land use effect and surplus land in the forage-grain scenario.

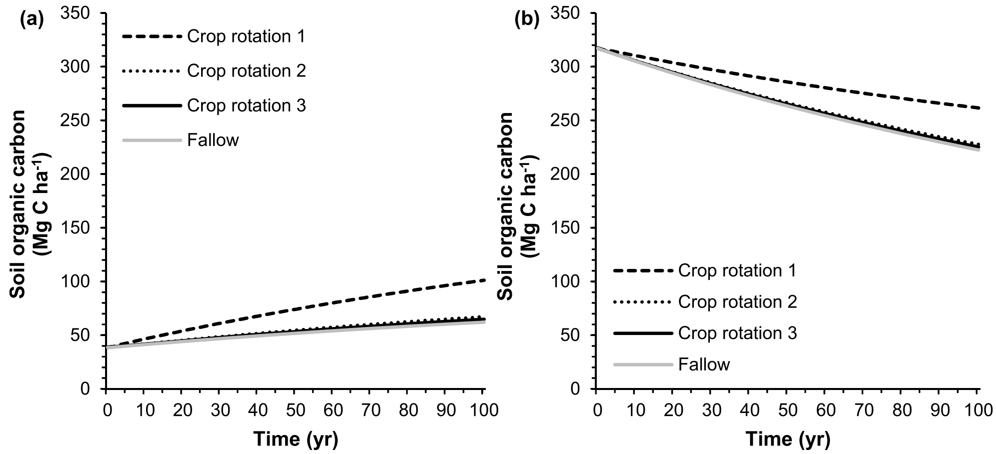

Fig. 10. Sensitivity analysis of (a) low initial soil organic carbon content (38.5 Mg ha^{-1}, 1.2%) and (b) high initial soil organic carbon content (318.7 Mg ha^{-1}, 10.1%), excluding net land use effect compared with continued fallow. For details of crop rotations, see Table 3.

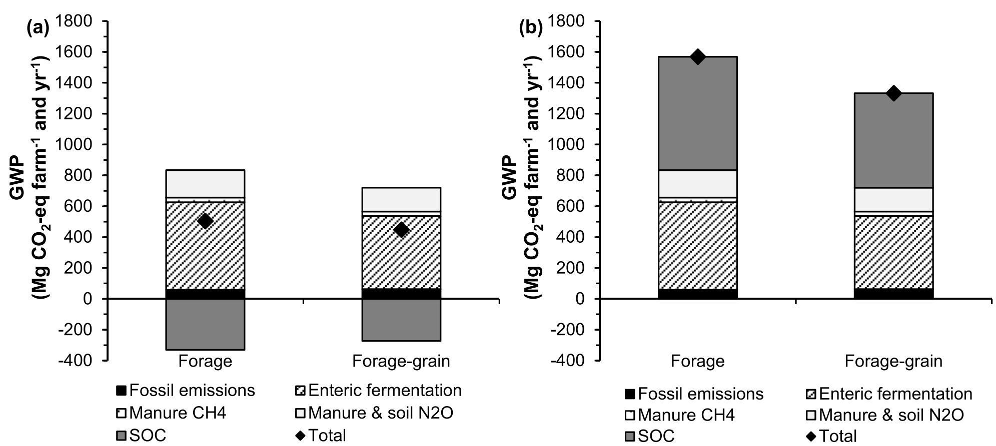

Fig. 11. Global warming potential (GWP) of the forage and forage-grain beef scenarios with (a) low initial soil organic carbon content (38.5 Mg ha^{-1}, 1.2%) and (b) high initial soil organic carbon content (318.7 Mg ha^{-1}, 10.1%), excluding net land use effect compared with continued fallow.

3.4. Sensitivity analysis

3.4.1. Initial soil organic carbon

The influence of initial SOC was tested in a sensitivity analysis. Low initial SOC resulted in carbon build-up over time (about 0.4 Mg C ha^{-1} and yr^{-1}) (Fig. 10a), while high initial SOC content resulted in decreased carbon stocks for all crop rotations (loss of about 0.8 Mg C ha^{-1} and yr^{-1}) (Fig. 10b).

