Bioresource Technology 440 (2026) 133401

Contents lists available at ScienceDirect

Bioresource Technology

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

Valorising digestate as bio-based fertiliser next to biogas: Environmental life cycle costing of three farm-scale anaerobic co-digestion scenarios

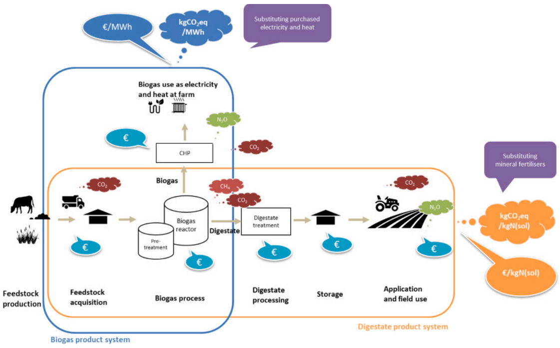
Karetta Vikki [a,*], Elina Tampio [b], Erika Winquist [b], Merja Saarinen [c], Sari Luostarinen [b]

[a] Natural Resources Institute Finland (Luke), Bioeconomy and Environment, Sustainability Assessment, Latokartanonkaari 9, 00791 Helsinki, Finland
[b] Natural Resources Institute Finland (Luke), Production Systems, Latokartanonkaari 9, 00791 Helsinki, Finland
[c] Natural Resources Institute Finland (Luke), Carbon Neutral Society, Tietotie 4, FI-31600 Jokioinen, Finland

HIGHLIGHTS

- The carbon footprint of the digestate (from cradle to grave) was lower than that of mineral fertiliser.
- Valorisation of digestate as a co-product decreased biogas emissions by 24–29 %.
- The net present value (NPV) was positive for all three biogas and digestate products.
- Monetising climate impacts increased the NPV for all biogas and digestate products.
- Different product specific hotspots were revealed for the biogas and digestate.

GRAPHICAL ABSTRACT



ARTICLE INFO

Keywords:
Biogas
Digestate
Climate impacts
Environmental life cycle costing
Life cycle assessment
Life cycle costing

ABSTRACT

Anaerobic digestion (AD) combines waste management with the production of renewable energy and recycled fertiliser products. Despite international targets to increase the recycling of nutrients for bio-based fertilisers (BBFs), there is a lack of studies assessing the environmental and economic impacts of the life cycle of digestate coming out of AD, and of associating the functional unit (FU) with the digestate, facilitating the comparability between mineral fertilisers. This study evaluated the environmental life cycle costs and climate impacts per FU of both biogas (MWh) and digestate (kg of soluble nitrogen) from feedstock procurement to the use on a farm (cradle-to-grave) which allowed comparison to purchased energy and mineral fertiliser. Three different farm-scale anaerobic digestion scenarios were studied utilizing slurry and different co-feedstocks. Results showed that biogas and digestate produced at farm had lower climate impacts and costs than the corresponding purchased products in all three scenarios. Valuing digestate as co-product decreased life cycle emissions of biogas by 24–29 %. Correspondingly, the carbon footprint of the digestate increased by 6–8% but was compensated by carbon sequestration potential when applied in soil. The NPV was positive for biogas and digestate product in all scenarios and highest for scenario utilising food industry side streams as co-feedstock. Monetising climate impacts increased the NPV of biogas and digestate because replaceable energy and mineral fertiliser had higher

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* Corresponding author at: Natural Resources Institute Finland (Luke), Bioeconomy and Environment, Sustainability Assessment, Latokartanonkaari 9, 00791 Helsinki, Finland.
E-mail address: karetta.vikki@luke.fi (K. Vikki).

https://doi.org/10.1016/j.biortech.2025.133401
Received 6 July 2025; Received in revised form 2 September 2025; Accepted 25 September 2025

climate impacts. Product level analyses revealed different product-specific hot spots for biogas and digestate for targeted improvements in sustainability.

1. Introduction

Anaerobic digestion (AD) as a microbiological process to convert biodegradable waste and residues as side streams into biogas and nutrient-rich digestion residues (digestate) has potential to help in achieving many sustainability targets (e.g. the EU Renewable Energy Directive (EC, 2009; EC, 2018), IEA (2017) and the US Renewable Fuel Standard (EPA, 2020). As a reaction to these renewable energy policies, literature reviews by Bacenetti et al., (2016), Balcioglu et al., (2022) and Egas et al. (2023) observed that the LCA studies of anaerobic digestion have been focusing only on biogas as a primary product because the digestate is seen as a secondary function of the anaerobic process, part of waste management process and not as a potential co-product alongside energy.

Only few LCA studies were found to cover the digestate utilisation on the field and associate the functional unit (FU) with the digestate product: per mass of digestate to be treated (Vazquez-Rowe et al., 2015, Styles et al., 2018; Spagnolo et al., 2019) or per nitrogen (N) contained in digestate (Timonen et al., 2019). However, according to the economic allocation guidance by Environmental Footprint initiative of the European Commission (EC, 2021) a product specific FU is a key component for meaningful comparison of the impacts of different products. The FU should also facilitate the comparability and interpretation of the results among systems with the same functionality (Ahlgren et al., 2013) and from a comparability point of view, it is essential to take the nutritional function into account if different fertiliser products are compared (Egas et al., 2023) e.g. soluble N available for plants.

There are drivers for the valorisation of the digestates as bio-based fertilisers (BBFs) through, for example EU Regulation on Fertilising Products (EC, 2019a), EU Green Deal (EC, 2019b) and Farm-to-Fork Strategy (EC, 2020), which all support the production and use of BBFs to supplement mineral fertilisers. However, despite the growing interest of markets and supporting policies, there is no Life Cycle Costing (LCC) method combined with LCA to valorise digestate product. In other words, studies of anaerobic digestion only combine LCA with LCC for the valorisation of the biogas and not digestate, considering only the phases of the life cycle from biogas plant construction to energy sales, excluding the end-of-life phase, i.e. digestate use (Britz and Delzeit, 2013, Bierer et al., 2015, Demichelis et al., 2022 and (Pasciucco et al., 2023)).

Finally, when digestate is given an economic value alongside biogas, the emissions are divided between the two products according to the economic allocation rule, which potentially reduces the carbon footprint of biogas (Timonen et al., 2019). Economic allocation is commonly used when co-products have very different physical relationships and end uses in the market (Kyttä et al., 2022). For example, biogas and digestate as by-products have a significant difference in mass, carbon balance and calorific value and therefore almost all emissions would be allocated to biogas or digestate despite both having value in the market (Timonen et al., 2019).

This study assesses the E-LCC for the valorisation of both biogas and digestate, from cradle-to-grave. The E-LCC is conducted by combining the LCA with the LCC in two different ways: 1) the life cycle environmental impacts are assessed next to life cycle costs in parallel and 2) the life cycle climate impacts are internalised in monetary terms as transfers (emission trade allowances) with the LCC results. Life cycle impacts and costs are allocated for both end-products: biogas and digestate by associating FU with both biogas (MWh) and digestate (kg of soluble N) and these are compared to the replaceable purchased energy and mineral fertiliser in finding potential for cost savings, emission reduction and value for the product (NPV). Finally, this study examines how the valorisation of digestate next to biogas product will decrease the emissions and costs of biogas product next to digestate. The scenarios are a typical Northern European (Finnish), small on-farm biogas plant processing manure (pig slurry) as the main feedstock and three different co-feedstocks relevant in the current Northern European conditions. The importance of on-farm biogas plants is growing in Finland. According to the statistics, the energy production from on-farm biogas plants increased by 31 per cent in 2023 compared to 2022 (StatFin, 2024). Furthermore, the various agricultural residues (manure, excess grass, straw, other crop residues) are considered to have the largest potential as unused feedstock for biogas production in Finland.

