Science of the Total Environment 918 (2024) 170629

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Cover crop cultivation strategies in a Scandinavian context for climate change mitigation and biogas production – Insights from a life cycle perspective

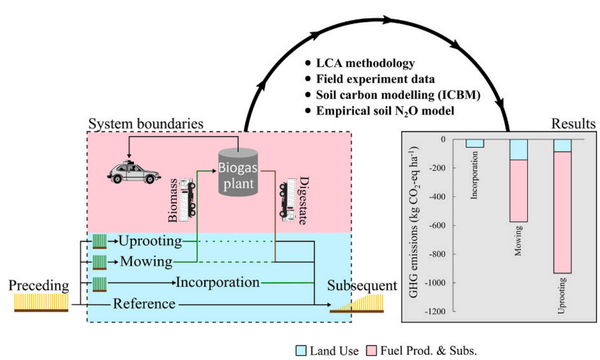
Johan Nilsson[a,*], Maria Ernfors[b], Thomas Prade[b], Per-Anders Hansson[a]

[a] Department of Energy and Technology, Swedish University of Agricultural Sciences (SLU), SE-750 07 Uppsala, Sweden
[b] Department of Biosystems and Technology, Swedish University of Agricultural Sciences (SLU), SE-234 22 Lomma, Sweden

HIGHLIGHTS

- The life cycle climate impact of cover crop cultivation was quantified.
- Trade-off between soil carbon sequestration and N_2O emissions was investigated.
- N_2O emission factor for oilseed radish cover crop was estimated.
- Harvesting the cover crop for biogas production increased the mitigation potential.
- Results showed high sensitivity to the timing of cover crop establishment.

GRAPHICAL ABSTRACT



ARTICLE INFO

Editor: Jacopo Bacenetti

Keywords:
Biomethane
Catch crop
Climate footprint
Intermediate crop
Soil N_2O emissions
Soil organic carbon sequestration

ABSTRACT

Cover crop cultivation can be a vital strategy for mitigating climate change in agriculture, by increasing soil carbon stocks and resource efficiency within the cropping system. Another mitigation option is to harvest the cover crop and use the biomass to replace greenhouse gas-intensive products, such as fossil fuels. Harvesting cover crop biomass could also reduce the risk of elevated N_2O emissions associated with cover crop cultivation under certain conditions, which would offset much of the mitigation potential. However, harvesting cover crops also reduces soil carbon sequestration potential, as biomass is removed from the field, and cultivation of cover crops requires additional field operations that generate greenhouse gas emissions. To explore these synergies and trade-offs, this study investigated the life cycle climate effect of cultivating an oilseed radish cover crop under different management strategies in southern Scandinavia. Three alternative scenarios (*Incorporation* of biomass into soil; *Mowing* and harvesting aboveground biomass; *Uprooting* and harvesting above- and belowground biomass) were compared with a reference scenario with no cover crop. Harvested biomass in the *Mowing* and *Uprooting* scenarios was assumed to be transported to a biogas plant for conversion to upgraded biogas, with the digestate returned to the field as an organic fertiliser for the subsequent crop. The climate change mitigation potential of cover crop cultivation was found to be 0.056, 0.58 and 0.93 Mg CO_2-eq ha^{-1} in the *Incorporation*, *Mowing* and *Uprooting* scenario, respectively. The *Incorporation* scenario resulted in the highest soil carbon

* Corresponding author.
E-mail address: johan.e.nilsson@slu.se (J. Nilsson).

https://doi.org/10.1016/j.scitotenv.2024.170629
Received 6 October 2023; Received in revised form 30 January 2024; Accepted 31 January 2024
Available online 4 February 2024

sequestration, but also the greatest soil N_2O emissions. Substitution of fossil diesel showed considerable mitigation potential, especially in the *Uprooting* scenario, where biogas production was highest. Sensitivity analysis revealed a strong impact of time of cover crop establishment, with earlier establishment leading to greater biomass production and thus greater mitigation potential.

1. Introduction

Strategies for removing and isolating greenhouse gases (GHGs) from the atmosphere in order to mitigate climate change are a critical component of work to meet the Paris Agreement targets (Babiker et al., 2022). Such strategies include soil organic carbon (SOC) sequestration, which is cost-efficient and has considerable mitigation potential (Minasny et al., 2017; Minx et al., 2018). One way to increase SOC stocks is to include cover crops (CC), also known as intermediate crops or catch crops, in cropping systems (Abdalla et al., 2019; Poeplau and Don, 2015). Cover crops are grown between main crops in a crop rotation, primarily to reduce nutrient leaching and improve soil quality, but can also have positive effects on biodiversity and help control weeds and pests (Blanco-Canqui et al., 2015; Constantin et al., 2010; Ellis and Barbercheck, 2015; Torstensson and Aronsson, 2000; Wilcoxen et al., 2018). According to Schipanski et al. (2014), CCs can improve the long-term resilience of cropping systems by promoting yield stability and reducing dependence on agronomic inputs, such as synthetic pesticides and fuels for tillage and cultivation. However, while some studies have shown that CCs reduce emissions of the potent GHG nitrous oxide (N_2O) (Aziz, 2022; Foltz et al., 2021), CC cultivation may also result in elevated emissions of N_2O (Guenet et al., 2021; Li et al., 2005, 2015). In a study conducted by Petersen et al. (2011), N_2O emissions more than doubled for CC cultivation compared with leaving the field bare over winter. The risk of elevated N_2O emissions is particularly high during freeze-thaw events in winter that cause nitrogen (N) and carbon (C) in CC biomass to be released into the soil in conditions favouring high denitrification levels (Olofsson and Ernfors, 2022). This can offset the climate mitigation effect achieved by increased SOC stock. Although CC cultivation is often recommended as a measure to reduce the climate impact of agricultural systems, few studies have quantified the effect, including all important emissions.

In high-latitude conditions, with a short growing season, CCs are normally left unharvested (Hansson et al., 2021). However, harvesting CCs could reduce the trade-off between SOC sequestration and N_2O emissions, since N is removed from the field before conditions become favourable for denitrification (Guenet et al., 2021) and since the contribution to long-term C storage is primarily from belowground biomass (Kätterer et al., 2011). However, the relationship between SOC sequestration and N_2O emissions in different CC cropping regimes first needs to be assessed (Launay et al., 2022). Harvesting CCs can increase resource efficiency in cropping systems by generating additional biomass, which can be used e.g. for fodder (Andersen et al., 2020), extraction of plant proteins (Muneer et al., 2021) or as substrate in biogas production (Launay et al., 2022). Biogas is a competitive form of renewable energy, while the digestate from biogas production can be used as fertiliser, so the CC system can provide further environmental benefits by replacing GHG-intensive synthetic fertilisers and fossil fuel. Improved resource efficiency can also benefit the agricultural sector by decreasing reliance on imported fuel and synthetic fertilisers, which is especially relevant in light of the current geopolitical situation in Europe (World Bank, 2022). Unlike dedicated energy crops, CCs are currently not regarded as competing with food production (Molinuevo-Salces et al., 2013; Prade et al., 2017; Styles et al., 2015). As a result, CCs are approved as biofuel substrates with potential tax exemptions in line with the European Union's renewable energy directive (RED), as long as CC cultivation "does not trigger demand for additional land" (EU, 2018).

To accurately account for the environmental impacts associated with different CC management options, it is essential to assess the entire life cycle of the system, including all relevant processes and inputs. Life cycle assessment (LCA) is a suitable comprehensive approach for such analysis, by evaluating all emissions and fluxes generated throughout the life cycle of a product or process (Cherubini and Strømman, 2011) and is a widely recognised method used by policymakers in both the public and private sectors (Brandão et al., 2022). Numerous LCA studies have been conducted on biogas systems using agricultural biomass as substrate (Hijazi et al., 2016), but only a few peer-reviewed studies have examined the life cycle climate footprint of CCs, and even fewer have investigated the synergies of combining these two systems (Launay et al., 2022).

The main aim of this study was to investigate the potential climate effect of cultivating CCs in different biomass management scenarios in a southern Scandinavian context. The model CC chosen was oilseed radish (*Raphanus sativus*), which is commonly used in Scandinavia owing to its fast growth rate, a particularly important trait in regions with a short vegetation period, and its deep roots that efficiently scavenge the soil for N and can help alleviate the effects of soil compaction (Norberg and Aronsson, 2020; Williams and Weil, 2004). However, due to its frost sensitivity, oilseed radish may be more susceptible to spikes in N_2O emissions during freeze-thaw events than other frost-tolerant CCs (Dörsch, 2000; Li et al., 2015; Olofsson and Ernfors, 2022). The biomass management scenarios compared were: (1) harvesting aboveground CC biomass for use in biogas production, by cutting; (2) harvesting aboveground and belowground CC biomass for use in biogas production, by pulling the plants out of the soil; (3) leaving the CC unharvested; and (4) no CC (reference scenario). The analyses specifically considered the relationship between SOC sequestration and soil-borne N_2O emissions and aimed to identify critical processes within the system making important contributions to the GHG balance.

2. Materials and methods

The study consisted of two parts. First, a field experiment was conducted in which oilseed radish CCs were established at different times (early to late). A systems study was then conducted to assess the climate footprints of different management strategies in CC cultivation, with the analysis based on biomass growth and N content data obtained from the field experiment and from the literature.

2.1. Field experiment

Field experiments were conducted in 2018 and 2019 to obtain data on biomass growth and N content of cultivated oilseed radish for use in further assessments. A short summary can be found in Supplementary Material (SM) to this paper and full details in Hansson et al. (2021) and Prade et al. (2022). The oilseed radish was grown without fertiliser as a sole CC, i.e. it was not mixed with other CC species. It was established after harvest of the preceding crop and terminated later in the autumn (Table S1 in SM). The plots in the field experiments were randomised and repeated in three blocks. Different dates for establishing the CC were investigated (early July (*Early*), late July (*Medium*), late August (*Late*)), with the CC sown at a seed rate of 15 kg ha^{-1} in all cases. The impact of establishment date was further explored in sensitivity analysis (see Section 2.4.1).

Aboveground biomass was hand-harvested in an area of 0.25 m^2 (4 rows 50 cm in length, 12.5 cm row spacing), leaving 10 cm of stubble. The biomass was dried at 65 °C for around 48 h to constant weight. Biomass yield was determined as amount of dry matter (DM, t ha^{-1}). To investigate the relationship between aboveground and belowground biomass, and that between stubble and harvested biomass, at 10 cm stubble height, 5–10 plants were dug up in each plot and separated into three fractions; roots (belowground), stubble (0–10 cm above ground) and harvested biomass (>10 cm above ground). To obtain the belowground fraction, all roots were extracted from the 0–20 cm soil layer. The field experiments were performed on sandy soil and therefore, even relatively fine roots were recovered. However, very fine roots were lost in the procedure, resulting in relatively conservative estimates of C contributions from the belowground fraction. The biomass was dried at 65 °C for around 48 h to constant weight. Representative subsamples of 10–20 g from both the harvest fraction and the root fraction were milled in an IKA knife grinder. Depending on expected N content, 3–8 mg ± 0.50 mg of plant material were used for C/N analysis.

The total content of C and N in the oilseed radish harvest fraction and root fraction was analysed using an elemental analyser (Flash 2000, Thermo Scientific) with external standards acetanilide (*N*-phenylacetamide) and known reference samples for quantification, see SM for details. The C concentration obtained was used to correct for contamination of the plant material with soil particles, by adjusting the weight of the plant material to a reference C content of 42.5 % (Ma et al., 2018). The N content was corrected similarly with the ratio between measured C and reference C content.

2.2. Systems study

The system boundary for the analysis was set from harvest of the preceding crop to fertiliser application in the subsequent crop (Fig. 1). In the climate footprint assessment, a *Reference* scenario, in which the field was assumed to be left bare during winter, was compared with three alternative cover cropping scenarios:

Incorporation: The CC was left in the field during winter and ploughed under and incorporated into the soil in spring.

Mowing: Aboveground CC biomass was harvested in the autumn, by mowing.

Uprooting: Both aboveground and parts of belowground CC biomass were harvested in the autumn.

In the *Mowing* and *Uprooting* scenarios, the harvested biomass was assumed to be transported to a biogas plant, where it was converted into upgraded biomethane that was used to replace fossil diesel as vehicle fuel. Identical processes in the *Reference* and alternative scenarios, such as seed production, sowing of main crops and ploughing after CC, were omitted from the calculations.

The CC system was divided into two subsystems, *Land Use* and *Fuel Production and Substitution* (Fig. 1). Net emissions (*E*) leaving the system boundary were calculated as the difference between each alternative scenario and the *Reference* scenario:

$$E_{Tot\,i} = \overbrace{E_{LU^{si}} - E_{LU^{R}}}^{E_{\Delta LU\,i}} + \overbrace{E_{FP^{si}} - E_{Sub}}^{E_{\Delta FP\,i}} \tag{1}$$

where $E_{Tot\,i}$ is total net GHG emissions in alternative scenario *i*, $E_{LU^{si}}$ is emissions from the *Land Use* subsystem in alternative scenario i, $E_{LU^{R}}$ is emissions from the *Reference* scenario in the *Land Use* subsystem, $E_{\Delta LU\,i}$ is net emissions from the *Land Use* subsystem for alternative scenario *i* (i.e. the difference between the alternative scenario and the *Reference* scenario), $E_{FP^{si}}$ is emissions from production and use of biomethane in alternative scenario *i*, E_{Sub} is emissions from production and use of the fossil diesel replaced by biomethane produced in system *i*, and $E_{\Delta FP\,i}$ is net emissions from the subsystem *Fuel Production and Substitution*.

