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Assessing the climate and eutrophication impacts of grass cultivation at five sites in Sweden

Johan Nilsson[a], Pernilla Tidåker[a], Cecilia Sundberg[a,b], Kajsa Henryson[a], Brian Grant[c], Ward Smith[c] and Per-Anders Hansson[a]

[a]Department of Energy and Technology, Swedish University of Agricultural Sciences (SLU), Uppsala, Sweden; [b]Division of Industrial Ecology, Department of Sustainable Development, Environmental Science and Engineering, KTH Royal Institute of Technology, Stockholm, Sweden; [c]Ottawa Research and Development Centre, Agriculture and Agri-Food Canada, Ottawa, Canada

ABSTRACT
In this study, Life Cycle Assessment (LCA) methodology was combined with the agro-ecosystem model DNDC to assess the climate and eutrophication impacts of perennial grass cultivation at five different sites in Sweden. The system was evaluated for two fertilisation rates, 140 and 200 kg N ha^{-1}. The climate impact showed large variation between the investigated sites. The largest contribution to the climate impact was through soil N_2O emissions and emissions associated with mineral fertiliser manufacturing. The highest climate impact was predicted for the site with the highest clay and initial organic carbon content, while lower impacts were predicted for the sandy loam soils, due to low N_2O emissions, and for the silty clay loam, due to high carbon sequestration rate. The highest eutrophication potential was estimated for the sandy loam soils, while the sites with finer soil texture had lower eutrophication potential. According to the results, soil properties and weather conditions were more important than fertilisation rate for the climate impact of the system assessed. It was concluded that agro-ecosystem models can add a spatial and temporal dimension to environmental impact assessment in agricultural LCA studies. The results could be used to assist policymakers in optimising use of agricultural land.

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Carbon sequestration; DNDC model; greenhouse gas emissions; life cycle assessment (LCA); perennial cropping systems; soil N_2O emissions

Introduction

Perennial grasses are one of the most commonly grown crops in humid and cold regions. They are primarily grown for forage in animal husbandry, but other alternative uses such as feedstock for bioenergy production have been proposed (Tilman et al. 2006; Auburger et al. 2017; Carlsson et al. 2017). Earlier studies have shown that soil organic carbon (SOC) is often more abundant in perennial than in annual cropping systems, an effect attributed to increased carbon (C) inputs due to high root biomass turnover, less exposure to ploughing and a longer growing season compared with annual crops (Baker et al. 2007; Bolinder et al. 2010; Börjesson et al. 2018). This feature is interesting from a global warming mitigation perspective (Smith et al. 2016; Minx et al. 2018) and soil C sequestration through grass cultivation has been suggested as a negative emission technology with large potential (Tidåker et al. 2014; Yang et al. 2018). However, grass-ley systems have been reported to act differently depending on climate and soil properties (Soussana et al. 2010; Kätterer et al. 2012; Jackson et al. 2017).

Grass cultivation also inevitably has environmental impacts due to different inputs during the system lifecycle, and it is important to determine these impacts in order to assess the full environmental burden of the system. For example, pure grass swards are reliant on inputs of fertilisers to promote high biomass yield and achieve high soil C sequestration (Yang et al. 2018). Mineral fertiliser use in agriculture is associated with environmental impacts, primarily global warming and eutrophication. The climate impact of mineral fertiliser is caused by both manufacturing and soil application, the latter by inducing increased terrestrial emissions of the greenhouse gas (GHG) nitrous oxide (N_2O). This GHG is of particular importance since it contributes significantly to the climate impact (Bouwman et al. 2002). Estimates of N_2O emissions are associated with

CONTACT Johan Nilsson johan.e.nilsson@slu.se Department of Energy and Technology, Swedish University of Agricultural Sciences (SLU), P.O. Box 7032, Uppsala 750 07, Sweden
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considerable uncertainty, due to substantial temporal and spatial deviations and because the underlying processes affecting emissions are still not fully known (Butterbach-Bahl et al. 2013). Management of the cultivation system, such as field operations, will also affect the total environmental impact (Tidåker et al. 2014).

Life Cycle Assessment (LCA) is a comprehensive approach for investigating the environmental impact of products and services. The method was originally developed as a site-independent tool for industrial processes, but has also been widely used for assessment of the environmental impact of agricultural systems (Garrigues et al. 2012). In contrast to the impacts of most industrial processes, the environmental impacts of agriculture are determined by, and embedded in, physical, climatological, social and environmental conditions. Moreover, these determinants vary over time and space. This means that where and when the cultivation takes place will affect the environmental impact of the studied system, for example due to variations in climate and soil properties (Miller et al. 2006). These variations have been proven to be important (Humpenöder et al. 2013; Hörtenhuber et al. 2014; Henryson et al. 2019), but are rarely included in LCA studies, often because of the extensive data demand and since measurements of these processes are time-consuming and costly. Thus in LCA analyses most practitioners rely on databases with low temporal and spatial resolution (Rebitzer et al. 2004).

One approach to include the spatial and temporal variations of the life-cycle impact of agricultural systems is to combine LCA methodology with agro-ecosystem modelling (e.g. Bessou et al. 2013; Goglio et al. 2014; Kløverpris et al. 2016; Deng et al. 2017). The DNDC model is a well-recognised, process-based biogeochemical model that has been used for sites all over the world (Giltrap et al. 2010; Gilhespy et al. 2014; Brilli et al. 2017; Ehrhardt et al. 2018). Since the first version was launched, developers have successively improved the model with additional agro-ecosystem mechanisms (Gilhespy et al. 2014). The DNDC model has been used for example to fill data gaps in LCAs in recent studies (Goglio et al. 2014, 2018).

In this study, we assessed the potential climate impact and eutrophication potential of grass cultivation at five sites in Sweden with different characteristics. The DNDC model was used to simulate C and N fluxes and calculate site-dependent impacts, in a life cycle perspective. The system boundary was set from cradle to farm gate, and the environmental impact was calculated per hectare and per Mg dry matter (DM) yield. Since estimates of N_2O emissions from soil sources have a high degree of uncertainty, we opted to compare three methods for calculating these emissions.

Material and methods

Site-specific data for each of the five sites were used to model life cycle inventory data, which were then used to evaluate the environmental impact of the grass cultivation system. The inventory data collected to assess the climate impact of the system comprised field operations (including sowing, rolling, cutting and ploughing), manufacturing of mineral N fertiliser and soil N_2O (direct and indirect), CH_4 and C fluxes, with the latter three estimated using the DNDC model. For the eutrophication assessment, the life cycle inventory was conducted using nitrogen (N) and phosphorus (P) leaching data from Johnsson et al. (2016).

Experimental sites

The five study sites selected were distributed over southern and central Sweden (Figure 1), to cover variations in climate and soil properties. The soils at the two most northerly sites, Kungsängen (59.8°N) and Karlslund (59.4°N), both had a high clay content (57% and 29%, respectively) and initial SOC content (6.0% and 2.6%, respectively). The soil at Lanna (58.5°N) was a silty clay loam with lower SOC content (2.0%) than the two soils at higher latitudes and

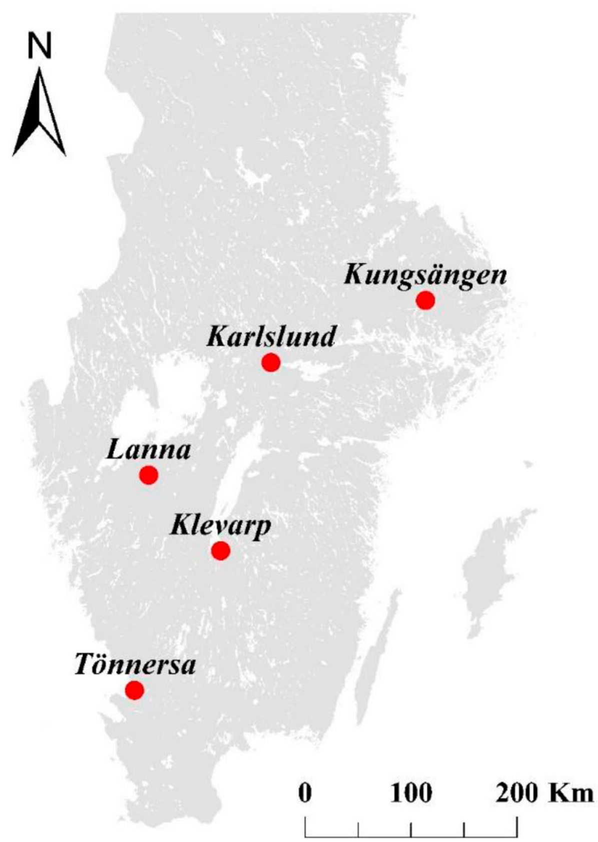

Figure 1. Map of southern and central Sweden indicating the location of the five study sites.