2. Materials and methods

The study followed the ISO 14040 (2006a) and ISO 14044 (2006b) standards and accordingly performed the E-LCC according to the LCA steps: goal and scope definition (2.1), life cycle inventory analysis (2.2.), life cycle impact assessment (2.3.) and interpretation (section 3). Regarding the LCC, the third step (life cycle impact assessment) was replaced by an economic analysis computing all the costs from the initial cost to its end of life.

2.1. Goal and scope

2.1.1. Product system scenarios

Three different theoretical anaerobic digestion chain scenarios 1–3 (S1-S3) were compared. The main feedstock in all scenarios was pig slurry (16,000 t), which was produced and stored on the same farm where the biogas plant was located. The scenarios differed with respect to external co-feedstocks (3,500 t) and their procurement prior to feeding into the biogas reactor. Both co-feedstocks in S1 (dry fraction of pig slurry) and S2 (grass from uncultivated fields) from a neighbouring farm were chosen based on the increase in biogas production of the plant compared to pure slurry produced as feedstock. The co-feedstock in S3 (food industry side stream) was considered as a source of gate fee, which is often considered economically desirable (Luostarinen et al., 2011; Rasi et al., 2012).

2.1.2. System boundaries and functional unit (FU)

The climate impacts (kg CO_2eq) and costs (€) of the anaerobic digestion chain are analysed for two different end products produced simultaneously– that is, biogas for energy, and digestate for fertilisers. The FU used for biogas is MWh of net energy produced for farms own use (total production minus energy consumed in biogas plant own processes). The FU used for digestate is kg of soluble N contained in digestate. This is because the total (tot) N content of the digestate fractions consists of N bound to organic matter and soluble (sol) N (mainly in the form of NH_4). Only soluble N is directly available to plants, but organic N must first be released in the soil before it is available to plants. The climate impacts (kg CO_2eq) and costs (€) of the anaerobic digestion chain are allocated between biogas and digestate according to allocation rules defined more precisely in section 2.1.3.2. Accordingly, the allocated impacts and costs as results for biogas is presented as kg CO_2eq per MWh and for digestate as kg CO_2eq per kg of $N_{(sol)}$.

In case of reference product for biogas, the result for replaced and purchased electricity and heat is presented as kg CO_2eq per MWh of energy with same net production shares of electricity and heat as for biogas (see net production shares of electricity and heat for farm's own use in Appendix A, Table A3) to be comparable with biogas product. In case of reference product for digestate, the result for replaced and purchased mineral fertiliser is presented as kg CO_2eq per kg of tot N in mineral fertiliser which is comparable only to the soluble N content in digestate.

The system boundaries of biogas and digestate product in scenarios 1–3 (S1-S3) were from cradle-to-grave, considering the stages from raw material acquisition to anaerobic digestion and to the use of biogas at farm as electricity and heat and separated fractions of digestate application on the field as bio-based fertilisers on field (Fig. 1). The calculation considers whether the raw material (e.g. slurry) is classified as waste or residue with no market value. In this case, it is not subject to emissions from its production chain — in other words, it has zero production emissions and costs. Emissions and costs relating to the production of raw materials (e.g. slurry produced in piggeries) and their management up to the farm/factory gate (e.g. slurry stored prior to transportation to a biogas plant) are allocated to other farm/factory outputs (e.g. pork meat), where the raw materials are produced (see section 2.1.3.1 for more information). Additionally, the system boundaries for biogas and digestate products vary depending on whether the resulting digestate from the digestion process is considered to be waste with no value (allocation option 1) or a valuable by-product of biogas production (allocation option 2). Further information on the allocation rules can be found in section 2.1.3.

In all scenarios S1-S3, the system boundary for biogas product (as well as digestate product in “allocation option 2”) starts when the pig slurry, as the main feedstock in all scenarios S1-S3, is transported via a pipeline from the storage of the same farm directly to the biogas plant reactor. In the case of co-feedstock, S1 starts with the mechanical

