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Cultivation of forage maize in boreal conditions – Assessment of trade-offs between increased productivity and environmental impact

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ABSTRACT

The cultivation of whole crop forage maize (*Zea mays* L.) for cattle feed has a potential for increased forage yield while reducing nitrogen (N) fertilisation compared to perennial grass-based systems. However, the possible environmental trade-offs of forage maize cultivation remain unknown in the boreal region due to the short growing season which limits cultivation practices. The aim of this study was to compare the environmental impact of forage maize with more widely cultivated forage crops in Finland that include perennial silage grass mixtures and whole crop spring cereal harvested as silage. The use of plastic mulch film in forage maize cultivation was included in the assessment as well. A life cycle assessment (LCA) was conducted including impact categories for global warming potential; marine and freshwater eutrophication; terrestrial acidification; freshwater, marine and terrestrial ecotoxicity; land use; and fossil resource depletion. Additionally, soil organic carbon (SOC) stock changes under long-term cultivation of the studied forage crops were simulated with the C-TOOL and Yasso20 models with methodological comparisons. The only clear differences between the studied crops were that the land use was lower (−26–48%) for forage maize, and the freshwater eutrophication (+59–67%) and terrestrial acidification (+10–57%) were higher for perennial grasses compared with other forages. A risk for decreased SOC stock under continuous forage maize cultivation was observed. Forage maize could be used to supplement perennial grass cultivation without major associated environmental risks. Future research shall be conducted on the effect of forage choices on the environmental impact of boreal dairy milk production and on decreasing the current high uncertainty associated with nitrous oxide (N_2O) emission factors and SOC stock modelling choices.

Abbreviations: a.i, active ingredient; C, carbon; CO, carbon monoxide; CO_2, carbon dioxide; DM, dry matter; ELY, Economic Development, Transport, and the Environment Centre; FM, fresh matter; GWP, global warming potential; HC, hydrocarbon; LCA, life cycle assessment; LCIA, life cycle impact assessment; LDPE, low-density polyethylene; K, potassium; MC, Monte Carlo analysis; ME, metabolisable energy; MJ, megajoule; N, nitrogen; NIR, near-infra-red; N_2O, nitrous oxide; NH_3, ammonia; NO_x, nitrogen oxide; P, phosphorus; SOC, soil organic carbon; SO_2, sulphur dioxide; TAN, total ammoniacal nitrogen.

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1. Introduction

The environmental impact of dairy cows is strongly related to feed production (Thomassen and De Boer, 2005; Lesschen et al., 2011; Nguyen et al., 2013; Mazzetto et al., 2022). In the boreal biogeographical region of Finland, dairy cattle feeding is traditionally based on grass silage (Virkajärvi et al., 2015), as the grazing period is short and the marginal climate conditions limit the number of crop species options. Regardless of the climatic limitations, the cultivation of forage maize (*Zea mays* L.) in the boreal region has increased during the past decades due to climate warming and enhanced availability of early maturing maize cultivars (Mussadiq, 2012; Liimatainen et al., 2022). Forage maize harvested as immature whole crop biomass is associated with high dry matter (DM) yield per hectare. In previous field experiments in the boreal region, forage maize yielded 10–20 DM Mg ha^{-1} (Hetta et al., 2012; Mussadiq et al., 2012; Seleiman et al., 2013; Liimatainen et al., 2022; Lehtilä et al., 2023), while perennial grasses yielded 8–12 DM Mg ha^{-1} within a growing season (Termonen et al., 2020; Luke, 2021). Regardless of the high yield, forage maize has a low protein content (O'Mara et al., 1998; Hart et al., 2015; Khan et al., 2015), and therefore it requires lower nitrogen (N) fertilisation compared with perennial grass stands as displayed in Finnish nitrate regulation (Nitraattiasetus 1250/2014, 2014). The need for field work, and thus, fuel use is reduced due to harvesting being conducted only once during a growing season for maize while it is typically 2–3 times for grass silages. Maize is an annual crop, and long-term annual crop cultivation is generally associated with lower soil organic carbon (SOC) stocks (Ledo et al., 2020; Heikkinen et al., 2022) and soil erosion (Cosentino et al., 2015; Vogel et al., 2016; Ruf et al., 2018) compared with long-term perennial crop cultivation. Furthermore, the generally greater herbicide use for forage maize than for perennial grasses may entail environmental risks (Mussadiq, 2012).

Currently, forage maize cultivation in Finland is limited to southern Finland, as the low temperature still restricts the production of forage maize yield with sufficient feeding quality (Lehtilä et al., 2023). Mulch films are used to protect maize seedlings from both cool temperatures and water deficit during the early growing season and to improve DM yield under cool climate conditions (Lehtilä et al., 2023). Nevertheless, an unexplored, potential trade-off remains between the increased biomass yield and emissions to the environment from mulch film manufacturing and use. In previous studies, biodegradable plastic mulch films have been associated with chemical pollution and microplastic contamination (Markowicz and Szymánska-Pulikowska, 2019) along with an increased total global warming potential (GWP) of grain maize under the climate conditions of Central China (Gao et al., 2022). Also, previous research has been conducted on mulch film manufacturing (Bos et al., 2007), without considering the yield effect of mulch use.

Our main study goal was to assess the impact of cultivating various forage crops under boreal climate conditions in Finland. For forage maize, the environmental impact of using mulch film was assessed to evaluate the trade-offs between increased yield and environmental impacts over the growing season in Finland. Furthermore, the potential for SOC stock change under long-term cultivation of assessed forage crops was simulated with different methodological scenarios, to assess the further impact of management on the environment. We formed a hypothesis that forage maize cultivation, especially with mulch film, is associated with a large environmental impact. However, the high biomass yield of forage maize will offset environmental impacts when assessed per yield unit.

2. Material and methods

2.1. Scope, system boundary, and functional units

An attributional life cycle assessment (LCA) methodology (ISO 14040, 2006) was applied to assess the environmental impact during the life cycles related to forage cultivation. The assessed system extended from cradle to silo, thus including manufacturing of cultivation inputs, field emissions from e.g., soil, field operations and fertilisers, ensiling and related transports throughout the life cycle (Fig. 1). The LCA was conducted for five forage crops:

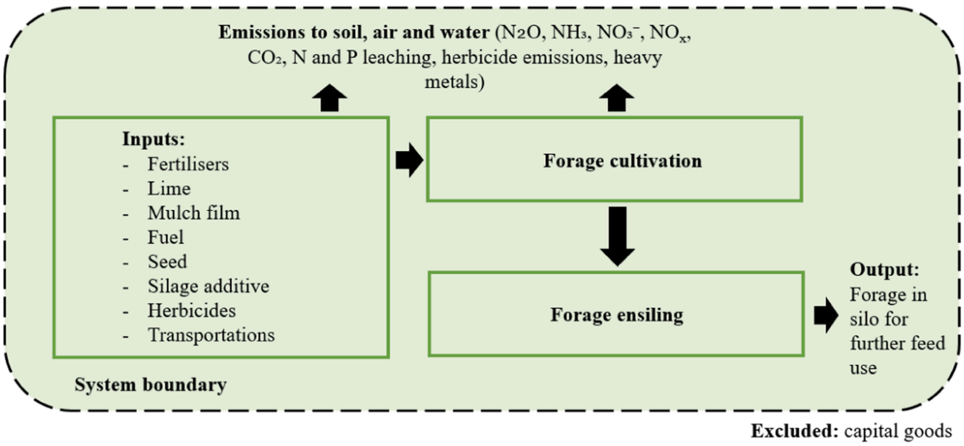

Fig. 1. System boundary of the studied forage cultivation system.

i) Forage maize grown with oxo-biodegradable plastic mulch film (abbreviated maize-MF)
ii) Forage maize grown without mulch film (maize-NM)
iii) Silage grass; a mixture of grass hay species (grass-H)
iv) Silage grass; a mixture of grass hay species and red clover (grass-CL)
v) Whole crop cereal silage, spring cereal (cereal)

The studied system represented forage cultivation in Finland with field experiment-level input and output rates for each forage crop. The systems had one product, i.e., harvested and ensiled forage yield, and no co-products. The functional units of the study were a tonne (1 Mg) of DM forage yield and a megajoule (MJ) of forage yield, which represented the metabolisable energy (ME) content of the forage yield.