2.2.1. Land use subsystem

The *Land Use* subsystem comprised emissions from agricultural field operations and soil processes affecting the GHG balance, such as soil C and N_2O fluxes. In all alternative scenarios, the CC was assumed to be sown immediately after ploughing, following the preceding crop. In the *Incorporation* scenario, the CC was left standing until spring, while in the *Mowing* and *Uprooting* scenarios it was harvested by mowing and uprooting, respectively, in late autumn. A machine for uprooting similar to that used to harvest carrots and beets with an energy use of 935 MJ ha^{-1} was assumed, based on data from Maskinkalkylgruppen (2023). Its capacity for uprooting the cover crop, in terms of ha per hour, was assumed to be 20 % higher than in harvesting of beets, given the considerably higher yields in beet cultivation per ha. The harvested biomass in the *Mowing* and *Uprooting* scenarios was assumed to be transported by truck to a biogas plant located 50 km away from the field, where the biomass was digested to produce upgraded biogas as a

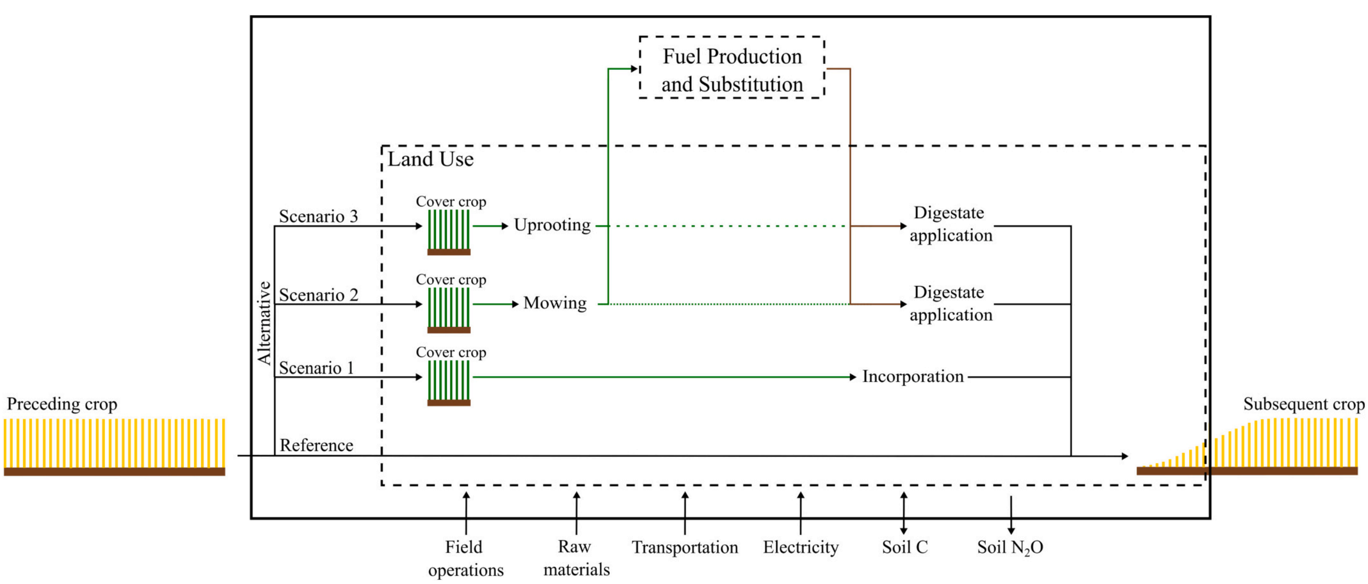


Fig. 1. Overview of the scenarios evaluated in this study. The *Reference* scenario involved leaving the field bare between two main crops in the crop rotation, while the three alternative scenarios involved cover cropping (oilseed radish). In *Incorporation* (scenario 1), the cover crop was left in the field during winter and ploughed under and incorporated into the soil in spring. In *Mowing* (scenario 2), aboveground biomass was harvested by mowing in autumn and the unharvested biomass was ploughed under during spring. In *Uprooting* (scenario 3, both aboveground and belowground biomass were harvested in autumn. Harvested biomass from the *Mowing* and *Uprooting* scenarios was assumed to be converted into upgraded biomethane and used to replace production and use of fossil diesel fuel.

substitute for fossil diesel in the transport sector. In all scenarios, the remaining CC biomass was assumed to be ploughed under and incorporated into the soil, followed by harrowing and sowing of the subsequent crop. In the *Mowing* and *Uprooting* scenarios, digestate from the biogas plant was assumed to be transported to the field (50 km) and applied to the subsequent crop using a slurry spreader. Data from Ecoinvent were used to assess GHG emissions from field operations, transportation and synthetic fertiliser (Table S2 in SM).

2.2.1.1. Soil organic carbon sequestration potential. The SOC sequestration potential of the scenarios was calculated using the Introductory Carbon Balance Model (ICBM) developed by Andrén and Kätterer (1997). This model divides the soil into two C pools, young and old. Carbon inputs, here in terms of aboveground and belowground CC residues, and digestate initially enter the young pool. From there, C is either transferred to the old pool or returned to the atmosphere as CO_2 through oxidation. Further oxidation occurs in the old pool, but at a slower rate. Transfer of C from the young to the old pool is determined by a humification coefficient (h), the value of which varies depending on the source of C input. In this study, we used humification coefficient values from Bolinder et al. (2018) of 0.155 and 0.395 for aboveground and belowground CC residues, respectively. Furthermore, we assumed an additional belowground carbon input in the form of exudates, root hairs and fine roots, which were not included in the sampled root material due to sampling limitations. Bolinder et al. (2007) estimated that this extra root C input constitutes approximately 65 % of the sampled belowground C. To our knowledge, no peer-reviewed humification coefficient has been established for digestate. Instead, we used the humification coefficient designated for farmyard manure, with a value of 0.266, as an approximate value. Retention time of C in each pool is determined by pool-specific first-order reaction coefficients (k_y and k_o), which were set to 0.756 and 0.005 year^{-1}, respectively (Bolinder et al., 2018). The degradation rate can be modified using parameter r_e, which describes the impact of external factors such as soil temperature and water-holding capacity (Andrén et al., 2004). In this study, a value of 1.05 was assigned to r_e in the base case, which corresponds to conditions in south-central parts of Sweden (Karlsson, 2012). The contribution of SOC input to the young pool in each scenario was calculated as:

$$Y_i(t) = \left(Y_{i_{t-1}}\right) \times e^{-k_y \times r_e} \quad (2)$$

where $Y_i(t)$ is amount of C from input source i remaining in the young pool at year t and $Y_{i_{t-1}}$ is amount of SOC in the previous time step, with the condition that $Y_i(0)$ is amount of C from input source i generated in the CC cultivation system.

The contribution of SOC to the old pool at each time step in each scenario was calculated as:

$$O_i(t) = \left(O_{i_{t-1}} - \left(\frac{h_i \times k_y}{k_o - k_y} \times Y_{i_{t-1}}\right)\right) \times e^{-k_0 \times r_e} + \frac{h_i \times k_y}{k_o - k_y} \times Y_i(t) \quad (3)$$

where $O_i(t)$ is the amount of C from input source i remaining in the old pool at year t, $O_{i_{t-1}}$ is the amount of C in the old pool in the previous time step, and $O_i(0)$ was set to 0. The potential contribution to SOC was defined as potential SOC sequestration in each scenario relative to the *Reference* scenario (which did not involve cultivation of CC, so its SOC contribution was assumed to be zero).

2.2.1.2. Soil nitrogen fluxes. Direct soil N_2O emissions induced by N inputs were calculated according to the method provided in the IPCC guidelines for national GHG inventories, i.e. the amount of N (total N) was multiplied by an emission factor (IPCC, 2019). The N_2O emissions generated by CC cultivation and digestate application were estimated separately. Direct N_2O emissions from CC cultivation ($N_2O_{Direct;CC}$) were calculated as:

$$N_2O_{Direct;CC} - N = \left[(AGB \times N_{AG}) + (BGB \times N_{BG})\right] \times EF_{1;CC} \quad (4)$$

where AGB is aboveground biomass (kg DM ha^{-1}), N_{AG} is N content in aboveground biomass (kg N kg^{-1} DM), BGB is belowground biomass (kg DM ha^{-1}), N_{BG} is N content (kg N kg^{-1} DM) and $EF_{1;CC}$ is a CC-specific emission factor representing kg N_2O-N kg^{-1} N in biomass residues.

Direct N_2O emissions from digestate application ($N_2O_{Direct;Dig}$) were calculated as:

$$N_2O_{Direct;Dig} - N = N_{Dig} \times EF_{1;Dig} \quad (5)$$

where N_{Dig} is N content in digestate (kg N ha^{-1}) and $EF_{1;Dig}$ is a digestate-specific emission factor (kg N_2O-N kg^{-1} N).

The emission factor for direct N_2O emissions induced by the CC was derived from reported N_2O emissions from unfertilised oilseed radish CCs under similar (southern Scandinavian) conditions. It was calculated using weighted linear regression analysis with N content in CC biomass as the independent variable and soil N_2O emissions as the dependent variable, with the intercept constrained to pass through the origin (Fig. 2). Weighting was applied to compensate for heteroscedasticity and was based on the variance of the residuals (Astivia and Zumbo, 2019). This resulted in a slope coefficient equal to 0.0153, which was used as the value of $EF_{1;CC}$ (Fig. 2). In cases where information on the N content in belowground biomass was unavailable, a belowground-to-aboveground biomass ratio of 0.18 and an estimated N content of 2 % in belowground biomass were used to approximate the N content, based on data obtained in field experiments (see Section 2.1). The emission factor used to estimate digestate-induced direct N_2O emissions was taken from a review study by Launay et al. (2022), who found the mean of published values for digestate to be 0.0052 (range 0.019–0.0008).

Digestate-induced indirect N_2O emissions were calculated based on the IPCC guidelines for national GHG inventories (IPCC, 2019). Eq. (6) was used for indirect N_2O emissions from volatilised N and Eq. (7) was used for emissions from leached N:

$$N_2O_{Indirect} - N = N_{Dig} \times Frac_{GAS} \times EF_4 \quad (6)$$

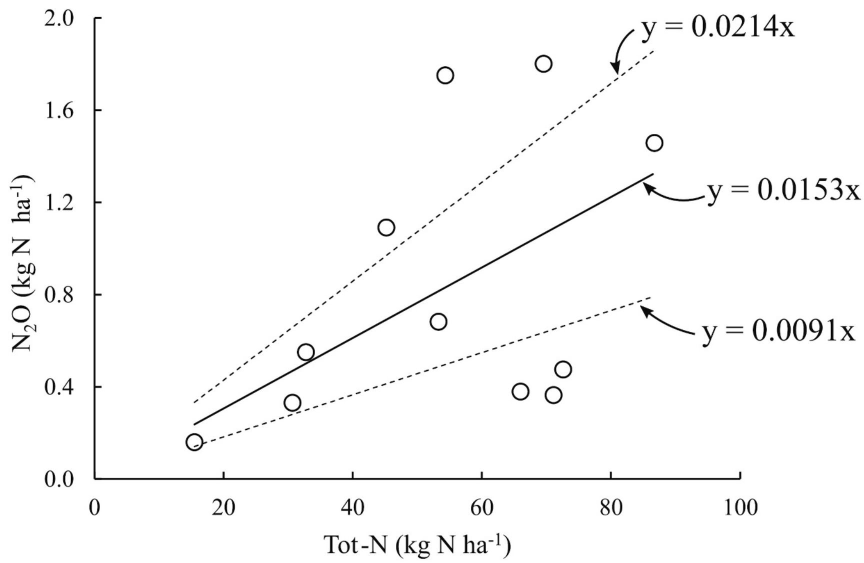
$$N_2O_{Indirect} - N = N_{Dig} \times Frac_{Leach-(H)} \times EF_5 \quad (7)$$

where $Frac_{GAS}$ and $Frac_{Leach-(H)}$ represents the fraction of the N input lost from the soil via volatilisation and leaching, respectively.

The value of $Frac_{GAS}$ and $Frac_{Leach-(H)}$ was set to 0.167 and 0.1396, respectively, based on average activity data for digestate presented by SEPA (2022). A default value of 0.014 and 0.011 was applied for EF_4 and EF_5, respectively (IPCC, 2019).