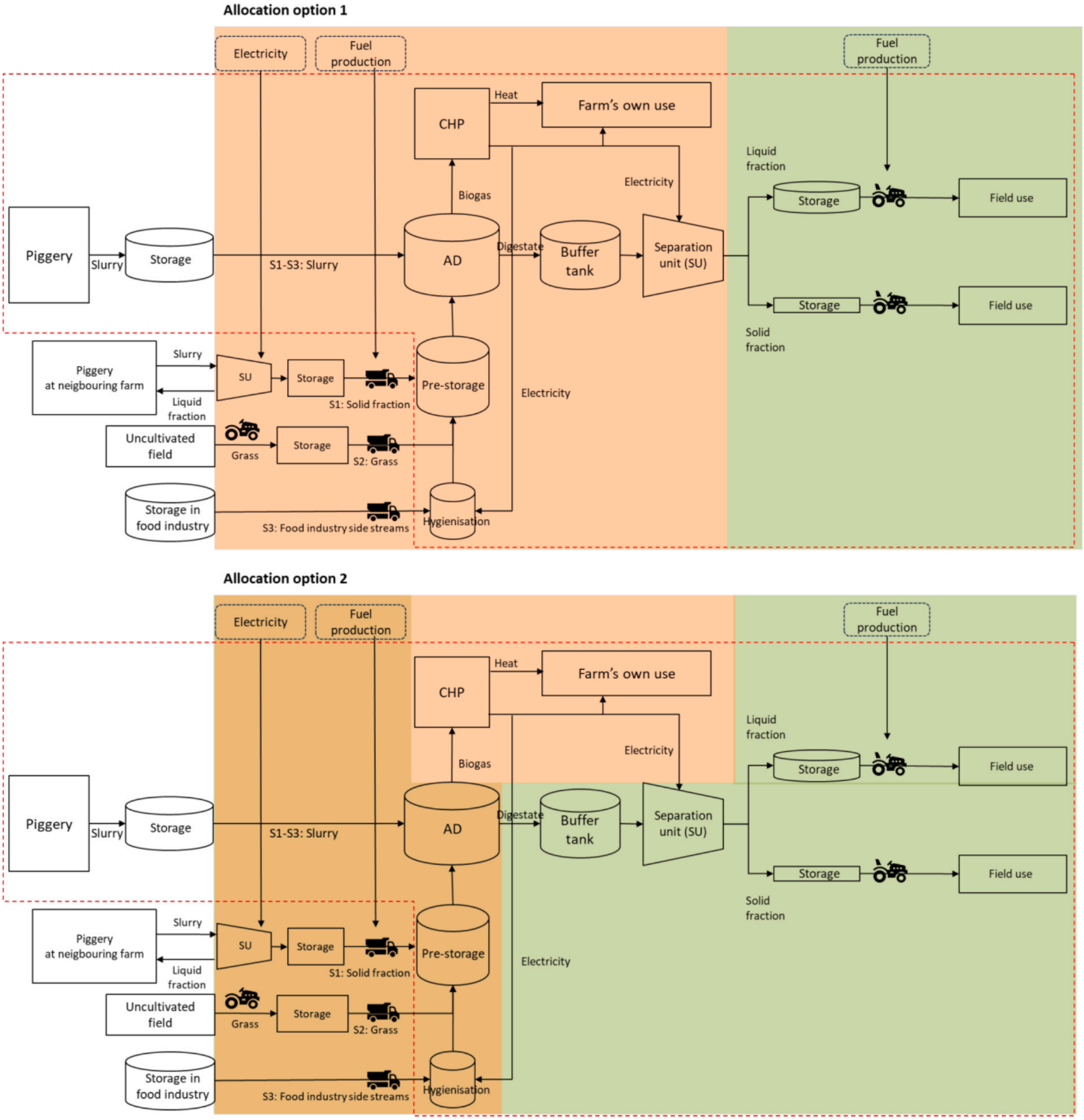

Fig. 1. The system boundary for biogas is presented in orange and for the digestate product in green. “Allocation option 1”is describing situation when digestate is seen as residual without market value and “Allocation option 2” when biogas and digestate are seen as co-products and both valorised with market value. System boundary for the biogas is presented in orange and for the digestate product in green. The area in “Allocation option 2” where the two colours overlap represents the part of the system of which emissions and costs are shared between biogas and digestate according to economic allocation rule. Co-feedstock is produced outside the farm system (dashed line in red) but acquisition emissions and costs are included in the system boundary of biogas and digestate according to economic allocation rules. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

separation of slurry into solid and liquid fractions on the neighbouring farm, after which the solid fraction is transported 20 km to the biogas plant by a semi-trailer truck. The liquid fraction is used as fertiliser on the neighbouring farm of origin and excluded from the system boundary. S2 starts when the originally mown grass left on the field is harvested from uncultivated fields, transported by a tractor and stored on the neighbouring farms and then again transported 20 km by a semi-trailer truck from the neighbouring farm to the biogas plant. S3 starts when the food industry side streams are transported 50 km by semi-trailer truck from the food industry.

After the transportation, co-feedstock is received in covered reception tank/pre-storage at the biogas plant. Only the food industry side stream as co-feedstock (S3) is hygienised (1 h at 70 °C) prior to the biogas process and the used energy produced by the biogas plant. Only the food industry's side stream is hygienised before the anaerobic digestion process in the reactor. After digestion in the reactor, the biogas is sent to the CHP unit to produce heat and electricity. Produced energy is partly used in the biogas plant's own operations (reactor heating and mixing, hygienisation and digestate separation) and partly for the farm's own needs to replace purchased electricity.

The digestate is briefly stored in a covered buffer tank with a short storage time as a conventional part of biogas plant operations before being separated into solid and liquid fractions using a screw press powered by electricity from the plant's CHP. The liquid fraction of the digestate is transported via a pipeline to the covered storage tank on the farm. The solid fraction is stored in a covered storage next to the biogas plant. Both fractions are stored for a maximum of 12 months (according to local regulations), and then transported 5 km and spread on the fields by tractor (12 m^3) in spring according to a phosphorus application rate of 14 kg P/ha (Nummela and Tuononen, 2009) – the average P value of clay soils in Finland is 8.03 mg/L (Eurofins Viljavuuspalvelu Oy, 2014).

In case for reference products, the mineral fertiliser was transported 200 km to the farm and 5 km to the field for application. The purchased electricity was from the grid and heat was produced at farm in a boiler.

2.1.3. *Allocation methods*

This study used economic allocation method to assess the impacts for feedstocks (see section 2.1.3.1.) and outputs (see section 2.1.3.2.), following the allocation hierarchy by ISO 14044 (2006) and EF method (EC, 2021).

2.1.3.1. Raw materials: feedstocks. When assessing the impacts of feedstock production, the economic allocation method was used because the avoidance of allocation through process subdivision or system expansion, was not seen to lead to one specific and comparable product. Also, feedstocks and their co-products have very different physical relationships and end uses in the market, and therefore allocation based on physical relationships would not be suitable (e.g. pig slurry and pork).

This study applied economic allocation for manure following the Environmental Footprint (EF) method (EC, 2021). According to this guidance, manure is regarded as residual (default option) without allocation of an upstream burden, if it is exported to another farm and does not have an economic value at the farm gate. The emissions from manure management up to farm gate are allocated to the other farm outputs (e.g. beef and milk) of the process in which feedstock is produced. Therefore, the emissions related to slurry production and conventional management practices up to farm gate are allocated to the other farm outputs (e.g. pork meat) of the process in which slurry is produced, and pig slurry without market value at farm gate is regarded as residual without allocation of an upstream burden (Table 1).

Solid fraction of slurry (S1) was considered having market value at neighbouring farm gate before being exported to a biogas plant. Since there was no market price considered for the liquid fraction of slurry, an allocation of the upstream burden (separation of slurry and storage of separated fractions) will only be used for the solid fraction. The separation of the slurry was not considered as a conventional management practice for slurry at the neighbouring farm, but was only done for the production of biogas, creating market value for the solid fraction, which was then exported to the biogas plant.

The grass from uncultivated fields (S2) is conventionally left on field but was considered to have a market value at the neighbouring farm gate before exported to the biogas plant because it needs to be harvested, transported and stored which creates value and therefore an allocation of the upstream burden was used.

In S3, the food industry side streams are considered as residual with no price on the market and no allocation of the upstream burden is considered. On contrary, gate fees to be paid for the biogas producer at factory gate.

2.1.3.2. End-products: biogas and digestate. In the case of biogas and digestate, the avoidance of allocation through process subdivision or system expansion, would not result in one specific and comparable product. Furthermore, biogas and digestate have very different physical relationships and end uses in the market, making allocation based on physical relationships inappropriate. Under the physical allocation rule, emissions are distributed disproportionately between biogas and digestate. More specifically, if emissions from the digestion chain are allocated according to calorific values, for instance, almost all emissions are allocated to biogas; if they are allocated according to mass, however, almost all emissions are allocated to digestate (Timonen et al., 2019). Economic allocation is commonly used if co-products have very different physical relationships and end uses in the market (Kyttä et al., 2022).

Following the same logic that economic allocation guidance for manure by EF method (EC, 2021), if digestate does not have an economic value at the plant gate, it is regarded as residual without allocation of an upstream burden. The emissions related to digestate management up to plant gate are allocated to the other plant outputs (e.g. energy) where digestate is produced (allocation option 1; Table 2). If digestate has an economic value at plant gate, emission of the upstream burden (from feedstock acquisition to the reactor gate) shall be allocated 62–70 % for biogas and 30–38 % digestate according to their respective productivity ratios in euros (allocation option 2, Table 2). The market value for net production of biogas and digestate is determined by the substitute products' unit prices (see Appendix B, Table B2) and presented in Table 2.

No allocation of emissions and costs was required further for electricity and heat produced from the biogas, nor for the separated solid and liquid fractions of the digestate, as it was assumed for simplicity that the total net production of biogas and digestate produced by the on-farm biogas plant (see more in Appendix A, Table A2, Table A3) is used on the same farm thus replacing purchased energy and mineral fertilisers.

Table 1
Allocation ratios (%) for inputs; feedstocks.

Feedstock type	Scenario	Market value	Exported	Allocation %	Upstream burden
Slurry	S1-S3	No	No	0 %	generation of slurry, storage
Dry fraction of slurry	S1	Yes	Yes	100 %	separation of slurry
Grass from uncultivated fields	S2	Yes	Yes	100 %	harvesting grass left on field, transportation, storage
Food industry side stream	S3	No (gate fees)	Yes	0 %	Food production

Table 2
Allocation ratios (%) for anaerobic digestion end-products; biogas and digestate.