2.2. *LCA inventory data collection*

2.2.1. *Crop production system*

The data regarding the yield and input use of forage cultivation were collected from field experiments conducted on mineral soils in Finland, except for the cereal data, which were supplemented with relevant Swedish field experiment results (see related references below). The field experiment data was from the years 2002–2021. Fertilisation of the field experiments was based fully on mineral fertilisers, and hence, manure application was not used in the baseline model.

Maize-MF and maize-NM data were collected from a three-year field experiment conducted in two locations in southern and central Finland (Lehtilä et al., 2023; Table 1). Grass-H data was collected from Finnish Official Cultivar Trials (Luke, 2021) results and Termonen et al. (2020) field experiments. Grass-H data represented grass hay species, timothy (*Phleum pratense* L.) and meadow fescue (*Fesctuca pratensis* L.). Grass-H was assumed to be perennial (four-year ley), to have three cuts per growing season, and to represent an average of three harvest years (grass ley establishment year excluded). Grass-CL yield data were calculated based on grass-H yield rate and the typical yield ratio (0.86) between grass-H and grass-CL (Huuskonen et al., 2012; Kykkänen et al., 2020) to enhance comparability between the two grass types, as grass-CL yields are often highly variable in Finland. Grass-CL was assumed to be perennial (four-year ley), to have two cuts per growing season, and to be composed of approximately 70% grass hay species and 30% red clover (*Trifolium pratense* L.) in the seed mixture. Cereal data were obtained from field experiments (Manni et al., 2021) conducted in northern Finland, supplemented with relevant Swedish field experiment data (Nadeau, 2007; Wallsten, 2008) due to Finnish data limitations. Cereal was assumed to be a pure stand of spring-sown barley (*Hordeum vulgare* L.), oats (*Avena sativa* L.), triticale (x *Triticosecale* Wittm.), or wheat (*Triticum aestivum* L.), as no consistent differences were observed between the yields of the cereal species.

Fertilisation rates (N, phosphorus (P), and potassium (K)) were based on the field experiment data, current crop-specific fertilisation recommendations and expert assumptions, all within the current N regulation (Nitraattiasetus 1250/2014, 2014) and P regulation (Fosforiasetus 64/2023, 2023) applied in Finland (Table 1). Maize-MF, maize-NM, and cereal stands were established with a combined seed and fertiliser drill (i.e., fertiliser incorporated into the soil), whereas grass-H and grass-CL stands were top-dressed (i.e., fertiliser spread on the soil surface).

The liming rate was based on Nordkalk's (2019) recommendation to fill the gap between the average pH of Finnish mineral soils and the recommended pH for the assessed forage crops. Liming was assumed to be conducted every five years, and the lime quantity was allocated uniformly for each year of the five-year rotation (Table 1).

Table 1
Input and output quantities used in the modelling of forage crops.

			Maize-MF	Maize-NM	Grass-H	Grass-CL	Cereal
Input	Fertilisation	N kg ha^{-1}	150	150	220	100	90
		P kg ha^{-1}	14	14	20	20	10
		K kg ha^{-1}	150	150	80	80	30
	Liming	kg ha^{-1}	1000	1000	1000	1000	1000
	Mulch film	kg ha^{-1}	60	0	0	0	0
	Seed	kg ha^{-1}	35	35	8[1]	8[1]	210
	Fuel use	kg ha^{-1}	82	76	39	33	60
	Herbicide	g a.i. ha^{-1}	2 015[2]	2 015[2]	520[3]	22.5[4]	520[3]
	Silage additive	l (FM Mg)$^{-1}$	5	5	5	5	5
Output	DM yield	Mg ha^{-1}	16.2	13.1	9.7	8.4	9.7
	DM content	g (FM kg)$^{-1}$	270	240	350	350	360
	ME content	MJ (DM kg)$^{-1}$	10.3	10.3	10.6	10.2	9.6

a.i. = active ingredient; DM = dry matter; FM = fresh matter; ME = metabolisable energy; MJ = megajoule.
Maize-MF = Forage maize grown with mulch film, Maize-NM = Forage maize grown without mulch film, Grass-H = Silage grass; a mixture of grass hay species, Grass-CL = Silage grass; a mixture of grass hay species and red clover, Cereal = Whole crop cereal silage, spring cereal.
[1]The seed rate for grass-H and grass-CL is the total seed rate on the establishment year (32 kg ha^{-1}) divided by the length of continuous cultivation of the grass ley (four years)
[2]Pendimethalin (2 000 g a.i. ha^{-1}), rimsulfuron (7.5 g a.i. ha^{-1}), thifensulfurol methyl 7.5 g a.i. ha^{-1})
[3]MCPA (400 g a.i. ha^{-1}), fluroxypyr (80 g a.i. ha^{-1}), clopyralid (40 g a.i. ha^{-1})
[4]Amidosulfuron (22.5 g a.i. ha^{-1}).

Seed rates were based on field experiments and expert assumptions (Table 1). Fuel use was calculated using estimates for various field operations (Tables S1, S2). Herbicide use rates and active ingredients (a.i.) were based on field experiment data and supplemented with Finnish rural advisory organisation ProAgria recommendations (Peltonen, 2021). The forage yields were assumed to be ensiled in bunker silos. The silage additive consisted of formic acid (75%) with a quantity of 5 litres (Mg forage yield fresh matter)$^{-1}$. The DM content and ME content (MJ (DM kg)$^{-1}$) of grass-H, grass-CL, and cereal were based on Finnish Feed tables for ruminants (Luke, 2023). The DM and ME contents of maize-MF and maize-NM were based on laboratory analyses conducted for samples collected from forage maize field experiments (Liimatainen et al., 2022; Lehtilä et al., 2023).

2.2.2. Input manufacturing and transports

Fertiliser emissions and emissions from seed, herbicide, fuel, and silage additive manufacturing were obtained from the Ecoinvent database v. 3.7.1. (Wernet et al., 2016). Limestone data was obtained from the Agri-footprint 5.0 database (van Paassen et al., 2019). A detailed description of the used data sets is presented in Table S3.

The mulch film used in the model was oxo-biodegradable clear film. Due to data limitations, the ingredients of the mulch film were assumed based on relevant literature. The modelled film consisted of low-density polyethylene (LDPE) and pro-oxidant with a ratio of 98:2 (Reddy et al., 2009; Grigale et al., 2010). The pro-oxidant composition was 33% iron, 33% manganese, and 33% cobalt (Grigale et al., 2010). Emissions from mulch film manufacturing were gathered from the literature (Grigale et al., 2010; Gironi and Piemonte, 2011; Liptow and Tillman, 2012; Benavides et al., 2020) supplemented with Ecoinvent data (Table S3).

Road transport distances from the manufacturer to the farm were 500 km for fertilisers and herbicides, 300 km for silage additive, 150 km for limestone, 400 km for fuel, 100 km for grass and cereal seeds, 3000 km for maize seeds, and 3500 km for mulch film. Sea transportation distances were 200 km for mulch film and 100 km for maize seed. The distances were based preferably on the expert assumptions described in Hietala et al. (2021) and secondly on own assumptions. Emissions from input transportation were modelled with Ecoinvent data (Table S3).

2.3. LCA calculation methods

2.3.1. Field emissions from forage cultivation

Direct nitrous oxide (N_2O) emissions from N fertilisation and crop residues were calculated using the Regina et al. (2013) method based on measurements conducted in Finland (more details about emission calculations in Table S4). Indirect N_2O emissions were calculated using the IPCC (2019) Tier 2 emission factors. Carbon dioxide (CO_2) emissions from limestone application were calculated with IPCC (2006) emission factor. Ammonia (NH_3) volatilisation from N fertilisers was estimated using the Grönroos et al. (2017)'s method. Nitrogen oxide (NO_x) emissions were calculated with EEA (2019) Tier 1 emission factors. The N and P leaching and run-off were calculated using the Saarinen et al. (2011)'s method.

Fuel combustion emissions to air (N_2O, CO_2, CH_4, sulphur dioxide (SO_2), NO_x, hydrocarbon (HC), and carbon monoxide (CO)) at field operations were estimated using Lipasto, the Finnish national database (VTT, 2017). Herbicide emissions to air, surface water, and groundwater were modelled with the PestLCI2.0 model (Dijkman et al., 2012). Heavy metal emissions to soil, surface water, and groundwater were modelled with the method in Nemecek and Schnetzer (2011).