Cover crop cultivation was assumed to reduce N leaching, and associated indirect N_2O emissions, compared with the *Reference* scenario. The N leaching reduction from growing a cover crop was estimated using the model from Aronsson and Torstensson (2004), implemented in the farm management tool VERA developed by the Swedish Board of Agriculture. In this approach, the N leaching reduction is quantified by multiplying N uptake in the CC by a soil type-specific leaching factor. In the present case, a clay content of 15–25 % was assumed in the default scenario. As earlier studies have reported considerable N losses (20–50 %) from oil crop biomass during winter (Aronsson and Torstensson, 2004), it was assumed that any aboveground biomass left in the field during winter lost 35 % of its N content.

Soil N volatilisation in the form of ammonia (NH_3) and N oxides (NO_x) induced by CC cultivation was assumed to correspond to 6 % of the N in aboveground biomass left in the field over winter (Ruijter et al., 2010; Janzen and McGinn, 1991). Nitrogen volatilisation in the form of NH_3 from the spreading of digestate was assumed to correspond to 15 % of ammonium-N (NH_4-N) in the digestate (Quakernack et al., 2012; Tidåker et al., 2016b). Nitrogen gas (N_2) emissions were estimated based on the calculated N_2O emissions and an N_2O:(N_2O + N_2) ratio of 0.15 (Butterbach-Bahl et al., 2013). Enhanced N availability, quantified as N fertiliser replacement value (NRFV), was modelled as a 15 % increase in mineralised N in digested crop residues compared with untreated



Reference	Tot N in biomass (kg N ha^{-1})	N_2O emissions (kg N ha^{-1})	Measuring period (number of days)	Treatment
Taghizadeh-Toosi et al. 2022*[D]	33	0.550	253	Ploughing
Taghizadeh-Toosi et al. 2022*[D]	31	0.330	253	Direct seeding
Li et al. 2015[D]	66	0.378	366	
Li et al. 2015[D]	45	1.090	366	CC harvested
Olofsson and Ernfors, 2022*[S]	70	1.800	43	
Petersen et al. 2011*[D]	87	1.457	240	Conventional ploughing
Petersen et al. 2011*[D]	71	0.363	240	Direct seeding
Petersen et al. 2011*[D]	53	0.681	240	Reduced ploughing
Lövgren 2022[S]	73	0.474	78	
Lövgren 2022[S]	16	0.159	78	CC harvested
Aziz 2022[N]	54	1.750	356	

* Nitrogen in belowground biomass was not measured.

D = Denmark, S = Sweden and N = Norway

Fig. 2. (Upper panel) Linear regression plot employed to derive a specific emission factor for direct soil N_2O emissions from oilseed radish cultivation as a cover crop in southern Scandinavia, where the solid black line indicates the slope and the dashed lines represent 95 % confidence interval. (Lower panel) Summary of literature data used in calculations. The data on N_2O emissions represent the difference between CC cultivation and no CC.

residues (Notaris et al., 2018). To assess accumulation of N stored within soil organic matter (SOM) attributable to the increased soil C stock, a C/N ratio of 12 in SOM was used (Batjes, 1996), meaning that for every kg of C stored in the soil, the equivalent of 83 g N was also sequestered. The difference in N balance ($\Delta N_{Balance^i}$) was calculated as:

$$\Delta N_{Balance^i}\left(\frac{kgN}{ha}\right) = \Delta N_{leach^i} + \Delta N_{gas^i} + \Delta N_{SOM^i} - \left(N_{loss\ biogas^{S_i}}\right) + \left(N_{NRFV^{S_i}}\right) \tag{9}$$

where ΔN_{leach^i} is the difference in leached N (kg ha^{-1}) from CC cultivation between the *Reference* scenario and the alternative scenario *i*, ΔN_{gas^i} is the difference in gaseous N emissions (N_2O, NH_3, NOx and N_2, kg N ha^{-1}) from CC cultivation between the *Reference* scenario and alternative scenario *i*, ΔN_{SOM^i} is stored N in SOM as a consequence of the increase in soil C stock, $N_{loss\ biogas^{S_i}}$ is N lost in the harvesting and biogas production life cycle stages (kg N ha^{-1}) and N_{NRFV} is the increased fertiliser value of the returned digestate.

Differences in N balance (Table S3 in SM) between the alternative scenarios and the *Reference* scenario were included in the climate footprint assessment by considering the impact of production and use of the corresponding amount of synthetic N fertiliser.

2.2.2. Fuel production and substitution subsystem

In the *Mowing* and *Uprooting* scenarios, the harvested biomass was assumed to be transported by truck to a biogas plant located 50 km away from the field where the CC was grown (the data used to estimate the GHG fluxes associated with this subsystem are presented in Table S4 and Table S5 in SM). The biomass was assumed to be digested in a large-scale co-digestion plant with capacity >10 GWh upgraded biomethane per year. Only the CC biomass flows were considered in this assessment, i.e. the other substrates included in co-digestion were excluded. Specific methane production of the CC was based on results from earlier studies on biogas production using oilseed radish biomass as substrate (Molinuevo-Salces et al., 2013). Biogas consisting of 55 % CH_4 and 45 % CO_2 was assumed to be produced from the CC, based on results from grass as substrate (Edström et al., 2008). Electricity consumption in the biogas process was set at 36 MJ Mg^{-1} DM of substrate (Börjesson et al., 2016). Nordic consumption electricity mix was adopted, with a climate impact of 57.7 g CO_2-eq kWh^{-1} (Table S2 in SM). Methane losses during the digestion process were assumed to represent 0.3 % of total CH_4 production and 5 % of CH_4 produced was assumed to be flared during the digestion process, whereof 5 % was incompletely combusted and lost to the atmosphere as CH_4. After anaerobic digestion, the biogas was assumed to be upgraded to biomethane using water-scrubber technology with electricity consumption of 0.9 MJ m^{-3} biogas (Börjesson et al., 2016) and the biomethane was compressed to 200 bar to facilitate transportation, which was assumed to require 0.025 MJ electricity MJ^{-1} biomethane (Björnsson et al., 2013). Methane losses during the upgrading process were set to 0.5 %. The upgraded gas was assumed to have a CH_4 concentration of 97 %. Heat demand in the anaerobic digestion process was 0.126 MJ kg^{-1} substrate and was assumed to be met using some of the biogas produced.

To estimate the CH_4 losses from storage of the digestate, we used an existing estimate for medium to large-sized biogas plants (Styles et al., 2016) that 1.5 % of potential CH_4 production from digestate is lost to the atmosphere. Potential methane production from the digestate, i.e. residual CH_4 potential, and the volatile solids (VS) and DM content of the digestate were based on results from Björnsson et al. (2016). Nitrogen losses in the form of N_2O were assumed to be 1 % of total NH_3 emissions from storage and NH_3 losses were assumed to correspond to 10 % of NH_4 concentration, which in turn corresponded to 60 % of total N content in the digestate (Styles et al., 2016; Tidåker et al., 2016a). The amount of digestate was assumed to follow the same mass balance approach as used in Nilsson et al. (2020), where outputs from DM substrate input to the biogas reactor were calculated during biogas production and during storage of digestate.

The *Fuel Production and Substitution* subsystem included replacement of fossil diesel with biomethane. In calculations, higher engine efficiency in the diesel engine was assumed, resulting in a substitution factor of 0.86, meaning that 1 MJ biomethane replaced 0.86 MJ diesel. The climate impact of the substituted diesel was set to 81 g CO_2-eq. MJ^{-1}, based on Gode et al. (2011). The digestate produced was assumed to be transported back to the field using the same type of truck as used for biomass transportation. The DM content in transported digestate was assumed to be 7.8 % (Björnsson et al., 2016).

2.3. Climate impact assessment

In climate footprint assessments, Global Warming Potential (GWP) is the default metric for quantifying impact (Cherubini et al., 2016). The emission factors applied in this study were 29.8 for fossil CH_4, 27.0 for non-fossil CH_4 and 273 for N_2O over a 100-year perspective (GWP_{100}) (Forster et al., 2021). Emissions in the form of biogenic CO_2 were considered not to cause increased radiative forcing over a 100-year perspective. However, negative emissions from SOC sequestration were considered by including the C remaining in the soil after 100 years. The impact of the chosen time horizon was further assessed in sensitivity analysis.

2.4. Sensitivity analysis

To evaluate the impact of the methodology and system assumptions on the results, the influence of changing time of establishment of the CC and the value of critical parameters identified in the climate footprint assessment was investigated. These parameters were varied based on literature data to develop high- and low-GHG emissions cases, which were compared with the base case described above.

2.4.1. Timing of cover crop establishment

Sensitivity of the results to the time of CC establishment was assessed using data on biomass growth (above- and belowground) and crop N content following *Early*, *Medium* and *Late* establishment (Table S1 in SM), where *Early* establishment referred to early July, *Medium* to late July and *Late* to late August. The data were obtained from the field experiments on unfertilised oilseed radish CCs in southern Sweden in 2018 and 2019 (see Section 2.1), with *Medium* establishment in 2019 being used as the base case.

2.4.2. Impact of critical parameters

For the sensitivity analysis, parameters known to be associated with relatively high uncertainty and identified as having a significant impact on the results were categorised into five distinct groups: i) Transport of biomass and digestate, ii) direct soil N_2O emissions, iii) reduced leaching from CC cultivation, iv) SOC sequestration potential, and v) CH_4 losses during biomass conversion at the biogas plant (Table S2 in SM). A literature review yielded a range of values for these parameters, where the most optimistic and pessimistic values were compared with the base case in sensitivity analysis. The impact of critical assumptions on the results was also analysed.

3. Results

3.1. Cover crop biomass growth

Biomass growth of the oilseed radish CC showed wide variation depending on the time of crop establishment (Fig. 3). Notably, the longer growth period following *Early* establishment resulted in considerably greater biomass growth compared with *Medium* and *Late* establishment. In 2018, biomass growth following *Late* establishment was insufficient for harvest (depicted as no biomass growth in Fig. 3). These findings emphasise the critical role of time of establishment in CC cultivation. Biomass growth following *Medium* establishment in 2019 (aboveground biomass 1.4 Mg ha^{-1} and belowground biomass 0.5 Mg ha^{-1}, which in total corresponded to 0.9 Mg C ha^{-1}) was selected as the base case to which other results were compared. However, the importance of time of CC establishment was further evaluated in sensitivity analysis (Section 3.5).

3.2. Carbon fluxes and soil organic carbon sequestration potential

The C fluxes associated with each scenario are presented in Fig. 4. The photosynthetic activity of the oilseed radish crop led to C entering the system boundary from the atmosphere and accumulating in above- and belowground biomass and via rhizodeposition. In the *Incorporation* scenario, all photosynthetically fixed C was transferred to the soil (Fig. 4a), resulting in the largest soil C input and thereby the highest SOC sequestration potential (Fig. 4d), with most of the contribution to SOC sequestration originating from the belowground C input. In the *Mowing* and *Uprooting* scenarios, C was removed from the field through crop harvesting and transformed into biogas and digestate. Some C was lost from the product system via biomass losses during transport and handling at the biogas plant and via C emissions during storage of the digestate (Fig. 4b, c). This resulted in lower soil C input and lower sequestration potential (Fig. 4d). Returning the digestate to the field proved to be important in reducing the difference in SOC sequestration potential between the scenarios. Estimated SOC sequestration potential after 100 years was used in the climate footprint assessment to quantify the potential climate effect of the different scenarios. The SOC sequestration potential shown in Fig. 4d is relative potential compared with the *Reference* scenario, rather than an absolute measure.

3.3. Nitrous oxide emissions

Soil N_2O emissions were estimated and classified into CC- and digestate-induced emissions, and N_2O emissions resulting from changes in the demand for synthetic N fertiliser based on differences in N balance between the *Reference* and alternative scenarios (Fig. 5). The highest direct N_2O emissions induced by CC cultivation occurred in the *Incorporation* scenario. Direct N_2O emissions from CC cultivation were lower in the other scenarios, where part or most of the CC biomass was removed from the field. Cover crop cultivation led to reduced N leaching, resulting in decreased indirect N_2O emissions compared with the *Reference*. The reduction in indirect N_2O emissions was lower when the CC was not harvested, due to assumed N losses in standing CC biomass during freeze-thaw events in winter.