End-product	Allocation option 1			Allocation option 2		
	S1	S2	S3	S1	S2	S3
Net bioenergy production (MWh/a)*	2,870	3,230	2,425	2,870	3,230	2,425
Digestate production (tonnes of soluble N/a)**	74,967	70,589	75,998	74,967	70,589	75,998
Biogas (€/a)	239,755	269,348	203,051	239,755	269,348	203,051
Digestate (€/a)	0	0	0	123,696	116.472	125,397
Biogas (allocation ratio)	100 %	100 %	100 %	66 %	70 %	62 %
Digestate (allocation ratio)	0 %	0 %	0 %	34 %	30 %	38 %

*Presented more specifically in Appendixes Table A3.
*Presented more specifically in Appendixes Table A2.

2.2. Life cycle inventory (LCI)

2.2.1. Data on feedstocks, energy and transportation

The mass and nitrogen balance of scenarios S1-S3 is presented in Fig. 2. More detailed information on scenarios' feedstocks, energy and transportation is presented, if not already in Fig. 4 and section 2.1.2, in the Appendix A. The characteristics of the feedstocks is presented in Table A1. The different feedstock mixes and their procurement are explained specifically in Appendix A and the mass, nutrient balances in Table A.2. The amounts of energy produced and consumed at the biogas plant and farm in each scenario are presented in Table A.3. The mass, nutrient and energy balances were based on calculations with the Biogas tool (2023). The transportation distances are shown in Table A.4 and estimates of tractor fuel consumption in Table A.5.

2.2.2. Emission factors

The assessment was conducted in using most representative emissions data available from literature (Table 3).

In case of electricity mix, this study used Ecoinvent model for country-specific electricity emission and the factors were updated with the 2022 electricity country mix for Finland.

The emissions associated with the biogas process were calculated using emission factors for advanced practices, which were collected by Lehtoranta et al. (2024) from the scientific literature on measured CH_4 emissions (see Appendix A, Table A6). More specifically, the average CH_4 emissions were selected, and the impact of choosing minimum and maximum values on the final life cycle of biogas and digestate was examined.

The CHP unit's N_2O emissions were assumed to be nearly zero (0.006 g N_2O/kWh, Liebetrau et al., 2013) due to the relatively low N content of the feedstocks and advanced operational practices (Appendix A, Table A2). Therefore, no ammonia was considered to end up in the biogas, though small amounts of N_2O emissions from the CHP unit occur due to high-temperature combustion, in which N from the air oxidizes. However, as N_2O is a significant greenhouse gas with high global warming potential, and due to general uncertainties in the ammonia content of biogas, sensitivity analyses were conducted in Appendix C (Fig. C1, Fig. C2) to show how life-cycle impacts would increase if biogas was assumed to consist of high amounts of ammonia, and the average and maximum values of N_2O emissions from a CHP unit were assumed, according to the literature (Liebetrau et al., 2013, Table 3).

Assessing emissions from the field use of digestate and mineral fertilizer, IPCC (2019) emission factors were used. Assessment of soil carbon (C) according to EC (2021) is considering only permanent C storage

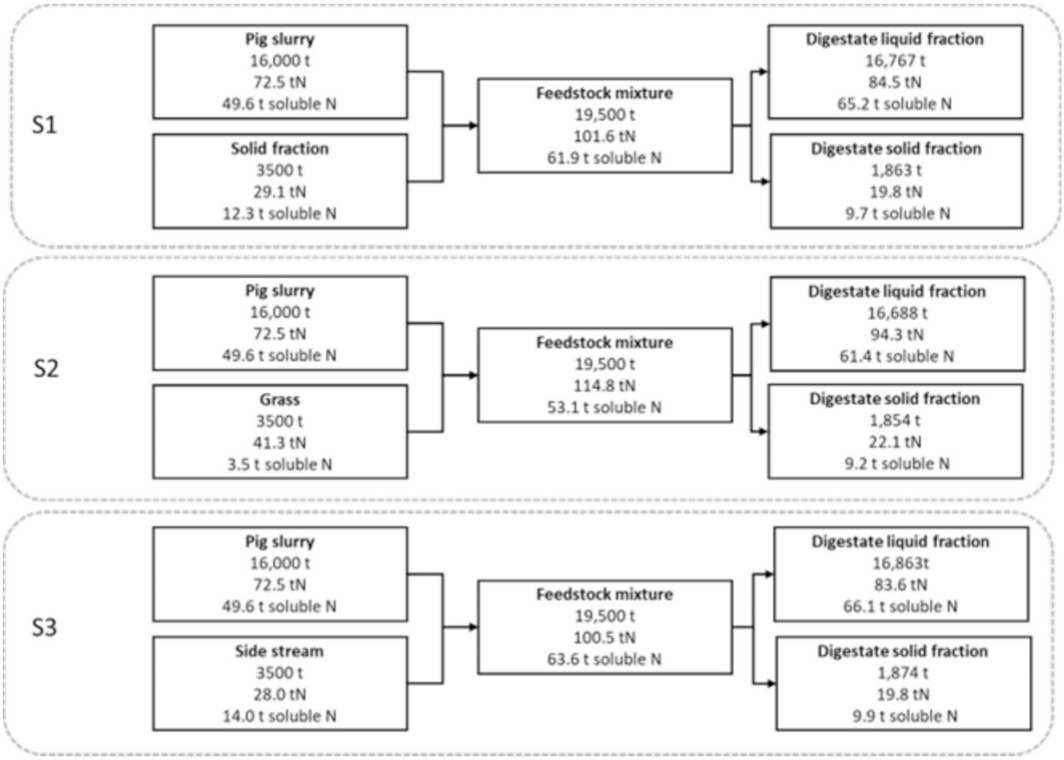

Fig. 2. Mass and nitrogen balance in scenarios S1-S3.

Table 3
Sources for main emission factors and assessment methods used in the assessment.

Chain stage	Emission factors inputs and utilisation of outputs	Value	Source
Feedstock handling	S1: Separation of slurry	168 kg CO_2eq/MWh	Ecoinvent, country-specific electricity emission factors updated with the 2022 electricity country mix for Finland
	S2: grass harvesting	0.029 kg CO_2eq/kg dry matter yield	Rasi et al. (2020), field machinery emissions incl. fuel production
Biogas plant: Operation and maintenance	Mixing and feeding unit	0.01–0.03 % of CH_4 produced	Lehtoranta et al. (2024)*
	Leakages from plant structures	0.017–0.1 % of CH_4 produced	Lehtoranta et al. (2024)*
	Pressure release valves	0.04–0.73 % of CH_4 produced	Lehtoranta et al. (2024)*
	Maintenance	0.1–0.5 % of CH_4 produced	Lehtoranta et al. (2024)*
	CHP emissions	0.17–0.4 % of CH_4 produced 0.006–0.345 g N_2O/kWh electricity produced	Lehtoranta et al. (2024)* Liebetrau et al. (2013)
Digestate field use	Direct N_2O emissions	0.006 kg N_2O-N/kg N	IPCC (2019) coefficients for organic amendments in wet climates
	Indirect N_2O emissions from ammonia	0.21 kg N_2O-N/kg N deposited	IPCC (2019) coefficients for an organic N fertiliser
	Indirect N_2O emissions from leaching of total N	0.011 kg N_2O-N/kg of N leaching	IPCC (2019) coefficients for an organic N fertiliser
Carbon sequestration	Digestate	0.11–0.15 kg CO_2eq/kg C in digestate (3–4 % of total C in digestate)	Tampio et al. (2024)
Purchased electricity	An average electricity grid mix in Finland	168 kg CO_2eq/MWh	Ecoinvent, country-specific electricity emission modelling modified with the 2022 electricity country mix for Finland
Purchased heat	Purchased wood chips burned in a boiler on the farm for heat	25.2 kg CO_2eq/MWh	Scrucca et al. (2023) (converted from an average value of 7 gCO_2eq/MJ)
Purchased mineral fertiliser	Nitrogen fertiliser production	5.3 kg CO_2eq/kg N	DNV GL 2015, (N25%, *P*3%)
Mineral fertiliser field use	Direct N_2O emissions	0.016 kg N_2O-N/kg N of fertiliser	IPCC (2019), coefficients for synthetic fertilisers in wet climates.
	Indirect N_2O emissions from ammonia	0.11 kg N_2O-N/kg N	IPCC (2019), coefficients for synthetic fertilisers
	Indirect N_2O emissions from leaching of N	0.011 kg N_2O-N/kg N	IPCC (2019), coefficients for synthetic fertilisers