The CO_2 emissions to the air following the degradation of mulch film were assumed based on the typical carbon content of LDPE film materials (80%, Smeaton et al., 2021); thus, the whole carbon content was assumed to be emitted to the air as CO_2 (2.9 kg CO_2 (kg mulch film)$^{-1}$). Other field emissions from mulch film use and degradation were excluded from the assessment due to limited data suitable for boreal climate conditions.

2.3.2. Simulation of SOC stock change

The potential of SOC change on mineral soil considering a response to long-term continuous forage crop cultivation was simulated with two process-based soil C models; C-TOOL (v.2.3; Taghizadeh-Toosi et al., 2014) and Yasso20 (Viskari et al., 2022). C-TOOL simulates SOC stock changes in topsoil and subsoil down to 1 m depth, and considers e.g., air temperature, soil C/N ratio and clay content. Yasso20 simulates SOC stock changes in 1 m layer, and considers e.g., air temperature, precipitation, and solubility (AWEN fractions) of C input to soil.

The weather data were gridded data obtained from the Finnish Meteorological Institute (unpublished data). The data included average annual rain sums and average monthly temperatures from regional Centres for Economic Development, Transport, and the Environment (ELY) in Finland from the year 1961 to the year 2021 (data from the year 1961 was used for the initialisation years before the year 1961). Weather data from the ELY Centres in Åland and Lapland were excluded, as most of the agricultural land in Finland is located in continental Finland between latitudes 60 and 65°N (Palosuo et al., 2015). For future scenarios, two approaches were used: i) continuously the same monthly temperatures as in 2021 and ii) a gradual increase totalling + 2.5 °C in the monthly temperatures between years 2022 and 2121. That was performed by assuming a linear increase in temperature from 2022 to + 2.5 °C in 2121 (Allen et al., 2018).

The models were initialised with a steady-state assumption. The weather during the initialisation period represented the average monthly temperature and precipitation in the simulation region for 1961–2021. Carbon (C) input during the initialisation period represented typical farm-level silage grass with annual cattle slurry manure application in Finland (total 2.8 Mg C ha^{-1} yr^{-1}). As a result of the initialisation, the initial steady-state SOC stocks were 120 Mg C ha^{-1} with C-TOOL and 57 Mg C ha^{-1} with Yasso20 for the whole soil profile (100 cm). These initial SOC stocks were used primarily for further simulations (i.e., C-TOOL initial SOC stock was used for C-TOOL simulations and vice versa). For C-TOOL and Yasso20 model comparisons, we carried out simulations with two initial SOC stocks for both models.

The C inputs of the simulated forage crops (maize, grass-H, grass-CL, and cereal) were calculated using allometric functions in the C-TOOL methodology (Taghizadeh-Toosi et al., 2014, 2020) (Table 2). For C-TOOL, the C input was presented separately for topsoil (0–25 cm) and subsoil (25–100 cm). For Yasso20, the C input was given for the whole soil profile (100 cm) as AWEN fractions that were based on Karhu et al. (2012) for grass-H, grass-CL, and cereal, and on the INRAE data set (Thiébeau et al., 2021) for maize-MF and maize-NM. An uncertainty of ± 50% from the baseline C input (Bolinder et al., 2002, 2007, 2008; Pausch and Kuzyakov, 2017) for each forage crop was considered in the simulation approach.

The change in SOC stock from initial SOC stock to SOC stock over 100 years (SOC $stock_{change}$; Mg C ha^{-1}) was calculated as:

$$\text{SOC stock}_{change} = \text{SOC stock}_{init} - \text{SOC stock}_{100}$$

where SOC $stock_{init}$ = initial SOC stock at the beginning of the simulation period (120 Mg C ha^{-1} with C-TOOL and 57 Mg C ha^{-1} with Yasso20), SOC $stock_{100}$ = SOC stock after cultivation of a forage crop continuously for 100 years.

The average annual CO_2 emission ($Annual_{CO2}$, kg CO_2 ha^{-1} yr^{-1}) of continuous cultivation of a forage crop, allocated for 20 years (adapted from IDF, 2022), and calculated as:

$$\text{Annual}_{CO2} = \text{SOC stock}_{change} / 20 \times 44/12 \times (-1)$$

Furthermore, $Annual_{CO2}$ was included in the GWP of forage crops.

2.4. Life cycle impact assessment

The environmental impact of forage crop cultivation was modelled using OpenLCA software (v. 1.10.3.) with Ecoinvent (v. 3.7.1) and Agri-footprint (v. 5.0) databases. The ReCiPe 2016 Midpoint (H) LCIA method (Huijbregts et al., 2017) was used to calculate GWP (kg CO_2-eq), freshwater and marine eutrophication potential (kg P-eq and kg N-eq), terrestrial acidification (kg SO_2-eq), freshwater, marine and terrestrial ecotoxicity (kg 1,4-DCB-eq), land use (m^2 crop-eq), and fossil resource depletion (kg oil-eq).

2.5. Sensitivity and uncertainty analyses

To estimate the uncertainty of the results, we conducted a common LCA uncertainty assessment method – the Monte Carlo (MC) assessment (Lloyd and Ries, 2007) to illustrate the uncertainty of the estimated LCA results. In MC, the chosen parameters of LCA are assigned uncertainty ranges, and the model is run a fixed number of times to provide random parameter combinations and, thus, the range of results. The uncertainty ranges designated in this study are shown in Table S5. The MC simulation was performed in Microsoft Office 365 Excel using 100 iterations (Heijungs, 2020). Finally, a standard deviation of the 100 iterations was calculated and attached as a probability distribution for the environmental impact results. The probability distribution used was standard deviation, which was attached in the figures showing results for the different impact categories.

The sensitivity of the environmental assessment was tested using the IPCC (2019) Tier 2 method for direct N_2O instead of the baseline method used in this study (Regina et al., 2013).

Additionally, the source of N fertilisation was modified using sensitivity analysis as below:

i) Baseline scenario = only mineral N fertiliser (Table 1)

Table 2
Values used in the calculation of annual C input to soil, calculated according to Taghizadeh-Toosi et al. (2014) with modifications for grass below-ground C input by Taghizadeh-Toosi et al. (2020). Values represent baseline C inputs.

			Maize-MF	Maize-NM	Grass-H	Grass-CL	Cereal
For C-TOOL and Yasso20	α	Ratio	0.85	0.85	0.80	0.80	0.75
	δ	Ratio	0	0	0	0	0
	β	Ratio	0.150	0.150	0.262	0.262	0.170
	C content	% of DM	45	45	45	45	45
	Total C input to soil (0–100 cm)	Mg ha^{-1} yr^{-1}	2.80	2.26	3.03	2.62	2.65
For C-TOOL	C input to topsoil (0–25 cm)*	Mg ha^{-1} yr^{-1}	2.35	1.90	2.45	2.12	2.29
	C input to subsoil (25–100 cm)*	Mg ha^{-1} yr^{-1}	0.45	0.37	0.58	0.50	0.36
	Soil clay content	%	5	5	5	5	5
	Soil C/N ratio	Ratio	13	13	13	13	13
For Yasso20**	A	Ratio	0.77	0.77	0.46	0.46	0.71
	W	Ratio	0.01	0.01	0.32	0.32	0.08
	E	Ratio	0.03	0.03	0.04	0.04	0.03
	N	Ratio	0.19	0.19	0.18	0.18	0.18

α = Harvest index of the main crop relative to above-ground biomass; δ = Biomass of secondary crop product as a yield proportion of the main crop product; β = Root and exudate C as a proportion of total C assimilation; DM = dry matter
Maize-MF = Forage maize grown with mulch film; Maize-NM = Forage maize grown without mulch film; Grass-H = Silage grass; a mixture of grass hay species; Grass-CL = Silage grass; a mixture of grass hay species and red clover; Cereal = Whole crop cereal silage, spring cereal
*Assumption: 90% of the total C input deposited to topsoil, 10% to subsoil
**Carbohydrate fractions: A = acid-soluble; W = water-soluble; E = ethanol soluble; N = non-soluble.

ii) Manure scenario = part of the mineral N fertiliser substituted with cattle manure (the sum of manure TAN and mineral fertiliser N same as presented in Table 1)

The manure application rate represented a typical quantity applied in Finland (30 m^3 manure ha^{-1}). The manure was assumed to be spread in spring before sowing for annual crops (maize-MF, maize-NM and cereal) and in summer after the first grass cut for perennial grasses (grass-H, grass-CL). The N content of applied manure was calculated using a default value for the total ammoniacal N (TAN) content of cattle slurry (Finnish Food Authority, 2022). The difference between manure TAN and the total N rate (Table 1) was fulfilled with mineral N fertiliser. For manure, only emissions released at manure application were considered. Emission factors used for manure spreading are presented in Table S4. SOC stock change simulation was not conducted for the manure application scenario.