The lower initial SOC content resulted in higher carbon uptake in soils, which lowered the overall climate impact (Fig. 11a). Conversely,

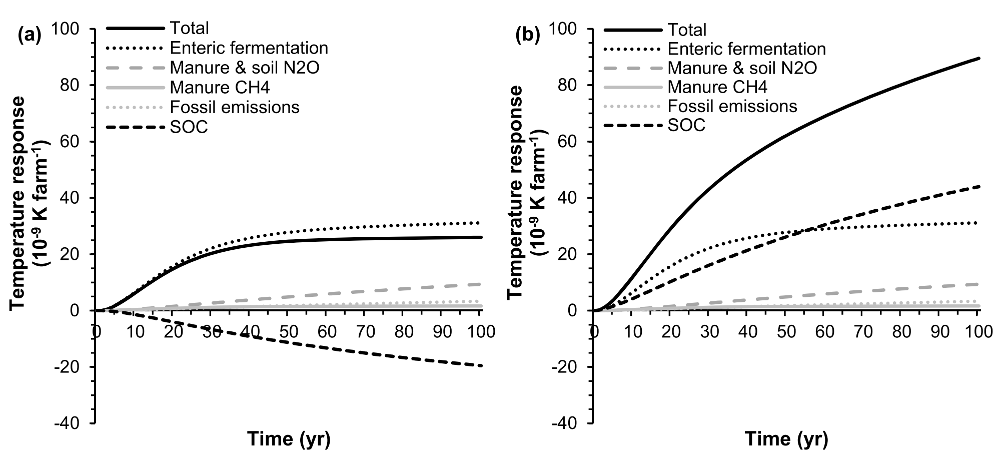

Fig. 12. Temperature response in the forage scenario of (a) low initial soil organic carbon content (38.5 Mg ha^{-1}, 1.2%) and (b) high initial soil organic carbon content (318.7 Mg ha^{-1}, 10.1%), excluding net land use effect compared with continued fallow.

the higher initial SOC content resulted in release of soil carbon, which increased the warming potential (Fig. 11b) and the temperature response (Fig. 12).

However, when the net land use effect was considered (*i.e.* the yearly difference compared with continued fallow), the initial SOC content was of minor importance, since the effect for the reference land use was similar. Thus initial SOC content had a small effect on the overall climate impact when the net land use effect was considered. Including soil carbon in the LCA counteracted around 13–16% of GHG emissions, irrespective of initial SOC content. The choice of reference land use for the assessment was therefore important. In this case, using fallow land as a reference can be justified by the fact that Sweden currently has approximately 130,000 ha of fallow land with little alternative use than cultivation of animal feed. There is currently a food strategy in place that promotes increased production of meat and dairy (Government Offices of Sweden, 2017), which could incentivise the use of low-value cropland for ruminant production. However, a potential alternative use of this land is forest. Using forest as a point of reference would heavily affect the results, as meat production would then come with a carbon opportunity cost considering the potential to store large amounts of carbon through afforestation (Hayek et al., 2021).

3.4.2. Carbon input from manure

The carbon input from liquid manure and grazing animals was calculated based on the C/N ratio of liquid manure in the Swedish fertilisation recommendations. However, C/N ratio can vary (from 6% to 20% in Carlsson and Uldal (2009)) and was therefore altered in a sensitivity analysis (Table 14). Higher C/N ratio increased the carbon input from liquid manure and grazing and resulted in carbon sequestration rates of around 0.3 Mg ha^{-1} and yr^{-1}, which counteracted around 31–32% of the total GWP in the two scenarios studied. The temperature response is shown in Fig. 13.

3.4.3. Carbon input from roots

The share of belowground biomass to aboveground biomass, *i.e.* shoot:root (S:R) ratio, of perennial grasses is an uncertain factor and was therefore tested in a sensitivity analysis. A higher S:R ratio means that more biomass is allocated to aboveground biomass, and consequently less carbon enters the soil via belowground crop residues (Table 15). The carbon input from aboveground residues was constant in the sensitivity analysis, *i.e.* the total amount of carbon input to the soil was altered.