*Collected by Lehtoranta et al. (2024) and presented more in detail in Appendix A Table A6.

as biogenic C emitted later than 100 years after its uptake. According to Tampio et al. (2024) 3–4 % of total C in digestate (produced from manure) is considered long term C. Theoretically, a kilo of C binds 3.667 kg of CO_2 (mass ratio of C to CO_2 is 44.01(g/mol)/12.01(g/mol)) and therefore, the permanent binding potential of digestate is 0.11–0.15 kg CO_2eq/kg C (tot) in digestate (Table 3).

The emission factors for transport and for fossil diesel used in working machines are based on the LIPASTO database of VTT Technical Research Centre of Finland Ltd (2017). By using LIPASTO (2017) database, the load factor of vehicles can be modelled more precisely, and it was modelled with a full load one way and an empty return. The coefficients assume 11 % of the fuel is renewable. NMVOCs were added to the Lipasto coefficients and inventoried as HC-Methane and distributed to the gases according to the EEA (2019). Diesel acquisition was also included by using Ecoinvent data (Diesel, low-sulphur {Europe without Switzerland}| market for | Cut-off, Ecoinvent).Vehicle service and road maintenance were not included.

2.3. *Impact assessment methods*

This study was using E-LCC method considering climate impacts and costs associated with the life cycle of a biogas and digestate product that are directly covered by all of the actors in the product life cycle (Hunkeler et al., 2008), for example the supplier of feedstocks, manufacturer of biogas, user or consumer, and/or the End of Life (EoL) actor. In E-LCC, the LCA use the same system boundaries and the same FU as the LCC (Norris, 2001, Carlsson Reich, 2005, Hunkeler et al., 2008; Swarr et al., 2011).

For the assessment of climate emissions in LCA, the Environmental Footprint 3.1. method (Bassi et al., 2023) and the Simapro calculation system were used. The life cycle inventory was characterised using factors for fossil $CO_2 = 1$, biogenic $CO_2 = 0$, fossil $CH_4 = 29.8$, biogenic $CH_4 = 27$ and $N_2O = 298$, presented as carbon dioxide equivalents, kg CO_2eq.

Life cycle costs of biogas and digestate product, produced in 2023, was considering annual costs from year 2023 in euros from feedstock acquisition, energy and fuel consumption, transportation and working machinery. In addition, annual cost of the biogas plant investment (annuity cost) is calculated from the estimated lifetime and interest rate (4 %) part of biogas plant profitability calculations presented in Appendix B. More specifically, investment costs were indexed, discounted and presented in a net present value (NPV) context as well as divided into annual costs and further targeted per FU to make each option comparable with each other from a life cycle perspective. An investment support of 50 % was also considered, which is the existing practice in Finland at the time of writing. The life cycle of a biogas plant in all scenario S1-S3 was estimated based on profitability calculations considering investment annuity, variable costs, revenue, gross margin, annual profit, payback time and is presented in Appendix B in more detail.

Furthermore, these abovementioned cost calculations were complemented with the LCA results next to them, enabling comparative assessments to be made between environmental and cost impacts. As a final step, this study completed the E-LCC assessment by monetising externalities and internalising them, that is calculating the climate impacts (LCA results) that will probably be manifested as actual direct costs, for example through European emission allowances (see Appendixes A, Table A6) to the relevant agents of the study (the farm as a biogas producer) in the near future, that is in a time perspective relevant for the decision-making process (Hunkeler et al., 2008, Martinez-Sanchez et al., 2015; De Menna et al., 2016). The E-LCC internalising monetised climate impacts were presented separately from the LCA results preventing risks for double counting of environmental impacts as emissions and costs (Luo et al., 2009, Martinez-Sanchez et al., 2015 and (Lu and El Hanandeh, 2017)).

The E-LCC did not provide feasibility or technological assessments

(TEA). However, the feasibility calculations for biogas plant were working as a base for selecting potential scenarios (see more in 2.1.1. and Appendix B).

3. Results and discussion

3.1. LCA of biogas and digestate

When a market value was created for the digestate product (allocation scenario 2), the carbon footprint of the biogas decreased by 24–29 % in all three scenarios (S1–S3) (Fig. 3). This is because, unlike in ’allocation option 1′, where the total emissions from the digestate chain are allocated to biogas, in ’allocation option 2′ part of these emissions are allocated to digestate according to productivity shares (see Fig. 1, Table 2, Figs. A1-A3 in Appendix A). Correspondingly, the carbon footprint of the digestate fractions increased, but only by 6–8 % (Fig. 4) and the estimated C sequestration potential of digestate fully compensates for this by reducing digestate life cycle emissions by up to 10–14 %. In all scenarios (S1-S3) and allocation options 1–2, the life cycle climate impacts of the biogas product (Fig. 3) and the separated digestate fractions (Fig. 4) were lower than those of purchased energy and mineral fertilisers. The results for biogas and digestate vary between scenarios depending on which feedstock mixture was used affecting the energy yield potential and the nutrient content of the digestate.

3.1.1. LCA of biogas product

In both allocation options 1 and 2 (Fig. 3), the climate impacts were higher for purchased energy (92.8 kg CO_2eq/MWh) than for biogas products in S1-S3 (26.8–44.4 kg CO_2eq/MWh) which is in line with results of previous studies where the anaerobic digestion led to emission reductions by replacing fossil fuels (Timonen et al., 2019, González et al., 2020; Lehtoranta et al., 2024). This is due to the fact that utilising biodegradable waste and residual streams as feedstocks in biogas production with zero market value, would cause zero emissions regarding raw material production. In addition, the transportation of main feedstock (pig slurry) was causing no emissions or costs because it was transported by a pipeline (Ghafoori et al., 2007). Moreover, it was interesting to note that producing biogas on one’s own farm resulted in significantly lower emissions, even though only 13 % of the purchased electricity was produced using fossil fuels; the rest came from renewable sources, nuclear power, or an unknown source due to a lack of data. Also, the purchased heat is assumed to be produced from wood chips, which are considered renewable with low procurement emissions according to emission estimate by Scrucca et al., (2023) used in this study (see Table 3). Sensitivity analyses showed in Fig. 3 that using the minimum and maximum CH_4 emission factors for advanced practices (see Table 3, Lehtoranta et al. 2024) did not result in emissions exceeding those of purchased electricity and heat. Despite the N_2O emissions from the CHP unit were assumed to be minimum in Fig. 3 due to advanced practices and no ammonia content of biogas, this study also demonstrated their significant contribution to the sustainability of the biogas product (see Appendix C, Fig. C1, Fig. C2). For example, the life cycle climate impacts of biogas (S1-S3) exceeded those of the reference product if an average or high level N_2O emissions from the CHP unit were assumed for example due to the ammonia contained in biogas (Liebetrau et al., 2013). In case of mineral fertiliser, there was only slight variation in life cycle impacts when the share of purchased electricity and heat per MWh changed in line with the net production shares of the biogas product being replaced in S1-S3 (see section 2.1.2 and Table A3).

