Nutri2Cycle
Nurturing the Circular Economy

Nutri2Cycle

D.3.4 Environmental and Social Life Cycle Assessment of Selected Innovations

Deliverable: Environmental and Social Life Cycle Assessment of Selected Innovations

Author(s): Beyers, M.[1], Bruun, S.[1], Jensen, L.S.[1]; Ravi, R.[2], Akyol, Ç.[2], Brienza, C.[2], Meers, E.[2]; Regelink, I.[3], Lesschen, J.P.[3], Duan, Y.F.[3], Groenendijk, P.[3]; Cerrillo, M.[4], Pereira, E.[4], Ballega A.[4], Montemayor, E.[4], Antón, A.[4], Bonmatí, A.[4]; Hajdu S.[5], Coudron, C.[6], Devlamynck, R.[6]; Corona, F.[7], Verdugo, F.[7]; Ashekuzzaman, S.M.[8], Forrestal, P.J.[8]

[1]University of Copenhagen (Denmark), [2]Ghent University (Belgium), [3]Wageningen University & Research (The Netherlands), [4]IRTA (Spain), [5]SOLTUB Ltd. (Hungary), [6]Inagro (Belgium), [7]CARTIF (Spain), [8]TEAGASC (Ireland)

Quality review: Final version (ver 3.)

Date: 30/09/2022 [revised ver 2 15/12/2023 and ver 3. 05/03/2024]

Grant Agreement N°: 773682

Starting Date: 01/10/2018

Duration: 60 months

Coordinator: Prof. Erik Meers, Ghent University

Contact details: Erik.meers@ugent.be

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 773682.

Table of Contents

List of Figures

List of tables

Abbreviations

Al-DPS	Aluminium-precipitated sludge
AP	Acidification terrestrial and freshwater (EF)
APR	Alternative P recovery
ATC	Atlantic-central (climatic zone)
ATW	Accidents at work
BE	Belgium
BOD	Biological oxygen demand
BSF	Black soldier fly (*Hermetia illucens*)
Ca-DPS	Calcium-precipitated lime-stabilised sludge
CAN	Calcium ammonium nitrate
CCP	Climate change potential (EF)
CHL	Child labour
CMS	Certified environmental Management System
COD	Chemical oxygen demand
COR	Public sector corruption
CTW	Continental-west (climatic zone)
D.x.x.	Deliverable (here: project reports to the EU)
DIR	Dashboard indicators
DK	Denmark
DM	Dry matter
DWC	Drinking water coverage
EBPR	Enhanced biological phosphorus removal
ECO	Contribution of the sector to economic development
EI	Environmental impact
EIns	Environmental indicator study
eLCA	Environmental life cycle assessment
EOE	Expenditures on education
ERP	Resource use, energy carriers (EF)
ES	Spain
EU	European Union
EvS	Evaporator systems
FAB	Freedom of association and collective bargaining
FCP	Fair competition
FEP	Eutrophication freshwater (EF)
FOL	Forced labour
FPR	European Fertilising Product Regulation
FSY	Fair salary
FTP	Ecotoxicity freshwater potential (EF)
FU	Functional unit
GEW	Gender wage gap
GHG	Greenhouse gas emissions
GZV	Groot Zevert Vergisting
HCP	Cancer human health effects (EF)
HEE	Health expenditure
HNP	Non-cancer human health effects (EF)
HPBE	High potential of beneficial effect
HPHE	High potential harmful effect
IE	Indifferent effect

ILL	Illiteracy
IMS	International migrant stock
INR	Indigenous rights
IRP	Ionising radiation, HH (EF)
LCA	Life cycle assessment
LCI	Life cycle inventory
LEB	Life expectance at birth
LF	Liquid fraction
LL	Longlist (solution)
LUP	Land use potential (EF)
MC	Mineral concentrate
MEP	Eutrophication marine (EF)
MF	Membrane filtration
MIG	Migration
MLF	Men in the sectoral labour force
MRP	Resource use, mineral and metals (EF)
N	Nitrogen
NDN	Nitrification-denitrification
NDVI	Normalised difference vegetation index
NFRV	Nitrogen fertiliser replacement value
NIRS	Near-infra red sensor
NL	the Netherlands
NRR	Nutrient reuse and recovery
OM	Organic matter
OP	Ozone depletion (EF)
P	Phosphorus
PAP	Processed animal protein
PBE	Potential beneficial effect
PEF	Product environmental footprint
PHE	Potential harmful effect
POL	Pollution
PP	Photochemical ozone formation, HH (EF)
PR	Phosphate rock
PSR	Promoting social responsibility
PUE	P use efficiency
ReNuRE	Recovered N from manuRE
RIP	Respiratory inorganics (EF)
RO	Reverse osmosis
ROC	Risk of conflicts
SA	Slurry acidification (here slurry with acid treatment)
SAM	Safety measures
SAN	Sanitation coverage
SAS	Stripping and scrubbing
SF	Solid fraction
sLCA	Social life cycle assessment
SOM	Soil organic matter
SP	Superphosphate
SSE	Social security expenditures
SSP	Single superphosphate
TEP	Eutrophication terrestrial (EF)

TIP	Trafficking in persons
TRL	Technology readiness level
TSP	Triple superphosphate
UA	Unacidified (here slurry without acid treatment)
UCPH	University of Copenhagen
UGENT	Ghent University
UNE	Unemployment
VAT	value added (total)
VCA	Value chain actors
VER	Violations of employment laws and regulations
WHW	Weekly hours of work per employee
WLF	Women in the sectoral labour force
WND	Workers affected by natural disasters
WP	Work package
WUP	Water use potential (EF)
WUR	Wageningen University & Research
WWS	Wastewater sludge
WWTP	Wastewater treatment plant
YIL	Youth illiteracy

Glossary

Allocation: means to split environmental impacts of products and by-products typically based on physical or economic factors. For example: if the product weighs 9 kg and the by-product weighs 1 kg, then 90% of the environmental burden could be allocated to the main product. In terms of economic values, the price one receives for either product or by-products are taken as baseline: impact allocation follows the same ratio as the weight ratio.

Attributional LCA: describes a method where the environmental impact of a product or service is assessed based on the quantification of all raw materials consumed and emissions made to produce or provide that very service. The view is retrospective and describes what the production of the product is "responsible" for. ***Consequential LCA***: modelling on the contrary is a prospective description of environmental impacts. It describes what would happen, if product X was to be produced, i.e., what is the consequence of producing it. The difference in data collection is that for attributional modelling, the average market is assumed (e.g., the electricity mix of a country), while in consequential modelling the marginal market is assumed (e.g., for an increase in demand, what would be the most likely additional source of electricity in a country, and for a decrease in demand, which electricity source is most likely to phase out first).

Biological treatment: see ***Nitrification-Denitrification***

By-product: material or substance created as secondary product when processing or manufacturing something else, typically the main product (Example: wheat is produced as main product and the remaining straw, the by-product, is used as cow bedding material)

Consequential LCA: see ***attributional LCA***

Crystallisation, Struvite: see ***struvite***

Dashboard indicators: qualitative environmental performance indicators assigned to technologies and solutions in Nutri2Cycle based on expert judgment and compared to a baseline practice.

Endpoint impact category: see ***environmental impact category***

Environmental impact category: divides environmental impacts into groups. The impact in each group is quantified by equivalence of one molecule or substance. For example, the impact category *Climate change potential* is expressed in CO_2 equivalences. All other molecules that contribute to climate change are expressed relative to CO_2. Impact categories can be divided into ***midpoint*** and ***endpoint impact categories***. While midpoint categories stretch across a variety of environmental impacts such as acidification, eutrophication and climate change, endpoint categories express effects of impacts on the endpoint level, such as reduced live expectancy.

Environmental indicator study: assessment method used in this report, which goes beyond assessing experimental data but cannot qualify as full ***environmental life cycle assessment***.

Environmental life cycle assessment: (eLCA) is a method to quantify and assess the environmental performance of a product or service throughout its entire or selected life cycle stages. To conduct eLCAs, all kinds of resource extractions from and emissions to the environment are quantified and transferred into types of environmental impacts such as global warming or eutrophication. Such impact quantification facilitates the comparison between like products and services and enables informed decision-making.

Frass, insect: insect excrements, residual feed, and dead insect bodies

Functional unit: quantitative description of a defined function or service, which the studied system should provide. All processes and flows in the environmental or social assessment are scaled to the functional unit, which forms the basis for determining the preferred option.

Impact category: see ***Environmental impact category***

Likert scale: is a scale used in questionaries to determine "degree of approval" (e.g., strongly agree to strongly disagree)

Midpoint impact category: see ***environmental impact category***

Multi-functionality: related to ***by-products*** of service or products and describes a situation where a process results in more than one valuable product or service. In eLCAs multi-functionality can be solved in typically two ways: ***system expansion*** or ***allocation***.

NDVI, normalised difference vegetation index: an index that describes the share of land covered by vegetation. The basis for such assessment are remote sensing images, which are assessed with regards to plant land coverage.

Nitrification-denitrification: Nitrification occurs under aerobic conditions and is the first step of biological wastewater treatment. Nitrification is a microbial process during which ammonium is converted to nitrite and then nitrate. Denitrification occurs under anaerobic conditions and is the second step in biological wastewater treatment. The nitrate (and nitrite) from the previous step is now reduced to molecular nitrogen (N_2) and nitric oxide. The objective of the couple process of nitrification-denitrification is the removal of reactive inorganic nitrogen from wastewater in a preferably harmless way.

Normalisation: method to express environmental impacts in relation to a reference. Such a reference could be the environmental impact caused by one average person during one year in the European Union. The question normalisation answers is thus: *What is the contribution of this product or service relative to the average environmental impact of an average citizen during one year?* Contributions are communicated in fractions per total impact, e.g., product x contributes to 1/10 to the total terrestrial eutrophication caused.

Precision fertilisation: is a method to apply variable rates of fertiliser across the area of a field of agricultural land, based on spatially differentiated information such as soil type and nutrient status, biomass and development/nutritional status of the crop or other environmental factors.

Social life cycle assessment (sLCA): is a method to standardize and assess the social performance of a product or service throughout its entire or selected life cycle stages. To conduct sLCA, all kinds of social consequences are collected and categorized. By standardizing all potential positive and negative effects of a product or service, comparisons, and improvements in terms of their social performance are facilitated.

Stripping and scrubbing: By increasing the pH of wastewater or the liquid manure fraction, dissolved ammonium is converted into gaseous ammonia and evaporates. To enhance the stripping process, air is bubbled though the liquid fraction. In the scrubbing stage, the ammonia-saturated air is then brought in contact with an acidic solution (typically sulphuric acid) that captures the ammonia and binds it into ammonium sulphate, which when concentrated enough can serve as a fertiliser.

Struvite: or magnesium ammonium phosphate (MAP), is a phosphate mineral that often forms in pipes and pumps in wastewater treatment plants, where it causes clogging. The process of struvite formation is called ***crystallisation***. This process can also be designed in the WWTP to produce struvite deliberately and then recover it from the wastewater stream, preventing undesirable struvite clogging of pipes. Due to its phosphate and ammonium content, struvite can be used as slow-release N and P fertiliser.

System expansion: is one way to deal with ***multi-functionality***, when comparing products or services that come with different ***by-products***. The systems are changed so that they include the additional functions of the by-product. Example: when comparing direct land application (A) against intermediate anaerobic digestion (B), then B results in energy as by-product. To equalise A and B, either a certain amount of natural gas could be added to A or the same amount of natural gas could be subtracted from B. In both cases, the systems would be 'balanced'. System expansion is typical for ***consequential LCAs***.

Weighting: method to express environmental impacts with regards to their severity. Weighting follows ***Normalisation*** and means to multiply the normalised score with a weighting factor. The weighting factor express the relative importance of an impact category and thus the severity of negative effects on each category. Example: contributions to climate change might be judged more severe than contributions to eutrophication and the weighting factor would be a means to express this judgment in terms of quantitative results.

Executive summary

1. Introduction: Background, context, and objectives

This report is one of the major outputs of Nutri2Cycle WP3 and consists of assessments of a selection of the shortlisted technologies or solutions aiming at closing nutrient loops. This deliverable analyses the technologies, solutions and management frameworks that, in previous Nutri2Cycle studies and deliverables been shown to bear the potential for contributing to closing gaps of N, P and C cycles in agricultural systems.

The overall aim is to quantitatively evaluate the advantages of the selected innovative agricultural technologies and management practices from a broad environmental and social impact perspective. The analysis can serve as policy-guidance regarding the environmental and social impacts from implementation of these selected technologies, solutions, and management systems.

2. Methods: Data inventory & collection, Assessment methods and Internal reviews

A total of 12 selected technologies or solutions from the Nutri2Cycle shortlist (D2.2) are analysed in the current report (see Table 1 below), with 6 of these also included on the Nutri2Cycle priority list (D3.2). The majority are assessed using environmental Life Cycle Assessment (eLCA), but those from Research Line 3 (two precision farming technologies) were not deemed suitable for eLCA and were instead examined as an Environmental Indicator Study (EInS). All 12 were qualitatively analysed by social LCA (sLCA). The individual studies were conducted in collaboration between different partners or groups of experts from different partners.

Table 1. Overview of the selected solutions analysed in this report (RL: Research line, SL: shortlist, LL: Long-list)

RL	SL#	LL#	Long-list abstract title
1	17	18	Slurry acidification with industrial acids to reduce NH3 volatilisation from animal husbandry
2	1	17	Crop farmer using a variety of manure and dairy processing residues to recycle and build soil C, N, P
3	19	30	Precision farming coping with heterogeneous qualities of organic fertilizers in the whole chain
3	23	13	Sensor technology to assess crop N status
4	4	1	Ammonium stripping / scrubbing and NH4NO3 as substitute for synthetic N fertilizers
4	4	2	Ammonium stripping / scrubbing and NH4SO4 as substitute for synthetic N fertilizers
4	4	6	Concentrate from vacuum evaporation/ stripping as nutrient-rich organic fertilizer
4	6	49	N & P recovery from pig manure via struvite crystallization and design of struvite tailor-made fertilizer
4	7	55	Manure processing and replacing mineral fertilizers in the Achterhoek region
4	7	20	Low temperature ammonium-stripping using vacuum
5	9	40	Insect breeding as an alternative protein source on solid agro-residues (manure and plant wastes)
5	12	41	Floating wetland plants grown on liquid agro-residues as a new source of proteins

Data inventories for each eLCA, EInS and sLCA were derived from deliverable reports from WP2, direct data provision from technology providers, field scale modelling (with baselines from WP1) where relevant, and the ecoinvent database. To ensure consistency and comparability between the studies, all eLCAs adhered to the guidelines and recommendations of the Product Environmental Footprint methodology. For each technology or solution, one or several implementation scenarios were compared against a relevant baseline.

To streamline assessments and ensure compliance with the guidelines, an internal review mechanism was implemented. Each study underwent a review process by one or two other partners, covering goals and scopes, graphical system diagrams, methodological choices (such as attributional versus consequential modelling, software used, etc.), and data collection methods. The aim was to discuss and align the fundamentals of each eLCA, EInS, and sLCA at an early stage to ensure appropriate and consistent system boundaries and methodologies.

Finally, a comparison was conducted between the earlier proposed Dashboard Indicators (DBI) results, assessed qualitatively by experts in D3.2, and the DBI equivalent indicators assessed quantitatively in each eLCA. This aimed to evaluate the reliability of the proposed DBIs when based on expert judgement.

3. Results, Discussion and overall Conclusions

Comparison of eLCAs results from the different studies

Due to varying functional units, system boundaries and analysed technologies in individual LCAs, it is challenging to draw general conclusions about which technology performs best in terms of improving C, N and P recovery and recycling. Such a comparison is actually not even desirable as the assessed solutions address different issues in distinct agricultural systems across Europe, as reflected in the diverse functional units and system boundaries. These technologies are not in direct competition; choosing one does not necessitate rejecting another.

Instead, the focus is on comparing their environmental performance to a baseline, enabling a relative assessment of their benefits and drawbacks. The selection of a baseline scenario significantly influences the comparison results, emphasising the need for tailored solutions based on specific requirements and the default reference.

Regarding the number of impact categories included in the different eLCA studies, there is variability across solutions. Most or all of the LCAs include impact categories such as *acidification*, *climate change* and *freshwater eutrophication*. However, the inclusion of categories like *land use*, *resource use of minerals* and *water use* varies and depends on the relevance for each particular technology or solution.

Based on the results, drawing general conclusions about the performance of the tested technologies or solutions is challenging. Each technology exhibits different environmental impact patterns, with some decreasing the environmental impact potential only for specific categories and others reducing it across all categories. Nevertheless, certain tendencies are observed in the different technology assessments. For *acidification potential* and *climate change potential*, many of the tested solutions (14 out of 23 for *acidification potential* and 15 out of 22 for *climate change potential*) demonstrated a reduction in impact potentials. However, concerning *non-cancer human toxicity* and *fossil and minerals and metals resource use*, the tested solutions performed worse than their established baselines (in 10 out of 14 for *non-cancer human toxicity* and 10 out of 15 for *fossil and minerals and metals resource use*). This discrepancy may be attributed to the fact that many assessed technologies are designed to mitigate greenhouse gases and reduce nitrogen losses through recycling, thereby impacting nutrient-related categories and climate change potential positively. Nevertheless, in many cases, these positive outcomes come with associated costs such as energy consumption for heating, pumping or transportation, and use of chemicals or materials, such as sulphuric acid and polymers. The production of these inputs is often linked to the use of energy resources and toxicity impacts. It is somewhat surprising that for eutrophication categories, such as *marine eutrophication*, *freshwater eutrophication* and *terrestrial eutrophication,* the impacts appear to increase and decrease in about an equal number of cases. This may be because the eutrophication impacts are influenced by upstream or downstream processes that might have been overlooked or disregarded during the development of the technology.

DBI qualitative assessment based on expert judgement vs. DBI based on eLCA quantitative assessment

The objective was to conduct an overall comparison between the Dashboard Indicators (DBI) proposed in D3.1 and qualitatively assessed by expert judgment, and the quantitative assessment of the LCA-based equivalent indicators. The goal was to analyse the degree of agreement, identify tendencies of deviations and reflect on potential causes of differences. This analysis is crucial because assessing technologies using the DBI allows for a rapid appraisal, whereas LCA is typically time-consuming, data-demanding, and costly. Therefore, an overarching analysis of the DBI vs. LCA results may offer valuable insights into areas for improvement in the guidelines for DBI assessment.

The comparison revealed that the agreement between DBIs based on expert judgment and the indicators derived from LCA results was less than half (39%). In over one-third of cases (43%), the expert assessment of DBI appeared to be overly optimistic about the environmental performance of a technology, while in about 19% of cases, it appeared to be overly pessimistic. The expert judgment of DBI was notably over-optimistic about N_2O emissions and rock phosphate consumption. However, a good agreement between DBI and LCA was found for the carbon footprint of the technologies as well as for nutrient recovery, where equal results were achieved in more than half of the cases. Surprisingly, over-pessimism was detected for soil quality.

One major lesson from this assessment is that a rapid appraisal or expert interviews may be sufficient for some impact categories (e.g., rock phosphate consumption), but for others (e.g., electricity consumption), there is a substantial risk of a rapid assessment being misleading. To address these issues, it might be advisable for experts to pay attention to impact categories or indicators with particularly poor agreement and to broaden their perspective to potentially linked processes and secondary effects. LCA studies enable the inclusion of such secondary effects and provide insights into whether the benefits achieved in a specific agricultural practice may be nullified by the additional materials or energy required to achieve those benefits. For C, N, and P recovery and recycling technologies, these indirect consequences can potentially be generalised and used to guide or inform rapid appraisals.

It is also possible that those developing, providing and suggesting technology may focus on specific benefits, such as directly reducing N_2O emissions resulting from the agricultural practice in question. This narrow focus could unintentionally lead to ignorance of side effects. It is crucial to avoid potential pollution-swapping effects by considering the broader environmental implications and unintended consequences of implementing technologies.

Another crucial difference between expert judgment and LCA assessments is that the system boundaries may not have been equally well defined. In LCA, all upstream and downstream effects in the background system are included to the extent possible. In contrast, expert judgments typically do not have well-defined system boundaries. This implies that while the most obvious up- and downstream effects may have been considered, some of the more inconspicuous but still substantial environmental impacts might have been overlooked.

In conclusion, the expert assessment of the Dashboard Indicators (DBI) was able to highlight some important aspects of the technologies. However, it is evident that important aspects can be missed, or the assessments can be biased. Decisions regarding the implementation of high Technology Readiness Level (TRL) environmental technologies on a larger scale should always be based on thoroughly conducted LCAs with well-defined system boundaries. However, for the initial assessment and prioritisation of technologies at a low TRL level, expert judgment of dashboard-type indicators may be used, especially if sufficient guidance and information are provided when assessing the more challenging or complex indicators, where upstream or downstream processes of importance may have a significant impact.

Social LCA

The social LCA study selected and tested a range of indicators for potential social hotspots and opportunities related to the novel technologies.

The qualitative expert assessment of these indicators highlighted the need for highly skilled workers, the attraction of a qualified labour force to agriculture, increased training and employee development and improved technology efficiency as some of the most positive impacts. Some technologies also contribute to reducing odours and other gaseous nuisances for local communities, minimising workplace accidents, and acting as drivers for more effective regulations of organic fertilisers. However, certain indicators, such as new jobs or a reduction in extra hours at farms, were site-dependent and varied depending on the technology or farmer behaviour. It was noted that new technologies may introduce new sources of damage, for instance, when using acids or working with heavy machinery, although these risks are controllable.

Qualitative scoring can serve as a starting point for sLCA, predicting the potential benefits and harms of new technologies. However, there is a need for the development of better methodologies for the quantitative assessment of sLCA indicators. Additionally, advancing concepts for weighting social and environmental indicators in simultaneous assessments is essential for comparing or aggregating results from the two dimensions in sustainability assessments.

1 Introduction

1.1 Background and context in the project

This report is a component of the innovation funnel, representing the second stage in the Nutri2Cycle project. The work in the innovation funnel, under WP2, involves investigating innovative technologies and management practices proposed by the Nutri2Cycle consortium to support carbon, nitrogen and phosphorus recovery and recycling at the farm level. These technologies and solutions primarily aim to achieve emission reductions and/or redirect farm residues and by-products. In the preceding phase of the innovation funnel, the first part of the work streamlined 76 initial propositions for technologies and solutions (D.2.1) into a shortlist of 47 solutions, categorised into 24 shortlist solution categories (D.2.2).

The research efforts in Nutri2Cycle were concentrated on generating research data for the solutions identified in the shortlist (D2.2). These solutions were categorised based on their technology readiness level (TRL) and detailed in deliverables D.2.3 (TRL < 6) and D.2.4 (TRL 6-9). The experiments and data outlined in these deliverables served as input for subsequent assessments related to the technical, environmental, economic and social performance of prioritised solutions (conducted in WP3 - WP5). Additionally, D.2.6 compiled supplementary information and discussed additional data needs to facilitate future assessments.

In the subsequent phase of the Nutri2Cycle project, the shortlisted solutions (D.2.2) underwent a thorough evaluation to form a priority list of 14 innovations chosen for upscaling, demonstration and further investigation (D.3.2). This process involved a Venn-diagram exercise, assessing the feasibility of Life Cycle Assessments (LCAs) for each solution and weighing it against experimental research in WP2 and demonstration activities in WP6 that produced relevant data. The solutions selected for LCA (both environmental and social) included those from both the priority list (D3.2) and the shortlist, which were investigated in WP2 and WP6 (refer to Figure 4 and 5 in D3.2 – solutions from the priority list are also indicated in Table 1 of this report).

The current report (D.3.4) is a component of WP3 and plays a role in describing and assessing the chosen selection of shortlisted technologies or solutions with the aim of closing nutrient loops. Previous studies and deliverables within Nutri2Cycle have indicated that the management frameworks outlined in this deliverable hold the potential to contribute to closing gaps in N, P and C cycles in agricultural systems. The primary objective of D.3.4 is to offer quantitative data and insights into selected innovative agricultural technologies and management practices, considering both environmental and social perspectives.

In terms of methodology, the current deliverable follows previous deliverable reports, starting with D.1.1, which screened and reviewed a broad array of potentially relevant indicator sets for comparison and benchmarking. Subsequently, D.3.1 prioritised and applied a subset of these indicators, termed dashboard indicators (DBI), from the D.1.1 list. This descriptive or qualitative assessment aimed to provide a swift appraisal of a solution's environmental performance compared to baseline agricultural management, utilising an easy-to-understand approach (baseline approach and case data described in D.1.5). Simple dashboards (summary presentation) were developed for straightforward communication of indicators.

In addition to the present report, deliverable report D.3.3 serves as a supplement, analysing the economic effects of implementing the prioritised technologies, solutions and management systems. D.3.3 employs cost-benefit analysis to compare the costs and revenues of a baseline agricultural management system against a system where the proposed solutions have been implemented.

1.2 Objectives

Deliverable report D.3.4 aims to assess the advantages of the selected innovative agricultural technologies and management practices from a comprehensive environmental and social impact perspective. The analysis is intended to provide policy guidance concerning the implementation of selected technologies, solutions and management systems.

The specific objectives of Deliverable 3.4 are:

- to describe the on- and off-farm systems related to the analysed technologies and solutions and define
  - I. the baseline agricultural system without any solution implemented
  - II. the (hypothetically) altered agricultural systems following the implementation of the innovative solution, including all its direct and indirect effects
- to describe the data collection methods & underlying models
- to present the results of the environmental assessments either in terms of
  - I. environmental life cycle assessments (eLCA), or
  - II. environmental indicator assessments (EIns), and
- to compare the results of the environmental assessment with the results of the same solution as assessed with the dashboard indicators (D3.1)
- to describe the results of the social life cycle assessment (sLCA)

Table 2. Overview of the priority list solutions analysed in this report.

D3.2 indicates the priority list solutions, in bold those included for environmental and social analysis in this report. EInS = Environmental indicator study, eLCA = Environmental life cycle assessment, sLCA = Social life cycle assessment. In the last column it is indicated for which solutions economic Cost Benefit Analysis (CBA) was also conducted.

RL	SL#	LL#	Long-list abstract title	D3.2	EInS	eLCA	sLCA	CBA
1	17	18	Slurry acidification with industrial acids to reduce NH3 volatilisation from animal husbandry			x	x	
1	13	10	Small/Farm scale AD of agro-residues to increase local nutrient cycling & improve nutrient use efficiency	x				x
1	15	24	Adapted stable construction for separated collection of solid manure and urine in pig housing	x				x
2	1	16	Using digestate, precision agriculture and no-till focusing on OM stocking in an area characterized by the lack of it.	x				x
2	2	17	Crop farmer using a variety of manure & dairy processing residues to recycle and build soil C, N, P fertility	X		x	x	x
3	19	30	Precision farming coping with heterogeneous qualities of organic fertilisers in the whole chain	X	x		x	x
3	23	13	Sensor technology to assess crop N status	X	x		x	
3	21	73	Precision arable farming using BBF in potato growing	x				x
4	4	1/2	Ammonium stripping / scrubbing and NH_4NO_3 as substitute for synthetic N fertilisers	X		x	x	x

RL	SL#	LL#	Long-list abstract title	D3.2	EInS	eLCA	sLCA	CBA
4	4/7	6/ 43	Concentrate from vacuum evaporation/stripping as nutrient-rich organic fertiliser	X		x	x	x
4	4	9	The liquid fraction of digestate substitute mineral N&K	x				x
4	6	49/ 65	N and P recovery from pig manure via struvite crystal-lization & design of struvite based tailor-made fertilisers	X		x	x	x
4	7	55	Manure processing and replacing mineral fertilisers in the Achterhoek region	X		x	x	
4	7	20	Low temperature ammonium-stripping using vacuum	X		x	x	x
4	8	22	BIO-PHOSPHATE: high temp. reductive thermal process recovery of concentrated P from animal bones	x				x
5	9	40	Insect breeding as an alternative protein source on solid agro-residues (manure and plant wastes)			x	x	
5	12	41	Floating wetland plants grown on liquid agro-residues as a new source of proteins	X		x	x	x

2 Methods

The approach outlined in this deliverable consists of two main parts, as depicted in Figure 1. Each part, detailed in the following sub-chapters, involves two major steps: (i) data collection and (ii) environmental and social assessments. The entire process was a collaborative effort between partners and incorporated insights from related deliverables. Close communication between technology providers and environmental assessors facilitated data recovery. The approach encompassed the modelling of emissions from agricultural fields as input data for the LCA and underwent a rigorous internal review process, as explained in more detail below.

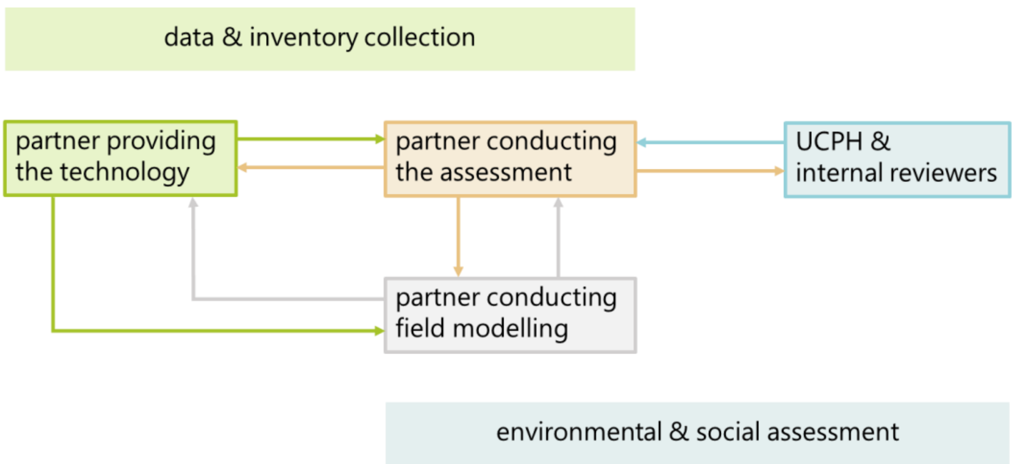


Figure 1. Schematic representation of workflow and collaboration for this report

2.1 Data inventory & collection

The foundation for data collection stemmed from the initial work reported in D.2.1, where the compilation of factsheets provided an initial understanding of the technology or solution. Subsequent reports, D.2.3 and D.2.4, offered more detailed descriptions of the longlist solutions, contributing to a deeper comprehension of the technology. Finally, D.2.6 and the associated data collection formed the basis for the subsequent assessments. To ensure reliable data collection in D.2.6, WP3 partners established 'minimum data set' descriptions as guiding principles, covering physical and chemical characteristics of materials and by-products, emission and environmental data, as well as work safety information. In some instances, environmental assessments had commenced before the completion of D.2.3, D.2.4, and D.2.6, necessitating closer collaboration with technology providers and cross-checking for data validity and consistency. Additional data was occasionally required and had to be sourced from the literature with the assistance of technology providers.

Finally, data from modelling studies using the baselines created in WP1 was used in several LCA studies in order to close data gaps relating to emissions taking place under field conditions. In the following a description of the collaboration regarding experimental and model data is given.

2.1.1 Data provision: technology provider

Some data was collected internally by partners that was both acting as technology providers and environmental assessors. Other data came from other partners and was shared between them. Below is a list giving a rough overview of the partners that have shared data and, in that way, collaborated on the assessment (technology provider & technology evaluator):

- LL#1+2: Detricon Inc. Provided data to UGENT and UCPH (private sector collaboration)
- LL6#: System project (EU-grant no. 730400) and Nitroman (Interreg V project) provided data to UGENT and UCPH) (cross-project data provision)
- LL#17: TEAGASC provided data to CARTIF
- LL#30: Thünen Institute provided data to UCPH and WUR
- LL#40: INAGRO provided data to UCPH and UGENT
- LL#41: INAGRO provided data to UCPH and UGENT
- LL#55: Systemic project (EU-grant no. 730400) provided data to WUR (cross-project data provision)
- LL#65: Aquafin Inc. provided data to UGENT and UCPH (private sector collaboration)
- Remaining eLCA and EI LL# data was collected internally by respective partners
- All LL#s provided data to IRTA for the SLCA

The collaboration ensured appropriate system boundaries, cross-validation of literature data and the coherent implementation of provided data.

2.1.2 Data provision: model provider

In addition to the described data inventories, agricultural field modelling was employed, along with direct measurements conducted by technology providers, for technologies where field emissions of C, N, and/or P to the environment were anticipated to be significantly influenced by the introduction of the technology. This modelling helped define farm baseline scenarios (D.1.5) and assess the change in field-scale emissions resulting from the implementation of innovative technologies. Daisy and SWAP/ANIMO agricultural models (as outlined in D1.2 and applied for baselines in D1.5) were utilised, relying on data provided by technology providers and additional information from technology evaluators regarding the specific circumstances under which the technology was to be modelled. Collaboration between modellers and technology evaluators took place in:

- LL#17: WUR conducted ANIMO modelling for CARTIF
- LL#18: internal Daisy modelling at UCPH

- LL#30: WUR and UCPH conducted Daisy modelling for WUR
- LL#55: internal ANIMO modelling at WUR

Agricultural modelling and collaboration ensured a better representation of the effect of the technologies in the field and emissions related to the field under specific circumstances than literature data could have done.

2.2 Assessment methods

2.2.1 Environmental life cycle assessment

Environmental life cycle assessment (eLCA or LCA) quantifies the environmental impacts of products and services across their entire life cycle stages. This assessment helps reveal impacts per stage, enabling the detection of environmental trade-offs resulting from adjustments or optimisations in one stage affecting other stages. In the Nutri2Cycle project, innovations focused on addressing harmful stages of agricultural production systems, such as technologies to reduce greenhouse gas emissions from handling livestock slurry. These innovations often require resources beyond those available to farmers, leading to potential problems beyond the farm boundaries, such as the use of additional machinery, chemical substances or energy.

LCA was employed to gain a comprehensive understanding of technologies addressing on-farm issues, extending the analysis beyond localised solutions. The LCAs aimed to quantify environmental impacts, identify burden-shifting and highlight areas for improvement. Results from the LCA were intended to complement and validate findings from the more qualitative dashboard indicator study outlined in D.3.1.

We selected technologies for LCAs based on data availability, expertise, and potential for scalability within the project. Further, the need for LCA studies was assessed based on published, existing studies. We further aimed to cover all research lines with at least one LCA to offer a greater variety and representation of studies. A detailed description of the selection process is given in D.3.2.

To ensure consistency between the LCAs that have been conducted in Nutri2Cycle we agreed upon some overall principles. These include:

1: All LCAs should follow the guidelines and recommendations on conceptualisation and interpretation of the Product Environmental Footprint (PEF) methodology.

2: All LCAs should use the impact assessment methodology and characterisation factors of the Environmental Footprint methodology.

3: Data inventories should be based on either / and

1) primary data from technology providers (described in D.2.4)
2) models and model baselines from the modelling conducted in WP1 D.1.5.
3) (peer-reviewed) literature
4) ecoinvent database.

For a more detailed description of methodologies applied in each LCA, please be referred to their respective chapters.

2.2.1.1 Dashboard indicators vs. environmental life cycle impact assessment

Within the frame of Nutri2Cycle, a set of Dashboard Indicators (DBI) has been developed (D3.1 report) that was used to facilitate a rapid appraisal of a technology's environmental performance compared against baseline agricultural management. The set of DBI presents a *qualitative* assessment of a technology, as opposed to the *quantitative* assessment of LCA studies in reference to the functional unit and system boundaries chosen to analyse a certain technology in specific settings.

The focus of the developed set of DBI in D3.1 was to evaluate the contribution of technologies to nutrient cycling, climate change mitigation, resource use reduction, and soil quality improvements. The DBIs were primarily based on indicators directly related to agricultural activities, although no precise system boundaries were defined. LCA results on the other hand aim to encompass a wider range of effects in- and outside the agricultural sphere, e.g., upstream procurement of materials, downstream consequences for food production, which required very well-defined system boundaries. We selected the set of DBI (as described in D3.1) because comprehensive LCA studies are not always feasible, given their extensive need for inventory data and thorough interpretation. The strengths of DBIs are that they are straight-forward to evaluate, facilitate fast screening of technologies regarding suitability and applicability, and are easy to communicate.

In the current report, we evaluate the earlier assessed DBI results against the LCA indicators of the selected studies in the current report. In order to do that, we took the set of DBI from D3.1, where they were assessed by expert judgement as a starting point, and then compared them against the equivalent indications/quantifications addressed in the LCAs. Table 3 is in part derived from Deliverable 3.1 and shows how we connected the qualitative set of DBI assessed by the experts with the equivalent indicators based on LCA results. We compared the DBI results with LCA results and either confirmed or disproved the findings of the DBI. Since in the LCA results are given as quantified impact potentials, we decided that a change (between baseline and solution) lower than 10% was negligible, while a change of >10% is considered a true change in impact. This 10% cut-off was arbitrary, but it was chosen based on our experience of typical uncertainty on quantified impacts in LCA; therefore a ±10% change in impact we would typically consider insignificant (=no change). This facilitates the comparison between DBI and LCA, as the DBIs only differentiate between no change and positive or negative effects.

Table 3. Principles between the comparison of dashboard indicator based on expert judgement and similar indicators based on life cycle assessments

	Dashboard indicator		LCA equivalent indication (per functional unit)	
Dimension	Name	Full description	Name	Comment
Use of primary resources	Rock phosphate	Reduction in mineral phosphorus consumption	Phosphorous, in ground | Resource	Change in P extraction within the system boundaries. Not limited to P consumed on the farm.
	Natural gas	Reduction in natural gas consumption in mineral fertiliser production	Gas, natural, in ground | Resource	Change in gas extraction within the system boundaries. Not limited to gas consumed during fertiliser production.
	Oil	Reduction in oil consumption in agricultural machinery	Oil, crude, in ground | Resource	Change in oil extraction within the system boundaries. Not limited to oil consumed in agricultural machinery.

Dimension	Dashboard indicator Name	Full description	LCA equivalent indication (per functional unit) Name	Comment
			Diesel burnt in agricultural machinery | Product	Not limited to machinery operated on the study farm.
	Electricity	Reduction in electricity consumption	All processes on 'electricity production' regardless of source and voltage (e.g., ecoinvent)	Not limited to electricity consumption on the study farm
	Water	Reduction in water consumption	Water scarcity as EF method category	Change in water consumption not limited to on-farm
	Soil quality	Improvement in soil quality	Land use as EF method category	No ideal representation – only a proxy – and this may influence correspondence with the DBI (soil quality is anyway vaguely defined); Change in soil quality not limited to soil on-farm
	Nutrient recovery	Nutrient recovered from agriculture and livestock systems	Market for N, P and K fertiliser	No ideal representation – only a proxy; including all 'Market for nitrogen fertiliser as N'
Emissions to the environment	Ammonia (air emission)	Reduction in NH_3 emissions	Ammonia | Emission to air	Including: low & high population density, long term and unspecified
	Nitrous oxide (air emission)	Reduction in N_2O emissions	Dinitrogen monoxide | Emission to air	Low & high population density, long-term, unspecified
	Methane (air emission)	Reduction in CH_4 emissions	Methane | Emission to air	Including: low & high population density, fossil & biomass stocks
	Nitrate (water emission)	Reduction in NO_3 emissions	Nitrate | Emission to water	Including: unspecified, ocean, ground water, surface water
	Phosphorus (water emission)	Reduction of P emissions	Phosphorous | Emission to water	Including: unspecified, ocean, ground water, surface water
			Phosphate | Emission to water	Including: unspecified, ocean, ground water, surface water
	Particulate matter	Reduction of particulate matter formation	Particulates [...] | Emission to air	Including: all < 10 um; low & high population density, lower + upper stratosphere, unspecified
Resilience to climate change	Carbon footprint	Reduction of carbon footprint	Climate change | EF method category	Greenhouse gas emissions not limited to on-farm
	Effective SOM	Effective Soil Organic Matter improvement	Carbon | Emission to soil	No ideal representation – only a proxy; including: agricultural, industrial, unspecified
			Carbon dioxide | Emission to soil	No ideal representation – only a proxy; including: to soil or biomass stock; agricultural & unspecified
	Renewable energy production	Renewable energy produced from biomass	-	*This cannot as such be seen in an LCA*

In the case of the set of DBIs, the performance of technologies was evaluated against a baseline without its implementation. The evaluation was scored qualitatively with regards to three possible effects a technology may have, namely:

- **+** Positive: the technology causes *improvements* compared to a baseline
- **-** Negative: the technology causes *deteriorations* compared to a baseline
- **o** Neutral / unknown: the technology has *no different* effect compared to a baseline or the effect is not known

The right-hand side of Table 3, listed at the end of each individual eLCA chapter, represents a quantitative assessment of the DBI-equivalents from the LCA, and thus an update of the qualitative assessment of DBIs in D3.2.

After presenting the findings of both evaluation methods, qualitative DBI assessment, and equivalent LCA indication, we have assessed the findings in relation to each other: their shortcomings and strengths, as well as their consensuses and contradictions. Results comparison and synthesis discussion are presented in the Chapter 4.

2.2.2 Environmental indicator assessments

In the assessment process, we realised that at this stage it was not relevant to conduct LCAs on all selected solutions. For solutions within research line 3: Tools, techniques & systems for higher-precision fertilisation, we judged that environmental LCA was an unsuitable methodology. As stated above, LCA is relevant for identifying environmental impacts and trade-offs of new technologies. The benefit of precision agriculture is that fertiliser is saved or perhaps losses are reduced, but there is no apparent trade-off apart from the impacts of producing a few sensors and managing and analysing data. However, we did not find reliable data on the manufacturing of sensors or on the energy and materials requirements for processing, storing, and providing data. We were thus unable to properly assess the environmental burdens resulting from the implementation of sensor technologies. When assuming them to have no or negligible impacts, which may well be true, then sensor technologies are obviously always a good idea because the use of less fertiliser, while maintaining yields at the same level will inevitably lead to lower environmental impacts. In this case it is more relevant to quantify for example the magnitude of leaching reduction. However, this requires agricultural mechanistic modelling or experimentation and cannot be answered by LCA.

Given the above, we decided to introduce another form of assessment, which we called environmental indicator (EInS) assessment. This type of assessment should go beyond the interpretation of experimental data and give a clearer picture of the implications of sensor technologies and their influence on field emissions.

2.2.3 Social life cycle assessment

To achieve sustainability, agriculture must comply with the principles of sustainability which are defined in the Brundtland Report (Brundtland, 1987). Like any other economic sector, agriculture must respect the needs of present and future generations while ensuring profitability, environmental health, and social and economic equity. Innovation in agricultural systems can have beneficial environmental impacts, especially when it comes to reducing emissions. However, associated social impacts, may not be apparent. Introducing novel agricultural technologies to reduce environmental impacts can create growth, jobs for local communities, training for workers, new outputs (e.g. biogas), and systems and innovative options involving science, technology and policy. Furthermore, the introduction of high levels of technology and innovation, presents an opportunity to attract young and skilled workers, making agriculture more interesting to this section of the population. However, it is not clear how adaptations and modifications to already established industries might evolve in a sustainable manner (Siebert et al., 2018).

It is difficult to obtain specific data to assess the social impacts over a whole production chain in agriculture. This lack of information can lead to an imbalance between the three dimensions of sustainability (environmental, social, and economic) (Darnhofer et al., 2010). However, there is growing awareness for the need for information on the social costs and opportunities of current activities and their related alternatives (Darnhofer et al., 2010). Social Life Cycle Assessment (sLCA) has shown to be a relevant methodology for the social evaluation of product systems, processes, and services (Chen & Holden, 2017; Pelletier, 2018; UNEP, 2020). sLCA helps to assess the socioeconomic impacts that directly and indirectly affect stakeholders during a product life cycle, providing short- and long-term information to help organisations understand their current situation and to develop future strategies (Kühnen & Hahn, 2017; Arcese et al., 2018). The Guidelines for Social Life Cycle Assessment of Products and Organizations (UNEP, 2009) were updated in 2020 (UNEP, 2020), and are used to assess social and socio-economic impacts, both positive and negative, of products over their life cycle.

In the present deliverable, a qualitative sLCA using the Likert scale (Albaum, 1997) and expert opinions was used to identify the potential social impacts of the implementation of solutions to recover nutrients in agricultural systems across Europe. In addition, a quantitative sLCA using the Product Social Impact Life Cycle Assessment (PSILCA) database was performed as a case-study for a technology for ammonia recovery using country and sector data.

2.3 Internal reviews

This process was also described in the 2nd Period Technical Report Part B.

The range in topics tackled in the different environmental and social assessments and the variation in approaches was a challenge for the collaborative work underlying this report. To streamline and harmonise the assessments, in terms of methodology, detail and presentation, we introduced an internal review mechanism. The internal reviews were particularly relevant for the environmental LCAs which were conducted by multiple partners. The review allowed for closer collaboration and ensured higher consistency and comparability of the studies.

To facilitate the review process, a review document was developed at the beginning of the work period. The form consisted of a brief description of the goals and scopes of the study, graphical system diagrams, an outline of methodological choices (such as attributional versus consequential modelling, software used, etc.), and an overview of data collection methods. The idea was to communicate the fundamentals of each eLCA, EInS, and sLCA at an early stage to ensure appropriate and consistent system boundaries and methodology.

The reviews ensured cross-checking at two stages. For each study, one or two partner institutions were selected for review. The following procedure was followed: (1) The conducting partner filled in the review form and sent it to the selected partners for review. (2) The reviewers commented on model choices, assessed the degree of revision needed (major, minor, none) and returned the form. (3) In an online meeting the conducting partners presented the changes they made and the new status of the assessment study. In most cases, a final discussion following this presentation was sufficient to satisfy everyone's expectations and remove all concerns. (4) The study was finalised. If open questions remained, steps (2) and (3) were iterated, until a compromise was reached, similar to a scientific peer-review process.

To keep track of the status of each study, we used an online spreadsheet accessible to all partners. Table 4 shows parts of the spreadsheet table to give an idea of the procedure.

Table 4. Example segment of online spreadsheet to follow-up on status of each assessment (contents are an example, reflecting the status in the middle of the assessment work process)

LL#	Who?	Responsible person	Type of analysis	Data provider	Sent for review?	Review A	Decision 1	Response to review	Clarifying meeting to discuss review	Decision 2
1,2,6	UGENT	Rahul Ravi	LCA	Claudio Brienza Anne Adriaens (Detricon, Strocon, AMPower)	yes	WUR	minor revision	No	no	no decision yet
11	SOLTUB	Zoltán Hajdu	LCA	Zoltán Hajdu (Soltub Ltd.)	yes	CARTIF	major revision	yes	no	no decision yet
13	SOLTUB	Zoltán Hajdu	Indicators	Zoltán Hajdu (Soltub Ltd.)	yes	IRTA	major revision	yes	no	no decision yet
18	UCPH	Miriam Beyers	LCA + daisy model	Lars Stoumann Jensen (UCPH)	yes	WUR	major revision	yes	yes	accepted
19-drop	UCPH		LCA		no	CARTIF	no decision yet	no	no	no decision yet
21-drop	IRTA		21		yes	UCPH	major revision	yes	no	no decision yet
27-drop	CARTIF		27		no	SOLTUB	no decision yet	no	no	no decision yet
30	UCPH/WUR	Y.F. Duan	Daisy modelling & indicators	Mareike Söder (Thünen Institute)	yes	UCPH	minor revision	no	no	no decision yet
40	UCPH + UGENT	Miriam Beyers	LCA	Carl Coudron (inagro)	yes	CARTIF	minor revision	yes	yes	accepted
49	CARTIF	Francisco Verdugo	LCA	Francisco Corona (CARTIF)	yes	UGENT	major revision	no	no	no decision yet
55	WUR	Y.F. Duan	LCA + ANIMO model		yes	UCPH	major revision	no	yes	no decision yet
65	UGENT + UCPH	Rahul Ravi	LCA	Bart Saerens (Aquafin Inc)	yes	IRTA	accepted	yes	yes	accepted
17 - new	CARTIF	Francisco Verdugo	Regionalyzed LCA + ANIMO model	SM Ashekuzzaman (teagasc)	yes	UCPH	major revision	yes	yes	major revision
41 - new	UCPH	Miriam Beyers	LCA	Reindert Devlamynck (inagro)	yes	(CARTIF)	major revision	yes	yes	accepted
20 - new	IRTA	August Bonmati	LCA	Miriam Cerrillo (IRTA)	yes	WUR	major revision	No	no	no decision yet

3 Results

3.1 Environmental life cycle assessment studies

3.1.1 LL#1+2+6: Ammonium stripping + scrubbing & Vacuum evaporation + stripping to produce alternative N fertiliser (UGENT + UCPH)

Longlist #1+2 title: Ammonium stripping / scrubbing and NH_4NO_3 as substitute for synthetic N fertilisers
Longlist #6 title: Concentrate from vacuum evaporation/ stripping as nutrient-rich organic fertiliser

Rahul Ravi[1,2], Miriam Beyers[2,1], Claudio Brienza[1], Sander Bruun[2], Erik Meers[1]

[1] Department of Green Chemistry and Technology, Faculty of Bioscience Engineering, Ghent University, Belgium

[2] Department of Plant and Environmental Sciences, Faculty of Science, University of Copenhagen, Denmark

This chapter builds on the published scientific paper by Ravi et al. (2023) In the quest for sustainable management of liquid fraction of manure - Insights from a life cycle assessment. Sustainable Production and Consumption. https://doi.org/10.1016/j.spc.2023.11.006

3.1.1.1 Introduction

As nitrate vulnerable zones, Flanders and the Netherlands are bound by legal application limits for animal manure and its derivatives. The Nitrates Directive limits nitrogen (N) application from manure to 170 kg N ha^{-1} y^{-1}. In addition, local legislation can limit phosphorus (P) application as low as 40 kg P_2O_5 ha^{-1} y^{-1}. As animal densities in Flanders and in the Netherland are particularly high, surplus manure is often processed. Solid-liquid separation of animal manure is used as a starting point where N is up-concentrated in the liquid fraction (LF) and P in the solid fraction (SF). The P-rich solid manure fraction is transported to P-deficient regions, and the N-rich liquid fraction is treated through conventional biological treatment, i.e., nitrification-denitrification (NDN). There are numerous innovative solutions for treating liquid manure fractions but biological treatment, specifically NDN remains the preferred method in Flanders due to its ease of operation and a lack of incentives for further manure valorisation. Disadvantages of the NDN technology are its high energy consumption and that all of the useful nitrogen is emitted as N_2.

Recently, there has been a push to promote a circular economy. This has resulted in the development of several novel manure valorisation techniques for nutrient reuse and recovery (NRR) in Flanders and the Netherlands. NRR technologies include: (a) ammonia (NH_3) stripping and scrubbing (SAS) to produce ammonium nitrate & ammonium sulphate, (b) membrane filtration (MF) to produce mineral concentrate (MC), and (c) evaporator systems (EvS) that may or may not be combined with membrane filtration to produce a mineral concentrate. In legal terms, the resulting products continue to be classified as animal manure, despite the fact that they contain a high proportion of mineral N relative to total N (Reuland et al., 2021). However, the European Union's joint research centre recently developed technical proposals for the safe use of such animal manure-derived products in nitrate-sensitive zones and reclassified them as "Renure" (Recovered Nitrogen from Manure).

The working principle of ammonia stripping-scrubbing installations involves shifting the NH_3:NH_4^+ equilibrium in liquid manure to NH_3. This is done by increasing either temperature, pH or both. The NH_3 is then scrubbed from the headspace with a strong acid (HNO_3 or H_2SO_4). There are many configurations of SAS (Sigurnjak et al. 2019), but we focus on an end-of-pipe pathway where digested manure is first separated into solid and liquid fractions, and subsequently, the liquid fraction is stripped and scrubbed.

Evaporator systems (EvS) are another long-standing and well-tested technology to concentrate the nutrients (mineral concentrate) and to distil purified water (condensate) from the liquid fraction (Vondra et al. 2018). EvS's working principle is based on vacuum evaporation, where the LF is boiled at sub-atmospheric pressure. Due to the pressure drop, the boiling point is lowered (50-60 degrees C) and a large proportion of the water evaporates. The benefits of using EvS is to reduce the LF volume, thus reducing transport costs. Usually, EvS are integrated with membrane filtration systems such as reverse osmosis (RO) to further retain N and potassium from LF.

This study uses primary data from installations processing liquid fraction of pig manure which are still at an early stage of development.

3.1.1.2 Materials & methods

3.1.1.2.1 Goal & scope

The goal of this study was to:

(i) identify the trade-offs of the aforementioned technologies to treat LF and measure their performance relative to the baseline, NDN

(ii) quantify the potential for the technologies in terms of environmental benefits and identify key parameters responsible for the uncertainty of the quantification.

The geographical scope has been set to Flanders in Belgium and the Netherlands.

The functional unit of the study is: treatment of 1 tonne of liquid fraction pig manure, the characteristics of which are listed in Table 5.

This functional unit was analysed for the scenarios described below, and a graphic representation can be found in Figure 2. The preceding step, i.e. solid-liquid separation has been cut-off from the system since centrifugation is common for all scenarios.

The ***Baseline (S1)*** represents the treatment of LF via NDN. The liquid fraction is stabilized through nitrification-denitrification. The effluent from the NDN system does not meet discharge requirements and the common practice is to apply it to local fields. The residual sludge is transported to P-deficient regions.

In ***Scenario S2 (LL#1+2)***, the NH_3 in the liquid fraction is "stripped and scrubbed" with HNO_3. The ensuing ammonium nitrate is considered as a mineral fertiliser substitute and field applied in non-nutrient surplus regions. The stripping residue, is pumped to an NDN system, followed by tertiary treatment in a constructed wetland. Post treatment, the effluent satisfies discharge norms (250 mg/l chemical oxygen demand (COD), 25 mg/l biochemical oxygen demand (BOD), 35 mg/l suspended solids, 15mg/l N and 1mg/l P).

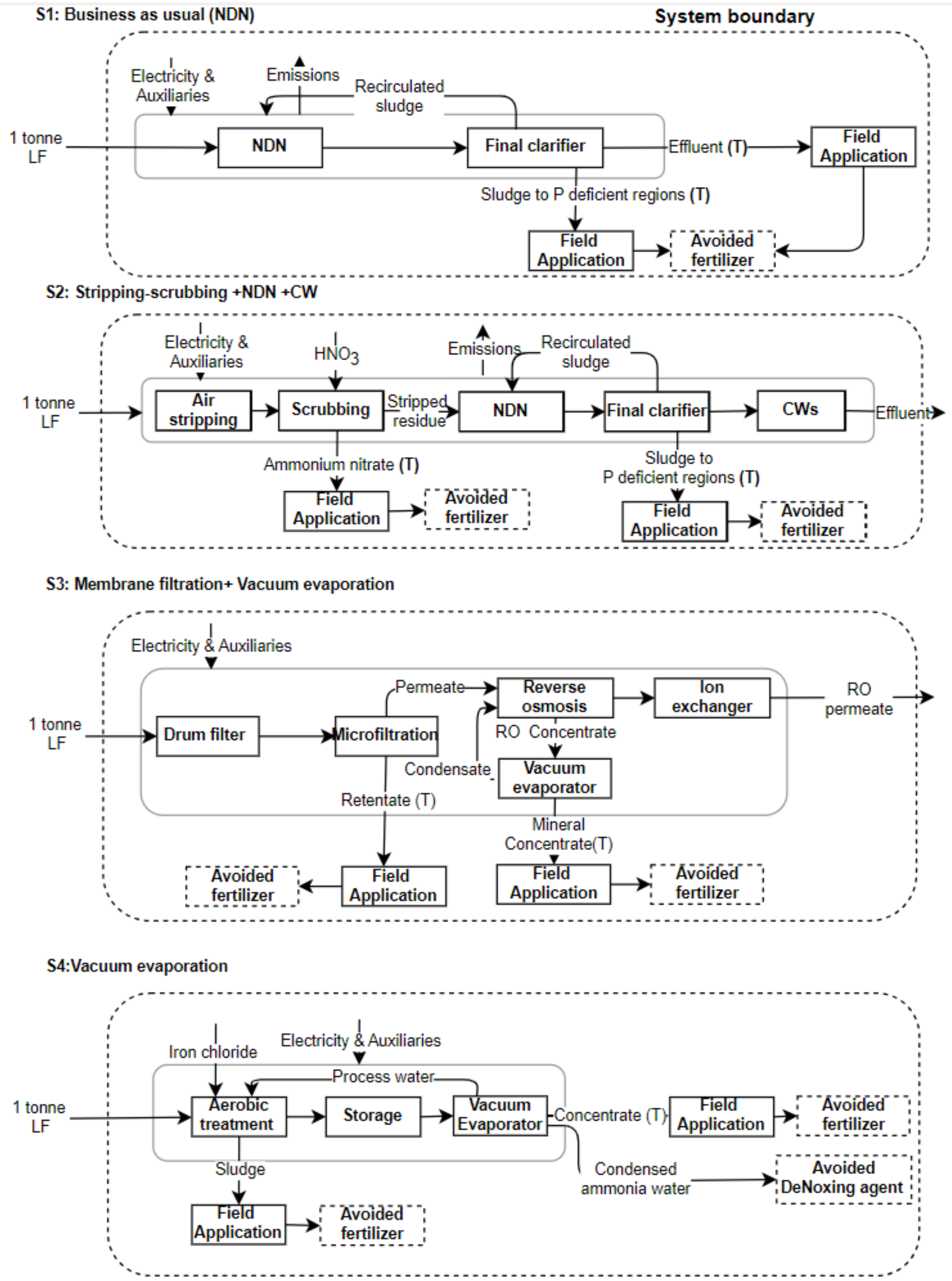


Figure 2. LL# 1+2+6: System boundaries.

Comparison of liquid fraction (LF) manure management. NDN represents nitrification-denitrification and CW is constructed wetlands. (T) denotes transport of the product and hashed blue boxes represent avoided processes.

In ***Scenario S3 (LL#6a)*** a combination of micro-filtration and reverse osmosis + evaporator system is used to concentrate the LF. The concentrate from the RO evaporator system is considered as a useful fertiliser, whereas the permeate is discharged. The filtrate from the micro-filtration unit is transported to non- nutrient surplus regions.

Scenario 4 (LL#6b) focuses on vacuum evaporation without a membrane filtration set-up. The outputs from the system include concentrate (an NK nitrogen-potassium fertiliser substitute), condensed ammonia water (can be used as a denoxing agent in incineration plants) and process water (that is partly recirculated and partly used as cleaning water).

In all the scenarios, the impacts from the infrastructure (machinery, construction, etc.) have been considered. The lifespan of the infrastructure in the NDN system is 40 years whereas in the other scenarios, it varies between 5 to 15 years.

3.1.1.2.2 Inventory

These characteristics were derived from weekly measurements conducted from May 9th to December 10th, 2019, at a pig manure treatment facility. The facility is located in Gistel-Zevekote, Belgium, and capable of supporting 11,000 fattening pigs and 5,400 piglets. The treatment facility's influent flow is estimated to be 120 m^3/day (IVACO, 2021) .

Table 5. LL# 1+2+6: Characteristics of liquid fraction pig manure

Values expressed in fresh matter basis.