Table 1. Specific characteristics of the five study sites.

Site	Karlslund	Klevarp	Kungsängen	Lanna	Tönnersa	Source
Latitude	59.4	57.7	59.8	58.5	56.5	Eckersten et al. (2004)
Mean temp (°C) 1986–2015	6.8	5.4	6.9	7.1	8.0	SMHI
Mean annual precipitation (mm) 1986–2015	691	679	568	598	791	SMHI
N in precipitation (ppm)	1.2	1.5	1.2	1.4	1.7	Krondroppsnätet (2018)
Soil texture	Clay loam	Sandy loam	Clay	Silty clay loam	Sandy loam	Eckersten et al. (2004)
Soil organic carbon at surface (%)	2.6	1.7	6.0	2.0	1.5	Eckersten et al. (2004)
Clay fraction (%)	29	2.1	57	33	7.2	Eckersten et al. (2004)
Sand fraction (%)	33	65	30	10	65	Assumption
Bulk density (g/cm^3)	1.29	1.37	1.39	1.24	1.43	Estimation based on Saxton and Rawls (2006)
Porosity (%)	51	48	48	53	46	Estimation based on Saxton and Rawls (2006)
Field capacity (water-filled pore space)	0.67	0.31	0.87	0.72	0.36	Estimation based on Saxton and Rawls (2006)
Wilting point (water-filled pore space)	0.38	0.09	0.71	0.39	0.14	Estimation based on Saxton and Rawls (2006)

Note: Data on nitrogen (N) concentration in precipitation were obtained from the national inventory database (Krondroppsnätet 2018). No values for the period of interest were available for the Kungsängen site and therefore the concentration for Karlslund, the nearest site to Kungsängen, was used. The sand fraction was assumed based on average soil texture values. SMHI: https://www.smhi.se/klimatdata & Krondroppsnätet: http://krondroppsnatet.ivl.se.

with 33% clay content. The two most southerly sites, Klevarp (57.7°N) and Tönnersa (56.5°N), both had sandy loam soils with low SOC content (1.7% and 1.5%, respectively). Tönnersa had the highest mean annual temperature and precipitation of all five sites and Klevarp, located in the centre of the south Swedish highlands, had the lowest mean annual temperature. Soil and climate properties for each site are shown in (Table 1).

Perennial cropping system

A five-year grass cultivation system was simulated over 30 years for each of the individual sites, using weather data for the period 1986–2015 (Table S1 in Supplementary Material (SM)). Each rotation started with sowing and rolling in the first year and ended with ploughing to 30 cm depth in year five (Figure 2). During the crop rotation, the grass was fertilised with mineral N fertiliser and cut twice a year. Two fertilisation rates were compared, F1 = 140 kg N ha^{-1} and F2 = 200 kg N ha^{-1}. Spreading of fertiliser was split between two occasions each year, with the first application (80/120 kg N ha^{-1}) on 1 May and the second (60/80 kg N ha^{-1}) on 10 June, shortly after the first cut.

Modelling and assumptions

Agro-ecosystem modelling

The DNDC model is driven by climate, soil, vegetation and management variables, which are used to simulate critical terrestrial processes such as crop growth, soil C dynamics, soil temperature and moisture regimes and emissions of greenhouse and trace gases. The simulation results are dynamically presented on a daily time step (Li et al. 1992, 2012). In this study, we used a model version that contains more detailed descriptions of crop biomass growth (Kröbel et al. 2011), soil temperature (Dutta et al. 2017) and evapotranspiration (Dutta et al. 2016), and has recently been modified for simulating perennial regrowth after each cut and in subsequent years (He et al. 2019). This version was chosen because it has been used to simulate perennial growth in similar cool-weather conditions to those in Sweden (He et al. 2019). The model was used to estimate life cycle inventory data for soil C fluxes, N_2O and CH_4 emissions and biomass yield, assuming that 85% of aboveground biomass was harvested at every cut. The parameterisation of the model is presented in (Table S2) in SM. Indirect N_2O emissions were calculated using the default emission factor (0.0075) from IPCC (2006) associated with N leaching and runoff, which were simulated using the DNDC model.

Field trials designed to study the growth pattern of a mixture of timothy grass (*Phleum pratense* L.) and meadow fescue (*Festuca pratensis* Huds.) over two consecutive years were conducted at the study sites between 1985 and 1988. At Kungsängen and Klevarp, the two-year trials were performed twice, i.e. for four years in total. All fields were treated equally in order to make the results comparable. For more information about the experimental set-up, see Eckersten et al. (2004, 2007). The DNDC model was evaluated for simulating the biomass growth pattern over the growing seasons. Data displayed in (Table 1) were used as input in the model to define the conditions at the different sites. Root:shoot ratio was assumed to be 1, i.e. 50% of total biomass, based on previous grass cultivation modelling studies by Eckersten et al. (2004) and Johnsson

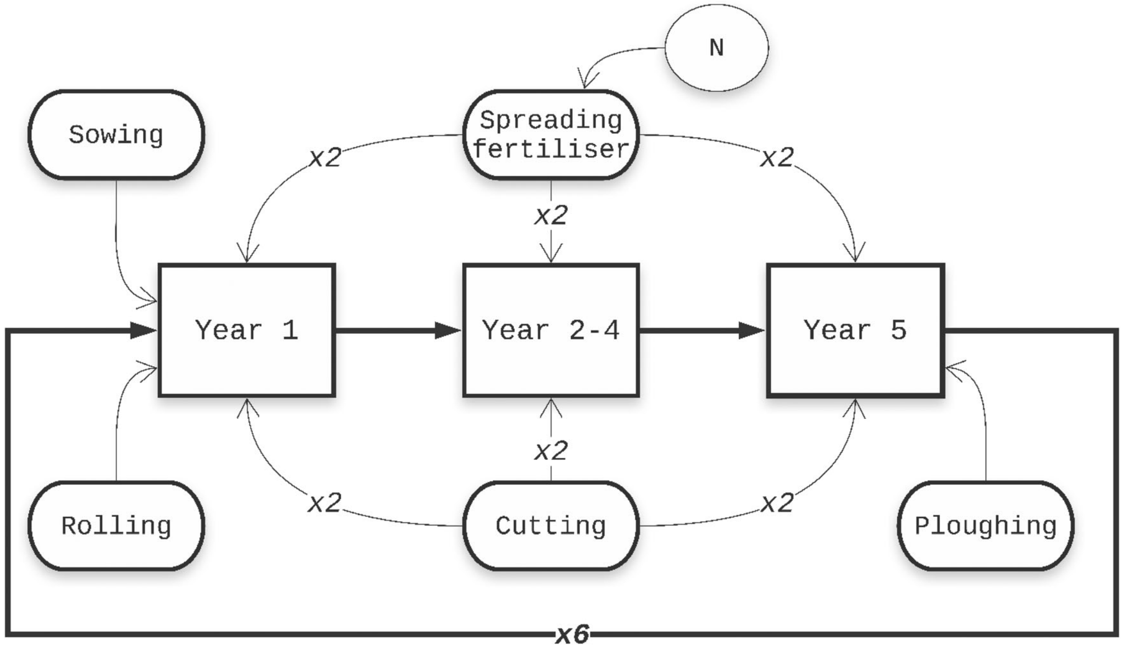

Figure 2. Overview of the crop rotation simulated for all study sites. The grass was sown and the soil was rolled in the first year, then growth continued for five more years. During this period, fertiliser was applied and the grass was cut twice every year. The crop rotation ended with deep ploughing to 30 cm. The rotation was repeated six times.