3. Results

3.1. Global warming potential

The average GWP (without SOC stock change) of cultivating the different forage crops varied between 217 and 369 kg CO_2-eq per Mg of DM, and the GWP tended to be lower for grass-CL compared with other forage crops (Fig. 2). Most GWP related to field N_2O emission, which accounted for approximately 44–50% of GWP for annual crops and 16–29% for perennial grasses. The second and third largest GHG emission sources were the manufacturing of N fertilisers (13–34%) and field CO_2 emissions from liming (8–24%), respectively. Manufacturing of silage additive (8–14%), fuel combustion of field operations (approximately 5%) and manufacturing of K fertiliser (2–10%) contributed to the remaining GWP together with other emission sources. For maize-MF, manufacturing and use of mulch film contributed 8% of the total GWP.

3.2. Eutrophication and acidification

Marine eutrophication varied between 0.31 and 0.47 kg N-eq per Mg of DM (Fig. 3) and was mainly related to field emissions,

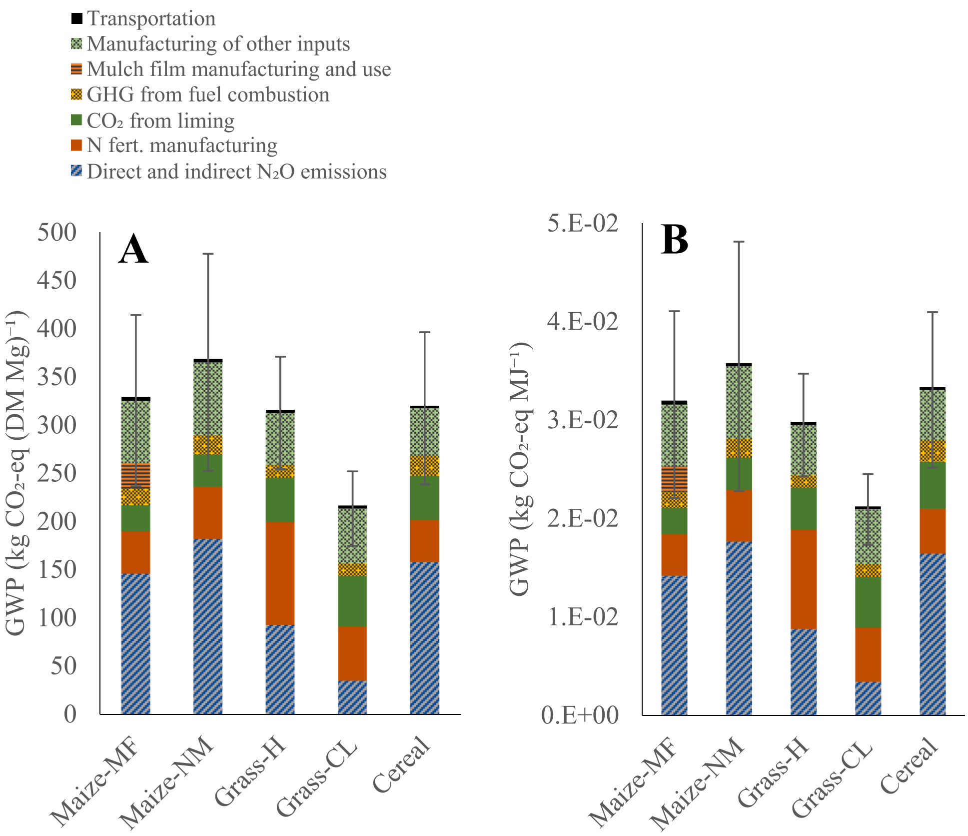

Fig. 2. Global warming potential (GWP, without soil organic carbon stock change) of cultivating different forage crops A) per 1 Mg dry matter (DM) yield and B) per 1 megajoule (MJ) of metabolisable energy yield. The error bars represent ± standard deviation based on Monte Carlo analysis. Maize-MF = forage maize grown with mulch film, Maize-NM = forage maize grown without mulch film, Grass-H = silage grass; a mixture of grass hay species, Grass-CL = silage grass; a mixture of grass hay species and red clover, Cereal = Whole crop cereal silage, spring cereal.

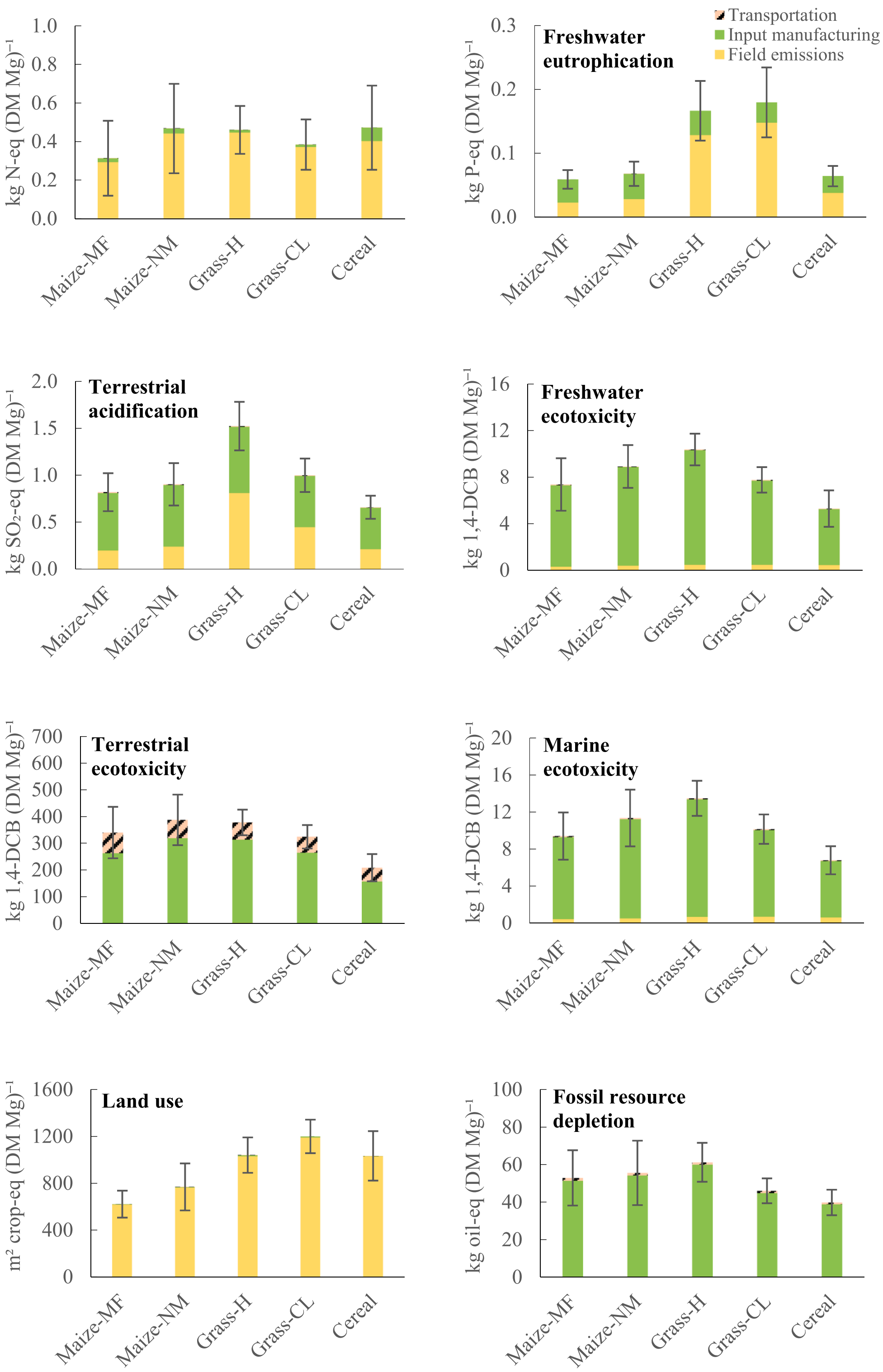

Fig. 3. Environmental impacts of forage crops per 1 Mg dry matter (DM) yield. The error bars represent ± standard deviation based on Monte Carlo analysis. DCB = dichlorobenzene. Maize-MF = Forage maize grown with mulch film, Maize-NM = Forage maize grown without mulch film, Grass-H = Silage grass; a mixture of grass hay species, Grass-CL = Silage grass; a mixture of grass hay species and red clover, Cereal = Whole crop cereal silage, spring cereal.