Parameter	Unit	Value
Dry matter (DM) (%)	%	3.61 ± 0.05
Total Nitrogen (N)	kg/tonne	4.34 ± 0.13
NH_4-N	kg/tonne	2.86 ± 0.09
NO_3-N	kg/tonne	0.06 ± 0.003
Total Phosphorus (P)	kg/tonne	0.42 ± 0.003
Biological oxygen demand (BOD)	kg/tonne	4.58 ± 0.19
Chemical oxygen demand (COD)	kg/tonne	34.03 ± 0.82
Total Potassium (K)	kg/tonne	4.1

Scenarios S3 and S4 (LL#6) are theoretical and do not take place at the treatment facility. The data for S3 was obtained from the technology provider itself (Strocon Inc) and the data for S4 was obtained from another full-scale treatment facility (Waterleau New Energy). The complete life cycle inventory (LCI) for all scenarios has been uploaded on Github.

The inventory table is given in Table 6.

Table 6. LL# 1+2+6 LCA Inventory for all scenarios.

Unit process	Inputs/Outputs per unit process	Unit	Probability distribution	S1	S2	S3	S4
Nitrification-denitrification (NDN)	**Influent LF (functional unit)**	tonne		1	1	1	1
	Methanol	kg	Triangular	1.9-6.81	1.4-1.9		
	Electricity	kWh	Triangular	7.05-11.37	6.45-7		

Unit process	Inputs/Outputs per unit process	Unit	Probability distribution	S1	S2	S3	S4
	Ammonia	kg		0.02	0.02		
	Dinitrogen monoxide	kg		0.05	0.03		
	Sludge	kg		250	250		
	Biological effluent	kg		750	750		
Field application	**Transport of product**	t-km	Normal	4.7± 2	0.25±.02		
	Liquid manure spreading, by vacuum tanker	m3		0.75	0.025		
	Inorganic potassium fertiliser, as K_2O	kg		-4.24			
	Inorganic nitrogen fertiliser, as N				-2.2		
	Nitrate	kg	Normal	0.12	0.35		
	Ammonia	kg	Normal	2.00E-03	0.02		
	Dinitrogen monoxide	kg	Normal	2.30E-03	0.02		
Sludge management	**Transport of sludge**	t-km		46	46		55
	Phosphate fertiliser, as P_2O_5	kg		-0.77	-0.82		-0.21
	Inorganic nitrogen fertiliser, as N						-0.72
	Inorganic potassium fertiliser, as K_2O						-0.605
	Manure spreading	kg		230	46		220
	Dinitrogen monoxide						0.01
	Nitrate						0.16
	Ammonia	kg		0.01	0.01		0.03
	Methane	kg		0.33	0.33		0.33
Stripping and scrubbing	**Market for nitric acid, without water, in 50% solution state**	kg			4.54		
	Electricity	kWh			2.13		
	Tap water	kg			16		
	Stripped effluent	kg			1000		
	Ammonium nitrate	kg			25		
Constructed wetlands	**Biological effluent**	kg			750		
	Dinitrogen monoxide	kg			6.70E-04		
	Transformation, from arable land	m2-year			0.69		
Microfiltration	**Electricity (Trommel filter)**	kWh				0.18	
	Trommel filter rejects	kg				50	
	Effluent from trommel filter	kg				950	
	Electricity	kWh	Lognormal			1.49 ± 0.45	
	Retentate	kg				95	
	Permeate	kg				855	
Reverse osmosis	**Electricity**	kWh				3.56	
	Sulfuric acid	kg				2.09	
	Sodium hypchlorite	kg				9.50E-03	
	RO permeate	kg				213.75	
	Condensate from evaporator	kg				106.88	
	RO concentrate	kg				748.13	

Unit process	Inputs/Outputs per unit process	Unit	Probability distribution	S1	S2	S3	S4
Evaporator	**Electricity**	kWh	Triangular			20-25	22-23
	Antifoam agent, adipicacid	kg				0.11	0.58
Field application Evaporator concentrate	**Transport of product**	t-km				1.0685	0.94
	Inorganic potassium fertilizer, as K_2O	kg				-2.35	-4.34
	Inorganic nitrogen fertiliser, as N	kg				-1.42	-0.76
	Fertilizer spreading	m3				0.10	0.09
	Ammonia	kg				0.01	0.00
	Dinitrogen monoxide	kg				0.02	0.01
	Nitrate	kg				0.28	0.14
Field application retentate	**Fertilizer spreading**	m3				0.10	
	Transport of product	t-km				23.75	
	Inorganic nitrogen fertiliser, as N	kg				-1.02	
	Inorganic potassium fertilizer, as K_2O	kg				-2.57	
	Phosphate fertiliser, as P_2O_5	kg				-1.72	
	Ammonia	kg				0.01	
	Dinitrogen monoxide	kg				0.03	
	Nitrate	kg				0.40	
Aeration tank	**Electricity use**	kWh					19.00
	Iron chloride	kg					0.07
	Recirculated process water	kg					276.63
	Dinitrogen monoxide	kg					0.01
	Ammonia	kg					0.00
	Sludge from aeration tank	kg					220
Credits for condensed ammonia water	Ammonia	kg					-1.68
Aerobic treatment of Process water	**Electricity**	kwh					5.83
	Ammonia	kg					3.00E-04
	Dinitrogen monoxide	kg					1.90E-03

**Note: S1 represents nitrification-denitrification (NDN) and field application of effluent, S2 represents stripping and scrubbing as pre-treatment with NDN followed by post-treatment in constructed wetlands, Scenario 3 represents Membrane filtration and vacuum evaporation, and Scenario 4 represents vacuum evaporation*

Gaps in the inventory were filled using literature values. All calculations were performed using a combination of Brightway2 and Activity Browser (Mutel 2017, Steubing, de Koning et al. 2020). All background processes were modelled using the consequential ecoinvent database 3.8 (Wernet et al., 2016). The impacts were quantified using Environmental Footprint methodology (EC 2021). The results from the midpoint indicators were normalised and weighted to represent the best- and worst-case scenarios through a single score.

Additionally, the contribution analysis for *acidification*, *climate change*, *freshwater ecotoxicity*, and *terrestrial eutrophication potential* is further elaborated.

3.1.1.3 Results

3.1.1.3.1 Impact assessment – at midpoint

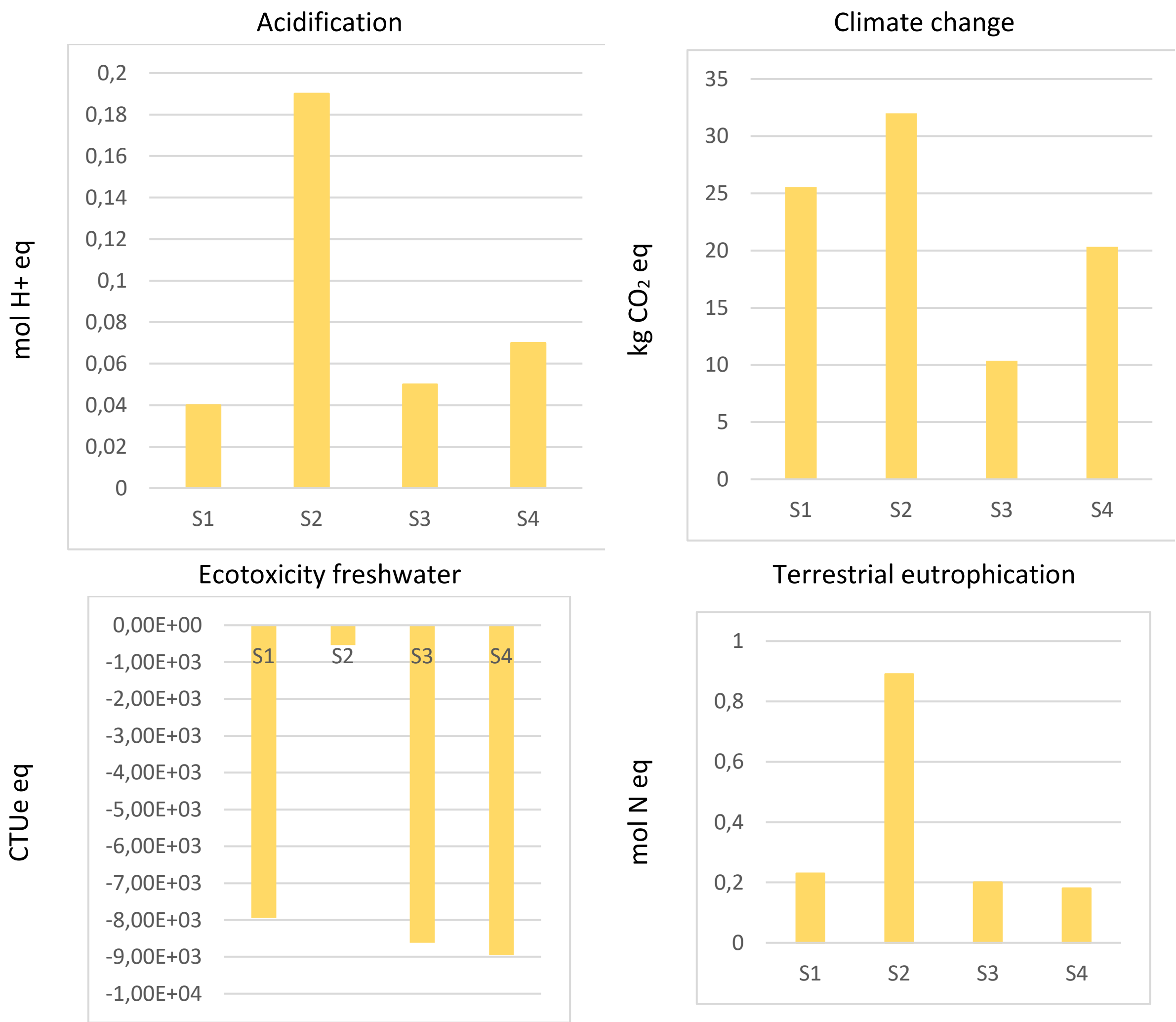


Figure 3. LL# 1+2+6 eLCA results – at midpoint.

Overall impacts at midpoint for select impact categories. S1: NDN; S2: Ammonium stripping and scrubbing as pre-treatment + NDN + constructed wetlands; S3: Reverse osmosis and Vacuum evaporation; S4: Vacuum evaporation

Climate change potential

The potential *climate change* impacts for S3 (median: 5.71 kg CO_2-eq) appeared to be the least relative to the other scenarios (S2: 31 kg CO_2-eq; S1: 24 kg CO_2-eq; S4: 16 kg CO_2-eq) (Figure 3).

The contribution analysis for the baseline, i.e. S1 showed that the majority of the burdens *from climate change potential* are due to fugitive N_2O emissions from NDN (14 kg CO_2-eq) as well as the energy demand for aeration (2 kg CO_2-eq) and methanol use for denitrification (4 kg CO_2-eq). Other significant contributors include the transportation and storage of sludge to P deficient regions (16 kg CO_2-eq). These burdens are offset by avoided synthetic K fertiliser use (-15 kg CO_2-eq) as a consequence of field application of the effluent from NDN. The results also showed that the environmental performance of S1 was highly dependent on the K fertilizer credits.

S2 (i.e. SAS + NDN + CW configuration) showed a 29% increase in potential *climate change* impacts relative to S1 (Figure 3). This increase is primarily due to the direct discharge of the effluent from the constructed wetlands and as a consequence, its constituent K leaves the system without fertiliser credits. Leaving the K fertiliser caveat aside, the inclusion of ammonia SAS in S2 reflected a benefit on the NDN step in S2, which showed a 38% reduction compared to the NDN in S1. This is primarily due to reduced N loading during NDN as a consequence of SAS, which harvests mineral N in the form of NH_4NO_3. The burdens from the SAS system (4.57 kg CO_2) were mostly as a consequence of nitric acid production. These emissions are partially offset by avoided production of synthetic N (-9 kg CO_2-eq) due to field application of NH_4NO_3.

S3, MF + ES (5.71 kg CO_2-eq) showed the least potential *climate change* impacts in relation to the other scenarios, although with high uncertainty. This can primarily be attributed to the up-concentration of N and K in the form of mineral concentrate (from RO and vacuum evaporator) and filtrate (residual fraction from microfiltration) and their subsequent field application (Retentate: -9 kg CO_2-eq and Mineral Concentrate: -9 kg CO_2-eq) create a net positive impact on *climate change potential*. The major burdens from S3 are due to the infrastructure for the evaporator (8 kg CO_2-eq) as well as its energy usage (7 kg CO_2-eq), which is higher compared to S1 and S2. The infrastructure burdens can be attributed to the stainless steel and brass needed for the evaporator, the RO, and ceramic membranes for microfiltration as well as the infrastructure modules for the ion exchanger.

S4 ranked as the second-best alternative in terms of *climate change potential*. Despite up-concentration of N and K, the energy use during aeration increases the net *climate change potential*. Furthermore, the burden from infrastructure is like S3, albeit lower since an ion exchanger is not present in this scenario. These burdens are however offset by benefits due to avoided N and K fertiliser from the mineral concentrate and process water. The use of ammonia water as a denoxing agent avoids the use of conventional ammonia thereby benefitting the system (-4 kg CO_2-eq).

Acidification potential

For *acidification potential*, S3 (median: 0.02 mol H^+-eq) performed the best relative to the other scenarios (S2: 0.18 mol H^+-eq; S1: 0.03 mol H^+-eq; S4: 0.09 mol H^+-eq) (Figure 3). The high acidification impacts in S2 can primarily be attributed to NH_3 emissions from field application of NH_4NO_3, since it has a higher emission factor (2.5% of TAN for arable land) when compared to mineral concentrates in S3 and S4 (0.64% of TAN).

Freshwater ecotoxicity potential

Freshwater ecotoxicity impacts are represented by the toxic effect on aquatic species in the water column and measured in comparative toxic unit equivalent (CTU-eq). The impacts due to *freshwater ecotoxicity potential* are a function of whether K fertiliser credits is awarded to the system. For scenarios S1, S3, and S4, K was supplemented through field application of products, reflecting an increased environmental benefit, whereas in S2, where the K is lost through effluent discharge, the scores showed a comparatively lower ecotoxicity benefit. The major influence on *freshwater* *ecotoxicity potential* is due to sulphur and chloride emissions during potassium chloride and potassium sulphate production respectively.

Terrestrial eutrophication potential

With respect to *terrestrial eutrophication*, N is the limiting factor and the impacts resulted from NH_3, and NO_3^- emissions due to field application of the ensuing products. Similar to *acidification potential*, S2 performed poorly relative to the other scenarios.

3.1.1.3.2 Impact assessment - at endpoint

The results at midpoint were normalised and weighted to a single score according to the Product Environmental Footprint guidelines (Figure 4). From the single scores it seems that using stripping and scrubbing (S2) as a pre-treatment prior to nitrification-denitrification is least beneficial to the environment.

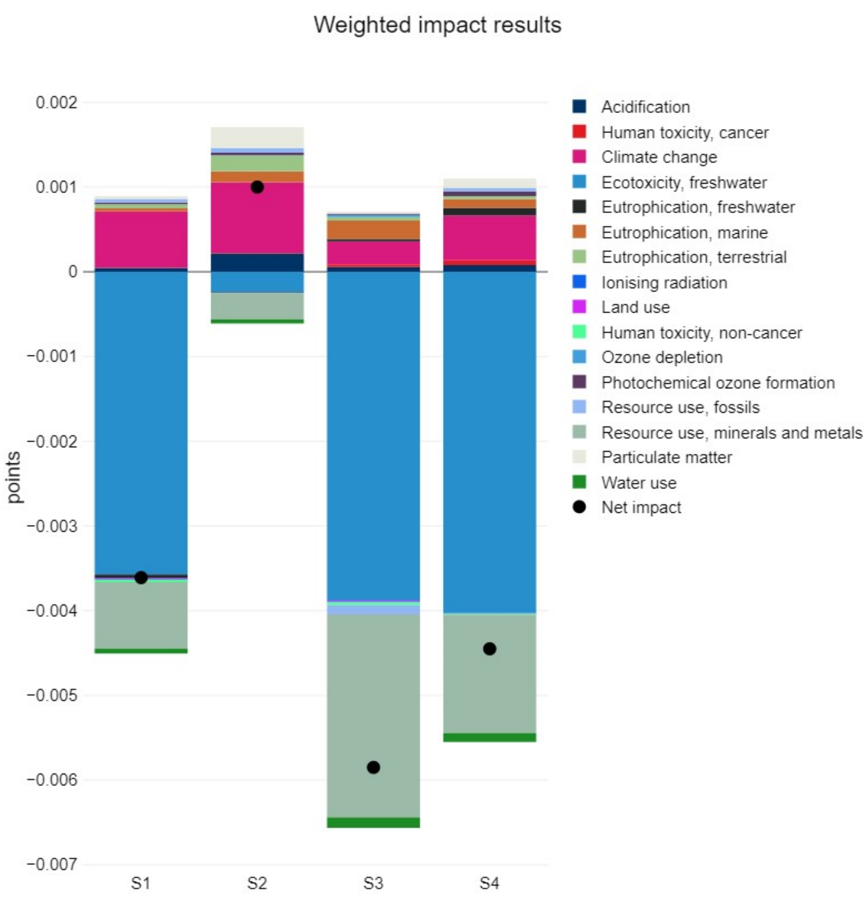


Figure 4. LL# 1+2+6: eLCA results – at endpoint.

Impacts at endpoint comparing scenarios for treating liquid manure. S1: NDN; S2: Ammonium stripping and scrubbing as pre-treatment + NDN + constructed wetlands; S3: Reverse osmosis and Vacuum evaporation; S4: Vacuum evaporation

3.1.1.4 Discussion

We conducted a comparative analysis of our study results with those from previous peer-reviewed works, despite differences in system perspectives and functional units specific to each study. In Table 7, we present an overview of our findings compared to scenarios and technologies from similar Life LCAs. The most relevant comparison appears to be with the study by Corbala-Robles et al. (2018), which examined the direct landspreading of pig manure versus treatment via NDN in Flanders using $1m^3$. Their study yielded inconclusive overall outcomes, with certain impact categories favouring direct landspreading while others favoured NDN. Notably, they identified NDN as an environmental hotspot for fugitive N_2O emissions and highlighted the impact of high energy demand, aligning with our results.

Our study revealed that incorporating a stripping and scrubbing process before NDN reduced its environmental burden due to decreased N loads. The burdens from stripping and scrubbing, related to acid and energy use, supported the observations of Vázquez-Rowe et al. (2015), who, however, focused on the direct field application of effluent from stripping and scrubbing. Additionally, we found that using HNO_3 partially offset the benefits of producing NH_4NO_3 and its associated fertiliser credits, as the production of HNO_3 through the Ostwald process has a high environmental footprint. To mitigate this, testing a scrubbing substitute with a lower environmental impact, such as organic acids (Brienza et al., 2020), could be explored.

Furthermore, our study involved a pilot facility implementing ammonia stripping and scrubbing without additional heat and pH control, resulting in a conservative NH_3 recovery from the LF at 29% N. Expert estimates suggest that on-site NH_3 recovery from LF could potentially be increased to around 50-60%, but this would require additional energy and auxiliary use.

Table 7. LL# 1+2+6: Comparison of results from this study versus other peer reviewed LCA studies.

Note that climate change potential has been considered since it is widely used impact category across the peer reviewed studies

Literature and technological process considered		Results from literature	Scenarios from this study that can be possibly compared
Finzi et al. (2020)	Anaerobic digestion, solid-liquid separation, nitrogen removal and field application	20.79 kg CO_2 eq $tonne^{-1}$ of treated manure	25.56 kg CO_2 eq$tonne^{-1}$ of LF manure using nitrification-denitrification and field application (S1)
Corbala-Robles et al. (2018)	Solid-liquid separation, nitrification-denitrification and field application of solid and liquid fraction	9.80 kg CO_2 eq/m^3	25.56 kg CO_2 eq/ $tonne^{-1}$ of LF manure using nitrification-denitrification and field application (S1)
Duan et al. (2020)	Composting solid fraction and using treated liquid fraction for microalgae cultivation and composting solid fraction and producing powder biofertilizers via struvite precipitation with ammonia stripping	-11 to 64.7 kg CO_2 eq $tonne^{-1}$ of PM treated	32 kg CO_2 eq/ $tonne^{-1}$ of LF manure using Ammonia stripping-scrubbing, nitrification-denitrification and treatment of effluent via constructed wetlands (S2)

Literature and technological process considered		Results from literature	Scenarios from this study that can be possibly compared
Vázquez-Rowe et al. (2015)	Solid, liquid separation, biological treatment, reverse osmosis and drying of digested PM	56.58 kg CO_2 eq/m^3	25.56 kg CO_2 eq/ $tonne^{-1}$ of LF manure using nitrification-denitrification and field application (S1) and 10.36 kg kg CO_2 eq/ $tonne^{-1}$ for reverse osmosis, vacuum evaporation and field application (S3)
	Ammonia stripping and drying of digested PM	68.41 kg CO_2 eq/m^3	32 kg CO_2 eq/ $tonne^{-1}$ of LF manure using Ammonia stripping-scrubbing, nitrification-denitrification and treatment of effluent via constructed wetlands (S2)
Feiz et al. (2022)	Solid liquid separation of digestate and ammonia stripping & scrubbing followed by field application	30 kg CO_2 eq/ $tonne^{-1}$	32 kg CO_2 eq/ $tonne^{-1}$ of LF manure using Ammonia stripping-scrubbing, nitrification-denitrification and treatment of effluent via constructed wetlands (S2)

3.1.1.5 Conclusion

According to the LCA, concentrating nutrients via reverse osmosis and/or vacuum evaporation outperforms the treatment trails of stripping-scrubbing, nitrification-denitrification and tertiary treatment in constructed wetlands as the most environmentally beneficial option for managing liquid fraction of manure.

We identified that fugitive N_2O emissions and energy demand during nitrification-denitrification of liquid fraction of manure are major environmental hotspots that can be reduced in part by introducing stripping and scrubbing as a pre-treatment step, but post-treatment of nitrification-denitrification effluent was the study's point of contention. The seemingly sub-optimal route of field application of nitrification-denitrification effluent demonstrated a net environmental benefit due to avoided K fertiliser as opposed to tertiary treatment in a constructed wetlands system, where the K is lost to surface water. The environmental benefits were mostly due to *freshwater ecotoxicity potential*, whose characterisation and weighting factors for inorganic compounds are highly uncertain. Therefore, these results must be cautiously considered. For liquid fraction management through reverse osmosis and vacuum evaporation, the production of mineral concentrate avoided the production of conventional N, K fertilisers causing a net environmental benefit. However, we recommend future studies to address the lifespan and data quality of the infrastructure concerning nutrient up-concentration.

Finally, because avoided synthetic fertiliser production affects the outcome of LCAs including nitrogen recovery from manure, future studies may include a socioeconomic variable connected to the geopolitical supply risk of crucial raw materials.

3.1.1.5 Dashboard indicators

LL#1+2+6 Ammonium stripping-scrubbing and vacuum evaporation of manure (*: Qualitative DBI assessment: **+** *improvement*, **o** *no change*, **-** *deterioration)*

Table 8. LL# 1+2+6: Comparison of dashboard indicator and life cycle assessment results

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication*	Comment	Quantitative estimate per FU (1 m^3 of liquid fraction of manure to be treated) [a]	LCA indication*	Re-evaluation of DBI irt. LCA results
Use of Primary Resources	**Rock Phosphate** (Reduction in mineral phosphorus consumption)	**LL1&2: +** **LL 6: +**	LL6: Fertiliser replacement	Phosphorus, 18% in apatite, 12% in crude ore, in ground S2: 0 kg S3: 0 kg S4: 0 kg	 **o** **o** **o**	Revised. LCA indicated no change for LL1+2 and LL6
	Natural Gas (Natural gas is consumed either as part of the electricity mix or to manufacture auxiliaries)	**LL1+2: +** **LL 6: +**	LL1+2: Reduction in relation to mineral fertilisers LL6: Fertiliser replacement	Gas, natural, in ground S2: -2.65 m^3 S3: -2.35 m^3 S4: -2.21 m^3	 **+** **+** **+**	Confirmed
	Oil	**LL1+2: o** **LL 6: -**	LL1+2: Both synthetic fertilisers and ammonium sulphate	Oil, crude, in ground [b] S2: -0.7 m^3 S3: -0.49 m^3	 **+** **+**	Revised. LCA indicated improvement for LL1+2 and LL6

[a]Scores reflect the difference between project and baseline scenarios. A negative value indicates improvement whereas a positive value indicates deterioration; S2(LL1+2): Ammonium stripping and scrubbing as pre-treatment + NDN + constructed wetlands; S3 (LL6): Reverse osmosis and Vacuum evaporation; S4 (LL6): Vacuum evaporation.

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication*	Comment	Quantitative estimate per FU (1 m^3 of liquid fraction of manure to be treated) [a]	LCA indication*	Re-evaluation of DBI irt. LCA results
	(Oil depletion is mostly linked to transport of the end-products)		/nitrate need machinery to be applied, so we assume no effect Fossil fuels are used during transport	S4: -1.2 m^3	+	
	Electricity	**LL1+2: -** **LL 6: +**	Electricity use to process LF LL6: Avoided electricity production due to biogas production	Electricity consumption S2: -1.04 kWh S3: 21.61 kWh S4: 31.57 kWh	+ - -	Revised. LCA indicated improvement for LL1+2 and deterioration for LL6
	Water (Reduction in water consumption)	**LL1+2: -** **LL 6: o**		Water scarcity S2: -5.719 m^3 S3: -1.259 m^3 S4: -1.519 m^3	+ + +	Revised. LCA indicated improvement for LL1+2 and 6
	Soil quality (Improvement in soil quality)	**LL1+2: +** **LL 6: +**		Soil quality S2: -15.67 pts S3: 13.73 pts S4: 29.03 pts	+ - -	Confirmed for LL1+2. Revised for LL6. LCA indicated deterioration
	Others? Please specify			no further info		

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication*	Comment	Quantitative estimate per FU (1 m^3 of liquid fraction of manure to be treated) [a]	LCA indication*	Re-evaluation of DBI irt. LCA results
Emissions to the environment	**Ammonia (air emission)** (Reduction in NH_3 emissions)	**LL1+2: o** **LL 6: o**		Ammonia emission to air S2: 0.01 kg S3: 0.01 kg S4: 0.03 kg	 - - -	Revised for LL1+2 and 6. LCA indicated deterioration
	Dinitrogen monoxide (air emission) (Reduction in N_2O Emissions)	**LL1+2: o** **LL 6: +**		Dinitrogen monoxide emission to air S2: 0.01 kg S3: -0.02 kg S4: -0.022 kg	 - + +	Revised for LL 1+2. LCA indicated deterioration Confirmed for LL6
	Methane (air emission) (Reduction in CH_4 emissions)	**LL1+2: o** **LL 6: -**		Methane emissions to air S2: 0.01 S3: -0.33kg S4: -0.33kg	 - + +	Revised. LCA indicated deterioration for LL1+2 and improvement for LL6
	Nitrates (water emission) (Reduction in NO_3 emissions)	**LL1+2: o** **LL 6: o**		Nitrate emission to water S2: 0.21 kg S3: -0.116 kg S4: -0114 kg	 - + +	Revised. LCA indicated deterioration for LL1+2 and improvement for LL6
	Phosphorus (water emission) (Reduction of P Emissions)	**LL1+2: -** **LL 6: o**		Phosphorous emission to water S2: -1.16E-07 kg S3: -1.30E-07 kg S4: -3.86E-07 kg	 + + +	Revised for LL1+2 and LL6. LCA indicated improvement

<table>
<tr><th>Indicator Dimension</th><th colspan="3">Qualitative Dashboard Indicator (DBI) assessment based on expert judgment</th><th colspan="3">Quantitative Dashboard Indicator based on LCA assessment</th></tr>
<tr><th></th><th></th><th>DBI Indication*</th><th>Comment</th><th>Quantitative estimate per FU (1 m^3 of liquid fraction of manure to be treated) [a]</th><th>LCA indication*</th><th>Re-evaluation of DBI irt. LCA results</th></tr>
<tr><td></td><td>Particulate matter
(Reduction of particulate matter formation)</td><td>LL1+2: NA
LL 6: -</td><td></td><td>Particulates emission to air, <2.5 µm

S2: -0.33E-03
S3: 1.53E-03
S4: -0.12E-03</td><td>

o
o
o</td><td>

Revised for LL1+2 and LL6. LCA indicated no change</td></tr>
<tr><td>Resilience to climate change</td><td>Carbon footprint
(Reduction of carbon footprint)</td><td>LL1+2: +
LL 6: +</td><td>The production of renewable energies and substitution of mineral fertilisers will reduce carbon footprint</td><td>Climate change:
S2: -4.96 kg
S3: -25.8 kg
S4: -16.75 kg</td><td>
+
+
+</td><td>
Confirmed</td></tr>
<tr><td></td><td>Effective SOM
(Effective Soil Organic Matter Improvement)</td><td>LL1+2: -
LL 6: o</td><td></td><td>Carbon | Emission from soil

Carbon dioxide | Emission from soil</td><td>o

o</td><td>Carbon:

Carbon dioxide:</td></tr>
<tr><td></td><td>Renewable energy production
(Renewable energy produced from biomass)
Others? Please specify</td><td>LL1+2: -
LL 6:</td><td>NA</td><td></td><td></td><td>This cannot as such be seen in an LCA</td></tr>
</table>

3.1.1.5.1 Comparison of LCA and DBI results

The outcome from the LCA was concurrent to the DBI outcome for rock phosphate (+), oil (+) and natural gas usage (-) for both LL1+2 as well as LL6. The increase in use of natural gas for LF processing is a consequence of electricity usage, since the grid mix in both BE as well as NL produces around 20% of electricity using natural gas. The DBI pointed out that processing LF through LL1+2 and 6 showed a reduction in carbon footprint relative to the baseline, and this aspect was corroborated by the LCA.

Also, the LCA results for electricity consumption showed deterioration, in contrast to the DBI outcome.

3.1.1.6 References

Aben, R. C. H., E. S. Oliveira Junior, A. R. Carlos, T. J. H. M. van Bergen, L. P. M. Lamers and S. Kosten (2022). "Impact of plant species and intense nutrient loading on CH4 and N2O fluxes from small inland waters: An experimental approach." Aquatic Botany 180: 103527.

AICE (2018). Life Cycle Assessment of REMEB MBR: 21.

Strocon Inc (2021). Reverse osmosis and evaporation to treat liquid manure-Mass and energy flows.

ATV-DVWK, S. (2000). Dimensioning of Single-Stage Activated Sludge Plants.

Brienza, C., J. van Puffelen, I. Regelink, H. Dedeyne, A. Giordano, M. Schepis, U. Bauermeister, T. Meier and I. Sigurnjak (2021). Fourth annual updated report on mass and energy balances, product composition and quality and overall technical performance of the demonstration plants (year 4).

Corbala-Robles, L., W. N. D. Sastafiana, E. I. Volcke and T. Schaubroeck (2018). "Life cycle assessment of biological pig manure treatment versus direct land application− a trade-off story." Resources, Conservation and Recycling 131: 86-98.

Corbella, C., J. Puigagut and M. Garfí (2017). "Life cycle assessment of constructed wetland systems for wastewater treatment coupled with microbial fuel cells." Science of The Total Environment 584-585: 355-362.

Doka, G. (2021). A model for composition-specific life cycle inventories of regionalised wastewater fates. Doka Life Cycle Assessments, Zurich, Switzerland.

Duan, N., Khoshnevisan, B., Lin, C., Liu, Z., & Liu, H. (2020). Life cycle assessment of anaerobic digestion of pig manure coupled with different digestate treatment technologies. Environment International, 137, 105522. https://doi.org/10.1016/j.envint.2020.105522

EC – European Commission, 2021. Commission Recommendation on the use of the Environmental Footprint methods to measure and communicate the life cycle environmental performance of products and organisations. https://ec.europa.eu/environment/publications/recommendation-use-environmental-footprint-methods_en

Fazio, S., Biganzioli, F., De Laurentiis, V., Zampori, L., Sala, S., & Diaconu, E. (2018). Supporting information to the characterisation factors of recommended EF Life Cycle Impact Assessment methods, version 2, from ILCD to EF 3.0 (E. Commission, Ed.). https://doi.org/10.2760/002447

Feiz, R., Carraro, G., Brienza, C., Meers, E., Verbeke, M., & Tonderski, K. (2022). Systems analysis of digestate primary processing techniques. Waste Management, 150, 352–363. https://doi.org/https://doi.org/10.1016/j.wasman.2022.07.013

Finzi, A., Mattachini, G., Lovarelli, D., Riva, E., & Provolo, G. (2020). Technical, Economic, and Environmental Assessment of a Collective Integrated Treatment System for Energy Recovery and Nutrient Removal from Livestock Manure. Sustainability, 12(7), 2756. https://doi.org/10.3390/su12072756

Guida, S., L. Conzelmann, C. Remy, P. Vale, B. Jefferson and A. Soares (2021). "Resilience and life cycle assessment of ion exchange process for ammonium removal from municipal wastewater." Science of The Total Environment 783: 146834.

Henze, M., W. Gujer, T. Mino and M. C. van Loosdrecht (2000). Activated sludge models ASM1, ASM2, ASM2d and ASM3, IWA publishing.

IVACO (2021). Daily inflow at the manure treatment facility at Gistel-Zevekote, Belgium.

Jie, Y., Buisonjé, F., Melse, R. 2017. Livestock manure treatment technology of the Netherlands and situation of China: white paper. Wageningen Livestock Research.

Mutel, C. (2017). "Brightway: an open source framework for life cycle assessment." Journal of Open Source Software 2(12): 236.

Ravi, R., Beyers, M., Vingerhoets, R., Brienza, C., Luo, H., Bruun, S., & Meers, E. (2023). In the quest for sustainable management of liquid fraction of manure - Insights from a life cycle assessment. Sustainable Production and Consumption. https://doi.org/10.1016/j.spc.2023.11.006

Reuland, G., Sigurnjak, I., Dekker, H., Michels, E., Meers, E. 2021. The Potential of Digestate and the Liquid Fraction of Digestate as Chemical Fertiliser Substitutes under the RENURE Criteria. Agronomy, 11(7), 1374.

Roefs, J. 2021. Treatment of surplus manure in the Netherlands, personal email communication.

Steubing, B., D. de Koning, A. Haas and C. L. Mutel (2020). "The Activity Browser—An open source LCA software building on top of the brightway framework." Software Impacts 3: 100012.

Tampio, E., Marttinen, S., Rintala, J. 2016. Liquid fertilizer products from anaerobic digestion of food waste: mass, nutrient and energy balance of four digestate liquid treatment systems. Journal of Cleaner Production, 125, 22-32.

Vázquez-Rowe, I., Golkowska, K., Lebuf, V., Vaneeckhaute, C., Michels, E., Meers, E., Benetto, E., & Koster, D. (2015). Environmental assessment of digestate treatment technologies using LCA methodology. Waste Management, 43, 442–459. https://doi.org/https://doi.org/10.1016/j.wasman.2015.05.007

Vondra, M., Máša, V., & Bobák, P. (2018). The energy performance of vacuum evaporators for liquid digestate treatment in biogas plants. Energy, 146, 141–155. https://doi.org/https://doi.org/10.1016/j.energy.2017.06.135

3.1.2 L#11: Alternative cow bedding from recycled manure (SOLTUB)

Longlist #11 title: Recycling fibres of manure as organic bedding material for dairy cows

Zoltán Hajdu[1], Francisco Verdugo[2]

[1] SOLTUB Ltd., Hungary

[2] CARTIF Technology Centre, Valladolid, Spain

3.1.2.1 Introduction

Greenhouse gases emitted from dairy farms, including CO_2, CH_4, and N_2O, exhibit varying levels, influenced by different manure management practices that generate products with different physical and chemical characteristics and environmental impacts. Common approaches for recycling animal slurries involve mechanical separation (1, 2) and biological treatment in biogas plants. The resulting and remaining digested solid manure can then be recycled into cow bedding.

An attributional LCA study was conducted to assess the environmental impacts of utilising organic bedding material in dairy cow production at two Hungarian farms, specifically at a conventional dairy farm three scenarios:

Scenario A: Using straw as bedding material (*short* value chain),

Scenario B: Using separated solid manure as bedding material (*medium* value chain)

Scenario C: Using separated solid manure as bedding material supplemented with 10% dried and sanitised solid digestate (*long* value chain)

For the environmental impact evaluation, two calculation methods were employed in the LCA study for each scenario. The first, without LCA software, utilised a reduced impact category assessment involving climate change, acidification, and freshwater eutrophication. The second LCA assessment employed dedicated LCA software, assessing 28 impact categories.

3.1.2.2 Materials & methods

3.1.2.2.1 Goal & scope

The study aims to offer manure management recommendations by assessing the environmental footprint of bedding materials in the three scenarios mentioned. It involves determining emission hotspots, prioritising reduction opportunities, and employing two assessment procedures: manual calculations using IPCC (2006) guidelines for CO2 emissions, soil acidification, and water eutrophication, and software (Sima Pro) for the same factors along with additional complementary factors.

The specific aim of the LL#11 LCA study was to assess the environmental impacts of recycled cow bedding material in two Hungarian dairy farms. The evaluation encompassed three scenarios representing short, medium, and long value chains, with diverse inputs, outputs, barn emissions, and energy consumptions. The use of straw as bedding involves straw collection from the fields, increasing the field carbon scarcity, but may also reduce direct and indirect N_2O emissions from the field derived from crop residues which differ between the slurry and digestate cases, where is an addition of nutrients and we count soil emissions due to crop residues. The functional unit was defined as one tonne of bedding material.

The LCA bedding study was conducted from 2019 to 2022, focusing on Hungary. In the case of digestate from the biogas plant, the calculation considered carbon savings attributed to the avoidance of external energy and heat use. Additionally, both direct and indirect emissions were assessed.

The main technologies in providing the recycled bedding material can be grouped as:

1. feed production and consumption:

    a) feed production: own feed, purchased feed, silage, or fodder feed (t/year),
    b) feed transportation: to farm and inside the farm, inventory of machinery (km/t),
    c) feed storage: type of storage and storage duration, (t/year),
    d) feed consumption: animal head, animal categories, feed rate composition (kg/head),
    e) animal products: manure, milk, meat (kg/head),
    f) product transport (km/t),

2. recycled bedding:

    a) number of cows using bedding: detailed on growing groups, animal rotations (head/year),
    b) bedding material: amount /per animal /year and quality (t),
    c) bedding transportation: into the stall and from the stall (km/t),
    d) bedding composition: straw, separated solid manure, separated solid digestate, (t/year),
    e) bedding treatment (sanitation by liming) (t/year),
    f) bedding collection (t/year),

3. manure processing (slurry separation by screw press and fermentation at biogas unit) as:

    a) amount and quality of inputs materials (t/year),
    b) amount and quality of output materials, solid fraction, and liquid fraction (t/year),
    c) energy consumption (kWh/t),
    d) digestate separation: separated digestate liquid fraction (m3) and solid fraction (t),
    e) energy consumption for separation digestate (kWh/t),
    f) storage of liquid fraction (m3/year) and duration (month),
    g) storage of solid fraction (t/year) and duration (month),
    h) solid manure transport within the farm (km/t),
    i) liquid manure transport within the farm (kWh/m3),

4. manure and bedding material distribution into fields:

    a) transportation of the liquid fraction onto fields (km/t),
    b) distribution of the liquid fraction onto fields (km/t),
    c) transportation of the solid fraction and bedding onto the fields (km/t),
    d) distribution of the solid fraction and bedding onto fields (km/t),

The system boundaries for the study were defined based on the following criteria::

1. the number of animals of a particular category that are present, on average, within the year, not taken into account the different feeding requirements for all livestock categories, e.g. calf, 12 months cattle or alder cattle, or selected cattle (non-productive) and the different amounts of produced manure based on animal categories. On average we deal with 860 cattle/year in all scenarios.

2. the consumed feed is an average amount of 45 kg per day, and the produced manure is an average of 38.7 tons per day, equivalent to 14,125 tons per year. Water consumption is not included in the calculations.

3. emissions during storage consider the proportion of manure in stalls for 6 months and the proportion of manure stored outside the stalls for another 6 months. Slurry storage tanks are emptied once a year. From the total produced manure, 50% is considered to be separated at farm in scenario B and 50% is considered to be separated digestate at farm in scenario C.

4. in scenarios B and C, after mechanical separation of the slurry, 30% is considered as the solid fraction, and 70% as the liquid fraction. In scenario C, 10% of the total dried digestate is used as dried solid digestate as a supplement to the separated solid digestate, along with liming as a sanitising material.

5. for the bedding application onto fields, the system boundaries do not account for emissions during field application for the entire emission period (next 1-3 years), which is part of the subsequent generation of crop production.

The system flows and boundaries for the three scenarios are illustrated in Figure 5 for straw, Figure 6 for separated solid manure and Figure 7 for separated solid digestate with 10 % dried solid digestate. The boxes represent the main flow steps, while the arrows indicate the flow of materials and energy.

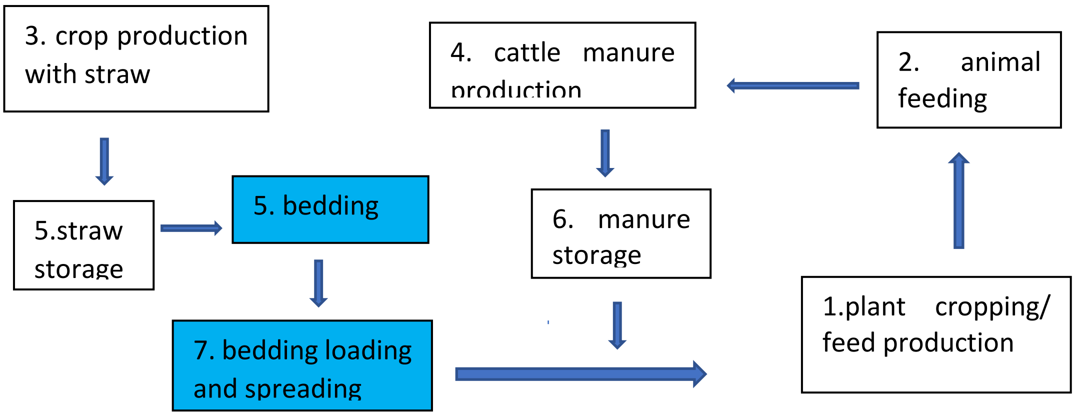


Figure 5. LL# 11: system boundaries - 1

LCA scenario for straw production as bedding

Note: the energy use for transportation is included for each bedding flow step. The feed storage, the manufacturing of the equipment used for cow feeding, manure collection and spreading on the fields and the construction of the manure and straw storage facilities and stalls are not included within the boundary.

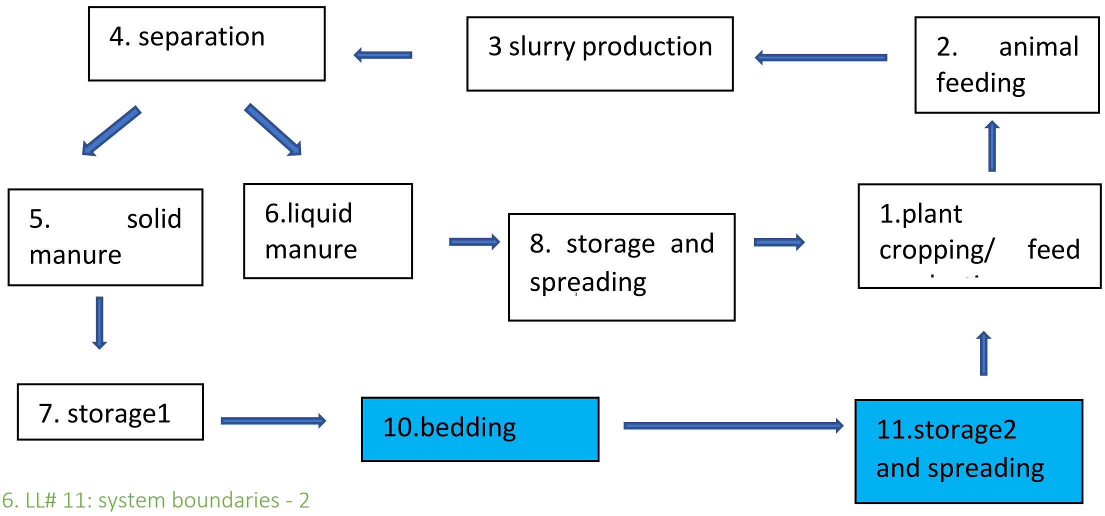


Figure 6. LL# 11: system boundaries - 2

LCA scenario for separated solid manure as bedding

Note: the energy use for transportation is included for each bedding flow step. The feed storage, the manufacturing of the equipment used for cow feeding, manure processing, recycled bedding spreading on the fields and the construction of manure storage facilities and stalls are not included within the boundary.

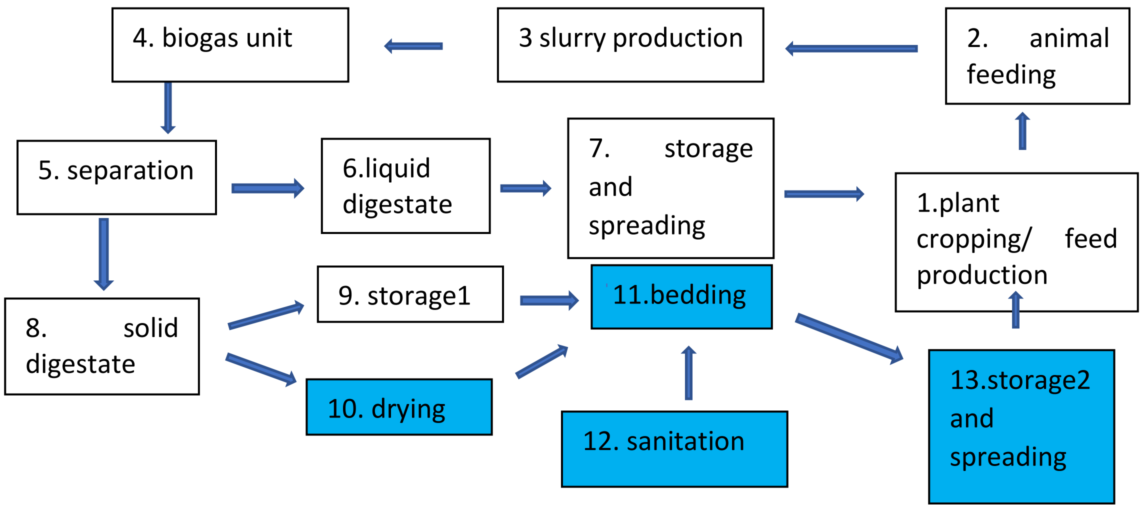


Figure 7. LL# 11: system boundaries - 3

LCA scenario for separated solid digestate as bedding

Note: The feed storage, the manufacturing of the equipment used for cow feeding, manure processing including the AD plant and the manure drying, the equipment for recycled bedding spreading on the fields, the construction of manure storage facilities and stalls are not included within the boundary.

In the manual computation of the three scenarios, we considered the impact on climate change, freshwater eutrophication, and acidification resulting from the production and utilisation of the recycled bedding material. In the review process, 28 impact categories were assessed using SimaPro software in collaboration with CARTIF.

3.1.2.2.2 Inventory

Primary data for the three scenarios were collected from the two involved Hungarian farms. This primary data was supplemented with secondary data from databases such as Ecoinvent 3.5. Secondary data were primarily used for transforming material and energy inputs and outputs into impact categories. In some cases, we utilised the results of other LCA calculations from sources like former projects, FADN, surveys, scientific papers, and experts' knowledge. For agricultural tractor fuel consumption, we referred to the Hungarian NAIK mechanical institute consumption list and farmer communications (primary data). Conversion factors were obtained from the CML Leiden database.

According to available studies on recycled manure solids (3,4) and available primary data for slurry composition we considered 9-11 % dry matter, 4,2-5,2 kg/t total N, 2-2,5 kg/t NH_4, 0,8-1,6 kg/t P_2O_5, 3,5-4,5 kg/t K_2O. For digestate composition we considered: 5-6% dry matter, 3-5 kg N total/t, the NH_4 ration of 60% from total N, 0,8-1,4 kg P_2O_5/t.

In the production of bedding material, we applied the IPCC 2006 guidelines for corn silo production, soil N_2O direct and indirect emissions during transport within and to the farm, CH_4 emissions due to digestion and animal housing, indoor direct and indirect N_2O emissions, as well as CH_4 emissions from manure storage. For the consumption of electric engines, raw data from farmers were utilised. In the utilisation of bedding material, we used the IPCC 2006 guideline for indirect N_2O emissions from bedding storage, soil direct and indirect N_2O emissions from manure, and emissions during transport within and to the farm, bedding transport, and field distribution based on farmers' raw data.

We followed the IPCC Guideline 2006, specifically Chapter 10 for emissions from livestock and manure management (Tier 1 approach) and Chapter 11 for N_2O emissions from managed soils and CO_2 emissions from lime and urea application (Tier 1 approach). Capital goods were excluded from the system boundaries as their impact was expected to be insignificant. In the GHG assessment, conversion rates for CH_4 (24 kg CO_2) and N_2O (298 kg CO_2, GWP 100 values) were used. The manual calculations involved multiplying activity data by emission factors, expressed in CO_2eq. *Climate change* impacts (CO_2eq) were used to calculate *acidification* (SO_4) and *freshwater eutrophication* (PO_4) impacts through conversion factors. For the *acidification potential* (SO_2) of bedding production and utilization, 16 kg NH_4NO_3/t separated solid manure was considered, with a transformation coefficient of 0.8 kg SO_2/kg NH_4NO_3. For *freshwater eutrophication potential* (PO_4), it was assumed that the separated fraction has 16 kg NH_4NO_3/t and 1 kg P_2O_5/t. The calculations used transformation coefficients of 0.33 kg PO_4/kg NH_4 and 3.07 kg PO_4/kg total P. For slurry and digestate, a ratio of 30% solid manure and 70% liquid manure was considered. A mass allocation was applied for slurry and digestate concerning the produced meat and milk, based on farmers' annual data.

3.1.2.3 Results

3.1.2.3.1 Impact assessment

Two different calculation procedures were applied as a manual calculation and a SimaPro software review. The results of the manual calculations are presented in Table 9.

Table 9. LL# 11: Impact assessment results (manual calculations) per ton of bedding material.

	scenario A. straw	scenario B. solid manure	scenario C. solid digestate
climate change (kg CO_2e/t)	122	58	59
acidification (kg SO_2e/t)	6	1.8	1.8
freshwater eutrophication (kg PO_4e/t)	5.1	1.6	1.5

The Figure 8 presents the same impact factors without credits by avoided inorganic fertilisers on year base.

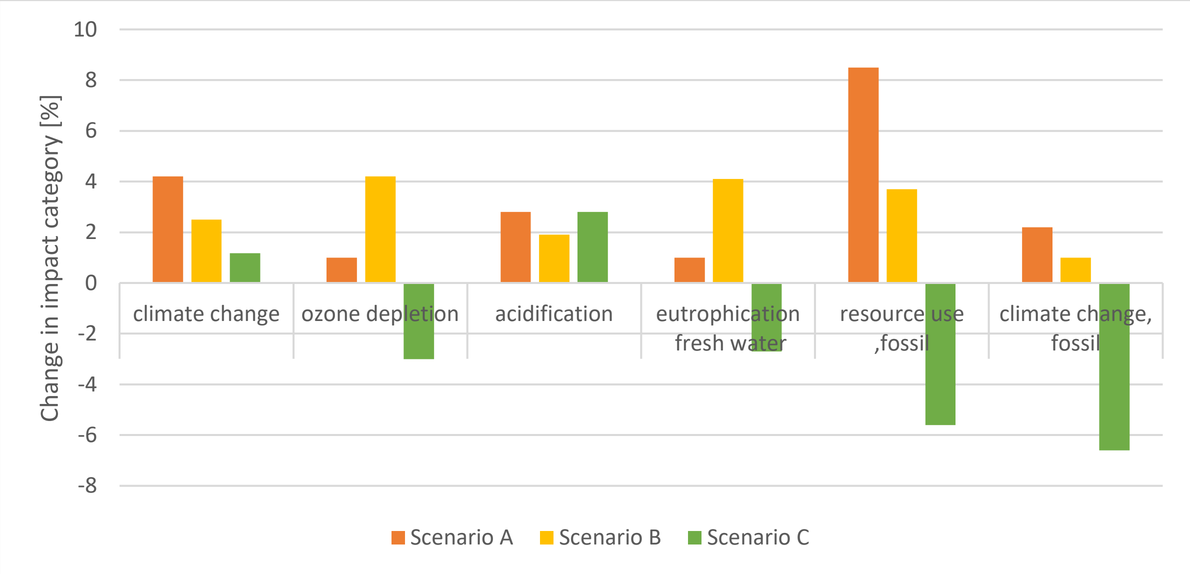


Figure 8. LL# 11: eLCA results – 1,

given as % change in impact category, relative to the baseline. The environmental impact of bedding in the three scenarios. Note: in the scenario C with digestate as recycled bedding in the software calculations 2.887.961 kg CO_2e avoided emissions are included due to the use of own green energy.

The data in Figure 9 indicates that there are no net benefits for straw bedding, solid manure as bedding, and digestate as bedding scenarios. In the straw scenario, *resource use (fossil)* has the highest impact at 8%, followed by *climate change* at 4%, and lower impacts for *climate change (fossil)* and *acidification* at 2%. The solid manure scenario mainly impacts *ozone depletion* and *water eutrophication* at 4%, with lower impacts on *resource use (fossil)*, *climate change* and *acidification* at 2%. Using digestate as bedding shows almost a net benefit for *climate change (fossil)* at -6%, *fossil resource use* at -5%, and negative impacts on *ozone depletion* and *eutrophication* at -2.5%. Scenario C has a positive impact on *terrestrial acidification* at 2.5% and less than 1% on *climate change*.

The data in Figure 9 indicates that there is no net benefit for the straw scenario, and the only net benefit is obtained for the solid manure scenario in *ozone depletion* with -6%. The straw scenario has the highest impact on *climate change*, almost 8%, followed by *acidification* and *resource use (fossil)* with 4%. There is less than 1% impact in *ozone depletion*. The scenario for solid manure as bedding is controversial since it has negative values in *ozone depletion* at -6%, and high positive values, almost 8%, in *acidification* and *eutrophication*. The best scenario is digestate as bedding, where there is a net benefit for five impact categories, with the only positive impact on *climate change* at 8%.

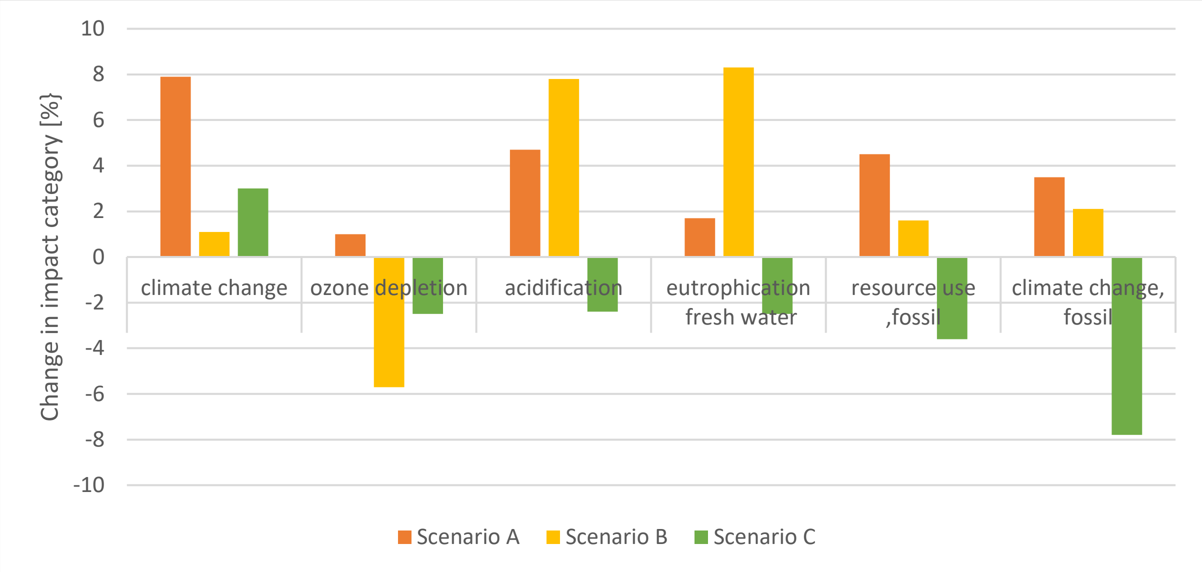


Figure 9. LL# 11: eLCA results – 2.

given as % change in impact category, relative to the baseline. The environmental impact of the performed three bedding scenarios, without credits by avoided inorganic fertilisers, on a yearly base.

3.1.2.4 3.1.2.4 Interpretation

Manure management technologies, including slurry separation and digestion in biogas plants, can offer environmental benefits. The net environmental benefits are most significant for digestate as bedding in the year-based case, except for *climate change* in the selected six impact categories. In the ton-based scenario, a similar situation of net environmental benefits occurs, except for *climate change* and *acidification*. For the solid manure as bedding scenario in the year-based case, net benefits occur for *ozone depletion*, with high positive values for *acidification* and *water eutrophication*. Similarly, in a ton-based case, the highest impact is on *eutrophication*, and non-similarly, on *ozone depletion*.

In a ton bedding impact assessment, the scenario on digestate as bedding is the most environmental friendly technology as presenting net benefit in four of six impact categories, except *climate change* and *acidification*. In the year-based calculation, we have the same net environmental benefits including the *acidification*. The only positive impact is for the *climate change*. Among the three scenarios assessed, the scenario with digestate as bedding is the most environmentally friendly technology, followed by the scenario with solid manure as bedding, and finally the scenario with straw as bedding. Beside the manure processing and storage technologies, other factors can have a great impact on emissions (due to lack of data it was less considered in the LCA study). Such factors can be related to animal feeding (mostly the variation in feed composition and feed storage), dairy herd composition (different composition of animal manure), bedding field application technologies (e.g. trailing shoes, trailing hose, shallow injection, deep injection). The water usage in the stalls have impact on the slurry composition (dry matter content, pH- value). The use of dried digestate and sanitation applied in case of scenario digestate as bedding proved its efficiency, more than that is a manure treatment option when the biogas heat is used to recycled manure sanitation treatment.

3.1.2.5 Conclusion

The LCA Study for LL11 compared two manure recycling strategies against the standard straw bedding application in two Hungarian dairy farms in order to assess the environmental performance of the bedding scenarios.

The scenario results show that the best environmental performance, providing net benefits for the selected six impact categories, is obtained with the digestate as bedding scenario for *water eutrophication* and *terrestrial acidification*. It is followed by the scenario with solid manure as bedding having environmental net benefits in the year-based ozone depletion. The worst scenario is the straw bedding scenario, having no net environmental benefits, rather a high *climate change* and *fossil resource use* impact.