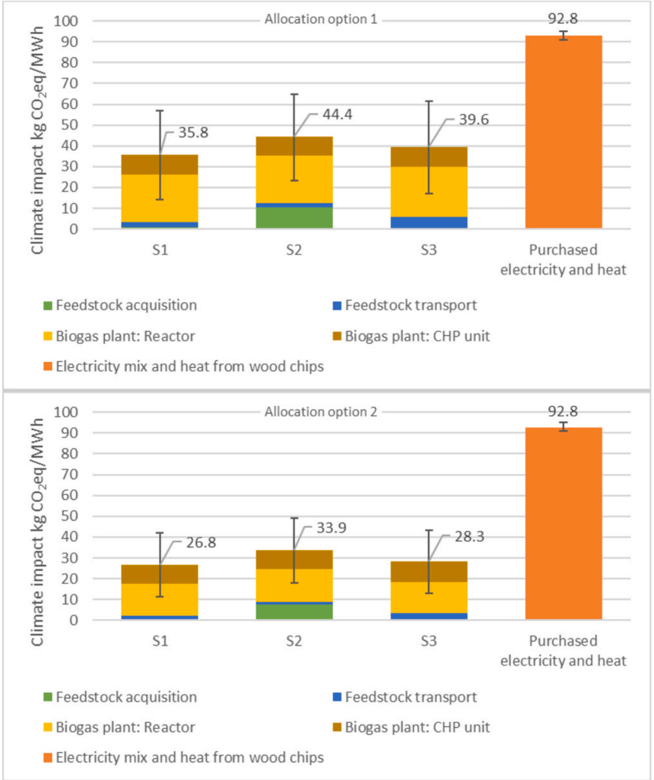

Fig. 3. Life cycle climate impacts (kg CO_2eq) of biogas produced (MWh) and purchased energy (MWh) to be substituted as reference. The line in scenarios S1-S3 illustrates how the outcome of the climate impact of the life cycle varies when choosing minimum or maximum range of CH_4 emissions with advanced practices (see Table 3). The line in reference product describes how the outcome varies when considering small differences in shares of purchased electricity and heat per MWh (see section 2.1.2.; Table A3).

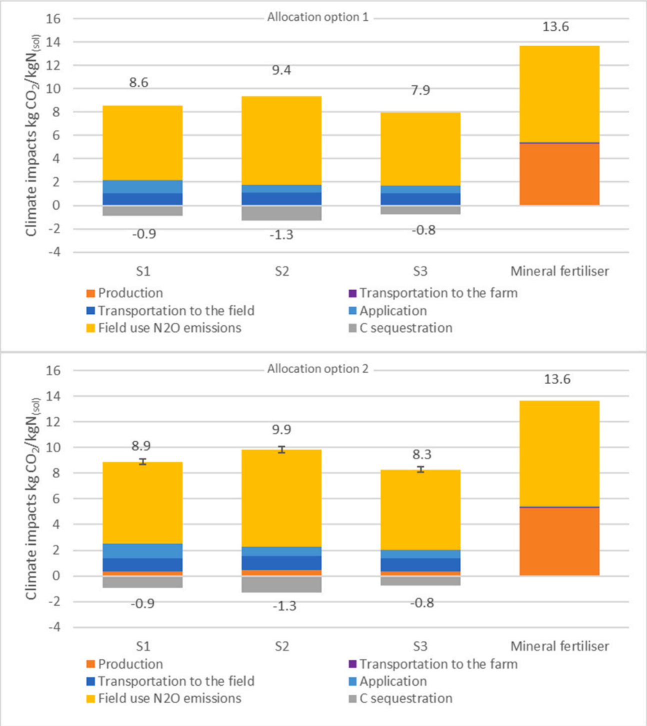

Fig. 4. Life cycle climate impacts (CO_2eq) for separated fractions of the digestate produced (kg soluble N) via anaerobic digestion (S1-S3) and mineral fertiliser (kg $N_{(sol)}$) as reference. The line in scenarios S1-S3 illustrates how the outcome of the climate impact of the life cycle varies when choosing minimum or maximum range of CH_4 emissions of biogas plant with advanced practices (see Table 3).

When looking more closely at biogas scenarios, the lowest emissions were for the biogas product utilising the dry fraction of slurry as co-feedstock (S1 in Fig. 3) because of the shorter transportation distance than in S3 and less steps for co-feedstock processing than in S2. In all three biogas scenarios S1-S3, the highest source for emissions was from biogas plant reactor and assumed to emerge mainly from the CH_4 leaks from pressure release valves and maintenance operations (see more Table 3, Liebetrau et al., 2013, Luostarinen et al., 2023; Lehtoranta et al., 2024). The second highest source of emissions was from CHP unit's CH_4 with minimum share of N_2O emissions (see more Table 2, Liebetrau et al. 2013; Luostarinen et al. 2023). In general, however, the climate impacts of biogas plant were still quite low as the theoretical biogas plant evaluated in this study was assumed to be operated with advanced practices, such as frequent maintenance to avoid malfunctions and using equipment with low emission leakage risk (Luostarinen et al., 2023; Lehtoranta et al., 2024). No more biogas was assumed to be produced than can be directed to the CHP unit, so the plant was not considered to generate flaring thus resulting in emissions (Luostarinen et al., 2023). Due to the covered storage conditions, no emissions were generated during storage of digestate which is usually one of the largest sources for methane emissions according to Liebetrau et al., (2013). Building infrastructure was not expected to generate emissions, but rather they were perceived as insignificant (Arias et al., 2020).

3.1.2. LCA of digestate fractions

The carbon footprint of the digestate was lower (7.9–9.9 kg $CO_2eq/kgN_{(sol)}$) than that of the mineral fertiliser (13.6 kg $CO_2eq/kgN_{(sol)}$) in both allocation options 1–2 (see Fig. 4). This is because the production emissions of mineral fertiliser (emissions from 'cradle-to-gate') are significantly higher (5.3 kg $CO_2eq/kgN_{(sol)}$), DNV GL, 2015) than production emissions of digestate which are considered zero in 'allocation option 1′ and increase somewhat (0.3–0.5 kg $CO_2eq/kgN_{(sol)}$) in 'allocation option 2′. Despite there was no specific data available on different emission sources of mineral fertiliser production (DNV GL, 2015), fossil ammonia is known to be an important raw material in the production of N mineral fertilisers. Also, Timonen et al. (2019), González et al. (2020), Balcioglu et al. (2022) and Lehtoranta et al. (2024) has showed that recycling nutrients by utilising and decreasing waste or residues in anaerobic digestion led to emission reductions, thus replacing mineral fertilisers. Sensitivity analyses for biogas plant with the minimum and maximum CH_4 emission factors for advanced practices (see Table 3; Lehtoranta et al. 2024) did not have significant impact on production emissions. One of the reasons is that biogas plant is limited outside the system boundaries of digestate product S1-S3 in 'allocation option 1′ and only biogas plant reactor is included in 'allocation option 2′ while CHP unit is excluded (see Fig. 1, Figs. A1-A3 in Appendix A).