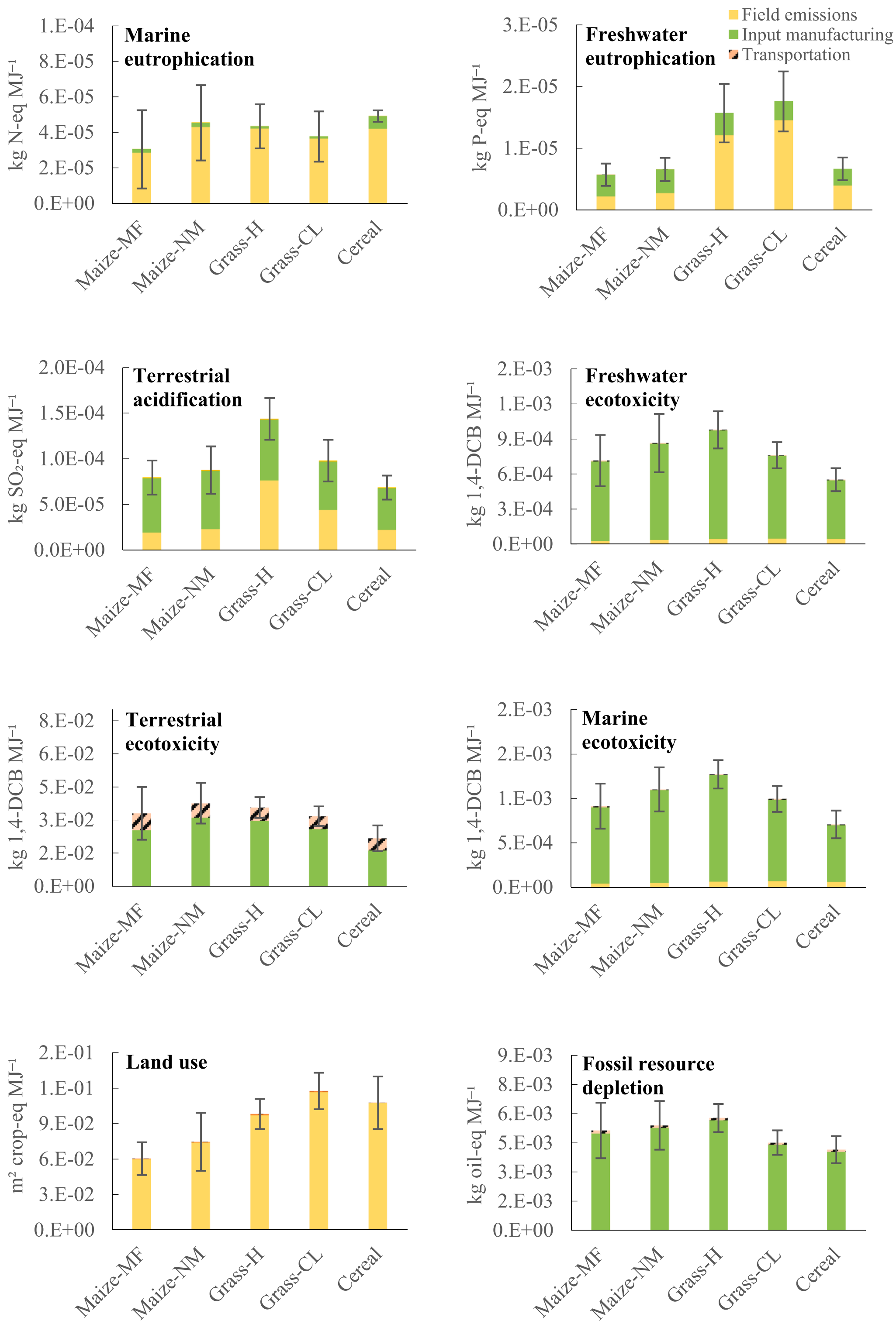

Fig. 4. Environmental impacts of forage crops per 1 megajoule (MJ) of metabolisable energy yield. The error bars represent ± standard deviation based on Monte Carlo analysis. DCB = dichlorobenzene. Maize-MF = Forage maize grown with mulch film, Maize-NM = Forage maize grown without mulch film, Grass-H = Silage grass; a mixture of grass hay species, Grass-CL = Silage grass; a mixture of grass hay species and red clover, Cereal = Whole crop cereal silage, spring cereal.

namely N leaching from the field (85–97%). Freshwater eutrophication varied from 0.06 to 0.18 kg P-eq per Mg of DM and was 2.5–3 times higher for perennial grasses compared with annual crops (Figs. 3 and 4). The P leaching from the field was the greatest source of freshwater eutrophication (38–59% for annual crops, 77–82% for perennial grasses), followed by silage additive manufacturing which contributed approximately 8%, 19%, and 28% of the freshwater eutrophication for perennial grasses, cereal, and both assessed types of maize, respectively.

Terrestrial acidification was highest for grass-H (Figs. 3 and 4). For perennial grasses receiving top-dressed fertilisation, most of the acidifying emissions were from field NH_3 emissions (23–28%), N fertiliser manufacturing (21–27%), and field NO_x emissions (17–21%). For annual crops with fertiliser incorporation, the greatest sources of acidifying emissions were N fertiliser manufacturing (20–23%) and field NO_x emissions (16–18%).

3.3. Ecotoxicity

Freshwater, terrestrial, and marine ecotoxicities were generally lower for cereal than for other forage crops, but the results contained high uncertainty (Figs. 3 and 4). The three largest sources of ecotoxic emissions were N fertiliser, K fertiliser, and silage additive manufacturing. Those together accounted for more than 85% of the freshwater and marine ecotoxicity, and more than 65% of the terrestrial ecotoxicity. For terrestrial ecotoxicity, road transportation contributed approximately 17–25% of the total emissions. Heavy metal emissions from cultivation contributed approximately 5–10% of the total freshwater and marine ecotoxicity. The role of herbicide use was minor (<1%) for all the assessed forage crops and ecotoxicity categories.

3.4. Land use and fossil resource depletion

The total (direct and indirect) land use varied between 621 and 1200 m^2 a crop-eq per Mg of DM and was lower for maize-MF