The study shows that the use of separated solid manure and separated solid digestate as bedding material in dairy production are approved technologies to close nutrient cycles as well as in reducing the environmental impact of agricultural technologies. The considered two manure processing scenarios compared with the basic straw option provide several opportunities to avoid emissions in dairy farms. Emissions hot spots are mostly due to CO_2, CH_4, and N_2O emissions.

3.1.2.6 Dashboard indicators

LL#11 Recycling fibres of manure as organic bedding material for dairy cows (*: Qualitative DBI assessment: **+** *improvement*, **o** *no change*, **-** *deterioration)*

Table 10. LL# 11 Comparison of dashboard indicator and life cycle assessment results

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication *	Comment	Quantitative estimate per FU (1 t of bedding)	LCA indication *	Re-evaluation of DBI irt. LCA results
Use of Primary Resources	**Rock Phosphate** (Reduction in mineral phosphorus consumption)	o	Not relevant	Scenario B: 0 kg P Scenario C: 0 kg P	O O	Not relevant
	Natural Gas (Natural gas is consumed either as part of the electricity mix or to manufacture auxiliaries)	o	Not relevant	Scenario B: 0 kg P Scenario C: 0 kg P	O O	Not relevant
	Oil (Oil depletion is mostly linked to transport of the end-products)	o	Not relevant	Negligible for the diesel fuel consumption	O	Not relevant
L	**Electricity**	-	Electricity is needed for separation, but with bioenergy production it could be +	Scenario B: 4380 kW Scenario C: 5146 kW	- -	Confirmed, actually electricity consumption is higher in both cases
	Water	+	Less water is used	Scenario B: -1.36 m3	+	Confirmed

	(Reduction in water consumption)			Scenario C: -5.12 m3	+	
	Soil quality (Improvement in soil quality)		Not relevant	Not assessed		
	Nutrient recovery (Increase in nutrient recycling)	+	Reduced impact by closing the nutrient loops by recycling	Scenario B: 0.3 t N Scenario C: 0.34 t N	+ +	Confirmed
	Others? Please specify					
Emissions to the environ-ment	**Ammonia (air emission)** (Reduction in NH_3 emissions)	+	Reduced ammonia emission	Scenario B: n.d. kg NH_3 Scenario C: n.d. kg NH_3		n.d. = not determined in this LCA
	Dinitrogen monoxide (air emission) (Reduction in N_2O Emissions)	+	Reducing the direct and indirect N2O losses due to soil and crop residue emissions, manure storage	Scenario B: n.d. kg N_2O Scenario C: n.d. kg N_2O		
	Methane (air emission) (Reduction in CH_4 emissions)	+	Reduced methane emission	Scenario B: n.d. kg CH_4 Scenario C: n.d. kg CH_4		
	Nitrates (water emission) (Reduction in NO_3 emissions)	-	Increased nitrate leaching	Scenario B: n.d. kg NO_3 Scenario C: n.d. kg NO_3		
	Phosphorus (water emission) (Reduction of P Emissions)	+	Decreased impact as reduce eutrophication of water bodies	Scenario B: 6.6 kg P eq. Scenario C: -4.8 kg P eq.	- +	Not confirmed Confirmed

	Particulate matter (Reduction of particulate matter formation) ***Others? Please specify***	+	Reduced impact	Scenario B: n.d. kg fPM Scenario C: n.d. kg fPM		
	Carbon footprint (Reduction of carbon footprint)	+	Reduced impact	Scenario B: -6.93 kg CO_2e Scenario C: -4.9 kg CO2e	+ +	Confirmed Confirmed
Resilience to climate change	**Effective SOM** (Effective Soil Organic Matter Improvement)	+	Increased SOM from slurry and digestate application	Not assessed		
	Renewable energy production (Renewable energy produced from biomass)	**0**		Scenario B Scenario C	**o** +	Confirmed Revised as C green energy is produced in the biogas unit

3.1.2.6.1 Comparison of LCA and DBI results

Comparing the LCA results with dashboard indicators there are similarities (+ and -), differences, (+ and -) and common non relevant aspects. The common non relevant aspects are the rock phosphate, natural gas and oil consumption, also. Common negative impact is for electricity, water use and human toxicity for scenario B and C. Common positive impact are on the reduction of the carbon footprint, effective SOM, N_2O, CH_4 and P emission, nutrient recovery, soil quality, particulate matter. For the dashboard indicators earlier assessed by IRTA not relevant criteria was considered for soil quality, nitrates, effective soil organic matter and P water emission.

3.1.2.7 References

1. Lpelc. (2019): Manure Separation: Bedding and Nutrient Recovery. Online article available at https://lpelc.org/manure-separation-bedding-and-nutrient-recovery/ (accessed Dec 2023)

2. Stallkamp (2018): Separation systems for Cow Manure (online resource available at https://www.stallkamp.de/en/references/separation-system-for-cow-manure (accessed Dec 2023)

3. Garcia A. and Diaz-Royón F. (2019): Recycled Manure Solids as Bedding. Online article available at https://dairy-cattle.extension.org/recycled-manure-solids-as-bedding/ (accessed Dec 2023).

4. Fournel, S., Godbout, S., Ruel P., Fortin, A., Genereux, M., Coté, C., Landry, C. & Pellerin, D. J. J. O. D. S. (2019): Production of recycled manure solids for bedding in Canadian dairy farms: I. Solid–liquid separation. Journal of Dairy Science, 102, 1832-1846.

3.1.3 LL#18: Pig slurry acidification in the outdoor storage under Danish, Dutch, and Spanish conditions (UCPH)

Longlist #18 title: Slurry acidification with industrial acids to reduce NH_3 volatilisation from animal husbandry

Miriam Beyers[1,2], Yun-Feng Duan[1,3], Lars Stoumann Jensen[1], Sander Bruun[1]

[1] Department of Plant and Environmental Sciences, Faculty of Science, University of Copenhagen, Denmark

[2] RE-SOURCE LAB, Laboratory of BioResource Recovery, Department of Green Chemistry and Technology, Faculty of Bioscience Engineering, Ghent University, Belgium

[3] Sustainable Soil Management Group, Wageningen Environmental Research, the Netherlands.

This chapter builds on the published scientific paper by Beyers et al. (2022) Effect of natural and regulatory conditions on the environmental impacts of pig slurry acidification across different regions in Europe: A life cycle assessment. J.Cle.Pro https://doi.org/10.1016/j.jclepro.2022.133072

3.1.3.1 Introduction

Slurry acidification aims to reduce ammonia emissions throughout the manure management chain (Fangueiro et al., 2015; Hou et al., 2015). By lowering the pH of animal slurry, the equilibrium between NH_4^+ and NH_3 shifts towards NH_4^+, inhibiting easy volatilisation of NH_3. A lower slurry pH also hinders microbial degradation of organic matter, reducing methane (CH_4) formation and emissions during storage (Petersen et al., 2014, 2012).

However, introducing an additional treatment step often involves trade-offs. In the case of slurry acidification, trade-offs arise from the production and provision of acid, leading to material and energy consumption. Moreover, more nitrogen is retained in the slurry, potentially enhancing field yields but posing the risks of increased losses of other forms of nitrogen (e.g. nitrate leaching or denitrification). These specific impacts may vary based on local environmental factors (climate, soils, agricultural crops) and legislative settings (legal regulations or taxes/subsidies on agricultural practices or emissions).

3.1.3.2 Materials & methods

3.1.3.2.1 Goal & scope

The goal of this life cycle assessment study was to assess the environmental impacts associated with implementing slurry acidification in three European regions characterised by intense pig production and subject to varying environmental and regulatory conditions. The study compares two scenarios in each region: unacidified (UA) and acidified slurry (AS). Case-study areas representing European regions include Denmark, Limburg in the Netherlands, and Catalonia in Spain.

We selected a functional unit (FU) of *handling of 1,000 kg of slurry-N entering the outdoor storage.* A graphic representation of system boundaries and essential processes can be found in Figure 10.

The modelled system serves two functions, referred to as multi-functionality: handling pig slurry and producing crops. Due to the chosen functional unit, crop yields are considered a 'by-product' of slurry handling. As these yields may vary between scenarios (UA vs. SA) due to the potential impact of acidification, the system needs to be balanced concerning available crops on the global market.

In consequential LCA modelling, multi-functionality is addressed through system expansion, assuming that the crops produced on the studied imaginary farm replace an equivalent quantity of crops produced elsewhere. This ensures that the difference in globally marketed crops (from the study farm or elsewhere) between scenarios (UA vs. SA) is zero, ensuring fair comparability. This approach allows us to evaluate how the environmental impact (EI) of our system changes relative to production elsewhere when introducing slurry acidification.

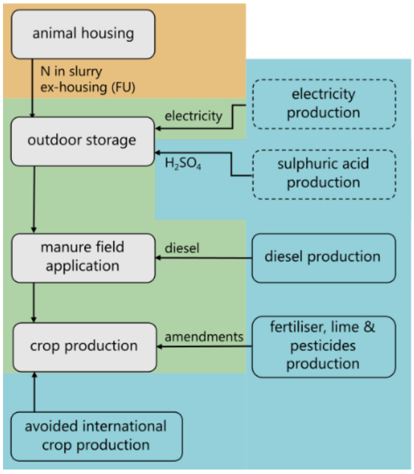


Figure 10. LL# 18: System boundaries

Structure of the analysed system. Boxes indicate main activities associated with the slurry acidification scenario (SA) and are not included in the scenario without acidification (UA). Grey boxes indicate activities happening on farm. Arrows indicate flows of products. FU: functional unit. Blue background: processes only considered in the LCA study, Yellow: processes only considered in the DBI assessment, Green: processes considered in both (Beyers et al. 2022, adjusted)

Following the overall trend of a growing global human population, crop production is increasing. We presume that the heightened crop production in our study serves to meet the growing demand on the global market, rather than causing an actual reduction in demand elsewhere. Following the principles of consequential LCA theory regarding the identification of marginal suppliers, supplying goods to a growing market may delay the implementation of new production technology of the fastest-growing market (Hauschild and Rosenbaum, 2018).

In our case, market segments equate to countries, and we have identified countries experiencing the strongest growth in crop production. We assumed that these countries would respond to the global increase in demand if we do not. Consistent with the ecoinvent database, we identified the long-term marginal. We consider like-for-like replacement, meaning for example potatoes replace potatoes, and crops are not substitutable by another crop in terms of nutritional values or other dietary factors.

3.1.3.2.2 Inventory

The inventory comprises literature, modelling, and ecoinvent data. The actual production of pigs, including feed, housing and construction/demolition of storage tanks or agricultural machinery, is excluded from the study due to assumed negligible differences between the studied scenarios. All analyses were conucted using openLCA software v.1.10.2, with background processes modelled using the consequential ecoinvent database 3.6 (Wernet et al., 2016).

Outdoor slurry storage

The studied system starts with a country-specific reference flow, representing an amount of pig slurry equivalent to the functional unit of 1,000 kg of slurry-N entering the outdoor storage. In the slurry acidification (SA) scenario, a portion of the slurry is pumped from the storage tank into a tank container mounted on a tractor trailer. There, it undergoes mixing with sulphuric acid (H_2SO_4) to achieve a pH of 5.5 before being returned to the outdoor storage.

Field application & crop production

Ammonia emissions during field application were calculated using the model ALFAM2 (Hafner et al., 2018). This involved considering application rates and methods, slurry composition as well as local environmental conditions in the three regions. The calculated NH_3 emissions, slurry compositions, representative crop rotations as well as weather data were utilised as inputs for the agro-ecosystem model Daisy (Abrahamsen and Hansen, 2000).

Daisy was used to estimate C and N related emissions (such as nitrous oxide emissions and nitrate leaching to freshwater ecosystems), as well as crop yields and residues returned to the soil. Baseline scenarios for the Daisy modelling were derived from Deliverable 1.5 of the Nutri2Cycle project (Duan et al. 2020). Chosen baseline scenarios were: Scenario 2 DK-pig for Denmark, Scenario 2 NL-arable for the Netherlands, and Scenario 1 ES-maize for Spain (all three including pig slurry application). To simulate the short-term turnover of slurries in the soil, the Daisy model was calibrated on the specific C and N mineralisation patterns of acidified slurry based on soil C and N mineralisation from laboratory incubation experiments (Fangueiro et al., 2010, 2009). Since, slurry acidification lowers nitrogen emissions, more N is retained in the slurry and is then field applied. In our simulations, we took account of increased N concentrations in the acidified slurry and as a consequence reduced mineral N fertilisation accordingly, assuming that there would be regulations in place in all the countries, if the technology is widely adopted.

To complement crop production processes with all the remaining emissions and materials needed, we selected existing ecoinvent processes for each respective crop and coupled them with the Daisy results. See Table 11, Table 12, and Table 13 for an overview of the inventory. Further information can be found in the supplementary material of Beyers et al. (2022).

Table 11. LL#18 LCA Inventory: Denmark

(adapted from Supplementary Material of Beyers et al. 2022)

		Weak P law				Strong P law		
		UA	UA_S	SA	SA_L	UA_P	SA_P	
	Unit	Amount	Amount	Amount	Amount	Amount	Amount	Exchange name in openLCA / ecoinvent
Outdoor storage								
Input								
slurry N ex-inhouse storage	t	1	1	1	1	1	1	
H2SO4 acidification agent	kg	-	-	1,095	1,095	-	1,095	sulfuric acid, market RER
acid transport to farm	t*km	-	-	5	5	-	5	transport, tractor and trailer, agriculture, CH
pumping/acidification	MJ	-	-	3,942	3,942	-	3,942	diesel, burned in agricultural machinary, GLO
Output								
CH4 emissions storage	kg	215	215	9	9	215	9	Ammonia, air, low population density
CO2 emissions storage	kg	402	402	33	33	402	33	Carbon dioxide, non-fossil, air, low population density
NH3 emissions storage	kg	24	24	6	6	24	6	Ammonia, air, low population density
Manure field application								
Input								
spring barley DK	kg	12,976	12,976	13,013	13,013	14,671	14,707	barley grain production, FR (changed)
winter barley DK	kg	14,826	14,826	14,876	14,876	16,739	16,805	barley grain production, FR (changed)
winter rape DK	kg	8,118	8,118	8,232	8,232	9,150	9,277	rape seed production, FR (changed)
winter wheat DK	kg	17,261	17,261	17,312	17,312	19,524	19,576	wheat production, FR (changed)
sulfuric acid as mineral fertiliser	kg	-	335.58	-	-	-	-	limestone and gypsum application, by speader, GLO
lime application	ha	-	-	-	6	-	-	transport, tractor and trailer, agriculture, CH
lime	kg	-	-	-	1,519	-	-	soil pH raising agent, as CaCO3, market for, GLO
Output								
CO2 emissions liming	kg	-	-	-	669	-	-	carbon dioxide, fossil, air, low population density
avoided spring barley FR	kg	12,976	12,976	13,013	13,013	14,671	14,707	barley grain production, FR

		Weak P law				Strong P law		
		UA	UA_S	SA	SA_L	UA_P	SA_P	
	Unit	Amount	Amount	Amount	Amount	Amount	Amount	Exchange name in openLCA / ecoinvent
avoided winter barley FR	kg	14,826	14,826	14,876	14,876	16,739	16,805	barley grain production, FR
avoided rape CA	kg	8,118	8,118	8,232	8,232	9,150	9,277	rape seed production, CA-CQ
avoided wheat RoW	kg	17,261	17,261	17,312	17,312	19,524	19,576	wheat production, RoW
Additional information								
slurry amount ex-inhouse storage	t	182	182	182	182	182	182	
Application area	ha	6	6	6	6	7	7	

Table 12. LL#18 LCA Inventory: The Netherlands

(adapted from Supplementary Material of Beyers et al. 2022)

		Weak P law				Strong P law		
		UA	UA_S	SA	SA_L	UA_P	SA_P	
	Unit	Amount	Amount	Amount	Amount	Amount	Amount	Exchange name in openLCA / ecoinvent
Outdoor storage								
Input								
slurry N amount ex-inhouse storage	t	1	1	1	1	1	1	
market for H2SO4	kg	-	-	786	786	-	786	sulfuric acid, market RER
acid transport to farm	t*km	-	-	4	4	-	4	transport, tractor and trailer, agriculture, CH
pumping/acidification	MJ	-	-	2,831	2,831	-	2,831	diesel, burned in agricultural machinary, GLO
Output								
NH3 emissions storage	kg	22	22	5	5	22	5	Ammonia, air, low population density
CH4 emissions storage	kg	217	217	9	9	217	9	Methane, non-fossil, air, low population density
CO2 emissions storage	kg	496	496	40	40	496	40	Carbon dioxide, non-fossil, air, low population density

		Weak P law				Strong P law		
		UA	UA_S	SA	SA_L	UA_P	SA_P	
	Unit	Amount	Amount	Amount	Amount	Amount	Amount	Exchange name in openLCA / ecoinvent
Manure field application								
Input								
Potato NL	kg	138,021	138,021	138,021	138,021	207,160	207,160	maize silage production, CH (changed)
Silage Maize NL	kg	57,665	57,665	57,665	57,665	86,333	86,333	potato production, CH (changed)
Sugar Beet NL	kg	166,766	166,766	166,766	166,766	249,673	249,673	sugar beet production, FR (changed)
Winter Wheat NL	kg	16,732	16,732	16,914	16,914	23,510	23,833	wheat production, FR (changed)
Sulfuric acid as mineral fertiliser	kg	-	561	-	-	-	-	sulfuric acid, market RER
Lime application	ha	-	-	-	9	-	-	limestone and gypsum application, by speader, GLO
Lime	kg	-	-	-	1,091	-	-	soil pH raising agent, as CaCO3, market for, GLO
Output								
CO2 emissions liming	kg	-	-	-	480	-	-	carbon dioxide, fossil, air, low population density
avoided Potato IN	kg	138,021	138,021	138,021	138,021	207,160	207,160	maize silage production, BR
avoided Silage Maize BR	kg	57,665	57,665	57,665	57,665	86,333	86,333	potato production, IN
avoided Sugar Beet RU	kg	166,766	166,766	166,766	166,766	249,673	249,673	sugar beet production, RU
avoided Winter Wheat RoW	kg	16,732	16,732	16,914	16,914	23,510	23,833	wheat production, RoW
Additional info:								
slurry amount ex-inhouse storage	t	131	131	131	131	131	131	
Application area	ha	9	9	9	9	14	14	

Table 13. LL#18 LCA Inventory: Spain

(adapted from Supplementary Material of Beyers et al. 2022)

		UA	UA_S	UA_urea	SA	SA_urea	
	Unit	Amount	Amount	Amount	Amount	Amount	Exchange name in openLCA / ecoinvent
Outdoor storage							
Input							
slurry N amount ex-inhouse storage	t	1	1	1	1	1	
market for H2SO4	kg	-	-	-	998	998	sulfuric acid, market RER
acid transport to farm	t*km	-	-	-	5	5	transport, tractor and trailer, agriculture, CH
pumping/acidification	MJ	-	-	-	3,594	3,594	diesel, burned in agricultural machinary, GLO
Output							
NH3 emissions storage	kg	34	34	34	8	8	Ammonia, air, low population density
CH4 emissions storage	kg	287	287	287	11	11	Methane, non-fossil, air, low population density
CO2 emissions storage	kg	536	536	536	43	43	Carbon dioxide, non-fossil, air, low population density
Manure field application							
Input							
silage maize ES	kg	159,576	159,576	159,576	172,081	172,081	maize silage production, CH (changed)
Sulfuric acid as mineral fertiliser	kg	-	579.05	-	-	-	sulfuric acid, market RER
Output							
avoided silage maize BR		159,576	159,576	159,576	172,081	172,081	maize silage production, BR
Additional info							
slurry amount ex-inhouse storage	t	166	166	166	166	166	
Application area	ha	5	5	5	6	6	

3.1.3.3 Results

3.1.3.3.1 Impact assessment

The impacts of each system were determined using Environmental Footprint methodology (EC 2021). In all three study regions, slurry acidification indicated both positive and negative environmental effects (Figure 11).

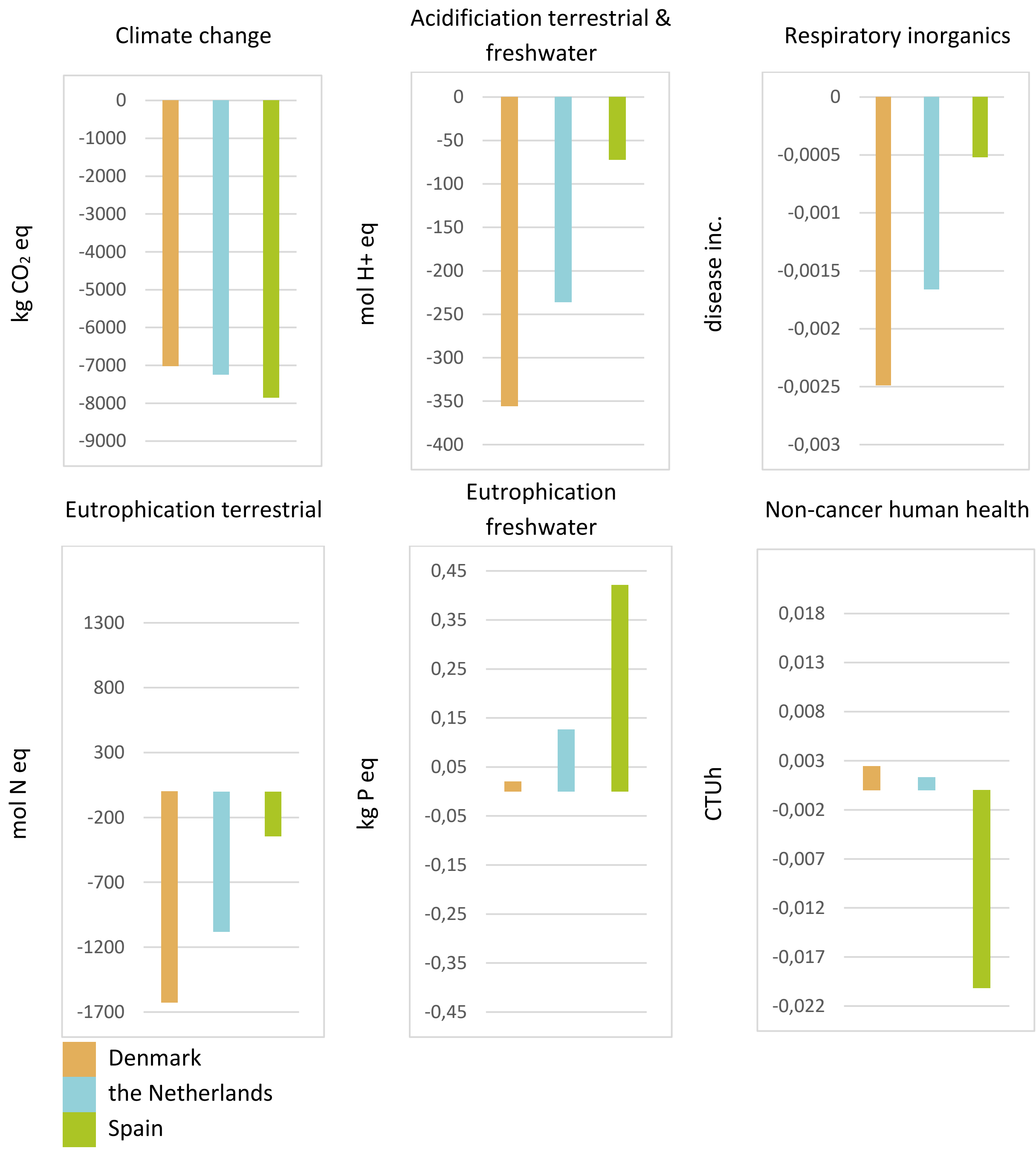


Figure 11. LL# 18: eLCA results

Results per functional unit of 1,000 pig slurry-N leaving the animal house. Change in potential environmental impact through shifting from: UA → SA (no acidification to acidification)

A decrease in impact potential in all countries was observed in the categories *climate change potential* (CCP), *terrestrial* & *freshwater acidification potential* (AP), *terrestrial eutrophication* (TEP), and *respiratory inorganics potential* (RIP). These reductions are achieved mainly because of lower CO_2, CH_4, and NH_3 emissions during storage and field application of acidified slurry.

Other categories suggested increases in environmental impacts as a result of an implementation of slurry acidification, namely *energy* & *mineral resource use potential* (ERP, MRP), *freshwater eutrophication potential* (FEP), and *human cancer potential* (HCTP).

In the UA scenarios of DK and NL, on-farm crop production was found to be less impactful in terms of fossil and mineral resource depletion per mass than international production of the crops they replace. Introducing slurry acidification increased the demand for fossil and mineral resources in the crop production of DK and NL and reduces this advantage. Spanish maize production was found to be more fossil and mineral resource intense per mass than international crop production and slurry acidification was found to further widen this gap.

3.1.3.4 Interpretation

Previous studies suggest that slurry acidification is a reliable treatment technology to reduce *terrestrial eutrophication potential* and also bears the potential to reduce CCP, when applied early in the management chain (ten Hoeve et al., 2016a, 2016b). A past LCA study comparing in-house and field acidification, found that in-house acidification resulted in a reduction in CCP whereas field acidification increased CCP of the slurry management chain (ten Hoeve et al., 2016a, 2016b). Our study suggests that storage acidification still suffices to achieve reductions in CCP. Given its lower investment costs, and easier implementation into existing systems compared to in-house acidification, storage acidification could be a viable technology to reduce GHG emissions in slurry management. Slurry acidification had a negligible impact on *marine eutrophication potential* (MEP), which is in accordance with ten Hoeve et al. (2016b), who also found the net change in MEP to be low, when comparing in-house or field acidification against no treatment. The same study suggested little difference in terms of *freshwater ecotoxicity* between treatment scenarios (ten Hoeve et al., 2016b), which is only in partial agreement with our findings as we did find increased impacts in Denmark and the Netherlands.

Another research study conducted under Danish conditions, focusing on in-house acidification, found that acidification led to decreases in both acidification and eutrophication. However, it also observed increases in emissions associated with global warming potential (GWP) and non-renewable energy and resource consumption potential (Pexas et al., 2020). Similar findings were derived in our study, with the exception of GWP. This discrepancy might be attributed to the fact that our study did not consider alterations in CO2 and CH4 emissions during storage and field application.

3.1.3.5 Conclusion

This study compared acidification of pig slurry against no treatment in three intensive pig production regions of Denmark, the Netherlands, and Spain in terms of their environmental performance.

The results suggest that slurry acidification reduces the environmental impact of slurry management in those categories mostly related to agriculture, such as *terrestrial* & *freshwater acidification* and *climate change potential*, and that it has the potential of contributing to an overall enhanced nutrient recycling.

However, our study also showed that slurry acidification can have negative impacts on those categories more related to the provision of energy (for example *energy resource use*) or manufacturing of inputs (for example *human health effects*). To justify slurry acidification on all levels, energy and material sources should be examined and carefully selected.

When comparing the effects acidification has on each country's performance, the Danish case showed greatest potential for improvements. However, also the Spanish case showed improvement potential, particularly in terms of climate change related emissions. The Netherlands on the other hand bear greatest risk of negative impacts, especially in categories not usually associated with agricultural activity.

This study can be used to support decision-making of different stakeholders involved the agricultural sector and interested in closing nutrient loops and improving the environmental performance of pig breeding and manure handling.

3.1.3.6 Dashboard indicators

LL#18 Slurry acidification with industrial acids to reduce NH3 volatilisation from animal husbandry (*: Qualitative DBI assessment: **+** *improvement (<-10%),* **o** *no change (-10% - +10%),* **-** *deterioration (>+10%))*

Table 14. LL# 18: Comparison of dashboard indicator and life cycle assessment results

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication*	Comment	Quantitative estimate per FU of 1,000 kg slurry N	LCA indication*	Re-evaluation of DBI irt. LCA results*
Use of Primary Resources	**Rock Phosphate** (Reduction in mineral phosphorus consumption)	**o**	No P is added or depleted unless phosphoric acid is used (not common though)	Phosphorous, in ground [a] DK: -1.78 kg P (-1%) NL: -1.36 kg P (-0.1%) ES: -11.09 kg P (-8%)	 **o** **o** **o**	 DK: confirmed NL: confirmed ES: confirmed
	Natural Gas (Reduction in natural gas consumption in mineral fertiliser production)	**o**		Gas, natural, in ground [b] DK: 13.38 m^3 (+1%) NL: 14.79 m^3 (+0.4%) ES: 24.44 m^3 (+9%)	 **o** **o** **o**	 DK: confirmed NL: confirmed ES: confirmed
	Oil (Reduction in oil consumption in agricultural machinery)	**o**	Negligible additional diesel for mixing and field application acidification	Oil, crude, in ground [b] DK: 105.25 kg (+12%) NL: 80.83 kg (+4%) ES: 138.74 kg (+29%) Diesel burnt in agricultural machinery [c]	 **-** **o** **-**	 DK: revised, more likely incr. NL: confirmed ES: revised, more likely incr.

[a] While slurry acidification does not change P application patterns on farm, it does slightly increase yields. The increase in yield results in the replacement of crops which production would have consumed more P.
[b] The increase in natural gas and oil results from the energy production needed to produce, transport, and mix the acid
[c] This is not necessarily limited to the farm implementing acidification

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication*	Comment	Quantitative estimate per FU of 1,000 kg slurry N	LCA indication*	Re-evaluation of DBI irt. LCA results*
				DK: 3941.48 MJ (>+100%) NL: 2830.50 MJ (+17%) ES: 3594.00 MJ (>+100%)	- - -	DK: revised, more likely incr. NL: revised, more likely incr. ES: revised, more likely incr.
	Electricity (Reduction in electricity consumption)	-	More pumping/mixing for in house/storage acidification	Electricity consumption [d] DK: 617.09 MJ (+4%) NL: 302.97 MJ (+2%) ES: 4349.78 MJ (+20%)	 o o -	 DK: revised, more likely negligible NL: revised, more likely negligible ES: confirmed
	Water (Reduction in water consumption)	o		Water scarcity [e] DK: 323.53 m^3 depriv. (+4%) NL: -880.8 m^3 depriv. (-0.1%) ES: 8982.67 m^3 depriv. (-9%)	 o o o	 DK: confirmed NL: confirmed ES: confirmed
	Soil quality (Improvement in soil quality)	-	Potential deterioration of soil quality from acidification still controversial	Land use DK: -86455 Pt (-2%) NL: 291596 Pt (-0.5%) ES: 291596 Pt (+8%)	 o o o	 DK: revised, more likely negligible NL: revised, more likely negligible ES: revised, more likely negligible
	Nutrient recovery (Increase in nutrient recycling)	+	Ammonia retained is converted to plant available N	DK N fertiliser: -1.34 kg (-0.4%) P fertiliser: -3.34 kg (-0.5%) K fertiliser: -2.10 kg (-1.4%) NL	 o o o	 N: revised, more likely negligible P: confirmed K: confirmed

[d] Electricity production is not limited to the process of acidification but includes for instance acid production and fertiliser production
[e] Changes in water use mainly stem from shifts in production

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication*	Comment	Quantitative estimate per FU of 1,000 kg slurry N	LCA indication*	Re-evaluation of DBI irt. LCA results*
				N fertiliser: 2.91 kg (+1%)	o	N: revised, more likely negligible
				P fertiliser: -2.64 kg (-0.1%)	o	P: confirmed
				K fertiliser: -0.003 kg (-0.0%)	o	K: confirmed
				ES		N: revised, more likely negligible
				N fertiliser: 24.93 kg (+8%)	o	P: confirmed
				P fertiliser: -21.43 kg (-8%)	o	K: confirmed
				K fertiliser: -21.68 kg (-8%)	o	
	Others? Please specify Lime for soil liming and sulphur for producing H_2SO_4	-	Additional soil liming required with acidification. Some resource consumption for the production of H_2SO_4	no further info		
Emissions to the environ-ment	**Ammonia (air emission)** (Reduction in NH_3 emissions)	+	Main objective of the technology; up to >90% reduction in NH_3 loss	Ammonia emission to air DK: -122.15 kg NH_3 (> -100%) NL: -81.29 kg NH_3 (-18%) ES: -27.83 kg NH_3[f] (>-100%)	 + + +	 DK: confirmed NL: confirmed ES: confirmed
	Dinitrogen monoxide (air emission) (Reduction in N_2O Emissions)	+	Also effects on N_2O - from literature	Dinitrogen monoxide emission to air DK: 1.12 kg N_2O (+14%) NL: 1.13 kg N_2O (+5%) ES: 2.95 kg N_2O (+8%)	 - o o	 DK: revised, more likely incr. NL: revised, more likely negligible ES: revised, more likely negligible

[f] Reduction mostly achieved at 'low population density' & on farm, where acidification was introduced.

Nutri2Cycle
Nurturing the Circular Economy

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication*	Comment	Quantitative estimate per FU of 1,000 kg slurry N	LCA indication*	Re-evaluation of DBI irt. LCA results*
	Methane (air emission) (Reduction in CH_4 emissions)	+	Also effects on CH_4 - from literature	Methane emission to air DK: -206.40 kg CH_4 (>-100%) NL: -208.38 kg CH_4 (>-100%) ES: -275.10 kg CH_4 (-96%)	 + + +	 DK: confirmed NL: confirmed ES: confirmed
	Nitrates (water emission) (Reduction in NO_3 emissions)	o	Effect depends on N scheme - is mineral fertiliser substituted or not	Nitrate emission to water DK: -17.96 kg NO_3 (-1%) NL: 24.72 kg NO_3 (+1%) ES: 17.29 kg NO_3 (+9%)	 o o o	 DK: confirmed NL: confirmed ES: confirmed
	Phosphorus (water emission) (Reduction of P Emissions)	o		Phosphorous emission to water DK: -0.14 kg P (-1%) NL: -0.06 kg P (-1%) ES: 0.09 kg P (+8%) Phosphate emission to water DK: 0.47 kg PO_4^{3-} (+73%) NL: 0.56 kg PO_4^{3-} (+3%) ES: 0.97 kg PO_4^{3-} (+15%)	 o o o - o -	Phosphorous: DK: confirmed NL: confirmed ES: confirmed Phosphate: DK: revised, more likely incr. NL: confirmed ES: revised, more likely incr.
	Particulate matter (Reduction of particulate matter formation)	o		Particulates emission to air DK: 1.41 kg fPM (+11%) NL: 1.35 kg fPM (+7%) ES: 1.78 kg fPM (+33%)	 - o -	 DK: revised, more likely incr. NL: confirmed ES: revised, more likely incr.
	Others? Please specify					

Nutri2Cycle
Nurturing the Circular Economy

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication*	Comment	Quantitative estimate per FU of 1,000 kg slurry N	LCA indication*	Re-evaluation of DBI irt. LCA results*
Resilience to climate change	**Carbon footprint** (Reduction of carbon footprint)	+	Overall reduction in N_2O, CH_4 etc. but also increased energy demand	Climate change [CO_2 eq.]		
				DK: -7017.91 kg (-20%)	+	DK: confirmed
				NL: -7250.96 kg (-69%)	+	NL: confirmed
				ES: -7856.97 kg (-19%)	+	ES: confirmed
	Effective SOM (Effective Soil Organic Matter Improvement)	o		Carbon | Emission from soil	-	DK: revised, more likely incr.
				DK: 0.18 kg C (+15%	o	NL: confirmed
				NL: 0.14 kg C (+5%)	-	ES: revised, more likely incr.
				ES: 0.13 kg C (+21%)		
				Carbon dioxide | Emission from soil		
				DK: 0.002 kg CO_2 (+2%)	o	DK: confirmed
				NL: 0.001 kg CO_2 (+0.4%)	o	NL: confirmed
				ES: 0.003 kg CO_2 (+31%)	-	ES: revised, more likely incr.

3.1.3.6.1 Comparison of LCA and DBI results

In terms of the consumption of natural gas and rock phosphate extraction, DBI results were in accordance with LCA results, and no change in consumption is to be expected.

On the system level, no degradation in soil quality was determined in the LCA as opposed to the DBI. Rather improvements in terms of *terrestrial eutrophication* and *acidification* were found in the LCA study. In the LCA study, the assessment is not limited to the soil of the study farm but encompasses industrial sites and agricultural soils of farms responding to changes in production of the study farm. In the qualitative expert assessment this is not well defined, but it is likely that the experts have focused more on impacts at the farms rather than outside.

Regarding indicator type Emissions to the environment, we found DBI and LCA results to agree mostly. Contradicting results mainly stem from the perspective of the experts assessing the DBI which may have focused more on the farm, thus accounting for direct effects of slurry acidification, versus the all-encompassing perspective of the LCA. In terms of impacts related to climate change, both methods came to the same conclusions.

3.1.3.7 References

Abrahamsen, P., Hansen, S., 2000. Daisy: An open soil-crop-atmosphere system model. Environ. Model. Softw. https://doi.org/10.1016/S1364-8152(00)00003-7

Beyers, M., Duan, Y.-F., Jensen, L.S, Bruun, S. 2022. Effect of natural and regulatory conditions on the environmental impacts of pig slurry acidification across different regions in Europe: A life cycle assessment. J.Cle.Prod 368, 133072 https://doi.org/10.1016/j.jclepro.2022.133072

Duan, Y.-F., Bruun, S., Jensen, L.Sl., van Gerven, L., Hendriks, C., Stokkermans, L., Groenendijk, P., Lesschen, J.P., 2020. D1.5 Mapping and characterization of CNP flows and their stoichiometry in main farming systems in Europe. Nutri2Cycle EU No. 773682. Available from www.nutri2cycle.eu

EC – European Commission, 2021. Commission Recommendation on the use of the Environmental Footprint methods to measure and communicate the life cycle environmental performance of products and organisations. https://ec.europa.eu/environment/publications/recommendation-use-environmental-footprint-methods_en

Fangueiro, D., Hjorth, M., Gioelli, F., 2015. Acidification of animal slurry– a review. J. Environ. Manage. 149, 46–56. https://doi.org/10.1016/J.JENVMAN.2014.10.001

Fangueiro, D., Ribeiro, H., Coutinho, J., Cardenas, L., Trindade, H., Cunha-Queda, C., Vasconcelos, E., Cabral, F., 2010. Nitrogen mineralization and CO_2 and N_2O emissions in a sandy soil amended with original or acidified pig slurries or with the relative fractions. Biol. Fertil. Soils 46, 383–391. https://doi.org/10.1007/s00374-010-0444-1 656

Fangueiro, D., Ribeiro, H., Vasconcelos, E., Coutinho, J., Cabral, F., 2009. Treatment by acidification followed by solid-liquid separation affects slurry and slurry fractions composition and their potential of N mineralization. Bioresour. Technol. 100, 4914–4917. https://doi.org/10.1016/j.biortech.2009.04.032

Hafner, S.D., Pacholski, A., Bittman, S., Burchill, W., Bussink, W., Chantigny, M., Carozzi, M., Génermont, S., Häni, C., Hansen, M.N., Huijsmans, J., Hunt, D., Kupper, T., Lanigan, G., Loubet, B., Misselbrook, T., Meisinger, J.J., Neftel, A., Nyord, T., Pedersen, S. V., Sintermann, J., Thompson, R.B., Vermeulen, B., Vestergaard, A. V., Voylokov, P., Williams, J.R., Sommer, S.G., 2018. The ALFAM2 database on ammonia emission from field-applied manure: Description and illustrative analysis. Agric. For. Meteorol. 258, 66–79. https://doi.org/10.1016/j.agrformet.2017.11.027

Hauschild, M.Z., Rosenbaum, 2018. Life Cycle assessement theory and practice. Springer

Hou, Y., Velthof, G.L., Oenema, O., 2015. Mitigation of ammonia, nitrous oxide and methane emissions from manure management chains: A meta-analysis and integrated assessment. Glob. Chang. Biol. 21, 1293–1312. https://doi.org/10.1111/gcb.12767

Petersen, S.O., Andersen, A.J., Eriksen, J., 2012. Effects of Cattle Slurry Acidification on Ammonia and Methane Evolution during Storage. J. Environ. Qual. 41, 88–94. https://doi.org/10.2134/jeq2011.0184 738

Petersen, S.O., Højberg, O., Poulsen, M., Schwab, C., Eriksen, J., 2014. Methanogenic community changes, and emissions of methane and other gases, during storage of acidified and untreated pig slurry. J. Appl. Microbiol. 117, 160–172. https://doi.org/10.1111/jam.12498

Pexas, G., Mackenzie, S.G., Wallace, M., Kyriazakis, I., 2020. Environmental impacts of housing conditions and manure management in European pig production systems through a life cycle perspective: a case study in Denmark. J. Clean. Prod. 253. https://doi.org/10.1016/j.jclepro.2020.120005

ten Hoeve, M., Gómez-Muñoz, B., Jensen, L.S., Bruun, S., 2016a. Environmental impacts of combining pig slurry acidification and separation under different regulatory regimes – A life cycle assessment. J. Environ. Manage. 181, 710–720. https://doi.org/10.1016/j.jenvman.2016.08.028

ten Hoeve, M., Nyord, T., Peters, G.M., Hutchings, N.J., Jensen, L.S., Bruun, S., 2016b. A life cycle perspective of slurry acidification strategies under different nitrogen regulations. J. Clean. Prod. 127, 591–599. https://doi.org/10.1016/J.JCLEPRO.2016.04.014

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., Weidema, B., 2016. The ecoinvent database version 3 (part I): overview and methodology. Int. J. Life Cycle Assess. 21, 1218–1230. https://doi.org/10.1007/s11367-016-1087-8

3.1.4 LL#40: Insect breeding on agro-residues to produce alternative livestock feed (UCPH+UGENT+inagro)

Longlist #40 title: Insect breeding as an alternative protein source on solid agro-residues (manure and plant wastes)

Miriam Beyers[1,2], Carl Coudron[3], Rahul Ravi[2], Erik Meers[2], Sander Bruun[1]

[1] Department of Plant and Environmental Sciences, Faculty of Science, University of Copenhagen, Denmark

[2] Department of Green Chemistry and Technology, Faculty of Bioscience Engineering, Ghent University, Belgium

[3] Inagro, Ieperseweg 87, 8800 Roeselare, Belgium

This chapter builds on the published scientific paper by Beyers et al. (2023) Black soldier fly larvae as an alternative feed source and agro-waste disposal route - A life cycle perspective. Resour. Conserv. Recycl. 192, https://doi.org/10.1016/j.resconrec.2023.106917

3.1.4.1 Introduction

Black soldier fly larvae have been recognised as a valuable protein source (e.g. Abd El-Hack et al. 2020; van Huis et al. 2020). Their ability to feed on a variety of organic substances, including waste materials, presents a management option for surplus manure and biowaste (Amrul *et al.*, 2022; Liu *et al.*, 2022; Lopes *et al.*, 2022).

Despite the potential benefits, a limited number of life cycle assessment (LCA) studies on insect production have revealed that feed provision and heat-related energy can offset the advantages. Even when the larvae are fed on side-streams from the food processing industry, the environmental impacts of rearing may surpass those of fish- or soymeal (Smetana *et al.*, 2019). However, insect production using waste products remains one of the few options within the EU to increase nutrient recycling and produce alternative feedstuff that require less land than current practices.

Flanders in Belgium faces challenges with a surplus of agro-residues, including endive roots, Brussels sprouts stems, and the solid fraction of pig manure. Vegetable waste left on fields contributes to ammonia emissions, and possibly nitrous oxide and carbon emissions during decomposition (Ruijter, F.J. de; Huijsmans, J.F.M.; Zanten, M.C.; Asman, W.A.H.; Pul, 2013). In addition, in these regions surplus manure has to be exported to neighbouring countries (D'Haene and Vannecke, 2020). Insect production on waste products may help reduce manure transportation. Former studies have not fully incorporated these benefits of insect production from agro-residues in their assessments.

The objective of this study was to assess potential environmental implications of black soldier fly production on agro-residues, identify environmental hotspots and provide recommendations for optimising the environmental aspects of larvae production.

This sub-chapter is a pre-publication summary of the full study presented in Beyers et al. (2023).

3.1.4.2 Material & methods

3.1.4.2.1 Goal & scope

The goal of this LCA study was to compare the environmental impacts of protein feed production through black soldier fly against conventional soy- and fishmeal protein feed. The functional unit is the *1,000 kg protein feed*. Figure 12 gives a schematic description of our assessment. The study includes the provision of feed to the insects, continues with the rearing process and ends with the harvesting of insects and disposal of insect frass. Frass refers to insect excrements, residual feed, and dead insect bodies.

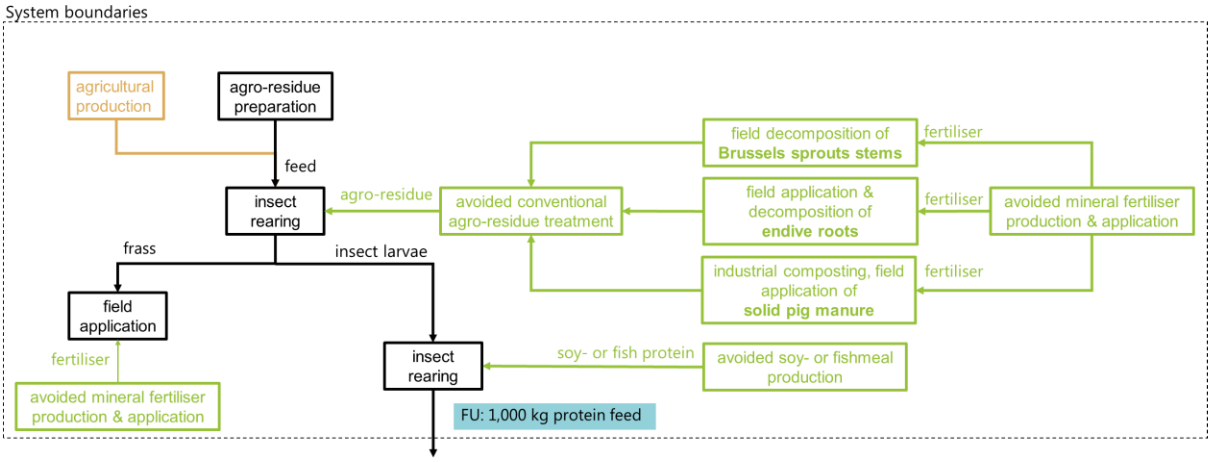


Figure 12. LL# 40: System boundaries.

Structure of the insect production system under study. Boxes indicate activities in the foreground system. Arrows indicate exchanges. FU = functional unit (blue background). Orange box stroke: only valid in non-manure diets. Green box stroke: credits for avoidance. Note: agro-residues indicated here are not all represented in all diets at the same time.

3.1.4.2.2 Inventory

Experimental data provided by Inagro, Belgium, formed the foundation of the inventory of this study. A detailed description of their experiments can be found in the Tier 2 Report of the Nutri2Cycle project. The experimental data was supplemented by literature data, and (modified) ecoinvent processes (ecoinvent v3.7.1 consequential database ((Moreno Ruiz E., Valsasina L., FitzGerald D., Symeonidis A. *et al.*, 2020)). For the impact assessment, the Environmental Footprint 3.0 methodology was selected (Fazio, S. Biganzioli, F. De Laurentiis and Zampori, L., Sala, S. Diaconu, n.d.). Recommendations of the Product Environmental Footprint Method were followed to the extent possible and included normalisation and weighting (European Commission, 2021).

We selected five diets (table included in Figure 13) and used them as basis for this LCA. The diets contain mixtures of agro-residues and non-agro-residue feedstuffs. Table 15 contains an overview of the inventory (for background information please be referred to the Supplementary Materials of Beyers et al. (2023).

Agro-residues

All additional steps required before feeding agro-residues to the insects, as well as the avoided conventional treatment of these residues, were included. In the conventional treatment scenario, Brussels sprouts stems are left in the field to decompose, while in the insect rearing scenario, they are collected and transported for feeding to insects. Endive roots, unlike Brussels sprouts, are reapplied to fields in the conventional treatment scenario since they are cultivated indoors. The solid fraction of pig manure is typically composted and exported from Belgium to P-deficient regions in France (conventional treatment scenario). In France, the compost serves as a fertiliser and soil enhancer.

In the insect rearing scenario, endive roots and manure only need to be transported to the insect farm. In all three cases, the avoided field application includes emissions from decomposition as well as the replacement of nitrogen, phosphorous, and potassium mineral fertilisers.

Non-agro-residues

For non-agro-residues, ecoinvent processes were used as a starting point and modified to suit the characteristics found in the feeding experiments. For instance, in the case of potato starch, the dry matter content in the ecoinvent process did not match that of the starch used in the feeding experiments. Therefore, the ecoinvent process for potato starch was scaled accordingly.

Insect rearing

The insect rearing process included the hatching of BSF eggs that are provided with a mixture of chicken feed and tap water, and the subsequent rearing of ready to harvest larvae. Here, we accounted for the energy, electricity and water needed during the rearing, and included emissions from feed and frass decomposition as well as insect metabolism.

Insect frass

The frass is assumed to be dried, transported and field applied as organic fertiliser in France. We followed the same logic as in the inventory for the avoidance of conventional treatment and the impacts from frass field application in terms of emissions resulting from application and decomposition as well the replacement of mineral fertilisers.

Energy & materials consumption, remaining emissions

To account for energy and material consumption as well as emission losses and mass flows between different compartments and insect rearing stages, we relied on literature sources and conducted mass flow analysis (e.g. following the example of Parodi et al., (2020, 2021) & Smetana et al., (2019)). We assumed the insect larvae to replace protein derived from soy- or fishmeal and used ecoinvent processes to model credits for avoidance.

Nutri2Cycle
Nurturing the Circular Economy

Table 15. LL#40 LCA Inventory: Insect rearing

		Diet 1	Diet 2	Diet 3	Diet 4	Diet 5	Comment	Reference
Insect production per Diet								
Input								
Diet	kg	10	10	10	10	10	See below (Provision Diet)	inagro
Endive root 'disposal'	kg	-4.94	-7.65		-1.44		See below (Endive root 'disposal')	inagro
Brussels sprout stem 'disposal'				-8.01	-6.53		See below (Brussels sprout stem 'disposal')	
Manure						-10	See below (Manure 'disposal')	inagro
Frass treated	kg	2.31	2.39	1.77	1.71	6.69	See below (Frass processing)	inagro
Larvae 7D	kg	0.08	0.08	0.08	0.08	0.08	See below (Larvae production)	inagro
Materials & Energy for 1 kg fresh larvae	kg	1.74	1.74	1.64	1.73	1.10	See below (Materials and energy for insect production)	
Output								
Carbon dioxide	kg	1.17	1.17	1.00	1.17	0.63		Parodi et al. (2020)
Dinitrogen monoxide	kg	4.20E-05	4.20E-05	3.44E-05	4.20E-05	1.85E-06		Parodi et al. (2020)
Larvae	kg	0.50	0.50	0.42	0.50	0.33		
Methane	kg	1.19E-05	1.19E-05	1.02E-05	1.19E-05	3.22E-03		Parodi et al. (2020)
Nitrogen, atmospheric	kg	2.38E-04	2.38E-04	1.95E-04	2.38E-04			Parodi et al. (2020)
Ammonia						2.22E-02		Parodi et al. (2021)
Diet Provision								
Input								
electricity, medium voltage	kWh	0.25	0.25	0.25	0.25	0.25		Smetana et al. (2019)
Endive root	kg	4.94	7.65		1.44		See below (Endive root provision)	inagro
Brussels sprout stem	kg			8.01	6.53		See below (Brussels sprout stem provision)	inagro
Potato starch	kg	1.60		1.99	1.99		See below (Potato starch provision)	inagro
Soy paste	kg	1.40					See below (Soy paste provision)	inagro
tap water	kg	6.70	6.70	6.70	6.70	6.70		Smetana et al. (2019)
Wheat bran	kg	2.06			0.04		See below (Wheat bran provision)	inagro
Chicken feed	kg		2.35				See below (Chicken feed provision)	inagro
transport, tractor & trailer, agri.	t*km					2.5		inagro
Manure	kg					10		inagro

		Diet 1	Diet 2	Diet 3	Diet 4	Diet 5	Comment	Reference
Output								
Diet	kg	10	10	10	10	10		
Endive root provision								
Input								
transport, tractor & trailer, agri.	t*km	2.5	2.5		2.5			Dobbelaere et al. (2015)
Output								
Endive root	kg	1000	1000		1000			
Potato starch provision								
Input								
potato starch	kg	1		1	1			adapted ecoinvent
transport, freight, lorry 7.5-16 metric ton, EURO6	t*km	0.05		0.05	0.05			own assumption
Output								
Potato starch	kg	2.70		2.70	2.70			
Soy paste provision								
Input								
soybean meal	kg	0.41						adapted ecoinvent
Output								
Soy paste	kg	1						
Wheat bran provision								
Input								
wheat bran	kg	1			1			ecoinvent
Output								
Wheat bran	kg	1			1			
Chicken feed production								
Input								

		Diet 1	Diet 2	Diet 3	Diet 4	Diet 5	Comment	Reference
energy feed, gross	MJ		18.70					adapted ecoinvent
protein feed, 100% crude	kg		0.15					adapted ecoinvent
Output								
Chicken feed	kg		1					
Brussels sprouts stems provision								
Input								
transport, tractor & trailer, agri.	t*km			2.50	2.50		transport to insect farm	Dobbelaere et al. (2015)
transport, tractor & trailer, agri.	t*km			0.20	0.20		harvest	Dobbelaere et al. (2015)
Output								
Brussels sprout stem	kg			1000	1000			
Endive root 'disposal' - avoided								
Input								
Ammonia	kg	-0.06	-0.06		-0.06			de Ruijter et al. (2013)
Dinitrogen monoxide	kg	-0.02	-0.02		-0.02			Velthof et al. (2002)
fertilising, by broadcaster	ha	0.04	0.04		0.04			own calculations
inorganic nitrogen fertiliser, as N	kg	1.10	1.10		1.10			own calculations
inorganic phosphorus fert., P2O5	kg	1.40	1.40		1.40			own calculations
inorganic potassium fert., as K2O	kg	3.49	3.49		3.49			own calculations
Nitrate	kg	-0.67	-0.67		-0.67			Smaling (1993)
transport, tractor & trailer, agri.	t*km	0.21	0.21		0.21			Dobbelaere *et al.*, 2015
Output								
Ammonia	kg	0.04	0.04		0.04			EEA (2019)
Dinitrogen monoxide	kg	0.02	0.02		0.02			Nemecek & Schnetzer (2011)
Endive root	kg	1000	1000		1000			
Nitrate	kg	0.63	0.63		0.63			Brockmann et al. (2018).
Brussels sprouts stem 'disposal' - avoided								
Input								

		Diet 1	Diet 2	Diet 3	Diet 4	Diet 5	Comment	Reference
Ammonia	kg			-0.23	-0.23			de Ruijter et al. (2013)
Dinitrogen monoxide	kg			-0.07	-0.07			Velthof et al. (2002)
fertilising, by broadcaster	ha			0.04	0.04			own calculations
inorganic nitrogen fertiliser, as N	kg			4.27	4.27			own calculations
inorganic phosphorus fert., P2O5	kg			1.51	1.51			own calculations
inorganic potassium fert., as K2O	kg			3.92	3.92			own calculations
Nitrate	kg			-2.60	-2.60			Smaling (1993)
Output								
Flow	Unit			Amount	Amount			
Ammonia	kg			0.37	0.37			EEA (2019)
Brussels sprout stem	kg			1000	1000			
Dinitrogen monoxide	kg			0.35	0.35			Velthof et al. (2002)
Nitrate	kg			2.70	2.70			Smaling (1993)
Manure 'disposal' - avoided								
Input								
Ammonia	kg					-0.41		de Ruijter et al. (2013)
Dinitrogen monoxide	kg					-0.12		Velthof et al. (2002)
electricity, medium voltage	kWh					47.50		Lemmens et al. (2007)
fertilising, by broadcaster	ha					0.05		own calculations
inorganic nitrogen fertiliser, as N	kg					7.55		own calculations
inorganic phosphorus fert., P2O5	kg					6.26		own calculations
inorganic potassium fert., as K2O	kg					4.68		own calculations
Nitrate	kg					-4.60		Smaling (1993)
solid manure loading & spreading, by hydraulic loader and spreader	kg					584.69		
transport, tractor & trailer, agri.	t*km					175.41		de Clercq et al. (2015)
transport, tractor & trailer, agri.	t*km					2.50		de Clercq et al. (2015)
Output								
Ammonia	kg					1.21		Zhang et al. (2021)
Ammonia	kg					2.11		Sommer et al. (2019)

		Diet 1	Diet 2	Diet 3	Diet 4	Diet 5	Comment	Reference
Carbon dioxide	kg					113.33		Zhang et al. (2021)
Dinitrogen monoxide	kg					0.13		Nemecek & Schnetzer (2011)
Dinitrogen monoxide	kg					0.24		Zhang et al. (2021)
Manure	kg					1000		
Methane	kg					1.07		Zhang et al. (2021)
Nitrate	kg					5.04		Brockmann et al. (2018)
Frass processing								
Input								
Ammonia	kg	-8.94E-04	-8.75E-04	-9.68E-04	-1.03E-03	-4.18E-04		de Ruijter et al. (2013)
Dinitrogen monoxide	kg	-6.02E-04	-6.10E-04	-5.00E-04	-5.18E-04	-8.16E-04		Velthof et al. (2002)
fertilising, by broadcaster	ha	2.70E-04	2.73E-04	2.24E-04	2.32E-04	3.66E-04		own calculations
inorganic nitrogen fertiliser, as N	kg	0.04	0.04	0.03	0.03	0.05		own calculations
inorganic phosphorus fert., P2O5	kg	0.02	0.02	0.02	0.02	0.02		own calculations
inorganic potassium fert., as K2O	kg	0.04	0.05	0.03	0.03	0.04		own calculations
Nitrate	kg	-0.02	-0.02	-0.02	-0.02	-0.03		Smaling (1993)
transport, tractor & trailer, agri.	t*km	0.46	0.48	0.35	0.34	1.34		own assumptions
Output								
Ammonia	kg	0.01	0.01	0.01	0.01	0.01		Sommer et al., (2019) - adapted
Dinitrogen monoxide	kg	0.01	0.01	0.01	0.01	0.01		Rummel et al., (2021)
Frass treated	kg	2.31	2.39	1.77	1.71	6.69		
Nitrate	kg	0.03	0.03	0.02	0.02	0.04		Brockmann et al. (2018)
Larvea production								
Input								
Chicken feed	kg	1.38	1.38	1.38	1.38	1.38	See (Chicken feed production)	
electricity, medium voltage	kWh	11.43	11.43	11.43	11.43	11.43		Smetana et al. (2019)
natural gas, high pressure	m3	37.27	37.27	37.27	37.27	37.27		Smetana et al. (2019)
tap water	kg	2.79	2.79	2.79	2.79	2.79		Smetana et al. (2019)
tap water	kg	50	50	50	50	50		Smetana et al. (2019)

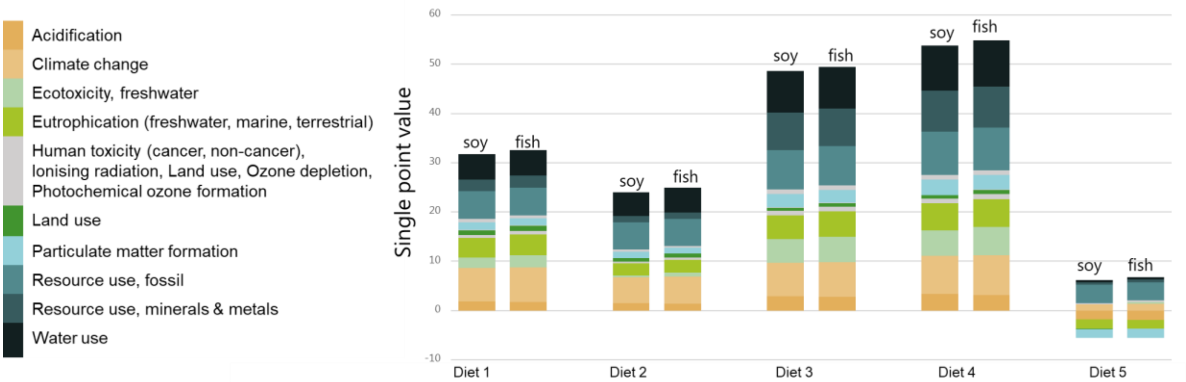
		Diet 1	Diet 2	Diet 3	Diet 4	Diet 5	Comment	Reference
Output								
Carbon dioxide	kg	0.49	0.49	0.49	0.49	0.49		Parodi *et al.* (2020)
Dinitrogen monoxide	kg	7.20E-06	7.20E-06	7.20E-06	7.20E-06	7.20E-06		Parodi *et al.* (2020)
Larvae 7D	kg	1.00	1.00	1.00	1.00	1.00		
Methane	kg	4.96E-06	4.96E-06	4.96E-06	4.96E-06	4.96E-06		Parodi *et al.* (2020)
Nitrogen, atmospheric	kg	1.63E-04	1.63E-04	1.63E-04	1.63E-04	1.63E-04		Parodi et al. (2020)
Materials and energy for insect production								
Input								
electricity, medium voltage	kWh	0.83	0.83	0.83	0.83	0.83		Smetana et al. (2019)
natural gas, high pressure	m3	0.10	0.10	0.10	0.10	0.10		Smetana et al. (2019)
tap water	kg	26.90	26.90	26.90	26.90	26.90		Smetana et al. (2019)
Output								
Materials & Energy for 1 kg fresh larvae	kg	1	1	1	1	1		

3.1.4.3 Results

3.1.4.3.1 Impact assessment

The overall normalised and weighted results from the production of of 1,000 kg of insect protein using the different diets are shown in Figure 13. The manure diet (diet 5) indicated lowest environmental impacts, while diet 3, including Brussels sprouts stems as agro-residue, indicated highest impacts. Of the non-manure-based diets, diet 2, including endive roots performed best.