et al. (2016). The model fit to observed growth data was evaluated as coefficient of determination (r^2, eq. 1), normalised root mean square error (nRMSE, eq. 2) and Nash-Sutcliffe model efficiency coefficient (Nash and Sutcliffe 1970) (ME, eq. 3). An ME value > 0 corresponds to goodness of fit better than the observed mean value, while ME = 1 corresponds to a perfect fit. To evaluate model performance, the goodness of fit statistics were calculated for all biomass data and the biomass observations closest to harvest.

$$r^2 = \left(\sum_{t=1}^{n}(S_t - \bar{S})(O_t - \bar{O})\right)^2 / \sum_{t=1}^{n}(S_t - \bar{S})^2 \sum_{t=1}^{n}(O_t - \bar{O})^2 \quad (1)$$

$$nRMSE = \left(\sum_{t=1}^{n}(O_t - S_t)^2/n\right)^{1/2} / \bar{O} \quad (2)$$

$$ME = 1 - \sum_{t=1}^{n}(O_t - S_t)^2 / \sum_{t=1}^{n}(O_t - \bar{O})^2 \quad (3)$$

where O denotes observed biomass and S simulated biomass.

Soil N_2O method comparison

Earlier studies have shown the importance of N_2O emissions when examining the climate impact of agro-ecosystems (e.g. Jury et al. 2010; Ruan et al. 2016). Because of the uncertainties associated with estimating soil-borne N_2O, we compared the results from the DNDC model with those obtained using two empirical approaches. These were: (i) the IPCC Tier 1 site-generic emissions factor, 0.01 kg N_2O-N kg N^{-1}, assuming no change in soil C stocks (IPCC 2006), and (ii) a site-specific approach developed by Rochette et al. (2018) who concluded, based on N_2O emissions observations in Canada, that cumulative emissions from synthetic N application, N_2O_{Roch} (kg N_2O-N ha^{-1}), can be predicted successfully ($R^2 = 0.68$) with the equation:

$$N2O_{Roch} = \exp(3.91 + 0.0022P + 0.0069MinN - 0.0032SAND - 0.747pH + 0.097T_{air}) \quad (4)$$

where P is growing season precipitation (mm), $MinN$ is mineral N application (kg), $SAND$ is soil sand content (g kg^{-1}), pH is soil pH and T_{air} is mean annual air temperature (°C) (Rochette et al. 2018).

The three methods were compared by calculating the yearly cumulative direct N_2O emissions at each of the five study sites.

Field operations and fertiliser manufacture

Diesel consumption for sowing, rolling and spreading fertiliser was assumed to be 2.3, 2.3 and 4.7 L ha^{-1}, respectively (Carlsson et al. 2017). Diesel consumption for cutting and ploughing was based on linear regression models with biomass yield and clay content, respectively, as the independent variable (Arvidsson and Keller 2011; Prade et al. 2015). The GHG emissions from production and use of diesel were set to 2.8 kg CO_2, 1.2 g CH_4 and 0.073 g N_2O L^{-1}, based on Gode et al. (2011). The GHG emissions during manufacture of mineral fertiliser were set to 3.5 kg CO_2-eq kg^{-1} N, where the climate impact was assumed to be 86% from CO_2 emissions, with the remaining 14% from N_2O (Bentrup et al. 2016).

Nitrogen and phosphorus leaching

Nitrogen and *P* leaching were estimated using data from Johnsson et al. (2016), who performed national simulations of mean leaching rates in 22 different regions in Sweden, using the models SOILNDB for N and ICECREAMDB for P leaching. The data represent leaching from the root zone and surface runoff for specific crops and soil textures (Johnsson et al. 2016).

Climate impact assessment

The climate impact was assessed using Global Warming Potential (GWP) and dynamically using Absolute Global Temperature Potential (AGTP), as defined by the IPCC (Myhre et al. 2013). The GWP methodology compares the cumulative radiative forcing of a GHG emission with the radiative forcing of an equal amount of emitted CO_2 over a specific period, typically 100 years (Myhre et al. 2013). The characterisation factor for CH_4 and N_2O is 34 and 298, respectively, with the inclusion of climate-carbon feedbacks (Myhre et al. 2013). One of the limitations with the GWP approach is that the method does not include the timing of emissions, which means that emissions which occur during different points in the life cycle are added together, although the endpoint of the impact differs (Kendall 2012).

The AGTP approach goes one step further by analysing the potential temperature change due to the change in radiative forcing caused by a pulse emission of GHGs, which is achieved by applying radiative forcing calculations in convolution with the climate temperature response to changes in radiative forcing. By investigating the cumulative temperature response from the yearly emissions modelled in the life cycle inventory, the climate impact can be assessed dynamically throughout a specified analytical time horizon. This approach to assessing the climate impact has been used previously in LCA studies to evaluate the climate impact of bioenergy systems (Ericsson et al. 2013; Hammar et al. 2017).

Table 2. Marine eutrophication and potential eutrophication at the study sites, calculated using nitrogen (N) and phosphorus (P) characterisation factors (CF) taken from CML (Guinée, 2002) and from Henryson et al. (2018), respectively.

	Marine eutrophication Henryson *et al.* (kg N-eq kg^{-1})		Potential eutrophication CML (kg N-eq kg^{-1})	
Sites	N CF	*P* CF	N CF	*P* CF
Karlslund	0.169	0.672	1	7.23
Klevarp	0.122	0.499	1	7.23
Kungsängen	0.435	2.48	1	7.23
Lanna	0.55	0	1	7.23
Tönnersa	0.835	0	1	7.23

Eutrophication impact assessment

The eutrophication caused by the leached N and P was assessed using two different, but complementary methods. First of all, we used the site-generic CML methodology (Guinée, 2002) to assess the potential eutrophication impact of estimated N and P leaching. This method places the indicator at the point of emission and thus neglects the fate of the eutrophying emissions. Furthermore, the method considers all N and P discharged to the environment as having eutrophying capacity and includes all recipients, such as terrestrial, freshwater and marine water bodies (Guinée, 2002). In reality, eutrophication is more complicated and highly dependent on spatial properties. One example is the Baltic Sea, which is the world's largest brackish water basin and, unlike most marine environments, is considered limited by both N and P, with variations between different sub-basins (Swedish EPA 2006). To account for this, we used site-specific marine eutrophication characterisation factors developed by Henryson et al. (2018) for different regions in Sweden. These include site and catchment properties and the P or N limiting status of the recipient, and were used here as a complement to the CML calculations to investigate the impact on the complex marine environment that surrounds Sweden. The characterisation factors used in the CML and Henryson *et al.* approach are listed in (Table 2).

Results

Life cycle inventory

The climate impact inventory was divided into change in SOC content, soil N_2O and CH_4 emissions, fertiliser manufacturing and field operations. The results of the life cycle inventory for soil C balance and soil N_2O emissions are presented in section 3.1.1 and the results of the inventory analysis of eutrophying N and P leaching rates in section 3.1.2.

SOC balance and N_2O emissions

The soil at all sites investigated showed an ability to sequester C over the complete simulation period and for both fertilisation rates, although the increase was low (0.035 Mg ha^{-1} in treatment F1) for the site with initial highest SOC content (Kungsängen). The largest increase in SOC content was for the silty clay loam at Lanna (4.3 and 6.5 Mg ha^{-1} over the 30-year simulation period for F1 and F2, respectively). As expected, the F2 application rate led to greater C sequestration in all soils than the F1 rate (Figure 3). At the end of each crop rotation, all living biomass (aboveground and belowground) was terminated through ploughing and thereby transferred to the SOC pool, which explains the large SOC increase every fifth year in (Figure 3). Yearly mean gross C input, i.e. before degradation, for all soils, was 2.7 and 3.4 Mg C ha^{-1} for F1 and F2, respectively.