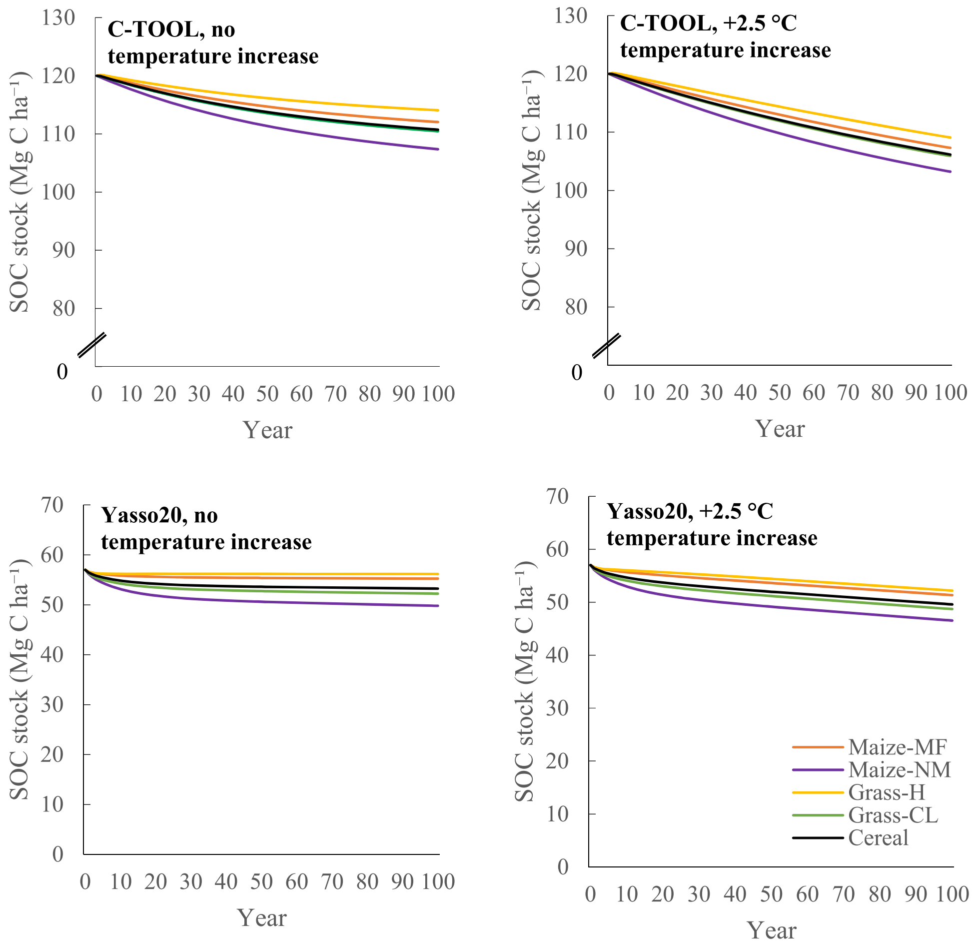

Fig. 5. Soil organic carbon (SOC) stock change under cultivation of forage crops, modelled with C-TOOL and Yasso20. Maize-MF = Forage maize grown with mulch film, Maize-NM = Forage maize grown without mulch film, Grass-H = Silage grass; a mixture of grass hay species, Grass-CL = Silage grass; a mixture of grass hay species and red clover, Cereal = Whole crop cereal silage, spring cereal.

compared with perennial grasses and cereal (Fig. 3). Direct land use in forage cultivation contributed approximately 99% of the land use, and the role of indirect land use was therefore minor.

Production of 1 Mg forage yield (DM) depleted fossil resources equal to 40–61 kg oil-eq. Furthermore, the depletion of fossil resources tended to be lower for cereal and grass-CL compared with the other forage crops (Figs. 3 and 4). Most fossil resource depletion was related to the manufacturing of N fertiliser (26–55%) and the silage additive (23–36%).

3.5. SOC stock change simulation

Cultivation of all the studied forage crops led to a decrease in SOC stock compared with the initial SOC stock in all scenarios and simulated by both selected models (Fig. 5). The decrease in SOC stock was largest for maize-NM based on both models and smallest for grass-H. Including the temperature increase of + 2.5 °C in the monthly temperatures decreased the SOC stocks on average by 65% compared with the scenario with no temperature increase (Fig. 5).

When SOC stock change compared with initial SOC stock was included in the GWP of forage crops, the GWPs increased for all forage crops (Fig. 6). The percentual increase in GWP was highest for grass-CL and lowest for maize-MF. A gap between the GWPs of maize-MF and maize-NM was observed. Including the temperature increase led to 11–20% higher GWPs compared with the scenario with no temperature increase.

The uncertainty of SOC stock change over 100 years ranged from approximately − 70% to + 130% when simulated with C-TOOL and from approximately − 325% to + 350% when simulated with Yasso20 when the uncertainty of the C input (± 50% from the baseline C input) was introduced in the models (Table 3). Generally, when the C input was increased by 50% from the baseline, the SOC stock changes under forage crop cultivation became positive compared with the initial SOC stock (except for maize-NM simulated with C-TOOL).

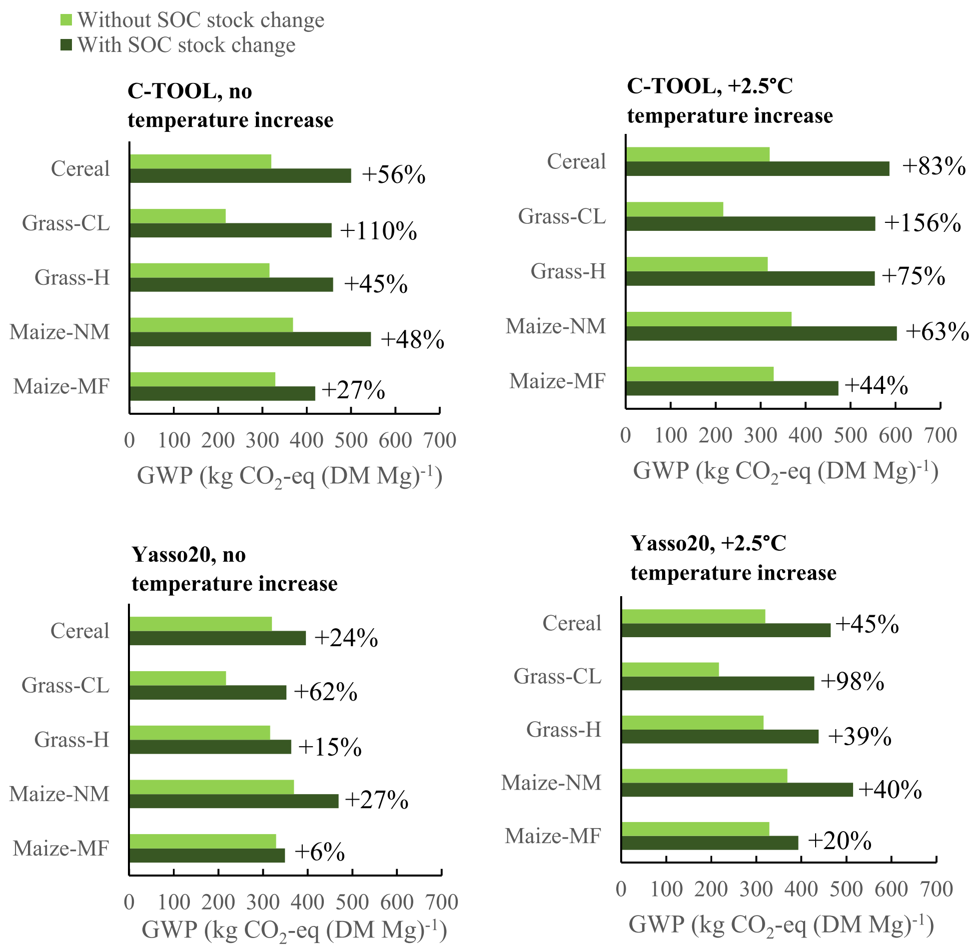

Fig. 6. Inclusion of the simulated soil organic carbon (SOC) stock change simulated with two models (C-TOOL and Yasso20) in the global warming potential (GWP) of forage crops. The SOC stock change represents the annual CO_2 emissions from SOC stock change over a 100-year period compared with the initial SOC stock (CTOOL = 120 Mg C ha^{-1}; Yasso20 = 57 Mg C ha^{-1}), with CO_2 emissions allocated for 20 years. Maize-MF = Forage maize grown with mulch film, Maize-NM = Forage maize grown without mulch film, Grass-H = Silage grass; a mixture of grass hay species, Grass-CL = Silage grass; a mixture of grass hay species and red clover, Cereal = Whole crop cereal silage, spring cereal.

Generally, Yasso20 showed a steeper decrease in SOC stocks compared with C-TOOL, when the two models were tested with both 57 and 120 Mg C ha^{-1} initial SOC stocks (Fig. A1). Considering an initial SOC stock of 120 mg C ha^{-1}, the average SOC stock changes over 100 years were − 9 and − 37 Mg C ha^{-1} for C-TOOL and Yasso20, respectively. With an initial SOC stock of 57 Mg C ha^{-1}, the average SOC stock changes were + 4 and − 8 Mg C ha^{-1} for C-TOOL and Yasso20, respectively.

3.6. Sensitivity analyses

Using the IPCC (2019) Tier 2 method for direct N_2O emissions calculation led to an average − 17% lower GWP per 1 DM Mg for annual crops compared with the method by Regina et al. (2013) (Table 4). However, for perennial grasses, the GWPs per 1 DM Mg increased by 44–55% when the IPCC (2019) method was applied.