A higher S:R ratio resulted in a higher temperature response (Fig. 14), due to lower carbon input from belowground biomass (roots). Conversely, a lower S:R ratio decreased the temperature response, since the carbon input from roots was higher (*i.e.* the total carbon input from crop residues was increased).

Table 14
Carbon input from manure (i_m) with different carbon/nitrogen (C/N) ratios.

C/N ratio	5	10
Liquid manure	1.2	2.4
Grazing	0.2	0.5

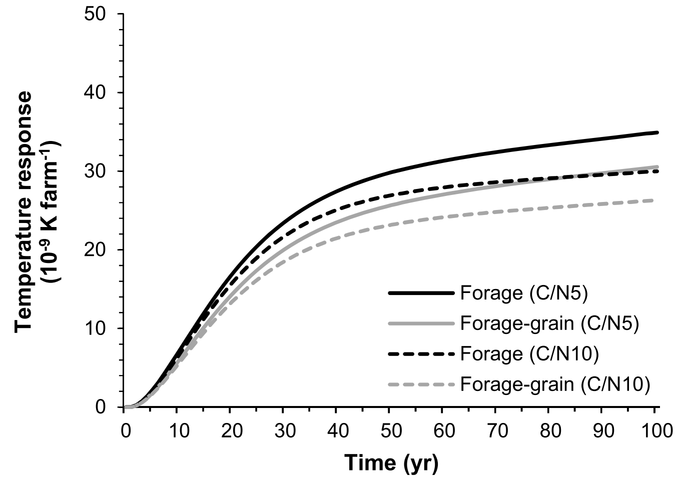

Fig. 13. Temperature response of different carbon/nitrogen (C/N) ratios in manure.

Table 15
Shoot:root (S:R) ratio (used in Eq. (15)) for perennial grasses (ley, grazing and fallow) varied in sensitivity analysis, and the effect on carbon input from belowground biomass (i_b) used in the ICBM model and on global warming potential (GWP).

	−20%		+20%
S:R ratio	1.6	2.0	2.4
Carbon input belowground (i_b)			
Fallow	1.4	1.1	0.9
Ley, year 1–2 (year 3)	1.9 (4.9)	1.6 (3.9)	1.3 (3.3)
Grazing	1.9	1.5	1.3
GWP incl. SOC (Mg CO_2-eq $farm^{-1}$ and yr^{-1})			
Forage-grain	590 (−10%)	650	700 (+7%)
Forage	520 (−9%)	570	610 (+6%)

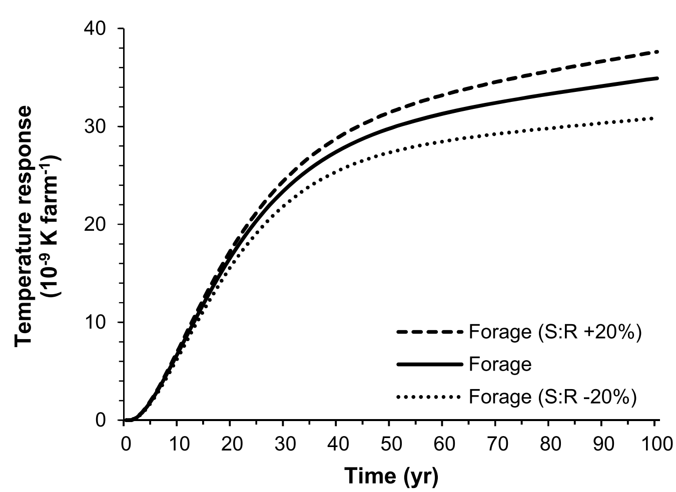

Fig. 14. Temperature response of different shoot:root (S:R) ratios in the forage scenario.