All diets follow a similar pattern with respect to the impact categories of greatest and least concern (Figure 13). *Fossil resource use potential* seems to be of great importance in all diets. Across all diets, the life cycle stage contributing most to *fossil resource use* is 'egg to larvae rearing', with natural gas causing the highest impact contributions.



	Diet 1	Diet 2	Diet 3	Diet 4	Diet 5
Brussels sprouts stems*			80.1	65.3	
Chicken feed		23.5			
Endive root*	49.4	76.5		14.4	
Potato starch	16		19.9	19.9	
Soy paste	14				
Wheat bran	20.6			0.4	
Pig manure (solid fraction)*					100
Total [%]	100	100	100	100	100

Figure 13. LL# 40: eLCA results: normalised and weighted

Environmental impacts per functional unit of 1,000 kg insect protein. Coloured bars shown contribution of different impact categories to the normalized and weighted result. Soy: environmental impacts of replacing soy with insects; fish: environmental impacts of replacing fish with insects. Table below: Dry matter share of dietary components per diet [%]

Apart from the use of fossil resources, the diets can be divided into two groups based on the impact categories they have the most effect on: non-manure-based (diet 1 to 4) and manure-based (diet 5) diets.

In the non-manure-based diets, *climate change potential* assumes high relevance. The life cycle stages contributing most to *climate change* are 'diet provision' followed by 'insect rearing' (Figure 14). *Climate change* and *fossil resource use potential* are linked to the consumption of natural gas, which is used for heating the insect production facilities. Natural gas consumption was identified as highly impacting process. Another highly impacted category in diets 1 to 4 is *water use potential*. *Water use* like *climate change* is dominated by 'diet provision' but in *water use* its contribution is more striking. In general, 'diet provision' suggested to be a life cycle stage of great importance. When further examined, it was found that whenever potato starch was part of the non-manure dietary mix its environmental impacts held the greatest overall impact share and dominated the categories *particulate matter formation*, *mineral resource use* and *acidification potential*.

While insect production from non-manure-based diets only results in net environmental burdens in each category, the manure-based diet is beneficial to the environmental in some categories. In the manure-based diet (diet 5), the two most important impact categories, following *fossil resource use* are *acidification* and *particulate matter formation potential*. Both impact categories show net negative impacts. The beneficial impacts in diet 5 arise from the avoidance of the conventional treatment of pig manure including transportation, industrial composting and field application and to a smaller extent the avoidance of mineral fertiliser through field application of insect frass.

The avoidance of the production of soy- and fishmeal protein feed does not suffice to make up for the environmental burdens resulting from insect production. Using soy- and fishmeal as protein feed seems to cause a lower environmental burden than using insect protein. Only when the insects are feed on pig manure, some environmental benefits can be identified, namely in the categories *acidification*, *particulate matter formation*, and *terrestrial eutrophication potential*.

Figure 14. LL# 40: eLCA results – 2

Environmental impact results per functional unit of 1,000 kg insect-derived protein feed

The results suggest that the dry matter content of the agro-residue is important because transportation is a great contributor to several impact categories (for example: *climate change* and *fossil resource use potential*). Transporting the water contained in the residue causes emissions but is of no or little benefit to the insects. When selecting suitable waste materials for insect rearing, the dry matter content should be part of the decision-making process.

Scenario analysis

In the scenario analysis, possibilities to decrease the environmental impact of insect production were explored. Due to their great contributions to overall impacts, the response to alternatives from the default Belgian natural gas mix as well as to a different source of potato starch were tested by means of scenario analysis.

<u>Energy supply</u>

When comparing different gas sources none of the alternatives appears neither clearly environmentally superior nor inferior to the default Belgian natural gas mix. None of the alternative energy sources including wood pellets and residual heat lowers *fossil resource use* of BSF production below that of soy- or fishmeal.

<u>Potato starch supply</u>

The shift in potato starch sourcing suggested a decrease in environmental impacts of 'diet provision'. The greatest reductions could be achieved in *freshwater ecotoxicity*, *water use*, *terrestrial eutrophication*, *acidification,* and *particulate matter formation potential*. Despite great reductions in environmental impacts, switching from the industrial starch market to sourcing starch derived from by-products is insufficient for insect larvae to compete with fish- or soymeal. In only one category (*freshwater eutrophication potential*) does diet 4 perform better than soymeal. Fishmeal exhibits lower impacts in all categories. Switching to potato starch as a by-product can only be part of the solution.

3.1.4.4 Interpretation

This study aimed at identifying an alternative and sustainable protein source in the form of BSF larvae and ways to upcycle three common agro-residues in Flanders, Belgium. The suitability of insects as protein source and waste management method will be discussed.

Black soldier fly versus soy- or fishmeal protein

A review study on 24 LCAs concluded that the environmental hotspots of insect production are feed and energy provision (Smetana *et al.*, 2021). In the present study, we found that even when replacing a part of the commercial feed with agro-residues, similar hotspots can be identified. Among the non-manure-based diets, diet provision and energy supply for insect production appear to have the greatest environmental impact. As found in the review, this translated into adverse effects on *climate change*, *fossil resource use* and *water use potential*. Only the great impacts on *land use potential* are contrary to presented findings as for example *acidification*, *mineral resource use* and *freshwater ecotoxicity* were found to be affected more intensely. In the reviewed studies, these last categories were however less represented.

It is worth noting that switching to 100% agro-residues, as in the case of the manure-based diet, changes the kind of environmental hotspots as well as where in the insect production process they occur. The aforementioned evaluation is limited to diets consisting entirely or in parts of commercial feed products. While energy consumption remains a concern, impacts arising from land application and decomposition of manure or frass gain in relative importance because the diet is provided at rather low costs and impacts from the alternative treatment of the waste products are avoided.

One work-around concerning the high impacts from energy provision was to opt for a renewable source of heating instead of natural gas. In another LCA study on Black soldier fly larvae production, energy supply from photovoltaic systems was found to decrease impacts (Smetana *et al.*, 2019). Biogas and wood pellets in the present study did not lead to clear improvements but could even result in larger impacts. Finding and most of all establishing sustainable energy sources is a cross-sector and cross-border challenge and not specific to insect production. With new energy and electricity mixes evolving in the coming years, it will be worth reassessing the environmental implications of insect production in regular intervals.

Contrary to the study conducted by (Smetana *et al.*, 2019), the present study concluded that soy- and fishmeal production are environmentally superior to all non-manure-based diets in all studied impact categories. Only in the case of manure as feed for insects, a switch to insect production could be beneficial in some categories.

A prior study on insect diets using manure found contrasting results for chicken and cow manure. While a chicken manure-based diet had impacts similar to other organic waste streams, cow manure showed lower and beneficial impacts (Smetana et *al.*, 2016). Cow manure's positive impacts from waste reduction outweighed negatives from insect production, while chicken manure, with its leftover treatment, posed environmental challenges. The environmental impacts of solid fraction pig manure fell between cow and chicken manure. Comparisons with Smetana et al. (2016) are complex due to differences in frass handling and modelling burdens or credits related to manure provision.

3.1.4.5 Conclusion

The aim of the current study was to assess the environmental implications of insect production from the angles of agro-residue management and feed source for livestock.

Regardless of the diet, insects as processed animal protein performed worse than conventional soy- and fishmeal feed in our models. However, it is still premature to disregard insect larvae production as a more sustainable pathway to provide protein for animal feed. Our results indicated that the balance between insect production and alternative changes drastically by varying the insect feed used and the source of energy for heating during insect production. Thus, investigations that identify the circumstances under which insect production is favourable, such as location, energy source, type of agro-residues and conditions of production are needed to make insect production more sustainable.

3.1.4.6 Dashboard indicators

LL#40 Insect breeding as an alternative protein source on solid agro-residues (manure and plant wastes) (*: Qualitative DBI assessment : **+** *improvement (<-10%)*, **o** *no change/unknown (-10% - +10%)*, **-** *deterioration (>+10%))*

The reference is the provision of 1,000 kg protein feed through soy

Table 16. LL# 40: Comparison of dashboard indicator and life cycle assessment results

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication *	Comment	Quantitative estimate per FU of 1,000 kg protein	LCA indication *	Re-evaluation of DBI irt. LCA results*
Use of Primary Resources	**Rock Phosphate** (Reduction in mineral phosphorus consumption)	**+**	If insect frass is applied as a fertiliser, less rock phosphate must be used	Phosphorous, in ground [g] Diet 2: +50.16 kg P (>+100%) Diet 5: +124.13 kg P (>+100%)	 - -	Non-manure diets: revised, more likely incr. Manure diets: revised, more likely incr.
	Natural Gas (Reduction in natural gas consumption in mineral fertiliser production)	**o**	Insect facilities must be heated	Gas, natural, in ground [h] Diet 2: +3.03E+04 m^3 (>+100%) Diet 5: +3.17E+04 m^3 (>+100%)	- -	Non-manure diets: revised, more likely incr. Manure diets: revised, more likely incr.

[g] Non-manure diet: The crops grown to feed the insects consume more P, than is reapplied the field via frass. Manure diet: Although the frass contains P, composted manure contains more

[h] All diets: The increase in natural gas results from heating requirements (as commented by technology provider) the most energy intense stage is the egg-to-larvae stage (prior to providing them the experimental diets)

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication *	Comment	Quantitative estimate per FU of 1,000 kg protein	LCA indication *	Re-evaluation of DBI irt. LCA results*
	Oil (Reduction in oil consumption in agricultural machinery)	o	Fuel will be necessary for manure transportation to the processing plant and transporting insect frass and insect products from the insect plant	Oil, crude, in ground[i] Diet 2: +1130.60 kg (>+100%) Diet 5: -40.46 kg (-31%) Diesel burnt in agricultural machinery [j] Diet 2: +1.18 MJ (>+100%) Diet 5: +8.59 MJ (>+100%)	- + - -	Non-manure diets: revised, more likely incr. Manure diets: revised, more likely decr. Non-manure diets: revised, more likely incr. Manure diets: revised, more likely incr.
	Electricity (Reduction in electricity consumption)	o	Powering insect facilities	Electricity consumption Diet 2: 7.42E+04 MJ (>+100%) Diet 5: 4.00E+04 MJ (>+100%)	- -	Non-manure diets: revised, more likely incr. Manure diets: revised, more likely incr.
	Water	o	Cleaning insect facilities	Water scarcity [k] Diet 2:	-	Non-manure diets: revised, more likely incr.

[i] Non-manure diets: the increase results from the non-agro-residue feed production for the insects. Manure diet: saving are achieved through the avoidance of soymeal

[j] This is not necessarily limited to the farm implementing acidification

[k] Non-manure diet: increase dominated by water consumed during crop production for feeding. Manure diet: water for cleaning (as commented by technology provider) & water consumption for feed production for egg-to-larvae rearing

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication *	Comment	Quantitative estimate per FU of 1,000 kg protein	LCA indication *	Re-evaluation of DBI irt. LCA results*
	(Reduction in water consumption)			1.69E+05 m^3 depriv. (>+100%) Diet 5: 2.14E+04 m^3 depriv. (>+100%)	-	Manure diets: revised, more likely incr.
	Soil quality (Improvement in soil quality)	+	Insect frass is rich in organic matter	Land use[l] Diet 2: 1.86E+06 Pt (>+100%) Diet 5: -2.38E+05 Pt (-69%)	- +	Non-manure diets: revised, more likely incr. Manure diets: confirmed
	Nutrient recovery (Increase in nutrient recycling)	+	Insects incorporate N into protein, but also P, K and other minerals. Efficiencies vary depending on the insect species and diet used	Diet 2:[m] N fertiliser: -49.91 kg (->100%) P fertiliser: 87.56 kg (>+100%) K fertiliser: -145.01 kg (>-100%) Diet 5: N fertiliser: +219.63	+ - +	Diet 2: N fertiliser: revised, negligible P fertiliser: revised, more likely, incr. K fertiliser: confirmed Diet 5:

[l] Non-manure diets: deterioration due to crop production for insect feed. Manure diet: improvements through the avoidance of soymeal production

[m] We looked at 'market for inorganic fertiliser as N, K2O and P2O5

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment	DBI Indication *	Comment	Quantitative Dashboard Indicator based on LCA assessment: Quantitative estimate per FU of 1,000 kg protein	LCA indication *	Re-evaluation of DBI irt. LCA results*
				kg (>+100%)	-	N fertiliser: revised, more likely, incr.
				P fertiliser: +299.55 kg (>+100%)	-	P fertiliser: revised, more likely, incr.
				K fertiliser: +47 kg (>+100%)	-	K fertiliser: revised, more likely, incr.
	Others? Please specify			no further info		
Emissions to the environ-ment	**Ammonia (air emission)** (Reduction in NH_3 emissions)	**o**	Dependant on insect species and diet	Ammonia emission to air Diet 2: 84.42 kg NH_3 (>+100%) Diet 5: -32.73 kg NH_3 (>-100%)	- +	Non-manure diets: revised, more likely incr. Manure diets: revised, more likely decr.
	Dinitrogen monoxide (air emission) (Reduction in N_2O Emissions)	**o**	Dependant on insect species and diet (7 - 118 mg / kg DM larvae for BSF)	Dinitrogen monoxide emission to air Diet 2: 68.21 kg N_2O (>+100%) Diet 5: 42.39 kg N_2O (>+100%)	- -	Non-manure diets: revised, more likely incr. Manure diets: revised, more likely incr.
	Methane (air emission) (Reduction in CH_4 emissions)	**o**	Dependant on insect species and diet (5.5 - 49 mg / kg / DM larvae for BSF)	Methane emission to air Diet 2: 0.09 kg CH_4 (>+100%) Diet 5: -60.91 kg CH_4 (>-100%)	- +	Non-manure diets: revised, more likely incr. Manure diets: revised, more likely decr.

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication *	Comment	Quantitative estimate per FU of 1,000 kg protein	LCA indication *	Re-evaluation of DBI irt. LCA results*
	Nitrates (water emission) (Reduction in NO_3 emissions)	+	Insects are kept in containers; no water can escape from it. Secondary water emissions are possible when the insect frass is applied as a fertiliser on the field	Nitrate emission to water Diet 2: 814.83 kg NO3 (>+100%) Diet 5: 40.94 NO3 (>+100%)	 - -	Non-manure diets: revised, more likely incr. Manure diets: revised, more likely decr.
	Phosphorus (water emission) (Reduction of P Emissions)	+	Insects are kept in containers; no water can escape from it. Secondary water emissions are possible when the insect frass is applied as a fertiliser on the field	Phosphorous emission to water Diet 2: 1.10 kg P (+62%) Diet 5: 0.13 kg P (-93%) Phosphate emission to water Diet 2: 17.60 kg PO_4^{3-} (+67%) Diet 5: 2.48 kg PO_4^{3-} (-91%)	 - + - +	Phosphorous: Non-manure diets: revised, more likely incr. Manure diets: confirmed Phosphate: Non-manure diets: revised, more likely incr. Manure diets: confirmed

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment	DBI Indication*	Comment	Quantitative Dashboard Indicator based on LCA assessment: Quantitative estimate per FU of 1,000 kg protein	LCA indication*	Re-evaluation of DBI irt. LCA results*
	Particulate matter (Reduction of particulate matter formation)	**o**	Has not been quantified, but particulate matter is a real possibility (insect allergies linked to fine dust inhalation is a known phenomenon for mealworm and locusts)	Particulates emission to air Diet 2: 2.20E-03 diseases incr. (>+100%) Diet 5: -4.38E-03 disease increase (>-100%)	**-** **+**	Non-manure diets: revised, more likely incr. Manure diets: revised, more likely decr.
	Others? Please specify		Unpleasant scent is possible nearby insect breeding facilities (especially BSF)	Not quantifiable through LCA		
Resilience to climate change	**Carbon footprint** (Reduction of carbon footprint)	**o**		Climate change [CO_2 eq] Diet 2: 1.06 kg (>+100%) Diet 5: 0.38 kg (>+100%)	**-** **-**	Non-manure diets: revised, more likely incr. Manure diets: revised, more likely incr.
	Effective SOM (Effective Soil Organic Matter Improvement)	**o**		Carbon | Emission from soil Diet 2: 1.05 kg C (>+100%) Diet 5: 0.38 kg C (>-100%)	**-** **+**	Non-manure diets: revised, more likely incr. Manure diets: revised, more likely decr.

3.1.4.6.1 Comparison of LCA and DBI results

In terms of rock phosphate extraction, the LCA study came to a different conclusion than the DBI study. That is mostly due to the scope of each study. In the LCA study, the P fertiliser needed to fertilise the crops produced to feed the insect larvae were included and the study took a closer look at the P mass balances of manure composting versus manure insect rearing. In the composting alternative, more P ends up in the fields than in the case of insect rearing. In terms of natural gas, the assumption of an increase in natural gas use could be confirmed and solidified. As for oil, the results of the LCA depended on the type of feed. In the manure-based diet a reduction could be achieved due to the avoidance of soymeal production. In the non-manure-base diet the increase resulted mostly from the production of insect feed. The assumption made in the DBI assessment of an increase in oil consumption due to the transportation of frass, manure and insect could not be confirmed – while likely true it has a minor impact on overall results.

Overall, the LCA study found that emissions to the environment increased in all emissions evaluated when looked at non-manure-based diets but to decrease in some cases when looking at the manure-based diet.

The agreement between DBI and LCA results varied greatly between categories. In many categories, the DBI study could either not conclude or found no change from a switch in technology. Here, the LCA study could provide additional information.

3.1.4.7 References

Abd El-Hack, M., Shafi, M., Alghamdi, W., Abdelnour, S., Shehata, A., Noreldin, A., Ashour, E., Swelum, A., Al-Sagan, A., Alkhateeb, M., Taha, A., Abdel-Moneim, A.-M., Tufarelli, V., & Ragni, M. (2020). Black Soldier Fly (Hermetia illucens) Meal as a Promising Feed Ingredient for Poultry: A Comprehensive Review. Agriculture, 10(8), 339. https://doi.org/10.3390/agriculture10080339

Amrul, N. F., Ahmad, I. K., Basri, N. E. A., Suja, F., Jalil, N. A. A., & Azman, N. A. (2022). A Review of Organic Waste Treatment Using Black Soldier Fly (Hermetia illucens). Sustainability (Switzerland), 14(8). https://doi.org/10.3390/su14084565

Beyers, M., Coudron C., Ravi R., Meers R., Bruun S., (2023). Black soldier fly larvae as an alternative feed source and agro-waste disposal route - A life cycle perspective. Resour. Conserv. Recycl. 192, https://doi.org/10.1016/j.resconrec.2023.106917

Brockmann, D., Pradel, M., Hélias, A., (2018). Agricultural use of organic residues in life cycle assessment: Current practices and proposal for the computation of field emissions and of the nitrogen mineral fertilizer equivalent. Resources, Conservation and Recycling. 133, 50-62

D'Haene, A., & Vannecke, T. (2020). VCM-ENQUETE OPERATIONELE STAND VAN ZAKEN MESTVERWERKING IN VLAANDEREN 2020. https://cdn.digisecure.be/vcm/202179144757785_rapport-mestverwerking-2020-finaal.pdf

De Clercq, L., Willeghems, G., Michels, E., Meers, E. Buysse, J. (2015) The impact of concentrated pig production in Flanders: a spatial analysis https://ec.europa.eu/information_society/newsroom/image/document/2016-48/buysse_presentation_workshop_40228.pdf

De Ruijter, F. J., Huijsmans, J. F. M., Zanten, M. C., & Asman, W. A. H. (2013). Ammonia emission from standing crops and crop residues: contribution to total ammonia emission in the Netherlands.

https://library.wur.nl/WebQuery/wurpubs/fulltext/290558%0Apapers3://publication/uuid/C2C35CDA-A263-4B07-81FA-AE6EA3E039CF

Dobbelaere, A.De, Vervisch, B., Annicaert, B., Ryckaert, B., Lebuf, V., Driessche, J.Van, 2015. Case study report - development of agro-sidestreams for bioenergy [WWWDocument]. URL https://www.izes.de/sites/default/files/publikationen/ARBOR_agroresidues_case_study_report.pdf

EEA (2019). EMEP/EEA air pollutant emission inventory guidebook 2019 (3.D Crop production and agricultural soils)

European Commission. (2021). COMMISSION RECOMMENDATION of 16.12.2021 on the use of the Environmental Footprint methods to measure and communicate the life cycle environmental performance of products and organisations.

Fazio, S. Biganzioli, F. De Laurentiis, V., & Zampori, L., Sala, S. Diaconu, E. (n.d.). Supporting information to the characterisation factors of recommended EF Life Cycle Impact Assessment methods. 49. https://eplca.jrc.ec.europa.eu/permalink/TR_SupportingCF_FINAL.pdf

Lemmens, B., et al. (2007). "Beste Beschikbare Technieken (BBT) voor mestverwerking." Ghent, Belgium 359

Liu, T., Klammsteiner, T., Dregulo, A. M., Kumar, V., Zhou, Y., Zhang, Z., & Awasthi, M. K. (2022). Black soldier fly larvae for organic manure recycling and its potential for a circular bioeconomy: A review. Science of the Total Environment, 833, 155122. https://doi.org/10.1016/j.scitotenv.2022.155122

Lopes, I. G., Yong, J. W., & Lalander, C. (2022). Frass derived from black soldier fly larvae treatment of biodegradable wastes. A critical review and future perspectives. In Waste Management (Vol. 142, pp. 65–76). Elsevier Ltd. https://doi.org/10.1016/j.wasman.2022.02.007

Moreno Ruiz E., Valsasina L., FitzGerald D., Symeonidis A., T., D., Müller J., Minas N., Bourgault G., Vadenbo C., I. D., & Wernet, G. (2020). Documentation of changes implemented in the ecoinvent database v3.7. 114. https://ecoinvent.org/wp-content/uploads/2021/01/change-report_v3-7_20200918.pdf

Nemecek, T., & Schnetzer, J. (2011). Methods of assessment of direct field emissions for LCIs of agricultural production systems. Agroscope Reckenholz-Tänikon Research Station ART

Parodi, A., De Boer, I. J. M., Gerrits, W. J. J., Van Loon, J. J. A., Heetkamp, M. J. W., Van Schelt, J., Bolhuis, J. E., & Van Zanten, H. H. E. (2020). Bioconversion efficiencies, greenhouse gas and ammonia emissions during black soldier fly rearing – A mass balance approach. Journal of Cleaner Production, 271, 122488. https://doi.org/10.1016/j.jclepro.2020.122488

Parodi, A., Gerrits, W. J. J., Van Loon, J. J. A., De Boer, I. J. M., Aarnink, A. J. A., & Van Zanten, H. H. E. (2021). Black soldier fly reared on pig manure: Bioconversion efficiencies, nutrients in the residual material, greenhouse gas and ammonia emissions. Waste Management, 126, 674–683. https://doi.org/10.1016/j.wasman.2021.04.001

Rummel, P. S., Beule, L., Hemkemeyer, M., Schwalb, S. A., & Wichern, F. (2021). Black soldier fly diet impacts soil greenhouse gas emissions from frass applied as fertilizer. Frontiers in Sustainable Food Systems, 5, 709993

Smaling E. (1993). An Agro-ecological Framework for Integrated Nutrient Management with Special Reference to Kenya

Smetana, S., Palanisamy, M., Mathys, A., Heinz, V. (2016). Sustainability of insect use for feed and food: life cycle assessment perspective. J. Clean. Prod. 137, 741–751. https://doi.org/10.1016/j.jclepro.2016.07.148

Smetana, S., Schmitt, E., & Mathys, A. (2019). Sustainable use of Hermetia illucens insect biomass for feed and food: Attributional and consequential life cycle assessment. Resources, Conservation and Recycling, 144, 285–296. https://doi.org/10.1016/j.resconrec.2019.01.042

Smetana, S., Spykman, R., & Heinz, V. (2021). Environmental aspects of insect mass production. Journal of Insects as Food and Feed, 7(5), 553–571. https://doi.org/10.3920/JIFF2020.0116

Sommer, S., Webb, J., Hutchings, N., (2019) New Emission Factors for Calculation of Ammonia Volatilization From European Livestock Manure Management Systems. Frontiers in Sustainable Food System,3 https://doi.org/10.3389/fsufs.2019.00101

van Huis, A., Oonincx, D. G. A. B., Rojo, S., & Tomberlin, J. K. (2020). Insects as feed: house fly or black soldier fly? Journal of Insects as Food and Feed, 6(3), 221–229. https://doi.org/10.3920/JIFF2020.X003

Velthof, G., Nelemans, J., Oenema, O. & Kukiman, P. (2005). Gaseous nitrogen and carbon losses from pig manure derived from different diets. Journal of Environmental Quality, 34(2), 698-706 https://doi.org/10.2134/jeq2005.0698

Zhang, Z., Liu, D., Yan, Q., Shuanglai, L., Yunfeng, C., Cheng, H. (2021) Mitigation of carbon and nitrogen losses during pig manure composting: A meta-analysis. Science of The Total Environment, 783, https://doi.org/10.1016/j.scitotenv.2021.147103

3.1.5 LL#49: Comparison of struvite production by crystallisation and untreated liquid digestate spreading: Identifying break-even points for transportation (CARTIF)

Longlist #49 title: Nitrogen and phosphorus recovery from pig manure via struvite crystallization and design of struvite based tailor-made fertilisers

Francisco Corona[1], Francisco Verdugo[1]

[1] CARTIF Technology Centre, Valladolid, Spain.

3.1.5.1 Introduction

Manure has a high potential as an organic fertiliser, due to its content of nitrogen, phosphorus, and potassium, among others. However, livestock intensification generates excess amounts of manure in very specific areas, making it difficult to manage. In such areas, the fields are not in need of nutrients and additional application only harms neighbouring ecosystems. The nutrients cannot be taken up by the crops and transporting the nutrients away from those fields can be a good idea. The nutrients can be applied to needing fields and inorganic fertiliser production can be avoided.

One promising method to recover and concentrate nitrogen and phosphorus from agricultural waste is precipitation. In chemical precipitation processes, nutrients are recovered and separated by crystallisation. One resulting product is struvite (an ammonium, phosphorus, and magnesium salt) which can be used as a biofertiliser (Le Corre et al. 2009). However, if the fields close to livestock production are in need of nutrients, it is unlikely that investing additional energy and materials in the crystallisation process makes sense.

The present life cycle assessment study looks at the interdependencies of transportation, fertiliser needs and manure treatment technology.

3.1.5.2 Material & method

3.1.5.2.1 Goal & scope

The goal of this LCA was to assess the environmental implications of struvite recovery from digestated liquid pig manure compared against a baseline without struvite recovery combined with different transportation and fertilisation scenarios (Figure 15). Application of the liquid fraction on fields nearby is not possible because they are already overfertilised, so all of the manure has to be transported to fields far away from the farm. Alternatively, the nutrients can be precipitated and then only the precipitate needs to be transported. However, crystallisation requires energy and resources and therefore only makes sense if the receiving field is so far away that sufficient environmental impacts are saved on the transportation. The current assessment will identify whether there is a 'break-even point' in terms of transport distance after which the transport of liquid fraction becomes so impactful on the environment that it makes sense to perform struvite recovery. Data collected for this assessment came from a pilot crystallisation plant. In addition, a sensitivity analysis was carried out in which we considered the energy consumption of a crystallisation plant at industrial scale. To achieve this, we considered an industrial plant located in Spain which produces 420 kg of struvite per day with a crystallisation energy consumption of 0.345 kWh/ton digestate managed (Sánchez et al. 2011).

The functional unit of this LCA is the '*disposal of 1 ton liquid fraction of digestate'*.

Following paragraphs describe the different scenarios considered in this LCA:

a) **Land application of the untreated liquid fraction of digestate *(Baseline system).*** This scenario involves the direct application of liquid digestate in fields. No treatment is applied and only transportation and field application are considered.

Regarding transportation of the liquid fraction, different distances have been considered to explore the relation between distance and overall environmental impacts. After transportation, the liquid digestate is spread on agricultural fields. Depending on the assumed N nutrient status, credits for the avoidance of mineral nitrogen fertiliser are given or not (Figure 15, blue box). Credits for the avoidance of P fertiliser are granted in all cases. Atmospheric emissions from field application are excluded from the system limits.

b) **Struvite crystallisation – pilot and industrial scale.** Again, the system starts with the digested liquid fraction which is now entering the crystallisation plant. During the crystallisation phase, the addition of chemicals ($MgCl_2 \cdot 6H_2O$ and NaOH) is necessary for the precipitation. This reaction takes place at 25ºC degrees. After the crystallisation phase, struvite is separated from the liquid fraction and dried in an oven. The liquid part is considered as liquid fertiliser which is transported and spread in fields close-by the place of production. These field are considered overfertilised; thus, inorganic fertiliser production is not avoided. We assumed a transport distance of 15 km (Corona, F. 2020) because transporting this product over longer distances is uneconomical. To explore sensitivity, the struvite is transported over varying distances and N fertiliser credits are included or excluded depending on the fertilisation status of receiving fields.

As for the baseline system, atmospheric emissions from the crystallisation process and the transportation and spreading of the liquid fraction are excluded from the assessment.

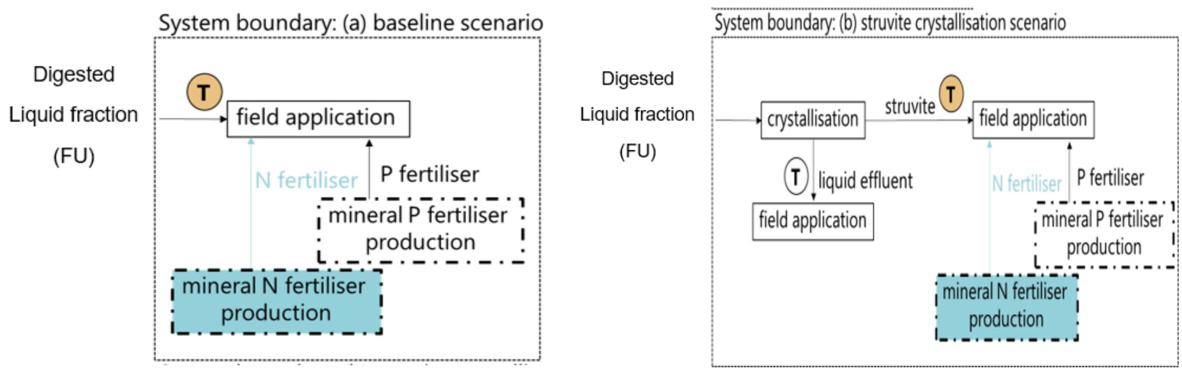


Figure 15. LL# 49: System boundaries

Structure of the analysed system. Boxes indicate main activities associated with direct land application of digested liquid manure (a) and struvite recovery plus land application of liquid effluent and struvite (b). Blue boxes indicate scenario analysis including nitrogen fertiliser credits. Dashed lines indicate credits through avoidance. Arrows indicate flows of products. T: transportation with varying distances. FU: functional unit.

3.1.5.2.2 Inventory

The foreground data required for the assessment was gathered during the experimental phase carried out at CARTIF. The experiments are described in more detailed in D.2.6. In short, struvite production, at the pilot plant at the CARTIF facilities starts with the reception of digestate from an anaerobic digestion plant and includes crystallisation and drying processes.

For the background data, ecoinvent 3.8 processes have completed the model.

All analyses were performed in SimaPro 9.3.0.2 LCA software with an attributional approach.

Table 17. LL# 49: Life cycle inventory

Struvite recovery from liquid fraction pig manure.

Process	Sub-process	Description	Unit	Value/UF
Untreated liquid fraction from digestate (baseline system)				
Transport		Liquid digestate	km	Under study
Spreading		Liquid digestate	kg	1000
Credits		From Liquid fertiliser[1] ertiliser	N kg	3.5
		From Liquid fertiliser	P_2O_5 kg	0.452
Struvite production by crystallisation - pilot scale (project system)				
Struvite production	Crystallization	Electricity	kWh	22.54 – (0.345[b])
		MgCl2 6 H2O	kg	1.9
		NaOH	kg	0.3
	Drying	Electricity	kWh	31.34
Products		Struvite production	kg	4.2
		Liquid fertiliser production	kg	995.8
Transport		Struvite	kg	Under study
		Liquid fertiliser	km	15
Spreading		Struvite applied in field	kg	4.2
		Liquid fertiliser applied in field	kg	995.8
Credits		From Struvite	N kg[a]	0.2
			P_2O_5 kg	0.39
[a]Only for N-deficient fields; [b]Energy consumption at industrial scale				

3.1.5.3 Results

3.1.5.3.1 Impact assessment

This section presents the results of the assessment carried out. The impacts of each system were determined using Environmental Footprint 3.0 methodology (EC 2021).

Table 18 and Figure 16 represent the environmental impact of 1 t of treated liquid digestate in a crystallisation plant (xxP), on a pilot or industrial scale, as well as, the application and transport of untreated liquid digestate (xxB). In addition, the fertilisation status of the fields has been considered. Thus, the assumption of N-deficient fields (UFx) or N-overfertilised fields (OFx) has been considered. The values in brackets indicate the brake-even transport distance between no treatment and crystallisation. Thus, as soon as the fields on which the untreated liquid digestate would be spread, are further away than the value in brackets, spending additional material and energy on the crystallisation process makes environmental sense. In contrast, if needing fields are closer than the value in brackets, then struvite crystallisation is unlikely to make sense from an environmental perspective.

Table 18. LL# 49: eLCA results

Environmental impact potentials of the different scenarios at the break-even point and distance at which the break-even point takes place for the impact category.

Impact category	Units Env. Impact (maximum? distance)	Pilot plant scale		Industrial scale	
		UF	OF	UF	OF
CCP	kg CO_2 eq (km)	2,38E+01 (245)	2,47E+01 (150)	7,21E+00 (143)	8,05E+00 (48)
OP	kg CFC^{11} eq (km)	6,43E-06 (216)	6,52E-06 (169)	5,36E-06 (187)	5,46E-06 (141)
IRP	kBq U-235 eq (km)	1,46E+01 (1199)	1,45E+01 (1139)	2,44E+00 (250)	2,48E+00 (198)
PP	kg NMVOC eq (km)	9,29E-02 (286)	9,43E-02 (220)	2,84E-02 (121)	3,01E-02 (56)
RIP	disease inc. (km)	8,30E-07 (128)	8,69E-07 (77)	4,34E-07 (98)	4,72E-07 (47)
HNP	CTUh (km)	1,69E-07 (191)	2,92E-07 (171)	1,06E-07 (158)	1,15E-07 (79)
HCP	CTUh (km)	1,01E-08 (337)	1,05E-08 (205)	3,60E-09 (223)	4,22E-09 (95)
AP	mol H+ eq (km)	1,78E-01 (614)	1,84E-01 (404)	3,34E-02 (302)	3,89E-02 (92)
FEP	kg P eq (km)	9,90E-03 (1245)	1,00E-02 (1055)	3,62E-03 (650)	3,73E-03 (460)
MEP	kg N eq (km)	3,29E-02 (449)	3,36E-02 (329)	9,81E-03 (204)	1,04E-02 (84)
TEP	mol N eq (km)	3,23E-01 (636)	3,42E-01 (305)	8,39E-02 (404)	1,03E-01 (73)
FTP	CTUe (km)	4,14E+02 (332)	4,21E+02 (232)	1,51E+02 (176)	1,60E+02 (97)
LUP	Pt (km)	1,09E+02 (82)	1,11E+02 (63)	5,38E+01 (50)	5,56E+01 (31)
WP	m^3 depriv. (km)	1,72E+01 (3133)	1,75E+01 (2264)	6,01E+00(1740)	6,40E+00 (871)
ERP	MJ (km)	5,26E+02 (305)	5,38E+02 (217)	1,27E+02 (143)	1,39E+02 (56)
MRP	kg Sb eq (km)	4,56E-05 (753)	5,98E-05 (237)	2,53E-05 (715)	3,94E-05 (239)

UF: Under N fertiliser baseline; UF: Under N fertiliser project; OF: Over N fertiliser baseline; OF: Over N fertiliser project;

At this point, considering the variation of different minimum distances between all the impact categories considered in this assessment, weighting has been applied. Weighting allows to determine a distance considering the importance of each impact category. The methodology selected for the weighting was Environmental footprint methodology.

After weighting, minimum transport distances were determined of a) 632 km for N-deficient fields and b) 441 km for over-fertilised fields at pilot plant scale and c) 356 km for underfertilised fields and d) 164 km in overfertilised fields at industrial scale.

Pilot plant scale

Nutri2Cycle
Nurturing the Circular Economy

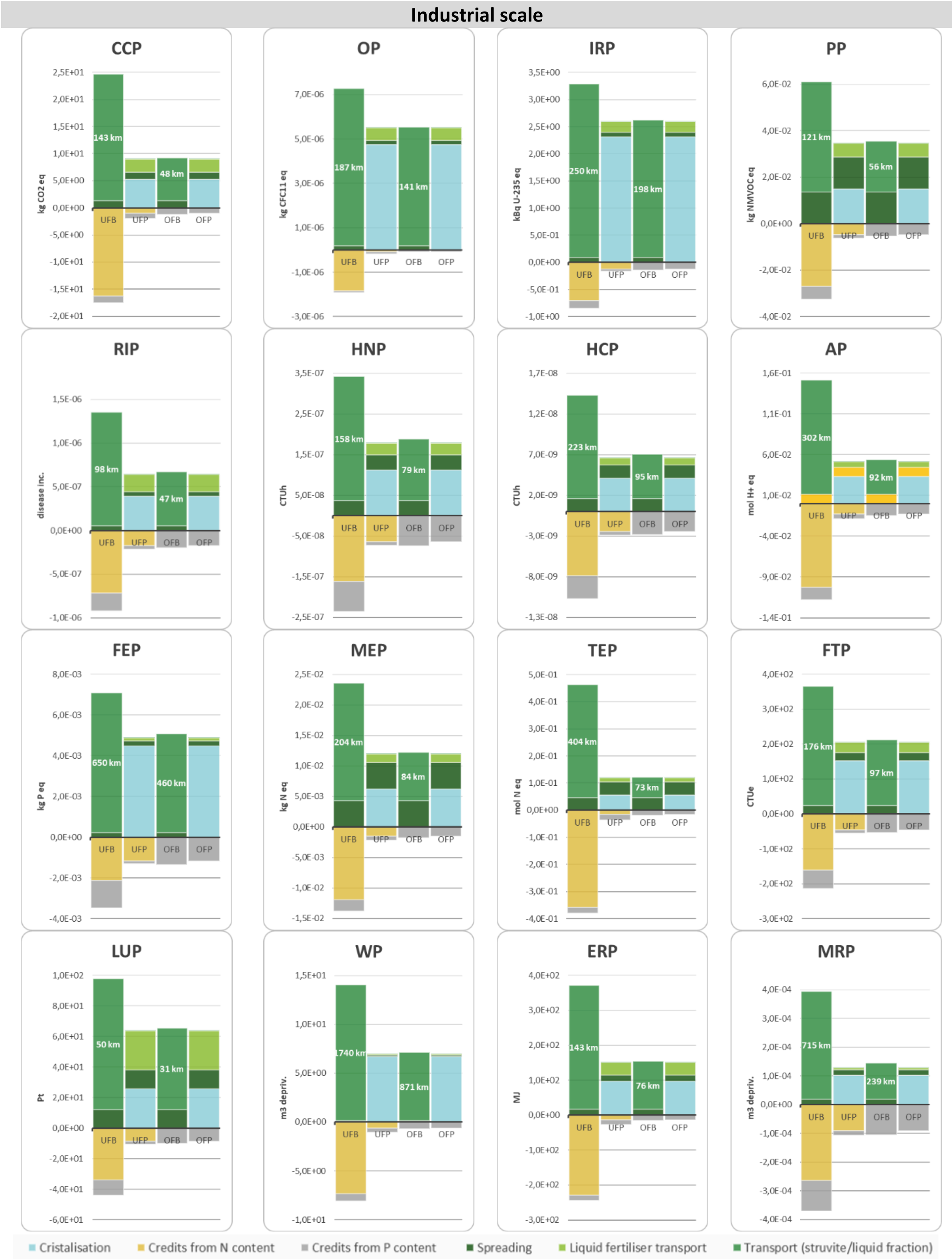


Figure 16. LL# 49: eLCA results.

Environmental impact result at the pilot plant and industrial scale in different life stages. In the transport bar the break-even point at which the B scenarios are becoming as impactful as the P scenarios are indicated

3.1.5.4 Interpretation

Struvite crystallisation at pilot scale: If the fields, receiving untreated digestate liquid fraction, are in need of nitrogen fertiliser, the maximum weighted reasonable transport distance is 632 km. If the fields are not in need of nitrogen fertiliser, the maximum weighted distance decreases to 441 km. This is due to the avoidance of inorganic nitrogen fertiliser production given the nitrogen content in liquid fraction of digestate when it is spread on N-needing fields.

In view of these results, it may be worth highlighting that a) *ionising radiation*; b) *freshwater eutrophication* and c) *water use* were most sensitive to transport distance. For these impact categories energy consumption for the crystallisation process contributed with more 50% to the total impact (Figure 17). In *ionising radiation*, the energy contribution reached more than 80%. In addition, with regards to *ozone depletion* and *water use*, sodium hydroxide consumption had great contributions Figure 17.

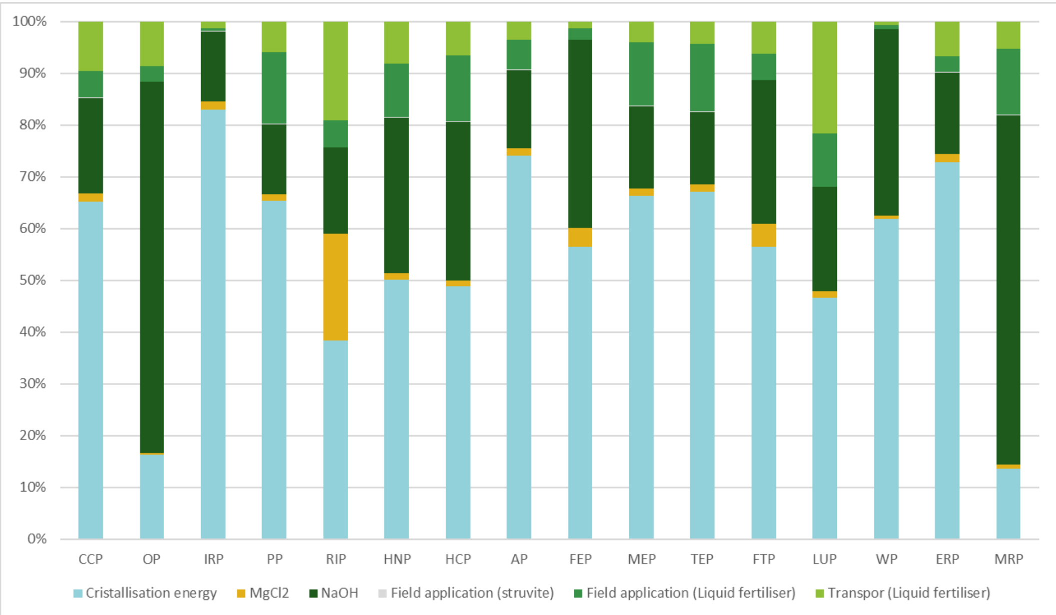


Figure 17. LL# 49: eLCA results – impact contributions from individual processes (% of total impact) - 1 Environmental impact pattern of struvite crystallisation at *pilot plant* scale.

Struvite crystallisation at industrial scale: When considering struvite crystallisation at an industrial scale, the burdens related to struvite production decrease considerable compared to the production at pilot scale. In the case of N-deficient fields, the maximum transport distance for the untreated digested liquid fraction is reduced by 50% (356 km) and struvite crystallisation starts to make sense for shorter distances. In the case of overfertilised fields, the transport distance even decreases by as much as 60% (to 164 km) (compare Table 18).

The main impact categories contributing to increased limit transport distances for untreated digestate liquid fraction are *ionising radiation* (IRP), *freshwater eutrophication* (FEP), *water use* (WUP) and *minerals and metals resource use* (MRP). The sensitivity analysis has shown that energy consumption has a high contribution to IRP, FEP and WP impact categories. However, for the *use of mineral and metal resources* the main contribution comes from the use of NaOH. Sodium hydroxide is used to increase the pH of digestate in the crystallisation reaction.

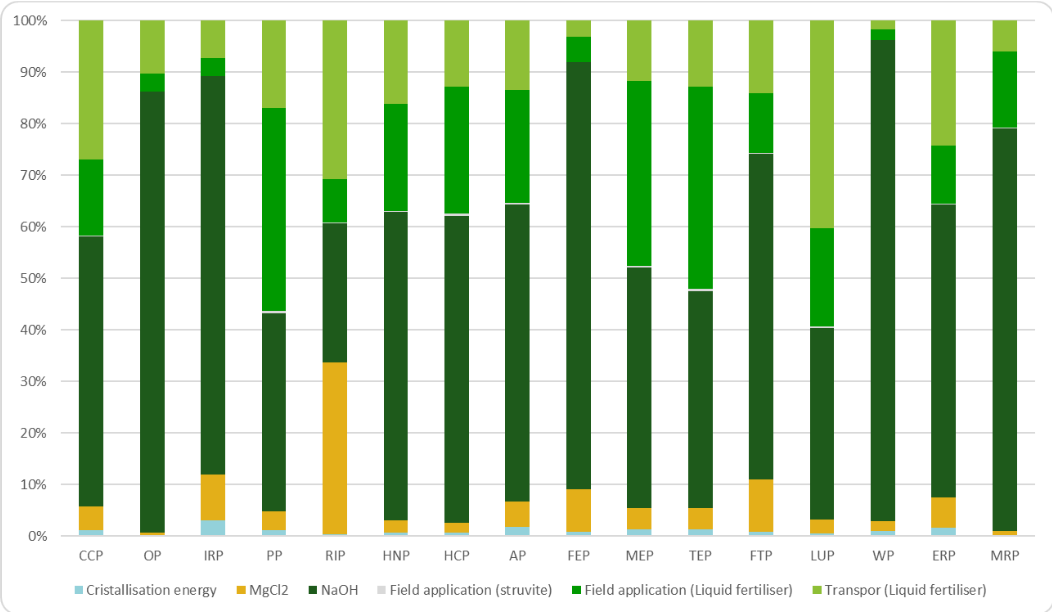


Figure 18. LL# 49: eLCA results – impact contributions from individual processes (% of total impact) - 2 Environmental impact pattern of struvite crystallisation at *industrial* scale

Figure 18 shows that once energy consumption has been reduced, most of the environmental impact contributions come from sodium hydroxide required in the crystallisation process.

At this point, given the strong influence of NaOH production on *water use potential* and the great role water use played in determining the maximum transport distance, it could be important to mention that this impact category has been assessed by the AWARE methodology (Sustainability. P., 2020). AWARE is a regionalised, water use midpoint indicator representing the relative Available WAter REmaining per area in a watershed after the demand of humans and aquatic ecosystems has been met. It assesses the potential of water deprivation, to either humans or ecosystems, building on the assumption that the less water remains available per area, the more likely another user will be deprived (Various authors, PRé Sustainability. 2020). Thus, the environmental burdens caused by NaOH production shall be allocated in the production region of this chemical.

Despite of the environmental burdens related to struvite production, its crystallization in fluidised beds is an economically sustainable option to process waste and mitigate environmental impacts (Sampat, A. M., et al., 2018). As has been shown in this work, several factors contribute to this fact, on the one hand, struvite is a more concentrate material than sewage sludge considering plant available P content, which contributes to a larger amount of avoided fertiliser.

At this point, aligned with Némethy, A. (2016), recovery of struvite reduces the impact of transportation, as struvite allowing the supply of organic fertilisers to more distant crop fields and, thus, avoiding nutrient losses due to Nitrogen over-fertilisation of fields close to the sludge production area.

These losses lead to environmental problems, such as the release of greenhouse gases, pollution of water bodies, soil acidification, or biodiversity reduction. The atmospheric level of N pollution is expected by, 2050, to be in the range 102–156% higher than in 2010 with the agricultural sector accounting for 60% of this increase (Martínez-Dalmau, J. et al, 2021)

3.1.5.5 Conclusion

In conclusion, this work has shown that on an industrial scale, the transport of struvite to fields located less than 200 km away is less impactful for 13 of 16 impact categories. Thus, this practice can be a sustainable management option that contributes to addressing the main challenges such as nitrogen over-fertilisation of crop fields, resource use, the environmental pollution or the transport of untreated digestate over long distances.

Nutri2Cycle
Nurturing the Circular Economy

3.1.5.6 Dashboard indicators

Longlist #49 title: Nitrogen and phosphorus recovery from pig manure via struvite crystallization and design of struvite based tailor-made fertilisers

(*: Qualitative DBI assessment : **+** *improvement (<-10%)*, **o** *no change/unknown (-10% - +10%)*, **-** *deterioration (>+10%))*

Table 19. LL# 49: Comparison of dashboard indicator and life cycle assessment results

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment	DBI Indication *	Comment	Quantitative Dashboard Indicator based on LCA assessment Quantitative estimate per FU of 1 t liquid fraction digestate	LCA indication*	Re-evaluation of DBI irt. LCA results
Use of Primary Resources	**Rock Phosphate** (Reduction in mineral phosphorus consumption)	+		Phosphorous, in ground (g) **Pilot scale** Under N: +31 (>+10%) Over N: +31 (>+10%) **Industrial scale** Under N: +31 (>+10%) Over N: +31 (>+10%)	**Pilot scale** Under N: **-** Over N: **-** **Industrial scale** Under N: **-** Over N: **-**	**Pilot scale** Under N: revised, increase. Over N: revised, increase. **Industrial scale** Under N: revised, increase. Over N: revised, increase.
	Natural Gas (Reduction in natural gas consumption in mineral fertiliser production)	+		Gas, natural, in ground (Nm^3) **Pilot scale** Under N: +6.88 (>+10%) Over N: +1.69 (>+10%) **Industrial scale** Under N: +5.1 (>+10%) Over N: -0.12 (<-10%)	**Pilot scale** Under N: **-** Over N: **-** **Industrial scale** Under N: **-** Over N: **+**	**Pilot scale** Under N: revised, increase. Over N: revised, increase. **Industrial scale** Under N: revised, increase. Over N: confirmed
	Oil (Reduction in oil consumption in agricultural machinery)	+		Oil, crude, in ground (tn.lg) **Pilot scale** Under N: -7.3 (<-10%) Over N: -7.6 (<-10%) **Industrial scale**	**Pilot scale** Under N: **+** Over N: **+** **Industrial scale**	**Pilot scale** Under N: confirmed Over N: confirmed **Industrial scale** Under N: confirmed

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication *	Comment	Quantitative estimate per FU of 1 t liquid fraction digestate	LCA indication*	Re-evaluation of DBI irt. LCA results
				Under N: -8.0 (<-10%) Over N: -8.3 (<-10%) Diesel burnt in agricultural machinery (MJ) **Pilot scale** Under N: +2.1 (>+10%) Over N: +2. 0 (>+10%) **Industrial scale** Under N: +1.1 (>+10%) Over N: +0.99 (>+10%)	Under N: **+** Over N: **+** **Pilot scale** Under N: - Over N: - **Industrial scale** Under N: - Over N: -	Over N: confirmed **Pilot scale** Under N: revised, increase. Over N: revised, increase. **Industrial scale** Under N: revised, increase. Over N: revised, increase.
	Electricity (Reduction in electricity consumption)	+		Electricity consumption (GJ) **Pilot scale** Under N: +1.7 $\cdot 10^6$ (>+10%) Over N: +1.0$\cdot 10^6$ (>+10%) **Industrial scale** Under N: +1.3 $\cdot 10^6$ (>+10%) Over N: +6.2$\cdot 10^5$ (>+10%)	**Pilot scale** Under N: - Over N: - **Industrial scale** Under N: - Over N: -	**Pilot scale** Under N: revised, increase. Over N: revised, increase. **Industrial scale** Under N: revised, increase. Over N: revised, increase.
	Water (Reduction in water consumption)	+		Water scarcity (m^3 depriv) **Pilot scale** Under N: +16.6 (>+10%) Over N: +4.5 (>+10%) **Industrial scale** Under N: +16.1 (>+10%) Over N: +4.9 (>+10%)	**Pilot scale** Under N: - Over N: - **Industrial scale** Under N: - Over N: -	**Pilot scale** Under N: revised, increase. Over N: revised, increase. **Industrial scale** Under N: revised, increase. Over N: revised, increase.

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment	DBI Indication *	Comment	Quantitative Dashboard Indicator based on LCA assessment Quantitative estimate per FU of 1 t liquid fraction digestate	LCA indication*	Re-evaluation of DBI irt. LCA results
	Soil quality (Improvement in soil quality)	+		Land use (Pt) **Pilot scale** Under N: -209 (<-10%) Over N: -266 (<-10%) **Industrial scale** Under N: -211 (<-10%) Over N: -264 (<-10%)	**Pilot scale** Under N: **+** Over N: + **Industrial scale** Under N: **+** Over N:**+**	**Pilot scale** Under N: confirmed Over N: confirmed **Industrial scale** Under N: confirmed Over N: confirmed
	Nutrients recovered (Nutrient recovered from agriculture and livestock systems)	+		-		No circularity indicator in LCA
	Renewable biomass	-		-		
	Others? Please specify	-		no further info		
Emissions to the environ-ment	**Ammonia (air emission)** (Reduction in NH_3 emissions)	+		Ammonia emission to air (kg) **Pilot scale** Under N: +20 (>+10%) Over N: +0.7 (>+10%) **Industrial scale** Under N: +19.4 (>+10%) Over N: +0.06 (>+10%)	**Pilot scale** Under N: **-** Over N: **-** **Industrial scale** Under N: **-** Over N: **-**	**Pilot scale** Under N: revised, increase. Over N: revised, increase. **Industrial scale** Under N: revised, increase. Over N: revised, increase.
	Dinitrogen monoxide (air emission) (Reduction in N_2O	+		Dinitrogen monoxide emission to air (kg) **Pilot scale**	**Pilot scale** Under N: **-** Over N: **+**	**Pilot scale** Under N: revised, increase. Over N: confirmed

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment	DBI Indication *	Comment	Quantitative Dashboard Indicator based on LCA assessment Quantitative estimate per FU of 1 t liquid fraction digestate	LCA indication*	Re-evaluation of DBI irt. LCA results
	Emissions)			Under N: +4.7 (>+10%) Over N: -0.4 (<-10%) **Industrial scale** Under N: +4.0 (>+10%) Over N: -1.1 (<-10%)	**Industrial scale** Under N: **-** Over N: **+**	**Industrial scale** Under N: revised, increase. Over N: confirmed
	Methane (air emission) (Reduction in CH_4 emissions)	**o**		Methane emission to air (mg) **Pilot scale** Under N: +56 (>+10%) Over N: -37 (<-10%) **Industrial scale** Under N: +20 (>+10%) Over N: -72 (<-10%)	**Pilot scale** Under N: **-** Over N: **+** **Industrial scale** Under N: **-** Over N: **+**	**Pilot scale** Under N: revised, increase. Over N: revised, increase. **Industrial scale** Under N: revised, increase. Over N: revised, increase.
	Nitrates (water emission) (Reduction in NO_3 emissions)	**+**		Nitrate emission to water (kg) **Pilot scale** Under N: +14.5 (>+10%) Over N: +12.9 (>+10%) **Industrial scale** Under N: +5.8 (>+10%) Over N: +4.2 (>+10%)	**Pilot scale** Under N: **-** Over N: **-** **Industrial scale** Under N: **-** Over N: **-**	**Pilot scale** Under N: revised, increase. Over N: revised, increase. **Industrial scale** Under N: revised, increase. Over N: revised, increase.
	Phosphorus (water emission) (Reduction of P Emissions)	**+**		Phosphorous emission to water (kg) **Pilot scale** Under N: +7.7 (>+10%)	**Pilot scale** Under N: **-** Over N: **-**	**Pilot scale** Under N: revised, increase. Over N: revised, increase. **Industrial scale**

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication *	Comment	Quantitative estimate per FU of 1 t liquid fraction digestate	LCA indication*	Re-evaluation of DBI irt. LCA results
				Over N: +6.3 (>+10%) **Industrial scale** Under N: +6.1 (>+10%) Over N: +4.7 (>+10%)	**Industrial scale** Under N: **-** Over N: **-**	Under N: revised, increase. Over N: revised, increase.
				Phosphate emission to water (kg) **Pilot scale** Under N: +33.7 (>+10%) Over N: +27.7 (>+10%) **Industrial scale** Under N: +15 (>+10%) Over N: +8.7 (>+10%)	**Pilot scale** Under N: **-** Over N: **-** **Industrial scale** Under N: **-** Over N: **-**	**Pilot scale** Under N: revised, increase. Over N: revised, increase. **Industrial scale** Under N: revised, increase. Over N: revised, increase.
	Particulate matter (Reduction of particulate matter formation)	**o**		Particulates emission to air (kg) **Pilot scale** Under N: +10.8 (>+10%) Over N: +0.7 (+10%) **Industrial scale** Under N: -20 (<-10%) Over N: -30 (<-10%)	**Pilot scale** Under N: **-** Over N: **o** **Industrial scale** Under N: **+** Over N: **+**	**Pilot scale** Under N: revised, increase. Over N: revised, increase **Industrial scale** Under N: revised, decrease. Over N: revised, decrease.
	Others? Please specify	-				
Resilience to climate change	**Carbon footprint** (Reduction of carbon footprint)	**+**		Climate change (kg CO_2 eq) **Pilot scale** Under N: +9.7 (>+10%) Over N: -5.7 (<-10%)	**Pilot scale** Under N: **-** Over N: **+**	**Pilot scale** Under N: revised, increase. Over N: confirmed **Industrial scale**

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication *	Comment	Quantitative estimate per FU of 1 t liquid fraction digestate	LCA indication*	Re-evaluation of DBI irt. LCA results
				Industrial scale Under N: -5.9 (<-10%) Over N: -22 (<-10%)	**Industrial scale** Under N: **+** Over N: **+**	Under N: confirmed Over N: confirmed
	Effective SOM (Effective Soil Organic Matter Improvement)	**o**		Carbon | Emission from soil (kg) **Pilot scale** Under N: -0.14 (<-10%) Over N: -0.42 (<-10%) **Industrial scale** Under N: -0.4 (<-10%) Over N: -0.7 (<-10%)	**Pilot scale** Under N: **+** Over N: **+** **Industrial scale** Under N: **+** Over N: **+**	**Pilot scale** Under N: revised, decrease. Over N: revised, decrease. **Industrial scale** Under N: revised, decrease. Over N: revised, decrease.
				Carbon dioxide | Emission from soil (kg) **Pilot scale** Under N: +637 (>+10%) Over N: +300 (>+10%) **Industrial scale** Under N: +357 (>+10%) Over N: +20 (>+10%)	**Pilot scale** Under N: **-** Over N: **-** **Industrial scale** Under N: **-** Over N: **-**	**Pilot scale** Under N: revised, increase Over N: revised, increase **Industrial scale** Under N: revised, increase Over N: revised, increase
	Renewable energy production (Renewable energy produced from biomass)	**o**		-		
	Others? Please specify	**-**				

3.1.5.6.1 Comparison of LCA and DBI results

The results presented in Table 19 show the comparison between DBI and LCA results. The criteria followed to check the relationship between LCA results against DBI were as follows. If the LCA shows a minimum of 10% reduction in impacts, we speak of true improvements, in contrast, if the LCA shows a min of 10% of increase in impacts, we say that the technology performs worse than the reference system. These results show a high difference between expected DBI results and LCA results obtained. That is due to the different baselines and scopes considered. At the beginning of the project the aim of the LCA was a comparison between: a) inorganic fertiliser application and b) struvite application. Thus, DBI table was filled considering these scenarios. However, during the project, the aim of the LCA was modified to get a suitable assessment of the main factor of this type of process, namely transport distances. Thus, the final scenarios assessed have been: a) untreated manure application and b) struvite application.