The N_2O emissions from digestate were greatest in the *Uprooting* scenario, where a larger quantity of digestate was returned to the soil (Fig. 5b). In this case, indirect emissions through leaching and volatilisation contributed more to total N_2O emissions. The *Incorporation* and *Mowing* scenarios both resulted in an N deficit compared with the *Reference*, mainly due to sequestration of N in SOM as a consequence of

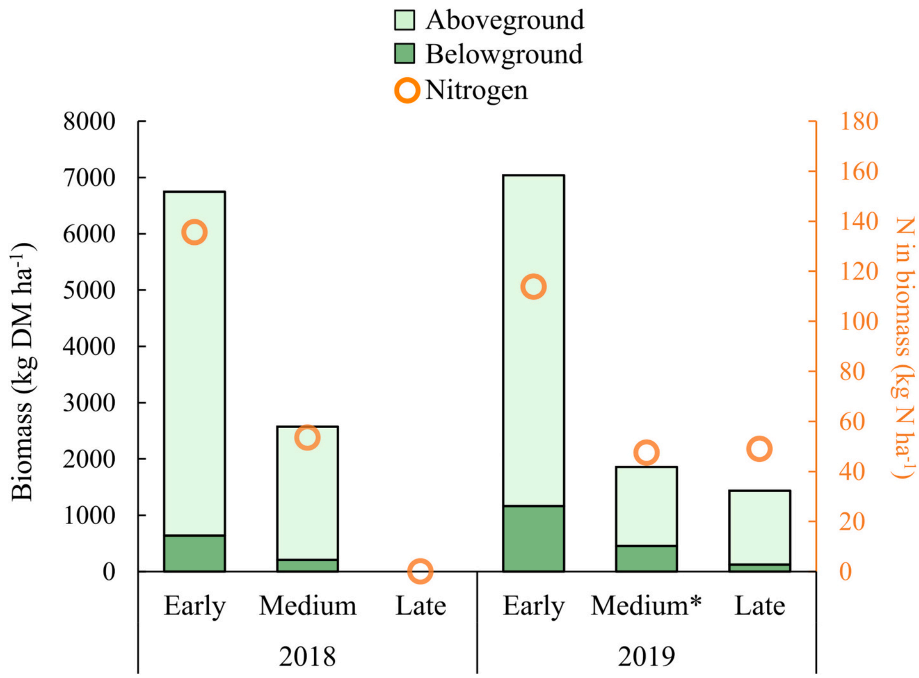


Fig. 3. Growth of aboveground (light green) and belowground (dark green) biomass following *Early*, *Medium* and *Late* establishment of the oilseed radish cover crop in 2020 and 2019 (left y-axis) and total nitrogen (N) content (orange rings) in cover crop biomass (right y-axis). *Indicates data used in the base case. The standard deviation of the total biomass growth was 560 and 160 kg DM ha^{-1} for Early and Medium, respectively, in 2018. In 2019, the value was 170, 10 and 30 kg DM ha^{-1} for Early, Medium and Late, respectively.

SOC sequestration. In contrast, the *Uprooting* scenario resulted in a small positive N balance compared with the *Reference*, due to emission reductions (Fig. 5c).

Overall, the *Incorporation* scenario led to the highest soil N_2O emissions and the *Uprooting* scenario the lowest. However, large variations in the literature data on N_2O emissions from oilseed radish CCs and from digestate application resulted in wide variation in the results obtained. The effects of these variations on the system-level climate impact were further investigated in a scenario analysis.

3.4. System greenhouse gas emissions

Cultivation of oilseed radish as a CC resulted in a modest GHG emissions reduction of 0.056 Mg CO_2-eq ha^{-1}. The mitigating effect was substantially greater when the CC was harvested, with reductions of 0.58 and 0.93 Mg CO_2-eq ha^{-1} in the *Mowing* and *Uprooting* scenario, respectively (Fig. 6c).

In the *Land Use* subsystem, the SOC sequestration potential resulted in a climate change-mitigating effect in all alternative scenarios, but this effect was counteracted by elevated emissions from soil N_2O emissions and field operations (Fig. 6a). The highest GHG emissions from field operations were in *Uprooting*, due to the more energy-intensive harvesting method assumed in this scenario. The *Incorporation* and *Mowing* scenarios both led to increased demand for synthetic N fertiliser, which increased GHG emissions, although this effect was low in the *Mowing* scenario. In contrast, the *Uprooting* scenario resulted in lower demand for synthetic N fertiliser, resulting in reduced GHG emissions compared with the *Reference*.

In the *Fuel Production and Substitution* subsystem (Fig. 6b), transportation was the largest contributor to GHG emissions, through transport of harvested biomass from the field to the biogas plant and of digestate from the biogas plant to the field. In addition, CH_4 emissions from losses during anaerobic digestion, upgrading and storage of the digestate made a large contribution to total GHG emissions in this subsystem. However, the largest contributor to GHG balance was substitution of fossil diesel fuel, with large mitigation potential from the 9 and 18 GJ upgraded biogas produced per ha in the *Mowing* and *Uprooting* scenario, respectively. The largest overall mitigation potential was found in the *Uprooting* scenario, where more of the biomass was harvested and consequently more biomethane was produced (Fig. 6c).

3.5. Sensitivity analysis

3.5.1. Time of establishment

Earlier establishment of the CC resulted in greater biomass production (Fig. 3), which in turn led to greater potential for SOC sequestration, lower leaching and greater potential for fossil fuel substitution. With early establishment, the *Mowing* and *Uprooting* scenarios gave emissions reductions of 3.0 and 3.5 Mg CO_2-eq ha^{-1}, respectively, based on field data from 2018, and 3.0 and 3.7 Mg CO_2-eq ha^{-1}, respectively, based on data from and 2019 (Fig. 7). In the *Incorporation* scenario, the GHG balance ranged from −0.45 to 0.12 Mg CO_2-eq ha^{-1} depending on time of CC establishment. Late establishment in 2018 led to insufficient biomass growth for harvest, so the elevated GHG emissions in Fig. 7 originate from sowing the CC.

3.5.2. Parameter sensitivity

The results displayed high sensitivity to the values of emission factors used for estimating direct soil N_2O emissions (Fig. 8). Specifically, results for the *Incorporation* scenario showed high sensitivity to the value of the emission factor used to estimate N_2O emissions from CC cultivation, the pessimistic and optimistic values of which led to a 265 % increase and 265 % decrease in climate change mitigation potential of the system, respectively. Results for *Mowing* and *Uprooting* scenarios exhibited considerably lower sensitivity to the CC-induced N_2O emission factor. Moreover, the results obtained for the *Incorporation* scenario demonstrated high sensitivity to the value of parameters used for estimating the reduction in N leaching and the selected time horizon for soil C sequestration, while the *Mowing* and *Uprooting* results showed lower parameter sensitivity in general. The results for these two scenarios were most sensitive to the values of the N_2O emission factor for digestate and for CH_4 losses during anaerobic digestion.

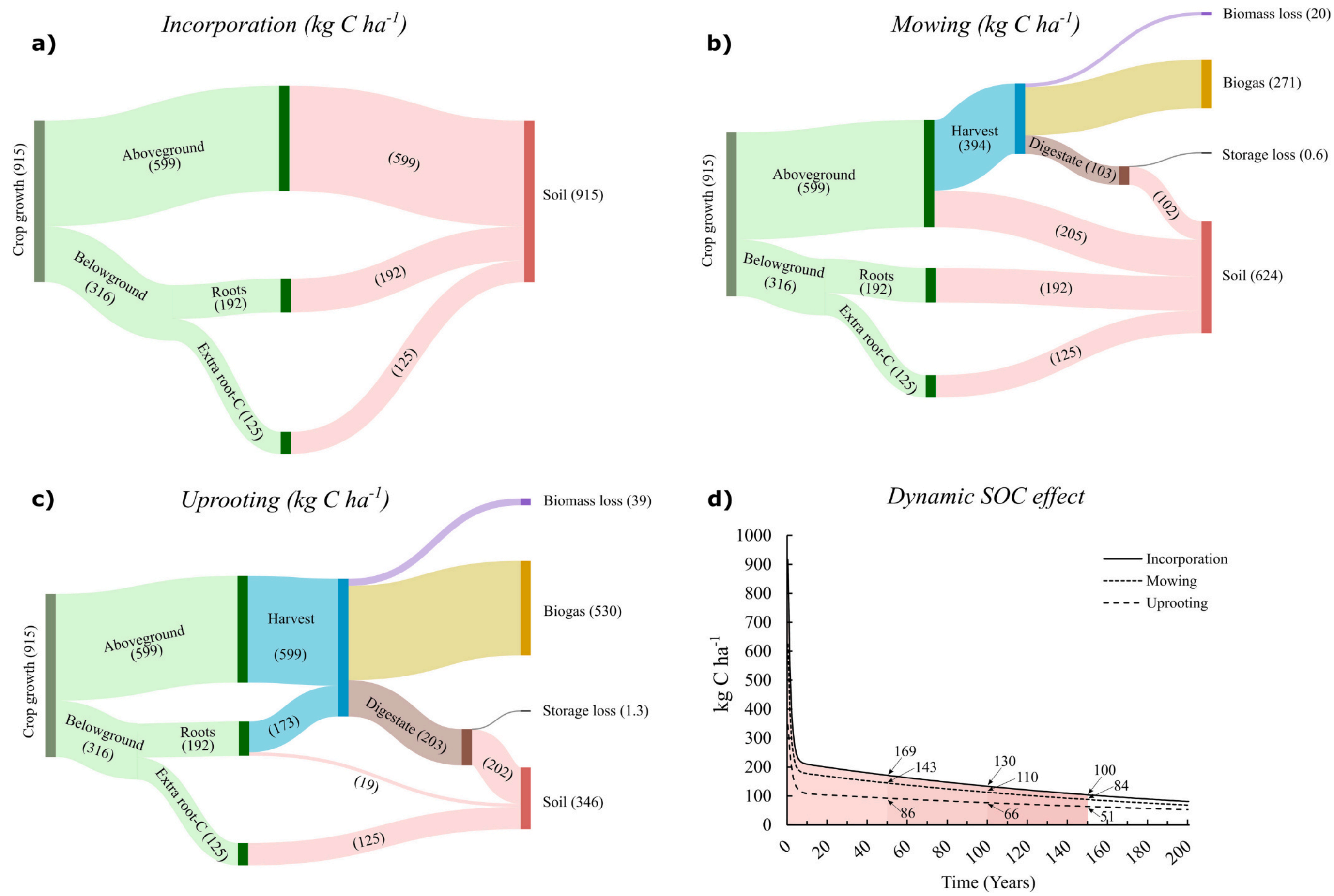


Fig. 4. Illustration of carbon balance and soil organic carbon (SOC) sequestration potential in the different cover crop scenarios. (a–c) Sankey charts of carbon flux (kg C ha^{-1}) within the system for the (a) *Incorporation*, (b) *Mowing* and (c) *Uprooting* scenarios, where each chart begins with (left) photosynthetic carbon from cover crop growth and ends (right) with either carbon outputs from the product system (in terms of biomass losses, emission losses during storage and the biogas product) or soil carbon input. (d) Changes in SOC effect over time in the three scenarios, indicated as residual soil carbon after 50, 100 and 150 years, where the soil carbon input at $t = 0$ was 915, 624 and 346 kg C ha^{-1} for the *Incorporation*, *Mowing* and *Uprooting* scenario, respectively. The Sankey charts were generated using the online tool provided by www.sankeymatic.com.

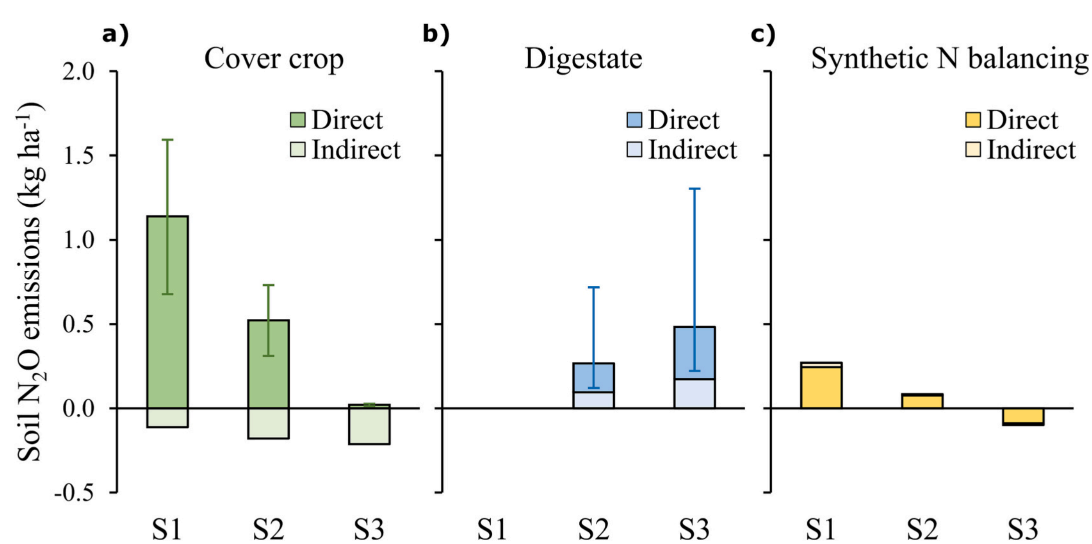


Fig. 5. Nitrous oxide (N_2O) emissions in the *Incorporation* (S1), *Mowing* (S2) and *Uprooting* (S3) scenarios compared with the *Reference* scenario without cover crop cultivation, divided into direct and indirect emissions induced by (a) cover crop cultivation, (b) application of digestate and (c) synthetic N fertiliser application to compensate for differences in N balance between scenarios S1-S3 and the *Reference*. The error bars in (a) represent ±95 % confidence interval in linear regressions analysis, while the error bars in (b) represent emissions when using maximum and minimum values of emissions factors in the literature.