Simulated cumulative N_2O emissions were highest for the clay and SOC-rich soil in Kungsängen (mean 5.2 kg N_2O ha^{-1} y^{-1} for F1 and 6.1 kg N_2O ha^{-1} y^{-1} for F2), while emissions were lower for the sandy loam soils at Klevarp and Tönnersa. The Klevarp site had the lowest estimated emissions (1.9 kg N_2O ha^{-1} y^{-1} for F1 and 2.1 kg N_2O ha^{-1} y^{-1} for F2). Higher emissions from soils containing more clay are consistent with findings in a meta-analysis based on observations from Rochette et al. (2018). There was considerable variation between simulated years, especially for the Kungsängen soil (Figure 4). This annual variation was attributed to weather fluctuations, for example differences in amount and pattern of precipitation. Mean N_2O emissions over the simulation period were slightly higher for the higher fertilisation rate (F2) at all study sites.

The different methods to estimate N_2O emissions were compared by calculating the emissions for each site. The two site-specific methods, DNDC and Rochette *et al.*, showed large variation between the different sites. Overall, the DNDC model predicted higher annual emissions than the Rochette *et al.* approach (Figure S1 in SM). The DNDC model predicted the highest emissions rate for the clay-rich soil at Kungsängen, while the Rochette *et al.* approach predicted the highest emissions for the field at Lanna, with the lowest soil sand content. Both site-dependent methods predicted the lowest emissions from the sandy loam soils at Klevarp and Tönnersa. Mean emissions across all sites calculated with the DNDC, Rochette *et al.* and IPCC Tier 1 approaches were 1.97 ± 0.83, 1.41 ± 0.93 and 1.63 ± 0.02, respectively, for F1 and 2.29 ± 0.98, 2.13 ± 1.41 and 2.31 ± 0.02 kg N_2O-N ha^{-1}, respectively, for F2. Mean estimates for each field are shown in (Table 3).

Nitrogen and phosphorus leaching

Nitrogen leaching was estimated to be higher for the sandy loam soils at Tönnersa and Klevarp than for the soils with higher clay content at Kungsängen, Lanna and Karlslund. The lowest N leaching rate was predicted for the soil with the highest clay content (Kungsängen). For P leaching, the trend was roughly the opposite, i.e.

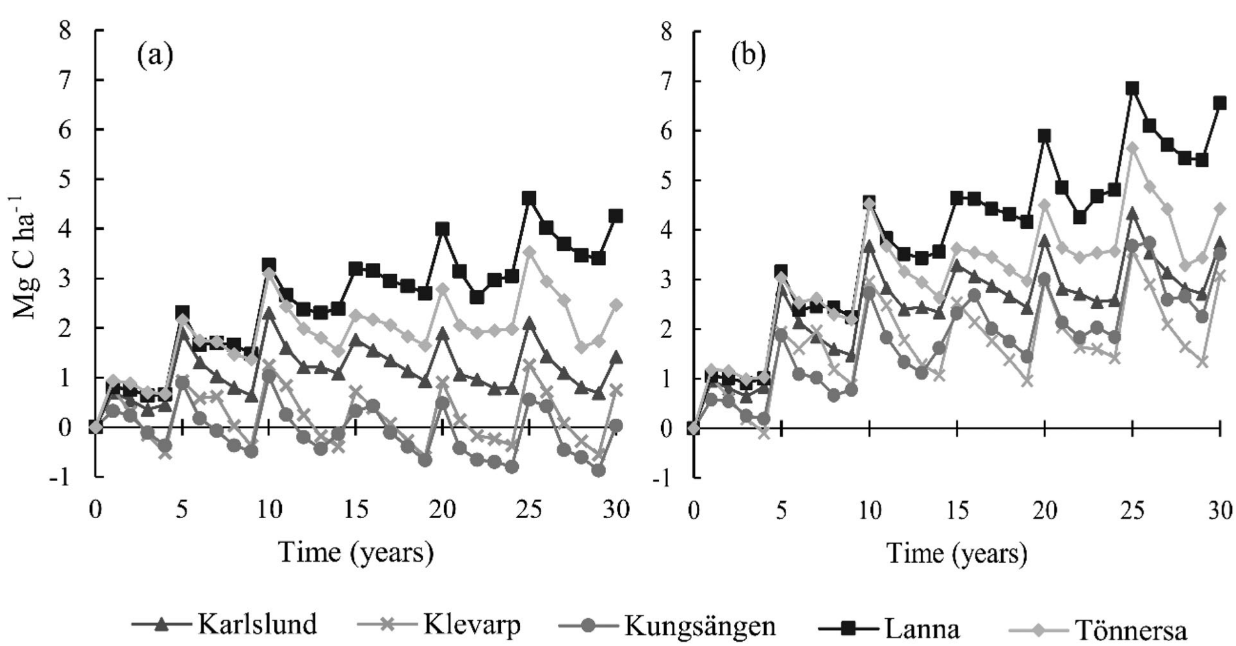

Figure 3. Simulated cumulative change in soil organic carbon (SOC) for fertiliser rate (a) F1 (140 kg N ha^{-1}) and (b) F2 (200 kg N ha^{-}) over the 30-year study period. The SOC change is presented in Mg C ha^{-1}.

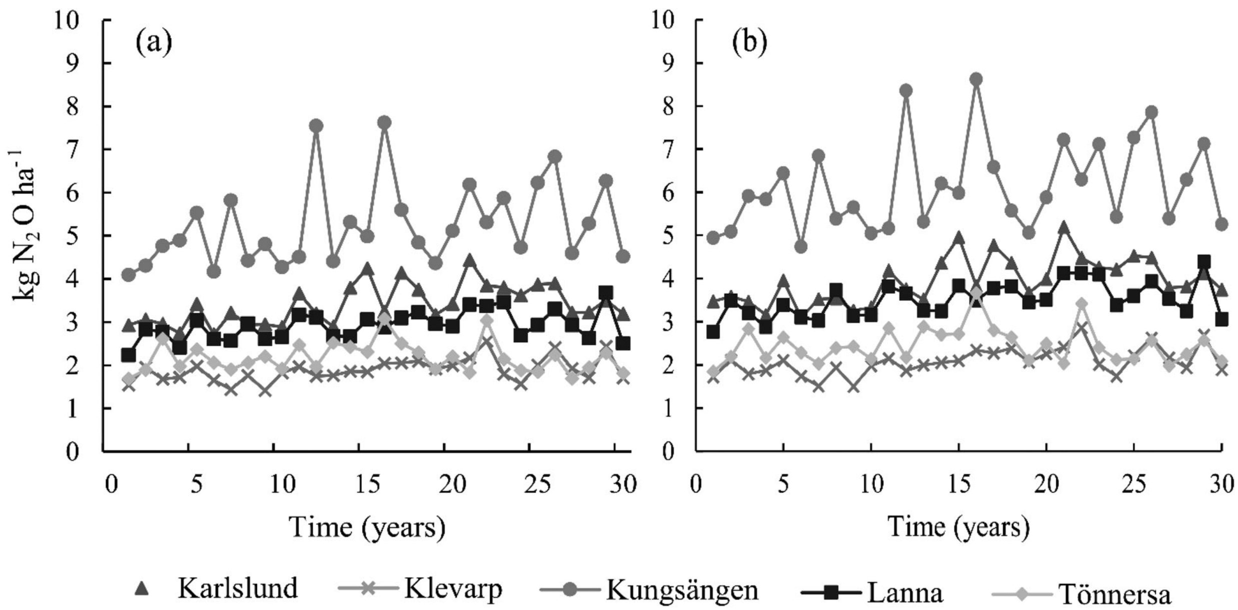

Figure 4. Annual cumulative nitrous oxide emissions from the study sites (kg N_2O ha^{-1}), estimated using the DNDC model, for fertilisation rate (a) F1 (140 kg N ha^{-1}) and (b) F2 (200 kg N ha^{-1}) over the 30-year simulation period.