4. Discussion

In this assessment of the climate impact of SOC fluxes in beef production, the GWP, excluding SOC changes, was found to be 25 and 30 kg CO_2-eq kg^{-1} bone-free meat for the forage-grain and forage scenario (with system expansion for surplus barley grain), respectively. This is somewhat lower than previously reported values for Swedish beef from suckler herds (Moberg et al., 2019). The difference is explained by not accounting for any losses of adult animals in this study and slight differences in calculations in CH_4 emissions from enteric fermentation and slaughter weight. The carbon sequestration potential was highest in the forage-fed suckler cow scenario, since it included a higher share of ley than the forage-grain scenario. The SOC changes reduced the GWP by around 21–22% for the two scenarios (15–16% when the net land use effect was considered, *i.e.* the difference compared with continuous fallow). Similar effects have been observed in studies in climates similar to Sweden. In a study by Alemu et al. (2017), emissions were reduced by 12–25% (beef system, western Canada), while in a study by Trydeman Knudsen et al. (2019) emissions were decreased by 5–18% (dairy systems, western Europe). It should be noted, however, that carbon sequestration levels off with time as the soil reaches a new equilibrium state (IPCC, 2019). In the present case, however, *i.e.* under the conditions at the specific site and in the production systems considered, modelling predicted that the soil will keep sequestering carbon for hundreds of years.

In line with previous LCAs of beef meat, the largest contributor to the climate impact was CH_4 emissions from enteric fermentation. Methane emissions increased when the animals were fed forage and grain compared with only forage (due to higher gross energy intake in the more intensive forage-grain production), but the overall emissions were higher in the forage-fed suckler cow system since the animals lived longer. Although soil carbon changes contributed to non-negligible negative emissions, they could not offset CH_4 emissions from the animals.

The highest carbon sequestration potential was found in the forage-fed scenario for beef cattle. However, the more intensive forage-grain suckler cow system required less land, and this surplus land could potentially be used for other purposes, *e.g.* bioenergy production or carbon sequestration, which could give additional climate impact benefits. Biomass productivity is another important factor for soil carbon changes. Higher productivity results in more residues, and thus higher carbon input to the soil. There are many factors influencing plant productivity, *e.g.* crop, soil texture, management practice, geographical location and climate. Variations in the results can thus be expected due to regional differences in these factors. There are also uncertainties in the proportion of aboveground to belowground biomass, which is why the influence of varying S:R ratio for perennial grasses was tested in a sensitivity analysis (Fig. 13).

There are large uncertainties in modelling SOC changes, especially for long-term grasses growing on pastureland. Estimating the yearly carbon input is more complex for perennial grasses than for annual crops, since only a certain part of the root biomass is turned over annually (Poeplau, 2016). There is also a lack of long-term data on biomass productivity on pastureland and on the pasture utilisation rate of grazing animals. To enable more accurate accounting for soil carbon changes in LCA on forage-based ruminant systems, further refinement of soil carbon models including improved calibration against relevant measurements is crucial.

In this study, it was assumed that all feed was produced on farm. That is not uncommon in Swedish ruminant production, but in more intensive ruminant systems, especially in dairy systems, imported feed including soy is commonly used (Cederberg et al., 2009). Accounting for changes in carbon stocks, both in soils and in standing biomass as a consequence of land use change, in all feed used is important to understand the net climate effect, but can be challenging as the exact origin of the feed is often unknown. In this study, this complication was avoided by using only local feeds.

In this study, all grazing was assumed to be performed on arable land included in the crop rotations. In an international perspective grazing on permanent grassland is more common, but grazing grass-clover ley in crop rotations is common in Sweden, where permanent grassland only makes up 1% of land area and where 40% of arable land is used for grass-clover leys in rotations (Swedish Board of Agriculture, 2020). This is also an important part of mixed agro-ecological systems (Karlsson and Röös, 2019). Another type of pasture in Sweden is semi-natural pasture, where land not suited for crop cultivation is grazed permanently. Previous assessments have shown that this type of pasture sequesters on average less than 0.1 Mg C ha^{-1} and yr^{-1} (Karltun et al., 2010), which is lower than the average found in this study (~0.2 Mg C ha^{-1} and yr^{-1} for the total land use). Conducting an LCA of a beef system with semi-natural pasture would also require another type of reference land use. One alternative could be afforestation, which could increase carbon stocks but would have detrimental effects on biodiversity, since semi-natural pasture in Sweden has high biological values and constitutes a relatively small share of the land area (Swedish Board of Agriculture, 2019).