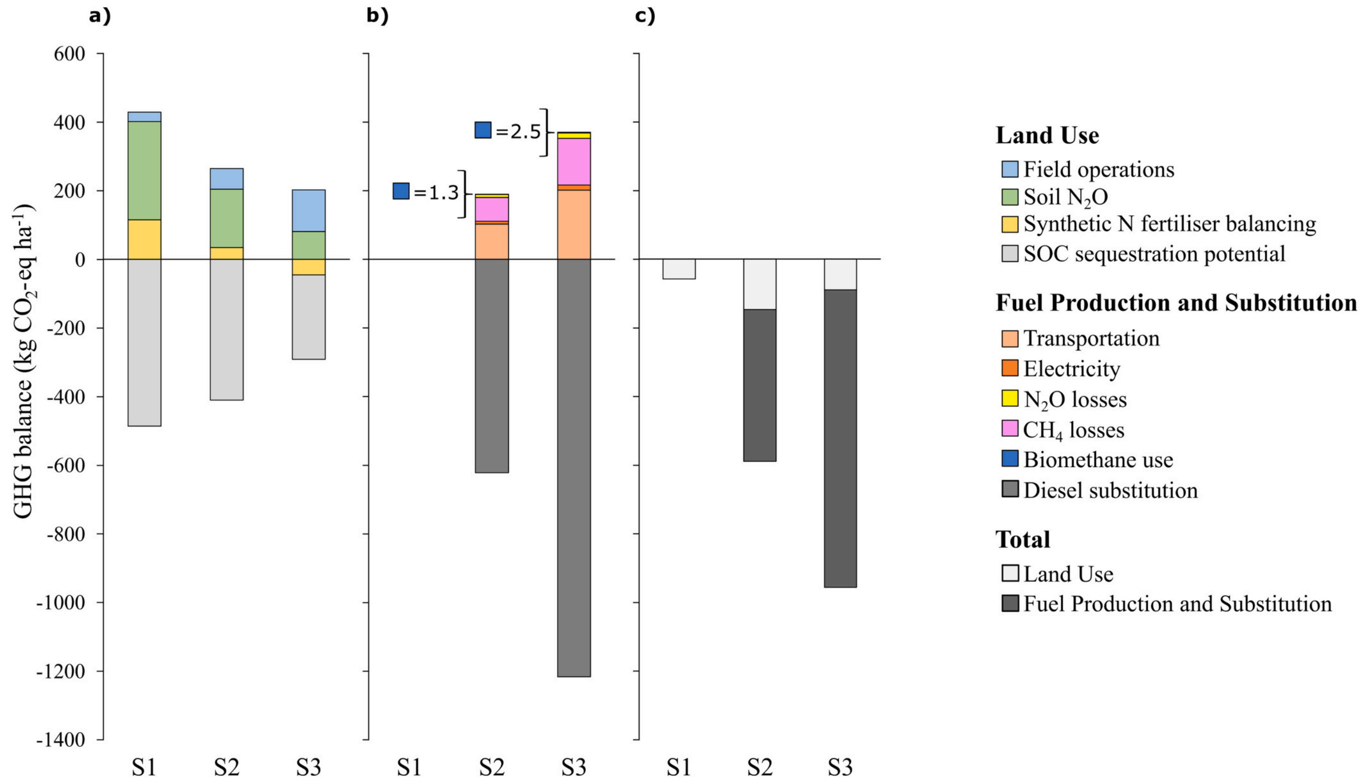


Fig. 6. Greenhouse gas (GHG) balance (kg CO_2-eq ha^{-1}) in (a) the land use and (b) fuel production and substitution subsystems in the *Incorporation* (S1), *Mowing* (S2) and *Uprooting* (S3) scenarios compared with the *Reference* scenario, and (c) total emissions from the product system. Different processes contributing to cumulative emissions are indicated by different colours.

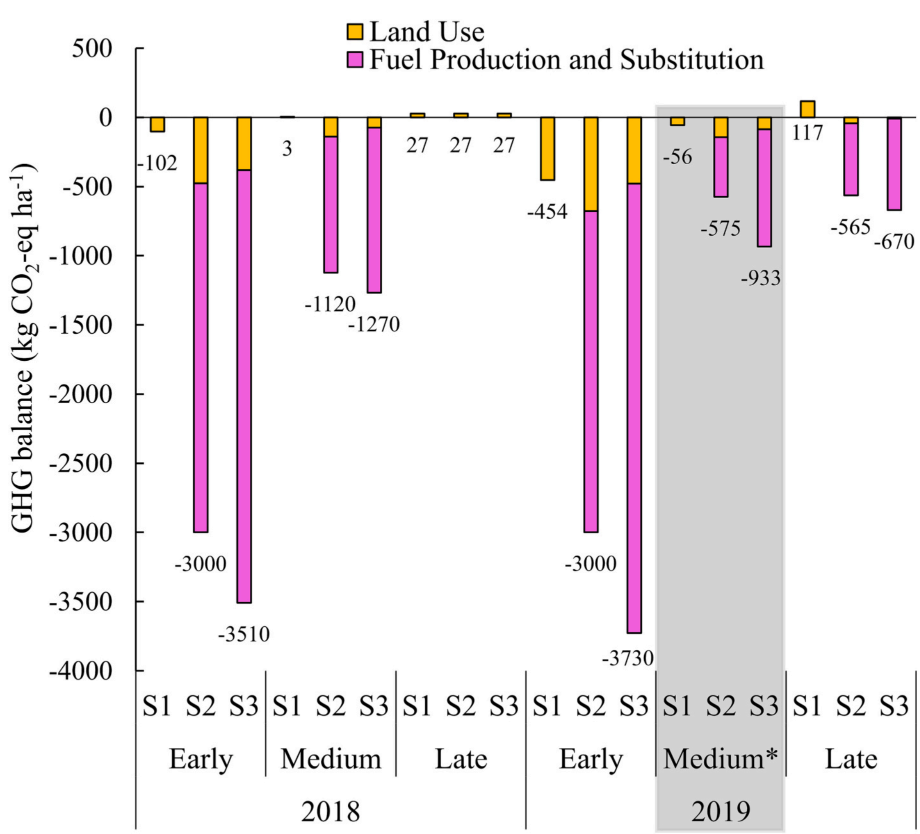


Fig. 7. Greenhouse gas (GHG) balance following *Early*, *Medium* and *Late* cover crop establishment in 2018 and 2019 in the land use and fuel production and substitution subsystems in the *Incorporation* (S1), *Mowing* (S2) and *Uprooting* (S3) scenarios. Grey shading indicates the base case.

Climate change mitigation potential (kg CO_2-eq ha^{-1})

Group	Parameter	S1	S2	S3
	Default	56	575	933
Transport	Increased transport distance (50%) [P]	56	524	833
	Decreased transport distance (50%) [O]	56	627	1034
Direct soil N_2O	High emission factor direct soil N_2O from CC [P]	-91	508	931
	Low emission factor direct soil N_2O from CC [O]	205	644	936
	High emission factor direct soil N_2O from digestate [P]	56	452	709
	Low emission factor direct soil N_2O from digestate [O]	56	615	1005
Reduced leaching	Low reduced leaching [P]	-17	483	831
	High reduced leaching [O]	78	604	965
	High N winter losses [P]	31	567	933
	Low N winter losses [O]	81	584	933
SOC sequestration	Low humification coefficient digestate [P]	56	557	898
	High humification coefficient digestate [O]	56	591	964
	High r_e-value [P]	15	541	913
	High r_e-value [O]	65	583	938
	150 yr SOC storage [P]	-24	508	893
	50 yr SOC storage [O]	160	663	986
	Low extra root C [P]	5	525	883
CH_4 losses	High CH_4 losses at AD [P]	56	426	642
	Low CH_4 losses at AD [O]	56	587	956
	High amount of biogas being flared [P]	56	559	902
	Low amount of biogas being flared [O]	56	588	958
	High CH_4 losses at digestate storage [P]	56	553	891
	Low CH_4 losses at digestate storage [O]	56	580	942
	Pessimistic (all [P] combined)	-335	-132	-43
	Optimistic (all [O] combined)	366	918	1297

Increase 2.0x

Decrease 2.0x

■ = No mitigation

Fig. 8. Impact of optimistic [O] and pessimistic [P] parameter assumptions and values sourced from the literature on the climate change mitigation potential of the *Incorporation* (S1), *Mowing* (S2) and *Uprooting* (S3) scenarios. Colour gradient ranges from green, representing more than a two-fold increase, to red, representing a two-fold decrease in the mitigation potential. Dark red boxes indicate an increased climate impact compared to the reference scenario.

4. Discussion

At the default time of establishment (*Medium*, 2019), CC biomass production (above- and belowground) was 2.0 Mg DM ha^{-1} (Fig. 3), which resulted in estimated biomass yield of 0.9 and 1.8 Mg DM ha^{-1} in the *Mowing* and *Uprooting* scenario, respectively. Previous studies of unfertilised oilseed radish as a CC under similar conditions have observed large variations in growth (Li et al., 2015; Lövgren, 2022; Olofsson and Ernfors, 2022; Petersen et al., 2011; Taghizadeh-Toosi et al., 2022), but generally somewhat lower aboveground biomass growth compared with our findings. In the context of energy cover cropping, where the biomass is used for energy production, obtaining sufficient yield is crucial for economic viability. Molinuevo-Salces et al. (2013) estimated that the economic threshold for harvesting a CC for biogas production is 2 Mg DM ha^{-1}, but this threshold depends on energy prices, which have become increasingly unstable due to the full Russian invasion of Ukraine (World Bank, 2022). Harvesting both aboveground and belowground biomass has the potential to increase yield per unit area, but may introduce challenges in the biogas production process by considerably increasing the risk of adding unwanted components, such as sand and grit, into the reactor, leading to operational difficulties and failures (Steffen et al., 1998). As indicated in the present study, earlier establishment of the CC can substantially increase biomass yield, but may be inconvenient as it also requires the preceding crop to be harvested early. This may not be possible in most crop rotations, but could be applied if the preceding crop is harvested before reaching maturity and used as fodder or if it matures early in the season, such as early-harvested peas (Hall et al., 2017). Another way to increase CC yield is to apply fertiliser (Launay et al., 2022), but this brings increased economic and environmental costs.

The highest SOC sequestration potential was seen for the *Incorporation* scenario, where it amounted to 130 kg C ha^{-1} after 100 years, corresponding to 477 kg CO_2 ha^{-1}. The SOC sequestration potential was lower when the CC was harvested, but some of the difference in sequestration potential was compensated for when the digestate was returned to the soil. In total, the sequestration potential amounted to 110 and 66 kg C ha^{-1} in the *Mowing* and *Uprooting* scenario, respectively. Previous meta-analyses of SOC changes under cover cropping have reported average SOC sequestration of 560 ha^{-1} year^{-1} (Jian et al., 2020) and 320 kg C ha^{-1} year^{-1} (Poeplau and Don, 2015), both of which are higher than in our study. This could be explained by these meta-analyses including studies conducted in regions with a longer vegetation period than Sweden, so the results are not directly comparable. In a study specifically in southern Sweden on the SOC effect of cover cropping with ryegrass, the average sequestration rate was found to be 320 kg C ha^{-1} $year^{-1}$, but with rather large variation (±280 kg C ha^{-1} $year^{-1}$) (Poeplau et al., 2015). Chaplot and Smith (2023) argued that the majority of field studies examining the SOC effect of CCs lack important features, such as appropriate controls and effects in a long-term perspective, and that the sequestration potential suggested in previous meta-analyses may be greatly overestimated. Overall, we found that SOC sequestration potential was dependent on biomass growth, which in turn was heavily dependent on time of establishment of the CC (Fig. 3). In general, earlier establishment led to greater biomass growth and hence greater SOC sequestration potential, but also to larger N_2O emissions, which offset some of the mitigation potential of the increased SOC sequestration. The results also indicated the high importance of the time horizon of the stored C (Fig. 4), which needs to be considered when comparing the results with those from other studies. The increased SOC stock also led to sequestration of N in the soil, causing a deficit in the N balance in both the *Incorporation* and *Mowing* scenarios compared with the *Reference*. As a result, more synthetic N fertiliser was needed in those scenarios to compensate for the deficit. The influence of SOC on the N balance of the systems is consistent with the underlying principle that, stoichiometrically, N must also be proportionally incorporated when C is sequestered (van Groenigen et al., 2017).

Previous studies have concluded that belowground biomass has greater potential to build SOC (Kätterer et al., 2011; Menichetti et al., 2015; Rasse et al., 2005). The humification coefficients applied in this study are consistent with this hypothesis of higher recalcitrance of CC residues belowground, i.e. roots and rhizodeposition, compared with aboveground. The humification coefficient used for belowground biomass was based on plants with a different root system to the oilseed radish taproot. In addition, recent findings have shown that rhizodeposition is correlated to root morphology, with more branched roots resulting in a larger share of rhizodeposition for belowground C input than less branched roots, such as those from oilseed radish (Engedal et al., 2023). Therefore, more research is needed to better quantify the effect of belowground biomass of oilseed radish under similar conditions. The humification coefficient for digestate was set to 0.266, based on a coefficient for manure from Bolinder et al. (2018), and this relatively large value resulted in rather small differences in SOC sequestration potential between the scenarios. This is in line with the concept that the most easily degradable material, i.e. the fraction of the biomass that would have degraded rapidly in the soil if it had not been harvested, is degraded during anaerobic digestion (Thomsen et al., 2013). However, there are currently no reliable scientific data on the actual SOC effect to contradict or corroborate this. Furthermore, despite belowground biomass being harvested in the *Uprooting* scenario, SOC sequestration potential was still relatively large due to the additional belowground C input, which was assumed to correspond to 65 % of the sequestration potential from the roots based on results from Bolinder et al. (2007). The practical implication of the *Uprooting* scenario were not studied further here. However, given the late potentially harvest date in autumn and the wet conditions in Scandinavia in that season, it is not very likely that either a self-propelled or tractor-drawn beet harvester could access the field without causing considerable soil compaction. This might even affect the feasibility of the *Mowing* scenario, since recovery of the relatively wet biomass would lead to high loads and a corresponding impact on the soil structure. Timing of harvest in suitable conditions and development of lightweight machinery and/or recovery processes would increase the likelihood of implementation of cover crop harvest.