Table 3. Mean nitrous oxide (N_2O) emissions at the five study sites under fertilisation rates F1 and F2 (140 and 200 kg N ha^{-1}, respectively), assessed with three different approaches: DNDC, Rochette and IPCC Tier 1.

Sites	DNDC (kg N_2O-N ha^{-1})	Rochette (kg N_2O-N ha^{-1})	IPCC Tier 1 (kg N_2O-N ha^{-1})
Karlslund F1	2.15 ± 0.29	1.50 ± 0.29	1.63 ± 0.01
Klevarp F1	1.19 ± 0.17	0.47 ± 0.09	1.64 ± 0.01
Kungsängen F1	3.32 ± 0.59	1.41 ± 0.24	1.61 ± 0.02
Lanna F1	1.84 ± 0.21	2.98 ± 0.55	1.62 ± 0.01
Tönnersa F1	1.37 ± 0.22	0.69 ± 0.17	1.63 ± 0.01
Karlslund F2	2.49 ± 0.33	2.27 ± 0.43	2.31 ± 0.01
Klevarp F2	1.30 ± 0.20	0.71 ± 0.13	2.32 ± 0.02
Kungsängen F2	3.89 ± 0.64	2.13 ± 0.36	2.28 ± 0.03
Lanna F2	2.22 ± 0.25	4.50 ± 0.83	2.30 ± 0.02
Tönnersa F2	1.54 ± 0.26	1.05 ± 0.26	2.32 ± 0.02

with higher leaching for the clay-rich soils than the sandy soils. The highest P leaching was predicted for the soil with 33% clay content (Lanna) (Table 4).

Life cycle impact assessment

The results from the life cycle inventory were used to assess the climate impact and potential eutrophication impact of the grass cultivation system at each of the five study sites.

Table 4. Predicted mean nitrogen (N) and phosphorus (*P*) leaching for the five study sites.

Site	N (kg ha^{-1})	*P* (kg ha^{-1})
Karlslund	3	0.41
Klevarp	15	0.27
Kungsängen	1	0.43
Lanna	3	0.79
Tönnersa	18	0.23

Climate impact

The GHG fluxes from the inventory analysis were divided into five categories and analysed with GWP_{100} (Figure 5). Mean total GHG emissions for all five sites were 1170 ± 460 and 1200 ± 460 kg CO_2-eq ha^{-1} y^{-1} for F1 and F2, respectively. Expressed per Mg DM, the mean emissions were 178 ± 77 and 136 ± 59 kg CO_2-eq for F1 and F2, based on the 30-year simulation. The large standard deviation indicates considerable variation between the sites. The highest emissions were simulated for Kungsängen (321 and 244 kg CO_2-eq Mg DM^{-1} for F1 and F2, respectively) and the lowest for Tönnersa (89 kg CO_2-eq Mg DM^{-1} for F2). The total climate impact of the system was mainly a balance between increased soil C stocks, i.e. C sequestration, and emissions of N_2O from soil processes and GHG emissions from manufacturing of the fertiliser. The grass cultivation resulted in a small CH_4 sink for all simulated sites (Figure 5). The higher fertilisation rate (F2) generated lower emissions per Mg DM in all fields, due to more soil C sequestration and higher grass yield. However, the variation between sites was greater than that between fertiliser rates.

The relationship between the emissions categories shown in (Figure 5) changed for different rotation periods over the 30-year simulation period. In other words, the climate impact assessed as GWP varied not only between sites and fertilisation rates, but also over time between consecutive rotations throughout the

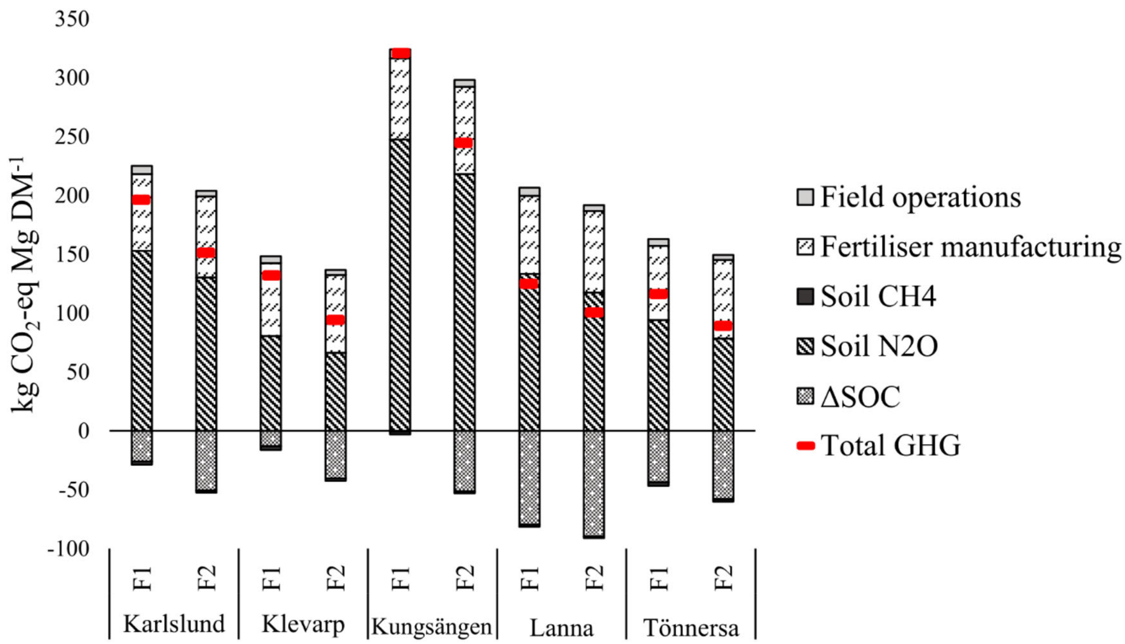

Figure 5. Total climate impact of grass cultivation during the 30-year simulation period at the five study sites for fertilisation rates F1 (140 kg N ha^{-1}) and F2 (200 kg N ha^{-1}), assessed as Global Warming Potential over 100 years (GWP100). SOC = soil organic carbon, GHG = greenhouse gases.

study period. For all fields except Kungsängen (F1 and F2) and Klevarp (F1), the first cropping sequence demonstrated a global warming mitigating effect, whereas the last rotation enhanced the global warming effect at all sites (Table 5). The main reason for this was that the soil C sequestration rate was higher during the first rotation compared with the last.

The climate impact was further investigated using the dynamic climate impact assessment model described in section 2.4. The yearly GHG fluxes from the system were used to calculate the cumulative temperature change for 100 years, expressed as pK ha^{-1} ($p = 10^{-12}$). The change in global mean temperature due to grass cultivation at the study sites is shown in (Figure 6). Similarly to (Table 5), it shows a lower temperature change at the beginning of the simulation period and an increasing rate of impact over time. At the temporal boundary of the system, i.e. after 30 years, the climate impact increased for a few years before it started to decline, which was due to the atmospheric inertia related to GHG emissions and temperature increase. Seventy years beyond the system's temporal boundary, grass cultivation still had a warming effect on the climate.

Table 5. Climate impact assessed as Global Warming Potential (GWP) for the first crop rotation (1) and the last (6) at the different sites under fertilisation rate F1 (140 kg N ha^{-1}) and F2 (200 kg N ha^{-1}).

Site and fertilisation rate	Crop rotation 1 (kg CO_2-eq Mg DM^{-1})	Crop rotation 6 (kg CO_2-eq Mg DM^{-1})
Karlslund F1	−4	290
Klevarp F1	39	206
Kungsängen F1	202	398
Lanna F1	−70	244
Tönnersa F1	−76	252
Karlslund F2	−42	245
Klevarp F2	−22	177
Kungsängen F2	119	322
Lanna F2	−85	215
Tönnersa F2	−99	227

Note: The results for each site are expressed per Mg DM.