When the mineral N fertiliser was partially substituted with cattle manure, the GWPs per 1 DM Mg increased slightly for annual crops and decreased slightly for perennial grasses (Table A1). Marine and freshwater eutrophication decreased for all forage crops. However, terrestrial acidification increased by approximately 76–187% with manure application for all forage crops.

4. Discussion

4.1. Environmental impacts of forage crops

The GWPs (without SOC stock change) of the forage crops were relatively similar except for the moderately low GWP of grass-CL. That resulted from the low field N_2O emissions and low GHG emissions from N fertilisation manufacturing, which related to a decreased use of mineral N fertiliser due to biological N fixation. The greatest GWP hotspots (without SOC stock change) for all forage crops were related to the use of N fertilisers, similar to previous field crop LCA studies (Parajuli et al., 2017; Joensuu et al., 2021; Hietala et al., 2022 among others). The crucial role and uncertainty related to the N_2O emission factors are acknowledged (Flysjö et al., 2011) and discussed below. The relatively high CO_2 emission related to the lime application was likely related to the high lime input rate in the modelling, as the rates were based on liming recommendations similar to Joensuu et al. (2021). Acid-based silage additive manufacturing was the third most important source of GWP, as previously observed by Tuomisto and Helenius (2008).

The GWPs (without SOC stock change) obtained in this study were mostly similar; regardless of mulch film use in maize cultivation. Although manufacturing as well as mulch film use accounted for approximately 8% of the GWP of maize-MF, the increase in maize yield decreased the total GHG emissions per yield unit. Mulch film use was not associated with an increased risk for other environmental impacts modelled in this study. Nevertheless, our results are limited to fossil-based, oxo-degradable mulch films, which are no longer available on the European market (Lehtilä et al., 2023). In previous studies, other degradable mulch materials have shown similar yield effects with oxo-biodegradable mulch films (Tofanelli and Wortman, 2020), including the mulch films currently allowed in the EU (e.g. starch-based and paper mulches). In the future, the environmental impact of using alternative, biobased film materials require further assessment. Also, field measurements would be required to validate the effect of mulch film on field GHG emissions, chemical pollution, and microplastic pollution under boreal conditions.

The GWPs of all studied forage crops included major uncertainty strongly related to field N_2O emissions. With the IPCC (2019) Tier 2 method, the GWPs of perennial grasses were higher than with the Regina et al. (2013) method, leading to uncertain differences in GWPs between annual crops and perennial grasses. The difference between the two methods relates to the consideration of N_2O from crop residues. Regina et al. (2013)'s method does not consider N_2O separately from crop residues, while IPCC (2019) method considers N_2O emissions from above-ground and below-ground crop residues, and thus, includes N_2O emitted at grass renewal. When using the IPCC (2019) method, the N_2O emissions of perennial grasses were higher compared with annual crops, as grasses typically have greater crop residue biomass compared with annual crops. However, the used methods did not consider the effect of vegetative soil cover or

Table 3
SOC stock change over 100 years (Mg C ha^{-1} yr^{-1}; SOC stock$_{change}$) compared with an initial SOC stock (C-TOOL = 120 Mg C ha^{-1}; Yasso20 = 57 Mg C ha^{-1}) under long-term cultivation of different forage crops, and the effect of C input uncertainty (±50% from the baseline C input rate, kg C ha^{-1} yr^{-1}) and temperature.

	Model	C input	Maize-MF	Maize-NM	Grass-H	Grass-CL	Cereal
No temperature increase	C-TOOL	-50%	-20	-23	-19	-21	-21
		Baseline	-8	-13	-6	-10	-9
		+50%	+4	-3	+7	+2	+2
	Yasso20	-50%	-16	-19	-15	-17	-17
		Baseline	-2	-7	-1	-5	-4
		+50%	+12	+4	+14	+8	+9
+2.5 °C temperature increase	C-TOOL	-50%	-23	-26	-23	-24	-24
		Baseline	-13	-17	-11	-14	-14
		+50%	-2	-8	-1	-4	-4
	Yasso20	-50%	-18	-21	-18	-20	-19
		Baseline	-6	-10	-5	-8	-7
		+50%	7	0	8	3	4

Maize-MF = Forage maize grown with mulch film, Maize-NM = Forage maize grown without mulch film, Grass-H = Silage grass; a mixture of grass hay species, Grass-CL = Silage grass; a mixture of grass hay species and red clover, Cereal = Whole crop cereal silage, spring cereal.

Table 4
A comparison of the effect of the direct N_2O emission calculation method on the global warming potential (GWP) of cultivating different forage crops. The baseline method = Regina et al. (2013), alternative = IPCC (2019) Tier 2. Difference = percentual difference of alternative method results compared with baseline method results for each crop.

		Maize-MF	Maize-NM	Grass-H	Grass-CL	Cereal
GWP, kg CO_2-eq (DM Mg)$^{-1}$	Regina et al. (2013)	329	369	316	217	320
	IPCC (2019) Tier 2	279	302	456	337	266
	Difference	-15%	-18%	+44%	+55%	-17%

Maize-MF = Forage maize grown with mulch film, Maize-NM = Forage maize grown without mulch film, Grass-H = Silage grass; a mixture of grass hay species, Grass-CL = Silage grass; a mixture of grass hay species and red clover, Cereal = Whole crop cereal silage, spring cereal.

tillage intensity as a separate parameter affecting N_2O emissions, which needs to be acknowledged. In the future, more research is needed on the accuracy of N_2O emission factors suitable for boreal conditions, especially for perennial grasses. Results also highlight the importance of considering grass renewal years in LCA.

In previous studies, Parajuli et al. (2017) obtained GWPs (without SOC stock change) of 273 and 480 kg CO_2-eq (DM Mg)$^{-1}$ for the whole crop maize and grass–clover mixture, respectively, both grown for biorefinery use in Denmark. Mogensen et al. (2014) suggested GWPs (without SOC stock change) of 224, 503, 404, and 285 kg CO_2-eq (DM Mg)$^{-1}$ for whole crop maize, grass, grass–clover, and whole crop barley grown in Denmark, respectively. When compared with these two studies, the differences between the GWPs in our studied forage crops followed a similar pattern, although the gap between the GWPs of annual and perennial crops was smaller in this study when the method by Regina et al. (2013) was applied. It must be noted that the N_2O emission calculations in the previous Danish studies were according to the IPCC (2006).

The higher risk for freshwater eutrophication and terrestrial acidification for perennial grasses compared with the assessed annual crops was mainly a result of field P leaching and NH_3 emission. The observations are mainly in accordance with Parajuli et al. (2017), who associated maize with lower NH_3 emissions and eutrophication potential (N and P) per yield unit compared with perennial grasses. In this study, the contrast between annual crops and perennial grasses originated mainly in the fertiliser application method. We assumed that perennial grasses were top-dressed, and the annual crops were established with a combined seed and fertiliser drill (fertiliser incorporated into the soil), a common practice in Finland. The top-dressing of P fertiliser increases the leaching of dissolved P (Saarinen et al., 2011). Also, NH_3 emissions were higher for perennial grasses with fertiliser top-dressing. According to the Grönroos et al. (2017)'s method, the NH_3 emission factor is higher for top-dressing than for fertiliser soil incorporation due to climatic conditions in Finland. Field NH_3 emissions seemed to be sensitive for fertiliser type, as including manure in the fertiliser led to distinctly higher – up to 2.8-fold – terrestrial acidification per Mg of DM for all studied forage crops. Regardless of fertiliser type, grass-CL had lower terrestrial acidification compared with grass-H. That indicates a reduction in mineral N fertiliser use (i.e., by biological N fixation), and could provide a way to balance the environmental impact of forage crop cultivation.

The land use requirement is relatively lower to produce forage maize compared with perennial grasses and cereal because of the high yield obtained per hectare of maize. The results are similar to Mogensen et al. (2014), who suggested direct land use of 0.90, 1.35, 1.21, and 1.1 m^2 per kg of DM for forage maize, whole crop barley, grass–clover, and grass; respectively. The results indicate possibilities to improve field area use efficiency with the cultivation of forage maize.