Considering field use, mineral fertilizer was subject to somewhat higher field use N_2O emissions than digestate. Furthermore, it is possible that N_2O emissions of digestate are even lower because the current study applied coefficients for organic fertilisers that are based on manure studies (IPCC, 2019) and might not adequately reflect realistic N_2O emissions from N in the digestate solid and liquid fractions. Mineral fertilizer on the other hand is based on coefficients for synthetic fertilisers (IPCC, 2019). Digestate is also considered to have C sequestration potential when applied on field. This study estimated that the C sequestration potential of digestate reduces its life-cycle climate impact by 10–14 %. In Fig. 4, carbon sequestration potential of digestate product as removals (−1.3-(−0.8) kg $CO_2eq/kgN_{(sol)}$) are reported separately from emissions ((7.9–10.1 kg $CO_2eq/ kgN_{(sol)}$) due to uncertainties in meeting the definition of permanent carbon stock as instructed in ISO 14067 and EC (2021). Assuming that the permanent C content of pig slurry-based digestate is 3–4 % of the total C content, as is the case for cow manure-based digestate (Tampio et al., 2024), the carbon footprint of the digestate would decrease to 7.1–8.4 kg $CO_2eq/ kgN_{(sol)}$ in 'allocation scenario 1′ and to 7.7–8.8 kg $CO_2eq/kgN_{(sol)}$ in 'allocation scenario 2′.

Transportation and application emissions of the digestate are not the most significant emission source when compared to N_2O emissions from field use, but they are significantly higher than emissions from mineral fertilizer transportation or application (Fig. 4). This is because the digestate products have a significantly lower nutrient content than the mineral fertiliser (see Table A4), and it needs to be applied more in volume, which then consumes more fuel and generates costs and emissions. The emissions of fuel consumption during the application of the digestate product in S1-S3 can be reduced through further processing the digestate products by increasing the N content relative to the P content.

The lowest emissions were for the digestate product utilising the food industry side streams as co-feedstock (S3) because of the lower application emissions than S1 and lower field use N_2O emissions than S2. More specifically, S1 contains less soluble N than S3, resulting in a higher application requirement and fuel consumption. Also, S2 contains more insoluble N relative to soluble N than S3 leading to higher N_2O emissions from field use. According to the sensitivity analysis performed on field transportation distances for digestate, scenario 3 (S3) allows an increase in transportation distance of up to 37 km, S1 up to 33 km, and S2 up to 22 km, provided that the life cycle impacts of the reference product are not exceeded (Fig. 4).

3.2. E-LCC of biogas and digestate

The results show that environmental life cycle cost of the biogas (S1-S3) decreased 24–30 % when digestate fractions were seen as valuable co-products in 'allocation option 2′ (Fig. 5). The decrease in the cost of biogas products is due to the fact that part of the life cycle costs from the anaerobic digestion chain were assigned not only for biogas but to digestate with certain allocated shares (see Table 2). Respectively, the environmental life cycle cost of the digestate fractions (S1-S3) increased 36–47 % (Fig. 6) in 'allocation option 1′ but did not exceed the life cycle costs for replaced mineral fertiliser purchased outside farm.

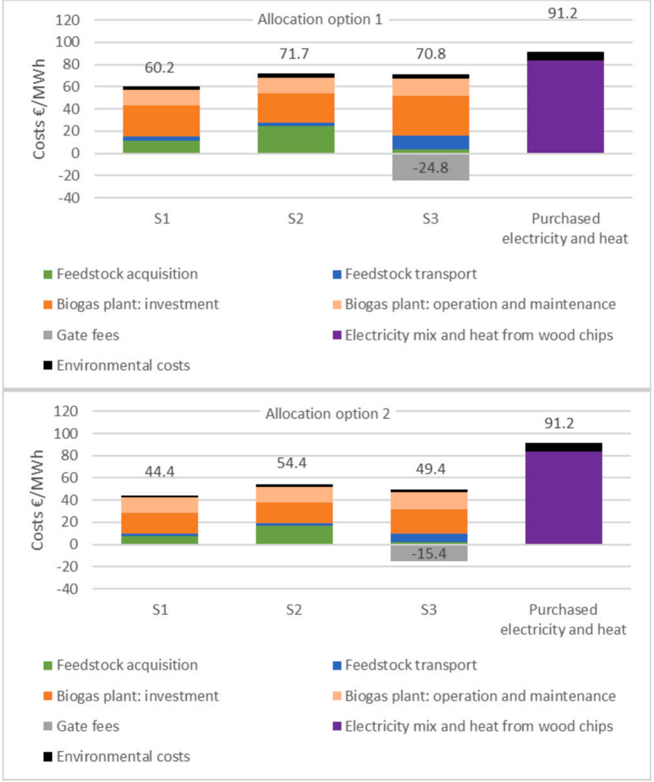

Fig. 5. The environmental life cycle costing (€) of biogas produced (MWh) via anaerobic digestion (S1-S3) and purchased energy (MWh) to be substituted as reference. Conventional costs (feedstock acquisition, transportation, biogas plant and gate fees) are complemented with environmental costs in black.

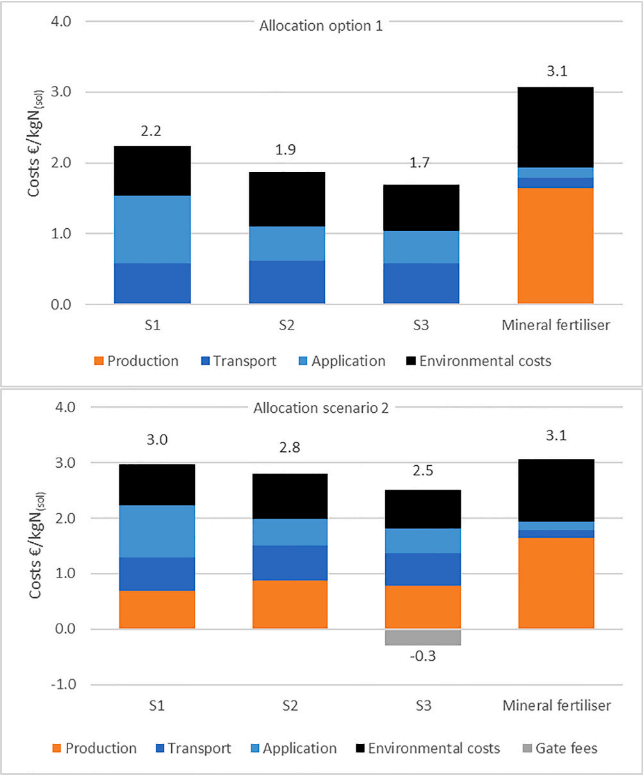

Fig. 6. Environmental life cycle costs (€) for separated fractions of the digestate produced ($kgN_{(sol)}$) via anaerobic digestion (S1-S3) and mineral fertiliser ($kgN_{(sol)}$) as reference. Conventional costs (production, transport, application and gate fees) are complemented with environmental costs in black.