In conclusion, the results may be grouped in different categories depending on the contribution of transportation. For instance, LCA indicators such as: Phosphorous (in ground), Electricity consumption, Phosphate emissions to water are not influenced by transportation, thus, if we considered these indicators, untreated liquid manure application would be a better option than struvite production. Whereas most of the LCA indicators: Oil, crude (in ground), Land use, Methane emission to air, Particulates emission to air, Climate change, Carbon (Emission from soil) and Carbon dioxide (Emission from soil) receive high contributions due to transportation, thus, short transport distances are required for struvite production to be an alternative to the application of untreated liquid manure. Finally, the remaining LCA indicators, show some variability when considering the different variables assessed: pilot/industrial scale or under/overfertiliser fields.

3.1.5.7 References

Adani F., D'Imporzano G., Herrera A., Ashekuzzaman S.M., Forrestal P.J., Fangueiro D., Hendriks C., Meers E., Michels E., Akyol C., Vandendriessche S., 2022. D2.5 report: Data collection from investigations feeding WP3-WP5. Nutri2Cycle EU No. 773682. Available from www.nutri2cycle.eu

Corona Encinas, F. (2020). Estudio de la recuperación de nutrientes presentes en efluentes de sistemas de digestión anaerobia en forma de estruvita. PhD thesis in Industrial Engineering, University of Valladolid. School of Industrial Engineering DOI: 10.35376/10324/45332

Corona, F., Hidalgo, D., Martín-Marroquín, J. M., & Meers, E. (2021). Study of pig manure digestate pre-treatment for subsequent valorisation by struvite. *Environmental Science and Pollution Research,* 28(19), 24731-24743. https://doi.org/10.1007/s11356-020-10918-6

Le Corre KS, Valsami-Jones E, Hobbs P, Parsons SA (2009) Phosphorus recovery from wastewater by struvite crystallization: a review. *Crit Rev Environ Sci Technol* 39(6):433–477. https://doi.org/10.1080/10643380701640573

Martínez-Dalmau, J., Berbel, J., & Ordóñez-Fernández, R. (2021). Nitrogen fertilization. A review of the risks associated with the inefficiency of its use and policy responses. *Sustainability*, 13(10), 5625.

Némethy, A. (2016). Analyzing the process of struvite recovery with life cycle assessment. MSc Thesis from Swedish University of Agricultural Sciences, Department of Energy and Technology. Available at https://stud.epsilon.slu.se/9562/1/nemethy_a_160922.pdf (accessed Dec 2023).

Sampat, A. M., Martin-Hernandez, E., Martín, M., & Zavala, V. M. (2018). Technologies and logistics for phosphorus recovery from livestock waste. *Clean Technologies and Environmental Policy*, 20, 1563-1579.

Sánchez, E., Rodríguez, B., Heredero, R., García Peñuela, R., Polo, A., García, M., ... & Sanz, R. (2011). Experiencias para la recuperación del fósforo de las aguas residuales en forma de estruvita en Canal de Isabel II. Available at https://www.madrid.org/bvirtual/BVCM018018.pdf

Sustainability. P. (2020). SimaPro Database Manual Methods Library. Pré Sustainability: Amersfoort. The Netherlands

3.1.6 LL#55: Manure processing through separation and reverse osmosis (WUR)

Longlist #55 title: Manure processing and replacing mineral fertilisers in the Achterhoek region

Inge Regelink[1], Jan Peter Lesschen[1], Yun-Feng Duan[1]

[1] Wageningen Environmental Research, Wageningen University and Research, Wageningen, The Netherlands

3.1.6.1 Introduction

In The Netherlands, the amount of manure that is being produced by livestock farming exceeds the amount of manure that can be used within the application rate limits. This has led to the development of manure treatment installations in regions with intensive livestock farming. Manure treatment can be preceded by anaerobic digestion of the manure. Anaerobic digestion is an effective approach to prevent methane emissions from manure storages, to produce renewable energy, reduce smell and increase the proportion of mineral nitrogen (N) available for plants. In the Netherlands, application of digestate from co-digested pig manure has to comply with application limits for phosphorus (P) and 'N from animal manure'. Application rate limits for P are differentiated according to soil P status and land use. The use of N from animal manure is restricted at 170 kg N/ha, or 230-250 kg in case of derogation, as set by the EU Nitrates Directive. On top of that, farmers can apply additional synthetic N until the N application rate limit is reached. The N application rate limit applies to the amount of effective N which is set at 100% for synthetic N fertilisers and varies between 40 and 80% for N from various types of animal manure.

Digestate can be used as an organic fertiliser without further treatment, but its transport from regions with a surplus of manure to regions with a demand for organic fertilisers is costly. As an alternative, digestate can be separated into various biobased fertiliser products with the goal to produce end-products that can be used in the vicinity of the anaerobic digestion plant, which reduces transport related emissions.

In the study region, the Achterhoek in the Netherlands, there is currently a demand for synthetic N fertiliser, due to intensive agricultural plant production. The European Commission is currently developing RENURE (Recovered Nitrogen from manURE) criteria for when recovered and recycled manure N can be allowed to replace synthetic N over and above the 'N from manure' limits of the EU Nitrates Directive. A RENURE fertiliser shall have a NH_4/TN ratio of at least 90% or a TOC/TN ratio of <3.0. (Huygens et al., 2020). Co-digestated pig manure can be processed into a RENURE fertiliser and hence in principle replace synthetic N. The European Commission has not yet implemented criteria for RENURE fertilisers but a few manure processing plants, including the plant of this study, have received a temporary exemption in advance of a definite implementation of the RENURE criteria.

The investigated manure treatment plant is processing digestate into a solid fraction, a reverse osmosis (RO) concentrate, purified water, and a residual slurry. The RO concentrate complies with RENURE criteria and is used on local fields replacing synthetic N fertiliser. Detailed descriptions of the treatment process and mass balances are published by Van Puffelen et al. (2022) and Brienza et al. (2022). The solid fraction is rich in P and is either exported to Germany or further processed into a soil improver or a peat replacer for use in potting soil. The production of a peat replacer includes an additional treatment step in which the solid fraction is diluted with water and thereafter separated by means of a screw press to recover the coarse organic material. This leads to a significant reduction in salt level, which is a prerequisite for use as potting soil, but also creates an additional side stream.

Here, environmental effects of digestate treatment and use of end products are evaluated as compared to a reference situation in which digestate is transported over long distances (250 km) to regions with a demand for animal manure.

3.1.6.2 Material & method

3.1.6.2.1 Goal & scope

The goal of this Life Cycle Assessment (LCA) study was to estimate the environmental impacts of implementing a digestate separation process producing a RENURE fertiliser and a peat replacer, as an alternative to the direct use of digestate in regions with arable farming at a distance of 250 km of the treatment plant.

To do this, three different scenarios were compared:

- Sc_1: Raw digestate is transported over a distance of 250 km to regions in Germany with a demand for organic fertiliser (to be applied under the limit of 170 kg N/ha for animal manure). This is the reference / baseline scenario.
- Sc_2: Raw digestate is processed into (i) a solid organic fertiliser (transported to Germany over 300 km), (ii) RO concentrate (a RENURE fertiliser), (iii) a residual organic fertiliser (transported to regions with arable farming within the Netherlands for direct use) and (iv) purified water. RO concentrate is used as an alternative for synthetic N and applied on grassland and arable land within 25 km distance from the plant. Mass balances and technical details are published in Brienza et al. (2022), Van Puffelen et al. (2022) and Regelink et al. (2021).
- Sc_3: As Sc-2 but with an additional treatment step for the solid fraction of digestate to produce a P-fertiliser and a low-P soil improver. The soil improver is used to replace peat in substrate or potting soil. Mass balances and technical details are published in Brienza et al. (2022) and Regelink et al. (2019).

We selected a functional unit (FU) of *handling of 1,000 kg of digestate leaving the anaerobic digester.* A graphic representation of system boundaries and essential processes can be found in Figure 19. The AD plant and the production of biogas are beyond the boundary of the LCA. The LCA covers emissions associated with the separation of digestate, transport of end products from the plant to the field and emissions during and after application of the fertilisers.

3.1.6.2.2 Inventory

Mass balances, fertiliser composition and energy consumption were taken from monitoring of a full scale digestate treatment plant, Groot Zevert Vergisting (GZV) located in The Netherlands, that participated as a demonstration plant in the H2020 SYSTEMIC project. Additionally, the inventory includes data collected from the GABI professional database, ANIMO modelling, and literature data. ANIMO is a process-based model that simulates the transport of nutrients to groundwater and surface water systems and the emission of greenhouse gasses for a wide range of soil types, land management practices and hydrological conditions (Groenendijk et al., 2014; Groenendijk et al., 2005). Emission data from the different sources were combined in MS Excel. The impacts of each system were determined using Environmental Footprint methodology (EC, 2021).

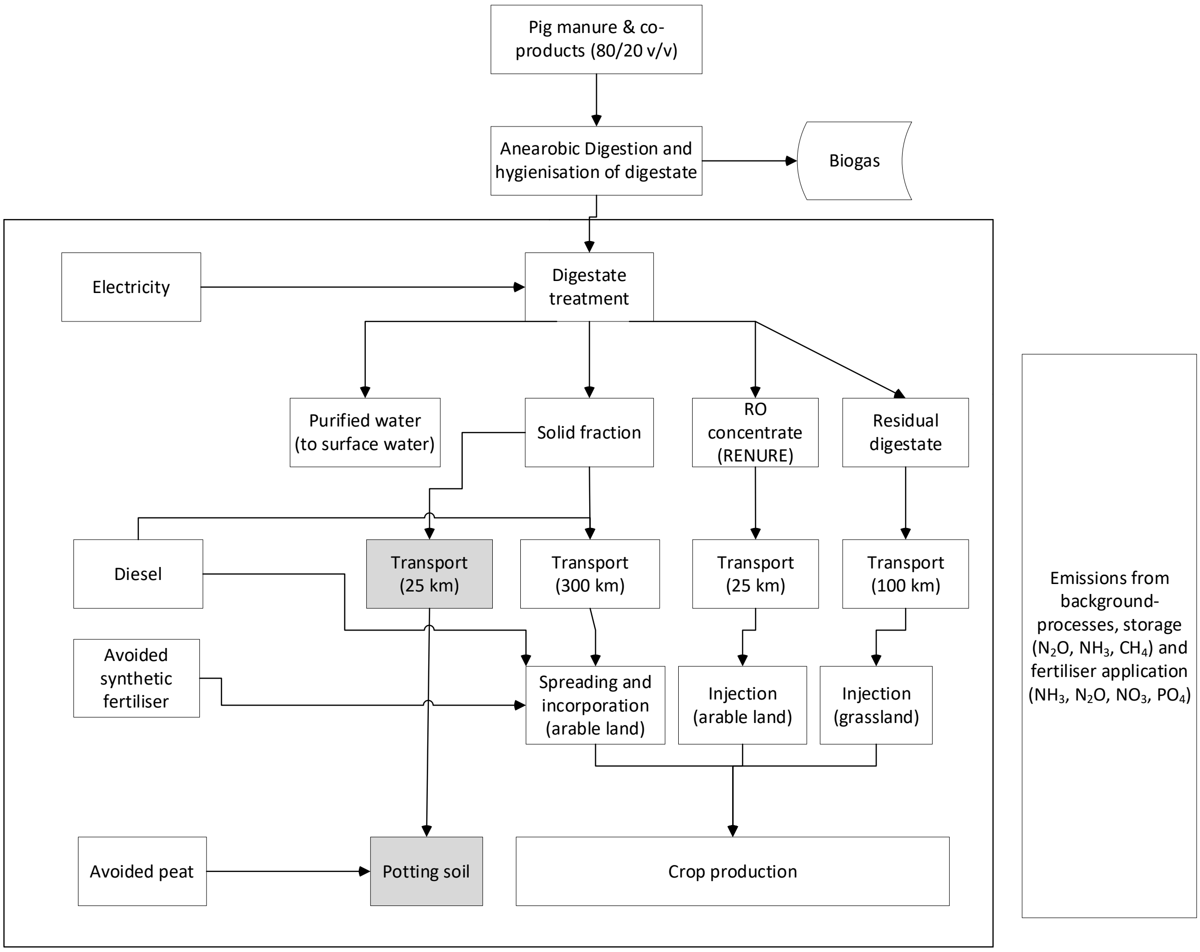


Figure 19. LL# 55: System boundaries

Structure of the analysed system for Sc_2 and Sc_3. White boxes indicate main activities associated with the separation of digestate and application of biobased fertilisers (Sc_2). Grey boxes indicate the additional treatment steps to upgrade the solid fraction to a peat replacer (Sc_3).

Digestate processing and transport

The studied system starts with the reference flow, i.e., one tonne of digestate obtained through co-digestion of pig slurry with residues from the agro-industry. On a volumetric basis, the feedstock consists for 80% of pig slurry and 20% of co-products, however, in terms of dry matter, pig slurry contributes 25% and co-products 75%.

Mass balances and transport distances in the three scenarios were as follows:

- In Sc_1, digestate is transported to Germany (250 km) and used on arable land.
- In Sc_2, digestate is processed by means of a decanter centrifuge, micro-filtration unit and reverse osmosis installation producing 0.16 tonne of solid fraction, 0.35 tonne of residual digestate fraction (liquid organic NPK fertiliser) and 0.33 tonne of RO concentrate (RENURE fertiliser), respectively. The residual digestate fraction consists of concentrate of the microfiltration unit. Additionally, 0.18 tonne of purified water is being produced per tonne of digestate. This water is discharged into surface water.

- In Sc_3, the treatment is extended with a washing step to lower the salt content of the solid fraction and to separately recover the course organic fibres that are suitable for use as potting soil ingredient replacing fossil peat. One tonne of solid fraction is mixed with 0.5 tonne of water producing 0.8 tonne of peat replacer and a 0.7 tonne residual fraction (including the recovered phosphate). The liquid fraction of the digestate is treated as in Sc_2. The overall mass balance per tonne of digestate amounts to: 0.12 tonnes of peat replacer, 0.46 tonnes of residual digestate fraction (liquid organic NPK fertiliser), 0.33 tonne of RO concentrate (RENURE fertiliser) and 0.18 tonne of purified water (discharged onto surface water).

Electricity consumption for digestate processing was based on monitoring data from the GZV plant (Brienza et al., 2022) and an emission factor of the EU energy mix was used (Gabi professional database, 0.396 kg CO_2/kWh). Electricity consumption in Sc_1, Sc_2 and Sc_3 amounted to 0, 21 and 24 kWh per tonne of digestate.
Emissions of CH_4 during processing, storage, and application of digestate and products thereof is assumed to be negligible as CH_4 from the digestate is already emitted in the digester and post-digester. Emissions of NH_3 during processing are assumed to be negligible as processing is performed in air-tight systems except for the decanters which are equipped with air treatment installations. For the solid fraction, N_2O emission is calculated as 0.0025 kg N-N_2O/kg N assuming 3 months of storage (Melse and Groenestein, 2016). For the liquid fertilising products, no emissions of N_2O occur during storage as these remain anaerobic (Melse and Groenestein, 2016).

The use of chemical additives (e.g. sulphuric acid, polymer, anti-foaming agents, cleaning agents) and consumables (e.g. RO membranes) were not included due to a lack of reliable emission factors in the GABI professional database.

Emissions associated to transport of end products are calculated assuming transport by a 24.7 tonne Euro 5 truck with a utilisation factor of 50% (i.e. truck being empty on the way back). Transport distances are included in Figure 19.

Field application & crop production

Environmental emissions following field application of the biobased fertiliser products, such as atmospheric emissions of NH_3 and N_2O, as well as leaching of NO_3 and PO_4, were calculated using the ANIMO model (Groenendijk et al., 2014; Groenendijk et al., 2005). The modelling was based on the baseline scenarios ATC-Arable and ATC-Dairy (Nutri2Cycle Deliverable 1.5 report, Duan et al., 2020), which simulate rotation of arable crops and perennial grassland production, respectively, under Dutch conditions. All organic fertilisers were applied on arable land (ATC-Arable) in compliance with the limit for N from animal manure (170 kg N/ha) and P (70 kg P_2O_5/ha), whereas RO concentrate was applied on grassland (ATC-Dairy) replacing synthetic N. The biobased fertiliser products were parameterised according to measurements of their compositions. Emission factors for ammonia volatilization were estimated using the ALFAM2 model (Hafner et al., 2018), considering the composition of the fertilisers, application method, as well as average climatic conditions at the time of application. Raw digestate, RO concentrate and the residual digestate fraction were assumed to be injected into the soil, whereas the solid fraction was assumed to be spread onto the soil and thereafter incorporated. Both synthetic and organic P fertilisers were assumed to have a relative P use efficiency (PUE) of 100%, implying that PO_4 leaching was equal between biobased and synthetic P fertilisers. To create realistic fertilisation scenarios, fertilisation with biobased fertilisers was complemented with synthetic fertilisers until the application rate limits for N and P were met. In addition, a reference scenario was included using only synthetic NP fertiliser which was used as an indicator for the baseline emissions. The simulation was run for 20 years, and emissions were estimated as the 20-year average.

Model estimated crop production was nearly similar in all treatments as N, P application rates were similar. Small differences in crop production between scenarios were not accounted for in the LCA. Diesel consumption for application of fertilisers was taken from VLM (2015). Diesel consumption amounted to 20 L diesel/ha for spreading and incorporation of the solid fraction, 13 L diesel/ha for injection of the liquid fertilisers and 1.2 L diesel/ha for spreading of CAN (granular product).

Avoided emissions from the avoided production of synthetic N fertiliser were calculating using emission factors for production of CAN (calcium ammonium nitrate, GABI professional database). The amount of N being avoided as calculated using a NFRV (nitrogen fertiliser replacement value) of 80%, 55% and 100% for digestate/residual digestate fraction, the solid fraction and the RO concentrate, respectively.

Purified water after RO containing 0.2 mg N/L and 0.01 mg P/L was discharged onto surface water but it's contribution to overall emissions of NO_3^- and PO_4^{-2} was negligibly low and hence not shown in the results.

Replacement of fossil peat

Sc_3 includes the production of a peat replacer from digestate. It was assumed that 1 m^3 of peat replacer substituted 1 m^3 of fossil peat thereby avoiding transport of peat to The Netherlands (500 km) and avoiding oxidation of fossil peat. It was assumed that all organic carbon contained in fossil peat is oxidized to CO_2, assuming and TOC/OM ratio of 45%. Avoided methane emissions due to excavation of peat were not included.

Further details of the input data for the LCA can be found in Table 20.

Table 20. LL#55 LCA inventory data: expressed per ton of digestate (functional unit = 1 ton of digestate)

		Sc_1 (reference) Export of digestate	Sc_2 Separation of digestate into a solid fraction and mineral concentrate	Sc_3 Sc_2 plus use of the solid fraction in potting soil replacing peat
Mass balance				
Digestate	(ton)	1		
Solid fraction	(ton)		0.16	116 (Peat replacer)
Sludge from micro-filtration	(ton)		0.35	461
Mineral concentrate	(ton)		0.33	330
Clean water	(ton)		0.17	175
Energy and avoided resources				
Electricity processing	(kWh)	0	21	23
Transport	(ton*km)	250	107	83
Diesel - field application	(L)	0.87	1.57	0.79
Avoided N fertilizer	(kg N)	5.9	6.0	6.1
Avoided fossil peat	(ton)	0	0	35

3.1.6.3 Results

3.1.6.3.1 Impact assessment

The impacts calculated for the scenarios with and without separation of digestate are presented in Figure 20. Environmental impacts of Sc_1 and Sc_2 were very similar for all considered impact categories.

In Sc_2, digestate is separated into a solid fraction, RENURE fertiliser, purified water, and a residual digestate fraction. The separation of digestate into three types of fertilising products led to a reduction in CO_2-eq emissions related to transport but to an increase in emissions related to the consumption of electricity. In addition, CO_2 emissions related to the injection or incorporation of the biobased fertilisers increased compared to injection of digestate. Avoided CO_2-eq emissions related to avoided production of synthetic N fertiliser remained similar in Sc_1 and Sc_2 as the total amount of effective N did not change. Hence, the production of a RENURE fertiliser has no benefits in terms of avoided N fertiliser as raw digestate also avoids use of synthetic N fertiliser. In Sc_3, in which the solid fraction was further processed into a peat replacer, a reduction in CO_2-eq emissions is predicted. This benefit is due to the avoided release of CO_2-eq from oxidation of fossil peat.

Terrestrial acidification is nearly similar in Sc_1 and Sc_2 and its impact is mostly due to emissions of NH_3 from the biobased fertilisers. In Sc_2, the RENURE fertiliser is supposed to have a relatively low emission factor for NH_3 of 13% of TAN but this is outweighed by the relatively high emission factors for NH_3 for the solid fraction (50% of TAN) and the residual digestate fraction (20% of TAN). Consequently, the overall NH_3 emissions remain similar to Sc_1 in which raw digestate was used directly as a fertiliser. A slight decrease is observed in Sc_3 where the solid fraction is not applied on land.

Marine eutrophication, which includes emissions of NO_3^- to fresh water, follows an opposite trend showing a small reduction in emissions of NO_3-eq for Sc_2 and Sc_3. The decreased leaching of NO_3^- is here explained by the increased atmospheric losses of NH_3 and hence no indication of an increased N use efficiency of the processed digestate.

Differences in *terrestrial eutrophication*, which comprises leaching of PO_4^{-2} to fresh water but not NO_3^- are negligibly small and solely related to the increased consumption of electricity in Sc_2 and Sc_3. Emissions from the field were expressed as a difference compared to emissions when using synthetic fertiliser only. There is no change in PO_4^{-2} leaching from agricultural soil as the applied model considers biobased fertilisers as effective as synthetic P fertiliser.

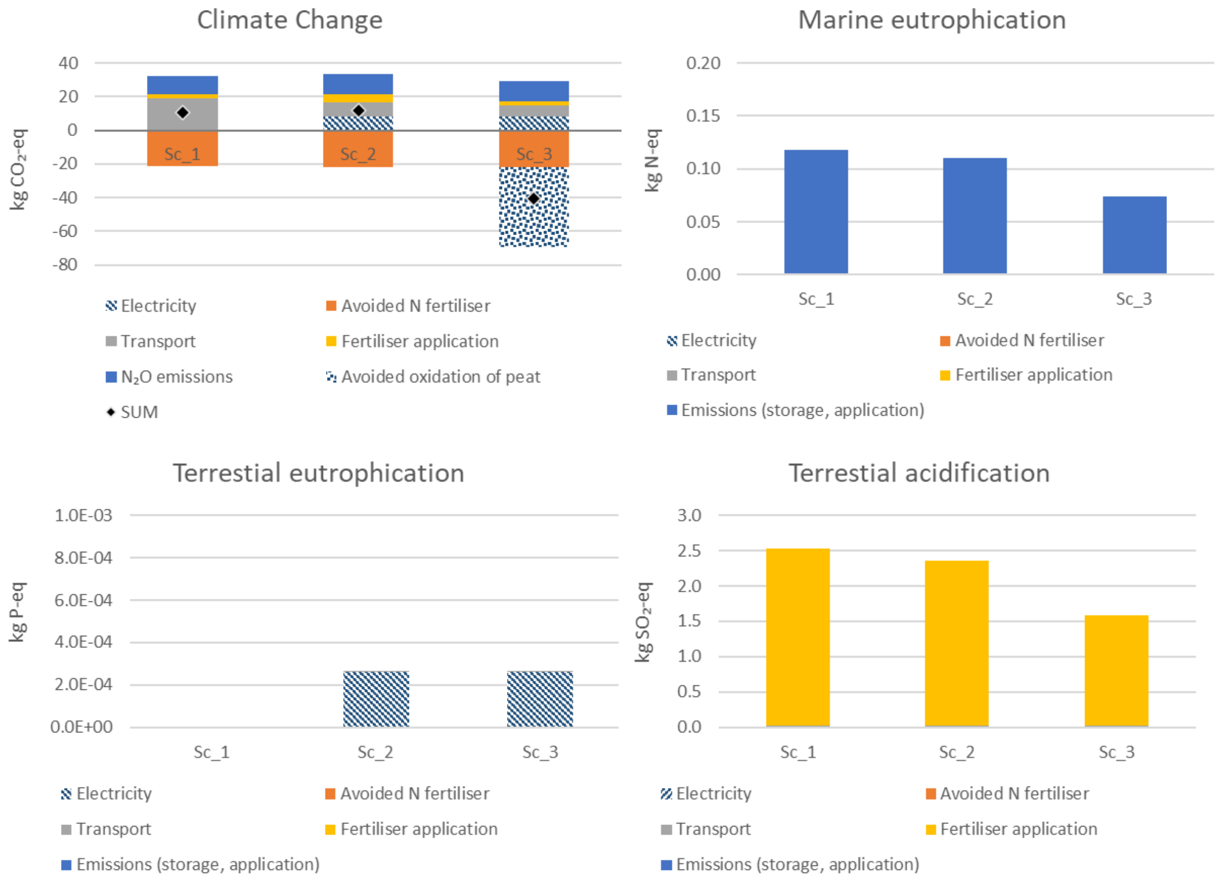


Figure 20. LL# 55: eLCA results

Environmental impact per functional unit of 1,000 kg digestate, with using raw digestate (Sc_1), separation of digestate (Sc_2) and separation plus production of a peat replacer (Sc_3) for four impact categories.

3.1.6.4 Interpretation

Separation of digestate producing a RENURE fertiliser, to be used locally as an alternative for synthetic N fertiliser, has little or no environmental benefits over transport of untreated digestate over 250 km and subsequent field application. In terms of CO_2-eq emissions, digestate treatment becomes beneficial only when the alternative is to transport the digestate over distances of more than 250 km. In the current calculations, the emission factors for the average EU energy mix have been used. If electricity from renewable sources would be used, the GHG savings of digestate treatment would be higher. As energy for heavy transport is more difficult to convert to renewable energy, manure treatment using renewable energy may be a better option. A similar conclusion was drawn by Duan et al. (2020) who evaluated different high-tech processing techniques for digestate of pig slurry but found little environmental advantages as compared to direct field application of digestate.

A RENURE fertilising product is an alternative for synthetic N and hence avoids production of synthetic N through the Haber-Bosch process. This is often used as an argument to promote production of RENURE fertilisers from manure or digestate. Raw manure and digestate must comply with the application rate limit for N from animal manure, but still deliver N to the soil and therefore avoid using a similar amount of synthetic N fertiliser. Following that reasoning, production of a RENURE fertiliser does not offer a benefit in terms of 'avoided synthetic N' as compared to the use of raw digestate. However, in regions with a surplus of manure and where the application of N from manure is limited by the Nitrates Directive, RENURE fertilisers offer an advantage as compared to long-distance (>250 km) transport of manure. The here evaluated process using MF and RO membranes, however, is relatively expansive due to the high investment and operational costs. A cheaper and simpler solution is to use liquid fraction of animal manure or digestate as a N fertiliser. A recent study showed that nearly 60% of the liquid fractions from digestate meet draft criteria for RENURE fertilisers (NH_4/TN>90% or TC/TN<3.0) (Reuland et al., 2021). Further purification of liquid fractions with membranes increases the NH_4/TN ratio of the fertiliser but it is questionable whether the additional efforts in terms of energy and chemicals outweigh the benefits of a slightly improved NH_4/TN ratio of the N fertiliser.

In term of emissions of N to groundwater and the atmosphere, it is important to consider not only the RENURE fertiliser but also all other fertilisers that are being produced at the treatment plant. For example, this particular installation produced a residual digestate fraction with a NH_4/TN ratio of 50% which therefore contributed to nitrate leaching and a solid fraction that contributed to ammonia emissions. As a consequence, net emissions of NH_3 and NO_3^- hardly decreased as compared to the reference scenario. However, digestate treatment has an impact on the spatial distribution of manure and derived products and its associated emissions. This aspect is not considered in LCAs, which typically calculates only the sum of the emissions regardless of their spatial distribution. It is advised to further assess the benefits in relation to local environmental targets for ground and surface waters.

3.1.6.5 Conclusion

This study compared scenarios for treatment of digestate against no treatment in an area with a surplus of animal manure in The Netherlands in terms of their environmental performance. The results suggest that digestate treatment producing a RENURE fertiliser has no benefits in terms of CO_2-eq emissions (*climate change potential*) and gives only a 10% reduction in emissions of SO_2-eq (*terrestrial acidification potential*) which is mostly emitted as NH_3. Other considered impact categories, *terrestrial eutrophication* and *marine eutrophication*, remain unaffected.

The more advanced digestate treatment scenario in which the solid fraction is further upgraded towards a peat replacer shows more pronounced environmental benefits. In this scenario, the impact on climate change is being reduced mostly due the avoided oxidation of fossil peat.

This study shows that policy makers shall be careful when considering stimulation of manure- or digestate treatment as environmental benefits may be absent or lower than anticipated.

3.1.6.6 Dashboard indicators

LL# 55 Pig manure processing and replacing mineral fertilisers (*: Qualitative DBI assessment: **+** *improvement*, **o** *no change*, **-** *deterioration)*

Table 21. LL# 55: Comparison of dashboard indicator and life cycle assessment results

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication *	Comment	Quantitative estimate per FU of 1000 kg digestate	LCA indication *	Re-evaluation of DBI irt. LCA results
Use of Primary Resources	**Rock Phosphate** (Reduction in mineral phosphorus consumption)	o		Phosphorous, in ground	o	No change as total amount of P application does not change among the scenarios
	Natural Gas (Reduction in natural gas consumption in mineral fertiliser production)	+	Less mineral N fertiliser is required, the process of making fertilisers is very energy intensive, using a lot of natural gas	Gas, natural, in ground	o	Locally N fertiliser and therefore natural gas can be saved, but elsewhere it will replace the digestate, so net savings are zero. The biogas produced can replace natural gas, but the anaerobic digester was outside the system boundary
	Oil (Reduction in oil consumption in agricultural machinery)	+	Less transport (diesel) is required for the export of manure	Oil, crude, in ground: Diesel burnt for transport Sc_2: -105 MJ (-41%) Sc_3: -171 MJ (-66%)	+ + +	Oil extraction can be reduced, due to less transport and diesel use Less transport and diesel use for exporting the digestate
	Electricity (Reduction in electricity consumption)	-	For the treatment process electricity is required	Electricity consumption Sc_2: 21 kWh (>100%) Sc_3: 24 kWh (>100%)	 - -	Increased electricity use for manure treatment

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment	DBI Indication *	Comment	Quantitative Dashboard Indicator based on LCA assessment: Quantitative estimate per FU of 1000 kg digestate	LCA indication *	Re-evaluation of DBI irt. LCA results
	Water (Reduction in water consumption)	o		Water scarcity:	o	Not evaluated, but no impact expected
	Soil quality (Improvement in soil quality)	o		Soil quality		Soil quality or land use change has not been included in the LCA
	Nutrients recovered (Nutrient recovered from agriculture and livestock systems)	+	Ammonia retained is converted to plant available N	Nitrogen recovery	o	For the RENURE product NH_3 emissions are lower, but thick fraction has higher emissions, net emissions do not change much
	Renewable biomass	o		Renewable biomass	+	For Sc_3 the peat replacer prevents the extraction of fossil peat
	Others? Please specify					
Emissions to the environ-ment	**Ammonia (air emission)** (Reduction in NH_3 emissions)	-	The mineral concentrate might have some risk of higher ammonia emissions, however these were not measured in the experiment	Ammonia emission to air Sc_2: -0.1 kg NH_3 (-7%) Sc_3: -0.6 kg NH_3 (-42%)	 + +	Ammonia emissions are slightly lower due to digestate treatment. NH_3 emission values are compared to fertilisation with only CAN.

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication *	Comment	Quantitative estimate per FU of 1000 kg digestate	LCA indication *	Re-evaluation of DBI irt. LCA results
	Dinitrogen monoxide (air emission) (Reduction in N_2O emissions)	o	Probably no effect as total N input remains the same	N_2O emission to air: Sc_2: 0.007 kg N_2O (20%) Sc_3: 0.005 kg N_2O (14%)	 - -	N_2O emissions are slightly higher in Sc_2 and Sc_3, due to more N_2O emissions from the solid fraction. Difference is small but >10%. N_2O emission values are compared to fertilisation with only CAN
	Methane (air emission) (Reduction in CH_4 emissions)	o		Methane emission:	o	No change, reduction of CH_4 emission as result of anaerobic digestion is outside the system boundary
	Nitrates (water emission) (Reduction in NO_3 emissions)	o	Nitrate leaching is being measured, but no results yet, remain the same, there will not be an increase in nitrate leaching	Nitrate emission to water Sc_2: 0.4 kg NO_3 (-17%) Sc_3: 2.2 kg NO_3 (-91%)	 - -	Nitrate leaching is slightly higher for Sc_2 and higher for Sc_3 as the solid fraction, which has lower leaching is not used. NO_3 emission values are compared to fertilisation with only CAN
	Phosphorus (water emission) (Reduction of P emissions)	o		P emission to water:	o	No change in PO_4 leaching as the model considers biobased fertilisers as effective as synthetic P fertiliser
	Particulate matter (Reduction of particulate matter formation)	o		Particulates emission to air		Not included in the LCA
	Others? Please specify					
Resilience to	**Carbon footprint** (Reduction of carbon footprint)	+	Overall reduction in N_2O, CH_4 etc. but also	Climate change: Sc_2: 1.0 kg CO_2 eq. (9,3%)	 o +	For Sc_3 with the peat replacer the footprint is much lower, because of

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment	DBI Indication *	Comment	Quantitative Dashboard Indicator based on LCA assessment Quantitative estimate per FU of 1000 kg digestate	LCA indication *	Re-evaluation of DBI irt. LCA results
climate change			increased energy demand	Sc_3: -51.3 kg CO_2 eq. (>-100%)		prevented fossil peat emissions, for Sc_2 the footprint is similar to the reference.
	Effective SOM (Effective Soil Organic Matter Improvement)	**o**		Carbon | Emission from soil	**o**	Not assessed
	Renewable energy production (Renewable energy produced from biomass)	**+**	The processing installation is linked to a digester, which produces bioenergy	Renewable energy production	**o**	The digester produces biogas, but this was outside the system boundary
	Others? Please specify					

3.1.6.6.1 Comparison of LCA and DBI results

The comparison shows that part of the results of the LCA and DBI were in line with each other, but for others differences occurred, which were mainly due to the different system boundary for the LCA analysis, compared to what was used for the establishment of the DBI. In the LCA the functional unit was one ton of digestate, so the anaerobic digestion process was beyond the system boundary. For the DBI the anaerobic digestion was included. Therefore, the use of natural gas and the production of renewable energy were positive in the DBI but had no effect in the LCA.

For N_2O emissions no effect was expected according to the DBI, whereas the LCA showed higher emissions due to the use of the solid fraction, which has a higher N_2O emissions compared to the digestate. On the other hand, the NH_3 emissions were lower according to the LCA results, whereas in the DBI an increase was expected. This was also due to the system boundary, as for the DBI the digestate was compared to undigested manure, and as digestion increases the pH and NH4 content, a higher NH3 emission was expected. In the LCA the anaerobic digestion was not included and only the comparison of treated versus untreated digestate was done. These results showed a decrease in NH_3 emissions for Scen_2 and Scen_3.

3.1.6.7 References

Brienza, C., J. van Puffelen, I. Regelink, H. Dedeyne, A. Giordano, M. Schepis, U. Bauermeister, T. Meier and I. Sigurnjak (2022) Technical performance and mass- and energy balances of five large-scale anearobic digesters applying nutrient recovery and reuse - A report within the H2020 project SYSTEMIC. Wageningen Environmental Research, Wageningen, the Netherlands. https://doi.org/10.18174/572613

Duan, Y.-F., Bruun, S., Jensen, L.S., van Gerven, L., Hendriks, C., Stokkermans, L., Groenendijk, P., Lesschen, J.P., 2020. D1.5 Mapping and characterization of CNP flows and their stoichiometry in main farming systems in Europe. Nutri2Cycle EU No. 773682. Available from www.nutri2cycle.eu

Duan, N., Khoshnevisan,B.,Lin, C., 2020. Life cycle assessment of anaerobic digestion of pig manure coupled with different digestate treatment technologies, *Environment International*, 137, https://doi.org/10.1016/j.envint.2020.105522

EC – European Commission, 2021. Commission Recommendation on the use of the Environmental Footprint methods to measure and communicate the life cycle environmental performance of products and organisations. https://ec.europa.eu/environment/publications/recommendation-use-environmental-footprint-methods_en

Groenendijk, P., Heinen, M., Klammler, G., Fank, J., Kupfersberger, H., Pisinaras, V., et al. (2014). Performance assessment of nitrate leaching models for highly vulnerable soils used in low-input farming based on lysimeter data. Science of The Total Environment, 499, 463-480. https://doi.org/10.1016/j.scitotenv.2014.07.002

Groenendijk, P., Renaud, L. V., & Roelsma, J. (2005). Prediction of nitrogen and phosphorus leaching to groundwater and surface waters: Process descriptions of the Animo 4.0 model (983). Retrieved from https://library.wur.nl/WebQuery/wurpubs/340937

Melse, R.W. and C.M. Groenestein, 2016. Emissiefactoren mestbewerking. Inschatting van emissiefactoren van ammoniak, methaan en lachgas uit mestbewerking. Wageningen Livestock Research, Livestock Research Rapport 962. https://edepot.wur.nl/386801

Hafner, S.D., Pacholski, A., Bittman, S., Burchill, W., Bussink, W., Chantigny, M., Carozzi, M., Génermont, S., Häni, C., Hansen, M.N., Huijsmans, J., Hunt, D., Kupper, T., Lanigan, G., Loubet, B., Misselbrook, T., Meisinger, J.J., Neftel, A., Nyord, T., Pedersen, S. V., Sintermann, J., Thompson, R.B., Vermeulen, B., Vestergaard, A. V., Voylokov, P., Williams, J.R., Sommer, S.G., 2018. The ALFAM2 database on ammonia emission from field-applied manure: Description and illustrative analysis. Agric. For. Meteorol. 258, 66–79. https://doi.org/10.1016/j.agrformet.2017.11.027

Hou, Y., Velthof, G.L., Oenema, O., 2015. Mitigation of ammonia, nitrous oxide and methane emissions from manure management chains: A meta-analysis and integrated assessment. Glob. Chang. Biol. 21, 1293–1312. https://doi.org/10.1111/gcb.12767

Huygens D, Orveillon G, Lugato E, Tavazzi S, Comero S, Jones A, Gawlik B, Saveyn HGM, *Technical proposals for the safe use of processed manure above the threshold established for Nitrate Vulnerable Zones by the Nitrates Directive (91/676/EEC)*, EUR 30363 EN, Publications Office of the European Union, Luxembourg, 2020, ISBN 978-92-76-21539-4, https://doi.org/10.2760/373351 , JRC121636.

Regelink, I.C., P. Ehlert, G.Smit, S. Everlo, A. Prinsen, O. Schoumans. 2019. Phosphorus recovery from co-digested animal manure. Development of the RePeat process. Wageningen Environmental report 2949. Wageningen. https://edepot.wur.nl/476731

Regelink, I.C., van Puffelen, J.L., Ehlert, P.A.I., Schoumans, O.F. 2021. Evaluatie van verwerkingsinstallaties voor mest en co-vergiste mest. Wageningen Environmental Research Rapport 3120. Wageningen. https://edepot.wur.nl/554452

Reuland, G., Sigurnjak, I., Dekker, H., Michels, E., Meers, E. 2021. The Potential of Digestate and the Liquid Fraction of Digestate as Chemical Fertiliser Substitutes under the RENURE Criteria. *Agronomy* 11(7).

Van Puffelen, Jasper L., et al. "Performance of a full-scale processing cascade that separates agricultural digestate and its nutrients for agronomic reuse." *Separation and Purification Technology* (2022): 121501. https://doi.org/10.1016/j.seppur.2022.121501

3.1.7 LL#65: Struvite recovery from wastewater sludge (UGENT+UCPH)

Longlist #65 title: Struvite as a substitute for synthetic P fertiliser

Rahul Ravi[1,2], Miriam Beyers[2,1], Sander Bruun[2], Erik Meers[1]

[1] Department of Green Chemistry and Technology, Faculty of Bioscience Engineering, Ghent University, Belgium

[2] Department of Plant and Environmental Sciences, Faculty of Science, University of Copenhagen, Denmark

This chapter builds on the published scientific paper by Ravi et al. (2022). Life cycle assessment of struvite recovery and wastewater sludge end-use: A Flemish illustration. Resources, Conservation and Recycling 182: 106325. https://doi.org/10.1016/j.resconrec.2022.106325

3.1.7.1 Introduction

Phosphate rock (PR) has been designated as a Critical Raw Material in the European Union (EU). This has led to increased emphasis on alternative P recovery (APR) from secondary streams like wastewater sludge (WWS). One of the most frequently proposed APR techniques in the EU is struvite (Magnesium ammonium phosphate) crystallisation from wastewater at wastewater treatment plants (WWTPs).

Struvite deposition is a natural process taking place in WWTPs and is known to clog pipes via encrustation and scaling, resulting in high operating and maintenance costs (De Boer et al., 2018; Doyle and Parsons, 2002). To address these concerns, WWTPs are increasingly implementing struvite recovery. The P recovery in the form of struvite is roughly 10−30% of the influent P (Egle et al., 2016). It is recovered from either (i) digested WWS or (ii) the centrate formed after WWS dewatering (Huygens et al., 2019). While the solubility of the recovered struvite can be lower than that of most mineral P fertilisers, it has shown to be an efficient fertiliser in plant growth trials (Möller et al. 2018; Vaneeckhaute et al. 2016). Also, struvite recovered from WWTPs has significantly lower cadmium and other heavy metal concentrations than synthetic P fertilisers (Egle et al. 2016; Kataki et al. 2016). Although there have been some questions regarding its market potential, especially amongst fertiliser companies (De Boer et al. 2018), struvite has achieved a secondary 'product' status, provided it complies with the minimum nutrient content, maximum limit values for inorganic contaminants, and biological pathogens (EC Regulation No 2019/ 1009).

The technology analysed in this study is the Nutrient Recovery System (NuReSys®), a full-scale struvite recovery plant, installed at Aquafin WWTP in Leuven (capacity: 120,000 inhabitants; 36,000 m^3 wastewater inflow/ day). The struvite here is recovered from digested WWS, preceded by an Enhanced Biological Phosphorus Removal (EBPR) system. The NuReSys® concept has been explained by Marchi et al. (2020) and further information regarding the techno-economic aspects is available in Saerens et al. (2021) and Marchi et al. (2015).

The main objective of this study is to assess whether the implementation of struvite recovery causes a net benefit or burden to the environment.

3.1.7.2 Material & method

3.1.7.2.1 Goal & scope

The study compares the present-day impacts of the WWTP (a) versus the environmental impacts before struvite recovery was implemented (b). Figure 21 illustrates the boundaries of both systems.

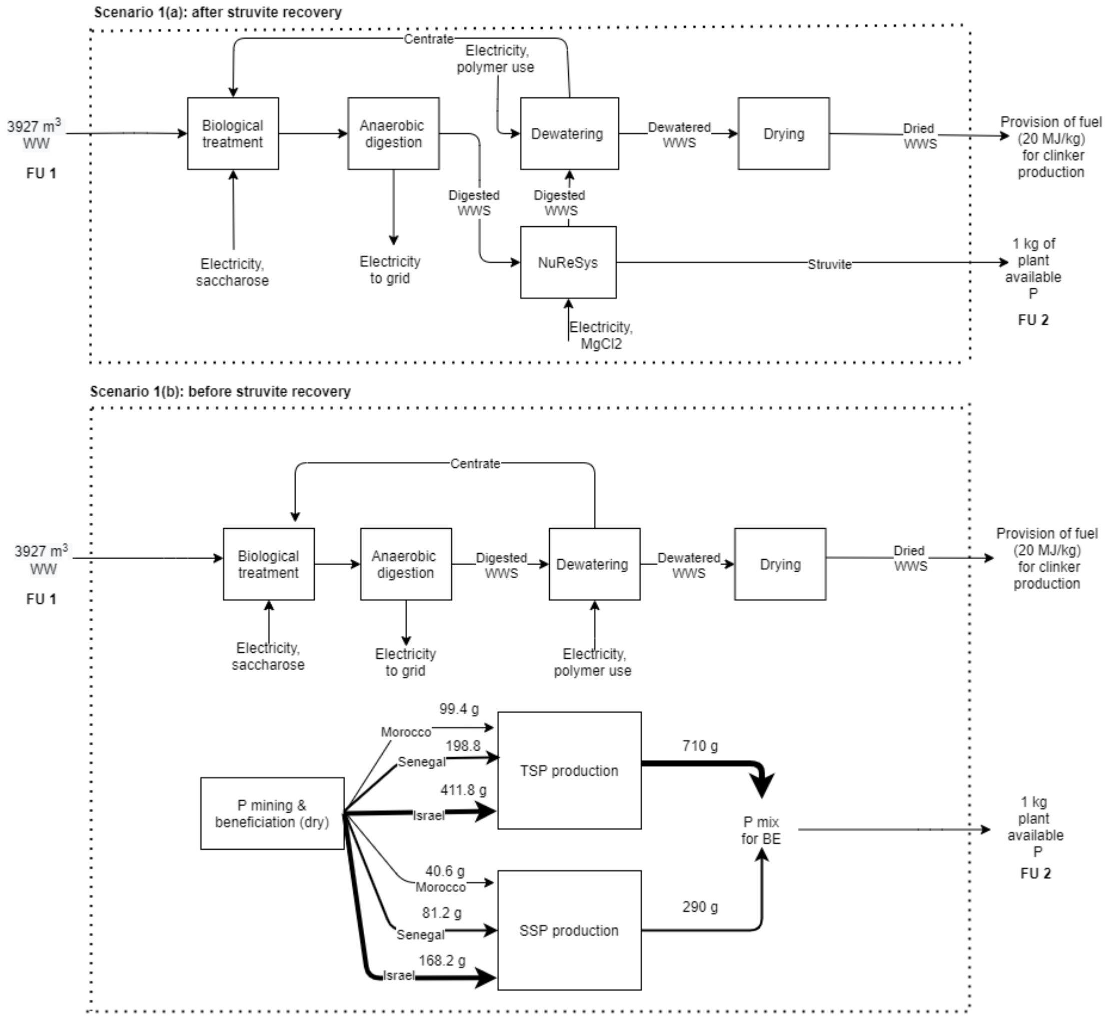


Figure 21. LL# 65: System boundaries

Systems comparing after (a) versus before (b) struvite recovery

In scenario (a), the system is already multi-functional, i.e. the influent is treated, and a 'useful' P product is generated. In (b), however, the system's normal function is to treat the influent. To ensure a fair comparison between (a) and (b), we expand the system by incorporating 1 kg of plant-available P in the form of synthetic fertilisers. To account for the two aspects, we choose a combined functional unit (FU) which considers a set amount of influent (3927 m3) entering the system and a set amount of P fertiliser leaving the system (1 kg). For synthetic P, only the fertilisers manufactured through the sulphuric acid route are considered (Triple superphosphate (TSP) and Single superphosphate (SSP)). The system boundary for TSP and SSP includes mining and beneficiation, transport, and processing of phosphate rock to the end-product.

According to the chosen attributional approach we expanded the system by using a market mix of P fertilisers. The estimated market share of marketable-phosphate rock in Germany is 58% from Israel, 28% from Senegal and 14% from Morocco and the market shares for TSP and SSP (fertilisers manufactured through the sulphuric acid route) are 71% and 29%, respectively (Kraus et al. 2019). We assume the same market share for Belgium. The life cycle inventory (LCI) for TSP, SSP, sulfuric acid production, and PR beneficiation process is based on Kraus et al. (2019), who updated the existing LCI in the ecoinvent database.

3.1.7.2.2 Inventory

To complete the inventory, a range of assumptions had to be made. One assumption is that all the P in the struvite is considered plant available. Generally, the P in struvite recovered from wastewater treatment plants is very available (Bogdan et al. 2021; Egle et al. 2016; Saerens, Geerts, and Weemaes 2021), and therefore this assumption is quite plausible.

The infrastructure component, i.e., building or equipment and the sewer network are assumed to be associated with minor impacts and have not been considered in the analysis. The system boundaries cover biological treatment, followed by WWS digestion and subsequent dewatering and drying of the WWS in (b), whereas in (a) the system also includes struvite recovery.

The foreground systems in the inventory were built from primary data available from Aquafin Inc (also see Table SI4 in Deliverable 2.6) and the background processes were modelled using the ecoinvent database (version 3.8) (Wernet et al. 2016). All analyses were performed using Activity Browser and Brightway2. An overview of key inventory data can be found in Table 22. The complete life cycle inventory is available in Ravi et al., 2022.

Table 22. LL# 65 LCA Inventory: key data

Stage	Exchange	Unit	*Scenario 1(a): post NuReSys®*		*Scenario 1(b): pre NuReSys®*	
			μ	σ	μ	σ
Biological treatment	WW influent	m^3	***3927***	***1398***	***3927***	***1398***
	Saccharose	Kg	1708	2152	1810	2152
	Electricity	kWh	1664	159	1670	10
Anaerobic digestion	Electricity to grid	kWh	-321.92		-321.92	
	Digested sludge	kg	1114.1	6.91	1017.21	6.92
NuReSys®	Digested sludge	kg	1011.83	6.92		
	Electricity usage	kWh	15.5			
	$MgCl_2$	kg	27.79			
	Struvite production (in terms of P)	kg	***1***			
Buffer tank	Digested sludge (After AD)	kg	1011.83	6.91	1017.21	6.92
	External sludge	kg	637.3		637.3	
Dewatering	WWS from buffer tank	kg	1649.13		1654.5	
	Electricity usage	kWh	333.49	10	340.14	10
	Polymer use	kg	44.31	32.66	54.84	32.66
P mix for BE	TSP production	kg			***0.71***	
	SSP production				***0.29***	

- μ refers to the mean and σ refers to the standard deviation
- *Values in bold indicate hybrid functional unit;
- Saccharose is used as a C-source for biological treatment. At Aquafin WWTP, the C-source comes burden free since it is a waste product from a confectionary factory in Turnhout. The transport to the WWTP has been considered.
- We used the production process for NaCl since there were no ecoinvent processes for $MgCl_2$ production. The rationale is explained below.

- Commonly used polymers at WWTPs include polyacrylamide and polyaluminium chloride. At Aquafin WWTP, polyacrylamide is used

3.1.7.2.3 Impact assessment

The impacts were quantified using Environmental Footprint methodology (EC 2021). The following impact categories are relevant for LCAs related to wastewater treatment (Niero et al. 2014; Renou et al. 2008) and were considered in the assessment:

- Climate change potential - in kg CO_2 equivalent (eq)
- Fossil depletion potential -in MJ eq
- Human toxicity potential - in CTUh eq
- Freshwater ecotoxicity potential - in CTUe eq
- Freshwater eutrophication potential - in kg P eq
- Marine eutrophication potential - in mol N^+ eq

3.1.7.3 Results

3.1.7.3.1 Impact assessment at midpoint

The results indicate that Scenario (a) i.e., after-struvite recovery, has slightly lower impacts than Scenario (b) i.e. before-struvite recovery for all impact categories (Figure 22). For *climate change potential*, a deeper analysis of the individual contributions revealed a negligible difference in impacts from biological treatment and anaerobic digestion between the scenarios. Leaving these impacts aside, the major impact contribution in Scenario (a) is the struvite recovery step (NuReSys® and $MgCl_2$ usage), which contributes to around 10 kg CO_2-eq. This is roughly 8 times higher than the *climate change* impact corresponding to conventional P fertiliser imports (1.2 kg CO_2-eq) in Scenario (b). However, these impacts are offset by the polymer (polyacrylamide) use in Scenario (b) which is 21% higher (157 kg CO_2-eq), when compared to (a) (127 kg CO_2-eq). The upstream impact contribution (79.8%) from polyacrylamide manufacturing is due to the ammoxidation process used to produce acrylonitrile.

The results for the other impact categories follow a similar trend to *climate change potential*; i.e. the increased polymer use in (b) caused higher environmental impacts.

As observed by Pradel et al. (2016) and Lam et al. (2020), life cycle assessment studies that viewed WWS from a waste perspective favoured APR over conventional PR, mostly because of the zero -burden assumption for the production of the influent. Studies that used the zero-burden assumption did not account for the upstream impacts (for instance, biological treatment, digestion of WWS) leading up to struvite recovery.

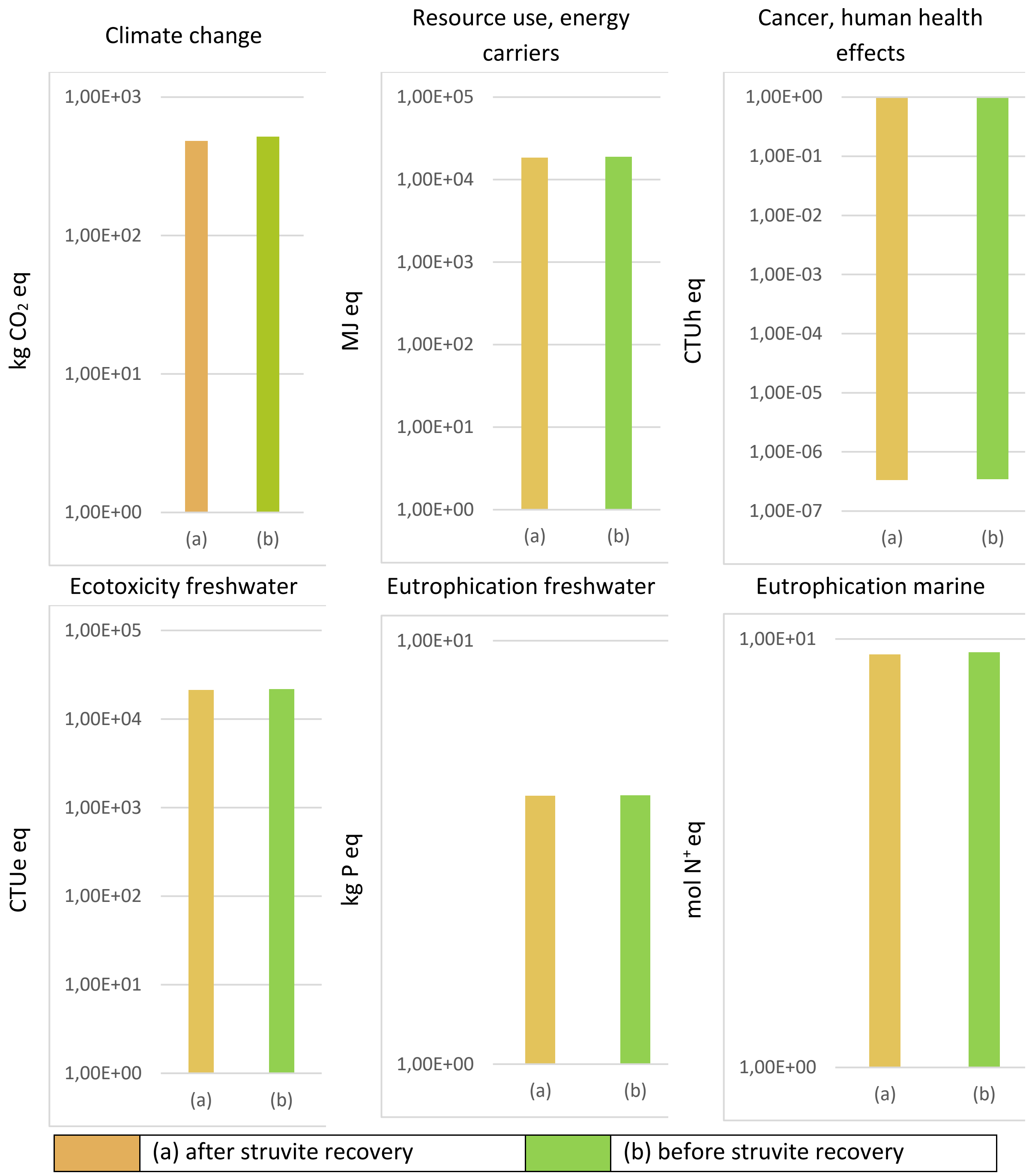


Figure 22. LL# 65: eLCA results – at midpoint

Overall impacts at midpoint for select impact categories. The functional units are 3927 m3 of wastewater treated and 1 kg of plant available fertiliser P. Notice the logarithmic scale.