For ecotoxicity, the clearest environmental impact hotspots were related to the manufacturing of inputs while herbicide application had only a minor effect on all the studied ecotoxicity impact categories. Our results are in accordance with Parajuli et al. (2017), who also used PestLCI 2.0 and obtained a pesticide contribution of 0.8% out of the total freshwater ecotoxicity. On the contrary, Fantin et al. (2017) observed an approximately 92% contribution of pesticides to maize cultivation freshwater ecotoxicity when applying the method by Margni et al. (2002). The conflicting results are likely related to varying pesticide application rates for maize and the differences in shares of pesticide emissions to air and soil in the different ecotoxicity LCA inventory methods.

4.2. SOC simulations and inclusion of SOC stock change in GWP

The simulated SOC stock decrease over 100 years corresponded to an average of 64 kg C ha^{-1} yr^{-1} (no temperature increase) and 105 kg C ha^{-1} yr^{-1} (temperature increase included). The results obtained align with the declining trend of SOC stocks of mineral soils in Finland (Heikkinen et al., 2013), and with the fact that SOC stocks are estimated to decrease even more due to climate change (Heikkinen et al., 2022). The higher risk of SOC losses for forage maize compared with high-yielding grass-H was similar to Parajuli et al. (2017), although they simulated a net SOC stock increase for grass–clover and ryegrass unlike that simulated in our study. Using organic amendments would be recommended for preventing SOC stock losses. Cover crops may also help to maintain SOC stocks (Poeplau and Don, 2015), and have shown positive effects on maize yields while legumes have been used as cover crops (Miguez and Bollero, 2005).

Following the SOC stock change inclusion, the GWPs of the studied forage crops increased either slightly or drastically, depending primarily on the forage crop, the model used, and the temperature scenario. The GWP of maize-MF increased the least due to SOC stock change, as maize-MF had a relatively high annual C input to soil due to high yield. The highest increase in GWP was observed for grass-CL because its GWP was originally low and because the annual C input to the soil was lower than for other forage crops. After the inclusion of SOC stock change, the gap in GWP between grass-CL and grass-H became less narrow. This was related to a lower C input to the soil from grass-CL, as the C-TOOL method assumes grass root biomass to correlate with above-ground biomass. However, because the above-ground and below-ground biomasses of grasses do not necessarily correlate (Kykkänen et al., 2022; Palosuo et al., 2015), the results of this SOC simulation contain recognised uncertainty that was considered with the ± 50% C input scenarios. In the future, the below-ground biomass estimates of perennial crops and maize grown under boreal conditions would require further research.

The steeper decrease in SOC stock simulated with Yasso20 compared with C-TOOL is likely related to the different decomposition rates in the models along with the different default parameters considered. Compared with C-TOOL, Yasso was originally developed for forest soil and provides more detailed decomposition rates for the AWENH fractions, and accounts for precipitation, unlike C-TOOL. On the other hand, C-TOOL – originally developed for agricultural mineral soils – considers soil clay content and C/N ratios of soil and C inputs, and these parameters were not considered in Yasso20. Another difference between the two models was the steady-state initial SOC stocks, as C-TOOL gave a higher SOC stock estimate (120 Mg C ha^{-1}) compared with Yasso20 (57 Mg C ha^{-1}), which was likely related to the initialisation assumption in those models. The C-TOOL model scales soil C by back calculation in initialisation processes, while the Yasso model may be initialised in multiple ways, including the steady-state model run used in this study. Based on our results, it seems most reasonable to use the same model for both initialisation and further modelling. Furthermore, the use of multi-model ensembles has been suggested to balance the well-known differences and uncertainties of the SOC models (Riggers et al., 2019; Bruni et al., 2022).

A limitation regarding our mulch film comparison was that C-TOOL and Yasso20 – similarly to most SOC models – are unable to estimate the effect that mulch film use has on decomposition and, thus, on soil CO_2 emissions. In previous studies, plastic mulch film use has increased the CO_2 release from soil (Cuello et al., 2015; Fan et al., 2019; Lee et al., 2019), but these studies were conducted under very different climatic conditions and using non-degradable plastic mulch. The biodegradable mulches used in this study began degrading early, approximately 3–4 weeks after sowing (Lehtilä et al., 2023). Thus, we assume that excluding the potential effect of mulch film on the soil CO_2 outflux is not a marked source of error in our study.

4.3. Practical viewpoints and application of the results

The cultivation of forage maize in boreal conditions does not increase the environmental impact compared with perennial grasses and cereal silage. The high DM yield of maize, leading to decreased input requirements per yield unit is most likely the underlying reason. However, in Finland, the cultivation of high-quality maize is currently limited to southern areas of the country due to the limited temperature during the growing season in central and northern Finland. Therefore, forage maize can be an environmentally relevant crop alternative for southern Finland cattle farms, to supplement forage production and/or to diversify grass-dominated cultivation. Large-scale replacement of perennial grass cultivation with forage maize is not feasible, as silage grass is typically only partially supplemented with maize silage in cattle feeding. Also, the risk of interannual maize yield variation (Seppälä et al., 2012; Epie et al., 2018; Lehtilä et al., 2023) and the potential for decreased SOC stock should be accounted for by including perennial grasses in forage maize crop rotation. Alternatively, legumes may be used as cover crops for maize or in legume–maize intercropping to provide e.g. C input to soil, weed suppression, and biological N fixation (Francis et al., 1986; Miguez and Bollero, 2005; Seran and Brintha, 2010). Forage maize offers the possibility of intensifying land use required for forage production, although a reduction in field area may not be possible on Finnish cattle farms due to area requirements for manure spreading defined by herd size and nitrate and P regulation. In cases where the excess manure could be used outside the farm, forage maize could be introduced to crop rotations, which has the potential to reduce the GWP of forage production systems.

In this study, the focus was on the cultivation of forages without attention on the animal production. Nevertheless, inclusion of forage maize on dairy cattle feeding has potential to reduce enteric methane emissions (Hart et al., 2015; van Gastelen et al., 2015) and increase milk yield (Khan et al., 2015). Thus, forage maize could – at least in theory – decrease the environmental impact of milk and beef production. Nevertheless, the impact likely depends on the quality of maize silage, which is often non-optimal (Liimatainen et al., 2022; Lehtilä et al., 2023) under the cool climate conditions in the boreal region.

The use rates of cultivation inputs are most likely somewhat lower on real-life farms in Finland, as illustrated in this assessment. The reason for the relatively high input use in this study was related to the use of field experiment data for field forage yields, which are often higher compared with farm-level yields, partly due to intensive resource use in the cultivation. As the aim of our study was to compare different forage crops to one another, the field experiment data were seen as more comparable and accurate compared with farm data. However, our results should not be directly used as estimates of forage environmental impacts, but to describe the relative differences between the forage crops.

5. Conclusions

Forage maize cultivation was not associated with a higher environmental impact compared with more widely cultivated forage crops in the boreal region. However, SOC stock losses may be a risk for the continuous cultivation of forage maize contrasting to high-yielding perennial grass, especially under an increased future temperature. Regardless of crop species, producing a high yield with suitable feeding quality and optimised input use is crucial for environmentally sustainable forage production. Special attention should be placed on the inclusion of legumes in grass mixtures, to avoid excessive N fertilisation, and to reduce SOC stock losses by increased C input to the soil in the form of crop residues and organic amendments. Further research is required concerning the effect of biobased mulch films on field emissions, especially GHG and ecotoxic emissions and microplastic pollution, as well as on the effect of forage maize feeding on the environmental impact of boreal dairy production. Also, the high uncertainties of N_2O emission factors related to annual and perennial crop comparisons, SOC stock change modelling choices, and ecotoxicity-related LCA methods need further attention.

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CRediT authorship contribution statement

Lehtilä Anniina: Conceptualization, Data curation, Formal analysis, Funding acquisition, Investigation, Methodology, Visualization, Writing – original draft. **Taghizadeh-Toosi Arezoo:** Methodology, Writing – review & editing. **Sairanen Auvo:** Funding acquisition, Writing – review & editing. **Tuomisto Hanna L.:** Conceptualization, Funding acquisition, Methodology, Supervision, Writing – review & editing. **Roitto Marja:** Writing – review & editing. **Kokkonen Tuomo:** Project administration, Supervision, Writing – review & editing, Funding acquisition. **Mäkelä Pirjo S.A.:** Funding acquisition, Supervision, Writing – review & editing.

Declaration of Competing Interest

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.anifeedsci.2024.115878.

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