Eutrophication assessment

The potential eutrophication (CML) and marine eutrophication (Henryson *et al.*) impact of N and P leaching were assessed on a per-hectare basis (Figure 7). Mean eutrophication potential for all sites, assessed with CML characterisation factors, was 11.1 ± 6.1 kg N-eq ha^{-1} (range 4.1 kg N-eq ha^{-1} for Kungsängen to 19.7 kg N-eq ha^{-1} for Tönnersa). The high eutrophication potential at Tönnersa was mainly due to the high N leaching rate at that site. In general, the eutrophication potential was higher for the sandy loam soils at Tönnersa and Klevarp and lower for the more clay-rich soils at the other sites.

Mean marine eutrophication at all sites, assessed with the Henryson *et al.* approach, was 4.2 ± 5.4 kg N-eq ha^{-1} (ranging from 0.1 kg N-eq ha^{-1} at Karlslund to 15.0 kg N-eq ha^{-1} at Tönnersa) (Figure 7). The lower impact compared with the CML approach is because the Henryson *et al.* characterisation factors assess marine

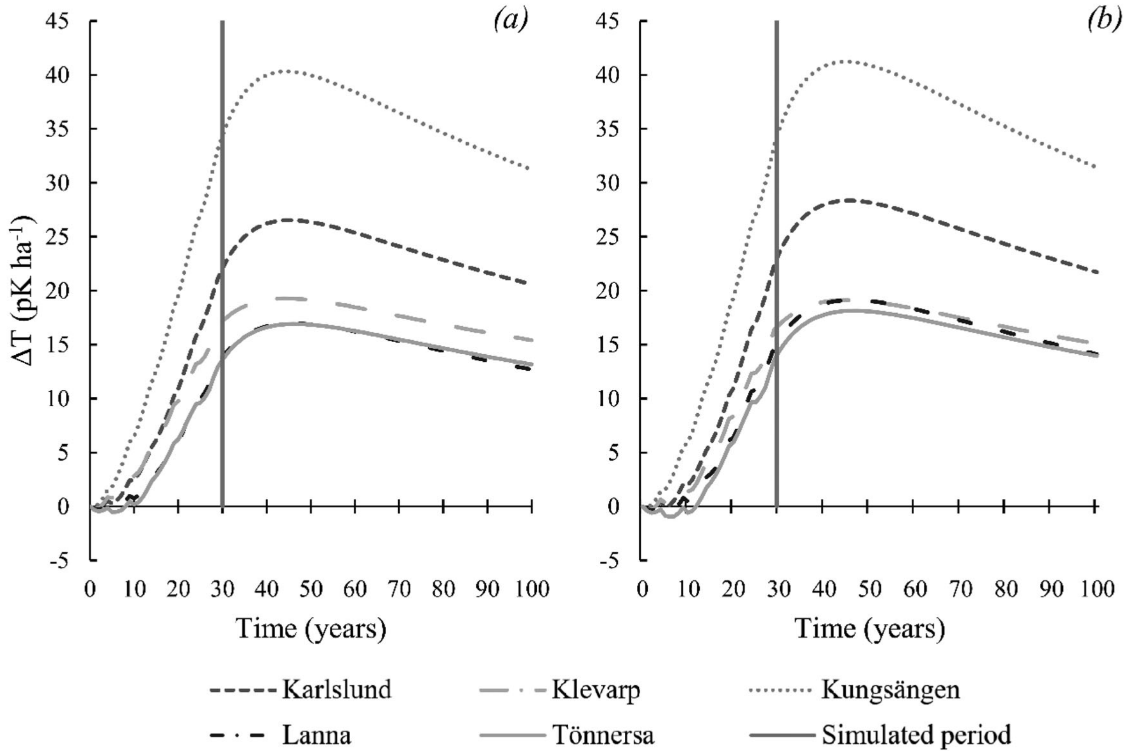

Figure 6. Simulated potential temperature response of the grass cultivation system during the 30-year study period with fertilisation rate (a) F1 (140 kg N ha^{-1}) and (b) F2 (200 kg N ha^{-1}) at the five study sites. The temperature response is expressed as pK ha^{-1} ($p = 10^{-12}$, K = Kelvin).

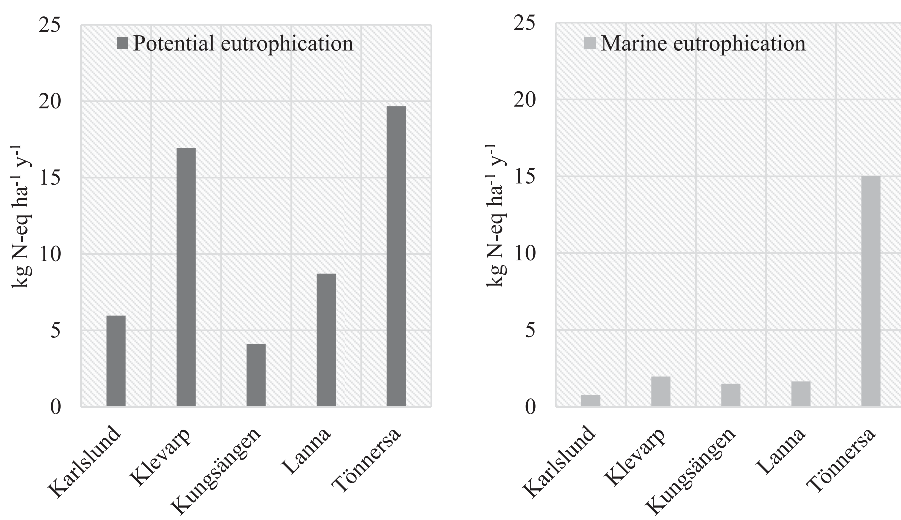

Figure 7. Potential eutrophication impact assessed using CML (potential eutrophication) and Henryson (marine eutrophication) methodology. The bar represents the eutrophication in kg N-eq ha^{-1} y^{-1}.

eutrophication exclusively, which means that results derived with the different methods should not be compared directly. However, it is relevant to analyse how the pattern of the estimated eutrophication differed between the two approaches. For instance, according to the Henryson *et al.* method, the eutrophication level for the sandy soil at Klevarp was similar to that for soils with a higher clay content. The other sandy loam soil (Tönnersa) was estimated to have the highest marine eutrophication, because of high N leaching rate and proximity to the recipient. Compared with the CML approach, relatively lower eutrophication was assessed for the field in Lanna, partly because of N-limiting characteristics of the recipient.

Model biomass growth goodness of fit

The goodness of fit of the DNDC model was analysed using mean simulated and observed aboveground biomass at each cutting occasion. This analysis showed that the mean simulated aboveground biomass was within the standard deviation of the observed data for each site (Figure 8). The goodness of fit for observations closest to harvest was 35 and 29% nRMSE for fertilisation application rate F1 and F2, respectively, and ME was 0.24 for both fertilisation rates (Table S3 in SM). Since ME was above zero, the model corresponded to the observed data more efficiently than the mean observed value (explained in section 2.3.1). The model fit to all observed biomass data for all fields was $r^2 = 0.61$, nRMSE = 49% and ME 0.53 for F1, and $r^2 = 0.71$, nRMSE = 38% and ME = 0.47 for F2 (Table S4 in SM).

Accurate simulation of biomass is important for estimating soil C inputs, which is a crucial driver for simulating soil C change.