The climate effect of SOC sequestration potential was assessed using the GWP_{100} approach, focusing on residual C after 100 years (Fig. 4d). However, the GWP method does not account for the timing of the emissions, which makes it less suitable for evaluating the climate effect of temporary C storage, such as C sinks in vegetation and soils (Brandão et al., 2013). For a more comprehensive evaluation of the climate impact of SOC sequestration potential, use of dynamic climate impact assessment approaches, such as the method employed by Ericsson et al. (2013), may be suitable. However, such methods are not commonly used in the field of LCA, which hampers comparison with other studies in the same domain. Using SOC sequestration in a cropping system as a tool to mitigate climate change also has finite capacity (Smith, 2014), i.e. there is a limit to how much C can be sequestered (Moinet et al., 2023). Without information about the specific soil, it is difficult to verify how much a certain measure will affect the SOC content under specific conditions. Furthermore, SOC sequestration is a reversible process, meaning that sequestered C may be re-emitted into the atmosphere, e.g. if the land use changes. This loss of SOC often happens faster than SOC build-up (Smith, 2005). In carbon credit schemes, where C offsets in one system are used to compensate for GHG emissions elsewhere, the permanence aspect, which refers to how long the C will be sequestered and kept out of the atmosphere, is of high importance (Paul et al., 2023). In such schemes, the benchmark is generally C storage for at least 100 years (Radley et al., 2021), but some apply a longer time frame, such as 1000 years. Our results showed that in a 100-year perspective, SOC sequestration via CC cultivation can be used to mitigate climate change, but over a longer period the sequestration effect will be low. This means that SOC sequestration can be an important tool for mitigating current elevated radiative forcing, by lowering CO_2 concentration in the atmosphere, but it cannot compensate for new fossil CO_2 emissions, which cause climate perturbations for thousands of years (Archer et al., 2009). Ensuring an equivalent timeframe for offset through SOC sequestration via CC cultivation would be challenging for any carbon credit scheme. Another important aspect of carbon credit schemes is the additionality (Paul et al., 2023). In best practice, it should be proven that credit is based on measures that would not have occurred without the credit system but, since cover cropping is associated with many other positive benefits from an agricultural perspective, it may be difficult to prove this.

Another important flux in the GHG balance of agricultural systems is soil-borne emissions of N_2O. Green manure crops with low C/N ratio that are terminated, tilled or frozen in late autumn or early winter add readily available C and N compounds to soils that are usually wet, increasing the risk of high N_2O emissions (Butterbach-Bahl et al., 2013; Groffman et al., 2009). Oilseed radish and other members of the Brassicaceae can produce substantially higher emissions than other CCs during winter when they are killed by freezing or ploughed under (Aziz, 2022; Dörsch, 2000; Olofsson and Ernfors, 2022). To account for the potential elevation in N_2O emissions from oilseed radish CC cultivation, we used field data from similar conditions to develop an emission factor in terms of N-N_2O emissions per kg added N in crop biomass, as done in the IPCC methodology. The emission factor showed a large variation between (Fig. 2), with an average value of 0.0153 kg N_2O-N kg^{-1} N, while the emission factor for crop residues in the IPCC methodology is set to 0.006 kg N_2O-N kg^{-1} N (IPCC, 2019). In this study, direct soil N_2O emissions induced by CC cultivation were highest in the *Incorporation* scenario (Fig. 5a), where the biomass was left unharvested and incorporated into the soil. When the biomass was harvested, the N_2O emissions from CC cultivation were considerably reduced, although returning digestate to the soil resulted in N_2O emissions that reduced the difference between harvesting and not harvesting the CC. However, the risk of N_2O emissions from field application of digestate may be low, as the digestate contains more recalcitrant C and is returned to the field in spring when there is uptake of N by the crop and the soil has started to dry out. This was reflected in our calculations, where the direct N_2O emission factor used for the digestate was considerably lower, based on literature data (Fig. 5b). Furthermore, CC cultivation led to avoided indirect N_2O emissions since it reduced N leaching from the field during winter compared with the *Reference* scenario (Fig. 5c). The amount of avoided indirect emissions was greater in the scenarios where the CC was harvested, since winter freeze-thaw events were assumed to add to N leaching losses. It was also a result of increased N availability in the digestate, which increased its fertiliser value. In the scenarios involving digestate, avoided N_2O emissions, direct and indirect, from synthetic fertiliser use were included in the GHG balance calculations. Overall, the results showed that N_2O emissions were highest in the *Incorporation* scenario, and 47 % and 85 % lower, respectively, in the *Mowing* and *Uprooting* scenarios. In contrast, Li et al. (2015) found no significant differences in N_2O emissions between removed and retained CCs, including oilseed rape. In a recent Swedish study, Lövgren (2022) examined the effect on N_2O emissions during winter when oilseed radish was cut and removed or uprooted and removed, compared with leaving the CC unharvested, and found that during the first 30 days, N_2O emissions were reduced by 66 % when the oilseed radish was uprooted and removed and by 61 % when it was cut and removed. However, when the full 79-day period was taken into account, there was a significant difference only when the oilseed radish was uprooted and removed, with N_2O emissions then reduced by 65 % (Lövgren, 2022). More studies are needed to clarify the effects on N_2O emissions of cultivation and biomass management of different CC. Moreover, since effects of biomass removal on N_2O emissions have been observed after the winter period, e.g. during spring cultivation (Li et al., 2015), measurements over longer periods or year-round measurements and evaluations of whole crop rotations are necessary.

Estimating soil N_2O emissions is associated with large uncertainties, due to the emissions varying widely over time and within the same field and to underlying processes being complex and still not fully understood (Butterbach-Bahl et al., 2013; Venterea et al., 2012). In this study, we applied the IPCC approach together with our own derived emission factors to estimate N_2O emissions (IPCC, 2019). Estimating N_2O emissions based on the total amount of N applied to soil in a year is associated with large uncertainties, as factors that play an important role in N_2O production, such as soil moisture, soil temperature, availability of readily degradable plant material, supply and mineralisation of N, and competition for available N, vary throughout the year (Butterbach-Bahl et al., 2013; Lashermes et al., 2022). In particular, N_2O emissions "hot-spots" and "hot-moments" can occur when different factors coincide (Groffman et al., 2009; Wagner-Riddle et al., 2020). The possibility to influence soil N_2O emissions through management practices, e.g. applying N at an appropriate time of year or adjusting application rates (Snyder et al., 2014), is not captured in detail with this approach. However, in the present study such potential variations were further assessed in a sensitivity analysis, using N_2O emissions values from the literature for oilseed radish cultivated as a CC.

The climate footprint assessment showed that all alternative scenarios, including CC cultivation, reduced GHG emissions compared with the *Reference* scenario with no CC. This reduction was attributed to an increase in SOC stock, while elevated soil N_2O emissions, emissions associated with a deficit in the N balance and field operations reduced the climate mitigation potential (Fig. 6a). Harvesting the CC biomass for biogas production resulted in even larger mitigation potential compared with the *Reference* scenario. This increased mitigation potential was mostly driven by the upgraded biogas produced replacing fossil diesel (Fig. 6b, c). The results were highly dependent on time of the establishment of the CC, with early establishment leading to greater SOC sequestration, and thus greater diesel substitution potential, due to the increased biomass growth compared with medium and late establishment (Fig. 7). This highlights the importance of high yield for energy CC systems. The sensitivity analysis showed that the climate change mitigation potential was highly sensitive to the estimated direct soil N_2O emissions (Fig. 8). This indicates that in conditions prone to high N_2O emissions, harvesting CC biomass may be a way of avoiding these emission peaks. Furthermore, the results showed high sensitivity to CH_4 losses throughout the upgraded biogas production value chain.

While other CC species may have a lower risk of high N_2O emissions (Dörsch, 2000; Li et al., 2015; Olofsson and Ernfors, 2022), the fast-growing oilseed radish produces larger above- and belowground biomass (Engedal et al., 2023), which increases the SOC sequestration potential and provides more biomass that can be used to replace products with a high climate impact. In addition, Molinuevo-Salces et al. (2013) found that oilseed radish, among 10 different CC species tested, had the highest specific CH_4 yield. This makes oilseed radish a relevant CC for climate change mitigation, provided that an appropriate management scheme is applied, as demonstrated in this study. However, in CC systems where the biomass is not harvested, other species may be more appropriate to avoid elevated N_2O emissions, which can otherwise result in an increased climate impact as shown in our sensitivity analysis.

Cover crop cultivation provides multiple advantages, such as increased domestic bioenergy production, reduced reliance on external fossil energy sources and increased resilience of cropping systems. The advantages of CCs extend beyond bioenergy, as they can be valuable resources for livestock fodder and as feedstock for biorefineries. Utilisation of biogas carries an array of benefits, including its capacity to serve as a reliable energy reservoir, an important characteristic in an energy landscape increasingly dominated by intermittent energy sources. To our knowledge, this is the first full life cycle climate footprint study to investigate the effect of an energy CC in a southern Scandinavian context.

5. Conclusions

Comprehensive analysis of the life cycle climate effects of oilseed radish-CC cultivation with different biomass management regimes in southern Scandinavia revealed that CC cultivation in all three alternative scenarios studied considerably reduced the climate impact compared with the *Reference* scenario with no CC. This reduction was partly due to the increased SOC stock resulting from introduction of the CC into the cropping system. In addition, harvesting the CC in the *Mowing* and *Uprooting* scenarios resulted in biogas production, providing an opportunity to reduce reliance on fossil fuels and mitigate climate change. Overall, the GHG emissions reduction was 0.056, 0.575 and 0.933 Mg CO_2-eq ha^{-1} in the *Incorporation*, *Mowing* and *Uprooting* scenario, respectively, compared with the *Reference*.

The timing of CC establishment was shown to have a large impact on climate change mitigation potential, with early establishment resulting in higher biomass yields, leading to greater SOC sequestration potential and a larger substitution effect. Sensitivity analysis revealed high sensitivity of the results to the value of the emission factor used for direct soil N_2O emissions during CC cultivation and digestate application, the parameters used to estimate reduced soil N leaching in CC cultivation and the amount of CH_4 lost during different processes in the biogas production life cycle stage.

The findings in this study offer valuable insights into creating sustainable agricultural systems in southern Scandinavia, and other regions with a similar climate, and underscore the importance of considering management practices and timing to achieve significant climate benefits through CC cultivation. Future research should focus on refining parameter choices and on identifying the specific mechanisms driving the climate effects of CC cultivation, to enhance its effectiveness in mitigating climate change.

CRediT authorship contribution statement

Johan Nilsson: Writing – original draft, Methodology, Conceptualization. **Maria Ernfors:** Writing – review & editing, Conceptualization. **Thomas Prade:** Writing – review & editing, Conceptualization. **Per-Anders Hansson:** Supervision, Project administration.

Declaration of competing interest

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

Data availability

Data will be made available on request.

Acknowledgements

This article was produced as part of the Mistra Food Futures research programme (www.mistrafoodfutures.se), Sweden (grant number DIA2018/24 #7).

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2024.170629.

References

Abdalla, M., Hastings, A., Cheng, K., Yue, Q., Chadwick, D., Espenberg, M., Truu, J., Rees, R.M., Smith, P., 2019. A critical review of the impacts of cover crops on nitrogen leaching, net greenhouse gas balance and crop productivity. Glob. Change Biol. 25, 2530–2543. https://doi.org/10.1111/gcb.14644.

Andersen, B.J., Samarappuli, D.P., Wick, A., Berti, M.T., 2020. Faba bean and pea can provide late-fall forage grazing without affecting maize yield the following season. Agronomy 10, 80. https://doi.org/10.3390/agronomy10010080.

Andrén, O., Kätterer, T., 1997. Icbm: the introductory carbon balance model for exploration of soil carbon balances. Ecol. Appl. 7, 1226–1236. https://doi.org/10.1890/1051-0761(1997)007[1226:ITICBM]2.0.CO;2.

Andrén, O., Kätterer, T., Karlsson, T., 2004. ICBM regional model for estimations of dynamics of agricultural soil carbon pools. Nutr. Cycl. Agroecosystems 70, 231–239. https://doi.org/10.1023/B:FRES.0000048471.59164.ff.