To avoid bias, we chose to compare our results with studies that considered a product perspective. Most studies that considered WWS from a product perspective (i.e. a product-based FU, for example, provision of 1 kg of plant-available P as fertiliser) either compared struvite recovery versus synthetic fertiliser or other secondary P recovery processes. Linderholm, Tillman, and Mattsson (2012) observed that struvite recovery had a lower *climate change potential* compared to synthetic P fertiliser. Amann et al. (2018) performed a study similar to ours, and, compared struvite recovery (Gifhorn and Stuttgart process) from WWS versus a reference system without nutrient recovery. Their results also showed lower *climate change potential* compared to the reference. While the impacts on *acidification potential* were insignificant in our case, the impacts from *acidification potential* in their study were higher, mostly due to the use of chemicals (sulphuric acid, lye and citric acid). Tonini, Saveyn, and Huygens (2019), who evaluated struvite recovery versus rock phosphate, observed lower impacts for *climate change*, *terrestrial acidification*, *ecotoxicity* and *human toxicity potential* for struvite recovery.

3.1.7.3.2 Impact assessment at endpoint

The results at midpoint were normalized and weighted to a single score according to the Product Environmental Footprint guidelines (Figure 23). The major contributor to the single score for both scenarios is from *freshwater eutrophication*, and this is mainly a consequence of effluent discharge after biological treatment. From the single scores it seems that recovering struvite creates a marginal net benefit to the environment.

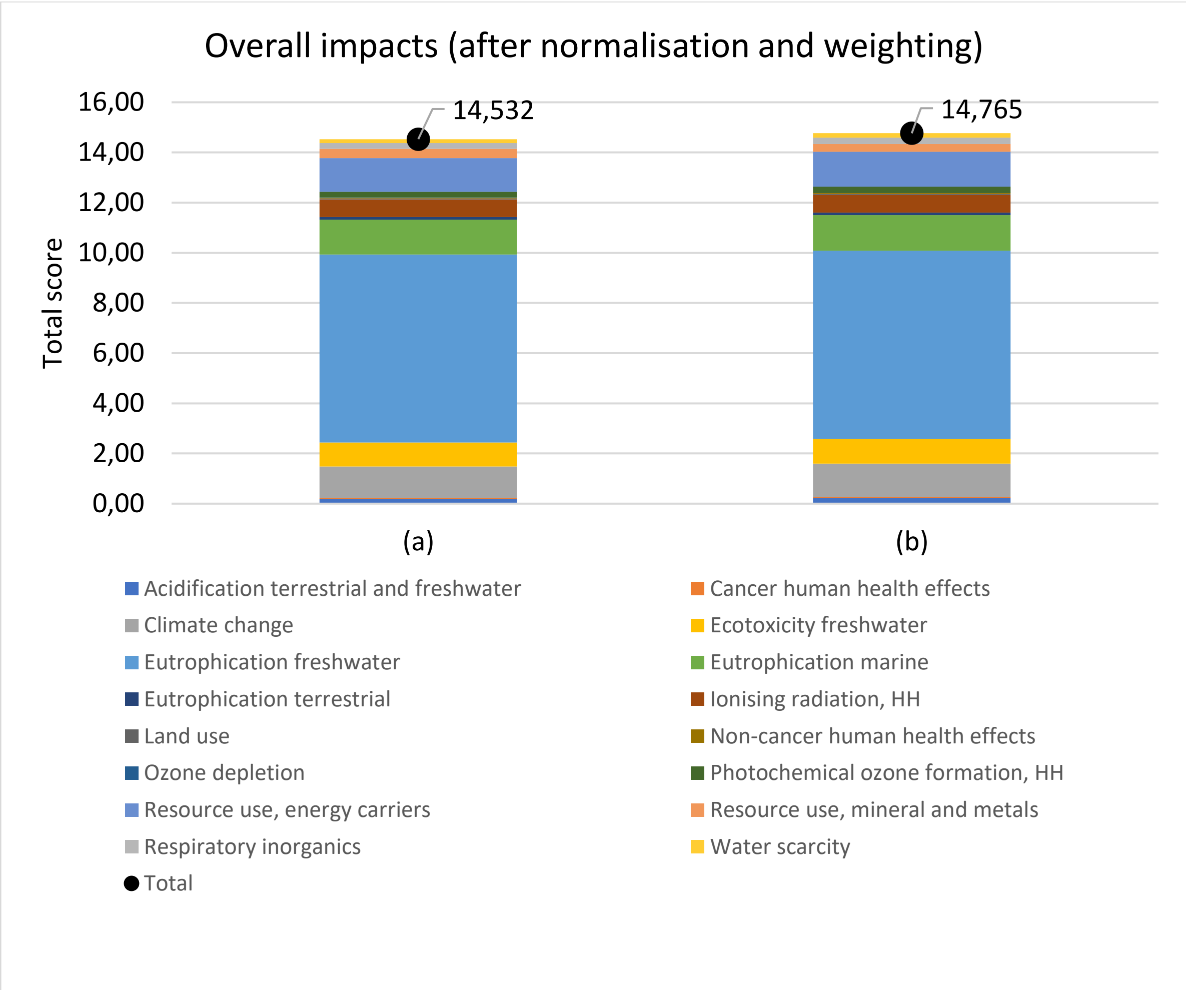


Figure 23. LL# 65: eLCA results: normalised and weighted

Single score results comparing after scenario(a) versus before scenario (b) struvite recovery

3.1.7.3 Discussion

A retrospective analysis of the environmental impacts of struvite recovery in a wastewater treatment plant (WWTP) reveals a slight reduction in the plant's environmental burdens following implementation. This positive outcome is attributed to the decreased use of polymers and reduced energy demand for dewatering. Notably, polymer usage, particularly polyacrylamide, emerges as the most influential parameter affecting overall environmental impacts. Future investigations could explore alternative polymers to assess their potential impact on WWTP performance. While peer-reviewed studies suggest the eco-friendly potential of biopolymers like chitosan and cellulose alginates, industrial-scale research on their usage is ongoing (Pandey, 2020).

Moreover, increasing the influent wastewater (currently recovering 5–6% of influent phosphorus) not only provides a substitute for synthetic phosphorus fertiliser but also enables a decrease in phosphorus load on the centrate. This, in turn, reduces electricity consumption (due to decreased aeration) and the use of saccharose in the biological treatment step.

Studies by Pradel et al. (2016) and Lam et al. (2020) indicate that life cycle assessments (LCAs) favouring struvite recovery over conventional phosphorus recovery (PR) often adopt a waste perspective, primarily due to the zero-burden assumption. To align with a product perspective, considering studies that account for upstream impacts leading to struvite recovery is crucial. Comparisons with studies adopting a product-based functional unit (e.g., providing 1 kg of plant-available phosphorus as fertilizer) show that struvite recovery, when compared with synthetic fertiliser or other phosphorus recovery processes, generally exhibits lower climate change potential. However, some variations exist among studies, such as Tonini et al. (2019) observing lower impacts for *climate change*, *terrestrial acidification*, *ecotoxicity*, and *human toxicity potential* for struvite recovery compared to rock phosphate.

While many studies employ system expansion and favour struvite recovery, Pradel and Aissani (2019) present a different perspective. They argue that phosphorus recovered from wastewater solids (WWS) should be considered a co-product resulting from a multifunctional system. In their approach, an allocation factor (45% of burdens to WWS management and 55% to wastewater treatment) is proposed to account for multi-functionality. Applying this factor, they contend that WWS-based phosphorus fertiliser, like struvite, appears less environmentally friendly than synthetic phosphorus fertilizer due to limited phosphorus yields, low phosphorus content, and high energy demand for recovery (Pradel and Aissani, 2019).

3.1.7.4 Conclusion

The current LCA indicated that struvite recovery slightly improved the environmental performance of a WWTP in Flanders. The hotspot analysis identified that, albeit marginal, reduced polymer use, improved dewaterability and avoided imports of synthetic P fertiliser resulted in a net benefit to the system as a consequence of the struvite precipitation. To further enhance the sustainability of WWTPs, plant operators may wish to focus on optimising polymer usage and at identifying sustainable substitutes. Struvite recovery at WWTPs have been sustainable from an economic standpoint, but from an environmental perspective, the difference is marginal. Therefore, future research could examine the effects of encrustation and scaling on infrastructure components and the benefits related to that, both prior to and following struvite recovery.

3.1.7.5 Dashboard indicators

LL# 65 Struvite recovery (*: Qualitative DBI assessment: **+** *improvement*, **o** *no change*, **-** *deterioration)*

Table 23. LL# 65: Comparison of dashboard indicator and life cycle assessment results

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication *	Comment	Quantitative estimate per FU (3927 m^3 of wastewater treated and 1 kg of plant available P)	LCA indication *	Re-evaluation of DBI irt. LCA results
Use of Primary Resources	**Rock Phosphate**	+	Direct Substitution	Phosphorous, in ground [n] -0.45 kg	+	Confirmed
	Natural Gas)	**o**		Gas, natural, in ground [o] -7.51 m^3	+	Revised. LCA indicated improvement
	Oil	+	Less transport needed (local P source instead of intercontinental P source)	Oil, crude, in ground [b] -7.4 m^3	+	Confirmed
	Electricity (Reduction in electricity consumption)	**o**		Electricity consumption 3 kWh	-	Revised. LCA indicated deterioration
	Water	**o**		Water scarcity	+	Revised. LCA indicated improvement

[n] Scores reflect the difference between after and before struvite recovery. A negative value indicates improvement whereas a positive value indicates deterioration

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication *	Comment	Quantitative estimate per FU (3927 m^3 of wastewater treated and 1 kg of plant available P)	LCA indication *	Re-evaluation of DBI irt. LCA results
	(Reduction in water consumption)			-15 m^3		
	Soil quality (Improvement in soil quality)	o		Land use -78.6 points	+	Revised. LCA indicated improvement
	Nutrients recovered (Nutrient recovered from agriculture and livestock systems)	+	Dedicated P recovery from sources that otherwise go to waste	no further info		
	Renewable biomass	-				
Emissions to the environ-ment	**Ammonia (air emission)** (Reduction in NH_3 emissions)	o		Ammonia emission to air -0.02 kg	+	Revised. LCA indicated improvement
	Dinitrogen monoxide (air emission) (Reduction in N_2O Emissions)	o		Dinitrogen monoxide emission to air 0 kg	o	Confirmed
	Methane (air emission) (Reduction in CH_4 emissions)	o		Methane emissions to air 0 kg	o	Confirmed
	Nitrates (water emission)	o		Nitrate emission to water		Confirmed

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment	DBI Indication *	Comment	Quantitative Dashboard Indicator based on LCA assessment: Quantitative estimate per FU (3927 m^3 of wastewater treated and 1 kg of plant available P)	LCA indication *	Re-evaluation of DBI irt. LCA results
	(Reduction in NO_3 emissions)			0 kg	o	
	Phosphorus (water emission) (Reduction of P Emissions)	o		Phosphorous emission to water: 0 kg	o	Confirmed
	Particulate matter (Reduction of particulate matter formation)	o		Particulates emission to air 0 kg	o	Confirmed
Resilience to climate change	**Carbon footprint** (Reduction of carbon footprint)	+	Savings on production and transport	Climate change: -28 kg	+	Confirmed
	Effective SOM (Effective Soil Organic Matter Improvement)	o		Carbon | Emission from soil	o	Confirmed
				Carbon dioxide | Emission from soil	o	Confirmed

3.1.7.5.1 Comparison of LCA and DBI results

Overall, the LCA results showed a strong correlation with the outcome from the DBI. The notable exceptions were electricity usage and water consumption. While electricity usage showed a deterioration (~3 kWh) due to struvite recovery, the outcome from water consumption was contrary. Struvite recovery showed reduced water consumption (~15 m^3). This is mostly as a consequence of reduced polymer usage for dewatering at the WWTP after implementing struvite recovery.

3.1.7.6 References

Bogdan, A., C. O' Donnell, A. A. Robles Aguilar, I. Sigurnjak, N. Power, E. Michels, J. Harrington and E. Meers (2021). "Impact of time and phosphorus application rate on phosphorus bioavailability and efficiency of secondary fertilisers recovered from municipal wastewater." Chemosphere **282**: 131017.

De Boer, M. A., A. G. Romeo-Hall, T. M. Rooimans and J. C. Slootweg (2018). "An Assessment of the Drivers and Barriers for the Deployment of Urban Phosphorus Recovery Technologies: A Case Study of The Netherlands." Sustainability **10**(6): 1790.

Egle, L., H. Rechberger, J. Krampe and M. Zessner (2016). "Phosphorus recovery from municipal wastewater: An integrated comparative technological, environmental and economic assessment of P recovery technologies." Science of The Total Environment **571**: 522-542.

Kataki, S., H. West, M. Clarke and D. C. Baruah (2016). "Phosphorus recovery as struvite: Recent concerns for use of seed, alternative Mg source, nitrogen conservation and fertiliser potential." Resources, Conservation and Recycling **107**: 142-156.

Kraus, F., M. Zamzow, L. Conzelmann, C. Remy, A. Kleyböcker, W. Seis, U. Miehe, L. Hermann, R. Hermann and C. Kabbe (2019). "Ökobilanzieller Vergleich der P-Rückgewinnung aus dem Abwasserstrom mit der Düngemittelprodukti-on aus Rohphosphaten unter Einbeziehung von Umweltfolgeschäden und deren Vermeidung."

Lam, K. L., L. Zlatanović and J. P. van der Hoek (2020). "Life cycle assessment of nutrient recycling from wastewater: A critical review." Water Research **173**: 115519.

Marchi, A., S. Geerts, B. Saerens, M. Weemaes, L. De Clercq and E. Meers (2020). Struvite Recovery from Domestic Wastewater. Biorefinery of Inorganics**:** 107-119.

Marchi, A., S. Geerts, M. Weemaes, S. Wim and V. Christine (2015). "Full-scale phosphorus recovery from digested waste water sludge in Belgium–part I: technical achievements and challenges." Water Science and Technology **71**(4): 487-494.

Niero, M., M. Pizzol, H. G. Bruun and M. Thomsen (2014). "Comparative life cycle assessment of wastewater treatment in Denmark including sensitivity and uncertainty analysis." Journal of Cleaner Production **68**: 25-35.

Pradel, M., L. Aissani, J. Villot, J.-C. Baudez and V. Laforest (2016). "From waste to added value product: towards a paradigm shift in life cycle assessment applied to wastewater sludge–a review." Journal of cleaner production **131**: 60-75.

Ravi, R., M. Beyers, S. Bruun and E. Meers (2022). "Life cycle assessment of struvite recovery and wastewater sludge end-use: A Flemish illustration." Resources, Conservation and Recycling **182**: 106325.

Renou, S., J. S. Thomas, E. Aoustin and M. N. Pons (2008). "Influence of impact assessment methods in wastewater treatment LCA." Journal of Cleaner Production **16**(10): 1098-1105.

Saerens, B., S. Geerts and M. Weemaes (2021). "Phosphorus recovery as struvite from digested sludge–experience from the full scale." Journal of Environmental Management **280**: 111743.

Tonini, D., H. G. Saveyn and D. Huygens (2019). "Environmental and health co-benefits for advanced phosphorus recovery." Nature Sustainability **2**(11): 1051-1061.

Vaneeckhaute, C., J. Janda, E. Meers and F. M. G. Tack (2015). Efficiency of Soil and Fertiliser Phosphorus Use in Time: A Comparison Between Recovered Struvite, FePO4-Sludge, Digestate, Animal Manure, and Synthetic Fertiliser. Nutrient Use Efficiency: from Basics to Advances. A. Rakshit, H. B. Singh and A. Sen. New Delhi, Springer India**:** 73-85.

3.1.8 LL#17: Comparison of dairy sludge derived fertilisers and conventional inorganic fertilisers (CARTIF)

Longlist #LL17 title: Crop farmer using a variety of manure and dairy processing residues to recycle and build soil C, N, P fertility

Francisco Verdugo-González[1], Ashekuzzaman S.M[2 3], Yun-Feng Duan[4], Patrick J. Forrestal[2], Francisco Corona[1]

[1] CARTIF Technological Centre, Parq. Tec. Boecillo, Valladolid, Spain.

[2] TEAGASC, Environment Research Centre, Johnstown Castle, Co., Y35 Y521 Wexford, Ireland

[3] Munster Technological University, Cork, Ireland

[4] Sustainable Soil Management Group, Wageningen Environmental Research, the Netherlands.

3.1.8.1 Introduction

The European dairy industry generates significant wastewater from milk and dairy processing. In wastewater treatment, removal of phosphorus (P) by complexing the P-rich wastewater sludge with metal cations (e.g., aluminium, calcium) can facilitate P recovery and recycling in agriculture (Ashekuzzaman, S.M. et al. , 2021). Wastewater sludge with metal cations (e.g., aluminium, calcium) can facilitate P recovery and recycling in agriculture (Ashekuzzaman, S.M. et al. , 2021).

This chapter presents a life cycle assessment study of two dairy sludge pathways: (1) aluminium-precipitated sludge (Al-DPS) and (2) calcium-precipitated lime-stabilised sludge (Ca-DPS) at field scale, applied to 1 ha of grassland in Ireland. The objective was to compare the environmental impact of providing P fertiliser to Irish grassland either through Al-DPS and Ca-DPS or through mineral P fertiliser.

3.1.8.2 Material & method

3.1.8.2.1 Goal & scope

The goal of this study was to estimate the environmental impacts of different fertilisation programmes on Irish grasslands with different inorganic and organic phosphorous sources:

1. **Inorganic fertilisation** (superphosphate - SP). Only inorganic fertilisers were applied during the experiment. The inorganic fertilisers were: calcium ammonium nitrate, triple superphosphate, potassium chloride and potassium sulphate.
2. **Organic fertilisation**. In these cases, two different dairy sludges were applied. The organic fertilisers were: (1) aluminium-precipitated sludge (Al-DPS) and (2) calcium-precipitated lime-stabilised sludge (Ca-DPS). In addition, some inorganic fertilisers were applied as source of nitrogen, sulphur, and potassium. Organic fertiliser application involves the avoidance of landfilling of dairy sludge, which is considered the conventional treatment in this assessment.

We selected a functional unit (FU) of “*100 tons of grass production*”. A graphic representation of system boundaries and essential processes of each programme considered in this assessment can be found in Figure 24.

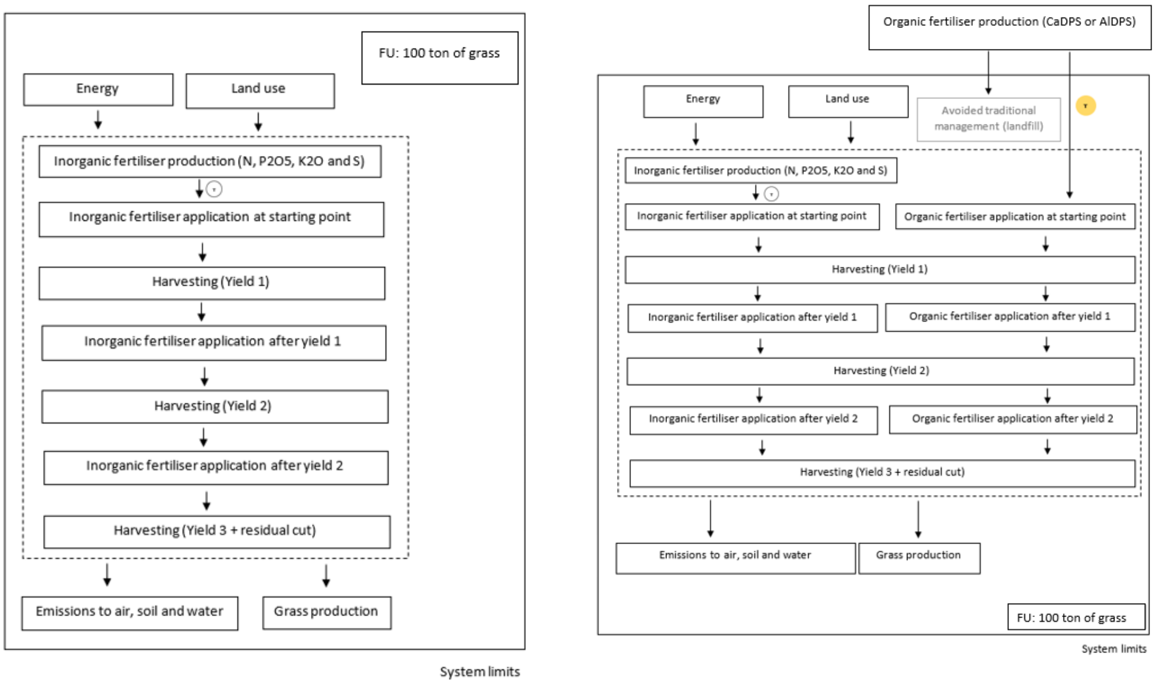


Figure 24. LL# 17: System boundaries

Structure of analysed system. Boxes indicate main activities associated with inorganic fertiliser program (a) and combination of inorganic and organic fertiliser program (b). Arrows indicate flows of products. T (yellow circle): transportation with varying distances. FU: functional unit.

Different volumes of inorganic fertiliser and dairy sludge result in different transport burdens in relation to a set amount of nutrients. Thus, the transport distance has to be considered in order to determine the maximum distance at which dairy sludge could be transported without causing an increment of the environmental burdens due to transportation.

3.1.8.2.2 Inventory

The data required for this assessment has been gathered during experimental field trials, through modelling and from the attributional ecoinvent database.

The field trails were conducted by TEAGASC on Irish grassland and are described in more detail in the study presented in Ashekuzzaman, S.M.et al. (2021) and in D.2.6 of the Nutri2Cycle project (F. Adani et al. 2022). Crop yield and P uptake were assessed for three grass harvests over three growing seasons (2019-2021). Initial soil testing showed low levels of P in the soil. The experimental treatments were super phosphate at 15, 30, 40, 50 and 60 kg P ha^{-1} (inorganic fertilisation) and two types of treated dairy sludge at 40 kg P ha^{-1} (organic fertilisation). The collected data includes: 1) inorganic fertiliser consumption; 2) organic (Ca-DPS and Al-DPS) fertiliser consumption and composition; 3) yield.

Nitrogen and phosphorous emissions to water and air were simulated by the SWAP/ANIMO models (Kroes et al., 2017; Groenendijk et al., 2005) using the ATC-Dairy baseline scenario from D.1.5 of the Nutri2Cycle project (Duan et al., 2021). The ATC-Dairy scenario models a perennial grassland on a sandy soil in the Atlantic Central (ATC) environmental zone, which covers the Benelux region, northern France, and most of Ireland. Three fertiliser treatments were modelled in accordance with the field trial (Ashekuzzaman et al., 2021): (1) inorganic P fertiliser (SP), (2) aluminium-precipitated sludge (Al-DPS), and (3) calcium-precipitated lime-stabilised sludge (Ca-DPS). Fertiliser application rates were identical in the field trial. The model was calibrated to reflect the differences in grass yields and P bioavailability between treatments as observed in the field trials. As soil and climate parameters used in the model differed from the specific conditions of the Irish site, it was difficult to calibrate the simulated yields to match those from the field trial exactly. Instead, the cumulative dry matter and P yields were calibrated based on relative differences between treatments, using the SP treatment as the baseline. To simulate the differences in P availability to crops between Al-DPS and Ca-DPS, composition and turnover rates of the slow- and fast-degrading organic fractions of Al-DPS and Ca-DPS were modified to achieve a 15-20% higher crop P uptake by Al-DPS than by Ca-DPS. The simulation was first run for a spin-up period of 20 years to prime the soil organic matter pools. Then, the simulation was run for 20 years to account for weather variations, and to collect results on crop production and environmental emissions. The 20-year average N and P emissions to water bodies and the atmosphere are presented in Table 24

To complete the dataset, background data from the attributional ecoinvent 3.8 database have been included. Derived data included: 1) inorganic fertiliser production, 2) fuel consumption for fertiliser spreading and 3) fuel consumption for harvesting processes.

The impacts of each system were determined using Environmental Footprint 3.0 methodology.

Table 24. LL# 17: Life Cycle Inventory.

SP: inorganic fertiliser; Ca-DPS: calcium-precipitated lime-stabilised sludge; Al-DPS: aluminium-precipitated sludge. If not indicated otherwise, data is derived from field experiments of TEAGASC.

Process	Description	Unit/F	SP	Ca-DPS	Al-DPS
Fertiliser production	NH_4NO_3	kg	975	959	909
	KCl		1098	1053	1059
	K_2SO_4		498	489	433
	TSP		341	-	-
	Ca-DPS	ton	-	6	-
	Al-DPS		-	-	25.5
Fertiliser spreading Harvesting	Surface	ha	20.8	21.3	21.8
Emissions to air[a]	NH_3	kg	15.8	15.8	15.6
	N_2O		18.6	18.6	19.2
Emissions to water[a]	NO_3 [b]		20.1	20.1	20.3
	NO_3 [c]		19.0	19.0	18.5
	P [2]		0.05	0.05	0.05
	P [3]		0.01	0.01	0.01
Grass production	Dry matter	ton	43	42	43
Land use	Surface	ha/FU	7.6 6.6 6.6	8.0 6.6 6.7	8.0 6.6 7.2

[a] SWAP/ANIMO results; [b] To surface water; [c] To groundwater

3.1.8.3 Results

3.1.8.3.1 Impact assessment

This section presents the results of the assessment carried out.

Table 25 and Figure 25 show the comparison between the different systems described in previous sections. Table 25 shows the environmental impact for each impact category excluding the transportation of organic fertiliser to the fields. The impact of transportation has been considered in terms of maximum distance (see values in brackets) at which dairy sludge could be spread to fields without resulting in higher environmental burdens than the reference scenario (SP).

Table 25. LL# 17: eLCA results.

Environmental impact results per functional unit of 100 t harvested grass. SP: inorganic fertiliser; Ca-DPS: calcium-precipitated lime-stabilised sludge; Al-DPS: aluminium-precipitated sludge. In brackets: maximum feasible transport distance from an environmental impact perspective in km.

Impact category	Units	SP	Ca-DPS	Al-DPS
Climate change pot. (CP)	kg CO_2 eq	6.62E+04	5.48E+04 (11648)	1.91E+04 (11528)
Ozone depletion pot. (OP)	kg CFC^{11} eq	3.76E-03	3.67E-03 (416)	3.47E-03 (308)
Ionising radiation pot. (IRP)	kBq U-235 eq	2.03E+03	1.94E+03 (1135)	1.83E+03 (619)
Photochemical ozone (PP)	kg NMVOC eq	1.07E+02	9.86E+01 (3485)	8.40E+01 (2321)
Respiratory inorganics (RIP)	disease inc.	4.03E-03	3.93E-03 (1280)	3.81E-03(652)
Non-chancer human health (HNP)	CTUh	6.56E-04	5.94E-04 (5390)	4.97E-04(3312)
Cancer human health effect (HCP)	CTUh	1.83E-05	1.67E-05 (4733)	1.56E-05 (1913)
Acidification pot. (AP)	mol H+ eq	5.36E+02	5.26E+02 (3516)	5.11E+02 (2198)
Freshwater eutrophication (FEP)	kg P eq	6.56E+00	4.55E+00 (31694)	-1.60E-01(25497)
Marine eutrophication (MEP)	kg N eq	1.11E+02	1.16E+02 (-8821)	3.14E+01(33558)
Terrestrial eutrophication (TEP)	mol N eq	2.12E+03	2.12E+03 (761)	2.08E+03(1770)
Freshwater ecotoxicity (FTP)	CTUe	8.43E+06	7.86E+06 (49053)	5.95E+06(51091)
Land use potential (LUP)	Pt	2.80E+05	2.63E+05 (1570)	2.36E+05(1016)
Water use potential (WUP)	m^3 depriv.	1.53E+04	1.42E+04 (22040)	1.33E+04(9717)
Resource use, energy (ERP)	MJ	4.14E+05	3.95E+05 (1287)	3.73E+05(674)
Resource use, minerals (MRP)	kg Sb eq	5.37E-01	4.83E-01 (16953)	4.56E-01(6168)

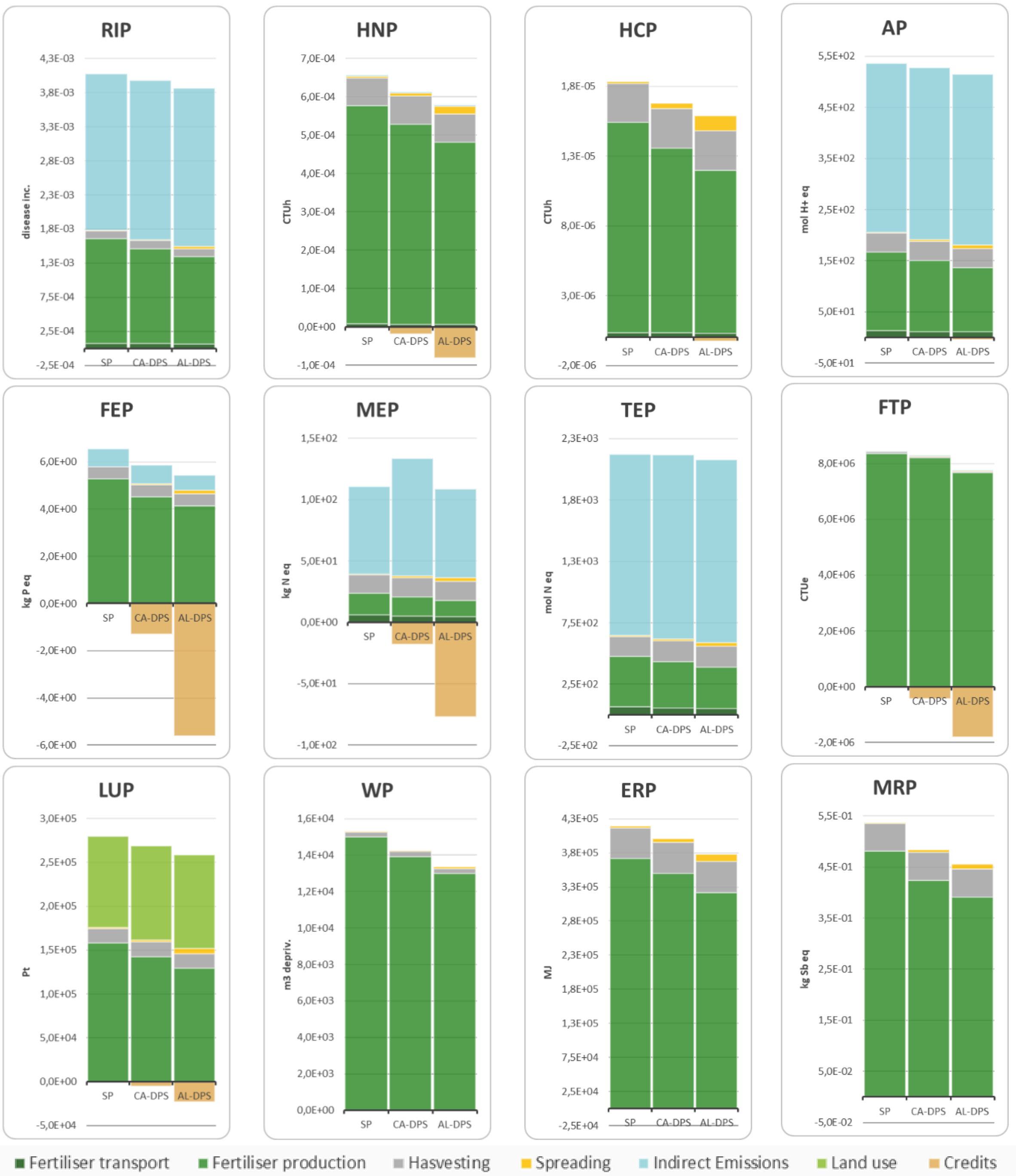


Figure 25. LL# 17: eLCA results -1

Environmental impact per functional unit 100 ton of grass, with using 100% inorganic fertiliser (SP); Inorganic fertiliser and Ca-DPS sludge (CaDPS) and Inorganic fertiliser and Al-DPS sludge (AlDPS) for impact categories considered in the assessment.

The SP column in Table 25 shows the impact assessment results of the inorganic fertilisation programme which is considered as reference system of this study. The environmental pattern of this system (Figure 25) shows that inorganic fertiliser production is the process which highly contributes to the impact categories: *ionising radiation (IRP), human toxicity (cancer and non-cancer), freshwater eutrophication (FEP), freshwater ecotoxicity (FTP), water use (WUP), resource use (fossils and minerals and metals)*. In addition, indirect emissions to atmosphere, soil and water, due to fertiliser uses, highly contribute to *climate change (CCP), particulate matter (RIP), acidification (AP)* and *eutrophication* *(marine and terrestrial)*. Finally, other processes, such as fertiliser transport, harvesting and land use contribute significantly to *ozone depletion (OP)*, *photochemical ozone formation (PP)* and *land use (LUP)*, respectively.

Ca-DPS and Al-DPS and the baseline scenario show a similar environmental impact pattern (Figure 25). The main difference lays in the reduction of environmental burdens due to the avoidance of traditional sludge management. The avoidance of landfilling reduces the environmental impacts in the following impact categories: *climate change, photochemical ozone formation, human toxicity (non-cancer eutrophication (marine and freshwater), freshwater ecotoxicity* and *land use*. When comparing, Ca-DPS and Al-DPS, Al-DPS shows higher reductions than Ca-DPS, due to the larger amount of sludge spread to fields instead of being landfilled.

Regarding Ca-DPS sludge, avoiding its landfilling results in an impact reduction, compared to the SP program, among others in CCP (94%), FTP (74%), FEP (65%), PP (40%) potential. Apart from that, environmental impact reductions can be achieved across all considered impact categories due to the avoidance of the production of inorganic P fertiliser. However, other processes such as harvesting and land use increase the burdens in some impact categories due to lower yields per area and greater needs for land. Another process, which contributes to an increase in environmental impact of the Ca-DPS system compared to the SP program, is the actual spreading of the sludge compared to the spreading of inorganic fertiliser to agricultural land. In the Ca-DPS system 8.5 t of sludge and in the SP program 2.5 t of inorganic fertiliser are applied on a per hectare basis to meet nutrient demands. Finally, the Ca-DPS program shows a higher environmental impact on *marine eutrophication potential* than the SP program. That is due to increased NO_3 emissions to water bodies, given the higher susceptibility to leaching of N bound in organic compared to N bound in inorganic fertilisers.

Similar to the Ca-DPS system, the Al-DPS sludge fertilisation program suggests lower environmental impacts compared to the inorganic P fertilisation program. Avoiding the production of inorganic phosphorus fertiliser and traditional sludge management (landfilling) results in an overall burden reduction. Main differences between both organic fertilisation programmes are caused by the different amounts of dairy sludge applied which is reflected in impacts on *climate change* and *freshwater* and *marine eutrophication*. In addition, like for Ca-DPS, considering only spreading the application of total Al-DPS sludge (25.5 ton) shows higher environmental impacts than the SP program, given that the inorganic fertiliser amounts (2.9 ton) required to meet fertilising requirements are lower.

At this point, considering the large amounts of sludge needed to meet plant-nutrient requirements, it is relevant to determine the tipping point and maximum transport distance after which the use of studied organic fertiliser becomes environmentally unacceptable. Thus, the maximum transport distances were calculated considering the difference between each system for the different impact categories considered, see Table 25 (values in brackets). In accordance with the results obtained, *ozone depletion* is the limiting impact category in terms of maximum transport distance. With regards to transport distances, Ca-DPS and Al-DPS could be transported 416 and 308 km, respectively, without surpassing the environmental impacts of inorganic P fertiliser.

3.1.8.4 Interpretation

In accordance with the results obtained, all scenarios show a similar environmental behaviour, in other words, like processes contribute with similar percentages in each impact category.

The most impactful process is the production of inorganic fertiliser, which highly contributes to all studied impact categories. In addition, emissions to soil, water, and atmosphere caused by fertiliser use contribute significantly to the following impact categories: *climate change*, *particulate matter*, *acidification, marine* and *terrestrial eutrophication*. Inorganic fertiliser transportation shows a high contribution to *ozone depletion* and harvesting contributes greatly to *photochemical ozone formation*.

It is well-known that not all the forms of P exhibit similar mobility and bio-availability in the sludge. Therefore, detailed information about P fractions is necessary, especially when land application of sludge is taken into consideration (Huang, W et al., 2015).

The main difference between organic and inorganic fertilisation lays in the avoidance of the conventional management of dairy sludge (landfilling) and in the avoidance of inorganic P fertiliser production. Theses avoidances result in a decrease in the environmental impact categories: CCP, PP, *human toxicity (non-cancer)*, *eutrophication (marine and freshwater)* and *freshwater ecotoxicity*. In contrast, and only in the Ca-DPS programme, burdens due to NO_3 emissions to water show greater environmental impact in terms of *marine eutrophication* than the reference system, see Figure 26.

The transportation of sludge between different treatment plants produces a considerable amount of greenhouse gases (GHG), mainly in the form of CO_2 (Mayer, F., 2021). Furthermore, Lam et al. (2016) investigated the correlation between transportation distance assumptions and climate change impacts, and it was validated that the default setting for the uniform distance between wastewater treatment plants and sludge disposal facilities lacked rationality for environmental impact analysis. For this reason, this study has not assumed a default transport distance and considered it as a variable to be studied, obtaining that ozone depletion potential is the limiting impact category in terms of maximum transport distance. Thus, as long as the transport distances are less than 416 km in Ca-DPS and 308 km in Al-DPS, the results suggest that replacing inorganic P fertiliser with dairy sludge is a sustainable way of managing this form of by-product.

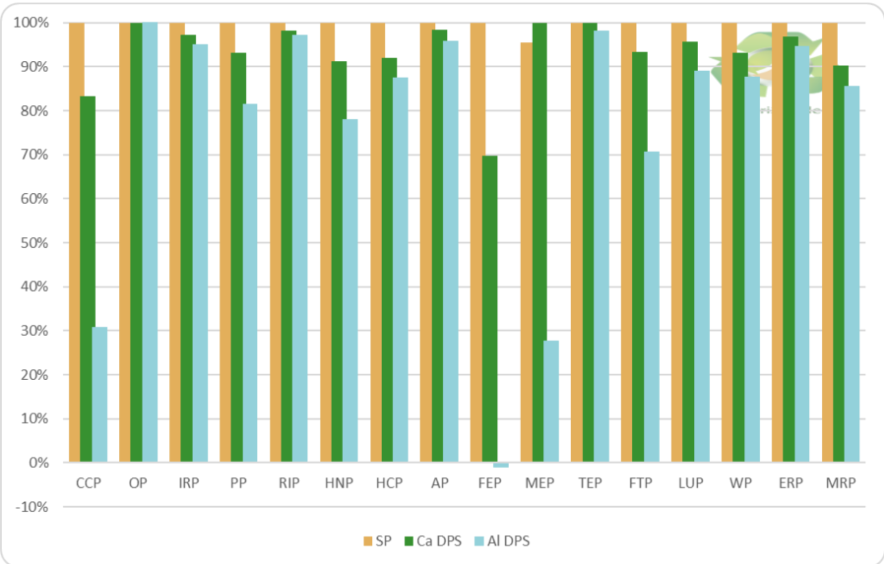


Figure 26. LL# 17: eLCA results – 2

Systems environmental impact comparison. Environmental impacts of shortest maximum transport distances obtained (from Ozone depletion impact category, OP) has been included.

3.1.8.5 Conclusion

In conclusion, the use of Ca-DPs and Al-DPS, as organic phosphorous source on Irish grasslands, shows a better environmental behaviour than inorganic P fertiliser. The avoidance of traditional sludge management and inorganic P fertiliser production decreases the environmental impacts of the organic fertilisation programmes.

The study shows how the environmental impact reductions compensate the environmental impact caused by the transport of great amounts of organic P fertiliser.

In accordance with the results obtained, *ozone depletion potential* is the limiting impact category in terms of maximum transport distance. In this way, as long as transport distances are below 416 km in Ca-DPS program and 308 km in Al-DPS, results suggest that the substitution of inorganic P fertiliser by dairy sludge is a sustainable way to manage this form of sub-product.

3.1.8.6 Dashboard indicators

#LL 17 Crop farmer using a variety of manure and dairy processing residues to recycle and build soil C, N, P fertility (*: Qualitative DBI assessment: **+** *improvement*, **o** *no change*, **-** *deterioration)*

Table 26. LL# 17: Comparison of dashboard indicator and life cycle assessment results

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication *	Comment	Quantitative estimate per FU of 100 t harvested grass at 200 km transport distance	LCA indication*	Re-evaluation of DBI irt. LCA results
Use of Primary Resources	**Rock Phosphate** (Reduction in mineral phosphorus consumption)	+	100 % of substitution of mineral fertiliser by recycling bio-based fertilisers	Phosphorous, in ground (g) Ca-DPS: -1.0 (-6%) Al-DPS: -1.2 (-8%)	Ca-DPS: **o** Al-DPS: **o**	Ca-DPS: revised, no variation Al-DPS: revised, no variation
	Natural Gas (Reduction in natural gas consumption in mineral fertiliser production)	+	Reduction in relation to mineral fertilisers	Gas, natural, in ground (Nm^3) Ca-DPS: -283 (-4%) Al-DPS: -647 (-9%)	Ca-DPS: **o** Al-DPS: **o**	Ca-DPS: revised, no variation Al-DPS: revised, no variation
	Oil (Reduction in oil consumption in agricultural machinery)	-	Energy consumption for transport of some locally available bio-based fertilisers	Oil, crude, in ground (tn.lg) Ca-DPS: -0.02 (-1%) Al-DPS: 0.12 (5%)	Ca-DPS: **o** Al-DPS: **o**	Ca-DPS: revised, no variation AlDPS: revised, no variation
				Diesel burnt in agricultural machinery (MJ): Ca-DPS: 4.4 (>10%) Al-DPS: 19.5 (>10%)	Ca-DPS: **-** Al-DPS: **-**	Ca-DPS: confirmed, increase Al-DPS: confirmed, increase.
	Electricity (Reduction in electricity consumption)	o		Electricity consumption (GJ) Ca-DPS: -16.7 (-9%) Al-DPS: -19.2 (-11%)	Ca-DPS: **o** Al-DPS: **+**	Ca-DPS: confirmed, no variation Al-DPS: revised, more likely red.
	Water	o		Water scarcity (m^3 depriv)	Ca-DPS: **o**	Ca-DPS: confirmed, no variation

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication *	Comment	Quantitative estimate per FU of 100 t harvested grass at 200 km transport distance	LCA indication*	Re-evaluation of DBI irt. LCA results
	(Reduction in water consumption)			Ca-DPS: $-1.1 \cdot 10^3$ (-7%) Al-DPS: $-1.8 \cdot 10^3$ (-12%)	Al-DPS: **+**	Al-DPS: revised, more likely red
	Soil quality (Improvement in soil quality)	+	We could assume that bio-based fertiliser application increases soil biological quality	Land use (Pt) Ca-DPS: $-1.5 \cdot 10^4$ (-5%) Al-DPS: $-3.3 \cdot 10^4$ (-12%)	Ca-DPS: **o** Al-DPS: **+**	Ca-DPS: revised, no variation Al-DPS: confirmed, reduction
	Nutrients recovered (Nutrient recovered from agriculture and livestock systems)	+	Valorisation of bio-based products as fertiliser	-		No circularity indicator in LCA
	Renewable biomass	-	-	-		
	Others? Please specify	-	-	no further info		
Emissions to the environ-ment	**Ammonia (air emission)** (Reduction in NH_3 emissions)	-	Some bio-based product application increases ammonia emissions	Ammonia emission to air (kg): Ca-DPS: 0.2 (0.1%) Al-DPS: -1 (-1%)	Ca-DPS: **o** Al-DPS: **o**	Ca-DPS: confirmed, no variation Al-DPS: confirmed, no variation
	Dinitrogen monoxide (air emission) (Reduction in N_2O Emissions)	-	Some bio-based product application increases dinitrogen monoxide emissions	Dinitrogen monoxide emission to air (kg): Ca-DPS: 2.1 (2%) Al-DPS: 6.1 (5%)	Ca-DPS: **o** Al-DPS: **o**	Ca-DPS: confirmed, no variation Al-DPS: confirmed, no variation

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication *	Comment	Quantitative estimate per FU of 100 t harvested grass at 200 km transport distance	LCA indication*	Re-evaluation of DBI irt. LCA results
	Methane (air emission) (Reduction in CH_4 emissions)	**+**	Reduction in relation to less use of mineral fertilisers	Methane emission to air (mg) Ca-DPS: -13.2 (-6%) Al-DPS: -22.1 (-10%)	Ca-DPS: **o** Al-DPS: **+**	Ca-DPS: revised, no variation Al-DPS: confirmed, reduction
	Nitrates (water emission) (Reduction in NO_3 emissions)	**o**		Nitrate emission to water (kg) Ca-DPS: 103 (+37%) Al-DPS: -2.6 (-2%)	Ca-DPS: **-** Al-DPS: **o**	Ca-DPS: revised, more likely inc. Al-DPS: confirmed, no variation
	Phosphorus (water emission) (Reduction of P Emissions)	**+**	Considering P recovery for some bio-based products and slow release	Phosphorous emission to water (kg) Ca-DPS: -0.1 (-7%) Al-DPS: -0.2 (-20%)	Ca-DPS: **o** Al-DPS: **+**	Ca-DPS: revised, no variation Al-DPS: confirmed, reduction
				Phosphate emission to water (kg) Ca-DPS: -6.2 (-37%) Al-DPS: -19.5 (-116%)	Ca-DPS: **+** Al-DPS: **+**	Ca-DPS: confirmed, reduction Al-DPS: confirmed, reduction
	Particulate matter (Reduction of particulate matter formation)	**o**	-	Particulates emission to air (kg) Ca-DPS: -4.5 (-12%) Al-DPS: -4.6 (-12%)	Ca-DPS: **+** Al-DPS: **+**	Ca-DPS: revised, more likely red. Al-DPS: revised, more likely red
	Others? Please specify	-	-			
Resilience to climate change	**Carbon footprint** (Reduction of carbon footprint)	**+**	Substitution of mineral fertilisers will reduce carbon footprint	Climate change (kg CO_2 eq) Ca-DPS: $-1{,}2\cdot10^4$ (-17%) Al-DPS: $-4{,}6\cdot10^4$ (-69%)	Ca-DPS: **+** Al-DPS: **+**	Ca-DPS: confirmed, reduction Al-DPS: confirmed, reduction

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication *	Comment	Quantitative estimate per FU of 100 t harvested grass at 200 km transport distance	LCA indication*	Re-evaluation of DBI irt. LCA results
	Effective SOM (Effective Soil Organic Matter Improvement)	**+**	We could assume that some bio-based fertiliser application increases SOC, and OM, but difficult to see the effect in the short-term	Carbon | Emission from soil (kg) Ca-DPS: -0.1 (-5%) Al-DPS: -0.1 (-5%)	Ca-DPS: **o** Al-DPS: **o**	Ca-DPS: revised, no variation Al-DPS: revised, no variation
				Carbon dioxide | Emission from soil (kg) Ca-DPS: -1.2 (-50%) Al-DPS: -1.2 (-50%)	Ca-DPS: **+** Al-DPS: **+**	Ca-DPS: confirmed, reduction Al-DPS: confirmed, reduction
	Renewable energy production (Renewable energy produced from biomass)	**(-)**		-		
	Others? Please specify	**(-)**				

3.1.8.6.1 Comparison of LCA and DBI results

For this solution, in which transport distance is under study, the results of the LCA and DBI assessments are quite different.

The assessment presented in this chapter, compared the LCA results for a transport distance of 200 km for dairy sludge vs. mineral fertilisation to produce 100 tons of grass against the DBI results. The assessment found agreement between the two assessments methods in terms of oil consumptions, i.e., diesel burnt in agricultural machinery, where both assessments pointed at increased consumption. There was also agreement in terms of the carbon footprint with both assessments concluding with reductions.

In some cases, the DBI suggested more positive results for the advanced technology than were indicated in the LCA indicators, namely in terms of phosphorous mining, natural gas consumption or carbon emissions from soil. In other cases, the DBI suggested more negative implications than the LCA indicators, namely with respect to crude oil consumption, and emissions of ammonia, nitrous oxide, and particulate matter. In the other categories, the LCA indicated different results for the Ca-DPS than for the Al-DPS treatment. A differentiation between the treatments was not made in the DBI assessment.

3.1.8.7 References

Ashekuzzaman, S. M., Fenton, O., Meers, E., & Forrestal, P. J. (2021). Differing phosphorus crop availability of aluminium and calcium precipitated dairy processing sludge potential recycled alternatives to mineral phosphorus fertiliser. *Agronomy*, 11(3), 427.

EC (European Commission), 2015. Factsheet: An ambitious EU circular economy package. Available at: https://commission.europa.eu/publications/ambitious-eu-circular-economy-package_en

Kroes, J. G., Van Dam, J. C., Bartholomeus, R. P., Groenendijk, P., Heinen, M., Hendriks, R. F. A., ... & Van Walsum, P. E. V. (2017). SWAP version 4: theory description and user manual. Alterra-rapport-Wageningen University and Research Centre, (2780).

Groenendijk, P., Renaud, L. V., & Roelsma, J. (2005). Prediction of nitrogen and phosphorus leaching to groundwater and surface waters; process descriptions of the Animo 4.0 model (No. 983). Alterra.

Huang, W., Cai, W., Huang, H., Lei, Z., Zhang, Z., Tay, J. H., & Lee, D. J. (2015). Identification of inorganic and organic species of phosphorus and its bio-availability in nitrifying aerobic granular sludge. *Water Research*, 68, 423-431.

Mayer, F., Bhandari, R., & Gäth, S. A. (2021). Life cycle assessment of prospective sewage sludge treatment paths in Germany. *Journal of Environmental Management*, 290, 112557.

Lam, C. M., Lee, P. H., & Hsu, S. C. (2016). Eco-efficiency analysis of sludge treatment scenarios in urban cities: the case of Hong Kong. *Journal of Cleaner Production*, 112, 3028-3039.

Regulation, E. U. (2019). Regulation (EU) 2019/1009 of the European Parliament and of the Council of 5 June 2019 laying down rules on the making available on the market of EU fertilising products and amending Regulations (EC) No 1069/2009 and (EC) No 1107/2009 and repealing Regulation (EC) No 2003/2003 (Text with EEA relevance). OJ L 170/1, 25.6. 2019. J. Eur. Union, 62, 1-132.

3.1.9 LL#41: Duckweed ponds as intermediate treatment step in liquid fraction pig manure management in Flanders (UCPH+UGENT+inagro)

Longlist #41 title: Floating wetland plants grown on liquid agro-residues as a new source of proteins

Miriam Beyers[1,2], Rahul Ravi[2,1], Reindert Devlamynck[3], Erik Meers[2], Lars Stoumann Jensen[1], Sander Bruun[1]

[1] Department of Plant and Environmental Sciences, Faculty of Science, University of Copenhagen, Denmark

[2] Department of Green Chemistry and Technology, Faculty of Bioscience Engineering, Ghent University, Belgium

[3] Inagro vzw, Ieperseweg 8, 8800 Rumbeke-Beitem, Belgium

This chapter builds on the published scientific paper Beyers et al. (2023) Constructed wetlands and duckweed ponds as a treatment step in liquid manure handling — A life cycle assessment. Sci. Total Environ. 889, https://doi.org/10.1016/j.scitotenv.2023.163956

3.1.9.1 Introduction

The region of Flanders in Belgium has a highly intensive animal husbandry industry. This creates unique challenges, because the industry produces high nitrogen (N) and phosphorus (P) surpluses from the livestock manure, while being dependent on the global market to meet its livestock's protein needs, also due to non-conducive climatic conditions to cultivate soy, a common protein feed supplement.

A multi-pronged approach is therefore necessary to manage the surplus nutrients as well as to reduce the reliance on soy imports. Manure management in Flanders starts with manure separation, with the idea of isolating the N and P fractions into the liquid and solid fractions, respectively. While the P-rich solid fraction is transported to P-deficient regions, the N-rich liquid fraction is either valorised or treated (See LL#1+2+6). In Flanders, it is common to treat the liquid fraction via biological treatment thereby converting the N to inert N_2. Post treatment, the residual N in the effluent from biological treatment is still well above discharge limits (250 mg/l COD, 25 mg/l BOD, 35 mg/l suspended solids, 15mg/l N and 1mg/l P). Therefore, it is typically applied to agricultural fields, where due to overfertilisation, the remaining N and P are of no benefit to receiving crops, but rather pose a threat to adjacent ecosystems. However, the effluent N as well as the potassium (K) in the effluent could serve as a valuable fertigation source, given the recurring drought in the growing season in recent years in Flanders.

A solution to polish N and P after biological treatment is to use free-floating aquatic plants, given their proven ability for nutrient removal (Harvey & Fox, 1973). Duckweed (*Lemna minor*) has been identified as suitable since it is (i) a valuable protein source and (ii) it has found recognition for its ability to contribute to wastewater treatment by incorporating excess nutrients into its plant body.

So far life cycle assessment (LCA) studies have been conducted on duckweed for biofuel production on municipal wastewater (Calicioglu et al, 2021) and different constructed wetland set-ups (e.g. Fuchs, et al. (2011), Corbella et al. 2017). However, to our knowledge no LCA study has been conducted that coupled duckweed production as a protein feed supplement, constructed wetlands and pig manure treatment. The present study aims to explore such combinations.

3.1.9.2 Materials & Methods

All analyses were performed using the openLCA software v.1.11. All background processes were modelled using the consequential ecoinvent database 3.7.1 (Wernet et al., 2016).

3.1.9.2.1 Goal & Scope

The goal of this life cycle assessment (LCA) study was to explore the environmental implications of different agricultural wastewater treatment pathways. The wastewater is the liquid fraction of pig manure after separation and biological treatment. The pathways are different combinations of land application, duckweed ponds and constructed wetlands. The functional unit (FU) in all cases was the treatment of 1 m^3 liquid fraction of pig manure after mechanical separation.

Figure 27 gives a description of the studied scenarios. In all scenarios, the studied system starts with the provision of the separated liquid fraction of pig manure. The treatment continues with a biological treatment step. After the biological treatment the effluent is either field applied (1 – light green), fed to a constructed wetland (2 - red), or mixed with the untreated liquid fraction and fed to a duckweed pond (3-5 – dark green, yellow, orange). Depending on the residence time of the effluent in the duckweed pond and management choices, the effluent either has to undergo additional polishing in a constructed wetland (3 - dark green), is directly field applied (4 – orange), or is discharged into natural water bodies (5 - yellow). For Scenarios 3 and 4, we made one further distinction based on the residence time of the biological effluent in the duckweed pond: We assume either a short residence time (a) or a medium residence time (b). Under (a), the duckweed is grown under optimal nutrient supply (nutrient concentration of N and P are never lower than 10% below the optimal). Under (b), more nutrients are taken up by the duckweed and the effluent stays in the pond until 50% below the optimum. Overall, it can be said that (a) focuses on duckweed production while Scenario 5 focuses on nutrient removal (here the effluent stays in the pond until it reaches discharge norms). (b) is located between the two extremes.

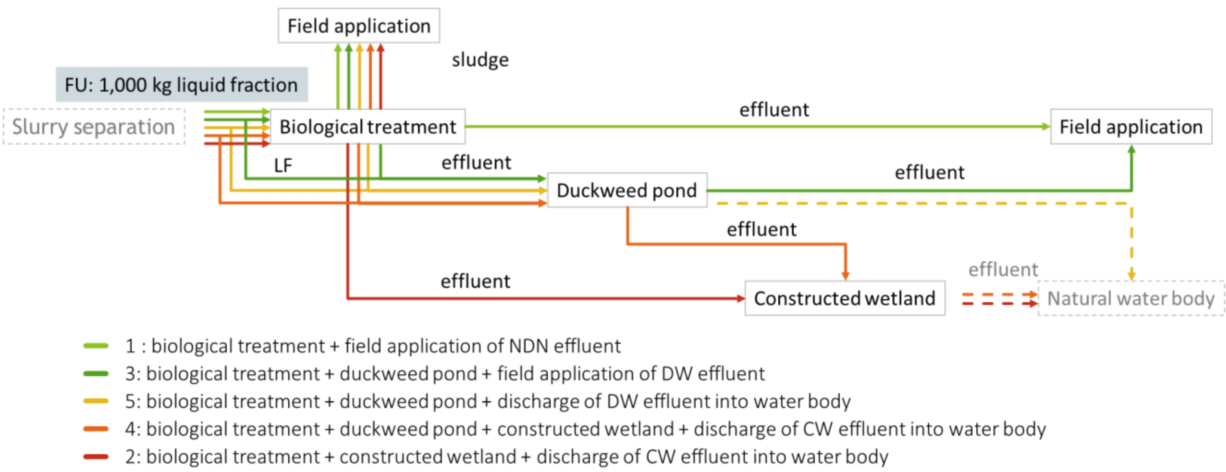


Figure 27. LL# 41: System boundaries

Scheme of the liquid pig manure treatment system under study. Boxes indicate activities in the foreground system. Arrows indicate exchanges – each colour represents one scenario. FU: functional unit (blue background). Dashed lines and grey font are activities outside the system boundary. Each colour represents one scenario.

3.1.9.2.3 Inventory

The data underlying this study were derived from experiments, literature data, mass balancing, and ecoinvent processes. In the following, a brief description of the three main stages and their inventories is given. Table 27 provides an overview of the inventory data.

Biological treatment

The inventory for the biological treatment includes infrastructure (Corbala-Robles *et al.*, 2018), the actual treatment, and sludge and effluent disposal. The mass balance of the biological treatment was performed using the STOAT model (Henze *et al.*, 2015). STOAT is a tool to model activated sludge systems and was used to represent the treatment of the liquid fraction of pig manure through a nitrification- denitrification pathway. The excess sludge is stored, transported, and field applied on P-deficient fields in France (credits for P and K fertiliser avoidance). The effluent is field applied in Flanders. We accounted for field emissions and gave credits for K fertiliser avoidance. Effluent discharge is viewed as disposal rather than fertiliser and receiving fields are typically overfertilised, therefore, no credits for P or N fertiliser were given (Lagerwerf *et al.*, 2019).