Discussion

Climate impact

Assessment of the climate impact categories revealed considerable variation between the study sites. The mean climate impact for all sites was 178 ± 77 kg CO_2-eq Mg DM^{-1} or 1170 ± 460 kg CO_2-eq ha^{-1} y^{-1} and 136 ± 59 kg CO_2-eq Mg DM^{-1} or 1200 ± 460 kg CO_2-eq ha^{-1} y^{-1} for the F1 and F2 fertilisation rate, respectively. The higher fertilisation rate resulted in higher yields, which reduced the climate impact per Mg DM compared with the F1 rate. However, the difference in climate impact between F1 and F2 was small when analysed per hectare. Overall, the site-specific properties were more important than fertilisation rate when assessing the climate impact of grass cultivation (Figure 5). The main emissions causing the climate impact were in the form of soil N_2O emissions and emissions from fertiliser manufacturing, while the increased soil C content reduced the climate impact of the system. Negative CH_4 emissions also contributed to reducing the climate impact, but at a very small scale compared with C sequestration. Soil can act as both a source and sink of CH_4, depending on the soil environment. However, less managed soils such as native prairie and forest soils are normally net consumers of CH_4 (Johnson et al. 2007).

Assessment of the climate impact over time showed lower impact during the first part of the simulation period, when C sequestration was higher and compensated for the impact of other emissions (Table 3). As yearly C sequestration decreased, the climate impact increased, which resulted in increased global mean temperature after both 30 and 100 years (Figure 6). The risk of soil C sequestration schemes transitioning from global warming mitigating to global warming

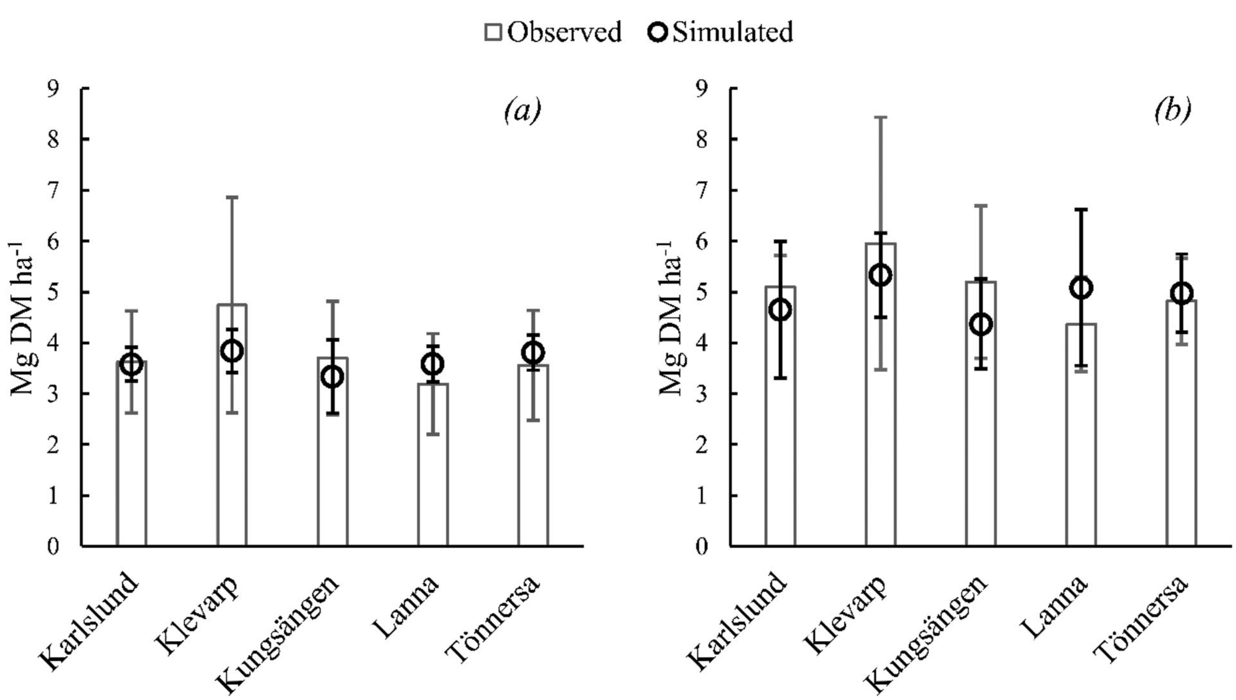

Figure 8. Aboveground mean biomass production at harvest, with fertilisation rate (a) F1 (140 kg N ha^{-1}) and (b) F2 (200 kg N ha^{-1}). Bars represent mean of the observations and the rings simulated means. The error bars represent the standard deviation of the observations (grey) and simulations (black). DM = dry matter.

forcing when the soil approaches SOC saturation has been discussed earlier (Lugato et al. 2018). This risk is especially high in agricultural systems that are dependent on mineral fertilisers to maintain SOC content, because of the large climate impact associated with fertiliser manufacturing and enhanced N_2O emissions from soil. Moreover, in terrestrial systems the C and N cycles are closely coupled, which means that a change in C stock will ultimately alter the conditions for N soil processes, such as nitrification and denitrification. Li et al. (2005) investigated the relationship between SOC content and N_2O emissions in both modelling studies and field trials. They concluded that strategies to increase SOC content, such as reduced tillage, enhanced crop residue incorporation and farmyard manure application, increase the N_2O emissions, offsetting the mitigating effect by 75-310% (Li et al. 2005). However, this pattern is not undisputed. For instance, studies in Canada have shown that reduced tillage in dry semi-arid and sub-humid soils can decrease N_2O emissions due to lower nitrification rates in poorly aerated soils, while reduced ploughing in more humid regions can result in increased N_2O emissions (Helgason et al. 2005; Rochette et al. 2008).

Soil C sequestration

The introduction of grass cultivation resulted in increased soil C stock at all sites over the 30-year simulation period. The F2 fertilisation rate induced more C sequestration than F1 (Figure 3). This was because of the increased mean gross C input, which was 2.7 and 3.4 Mg ha^{-1} y^{-1} for F1 and F2, respectively. The greatest increase in soil C stocks was predicted for the silty clay loam at Lanna (0.14 and 0.22 Mg ha^{-1} y^{-1} for F1 and F2, respectively). Goglio et al. (2014) used DNDC to assess soil GHG emissions in an LCA study and concluded that it can accurately simulate C inputs. Furthermore, in a study using dry combustion analysis to determine C sequestration in long-term grass and cereal rotations at two sites, including Lanna, Börjesson et al. (2018) concluded that a mean C input of roughly 2.5 and 3.5 Mg C ha^{-1} y^{-1} increased the soil C stock by 0.11 and 0.17 Mg C ha^{-1} y^{-1}, respectively. The grass rotation in that study included three years of grass-clover mixture and one year of cereals (Börjesson et al. 2018), and thus less C sequestration could be expected since the perennial period was shorter and an annual crop was present in the rotation.

Clay and SOC content are two important soil properties that influence the C sequestration potential. Soils with a high SOC content are usually closer to their C saturation concentration, which means lower capacity to sequester C, while a high clay content affects the decomposition rate by making organic material physically unavailable to the soil decomposers (Li et al. 1992). The effects of the interaction between SOC and clay content are not always trivial. For instance, in the present study C sequestration was estimated to be greatest in the soil with the second highest clay content (33%) and moderate SOC content (2%), while the soil with the highest clay and SOC content (57% and 6%, respectively) had the lowest soil C sequestration under F1 fertilisation (Figure 3). However, the F2 fertilisation rate induced increased soil C stocks in the same soil, by 0.12 Mg C ha^{-1} y^{-1}, due to the increased C input and the high SOC binding capacity associated with the high clay content. The soil with the lowest clay content showed low C sequestration ability, even though the initial SOC content was low.

In this study, we did not include CO_2 assimilated in the biomass yield, which corresponded to 9.8 and 13.2 Mg CO_2-eq ha^{-1} y^{-1} for fertilisation rate F1 and F2, respectively. This means that, although production of grass increased the global mean temperature, there is potential for creating climate-mitigating systems depending on how the harvested biomass is utilised.