Archer, D., Eby, M., Brovkin, V., Ridgewell, A., Long, C., Mikolajewicz, U., Caldeira, K., Matsumoto, K., Munhoven, G., Montenegro, A., Tokos, K., 2009. Atmospheric lifetime of fossil fuel carbon dioxide. Annu. Rev. Earth Planet. Sci. 37, 117–134.

Aronsson, H., Torstensson, G., 2004. Beräkning av olika odlingsåtgärders inverkan på kväveutlakningen - Beskrivning av ett pedagogiskt verktyg för beräkning av kväveutlakning från enskilda fält och gårdar, vol. No. 78. Ekohydrologi. Swedish University of Agricultural Sciences, Uppsala, Sweden.

Astivia, O., Zumbo, B., 2019. Heteroskedasticity in multiple regression analysis: what it is, how to detect it and how to solve it with applications in R and SPSS. Pract. Assess. Res. Eval. 24 https://doi.org/10.7275/q5xr-fr95.

Aziz, K.. Effects of cover crops on nitrous oxide (N2O) emissions in cereal cropping (MSc thesis). SLU. https://stud.epsilon.slu.se/18283/.

Babiker, M., Berndes, G., Blok, K., Cohen, B., Cowie, A., Geden, O., Ginzburg, V., Leip, A., Smith, P., Sugiyama, M., Yamba, F., 2022. Cross-sectoral perspectives. In: Shukla, P. R., Skea, J., Slade, R., Al Khourdajie, A., van Diemen, R., McCollum, D., Pathak, M., Some, S., Vyas, P., Fradera, R., Belkacemi, M., Hasija, A., Lisboa, G., Luz, S., Malley, J. (Eds.), Climate Change 2022: Mitigation of Climate Change. Contribution of Working Group III to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK and New York, NY, USA.

Batjes, N.H., 1996. Total carbon and nitrogen in the soils of the world. Eur. J. Soil Sci. 47, 151–163. https://doi.org/10.1111/j.1365-2389.1996.tb01386.x.

Björnsson, L., Prade, T., Lantz, M., Börjesson, P., Svensson, S.-E., Eriksson, H., 2013. Impact of Biogas Energy Crops on Greenhuse Gas Emissions, Soil Organic Matter and Food Crop Production - a Case Study on Farm Level (The Swedish knowledge centre for renewable transportation fuels).

Björnsson, L., Prade, T., Lantz, M., 2016. Grass for Biogas – Arable Land as a Carbon Sink; an Environmental and Economic Assessment of Carbon Sequestration in Arable Land through Introduction of Grass for Biogas Production. Energiforsk.

Blanco-Canqui, H., Shaver, T.M., Lindquist, J.L., Shapiro, C.A., Elmore, R.W., Francis, C. A., Hergert, G.W., 2015. Cover crops and ecosystem services: insights from studies in temperate soils. Agron. J. 107, 2449–2474. https://doi.org/10.2134/agronj15.0086.

Bolinder, M.A., Janzen, H.H., Gregorich, E.G., Angers, D.A., VandenBygaart, A.J., 2007. An approach for estimating net primary productivity and annual carbon inputs to soil for common agricultural crops in Canada. Agric. Ecosyst. Environ. 118, 29–42. https://doi.org/10.1016/j.agee.2006.05.013.

Bolinder, M.A., Menichetti, L., Meurer, K., Lundblad, M., Kätterer, T., 2018. New Calibration of the ICBM Model & Analysis of Soil Organic Carbon Concentration from Swedish Soil Monitoring Programs (No. 20 2018). SMED.

Börjesson, P., Lantz, M., Andersson, J., Björnsson, L., Möller Fredriksson, B., Fröberg, M., Hanarp, P., Hulteberg, C.P., Iverfeldt, E., Lundgren, J., Röj, A., Svensson, H., Zinn, E., 2016. Methane as Vehicle Fuel – A Well to Wheel Analysis (METDRIV) (The Swedish knowledge centre for renewable transportation fuels).

Brandão, M., Levasseur, A., Kirschbaum, M.U.F., Weidema, B.P., Cowie, A.L., Jørgensen, S.V., Hauschild, M.Z., Pennington, D.W., Chomkhamsri, K., 2013. Key issues and options in accounting for carbon sequestration and temporary storage in life cycle assessment and carbon footprinting. Int. J. Life Cycle Assess. 18, 230–240. https://doi.org/10.1007/s11367-012-0451-6.

Brandão, M., Weidema, B.P., Martin, M., Cowie, A., Hamelin, L., Zamagni, A., 2022. Consequential Life Cycle Assessment: What, Why and How?, in: Reference Module in Earth Systems and Environmental Sciences. Elsevier. https://doi.org/10.1016/B978-0-323-90386-8.00001-2.

Butterbach-Bahl, K., Baggs, E.M., Dannenmann, M., Kiese, R., Zechmeister-Boltenstern, S., 2013. Nitrous oxide emissions from soils: how well do we understand the processes and their controls? Philos. Trans. R. Soc. Lond. B Biol. Sci. 368, 20130122. https://doi.org/10.1098/rstb.2013.0122.

Chaplot, V., Smith, P., 2023. Cover crops do not increase soil organic carbon stocks as much as has been claimed: What is the way forward? Glob. Change Biol. 29, 6163–6169. https://doi.org/10.1111/gcb.16917.

Cherubini, F., Strømman, A.H., 2011. Life cycle assessment of bioenergy systems: state of the art and future challenges. Bioresour. Technol. 102, 437–451. https://doi.org/10.1016/j.biortech.2010.08.010.

Cherubini, F., Fuglestvedt, J., Gasser, T., Reisinger, A., Cavalett, O., Huijbregts, M.A.J., Johansson, D.J.A., Jørgensen, S.V., Raugei, M., Schivley, G., Strømman, A.H., Tanaka, K., Levasseur, A., 2016. Bridging the gap between impact assessment methods and climate science. Environ. Sci. Policy 64, 129–140. https://doi.org/10.1016/j.envsci.2016.06.019.

Constantin, J., Mary, B., Laurent, F., Aubrion, G., Fontaine, A., Kerveillant, P., Beaudoin, N., 2010. Effects of catch crops, no till and reduced nitrogen fertilization on nitrogen leaching and balance in three long-term experiments. Agric. Ecosyst. Environ. 135, 268–278. https://doi.org/10.1016/j.agee.2009.10.005.

Dörsch, P., 2000. Nitrous Oxide and Methane Fluxes in Differentially Managed Agricultural Soils of a Hilly Landscape in Southern Germany. München.

Edström, M., Jansson, L.-E., Lantz, M., Johansson, L.-G., Nordberg, U., Nordberg, Å., 2008. Gårdsbaserad biogasproduktion - System, ekonomi och klimatpåverkan (No. 42), Kretslopp & Avfall. JTI – Institutet för jordbruks- och miljöteknik.

Ellis, K.E., Barbercheck, M.E., 2015. Management of Overwintering Cover Crops Influences Floral Resources and Visitation by native bees. Environ. Entomol. 44, 999–1010. https://doi.org/10.1093/ee/nvv086.

Engedal, T., Magid, J., Hansen, V., Rasmussen, J., Sørensen, H., Stoumann Jensen, L., 2023. Cover crop root morphology rather than quality controls the fate of root and rhizodeposition C into distinct soil C pools. Glob. Change Biol. 29, 5677–5690. https://doi.org/10.1111/gcb.16870.

Ericsson, N., Porsö, C., Ahlgren, S., Nordberg, Å., Sundberg, C., Hansson, P.-A., 2013. Time-dependent climate impact of a bioenergy system – methodology development and application to Swedish conditions. GCB Bioenergy 5, 580–590. https://doi.org/10.1111/gcbb.12031.

EU, 2018. DIRECTIVE (EU) 2018/2001 OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 11 December 2018 on the Promotion of the Use of Energy from Renewable Sources.

Foltz, M.E., Kent, A.D., Koloutsou-Vakakis, S., Zilles, J.L., 2021. Influence of rye cover cropping on denitrification potential and year-round field N2O emissions. Sci. Total Environ. 765, 144295 https://doi.org/10.1016/j.scitotenv.2020.144295.

Forster, P., Storelvmo, T., Armour, K., Collins, W., Dufresne, J.-L., Frame, D., Lunt, D.J., Mauritsen, T., Palmer, M.D., Watanabe, M., Wild, M., Zhang, H., 2021. The earth's energy budget, climate feedbacks, and climate sensitivity. In: Masson-Delmotte, V., Zhai, P., Pirani, A., Connors, S.L., Péan, C., Berger, S., Caud, N., Chen, Y., Goldfarb, L., Gomis, M.I., Huang, M., Leitzell, K., Lonnoy, E., Matthews, J.B.R., Maycock, T.K., Waterfield, T., Yelekçi, O., Yu, R., Zhou, B. (Eds.), In Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 923–1054.

Gode, J., Martinsson, F., Hagberg, L., Öman, A., Höglund, J., Palm, D., Ekvall, T., 2011. Miljöfaktaboken 2011 - estimated emission factors for fuels. In: Electricity, Heat and Transport in Sweden (No. A08-833), ANLÄGGNINGS- OCH FÖRBRÄNNINGSTEKNIK. VÄRMEFORSK Service AB, Stockholm.

Groffman, P.M., Butterbach-Bahl, K., Fulweiler, R.W., Gold, A.J., Morse, J.L., Stander, E. K., Tague, C., Tonitto, C., Vidon, P., 2009. Challenges to incorporating spatially and temporally explicit phenomena (hotspots and hot moments) in denitrification models. Biogeochemistry 93, 49–77. https://doi.org/10.1007/s10533-008-9277-5.

Guenet, B., Gabrielle, B., Chenu, C., Arrouays, D., Balesdent, J., Bernoux, M., Bruni, E., Caliman, J.-P., Cardinael, R., Chen, S., Ciais, P., Desbois, D., Fouche, J., Frank, S., Henault, C., Lugato, E., Naipal, V., Nesme, T., Obersteiner, M., Pellerin, S., Powlson, D.S., Rasse, D.P., Rees, F., Soussana, J.-F., Su, Y., Tian, H., Valin, H., Zhou, F., 2021. Can N2O emissions offset the benefits from soil organic carbon storage? Glob. Change Biol. 27, 237–256. https://doi.org/10.1111/gcb.15342.

Hall, C., Hillen, C., Garden Robinson, J., 2017. Composition, nutritional value, and health benefits of pulses. Cereal Chem. 94, 11–31. https://doi.org/10.1094/CCHEM-03-16-0069-FI.

Hansson, D., Svensson, S.-E., Prade, T., 2021. Etableringstidpunktens inverkan på sommarmellangrödors ogräsbekämpande egenskaper, markkolsbidrag och potential som biogasråvara – fältförsök Norra Åsum 2018 (No. 2021–1). Landskapsarkitektur, trädgård, växtproduktionsvetenskap: rapportserie. Alnarp, Biosystem och teknologi, Sveriges lantbruksuniversitet.

Hijazi, O., Munro, S., Zerhusen, B., Effenberger, M., 2016. Review of life cycle assessment for biogas production in Europe. Renew. Sustain. Energy Rev. 54, 1291–1300. https://doi.org/10.1016/j.rser.2015.10.013.

IPCC, 2019. 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse gas Inventories. IPCC.

Janzen, H.H., McGinn, S.M., 1991. Volatile loss of nitrogen during decomposition of legume green manure. Soil Biol. Biochem. 23, 291–297. https://doi.org/10.1016/0038-0717(91)90066-S.

Jian, J., Du, X., Reiter, M.S., Stewart, R.D., 2020. A meta-analysis of global cropland soil carbon changes due to cover cropping. Soil Biol. Biochem. 143, 107735 https://doi.org/10.1016/j.soilbio.2020.107735.