Duckweed pond

The inventory of the duckweed pond treatment step includes infrastructure (Calicioglu *et al.*, 2021), sludge disposal, and duckweed production. The mass balance was primarily based on experimental data provided by Inagro. Inagro performed rearing trials to assess the ability of duckweed to serve as polishing step in pig manure treatment. The experiments included indoor and outdoor trials and focused on water quality parameters as well as duckweed composition with regards to nutritional values. Descriptions of these studies can be found in Devlamynck *et al.* (2020, 2021a, 2021b) and in Nutri2Cycle deliverable report D2.1 (Luo *et al.* 2019). To achieve optimal N and P concentrations (N: 2.8 – 350 mg/l, P: 0.4 – 11 mg/l) for duckweed growth (Elias Landolt and Kandeler, 1987), the biological effluent is mixed with a proportion of untreated liquid fraction and rainwater before being fed into the duckweed pond. To represent different treatment and production strategies, we explored three different residence times and their influence on the overall environmental performance of the system:

Scenario 5 (long residence time): the mixture stays in the duckweed pond until the lower optimum concentrations of either N or P is reached

> → This requires for a larger area, but the 'maximum plant-available' N and P are taken up by duckweed. The resulting effluent does not require for additional treatment and can be discharged into natural water bodies.

Scenario 3a+4a (short residence time): the mixture stays in the duckweed pond until below 10% of the upper optimum concentration of either N or P is reached

> → This requires for a smaller area and facilitates optimal growth conditions; but the effluent needs to either undergo a polishing step in a constructed wetland (3) or be field applied (4).

Scenario 3b+4b (intermediate residence time): the mixture stays in the duckweed pond until the middle between upper and lower optimum concentrations of either N or P is reached

> → This requires for a 'medium sized' duckweed pond and results in trade-offs on both ends: duckweed yields a lower than in (a) but the effluent water quality is better and duckweed

Table 27. LL#21 LCA inventory for the treatment of liquid fraction pig manure for the baseline scenarios.

Values refer to the functional unit of 1 m3 liquid fraction entering the system (adapted from Beyers et al. 2023).

	1 | NDN + F	2 | NDN + CW	3a | NDN + DW (short) + CW	3b | NDN + DW (medium) + CW	4a | NDN + DW (short) + F	4b | NDN + DW (medium) + F	5 | NDN + DW (long)
	• Nitrification-denitrification treatment • Field application of the NDN effluent	• Nitrification-denitrification treatment • Treatment of the NDN effluent in a constructed wetland	• Nitrification-denitrification treatment • Treatment of the NDN effluent in a duckweed pond for 4 days • Treatment of DW effluent in constructed wetlands	• Nitrification-denitrification treatment • Treatment of the NDN effluent in a duckweed pond for 19 days • Treatment of DW effluent in constructed wetlands	• Nitrification-denitrification treatment • Treatment of the NDN effluent in a duckweed pond for 4 days • Field application of DW effluent	• Nitrification-denitrification treatment • Treatment of the NDN effluent in a duckweed pond for 19 days • Field application of DW effluent	• Nitrification-denitrification treatment • Treatment of the NDN effluent in a duckweed pond for 31 days • Discharge of DW effluent into natural waterbody
INVENTORY							
Nitrification-denitrification treatment							
Infrastructure				derived from Corbala-Robles et al. (2018) & see SI Table A1			
Sludge treatment [kg]	60	60	54	54	54	54	54
Storage emissions							
Ammonia [kg]	0.01	0.01	0.009	0.009	0.009	0.009	0.009
Methane [kg]	0.33	0.33	0.297	0.297	0.297	0.297	0.297
Field application							
Sludge transportation to France [kg*km]	60*250	60*250	54*250	54*250	54*250	54*250	54*250
Field application [kg]	60	60	54	54	54	54	54
Avoidance of P fertiliser [kg]	-0.23	-0.23	-0.207	-0.207	-0.207	-0.207	-0.207
Avoidance of K fertiliser [kg]	-0.17	-0.17	-0.153	-0.153	-0.153	-0.153	-0.153

	1 | NDN + F	2 | NDN + CW	3a | NDN + DW (short) + CW	3b | NDN + DW (medium) + CW	4a | NDN + DW (short) + F	4b | NDN + DW (medium) + F	5 | NDN + DW (long)
Effluent field application	0.75	-	-	-	-	-	-
Spreading to field [m³]	0.75		-	-	-	-	-
Avoidance of K fertiliser [kg]	-4.3		-	-	-	-	-
Ammonia emissions [kg]	0.0002		-	-	-	-	-
N_2O emissions [kg]	0.0023		-	-	-	-	-
Nitrate leaching [kg]	0.12		-	-	-	-	-
Phosphate leaching [kg]	0.0001		-	-	-	-	-
Duckweed pond							
Infrastructure	-	-	derived from Calicioglu et al. (2021) & see SI Table A6				
Electricity consumption for harvesting [kWh]	-	-	0.04	0.90	0.04	0.90	3.30
Sludge treatment [kg]	-	-	31	153	31	153	254
Field application [m3]	-	-	0.31	0.153	0.31	0.153	0.254
Avoidance of K fertiliser [kg]	-	-	-0.26	-1.3	-0.26	-1.3	-2.2
Ammonia emissions [kg]	-	-	0.002	0.009	0.002	0.009	0.015
N_2O emissions [kg]	-	-	0.0009	0.005	0.0009	0.005	0.008
Nitrate leaching [kg]	-	-	0.01	0.065	0.01	0.065	0.1
Phosphate leaching [kg]	-	-	2.66E-05	0.0001	2.66E-05	0.0001	0.0002
Avoidance of protein feed [kg]	-	-	0.1	0.5	0.1	0.5	0.9
Effluent field application							
Spreading to field [m³]	-	-	-	-	9.4	9.4	-
Avoidance of K fertiliser [kg]	-	-	-	-	-3.3	-2.3	-
Ammonia emissions [kg]	-	-	-	-	0.02	0.006	-
N_2O emissions [kg]	-	-	-	-	0.009	0.004	-
Nitrate leaching [kg]	-	-	-	-	0.13	0.06	-
Phosphate leaching [kg]	-	-	-	-	0.0002	9.49E-05	-
Constructed wetland							

	1 | NDN + F	2 | NDN + CW	3a | NDN + DW (short) + CW	3b | NDN + DW (medium) + CW	4a | NDN + DW (short) + F	4b | NDN + DW (medium) + F	5 | NDN + DW (long)
Infrastructure	-	Corbella et al. (2017), Scenario S1 & see SI Table A10					-

yields are higher than in 5 but the water quality is worse. It can be considered a 'middle-way' solution.

The sludge from the duckweed ponds is handled the same way as the sludge from the biological treatment. The harvested duckweed is assumed to replace soybean meal based on its protein content, so no account was taken of differences in digestibility or amino acid composition.

Constructed wetland

The inventory for the constructed wetland treatment step includes infrastructure (Corbella *et al.*, 2017), mass balancing (Meers *et al.*, 2008), and emissions of CH_4 and N_2O (Aben et al., 2022).

3.1.9.3 Results

Scenario 1 (direct field application after the biological treatment) indicated the lowest overall (normalised and weighted) environmental impact. The highest environmental impact appears to result from scenario 2, where the biological treatment is followed by effluent treatment in a constructed wetland. Of the scenarios entailing duckweed treatment, scenario 4b indicated lowest environmental impacts. Here, an intermediate residence time in the duckweed pond is assumed and the effluent is applied to fields following the duckweed treatment.

Normalisation and weighting suggest that the impact categories *climate change*, *freshwater ecotoxicity*, and *minerals and metals resource use* are of greatest interest in most scenarios (Figure 28). Potential impacts on *climate change* mostly steam from emissions of N_2O emissions during the biological treatment. Savings in *freshwater ecotoxicity* and *minerals and metals resource use potential* in scenario 1 mostly result from the avoidance of potassium (K) fertiliser through the application of the biological effluent to agricultural fields and scenarios including duckweed from K fertiliser avoidance through the application of sludge (3 - 5) or effluent (3) from the duckweed pond. Beneficial results in *freshwater ecotoxicity* increase with residence time in the duckweed pond because more sludge can accumulate at the bottom of the pond. Instead of being discharged to a constructed wetland, where it is lost for further agricultural use, parts of the potassium can be retrieved and applied to fields where it serves as a K fertiliser. Environmental savings achieved through the avoidance of soy feed by replacing it with duckweed protein are minor and have little effect on the overall performance of the treatment.

Above analysis suggests that the avoidance of potassium fertiliser plays a major role in the environmental performance of liquid fraction pig manure treatment. The results for most relevant impact categories suggest that retrieving the remaining K from the duckweed effluent is advisable from an environmental point of view (Figure 29) and it seems advisable to utilise the duckweed pond effluent as fertigation source, whenever possible.

3.1.9.4 Discussion

These LCA results suggests that if farmers perceive and use the biological effluent as a K fertiliser, utilising the effluent in that manner outperforms the other treatment paths such as duckweed pond and/or constructed wetlands. Previous LCA studies have either disregarded the alternative handling of wastewater (Corbella et al. 2017) or disregarded the K fertiliser value of the biological effluent when field applied (Corbala-Robles et al. 2018). Current LCA points at the importance of including those.

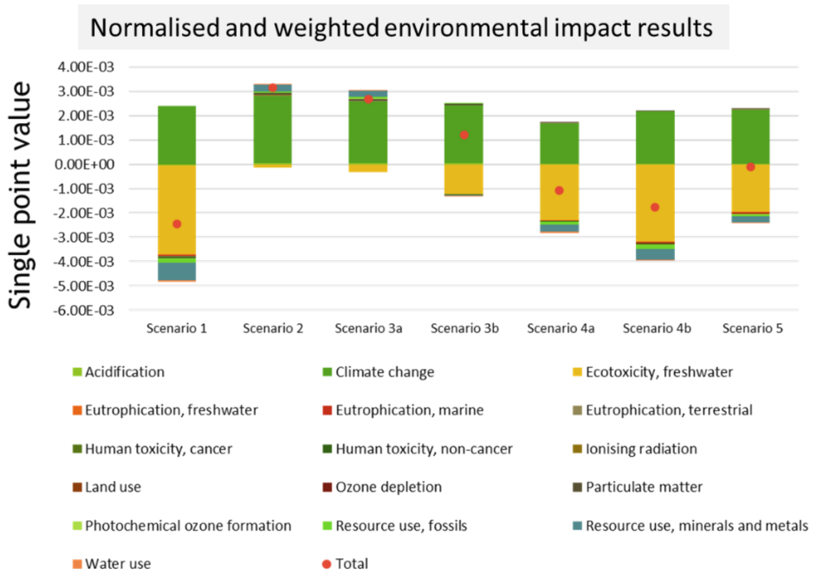


Figure 28. LL# 41: eLCA results - normalised and weighted

Normalised and weighted environmental impacts per function unit of 1 m3 liquid manure treatment. Scenario 1: biological treatment + field application; Scenario 2: biological treatment + constructed wetland; Scenario 3: biological treatment + duckweed pond + constructed wetland; Scenario 4: biological treatment + duckweed pond + field application; Scenario 5: biological treatment + duckweed pond. a: short residence time, b: medium residence time in the duckweed pond.

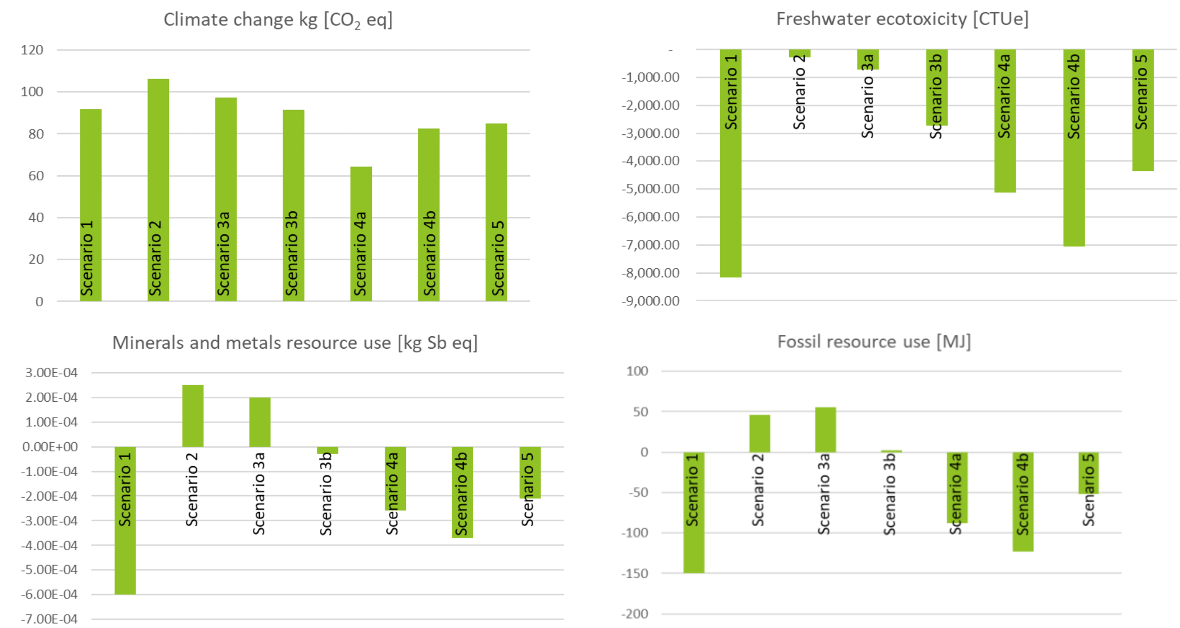


Figure 29. LL# 41: eLCA results

Future LCA studies could look deeper into the environmental impacts of K fertiliser production and at ways to recover potassium from other K-rich waste streams.

A study by Bayo et al. (2012), like ours, focused on intensive pig production in Murcia, Spain, concluding that direct field application is more environmentally favourable than a constructed wetland treatment. Unlike us, they didn't include an NDN treatment step, assuming the liquid fraction entered the wetland untreated. Despite similarities, key differences exist. They credited not only potassium (K) but also nitrogen (N) and phosphorus (P), assuming fertiliser credits post-constructed wetland treatment, unlike our assumption of effluent cleanliness for discharge. On climate change potential, their study favoured constructed wetland treatments, unlike our finding where constructed wetlands had greater impacts than direct field application. This difference may arise from our exclusion of fertiliser credits after the wetland treatment and the greater emissions from the untreated liquid fraction compared to NDN effluent. They also didn't consider freshwater ecotoxicity.

3.1.9.5 Dashboard indicators

LL# 41 Floating wetland plants grown on liquid agro-residues as a new source of proteins (*: Qualitative DBI assessment: **+** *improvement (<-10%)*, **o** *no change (-10% - +10%)*, **-** *deterioration (>+10%))*

Table 28. LL# 41: Comparison of dashboard indicator and life cycle assessment results

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication*	Comment	Quantitative estimate per FU of 1,000 kg liquid fraction pig manure	LCA indication*	Re-evaluation of DBI irt. LCA results*
Use of Primary Resources	**Rock Phosphate** (Reduction in mineral phosphorus consumption)	+	Using waste streams, allows duckweed to use P that normally would be lost. No addition of rock phosphate is necessary and thus a reduction would be expected compared to conventional feed crops that are fertilised with rock phosphate.	Phosphorous, in ground [p] 4b: +0.13 kg (+13%) 5: +0.42 kg (+43%)	 - -	 4b: revised, more likely incr. 5: revised, more likely incr.
	Natural Gas (Reduction in natural gas consumption in mineral fertiliser production)	o	unknown	Gas, natural, in ground [p] 4b: +0.7 m^3 (+18%) 5: +2.19 m^3 (+56%)	 - -	 4b: revised, more likely incr. 5: revised, more likely incr.
	Oil	o	unknown	Oil, crude, in ground [p,q]	-	4b: revised, more likely incr.

[p] linked to K fertiliser production
[q] Transportation of sludge

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication*	Comment	Quantitative estimate per FU of 1,000 kg liquid fraction pig manure	LCA indication*	Re-evaluation of DBI irt. LCA results*
	(Reduction in oil consumption in agricultural machinery)			4b: +0.02 kg (+21%) 5: +0.28 kg (+83%)	-	5: revised, more likely incr.
				Diesel burnt in agricultural machinery 4b: -1.20E-04 MJ (-32%) 5: -3.20E-04 MJ (-86%)	+ +	4b: revised, more likely decr. 5: revised, more likely decr.
	Electricity (Reduction in electricity consumption)	o	unknown	Electricity consumption 4b: +13.3 MJ (+84%) 5: +20.07 MJ (>100%)	- -	4b: revised, more likely incr. 5: revised, more likely incr.
	Water (Reduction in water consumption)	o	unknown	Water scarcity [p] 4b: 1.3 m^3 depriv. (+14%) 5: 4.35 m^3 depriv. (+48%)	- -	4b: revised, more likely incr. 5: revised, more likely incr.
	Soil quality (Improvement in soil quality)	o	unknown	Land use [r] 4b: -121.69 Pt (>-100%) 5: -189.57 Pt (>-100%)	+ +	4b: revised, more likely decr. 5: revised, more likely decr.
	Nutrient recovery (Increase in nutrient recycling)	+	NitrogenUseEfficiency can be around of 85% kg N/kg N applied (Guo et al. 2020). However, 28-96% is also reported Mohedano depending	4b N fertiliser -5.74E-04 kg (>-100%) P fertiliser: +0.02 kg (+7%)	+ o -	4b N: confirmed P: revised, more likely negligible K: revised, more likely incr.

[r] Soy replacement by duckweed

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication*	Comment	Quantitative estimate per FU of 1,000 kg liquid fraction pig manure	LCA indication*	Re-evaluation of DBI irt. LCA results*
			on the loading rate. the more n applied, the lower the relative uptake of duckweed and the higher the relative removal by bacteria.	K fertiliser: +0.61 kg (+14%) 5 N fertiliser: -157.13 kg (>-100%) P fertiliser: +0.01 (+4%) K fertiliser: +2.1 (+48%)	+ o -	5 N: confirmed P: revised, more likely negligible K: revised, more likely incr.
	Others? Please specify Lime for soil liming and sulphur for producing H_2SO_4			no further info		
Emissions to the environ-ment	**Ammonia (air emission)** (Reduction in NH_3 emissions)	0	Ammonia volatilsation is possible, but at neutral pH this does not occur (Körner, 2003). (<1.5% Zimmo et al. 2003) Importance of pH	Ammonia emission to air 4b: +0.02 kg NH_3 (>+100%) 5: +0.02 kg NH_3 (>+100%) Ammonia, BE[s] 4b: +0.01 kg NH_3 (>+100%) 5: +0.01 kg NH_3 (>+100%)	- - - -	4b: revised, more likely incr. 5: revised, more likely incr. 4b: revised, more likely incr. 5: revised, more likely incr.
	Dinitrogen monoxide (air emission) (Reduction in N_2O Emissions)	-	(0.63 - 4 mg $N_2O/m^2/d$) sims et al 2013	Dinitrogen monoxide emission to air 4b: -0.02 kg N_2O (-9%) 5: -0.02 kg N_2O (-9%)	o o	P: revised, more likely negligible P: revised, more likely negligible

[s] Treatment emissions in the foreground system

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication*	Comment	Quantitative estimate per FU of 1,000 kg liquid fraction pig manure	LCA indication*	Re-evaluation of DBI irt. LCA results*
	Methane (air emission) (Reduction in CH_4 emissions)	0	502 - 1900 mg CH4/m²/d sims et al 2013	Methane emission to air [t] 4b: -0.03 kg CH_4 (-11%) 5: -0.03 kg CH_4 (-11%)	 + +	 4b: revised, more likely decr. 5: revised, more likely decr.
	Nitrates (water emission) (Reduction in NO_3 emissions)	+	Using a plastic foil to cover the bottom of the pond prevents any leaching of nutrients	Nitrate emission to water [p,u] 4b: -0.03 kg NO_3 (>-100%) 5: +0.05 kg NO_3 (>-100%)	 + +	 4b: confirmed 5: confirmed
	Phosphorus (water emission) (Reduction of P Emissions)	+	Using a plastic foil to cover the bottom of the pond prevents any leaching of nutrients	Phosphorous emission to water [p,r] 4b: -2.48E-04 kg P (-55%) 5: -3.65E-04 P (-81%) Phosphate emission to water [p,v] 4b: +9.39E-04 kg PO_4^{3-} (+16%) 5: +3.42E-03 kg PO_4^{3-} (+58%)	 + + - -	 4b: confirmed 5: confirmed 4b: revised, more likely incr. 5: revised, more likely incr.
	Particulate matter	0	unknown	Particulates emission to air		

[t] construction of biological treatment plant & emissions from biological sludge storage in the foreground system
[u] to groundwater: soy replacement
[v] linked to P fertiliser production

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment			Quantitative Dashboard Indicator based on LCA assessment		
		DBI Indication*	Comment	Quantitative estimate per FU of 1,000 kg liquid fraction pig manure	LCA indication*	Re-evaluation of DBI irt. LCA results*
	(Reduction of particulate matter formation)			4b: -7.02E-04 kg fPM (-5%) 5: +4.19E-03 kg fPM (+31%)	**o** **-**	4b: confirmed 5: revised, more likely incr.
	Others? Please specify					
Resilience to climate change	**Carbon footprint** (Reduction of carbon footprint)	**0**	unknown	Climate change [CO_2 eq.][w] 4b: -9.10 (-11%) 5: -6.74 (-8%)	 **+** **o**	 4B: revised, more likely decr. 5: confirmed
	Effective SOM (Effective Soil Organic Matter Improvement)	**0**	unknown	Carbon | Emission from soil [p] 4b: 8.26E-06 kg C (+6%) 5: 8.64E-05 kg C (+62%) Carbon dioxide | Emission from soil [r] 4b: -2.52 kg (>-100%) 5: -4.52 kg (>-100%)	 **o** **o** **+** **+**	 4b: confirmed 5: confirmed 4b: revised, more likely decr. 5: revised, more likely decr.

[w] N2O emissions during biological treatment

3.1.9.5.1 Comparison of LCA and DBI results

With regards to the consumption of primary resources, the LCA concluded that the introduction of a duckweed pond would result in increased rock phosphate mining, while the DBI suggested a decrease. In the DBI the argument is that the duckweed takes up P that would have been lost (or of no use) otherwise. While this is true, the DBI failed to properly account for the missed opportunity of using the effluent as K fertiliser and thus avoiding its production. Including related processes result in greater P mining in the duckweed pond scenarios than in the baseline, where the effluent of the biological treatment is field applied.

In terms of natural gas and oil consumption, the DBI could not provide an evaluation. The LCA concluded an increase in natural gas and oil consumption due to the need for K fertiliser. The consumption of 'diesel burnt in agricultural machinery' could be decreased through the introduction of a duckweed pond potentially due the decreased demand for soy.

While the DBI could not inform about electricity or water use and soil quality changes, the LCA found both uses to increase and soil quality to increase, too. The increase in soil quality is due to the decreased demand for soy imports.

In terms of nutrient recovery both assessments agreed on reduced needs for N fertiliser.

Regarding emissions it is a mixed picture. While both assessments agree on reduced nitrate and phosphorous emissions to water, the LCA also concluded a reduction in methane emissions (due to reduced sludge treatment after NDN as some of the liquid fraction is lead to the duckweed pond without being treated biologically) which were deemed negligible in the DBI assessment. However, in terms of ammonia emissions the DBI suggested no change, but the LCA suggested an increase (related to the increased need for fertilisers in the treatment scenarios). Emissions on N_2O were predicted to increase in the DBI assessment but are suggested to change very little in the LCA. Emissions related to CO_2 are assumed unaffected in both studies.

3.1.9.6 References

Aben, R. C. H., Oliveira Junior, E. S., Carlos, A. R., van Bergen, T. J. H. M., Lamers, L. P. M., & Kosten, S. (2022). Impact of plant species and intense nutrient loading on CH4 and N2O fluxes from small inland waters: An experimental approach. *Aquatic Botany*, *180*, 103527. https://doi.org/10.1016/j.aquabot.2022.103527

Bayo, J., Gómez-López, M.D., Faz, A., Caballero, A., 2012. Environmental assessment of pig slurry management after local characterization and normalization. *J. Clean. Prod*. *32*, 227–235. https://doi.org/10.1016/j.jclepro.2012.04.003

Beyers M., Ravi R., Devlamynck R., Jensen L.S., Meers E., Bruun S. (2023) Constructed wetlands and duckweed ponds as a treatment step in liquid manure handling — A life cycle assessment. *Sci. Total Environ.* 889, https://doi.org/10.1016/j.scitotenv.2023.163956

Calicioglu, O., Femeena, P. V., Mutel, C. L., Sills, D. L., Richard, T. L., & Brennan, R. A. (2021). Techno-economic Analysis and Life Cycle Assessment of an Integrated Wastewater-Derived Duckweed Biorefinery. *ACS Sustainable Chemistry and Engineering*, *9*(28), 9395–9408. https://doi.org/10.1021/acssuschemeng.1c02539

Corbala-Robles, L., Sastafiana, W. N. D., Van linden, V., Volcke, E. I. P., & Schaubroeck, T. (2018). Life cycle assessment of biological pig manure treatment versus direct land application − a trade-off story. *Resources, Conservation and Recycling*, *131*, 86–98. https://doi.org/10.1016/j.resconrec.2017.12.010

Corbella, C., Puigagut, J., & Garfí, M. (2017). Life cycle assessment of constructed wetland systems for wastewater treatment coupled with microbial fuel cells. *Science of the Total Environment*, *584–585*, 355–362. https://doi.org/10.1016/j.scitotenv.2016.12.186

Devlamynck, R., de Souza, M. F., Leenknegt, J., Jacxsens, L., Eeckhout, M., & Meers, E. (2021a). Lemna minor cultivation for treating swine manure and providing micronutrients for animal feed. *Plants*, *10*(6). https://doi.org/10.3390/plants10061124

Devlamynck, R., de Souza, M. F., Michels, E., Sigurnjak, I., Donoso, N., Coudron, C., Leenknegt, J., Vermeir, P., Eeckhout, M., & Meers, E. (2021b). Agronomic and Environmental Performance of Lemnaminor Cultivated on Agricultural Wastewater Streams—A Practical Approach. *Sustainability*, *13*(3), 1570. https://doi.org/10.3390/su13031570

Devlamynck, R., Fernandes de Souza, M., Bog, M., Leenknegt, J., Eeckhout, M., & Meers, E. (2020). Effect of the growth medium composition on nitrate accumulation in the novel protein crop Lemna minor. *Ecotoxicology and Environmental Safety*, *206*, 111380. https://doi.org/10.1016/j.ecoenv.2020.111380

Elias Landolt, & Kandeler, R. (1987). *The Family of Lemnaceae: A Monographic Study. Vol. 2*. Geobotanischen Institutes Der ETH. https://books.google.dk/books/about/The_Family_of_Lemnaceae.html?id=oUf9mQEACAAJ&redir_esc=y

Fuchs, V. J., Mihelcic, J. R., & Gierke, J. S. (2011). Life cycle assessment of vertical and horizontal flow constructed wetlands for wastewater treatment considering nitrogen and carbon greenhouse gas emissions. *Water Research*, *45*(5), 2073–2081. https://doi.org/10.1016/j.watres.2010.12.021

Harvey, R. M., Fox, J. L. (1973) Nutrient removal using Lemna minor. Water Pollution Control Federation. Vol. 45, No. 9

Henze, M., Gujer, W., Mino, T., & van Loosedrecht, M. (2015). Activated Sludge Models ASM1, ASM2, ASM2d and ASM3. *Water Intelligence Online*, *5*(0), ISBN: 9781780402369. https://doi.org/10.2166/9781780402369

Lagerwerf, L. A., Bannink, A., van Bruggen, C., Groenestein, C. M., Huijsmans, J. F. M., van der Kolk, J. W. H., Luesink, H. H., van der Sluis, S. M., Velthof, G. L., & Vonk, J. (2019). *Methodology for estimating emissions from agriculture in the Netherlands : Calculations of CH4, NH3, N2O, NOx, NMVOC, PM10, PM2.5 and CO2 with the National Emission Model for Agriculture (NEMA), Update 2019*. Statutory Research Tasks Unit for Nature & the Environment. https://doi.org/10.18174/472366

Luo, H., Michels, E., Meers, E. (2019). *Longlist of innovations & compilation of factsheets*. Nutri2Cycle report D.2.1.

Meers, E., Tack, F. M. G., Tolpe, I., & Michels, E. (2008). Application of a full-scale constructed wetland for tertiary treatment of piggery manure: Monitoring results. *Water, Air, and Soil Pollution*, *193*(1–4), 15–24. https://doi.org/10.1007/s11270-008-9664-5

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., Weidema, B., 2016. The ecoinvent database version 3 (part I): overview and methodology. *Int. J. Life Cycle Assess*. 21, 1218–1230. https://doi.org/10.1007/s11367-016-1087-8

3.1.10 LL#20: Replacement of mineral fertilisers with ammonia retrieved through by stripping – Scrubbing (IRTA)

Longlist #20 title: Low temperature ammonia-stripping using vacuum

Miriam Cerrillo, Edilene Pereira, Ariadna Ballega, Erica Montemayor, Assumpció Antón, August Bonmatí

Sustainability in Biosystems. IRTA, Torre Marimon, Caldes de Montbui, Barcelona, Spain

3.1.10.1 Introduction

Livestock manure requires correct management due to its valuable organic matter and nutrient content, especially phosphorus (P) and nitrogen (N). Ammonia recovery from livestock manure can produce marketable products, such as fertilisers, whilst allowing for nitrogen loop closure. The present demo investigation develops a low temperature vacuum evaporation system for the recovery of ammonia from livestock manure, in the form of a salt solution that can be used as a fertiliser.

When a vacuum is applied to an enclosed reactor, the ammonia boiling point temperature decreases. This reduces energy costs because of lower heating requirements. In addition, gas-phase ammonia mass transfer is boosted by the suction effect of the applied vacuum. The evaporated ammonia can be recovered in the form of an ammonium sulphate, nitrate or lactate salt solution, among others. Up-to-date, lab-scale and small pilot plants (up to 40 L) have been tested using anaerobic digester effluent, livestock manure or urine (Tao et al., 2018).

The N recovery plant evaluated here belongs to the category of low temperature vacuum ammonium-evaporation solutions and is included under sub-research line 7: “pig manure processing and replacing mineral fertilisers” (see deliverable 2.6).

3.1.10.2 Material & methods

3.1.10.2.1 Goal & scope

Intended application: Assessment of the environmental impact of the low temperature vacuum ammonium-evaporation technology that recovers ammonia from livestock slurry, resulting in an ammonia salt which can be reused as a fertiliser.

LCA approach: attributional

Target audience: Nutri2Cycle partners; local administration and farmers; scientific community

Limitations of study: use of secondary datasets; use of average data from slurry treatment installations; use of default emission factors and impact methods not fully developed.

Are results intended to be used in comparative assertions and disclosed to the public? Assessment is conducted to obtain a snapshot of the current technology. It could be compared with other manure treatments, if methods, criteria, system boundaries and models applied in the assessment are the same.

Commissioner of the study and other influential actors: European Commission through N2C project
Identification of the product system to be studied: slurry treated.

Location: Viver i Serrateix, Catalonia (Spain).

Time period: Data from technology tested correspond to experimental trials conducted during 2020-2021

The functions of the product system/s: livestock slurry treatment

The functional unit(s): treatment of 1 m^3 of raw pig slurry

The system boundaries: From farm gate to treatment plant gate. Inclusion of inputs and energy required during slurry treatment. Fertiliser application has been excluded because we do not have information on specific emissions depending on the type of fertiliser. Applying the default emission factors, recommended by the Product Environmental Footprint guidelines, would not result in differences in field emissions as far as we are dealing with 1 kg of N, no matter the origin. Figure 30 shows the scheme of the system and flows, inputs and outputs considered in the low temperature vacuum ammonium-stripping technology.

Allocation procedures: Following ISO 14040 (2006, 2020), we have applied biophysical criteria. Two outputs are obtained from the solid-liquid separator: solid and liquid fractions. Inputs and emissions are allocated based on the volume obtained, thus, allocating 96% to the liquid fraction and 4% to the solid fraction. An ammonia-evaporation treatment is applied to the liquid fraction and two more products are obtained: lactate and treated pig slurry liquid fraction with reduced N (reduction of 60%). Allocation between these two co-products has been performed based on N content of lactate (24%) and treated slurry (76%) (Table 31). As shown in Figure 30, a 1st co-product was obtained (solid fraction), and after the solid-liquid separator a 2nd coproduct was obtained (treated liquid fractions). After the evaporator the final product, lactate, was attained. Figure 30 also shows the amount obtained from 1 m^3 of raw slurry treated.

Impact categories to be covered and methodology of impact assessment: EF 3.0 Method (adapted) V1.00 (Fazio et al 2018). Normalisation and weighting have not been performed because accordingly to ISO 14044 (2006, 2020), it is not recommended to apply these steps in LCA studies intended to be used in comparative assertions.

Data requirements: Primary data comes from IRTA experiments at pilot trial; Secondary data: ecoinvent + agribalyse databases. Secondary data has been adapted to Spanish conditions if needed.

Assumptions: The organic fertilisers themselves are considered without burdens (which have been attributed to the pig farm that generates the waste). Only the treatment applied to the slurry has an associated environmental burden. Any treatment as well as transport to agricultural fields is included.

Specifications: Primary data corresponds to a pilot plant, where emissions have been measured, showing a high variability, therefore a sensitivity assessment has been conducted providing minimum and maximum values. Table 30 shows emission values considered.

Alternative scenarios: Baseline scenario considered is manure without treatment with an average content of 4.6 kg N/ton and synthetic fertilisers.

Type of critical review, if any: Internal review by N2C partners.

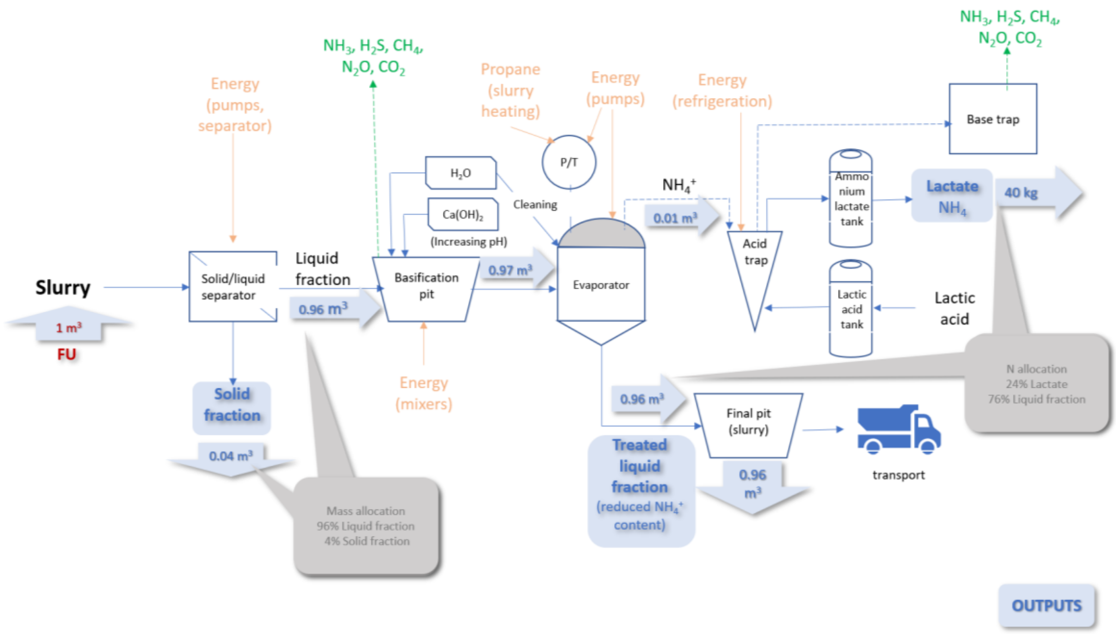


Figure 30. LL# 20: System boundaries

System diagram detailing the low temperature vacuum ammonium-stripping technology, including the allocation criteria considered.

3.1.10.2.2 Inventory

Data collection included general information about the activity and the specific inputs used in the pilot plant. These inputs have been split into infrastructures, and materials and energy (e.g., electricity, heat) (

Table 29). For secondary data, we used ecoinvent 3.6 (Wernet et al 2016) and agribalyse (2017) databases, as well as bibliographic sources. Software used was Simapro 9.1.1.7 (PRéConsultants, 2020). Emissions of NH_3, H_2S, CH_4, N_2O, CO_2 were measured at the basification pit and base trap (Figure 30 and Table 30).

Table 31 shows the nutrient content of the three outputs obtained, i.e., solid fraction, liquid fraction, and lactate with ammonium.

Table 29. LL# 20: Life cycle Inventory

Inventory corresponding to low temperature ammonium-stripping.

Inputs	material	amount units
Pig slurry		1 m^3
Infrastructure		4.2E-05 p/m^3 slurry
Lactic acid	Lactic acid 80%	6,4 kg/m^3 slurry
Calcium hydroxide	Calcium hydroxide, $Ca(OH)_2$	1 kg/m^3 slurry
Water	H_2O, for calcium hydroxide solution	0.02 m^3/m^3 slurry
Water	H_2O, for cleaning	0.43 m^3/m^3 slurry
Electrical power	Pumps	3 kWh/m^3 slurry
Propane	Water heating	3 m^3/m^3 slurry
Outputs transport	Average SP lorry transport	20 km

Outputs	
Lactate with N	0.04 tn/m^3 slurry
Pig slurry liquid fraction with reduced N (reduction of 60%)	0.96 tn/m^3 slurry
Pig slurry solid fraction	0.04 tn/m^3 slurry

Table 30. LL# 20: Measured emissions at basification pit and basic trap.

	N-NH_3	S-H_2S	N-N_2O	C-CH_4	C-CO_2
Basification pit			g/m^3 slurry		
Minim	0.03	0	0.006	0.02	1.66
Maxim	0.27	0	0.009	2.7	2.60
Average	0.14	0.002	0.007	0.92	1.90
Basic trap			g/m^3 slurry		
Minim	0.01	0	0.01	0.01	0.96
Maxim	0.07	0.3	0.05	18.1	18.6
Average	0.03	0.1	0.04	14.3	11.9

Table 31. LL# 20: Nutrient content of the three fertilisers obtained

	NTK	N-NH_4^+	SO_4^{-3}	K^+	P_{total}
	kg/m^3	kg/m^3	kg/m^3	kg/m^3	kg/m^3
Solid fraction	7.38	3.14	0.22	0.21	1.21
Liquid fraction, basification	1.16	0.89	0.06	0.83	0.01
Lactate	12.29	12.16	0.47	0.06	

3.1.10.3 Results

3.1.10.3.1 Impact assessment

Table 32 presents the results for the different impact categories for 1 m^3 of treated raw slurry. In addition, we assume that the use of these nutrients as a fertiliser would mean an avoided impact for mineral fertilisation, which we have added separately in Table 32. Avoided nutrients were calculated considering the N obtained from the different co-products acquired. According to the amount of the different outputs (

Table 29) and corresponding to the N content (Table 31). After treatment of 1 m^3 of raw slurry, we obtained fertilisers with a total amount 1.90 kg N, 0.266 kg P_2O_5 and 0.971 kg K_2O. The amount of mineral fertiliser that would be avoided was calculated based on the defined Mineral Nitrogen Equivalence of 0.5 kg of N from mineral fertiliser for each kg of N from slurry fertiliser. For P_2O_5 and K_2O we have kept an equivalent ratio of 1.

Table 32. LL# 20: eLCA results

Impact category results for 1 m3 of raw slurry treated through ammonia stripping-scrubbing technology. Results are expressed in relation to kg N obtained. In addition, impact results for avoided mineral fertilisers substitution are added.

		LL#20	Avoided impacts		
Impact category	Units	AVG /1 m^3 raw slurry	N *[/0.95 kg N]*	P_2O_5 */0.266 kg P_2O_5*	K_2O */0.971 kg K_2O*
Climate change	kg CO_2 eq	**3.50E+01**	2.65E+00	7.49E-01	8.18E-01
Ozone depletion	kg CFC11 eq	**4.49E-06**	3.58E-07	9.05E-08	1.34E-07
Ionising radiation	kBq U-235 eq	**2.88E+00**	1.63E-01	8.75E-02	6.70E-02
Photochemical ozone formation	kg NMVOC eq	**1.01E-01**	5.72E-03	3.40E-03	3.47E-03

Impact category	Units	LL#20 AVG /1 m^3 raw slurry	Avoided impacts N /0.95 kg N	P₂O₅ /0.266 kg P_2O_5	K_2O /0.971 kg K_2O
Particulate matter	disease inc.	**1.54E-06**	1.34E-07	7.51E-08	8.73E-08
Human toxicity, non-cancer	CTUh	**7.75E-07**	3.62E-08	1.92E-08	5.25E-08
Human toxicity, cancer	CTUh	**5.35E-08**	1.71E-09	7.91E-10	7.10E-10
Acidification	mol H+ eq	**1.43E-01**	2.11E-02	1.49E-02	6.17E-03
Eutrophication, freshwater	kg P eq	**7.10E-03**	4.81E-04	2.71E-04	1.84E-04
Eutrophication, marine	kg N eq	**3.52E-02**	2.42E-03	9.83E-04	1.09E-03
Eutrophication, terrestrial	mol N eq	**3.18E-01**	4.80E-02	1.38E-02	1.43E-02
Ecotoxicity, freshwater	CTUe	**5.35E+02**	6.36E+01	1.28E+01	1.21E+03
Land use	Pt	**1.33E+02**	1.15E+01	1.09E+01	1.05E+01
Water use	m^3 depriv.	**2.89E+01**	2.64E+00	7.48E-01	3.94E-01
Resource use, fossils	MJ	**5.63E+02**	4.44E+01	1.12E+01	1.27E+01
Resource use, minerals & metals	kg Sb eq	**8.09E-04**	5.81E-05	1.79E-05	2.02E-05
Climate change – Fossil	kg CO_2 eq	**3.46E+01**	2.65E+00	7.44E-01	8.17E-01
Climate change – Biogenic	kg CO_2 eq	**4.16E-01**	2.62E-03	1.01E-03	1.05E-03
Climate change – Land use & change	kg CO_2 eq	**1.87E-02**	2.64E-03	4.37E-03	6.72E-04

3.1.10.4 Interpretation

The results show the impacts associated with the slurry treatment applying the low temperature ammonia-stripping technology and the comparison to avoided impacts from the avoidance of chemical fertilisers as a consequence of the nutrients recovered.

To see the major contributions of the different life stages of the technology, Figure 31 provides contribution percentages for the different impact categories of the different processes involved in the ammonia stripping treatment. Acid trap and treated liquid fraction transport contribute with more than 80% for most of the impact categories. The former is explained by the use of lactic acid and the later by the relatively large volume to be transported. Clear exceptions were for: i) *human toxicity, cancer category* for which steel used in infrastructure shows the major contribution and ii) *water consumption* impact category, where water consumed in the evaporator represents the major contribution. Measured emissions show a minor contribution, therefore, the sensitivity assessment conducted on measured emissions was not significant for the total results.

On the other hand, if we establish "the use of mineral fertilisers" as reference scenario and compare it with the current technology, we observe that the impact potential is increasing as a consequence of the technology for most of the environmental categories. The only exception is *ecotoxicity freshwater*, for which the production of potassium fertilisers presents a major contribution. Several aspects needs to be pointed out to understand the results correctly: i) Different assumptions for Mineral Nutrient Equivalence may change quantitative results, but not the tendency of the superior environmental efficiency of mineral fertilisers; ii) It can be observed that, for those categories related to NH_3 emissions such as *acidification*, *eutrophication* and *particulate matter*, the ratio between treatment technology and mineral fertilisers is lower than for the rest; iii) The rest of the categories are especially affected by the infrastructure and inputs needed to conduct the treatment, showing that mineral fertiliser production is more efficient; iv) the reader needs to be aware that comparing the current technology with mineral fertilisers is a simplification of the reality.

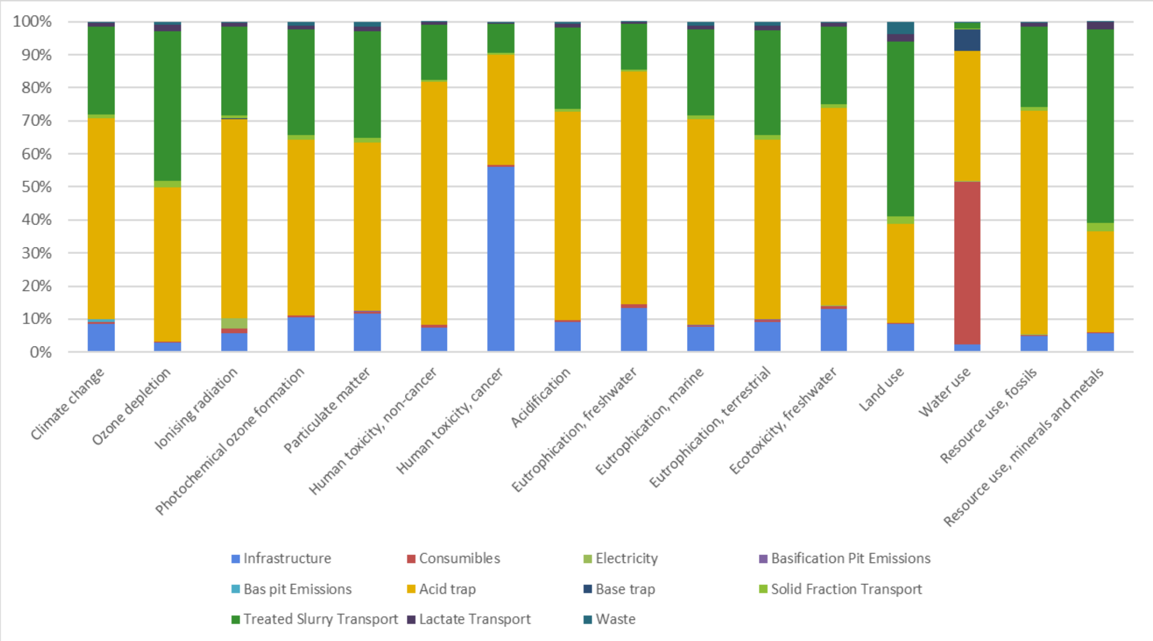


Figure 31. LL# 20: eLCA results - impact contributions

Contribution percentages for the different impact categories of the different processes involved in the ammonia stripping treatment of 1 m^3 of pig slurry

3.1.10.5 Conclusion

This study has assessed the technology of low temperature ammonium-stripping using a vacuum to treat pig slurry. The results suggest that slurry treatment may reduce the environmental impact in those categories mostly related to an overall enhanced nutrient recycling and that it has a potential to contribute to a reduction in ammonia emissions. In addition, our study also showed, which aspects should be improved. These are related to infrastructure and other inputs used such as lactic acid.

3.1.10.6 Dashboard indicators

Dashboard indicators were applied, taking synthetic fertilisers as a reference, so following the LCA conducted, we compared qualitative and quantitative results of 1 m^3 treated raw slurry according to the Low temperature ammonium stripping process using vacuum technology with the corresponding amount of avoided synthetic fertilisation, 0.95 kg N, 0.266 kg P_2O_5 and 0.971 kg K_2O.

LL#20 Low temperature ammonium stripping process using vacuum (*: Qualitative DBI assessment: "+" means improvement (<-10%), "o" means no change (-10% - +10%), "-" means deterioration (>+10%))

REF: reference scenario synthetic fertilisation; TEC: ammonium stripping technology

Table 33. LL# 20: Comparison of dashboard indicator and life cycle assessment results

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment	DBI Indication*	Comment	Quantitative Dashboard Indicator based on LCA assessment: Quantitative estimate per FU of 1 m^3 raw slurry	LCA indication*	Re-evaluation of DBI. LCA results*
Use of Primary Resources	**Rock Phosphate** (Reduction in mineral phosphorus consumption)	+	Mineral Fertiliser replacement	**Phosphorous, in ground** TEC: -636.4 g P (-99%)	+	**Confirmed**, there is a mineral fertiliser replacement, which reduces resources of Rock Phosphate
	Natural Gas (Reduction in natural gas consumption in mineral fertiliser production)	+	Mineral Fertiliser replacement	**Gas, natural, in ground** TEC:6.83 m^3 (>100%)	-	**Revised**, there are other processes involved, such as the electricity use, which involve Natural gas inputs and means an increase of its consumption
	Oil (Reduction in oil consumption in agricultural machinery)	o	Negligible	**Oil, crude, in ground** TEC: 30.3 kg (>100%)	-	**Revised**, again processes involved in the treatment seems to be less efficient than mineral fertilisers production
	Electricity (Reduction in electricity consumption)	-	Energy consumption of the treatment process	**Electricity consumption** TEC: 3 kWh + backg. proc – REF	n.d.	Not determined (n.d.), since It is not possible to determine the life cycle electricity consumption for mineral fertilisers, nor for background processes of new technology
	Water (Reduction in water consumption)	o	Negligible	**Water scarcity** TEC: Not available	n.d.	Idem
	Soil quality (Improvement in soil quality)	o	Negligible	**Soil quality**	n.d.	Not available in this LCA
	Nutrient recovery	+	Valorisation of manure as fertiliser	**N fertiliser**[d] -0.95 kg N (> -100%)	+	**Confirmed**

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment	DBI Indication*	Comment	Quantitative Dashboard Indicator based on LCA assessment Quantitative estimate per FU of 1 m^3 raw slurry	LCA indication*	Re-evaluation of DBI. LCA results*
	(Increase in nutrient recycling)			**P fertiliser**: 0.266 kg P_2O_5 (> -100%) **K fertiliser:** 0.971 kg K_2O (> -100%)		
Emissions to the environ-ment	**Ammonia (air emission)** (Reduction in NH_3 emissions)	+	N recovery and use as fertiliser in non-volatile form	**Ammonia emission to air** TEC: 0.21 g NH_3 (-88%)	+	Confirmed
	Dinitrogen monoxide (air emission) (Reduction in N_2O Emissions)	o	Unknown	**Dinitrogen monoxide emission to air** TEC: -0.72 g N_2O (-90%)	+	**Revised**, because of the inclusion of background processes
	Methane (air emission) (Reduction in CH_4 emissions)	o	Negligible	**Methane emission to air** TEC: 20.2 g CH_4 (>100%)	-	**Revised** considering background processes
	Nitrates (water emission) (Reduction in NO_3 emissions)	+	N recovery	**Nitrate emission to water**		N/A (And not relevant as far as field application is not considered
	Phosphorus (water emission) (Reduction of P Emissions)	+	P recovery	**Phosphorous emission to water**		Idem
	Particulate matter (Reduction of particulate matter formation)	o		**Particulates emission to air**	-	**Revised,** although technology means a reduction of ammonia emissions, which are precursor of PM formation, datasets from background

Indicator Dimension	Qualitative Dashboard Indicator (DBI) assessment based on expert judgment	DBI Indication*	Comment	Quantitative Dashboard Indicator based on LCA assessment Quantitative estimate per FU of 1 m^3 raw slurry	LCA indication*	Re-evaluation of DBI. LCA results*
				TEC: $1.23 \cdot 10^{-6}$ disease inc. (>100%)		processes report higher quantity PM emissions. On the other hand, during trial experiments PM emissions have not been estimated.
Resilience to climate change	**Carbon footprint** (Reduction of carbon footprint)	**+**	Increased energy demand but will contribute to reduction of transport amount and distance and substitution of mineral fertilisers	**Climate change [CO_2 eq.]** TEC: 30.8 kg CO_2 eq (>100%)	**-**	**Revised**, there is an increase in energy demand and treated slurry results with a major volume per Nitrogen unit to be transported, especially in relation to treated liquid fraction in reference to synthetic fertilisers
	Effective SOM (Effective Soil Organic Matter Improvement)	**o**		**Effective SOM**	**o**	Unknown with current LCA methods

3.1.10.6.1 Comparison of LCA and DBI results

When we defined the DBI, we used mineral fertilisers as reference scenario. As previously mentioned, this is a simplification of reality. 1 kg of organic nutrients is not directly proportional to 1 kg on synthetic fertiliser, so for the quantitative LCA comparison we have considered 0.5 kg of Nitrogen synthetic fertiliser/kg nutrient of organic fertiliser. On the other hand, an equivalent ratio for P_2O_5 and K_2O were considered. Main differences between DBI and LCA results come from the inclusion of background processes. For instance, in terms of natural resource consumption, some aspects of the DBI results needs to be revised, particularly those in relation to natural gas consumption. The avoided use of natural gas because of the substitution of mineral fertiliser is not confirmed. As far as energy is needed to conduct the treatment, which was foreseen with the DBI, the use of natural gas increases because it is included in the electricity mix production.

In relation to nutrient recovery, the LCA results support the DBI results.

Regarding indicators related to the emissions to the environment, we found DBI and LCA results to agree mostly in relation to ammonium emissions, but not dinitrogen monoxide and methane. In fact, for the DBI, those emissions were considered unknown or negligible, respectively. The LCA has shown that the technology improves NH_3 and N_2O emissions and increases CH_4, due to the background processes involved. In relation to the leaching of N and P to water, assumptions made in the DBI should be corrected, because these emissions are mostly due to fertiliser application practices, more than the fertiliser itself.

In relation to Particulate matter, for which no change was assumed in DBI, LCA results show that background fertiliser manufacture processes report a higher quantity of particulate matter emissions. On the other hand, during trial experiments PM emissions were not estimated.

Regarding resilience to climate change, a major impact has been observed. This was, as expected, due to energy consumption.

No data on this technology, nor existing models in LCA are available that can verify the SOM increases.

Main differences between DBI and LCA come from the change of perspective, foreground system for the former and whole supply chain for the latter. This means that for processes involved in the foreground, such as ammonia emissions or nutrient recovery, DBI has foreseen improvements compared to the reference, but it is more difficult for the experts assessing the DBI for a technology to foresee changes in background processes. Finally, it should be noted that the datasets for background processes could be averages more or less adjusted to the specific scenarios. For instance, it is worth mentioning that the average N, P_2O_5, K_2O production in Spain was used as the reference for synthetic mineral fertilisers. Results may change depending on the specific fertiliser used for substitution.

3.1.10.7 References

Agribalyse (2017). Database v1.3. France: ADEME. Available at: www.ademe.fr/agribalyse-en

Fazio, S., Castellani, V., Sala, S., Schau, E. M., Secchi, M., Zampori, L., & Diaconu, E. (2018). Supporting information to the characterisation factors of recommended EF Life Cycle Impact Assessment method. In Supporting information to the characterisation factors of recommended EF Life Cycle Impact Assessment methods,. https://doi.org/10.2760/671368

ISO 14040 (2006). Environmental management — Life cycle assessment — Principles and framework.

ISO 14040 (2020). 2006/AMD1:2020 Environmental management — Life cycle assessment — Principles and framework — Amendment 1. https://www.iso.org/standard/76121.html

ISO 14044 (2006). Environmental management — Life cycle assessment — Requirements and guidelines. https://doi.org/10.5594/J09750

ISO 1044 (2020). 2006/AMD 2:2020 Environmental management — Life cycle assessment — Requirements and guidelines — Amendment 2. https://www.iso.org/standard/76122.html

PRéConsultants. 2020. LCA software SimaPro 9.1.1.7

Tao, W; Ukwuani, A.T.; Agyeman, F. (2018). Recovery of ammonia in anaerobic digestate using vacuum thermal stripping – acid absorption process: scale-up considerations. *Water Sci Technol*, 78 (4), 878–885

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., and Weidema, B., (2016) The ecoinvent database version 3 (part I): overview and methodology. *The International Journal of Life Cycle Assessment*, [online] 21(9), pp.1218–1230.

3.2 Environmental Indicator Assessments

3.2.1 LL#13: Application of sensor technology to assess crop N status in Hungary (SOLTUB)

Longlist #13 title: Sensor technology to assess crop N status

Zoltán Hajdu, SOLTUB Ltd., Hungary

3.2.1.1 Introduction

In modern agricultural engineering, sensor technologies, digital positioning, optical recognition systems and data visualisation are innovative technologies that are developed and provided by several multinational companies such as Yara, Claas, John Deere and Krone. Such technologies can contribute to reductions in fertiliser and pesticides application. The tractor-mounted N sensor technology is an example for these precision technologies. The sensor measures crop leaves' light reflectance (e.g. NDVI, normalised difference vegetation index) and indirectly approximates the crop nitrogen (N) status and hence fertiliser N requirements. The sensor itself cannot measure the level of N in the soil. However, together with a dedicated software, the sensor technology can provide an estimation since the colour intensity of crop leaves gives an indirect indication of the soil N status. By doing so, the sensor technology can establish a link between spatially distributed variables and appropriate farming practices such as fertilisation, herbicide, and pesticide application, and harvesting (1).

In this study, we conducted an indicator analysis of a tractor-mounted N sensor, the Yara N-Sensor(TM) and the NEXT GreenSeeker. After a few preliminary analyses it was decided that the environmental assessment of LL13 research line on sensor technologies will be performed with Dashboard Indicators as well as by a simpler environmental indicator study. The main reason for not conducting a full LCA for LL13 sensor technology is that given the scope of this study it was impossible to get reliable data from involved farmers for the technology-related environmental trade-offs of the sensor technology - for example data on material and energy costs for using the sensor technology. Another reason was that the difference between the two chemical fertilisation scenarios only relates to reduced fertiliser use. The technology thus provides some fertiliser savings, but there is very little disadvantages apart of a little computational power needed to run the sensors and do the calculations. The trade-offs of the technology are therefore most probably negligible, and anyway difficult to assess.

3.2.1.2 Materials & methods

3.2.1.2.1 Goal & scope

The goal of this environmental indicator assessment study was to compare the environmental performance of wheat production with and without sensor technology.

In the present study, we assessed the environmental impacts of tractor-mounted N sensor technology on the example of two sensor technologies: One provided by Yara (5) and one by John Deere (6). Both systems are based on measuring crops light reflectance, the NDVI (normalised difference vegetation index), with values between zero and one – zero meaning no leaf cover and one meaning complete coverage of the soil by leaves. Their usage and performance were assessed under Hungarian conditions on three sample farms, two of these have the same Yara N-sensor, while one has the GreenSeeker.

The system boundaries define which part of the wheat product life cycle plus associated processes and activities belong to the main product analysis and which parts of the life cycle are excluded from the analysis (Figure 32). For example, the manufacturing of the tractor, the N-sensor as well as of chemical fertilisers and pesticides were excluded. We assumed those to have minor impacts and to vary greatly between manufacturers. Other out-of-scope processes include post-harvest technologies such as grain storage as well as grain drying or chemical treatment against mycotoxin contamination. Field scanning was also not included as a separate operation because it is done together with the spraying of plant protection products. We also did not consider the possible residual nutrient value of the fertilisers for the next crop.