The importance of time perspective for short-lived GHGs like CH_4 and the limitations of GWP have been discussed previously (*e.g.* in Allen et al., 2016). In the present analysis, the climate impact of beef was assessed using two climate metrics. In addition to GWP, the temperature response of greenhouse gas emissions was calculated using AGTP. The results showed that the temperature response of CH_4 emissions increased during the first five decades, after which it started to level off as a result of the relatively short atmospheric lifetime of methane (Fig. 8). This illustrates an important characteristic of short-lived GHGs whereby a change in emissions rate has a large influence on the temperature, while constant emissions of CH_4 over time do not lead to (much) additional warming (Allen et al., 2016). However, there is still potential to mitigate climate change by decreasing CH_4 emissions.

Depending on time perspective, the relative influence of different GHGs on the total climate impact varies. This variation would have been missed if only the GWP metric had been used. In addition, the temperature response metric (AGTP) displays the time-dependent climate impact and considers the timing of GHG emissions. The GWP value does not reflect when in a time period a specific gas is emitted, as emissions are multiplied by the same characterisation factor independently of emission (or uptake) year. The AGTP, on the other hand, considers the year of the flux, so that the climate response of one kg GHG emissions depends on the emission year. This timing can have large effects for biobased systems, especially forestry systems, but also for agricultural systems with varying fluxes over years. In the present study however, this timing was not crucial, as emissions and uptake were rather constant over the years, but AGTP still added additional information in comparison with the GWP, especially with regard to methane. The GWP value also fails to capture the great increase in temperature response that would follow from an increase in CH_4 emissions with an increase in ruminant livestock numbers (Lynch et al., 2021).

Despite higher soil carbon sequestration in the forage scenario, the forage-grain scenario modelled here performed better in terms of climate impact, due to shorter animal lifetimes leading to lower CH_4 emissions from enteric fermentation. However, the differences are small when considering the large uncertainties in LCA of livestock systems (Röös, 2013). Therefore, it is possible that some forage-based systems can perform better than some more grain-based systems, depending above all the soils potential to sequester carbon and growth rates of animals. New technologies, for example the promising red seaweed feed supplement (Roque et al., 2021), could potentially drastically reduce CH_4 emissions, which could heavily influence the climate impact. However, when designing sustainable livestock systems a wide range of sustainability aspects needs to be accounted for, including other environmental aspects, animal welfare and a range of socio-economic aspects.

5. Conclusions

Time-dependent modelling of climate impact showed that CH_4 emissions from enteric fermentation in beef cattle caused a rapid temperature response during the first 50 years, after which the response levelled off as a result of the limited atmospheric life time of methane. However, sustained production and associated CH_4 emissions would maintain the temperature response and contribute to climate damage. After 50 years, emissions of N_2O from manure, fertiliser and crop residues, and of carbon dioxide from fossil energy use, caused the temperature response to continue to increase slowly, due to the long atmospheric lifetime of these gases. Soil organic carbon sequestration offset some of the temperature response. Including soil organic carbon counteracted around 15–22% of the global warming potential of beef meat, depending on the system boundaries selected for land use and production intensity. Suckler cow systems using only forage (silage) had greater potential to sequester carbon than forage-grain systems, since a larger share of grass-clover ley was included in the crop rotation, but overall emissions were higher since the animals took longer to reach slaughter weight.

CRediT authorship contribution statement

Torun Hammar: Conceptualization, Methodology, Formal analysis, Writing – original draft. **Per-Anders Hansson:** Conceptualization, Writing – review & editing, Supervision. **Elin Röös:** Conceptualization, Methodology, Validation, Writing – review & editing, Supervision.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2021.129948.

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