3.2.1. *E-LCC of biogas product*

As also with life-cycle environmental impacts in Fig. 3, in both allocation options 1 and 2 the life cycle costs were higher for purchased energy and for biogas products in S1-S3 (Fig. 5). Cost savings, as well as emission reduction, in biogas production are achieved when biodegradable waste and residual streams, with zero market value and emissions were used as feedstocks in biogas production in S1-S3. In addition, the main feedstock (pig slurry) was obtained from the farm itself via pipeline causing no transportation costs nor emissions. The lowest life cycle cost, as well as climate impacts, was for the biogas product utilising the dry fraction of slurry as co-feedstock (S1) mainly because of a shorter transportation distance of co-feedstock to the biogas plant (20 km) compared to S3 (50 km) and less need for co-feedstock processing (only separation of slurry) than in S2 (harvesting, harvesting and storage of grass). In addition, the scenario S3 utilising food waste as co-feedstock (S3) had higher investment costs due hygienisation unit.

By including returns (gate fees) and replacement benefits of purchased energy to the calculations, the NPV was positive for each biogas scenario S1-S3 with the highest value being for S3 in allocation option 1 (45.2 €/MWh) and even higher in allocation option 2 (57.2 €/MWh). Monetising climate impacts as environmental costs, increased the NPV of biogas. This is because environmental costs of purchased energy increased more than that of biogas, due to the higher climate impacts of the purchased energy presented in Fig. 3.

A chain-specific analysis revealed that the majority of the costs (Fig. 5) and emissions (Fig. 3) associated with a biogas product originate from the biogas plant itself (reactor and CHP unit). The highest source for costs in all three scenarios S1-S3 emerged from the investment costs of biogas plant infrastructure. This is the case even though 50 % investment subsidy was already considered in the calculations and deducted from the incurred costs. The second highest source for costs was from operation and maintenance of biogas plant. On contrary, building infrastructure was not expected to generate emissions, but rather they were perceived as insignificant (Arias et al., 2020). However, leakages from plant structures were one of the most significant emissions sources during operation and maintenance of biogas plant (see more in Table 3). All in all, costs from investment and operation and maintenance in biogas plant were significant because of advanced operational practices which on the other hand led to lower emissions in comparison to conventional practices (Lehtoranta et al. 2024). In other words, without the higher investment costs to advanced practices the emissions would be even higher. Investments in new technologies for energy production could lead to further savings in costs as well in addition to emissions (Ristimäki et al., 2013) e.g. through lower maintenance costs in new technologies or lower environmental costs.

3.2.2. *E-LCC of digestate fractions*

As shown in Figs. 5 and 6, the results indicated that both life cycle climate impacts and costs of the digestate fractions in all three scenarios (S1–S3) were lower than the life cycle costs of purchased mineral fertiliser. One of the main reasons in ‘allocation option 1′ is that the digestate is considered as residual product with zero market value and emissions in relation to significant production emissions and purchase cost of mineral fertiliser. In ‘allocation option 2′ on the other hand, 30–38 % of the emissions and costs of digestate production are borne by the digestate product, while the rest being covered by the biogas product (see Fig. 1, Table 2, Figs. A1-A3 in Appendix A). The lowest life cycle cost was for S3 which produced the highest amount of soluble N considered as the functionality of the digestate product ie. costs are targeted per higher amount of product. There was no significant differences in transportation costs of digestate (S1-S3) to the field but application costs were significantly higher in case of S1 than in S2, S3 and mineral fertilizer. The digestate fractions in S1 have a higher P content in relation to soluble N and needs to be applied more in volume, which then consumes more fuel and generates costs and emissions. This is because P is one of the most common limiting factors in plant production systems (Elhawat et al., 2023).

Replacing mineral fertiliser with digestate fractions produced on the farm resulted in cost savings. When including returns and replacement benefits of mineral fertiliser to the costs of digestate fractions, the NPV was positive for each scenario S1-S3 with the highest positive value being for S3 in allocation option 1 (1.4 €/MWh) and slightly lower in allocation option 2 (0.9 €/MWh). Since the products are used to replace purchased mineral fertiliser rather than being sold on, NPV reflects same as the cost savings. Monetising climate impacts as environmental costs increases the NPV of digestate because environmental costs of mineral fertiliser increase more than that of digestate, due to the higher climate impacts of the mineral fertiliser (see Fig. 6).

A chain-specific analysis revealed that the majority of the costs (Fig. 6) and emissions (Fig. 4) associated with a digestate fractions S1-S3 varied between allocation option 1 and 2. Application costs (Fig. 6) and emissions (Fig. 4) of the digestate fractions are significantly higher in S1. Both transportation and application costs (Fig. 6) and emissions (Fig. 4) of the digestate are higher in all digestate scenarios S1-S3 when compared to a mineral fertiliser (in line with the findings of a study by Balcioglu et al. (2022)). This is because the digestate products have a significantly lower nutrient content than the mineral fertiliser (Table A4), and it needs to be applied more in volume, which then consumes more fuel and generates costs and emissions. A sensitivity analysis showed that the digestate fractions in S1 could not be distributed over a distance longer than 5 km without the costs rising to the same level as those of mineral fertiliser (Fig. 6). For scenario S2, the distance could be increased to 7 km, and for scenario S3, to 9 km without being equal or higher than the life cycle costs of mineral fertiliser. If the gate fee is also considered as income, the maximum distance for scenario S3 is 12 km.

Both emissions and costs of fuel consumption during the application of the digestate product in S1-S3 can be reduced even more through further processing the digestate products by increasing the N content relative to the P content. Digestate treatments, such as composting, drying or evaporation of the digestate should be considered (Balcioglu et al., 2022) or by adding plant growth-promoting rhizobacteria (González et al., 2020), if profitable for the investigated small biogas plants (Herbes et al., 2020). Another potential option is to use biofuel instead of diesel or better application technologies (Balcioglu et al., 2022), for example using low-emission vehicles and improving vehicle efficiency (Krause et al., 2020).

4. Conclusions

This study emphasises the importance of valorising the digestate to reduce the carbon footprint of biogas as co-product. Conversely, the carbon footprint of digestate increases, but is offset by the carbon sequestration potential when applied in soil. Environmental and economic benefits can be achieved by producing energy and fertiliser products from biodegradable waste and residuals in a typical Northern European on-farm biogas plant, replacing the purchased energy mix and mineral fertiliser. Monetising climate impacts increased the already positive net present value of biogas and digestate products. Product-specific calculations revealed different hot spots for biogas and digestate product for targeted improvements in sustainability.

CRediT authorship contribution statement

Karetta Vikki: Writing – original draft, Visualization, Software, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Elina Tampio:** Writing – original draft, Visualization, Resources, Investigation, Data curation. **Erika Winquist:** Writing – review & editing, Software, Resources, Project administration, Investigation, Funding acquisition, Data curation. **Merja Saarinen:** Writing – review & editing, Supervision, Methodology. **Sari Luostarinen:** Writing – review & editing, Supervision.

Declaration of competing interest

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

Acknowledgements

This study was conducted as part of the Biokiertomoduuli project funded by the Development Fund for Agriculture and Forestry (Makera). For their assistance, we would especially like to thank Ville Pyykönen (Natural Resources Institute Finland (Luke)).

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.biortech.2025.133401.

Data availability

Data will be made available on request.

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