Soil N_2O emissions

The yearly cumulative N_2O emissions showed large variation between sites and fertilisation intensities (Figure 4). In general, the N_2O emissions were higher from fine-textured soils than from coarser-textured soils. Soil water content and water-filled pore space have been shown to be appropriate parameters for describing soil redox potential, and thus the conditions for soil N_2O formation (Li et al. 2000). Soils with high water content are often characterised by low redox potential, which favours the formation of N_2O through denitrification. For the DNDC simulations in the present study, data on water retention parameters such as porosity, field capacity and wilting point for each soil were input directly into the model. In contrast, the Rochette *et al.* approach uses soil sand content to describe soil water-filled pore space (Rochette et al. 2018). This explains why the Rochette *et al.* approach gave the highest N_2O emissions for the soil with the lowest soil sand content, while the DNDC model gave the highest emissions for the soil with the greatest water holding capacity (Table 1). Both site-specific methods gave the lowest emissions for the sandy loam soils. When measurements of N_2O emissions are not available, estimation using the IPCC Tier 1 approach is common in LCA studies. For all sites in this study, the Tier 1 approach predicted similar mean N_2O emissions to the other methods tested, which indicates that IPCC Tier 1 could be an adequate tool for estimating mean emissions in site-independent studies. However, since it does not consider how the emission rate is affected by spatial variations, e.g. soil properties and climate, it may not produce reliable results for site-specific LCAs.

Eutrophication

Mean potential eutrophication at the five sites included in the present study was 11.1 ± 6.1 kg N-eq ha^{-1}, while mean marine eutrophication was 4.2 ± 5.4 kg N-eq ha^{-1}. Use of the site-independent CML method to assess the eutrophication potential of the grass cultivation system at different locations in Sweden revealed the most substantial impacts for sandy loam soils, due to the relatively high N leaching rate (Table 4 and Figure 7). One of the advantages of the CML approach is that it includes all types of recipients. The main disadvantage is that it does not consider how eutrophying emissions affect different types of environments. Eutrophication impact is highly spatially dependent and therefore site-specific methods are preferable, especially in regions with complex environments such as the Baltic Sea. The site-specific method used in this study to assess marine eutrophication accounts for site and catchment properties, as well as the limiting nutrient in the recipient. The Henryson *et al*. approach estimated by far the highest marine eutrophication impact for the Tönnersa site, because of high N leaching and the proximity of the site to an N-limited recipient. The other sandy loam soil (Klevarp), also with high N leaching rates, was estimated to have much lower marine eutrophication impact, due to high N retention in freshwater along the transport pathway to the marine recipient. Furthermore, the Henryson *et al.* approach does not include the eutrophication effects on freshwaters, primarily caused by P addition, which is covered with the CML method. These two different methods to assess the eutrophication effect of grass cultivation should not be directly compared, since they are used to assess different types of eutrophication. They should instead be viewed as complements to each other and used to provide a more complete picture of the eutrophication situation of study systems.

Concluding discussion

Climate impact assessment showed substantial variation between five study sites at different locations in Sweden. The mean climate impact was 1170 ± 460 and 1200 ± 460 kg CO_2-eq ha^{-1} y^{-1} for a fertilisation rate of 140 and 200 kg N ha^{-1}, respectively. The difference in climate impact between the two fertilisation rates was greater when expressed per Mg DM (178 ± 77 and 136 ± 59 kg CO_2-eq for F1 and F2, respectively). The climate impact was greatest for a heavy clay and SOC-rich soil, while it was lower for sandy loam and silty clay loam soils. In general, soil properties and weather conditions were more important than fertilisation rate for the estimated climate impact of the system.

The climate impact increased over time, with a low impact during the first part of the simulation period for most fields and an increased impact during the latter part due to decreased C sequestration rate. This pattern was not captured with the GWP method, which does not account for the timing of emissions.

There were only small differences in the results when overall mean N_2O emissions were compared between modelling approaches. However, the two site-specific methods, DNDC and Rochette *et al.*, showed large variations between sites, which were not captured with the IPCC Tier 1 approach. The DNDC model predicted the highest emissions for the soil with the highest water-holding capacity, while the Rochette *et al.* approach predicted the highest emissions for the soil with the lowest sand content. This was due to their different inherent approaches to estimating water-filled porosity in soil. Both site-specific methods predicted the lowest emissions from sandy loam soils.

Mean potential eutrophication estimated with the CML method was 11 ± 6.1 kg N-eq ha^{-1}, with the high standard deviation indicating considerable variation between sites. Potential eutrophication was highest for sandy loam soils and lowest for soils with a higher clay content. Marine eutrophication assessed with a site-specific method was greatest for a sandy soil with high N leaching rate at a site in close proximity to the recipient.

Simulation of grass cultivation is known to be complex, primarily because grasses are generally grown in a mixture of species. It is difficult to predict how the proportions of the species vary between years and locations. In the model set-up for this study, the grass mixture was simulated as one crop. Despite this, the DNDC model managed to reproduce observed biomass growth with positive model efficiency values, both for all observations and for observations closest to harvest (Figure 8 and Table S3).

Overall, the great variation found between sites in this study stresses the importance of including temporal and spatial dependency in agricultural LCAs. When important data are lacking, agro-ecosystem models such as DNDC can be a useful tool in completing the life cycle inventory.

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Disclosure statement

No potential conflict of interest was reported by the author(s).

Notes on contributors

Johan Nilsson is a PhD candidate at the Swedish University of Agricultural Sciences, Uppsala Sweden. He received a master degree in environmental engineering from the Uppsala University and Swedish University of Agricultural Science. His current research field is Life Cycle Assessment (LCA) of agricultural systems. He is especially interested in climate impact of land use and land use change and how to include spatial and temporal variation of the impact in the assessment.

Pernilla Tidåker is a senior lecturer at the Swedish University of Agricultural Sciences, Uppsala Sweden. She holds a PhD in agricultural engineering: Her research was initially focusing on systems integrating farming and wastewater management and is now mainly emphasising life cycle assessment of food production system and evaluation of environmental impact, ecosystem services and biodiversity in agricultural production using sustainability indicators and tools.

Cecilia Sundberg is Associate Professor in Bioenergy Systems at the Swedish University of Agricultural Sciences (SLU) in Uppsala, Sweden. She also holds a research position at KTH Royal Institute of Technology in Stockholm. She has a special research interest in climate change mitigation through transformation of land use and bioenergy systems.

Kajsa Henryson holds a PhD from the Swedish University of Agricultural Sciences and is currently a postdoctoral researcher. She specialises in life cycle assessment of crop cultivation, particularly environmental impacts related to soil carbon and nitrogen cycling.

Brian Grant works as a model developer and ecosystem modeller at the Ottawa Research and Development Centre, Agriculture and Agri-Food Canada. He primarily focuses on the application of mechanistic models to conduct assessments of GHG emissions from various agricultural practices along with evaluating the long-term sustainability of crop production and soil health under present and future climate variability/change. Recent focus is on understanding and improving models for simulating nutrient/water flows in cropping systems, particularly in cool weather conditions. Brian participates in several international studies focusing on inter-comparison and improvement of agricultural models.

Dr Ward Smith is a Physical Scientist, Project lead in Agri-Environmental modelling at the Ottawa Research and Development Centre, Agriculture and Agri-Food Canada. He has 25 years of experience working with scientists from many disciplines (agronomy, soil science, atmospheric research) to integrate new knowledge into process-base agricultural models with a focus on estimating management impacts on crop yields, soil nutrient cycling, soil carbon sequestration, GHG emissions, nutrient loss to volatilization, runoff and drainage. His research focuses on 1) Development and validation of process-based model mechanisms, 2) Investigation of management impacts on crop production, environmental sustainability, 3) Investigation of the impact of climate variability and climate change on cropping system resilience, 4) International activities on assessing the current state and improvement of agricultural models to enhance understanding and 5) Integration of new modelling approaches into programs in Canada to estimate national GHG inventories and emission intensities from Agriculture.

Per-Anders Hansson is professor at the Department of Energy and Technology at the Swedish University of Agricultural Sciences, Uppsala, Sweden. He has a background in Biosystems Engineering and has worked as professor since year 1997. He leads a group with approx. 25 researchers, strong especially in environmental systems analyses of food, biomaterials and energy systems. Main methodology is LCA (life cycle assessment) and one research aim is to further develop the methodology to be better suited for evaluation of bio-based production systems.

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