Karlsson, T., 2012. Carbon and Nitrogen Dynamics in Agricultural Soils Model Applications at Different Scales in Time and Space. Swedish University of Agricultural Sciences, Uppsala, Department of Soil and Environment.
Kätterer, T., Bolinder, M.A., Andrén, O., Kirchmann, H., Menichetti, L., 2011. Roots contribute more to refractory soil organic matter than above-ground crop residues, as revealed by a long-term field experiment. Agric. Ecosyst. Environ. 141, 184–192. https://doi.org/10.1016/j.agee.2011.02.029.
Lashermes, G., Recous, S., Alavoine, G., Janz, B., Butterbach-Bahl, K., Ernfors, M., Laville, P., 2022. N2O emissions from decomposing crop residues are strongly linked to their initial soluble fraction and early C mineralization. Sci. Total Environ. 806, 150883 https://doi.org/10.1016/j.scitotenv.2021.150883.
Launay, C., Houot, S., Frédéric, S., Girault, R., Levavasseur, F., Marsac, S., Constantin, J., 2022. Incorporating energy cover crops for biogas production into agricultural systems: benefits and environmental impacts. A review. Agron. Sustain. Dev. 42, 57. https://doi.org/10.1007/s13593-022-00790-8.
Li, C., Frolking, S., Butterbach-Bahl, K., 2005. Carbon sequestration in arable soils is likely to increase nitrous oxide emissions, offsetting reductions in climate radiative forcing. Clim. Change 72, 321–338. https://doi.org/10.1007/s10584-005-6791-5.
Li, X., Petersen, S.O., Sørensen, P., Olesen, J.E., 2015. Effects of contrasting catch crops on nitrogen availability and nitrous oxide emissions in an organic cropping system. Agric. Ecosyst. Environ. 199, 382–393. https://doi.org/10.1016/j.agee.2014.10.016.
Lövgren, 2022. Complete Removal of Biomass from Oilseed Radish as a Cover Crop Decreased Nitrous Oxide Emissions (MSc Thesis). SLU. https://stud.epsilon.slu.se/18483/.
Ma, S., He, F., Tian, D., Zou, D., Yan, Z., Yang, Y., Zhou, T., Huang, K., Shen, H., Fang, J., 2018. Variations and determinants of carbon content in plants: a global synthesis. Biogeosciences 15, 693–702. https://doi.org/10.5194/bg-15-693-2018.
Maskinkalkylgruppen, 2023. Maskinkostnader 2023. Kalmar, Sweden, Hushållningssällskapet Kalmar - Kronoberg - Blekinge.
Menichetti, L., Ekblad, A., Kätterer, T., 2015. Contribution of roots and amendments to soil carbon accumulation within the soil profile in a long-term field experiment in Sweden. Agric. Ecosyst. Environ. 200, 79–87. https://doi.org/10.1016/j.agee.2014.11.003.
Minasny, B., Malone, B.P., McBratney, A.B., Angers, D.A., Arrouays, D., Chambers, A., Chaplot, V., Chen, Z.-S., Cheng, K., Das, B.S., Field, D.J., Gimona, A., Hedley, C.B., Hong, S.Y., Mandal, B., Marchant, B.P., Martin, M., McConkey, B.G., Mulder, V.L., O'Rourke, S., Richer-de-Forges, A.C., Odeh, I., Padarian, J., Paustian, K., Pan, G., Poggio, L., Savin, I., Stolbovoy, V., Stockmann, U., Sulaeman, Y., Tsui, C.-C., Vågen, T.-G., van Wesemael, B., Winowiecki, L., 2017. Soil carbon 4 per mille. Geoderma 292, 59–86. https://doi.org/10.1016/j.geoderma.2017.01.002.
Minx, J.C., Lamb, W.F., Callaghan, M.W., Fuss, S., Hilaire, J., Creutzig, F., Amann, Thorben, Beringer, T., Garcia, W. de O., Hartmann, J., Khanna, T., Lenzi, D., Luderer, Gunnar, Nemet, G.F., Rogelj, J., Smith, P., Vicente, J.L.V., Wilcox, J., Dominguez, M. del M.Z., 2018. Negative emissions—part 1: research landscape and synthesis. Environ. Res. Lett. 13, 063001 https://doi.org/10.1088/1748-9326/aabf9b.
Moinet, G.Y.K., Hijbeek, R., van Vuuren, D.P., Giller, K.E., 2023. Carbon for Soils, Not Soils for Carbon. Change Biol. n/a, Glob. https://doi.org/10.1111/gcb.16570.
Molinuevo-Salces, B., Larsen, S.U., Ahring, B.K., Uellendahl, H., 2013. Biogas production from catch crops: evaluation of biomass yield and methane potential of catch crops in organic crop rotations. Biomass Bioenergy 59, 285–292. https://doi.org/10.1016/j.biombioe.2013.10.008.
Muneer, F., Hovmalm, H.P., Svensson, S.-E., Newson, W.R., Johansson, E., Prade, T., 2021. Economic viability of protein concentrate production from green biomass of intermediate crops: a pre-feasibility study. J. Clean. Prod. 294, 126304 https://doi.org/10.1016/j.jclepro.2021.126304.
Nilsson, J., Sundberg, C., Tidåker, P., Hansson, P.-A., 2020. Regional variation in climate impact of grass-based biogas production: a Swedish case study. J. Clean. Prod. 275, 122778 https://doi.org/10.1016/j.jclepro.2020.122778.
Norberg, L., Aronsson, H., 2020. Effects of cover crops sown in autumn on N and P leaching. Soil Use Manage. 36, 200–211. https://doi.org/10.1111/sum.12565.
Notaris, C., Sørensen, P., Møller, H.B., Wahid, R., Eriksen, J., 2018. Nitrogen fertilizer replacement value of digestates from three green manures. Nutr. Cycl. Agroecosystems 112, 355–368. https://doi.org/10.1007/s10705-018-9951-5.
Olofsson, F., Ernfors, M., 2022. Frost killed cover crops induced high emissions of nitrous oxide. Sci. Total Environ. 837, 155634 https://doi.org/10.1016/j.scitotenv.2022.155634.
Paul, C., Bartkowski, B., Dönmez, C., Don, A., Mayer, S., Steffens, M., Weigl, S., Wiesmeier, M., Wolf, A., Helming, K., 2023. Carbon farming: are soil carbon certificates a suitable tool for climate change mitigation? J. Environ. Manage. 330, 117142 https://doi.org/10.1016/j.jenvman.2022.117142.
Petersen, S.O., Mutegi, J.K., Hansen, E.M., Munkholm, L.J., 2011. Tillage effects on N2O emissions as influenced by a winter cover crop. Soil Biol. Biochem. 43, 1509–1517. https://doi.org/10.1016/j.soilbio.2011.03.028.
Poeplau, C., Don, A., 2015. Carbon sequestration in agricultural soils via cultivation of cover crops – a meta-analysis. Agric. Ecosyst. Environ. 200, 33–41. https://doi.org/10.1016/j.agee.2014.10.024.
Poeplau, C., Aronsson, H., Myrbeck, Å., Kätterer, T., 2015. Effect of perennial ryegrass cover crop on soil organic carbon stocks in southern Sweden. Geoderma Reg. 4, 126–133. https://doi.org/10.1016/j.geodrs.2015.01.004.
Prade, T., Björnsson, L., Lantz, M., Ahlgren, S., 2017. Can domestic production of iLUC-free feedstock from arable land supply Sweden's future demand for biofuels? J. Land Use Sci. 12, 407–441. https://doi.org/10.1080/1747423X.2017.1398280.
Prade, T., Hansson, D., Svensson, S.-E., 2022. Etableringstidpunktens inverkan på sommarmellangrödors markkolsbidrag och ogräsbekämpande egenskaper - fältförsök på Helgegården 2019. Landskapsarkitektur, trädgård, växtproduktionsvetenskap: rapportserie. Alnarp, Biosystem och teknologi, Sveriges lantbruksuniversitet.
Quakernack, R., Pacholski, A., Techow, A., Herrmann, A., Taube, F., Kage, H., 2012. Ammonia Volatilization and Yield Response of Energy Crops after Fertilization with Biogas Residues in a Coastal Marsh of Northern Germany. Agric. Ecosyst. Environ.
Radley, G., Keenleyside, C., Frelih-Larsen, A., McDonald, H., Pyndt Andersen, S., Qwist-Hoffmann, H., Strange Olesen, A., Bowyer, C., Russi, D., European Commission, Climate Action DG, COWI, Ecologic Institute, IEEP, 2021. Setting up and Implementing Result-Based Carbon Farming Mechanisms in the EU: Technical Guidance Handbook.
Rasse, D.P., Rumpel, C., Dignac, M.-F., 2005. Is soil carbon mostly root carbon? Mechanisms for a specific stabilisation. Plant and Soil 269, 341–356. https://doi.org/10.1007/s11104-004-0907-y.
Ruijter, F.J., Huijsmans, J.F.M., Rutgers, B., 2010. Ammonia volatilization from crop residues and frozen green manure crops. Atmos. Environ. 44, 3362–3368. https://doi.org/10.1016/j.atmosenv.2010.06.019.
Schipanski, M.E., Barbercheck, M., Douglas, M.R., Finney, D.M., Haider, K., Kaye, J.P., Kemanian, A.R., Mortensen, D.A., Ryan, M.R., Tooker, J., White, C., 2014. A framework for evaluating ecosystem services provided by cover crops in agroecosystems. Agr. Syst. 125, 12–22. https://doi.org/10.1016/j.agsy.2013.11.004.
Smith, P., 2005. An overview of the permanence of soil organic carbon stocks: influence of direct human-induced, indirect and natural effects. Eur. J. Soil Sci. 56, 673–680. https://doi.org/10.1111/j.1365-2389.2005.00708.x.
Smith, P., 2014. Do grasslands act as a perpetual sink for carbon? Glob. Change Biol. 20, 2708–2711. https://doi.org/10.1111/gcb.12561.
Snyder, C., Davidson, E., Smith, P., Venterea, R., 2014. Agriculture: sustainable crop and animal production to help mitigate nitrous oxide emissions. Curr. Opin. Environ. Sustain. SI: System Dynamics and Sustainability 9–10, 46–54. https://doi.org/10.1016/j.cosust.2014.07.005.
Steffen, R., Szolar, O., Braun, R., 1998. Feedstocks for Anaerobic Digestion. University of Agricultural Sciences Vienna, Institute for Agrobiotechnology Tulln.
Styles, D., Gibbons, J., Williams, A.P., Dauber, J., Stichnothe, H., Urban, B., Chadwick, D. R., Jones, D.L., 2015. Consequential life cycle assessment of biogas, biofuel and biomass energy options within an arable crop rotation. GCB Bioenergy 7, 1305–1320. https://doi.org/10.1111/gcbb.12246.
Styles, D., Dominguez, E.M., Chadwick, D., 2016. Environmental balance of the UK biogas sector: an evaluation by consequential life cycle assessment. Sci. Total Environ. 560–561, 241–253. https://doi.org/10.1016/j.scitotenv.2016.03.236.
Taghizadeh-Toosi, A., Hansen, E.M., Olesen, J.E., Baral, K.R., Petersen, S.O., 2022. Interactive effects of straw management, tillage, and a cover crop on nitrous oxide emissions and nitrate leaching from a sandy loam soil. Sci. Total Environ. 828, 154316 https://doi.org/10.1016/j.scitotenv.2022.154316.
Thomsen, I.K., Olesen, J.E., Møller, H.B., Sørensen, P., Christensen, B.T., 2013. Carbon dynamics and retention in soil after anaerobic digestion of dairy cattle feed and faeces. Soil Biol. Biochem. 58, 82–87. https://doi.org/10.1016/j.soilbio.2012.11.006.
Tidåker, P., Bergkvist, G., Bolinder, M., Eckersten, H., Johnsson, H., Kätterer, T., Weih, M., 2016a. Estimating the environmental footprint of barley with improved nitrogen uptake efficiency—a Swedish scenario study. Eur. J. Agron. 80, 45–54. https://doi.org/10.1016/j.eja.2016.06.013.
Tidåker, P., Rosenqvist, H., Bergkvist, G., 2016b. Räkna med vall. Hur påverkas ekonomi och miljö när vall införs i spannmålsdominerade växtföljder, No. 445 (JTI – Institutet för jordbruks- och miljöteknik, Uppsala).
Torstensson, G., Aronsson, H., 2000. Nitrogen leaching and crop availability in manured catch crop systems in Sweden. Nutr. Cycl. Agroecosystems 56, 139–152. https://doi.org/10.1023/A:1009821519042.
van Groenigen, J.W., van Kessel, C., Hungate, B.A., Oenema, O., Powlson, D.S., van Groenigen, K.J., 2017. Sequestering soil organic carbon: a nitrogen dilemma. Environ. Sci. Technol. 51, 4738–4739. https://doi.org/10.1021/acs.est.7b01427.
Venterea, R.T., Halvorson, A.D., Kitchen, N., Liebig, M.A., Cavigelli, M.A., Grosso, S.J.D., Motavalli, P.P., Nelson, K.A., Spokas, K.A., Singh, B.P., Stewart, C.E., Ranaivoson, A., Strock, J., Collins, H., 2012. Challenges and opportunities for mitigating nitrous oxide emissions from fertilized cropping systems. Front. Ecol. Environ. 10, 562–570. https://doi.org/10.1890/120062.
Wagner-Riddle, C., Baggs, E.M., Clough, T.J., Fuchs, K., Petersen, S.O., 2020. Mitigation of nitrous oxide emissions in the context of nitrogen loss reduction from agroecosystems: managing hot spots and hot moments. Curr. Opin. Environ. Sustain. Climate Change, Reactive Nitrogen, Food Security and Sustainable Agriculture 47, 46–53. https://doi.org/10.1016/j.cosust.2020.08.002.
Wilcoxen, C.A., Walk, J.W., Ward, M.P., 2018. Use of cover crop fields by migratory and resident birds. Agric. Ecosyst. Environ. 252, 42–50. https://doi.org/10.1016/j.agee.2017.09.039.
Williams, S.M., Weil, R.R., 2004. Crop cover root channels may alleviate soil compaction effects on soybean crop. Soil Sci. Soc. Am. J. 68, 1403–1409. https://doi.org/10.2136/sssaj2004.1403.
World Bank, 2022. Commodity Markets Outlook - The Impact of the War in Ukraine on Commodity Markets. International Bank for Reconstruction and Development. World Bank, 1818 H Street NW, Washington, DC 20433.