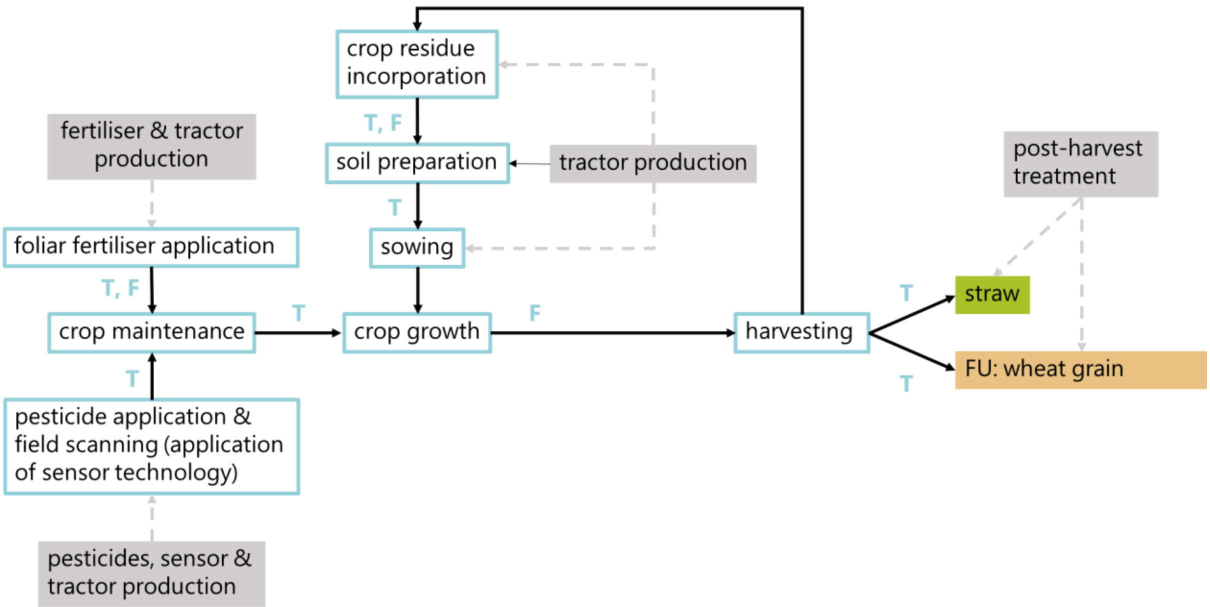


Figure 32. LL# 20: System boundaries

Boxes with blue stroke indicate processes included in the analysis, specified with T: transport related emissions included; F: field emissions included (of crop residues or mineral fertiliser). Grey boxes indicated processes outside of scope. FU: functional unit. Green = straw (with impacts allocated between straw and crop 1:1)

In brief, we included the following (compare Figure 32):

- crop residue management e.g. tractor transport on field and work on field, fuel consumption,
- soil preparation: e.g. tractor transport on field and work on field, fuel consumption,
- seed sowing e.g. tractor transport on field and work on field, fuel consumption, seed amount,
- liquid, chemical fertiliser application e.g. tractor transport on field and work on field, N2O emission due to fertiliser use,
- plant protection e.g. tractor transport on field and work on field, fuel consumption, plant protection material consumption, causing soil acidification and eutrophication,
- harvesting e.g. combine harvester transport on field and work on field, fuel consumption, harvested wheat amount,
- grain harvest transport e.g. tractor transport on field and work on field, fuel consumption.

3.2.1.2.2 Environmental indicator assessment

The environmental indicator assessment followed an attributional approach using allocation instead of system expansion.

Mass allocation was applied between grain production and straw production in a ratio of 1:1 (6 t/ha wheat : 6 t/ha straw). The functional unit is one tonne of wheat grain. We focused on the three impact categories: *climate change, terrestrial acidification* and *freshwater eutrophication*.

In the assessment, we applied GWP 100 (IPCC, 2006). For energy consumption we assumed 66.2 gCO_2e/MJ. The environmental impact assessment calculations are based on the multiplication of activity data with emission factors or emission equivalent factors. To define the *acidification potential* (SO_2e) of fertilisers, a transformation coefficient of 0.8 Kg SO_2/kg NH_4NO_3 was used. For *freshwater eutrophication potential* (PO_4e), a transformation coefficient of 0.33 kg PO_4/kg NH_4 and 3.07 kg PO_4/kg total P was considered according to CML Leiden database.

3.2.1.2.3 Inventory

Primary data was collected from the participating farms, and secondary data was calculated by means of emissions factors or derived from databases such as Ecoinvent 3.0.

Farm data / Experimental data

The primary data was collected on three participating farms during 2019 and 2021. Two of the farms (Recrea Ltd. and Gábor major Ltd.) used the tractor mounted Yara N-sensor technology and one farm (Intermező Ltd.) used the tractor mounted GreenSeeker sensor technology. The data included (solid and liquid) fertiliser use, sowing details, and working hours and use of tractors and other equipment for the production of wheat. The amount of chemical fertilisers applied differs from field to field and depends on weather conditions, the soil type and plant nutrient requirements. Within each field, there is a typical saving of 30-80 kg/ha chemical fertiliser by using the N-sensor.

Emission data / Literature data

To account for field emissions, we followed the IPCC 2006 Guidelines, Chapter 11. Field emissions accounted in this study include: direct N_2O emissions from N in chemical fertiliser and above- & below-ground crop residue decomposition and indirect N_2O emissions from chemical fertiliser N lost by volatilisation and run-off. For wheat crop residue emissions, we used a N content of 0.007 kg N/kg for the above ground part (straw) and 0.014 kg N/kg for the below ground part (roots). The ratio between the below and the above ground part of crops is 0.22. We did not take into account the N mineralisation associated with a loss of soil organic matter resulting from land use change or management of mineral soils.

Secondary data

Secondary data were used mostly for the transformation of material and energy inputs and outputs into impact categories. In some cases, we used LCA calculation results from former projects or from farm accountancy data networks, from different surveys, scientific papers, from field experiments and experts' knowledge. For fuel consumption of agricultural tractors, the Hungarian NAIK mechanical institute consumption list is used, but also farmer's communications (primary data). For conversion factors the CML Leiden database was also used. The emission factors are taken from IPCC, 2006 guideline.

3.2.1.3 Results

The climate change potential of wheat in the baseline (no sensor technology) was estimated at 0.37 kg CO_2e/kg product and with the sensor technology at 0.34 kg CO_2e/ kg product. The impact on freshwater eutrophication was 0.05 kg PO_4/kg product for the baseline without sensor technology and 0.04 kg PO_4e/kg product with the sensor technology. The impact on soil acidification was 0.021 kg SO_2e and 0.017 kg SO_2e without and with the sensor technologies, respectively. The results are in accordance with other LCA calculations for cereal grain production. In the literature, the values range between 0.4-0.9 kg CO_2e/kg product. For example, in the reference case, the wheat production had 0.83 kg CO_2e/kg with a 5 t/ha grain yield, lower than in this current study. After making the corrections to the same 6 t/ha year in this case, the CF climate change potential was 0.408 kg CO_2e/kg and the baseline reference (without technology) had 0.691 kg CO2e/kg (2)

3.2.1.4 Conclusion

The utilisation of the N sensors in the precision agriculture is a recognised technology to reduce the average application rate of N fertiliser as well as the environmental impact of agricultural practices in general (3,4,5). By using sensor technologies, greenhouse gas emissions can be reduced, and a more uniform crop production can be achieved. Accordingly, the technology is a prospective solution for farmers. In general, applied technologies including the applied materials and equipment can contribute to emission reductions in wheat production.

3.2.1.5 References

1. Athanasios B. et al (2017): Precision Agriculture Technologies Positively Contributing to GHG Emissions Mitigation, Farm Productivity and Economics, in *Sustainability* 9, https://doi.org/10.3390/su9081339

2. EIP-AGRI Focus Group (2015): Precision Farming, Final report

3. Jose L. Gabriel et al. (2017) Airborne and ground level sensors for monitoring nitrogen status in a maize crop, in *Biosystems engineering*, vol 160, https://doi.org/10.1016/j.biosystemseng.2017.06.003

4. Max Grell et al (2020). Determining and predicting soil chemistry with a point-of-use sensor toolkit and machine learning model, in Biorxiv, https://doi.org/10.1101/2020.10.08.331371

3.2.2 LL#30: Precision farming and field application of heterogeneous organic fertilisers (WUR)

Longlist #30 title: Precision farming coping with heterogeneous qualities of organic fertilisers in the whole chain

Kevin Duan [1 2], Sander Bruun [1], Lars Stoumann Jensen [1], Jan Peter Lesschen [2], Piet Groenendijk [2]

[1] Department of Plant and Environmental Sciences, University of Copenhagen, Denmark.

[2] Sustainable Soil Management Group, Wageningen Environmental Research, the Netherlands.

3.2.2.1 Introduction

The composition of livestock slurry varies from batch to batch. During storage and long-distance transport, slurries can stratify, resulting in an uneven distribution of nutrient content within the tank. In current farming practice, farmers usually lack knowledge on the slurry nutrient variation, and the slurry nutrient content is assessed based on tabulated empirical values or limited sampling. During field application, slurry is usually dispensed at a fixed flow rate. However, due to the intrinsic heterogeneity in slurry composition, the actual N applied will fluctuate around the target rate. Depending on farmer knowledge and availability of manure analyses, farmers in some areas tend to estimate their application rates based on the lower bound of the nutrient concentration values to avoid potential yield reduction due to insufficient fertilisation. This practice, however, often leads to an excessive application of slurry N.

A near-infra red sensor (NIRS) technology has been developed to measure the N content of slurry on-the-fly during application, allowing tankers to change the flow rate or travelling speed to adjust the N application rate. With the NIRS technology, slurry N may be applied at the intended N application rate, eliminating excessive N application and overcoming fluctuations caused by slurry heterogeneity.

Precision fertilisation (slurry application) is an important component of the practice of precision farming. The NIRS technology provides farmers with information on how much slurry N is applied, but knowledge of how much N is needed to achieve optimal crop yield is equally important. Nutrient requirements by crops depend on a multitude of factors, including crop variety, soil type and condition, weather, pests, diseases, etc. Soil conditions determine water and nutrient availability to crops, but detailed soil mapping is not always available to farmers. Alternatively, some important soil conditions, such as organic matter (OM) content and groundwater level, may be qualitatively derived from the topography of the field.

The landscape within a field may be categorised into three basic topographies: plain, shoulder (small hill), and depression. Due to tillage erosion and wind/rain depositions, soil is removed from crests and shoulder slopes and deposited at foot slopes and depressions, resulting in a thicker top layer and a higher OM content in depressions than shoulders (Govers et al., 1996). Groundwater levels also differ between topographies. For example, depressions tend to have a higher groundwater level than plains and shoulders. During prolonged precipitation, waterlogging and surface ponding can occur in depressions, which impedes crop root development during the growing season and can eventually result in yield reduction. On the other hand, high groundwater in depressions may be beneficial to crop production during dry spells.

The benefit of NIRS technology is directly related to fertiliser application and the associated environmental emissions, but the environmental footprints related to the manufacturing and use of NIRS are unclear or difficult to define. Therefore, a life cycle analysis is not suitable to assess the environmental impact of NIRS technology. Instead, we opted to perform an environmental indicator assessment, using agri-environmental modelling to investigate the effect of precision slurry application with NIRS on crop yields and field environmental emissions as compared to conventional practice. We also modelled key soil conditions derived from field topographies and evaluated the potential use case of precision farming with the combination of NIRS and soil information. We chose the monoculture of winter wheat on various land topographies in Denmark as an example case to demonstrate the setup and capability of the modelling approach. With proper information, similar modelling practice can also be performed with other farming systems and environmental conditions.

3.2.2.2 Materials & methods

3.2.2.2.1 General model setup

Daisy, a field-level mechanistic agri-environmental model (Abrahamsen & Hansen, 2000), was used to assess the effect of precision slurry application on crop yields and field environmental emissions. The simulation was set up on the baseline scenario CTW-Pig as specified in the Nutri2Cycle deliverable 1.5 report (Duan et al., 2021), which models pig farm production in the continental west (CTW) geoclimatic region. Soil and climate were mostly based on eastern Denmark, but similar conditions are also common in eastern Germany and Poland (Metzger et al., 2012). The production of winter wheat monoculture was simulated using a winter wheat crop model calibrated by Gyldengren et al. (2020). Following a 30-year spin-up period to prime the soil OM pools, simulation was run for 20 years continuously to collect results, accounting for the effect of weather variation.

3.2.2.2.2 Simulation of variable field topography

For simplification, three different topographies were defined: plain, shoulder, and depression (Table 34). The plain topography represents the “normal” conditions in a field, with medium OM content and thickness in the topsoil, and medium groundwater level. Topsoil thickness and OM content were highest in the depressions, and lowest in the shoulders. The shoulders have low groundwater levels, whereas three levels of groundwater were considered for the depressions: medium, high, and waterlogged. The different groundwater levels were simulated by modifying the depths of drainage pipes, horizontal distances between drainage, conductivity of bottom aquitard layer, etc. By increasing root death rate during waterlogging period, the deterioration of crop growth and production due to waterlogging was simulated. The root death rate was assumed to be positively correlated to the fraction of water saturated root zone.

Table 34. LL# 30: Simulation set-up

Setup of topsoil and groundwater conditions to simulate field topographies in the Daisy model. Parameters were derived from the Daisy handbook by Styczen et al. (2006).

	Plain	Shoulder	Depression		
			Medium groundwater	High groundwater	Waterlogged
Topsoil OM content (%)	2.50	1.38	4.06	4.06	4.06
Topsoil thickness (cm)	30	25	40	40	40

	Plain	Shoulder	Depression		
			Medium groundwater	High groundwater	Waterlogged
Groundwater level	Medium	Low	Medium	High	High with occasional surface ponding
Drainage depth (m)	1.2	2.0	1.2	0.6	0.3
Distance between drainage pipes (m)	18	100	18	18	48
Aquitard conductivity	2.33E-3	2.0E-2	1.10E-4	1.10E-4	1.10E-4

3.2.2.2.3 Simulation of conventional and precision slurry application scenarios

Per Danish regulation and convention, the average N application rate for winter wheat on loamy soil was 255 kg N/ha in total, which consists of 170 kg N from pig slurry (maximum rate allowed by EU Nitrates Directive, with a mineral fertiliser equivalent coefficient of 0.75), and 85 kg mineral N fertiliser. This gives an effective N fertilisation rate of 212.5 kg N/ha (170 × 0.75 + 85, the statutory N-norm for winter wheat according to fertiliser regulations). Therefore, the target rate for slurry was 170 kg slurry N/ha for both precision and conventional applications.

For each field topography, the simulation consisted of two fertilisation scenarios: precision and conventional. For precision application, slurry was always applied exactly to the target rate. For conventional application, two sub-scenarios were considered: (a) farmers have some knowledge on the average nutrient content of the slurry and can apply slurry according to the target rate (Conv_100), despite intrinsic compositional fluctuations; and (b) farmers do not have accurate information on slurry nutrient content and over-apply slurry N as an aversion tactic for potential yield loss (the "over-application" scenario). Furthermore, for the over-application scenario, we considered a surplus of 25, 50, 75, and 100% over the target rate (Conv_125 to Conv_200), to explore the upper boundaries of environmental emissions (or "worst-case" scenario) by conventional application method.

Total N input rates ranging from 0 to 500 kg N/ha were simulated to construct response curves of crop yields and environmental emissions to N fertilisation. Only slurry was applied when N input rates were ≤ 170 kg N/ha, and mineral N was added to fulfil the quota when N input rates were > 170 kg N/ha.

To simulate the fluctuations in actual N application rates by conventional method, for each N input level the simulation was repeated 10 times, and the slurry N rate on each application event was drawn randomly from a normal distribution with a mean of the N input level and a coefficient of variation (CV) of 0.1. The CV was derived from measurements of conventional slurry application in field trials (data from John Deere Germany, personal communication). Mineral fertiliser was assumed to always be applied by the precise amount in any case.

In the results and discussions hereafter, the N input rates referred to are target or "intended" rates, not the actual application rates. For example, in the conventional over-application scenarios, when farmers "intend" to apply 100 kg slurry N/ha, they may, unknowingly, actually apply around 150 kg slurry N/ha in scenario Conv_150, and 200 kg slurry N/ha in scenario Conv_200. However, during analysis, these rates were treated as 100 kg N/ha, not their actual values. The intended rates were used instead of actual application rates to reflect that farmers do not have accurate knowledge on actual N application rates in conventional scenarios as discussed above. This would be consistent with situations in a field study, where the intended rates are the ones that farmers report, and the ones used for analysis.

3.2.2.3 Results & discussion

3.2.2.3.1 Precision vs. conventional slurry application in a single year and in the long term

Figure 33a and c show crop dry matter yields and N losses to water following precision and conventional slurry applications within a single year (2000 as an example). Slurry was applied at the target rate uniformly across the entire field with the precision method. Assuming no other environmental variations within the field, crop yields were also uniform across the entire field with precision application. Conventional scenarios, on the contrary, applied slurry unevenly. This introduced some variation in crop yields within the field (Figure 33a). The average yields for the entire field were equal to the expected yield under the target rate, as the yield variations would often cancel out each other. Similarly, the differences in N emissions between precision and conventional (Conv_100) scenarios were also marginal. Conventional over-application scenarios (Conv_150 and Conv_200) generally had higher yields than precision and regular conventional (Conv_100) scenarios, due to the unintended excess slurry N over the target rate. However, this also led to increased N emissions to the environment (Figure 33c).

Weather variation was considered by averaging the simulation results over 20 continuous years. The long-term simulation showed some variation in yields and environmental emissions to the precision scenario as well, as shown in Figure 33b and d. Interestingly, the intensity of variation between precision and Conv_100 conventional scenarios are similar, suggesting that the effect by weather and N input rates are not simply cumulative. In fact, the two factors can counteract each other. For example, in the conventional scenario, yield reduction caused by unfavourable weather may be compensated by over-application of slurry N. However, as farmers do not have the ability to predict climate at the time of fertilisation, the interaction between weather and fertilisation rate can only be random.

In regions where farmers have access to more accurate estimates of slurry nutrient content, as modelled in the Conv_100 scenario, precision slurry application generally offers little to no benefit in terms of improving crop yields or reducing N emissions as compared to conventional methods. Still, precision slurry application in these regions may reduce the uncertainty in crop yields and environmental emissions and help farmers to avoid accidental under- or over-application of slurry.

The benefit of precision slurry application with NIRS is apparent in regions where over-application is common, as precision slurry application is effective in reducing excess slurry N application and the associated emissions without compromising crop yields.

Nonetheless, NIRS technology by itself is inadequate for the practice of precision farming. To better estimate how much fertiliser should be applied where, other factors, such as soil conditions, must be considered as well.

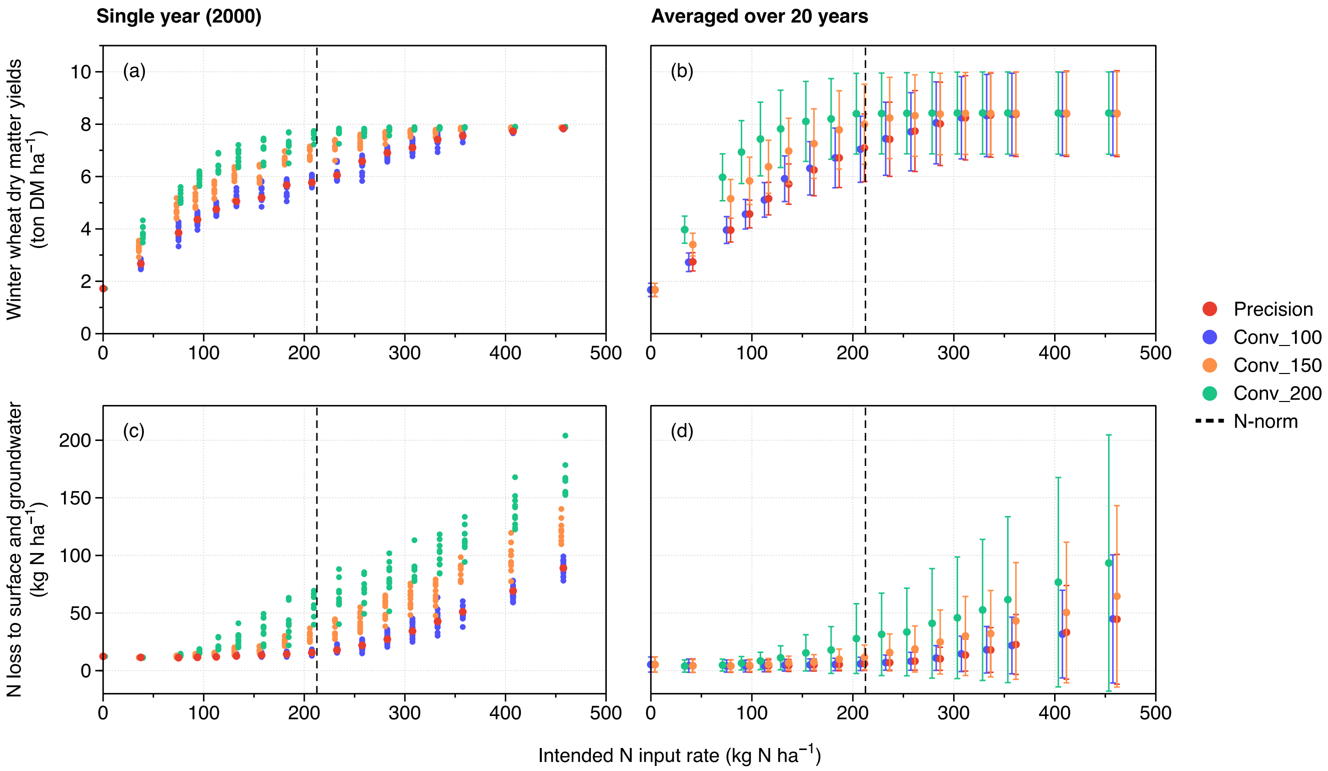


Figure 33. LL# 30: Modelling results - 1

Winter wheat dry matter yields (a & b) and N loss to water (c & d) in plain profile in response to N input rates. Panels a & c show simulations from a single year, in which each dot represents one simulation. Panels b & d show the averaged values over the 20-year simulation period, where each dot represents the mean and error bars show standard deviation. Conventional or precision slurry application scenarios are represented by different colours. For visibility reasons, only Conv_150 and Conv_200 are kept in the figure to represent over-application scenarios.

3.2.2.3.2 Crop yields and emissions in different field topographies

Figure 34 shows the response of crop dry matter yield to N input rates in different topographical profiles. Crop yields in plain, shoulder, and depression (with both medium and high groundwater levels) profiles showed similar response to N input, with increasing crop yields as N input increases, reaching maximum yields at approximately 310 kg effective N/ha input for the precision and Conv_100 scenarios, after which crop yields levelled off. Maximum yields were reached at lower intended N input rates for the over-application scenarios.

At the winter wheat N-norm, depression with medium groundwater level had the highest yields, followed by plain and shoulder. Yields at N-norm were comparable to yields measured at field experiments under similar conditions (Jensen et al., 2021), and were approximately 84% of the maximum yields. The simulated yields at high N input rates (> 300 kg N/ha) may be overestimated, as yield reduction by crop lodging cannot be simulated by Daisy yet. Yields become reduced in depressions with rising groundwater levels, as elevated groundwater tends to restrict crop root growth and thus limits crop production. Waterlogging is an extreme case of high groundwater condition in which the entire or most of the soil column was saturated, causing root death and eventually resulting in a severe decline in crop yields (a 64% reduction as compared to that in the plain at N-norm).

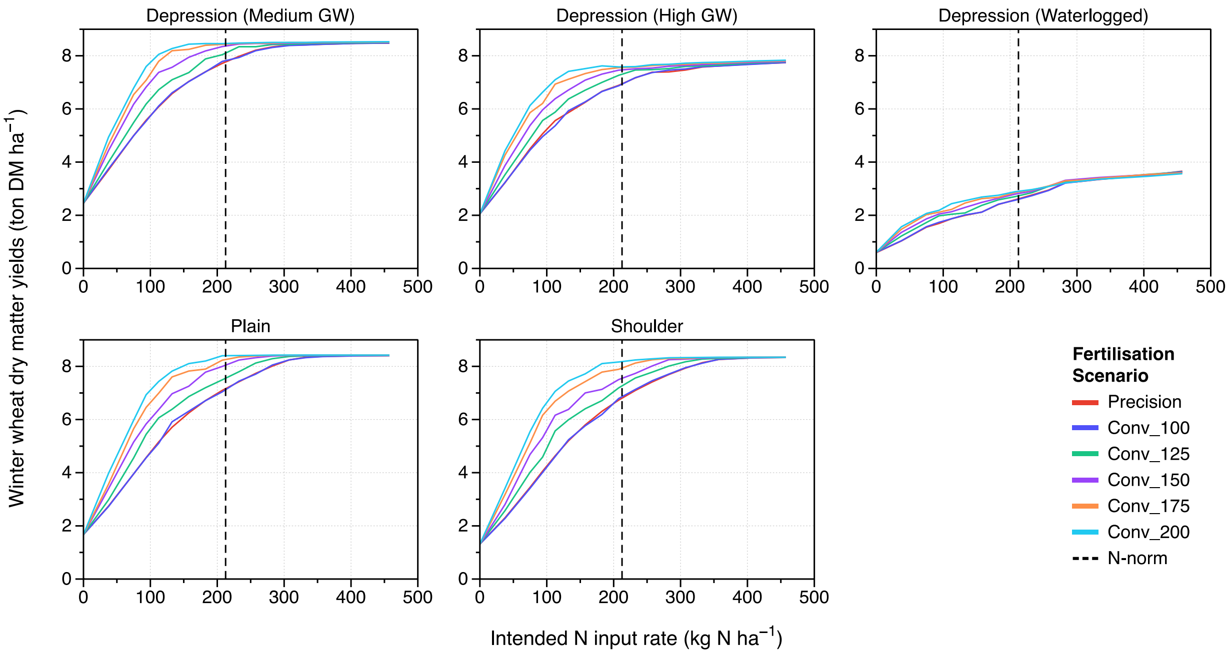


Figure 34. LL# 30: Modelling results - 2

Winter wheat dry matter yields in response to N input rates in different field topographies, and under conventional or precision slurry application scenarios. Note that the x-axes represent intended total N input rates (combination of both slurry and mineral N). For conventional over-application scenarios, the actual input rates are higher than the intended rates.

Figure 35 shows the response of N loss to water in different topographies. N losses to water were low in plain and shoulder profiles (approx. 6 kg N/ha at application rates below the N-norm) and became slightly elevated in depressions with medium to high groundwater (10–30 kg N/ha). In waterlogged depression, N loss was several folds higher than that in other profiles (> 100 kg N/ha), and the loss was significant even with no N fertilisation (N input level 0) but increased more or less proportionally with increasing N inputs.

N emissions to the atmosphere included NH_3 volatilization and N_2O emissions from both nitrification and denitrification. All topographical profiles showed similar NH_3 volatilisation levels under the same N input, as volatilisation depends mostly on the properties of the fertiliser and application conditions. Nitrification-related N_2O emissions were higher in depressions than in plain and shoulder profiles, which may be attributed to higher mineralisation of organic matter content in depression topsoil. Nitrous oxide originating from denitrification was estimated by assuming a N_2/N_2O ratio of 4 in the denitrification product (Vinther & Hansen, 2004), and it was also higher in depressions, and 3–4 times higher in waterlogged depression than in other profiles. Denitrification is an anaerobic process and is likely to proliferate in waterlogged soils where aeration is impaired.

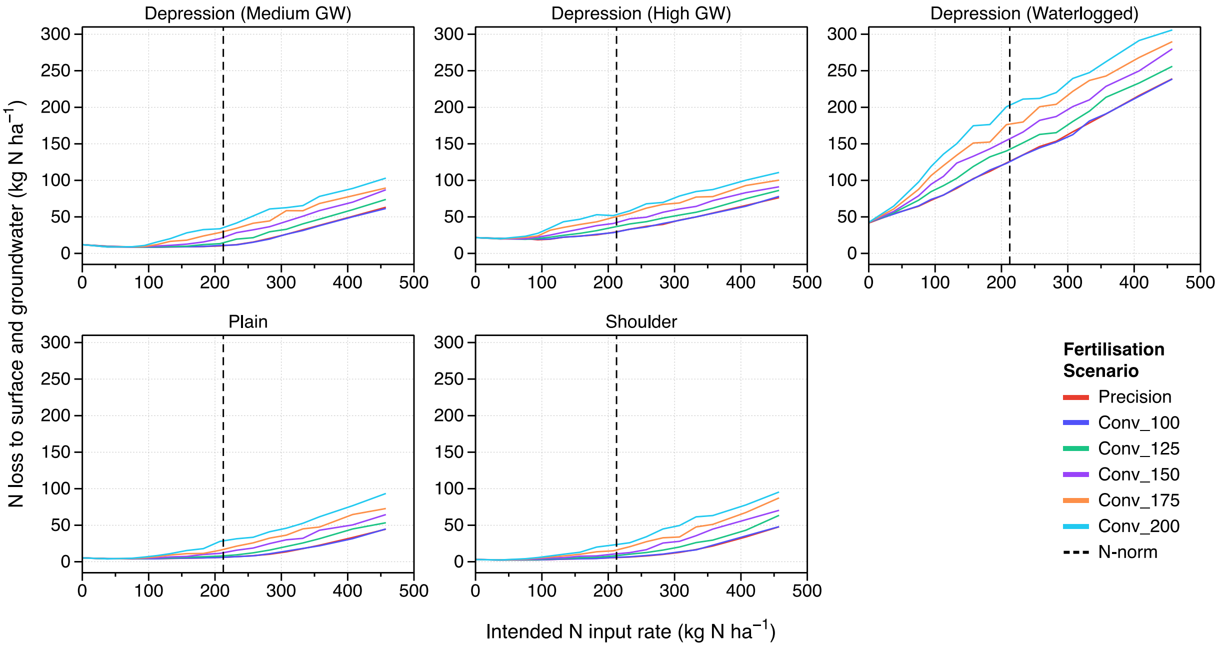


Figure 35. LL# 30: Modelling results - 3

Combined N loss to surface and groundwater, including leaching and loss via drainage pipes, as response to total N input rates on different field topographies, and under conventional or precision slurry application scenarios. Note that the x-axes represent intended total N input rates (combination of both slurry and mineral N), and for conventional over-application scenarios, the actual input rates are higher than the intended rates.

3.2.2.3.3 Precision farming combining NIRS and field topography

With the knowledge on nutrient content variation in the slurry, as well as the yield response to N input in different field topographies, farmers can better estimate how much fertiliser should be applied to which topographical profile to achieve optimal yield or revenue. The optimisation of fertilisation scheme is a field-specific task, requiring information on the specific topographical composition of the field, the cost of field management, the sales price for the crop, etc. Therefore, it is difficult to provide a general solution. However, in the text below, we demonstrate how such an optimisation may be performed using a theoretical example.

As waterlogged depressions are unfavourable to crop production and has a great potential for N losses, it may be advisable to reduce or completely avoid fertilisation in waterlogged depressions. The saved fertiliser can be reallocated to other topographical profiles within the field. In this study, we consider a hypothetical field of 1 hectare with 0–50% waterlogged depression in the field, and evaluate how fertiliser may be rearranged within the field to achieve optimal revenue. We assumed the moisture content of the grain was 15%, the sales price of winter wheat was €161 per tonne (average farm-gate price from 2010 to 2020, Danmarks Statistik), the cost for N fertiliser was €1.15/kg mineral N (based on average ammonium nitrate price from 2010 to 2020, Danmarks Statistik) and free for slurry N. All prices were converted to Euro based on most recent exchange rate. The cost of fertiliser application was not included. The field revenue was calculated as the sales of crop minus the cost of fertilisers.

Waterlogged depression may be fertilised in two scenarios: (a) completely avoid fertilisation (zero N input, the "avoidance" scenario), and (b) reduce fertilisation rate in the waterlogged depression between maximum legal allowance and complete avoidance (the "reduction" scenario). In both scenarios, the saved fertiliser from waterlogged depression was evenly distributed to other parts of the field. A Python script was developed to calculate field revenue based on the area of different topographies and the crop yields in each topography as a response to N input level. For the "reduction" scenario, the fertilisation scheme was optimised using the `minimize` function and the "SLSQP" optimisation algorithm in Python package `scipy`. During optimisation, the algorithm tried to reduce fertilisation in waterlogged depression and reallocate the fertilizer to other topographies, resulting a yield loss in waterlogged depression and a yield increase in other topographies. This process was iterated until the maximum field revenue was found. A constraint condition was applied during optimisation to limit the average N application rate over the entire field to not exceeding the N-norm.

The optimised field revenue results are illustrated in Figure 36, which shows the change in optimal revenue in a theoretical field when the proportion of waterlogged depression increases from 0 to 50%. The highest revenue was achieved when there was no waterlogged depression in the field, and the revenue continuously decreased as the area of waterlogged depression increased in the field. When the area of waterlogged depression was < 25% in the field, there was none or only marginal difference in optimal field revenue between avoiding fertilisation ("avoidance" scenario) and reducing fertilisation rates to waterlogged depression ("reduction" scenario). In fact, in these cases, N input to waterlogged depression was reduced to 0 in the "reduction" scenario, making them essentially identical to the "avoidance" scenario. It suggests that with a relatively small waterlogged depression in the field, farmers may simply avoid fertilisation in waterlogged depressions without the risk of significant revenue loss. However, as the area of waterlogged depression increases, there was a higher risk of revenue loss by avoiding fertilisation to waterlogged depression, and the optimal revenue for avoidance scenario was 14% lower than the reduction scenario with 50% waterlogged depression in the field. In the case of a large waterlogged depression in the field, completely avoiding fertilising the depression would cause a yield loss potential that cannot be mitigated by the yield increase from reallocating fertilisers to other topographies. Therefore, here it makes economic sense to reduce N input rather than avoid fertilisation to the waterlogged depression. However, farmers may still prefer to avoid fertilisation in large, waterlogged depressions, considering practical difficulties such as driving the tractor into a waterlogged depression.

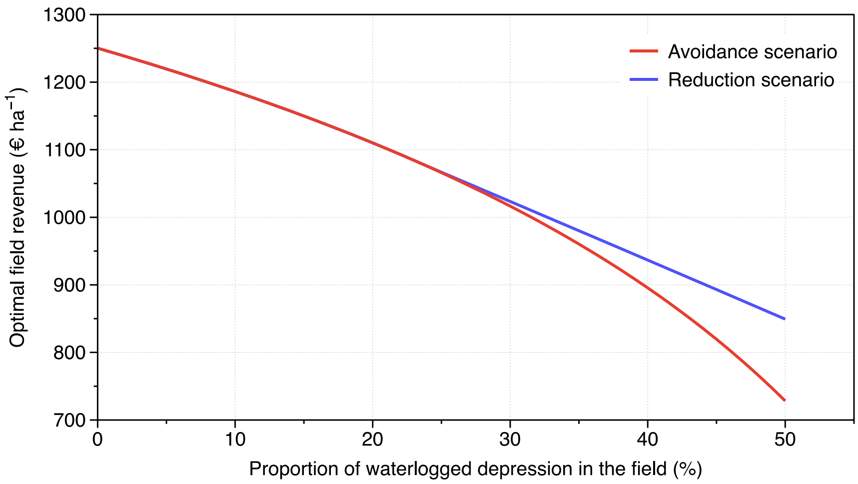


Figure 36. LL# 30: Modelling results - 4

Optimal field revenue generated on a theoretical field with different proportions of waterlogged depressions under reduction and avoidance fertilisation scenarios.

3.2.2.4 Conclusion

Precision slurry application with NIRS technology provides farmers with insight on the nutrient content variation in the slurry and allows farmers to apply slurry more precisely to meet the requirements by regulation and crop needs. In regions where farmers have access to more accurate estimates of slurry nutrient content (chemical analysis or reliable tabulated values), precision slurry application generally offers little to no benefit in terms of improving crop yields or reducing N emissions as compared to conventional methods. However, in cases of poor farmer information level, where over-application is common, precision slurry application can markedly reduce the risk of over-application and hence emissions to the environment, without compromising crop yields. NIRS can therefore be an important tool in the practice of precision farming, especially in combination with knowledge on other field environmental factors, such as soil conditions derived from field topography. In the example demonstrated in this study, NIRS and knowledge on field topography could potentially enable farmers to rearrange fertiliser application within the field to achieve more optimal revenue and less loss to the aquatic environment. To validate effectiveness and advantage of precision farming, future studies may focus on the modelling of actual fields of different farming types with proper quantification of soil variations.

3.2.2.5 References

Abrahamsen, P., Hansen, S., 2000. Daisy: an open soil-crop-atmosphere system model. Environmental Modelling & Software. 15(3), 313-330. https://doi.org/10.1016/S1364-8152(00)00003-7

Duan, Y.F., Bruun, S., Jensen, L.S., van Gerven, L., Hendriks, C., Stokkermans, L., Groenendijk, P., Lesschen, J.P., Prado, J., Fangueiro, D., 2021. Mapping and characterization of CNP flows and their stoichiometry in main farming systems in Europe. https://edepot.wur.nl/547940

Govers, G., Quine, T.A., Desmet, P.J.J., Walling, D.E., 1996, The relative contribution of soil tillage and overland flow erosion to soil redistribution on agricultural land. Earth Surf. Process. Landforms, 21: 929-946. https://doi.org/10.1002/(SICI)1096-9837(199610)21:10<929::AID-ESP631>3.0.CO;2-C

Gyldengren, J.G., Abrahamsen, P., Olesen, J.E., Styczen, M., Hansen, S., Gislum, R., 2020. Effects of winter wheat N status on assimilate and N partitioning in the mechanistic agroecosystem model DAISY. J Agro Crop Sci. 206:784– 805. https://doi.org/10.1111/jac.12412

Jensen, K.J.S., Hansen, S., Styczen, M.E., Holbak, M., Jensen, S.M., Petersen, C.T., 2021. Yield and development of winter wheat (Triticum aestivum L.) and spring barley (Hordeum vulgare) in field experiments with variable weather and drainage conditions. European Journal of Agronomy, 122. https://doi.org/10.1016/j.eja.2020.126075

Metzger, M.J., Shkaruba, A.D., Jongman R.H.G., Bunce, R.H.G., 2012. Descriptions of the European Environmental Zones and Strata. Alterra Report 2281. https://edepot.wur.nl/197197

Styczen, M., Hansen, S, Jensen, L. S., Svendsen, H., Abrahamsen, P., Børgesen, C. D., Thirup, C., Østergaard, H. S., 2004. Standardopstillinger til Daisy-modellen. Vejledning og baggrund. Version 1.2, April 2006. DHI Institut for Vand og Miljø. 62 pp.

Vinther, F. P., & Hansen, S. (2004). SimDen -en simpel model til kvantificering af N2O-emission og denitrifikation (104). Retrieved from https://orgprints.org/3139/

3.3 Social Life Cycle Assessment of Shortlisted Technologies (IRTA)

Edilene Pereira Andrade, August Bonmati, Assumpció Anton

Institute of Agrifood Research and Technology, Torre Marimon, 08140 Caldes de Montbui, Barcelona, Spain

This chapter builds on the published scientific paper Andrade et al. (2022) Assessment of social aspects across Europe resulting from the insertion of technologies for nutrient recovery and recycling in agriculture. Sustainable Production and Consumption, 31, 52-66, https://doi.org/10.1016/j.spc.2022.01.025

3.3.1 Introduction

The potential positive and adverse social impacts generated by the introduction of novel technologies, in this case nutrient recovery and recycling technologies and / or solutions, are of great concern considering the uptake and adoption of these technologies (Andrade et al., 2022). This study investigated the potential social impacts of the selected Nutri2Cycle technologies introduced in agriculture to reduce nutrient losses and close CNP loops.

3.3.1.1 Technologies included in the sLCA

In this section, 12 technologies for nutrient recovery and improvement of nutrient efficiency are assessed. The technologies and their corresponding baseline scenarios are listed in Table 35.

Table 35. sLCA: assessed technologies and respective baseline

ID	TECHNOLOGY	BASELINE
RL4.LL1	Ammonium stripping / scrubbing and NH_4NO_3 as substitute for synthetic N fertilisers	Impacts from crop production with mineral fertilisation.
RL4.LL2	Ammonium stripping/scrubbing & $(NH_4)_2SO_4$ as substitute for synthetic N fertilisers	Impacts from crop production with mineral fertilisation.
RL4.LL6	Concentrate from vacuum evaporation/ stripping as nutrient-rich organic fertiliser	Mono-cultivation of maize was compared to the no-fertiliser & synthetic fertiliser (calcium ammonium nitrate (CAN)) treatments.
RL3.LL13	Sensor technology to assess crop N status	Traditional chemical fertilisation technology without N sensors.
RL2.LL17	Crop farmer using a variety of manure and dairy processing residues to recycle and build soil C, N, P fertility	Crop production with mineral fertilisation.
RL1.LL18	Slurry acidification with industrial acids to reduce NH_3 volatilisation from animal husbandry	Pig/cattle slurry management without acidification.
RL4.LL20	Low temperature ammonium-stripping using vacuum	Pig manure management without processing.
RL3.LL30	Precision farming coping with heterogeneous qualities of organic fertilisers in whole chain	Current practice of applying liquid manure without precise info on nutrient content & traditional nutrient testing and storage time.
RL5.LL40	Insect breeding as an alternative protein source on solid agro-residues (manure and plant wastes)	Manure management without processing.
RL5.LL41	Floating wetland plants grown on liquid agro-residues as a new source of proteins	1) Tertiary treatment of constructed wetlands using reed + 2) Imports of protein rich soy from North and South America
RL4.LL49	Nitrogen and phosphorus recovery from pig manure via struvite crystallization and design of struvite based tailor-made fertilisers	Crop production with mineral fertilisation.

RL4.LL55	Manure processing and replacing mineral fertilisers – The Netherlands	Mineral N fertilisers being applied on grassland in the region.

3.3.1.2 Social Life Cycle Assessment of shortlisted technologies

3.3.1.2.1 Set of indicators

Although several studies about suitable social indicators for sLCA are available (UNEP, 2020; Pelletier, 2018; Chen & Holden, 2017; Revéret et al., 2015), there is no commonly accepted set of indicators (Hauschild et al., 2008). In addition, it could even be questioned whether such set of indicators is feasible due to the high variety of existing systems that such a methodology could be relevant for.

In the current study, a set of indicators relevant to agriculture and nutrient recovery were selected. This includes both social and environmental indicators (midpoint indicators) with social consequences (endpoint indicators). The set of indicators is summarised in Figure 37. It is important to note that caution should be exercised to avoid overlaps or double counting when carrying out an environmental LCA and a social LCA. It is necessary to clarify how the indicator can have social and environmental consequences or to eliminate the indicator from one or the other assessment, as in Werker et al. (2019).

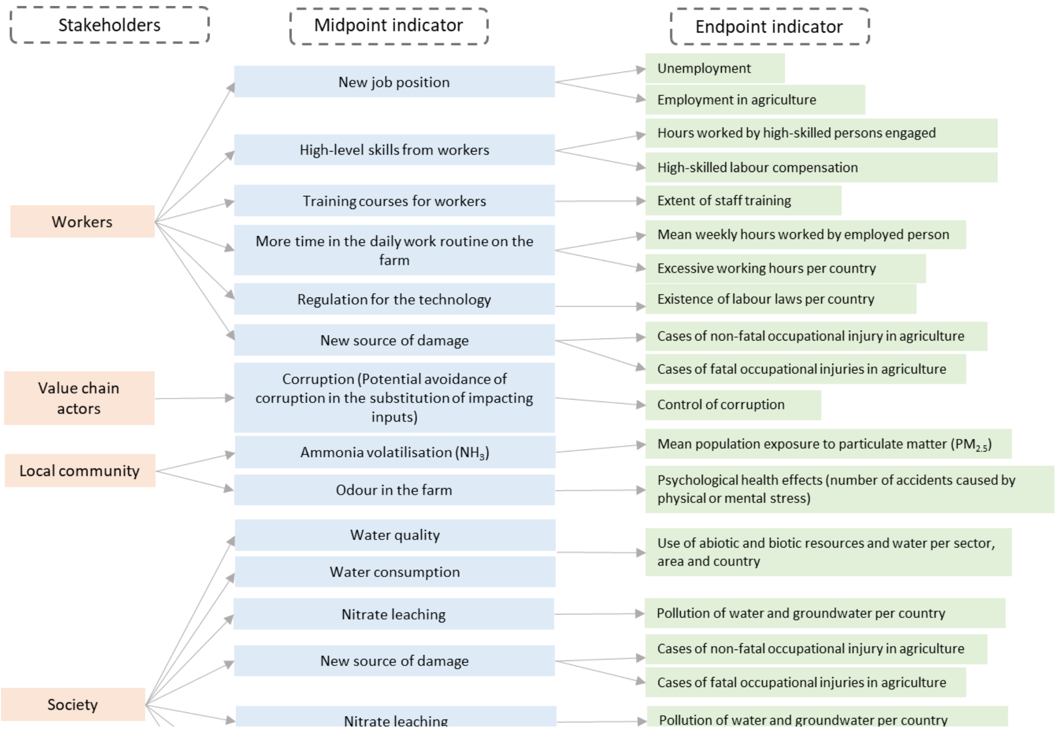


Figure 37. sLCA: Set of indicators applied

3.3.1.2.2 Collecting inventory for the sLCA and impact assessment method

First, an Excel questionnaire featuring the selected indicators was sent to experts on each technology to collect data.

The questionnaire was based on the Likert scale (Albaum, 1997). The Likert scale is used to measure the attitude of the experts towards the technologies. The questionnaire consists of a series of statements to which a respondent indicates a degree of agreement or disagreement using the following options: strongly agree, agree, neither agree nor disagree, disagree or strongly disagree (Table 36).

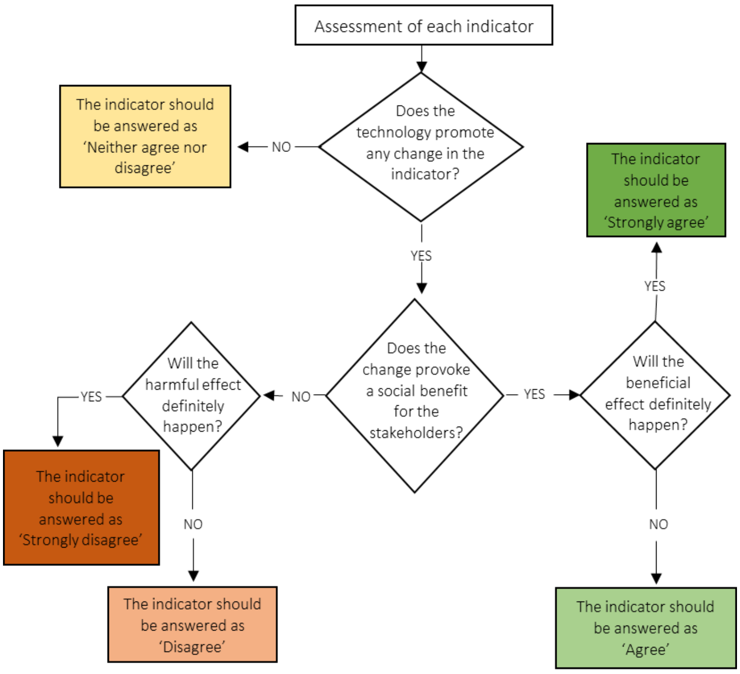


Figure 38. sLCA: Decision tree for social Life Cycle Inventory (sLCI)

Identification of potential social impacts from solutions for nutrient recovery in agriculture and livestock

For the impact assessment, an adapted version of the approach used in Franze and Ciroth (2011) was applied. Considering the complexity of social phenomena and the difficulty of avoiding ordinal scales in sLCA (Arvidsson, 2019), the scale in Table 36 was used in the impact assessment, ranging from 'potentially large beneficial effect' to 'potentially large harmful effect'. The indicators were not aggregated, and the technologies were not ranked.

Table 36. sLCA: Qualitative assessment of social indicators using Likert scale parameters

Level (Likert scale)	Strongly agree	Agree	Neither agree nor disagree	Disagree	Strongly disagree
Impact assessment	High potential of beneficial effect (**HPBE**)	Potential beneficial effect (**PBE**)	Indifferent effect (**IE**)	Potential harmful effect (**PHE**)	High potential of harmful effect (**HPHE**)

The qualitative information obtained from the questionnaires for the midpoint indicators underwent a review round owing to the importance of data triangulation in sLCA (Ramirez et al., 2016), especially when qualitative data are used since there is no guarantee that the respondents have interpreted the potential effect in the same way.

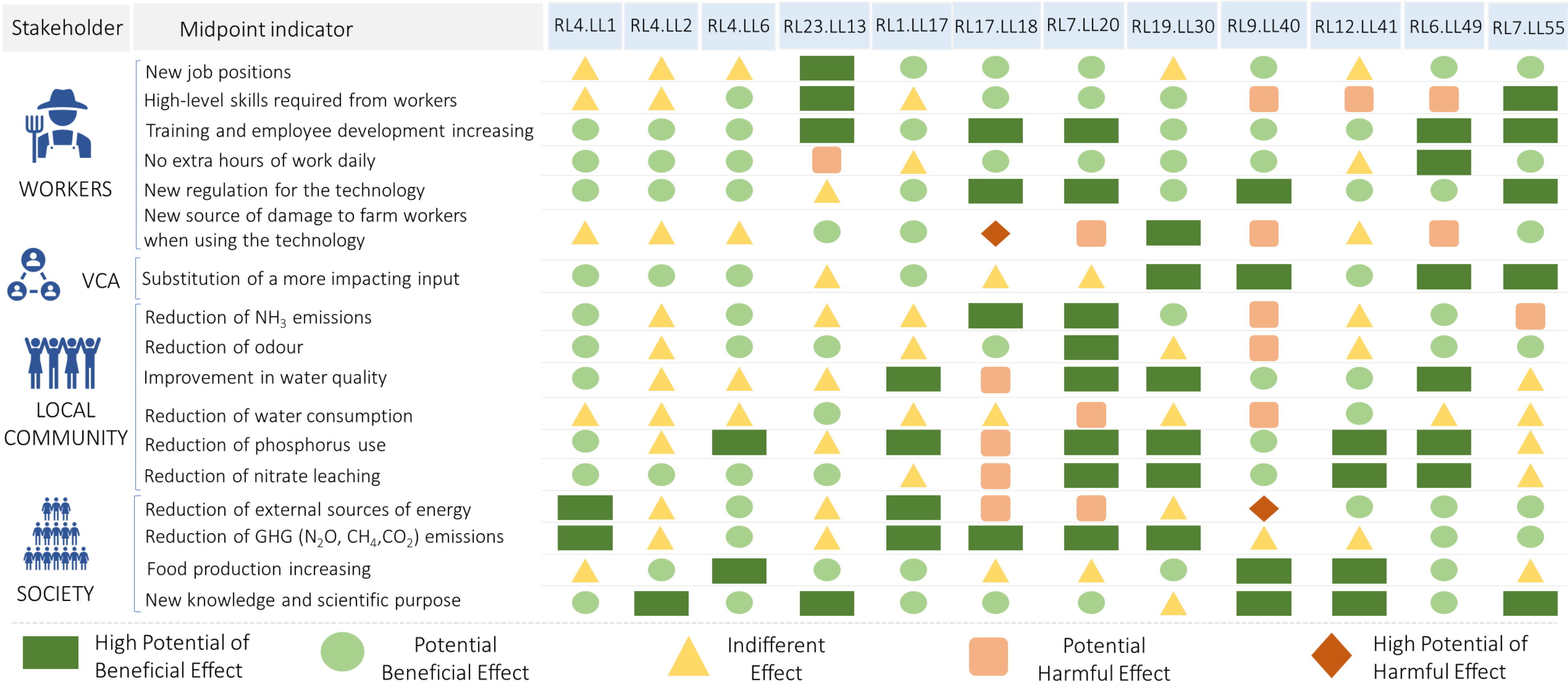
3.3.1.3 Impact assessment

The technologies were prospectively evaluated, bearing in mind that they can vary greatly according to the context (country/farm) in which they are applied or the baseline with which they are compared. In the present study, social impacts were assessed considering where the technology is developed. Results after the two rounds of questioning are presented in Figure 39. It is important to highlight that the social assessment provided in this section is not based on experience. Due to the low level of adaptation of the technologies, they were assessed as *potential* effect.

- ***RL4.LL1 'Ammonium stripping / scrubbing and NH_4NO_3 as substitute for synthetic N fertilisers'*** It has been scored as 12% HPBE, 65% PBE, 6% IE, 18% PHE and 0% HPHE. It is expected that the implementation of ammonium stripping and scrubbing will increase training and employee development in the agricultural sector since extensive knowledge about product characteristics and fertiliser application is necessary. In addition, regulations are continuously updated which means that people working with the technology will have to be trained continuously (e.g., European Fertilising Product Regulation (FPR) and REcovered Nitrogen from manURE (ReNuRe) that might influence the use of the fertiliser). Treatments for organic fertiliser can have positive impacts when compared to raw animal manure as emissions are better controlled and managed. Also, due to a decreased use of fossil fuel energy to produce synthetic chemical fertilisers, it is expected that stripping-scrubbing reduces external sources of energy and GHG emissions. The technology is also expected to have soil health benefit, to increase soil organic matter and carbon, and to thus contribute to an increase in food production. On the other hand, new job positions or high-level skilled workers are not necessary. Thus, improvements on unemployment and on workers with high skills in agriculture are unlikely to be achieved with this technology.
- ***RL4.LL2 'Ammonium stripping / scrubbing and $(NH_4)_2SO_4$ as substitute for synthetic N fertilisers'*** It has been scored as 12% HPBE, 65% PBE, 6% IE, 18% PHE and 0% HPHE. Potential impacts of this technology are expected to be the same as in RL4.LL1.
- ***RL4.LL6 'Concentrate from vacuum evaporation/ stripping as nutrient-rich organic fertiliser'*** It has been scored as 18% HPBE, 59% PBE, 24% IE, PHE and HPHE 0%. The technology promotes an improved organic fertiliser in the form of an evaporation concentrate. Thus, compared to unprocessed pig manure, the production of evaporator concentrate can reduce the GHG emissions during storage and field application. In addition, the product can be used as substitution for synthetic mineral NPK fertiliser as its production requires high inputs of energy and fossil fuels. Although it is not expected to create new job positions, the technology can contribute to introducing high-level skilled workers in agriculture to better handle the technology, and training is necessary regarding how product characteristics affects its use. Regarding the production of the evaporator concentrate, there should be regulations related to the working environment, but for the application, there is no difference compared to conventional farming. Finally, the produced evaporator concentrate is rich in labile organic N but contains low NH_4-N which means that there is a potential to reduce the NH_3 emission during field application compared to field application of unprocessed pig manure. In addition, using the technology can reduce the odour on the farm.

RL3.LL13 'Sensor technology to assess crop N status' It has been scored as 24% HPBE, 29% PBE, 35% IE, 12% PHE and HPHE 0%. The technology has the potential to create new job positions for skilled workers with knowhow about the use and production of the sensor. This could contribute to decreasing unemployment. In addition, to deal with the sensor, training is required, also for the calibration of the equipment for a proper function, and there will be no or very little harm on any employees or the environment from using the technology. It is also expected that precision fertilisation improves food production. On the other hand, more time is required in preparing and planning of the application, as the planning is at plot level, but no regulation is necessary regarding the use of the technology.

- **RL2.LL17 *'Crop farmer using a variety of manure and dairy processing residues to recycle and build soil C, N, P fertility'*** It has been scored as 24% HPBE, 35% PBE, 41% IE, PHE and HPHE 0%. Considering the end user (i.e., farmers) of this solution, new job positions are unlikely to be created, but for local distribution and for making the products widely available, some jobs may be needed. Since nutrient availability varies between the different recycling products, the application rates need to be adjusted compared to conventional chemical fertilisers, and nutrient advisors and farmers need more practical knowledge on the use and availability of nutrients from recycling products so that expected crop yields are obtained. The technology can contribute to new regulations since existing health and safety regulations may need to follow when applying recycling products. In addition, by using the proposed biobased fertilisers, the use of synthetic chemical fertilisers would be reduced. Consequently, there would also be a reduction of external sources of energy. It can also be assumed that methane emission will be reduced due to less use of chemical fertiliser (i.e., less emission from fossil fuel burning) which is likely to outweigh methane emission from organic or bio-based fertiliser application.
- ***RL1.LL18 'Slurry acidification with industrial acids to reduce NH_3 volatilization from animal husbandry'*** It has been scored as 24% HPBE, 29% PBE, 18% IE, 24% PHE and 6% HPHE. There is a potential to create new job positions by implementing this technology, because high-level skills and specific training are required for handling strong acids and to control the acidification. New regulations are required for handling acid and permission to acidify in stables (e.g., not allowed in Sweden). A new source of damage to workers is expected with this technology since farmers will have to handle acid, although with training the risks are very low. Since methane and ammonia emissions are expected to decrease, odour is also expected to be reduced. The technology will also contribute to a healthier work environment. On the other hand, additional nitrate leaching is possible because more N is contained in the slurry. Phosphorus leaching can increase since P becomes more mobile, and the whole process to acidify the slurry consumes electricity for pumps and the production of sulphuric acid contributes to a higher electricity consumption.



Legend: VCA = value chain actors; GHG = Greenhouse gas; N_2O = nitrous oxide; CH_4 = methane; NH_3 = ammonia; CO_2 = carbon dioxide

Figure 39. sLCA: Assessment results - 1

Social assessment of the potential impacts of solutions used to recover nutrients from agricultural and livestock practices

Nutri2Cycle
Nurturing the Circular Economy

- ***RL4.LL20 'Low-temperature ammonium-stripping'*** It has been scored as 35 % in HPBE, 24 % PBE, 24 % IE, 12 % PHE and 6 % in HPHE. It is expected that new job positions will be created since this technology will need technicians for installation and maintenance. Training farmers to operate the technology is important, and technicians must be trained to maintain the plant. This technology was developed to work automatically and to be controlled remotely, requiring only a brief supervision, which can save some work time, but it is still recommended that a technician operates and checks the proper functioning of the plant. When the technology is correctly used, no air pollution is expected from the reaction of ammonia and sulphur dioxide, but in case it is not this can be considered a potential source of damage to workers. Proper handling of acidic or basic substances will prevent personal injury. The main aim of this technology is the recovery of ammonia from livestock manure, avoiding manure storage in open pits for long periods, and uncontrolled ammonia emission to atmosphere, consequently reducing odour on the farm.
- ***RL3.LL30 'Precision farming coping with heterogeneous qualities of organic fertilisers in the whole chain'*** It has been scored as 41% HPBE, 35% PBE, 24% IE, PHE and HPHE 0%. Jobs will not be created due to this technology, but possibly, in case of a wide dissemination of the technology, new jobs can be created in the production units. The Association of German Agricultural Analytic and Research Institutes (VDLUFA) recommends a frequent control and cleaning of the optical window for the usage of NIR sensor where necessary and the daily inspection and maintenance should be done by specialised staff. In addition, the technicians should have some skills for maintenance and installation. At the current state, the Fertilisation Ordinance does not accept NIRS-technology for the quantification/control of applied nutrients. Some federal states however have "accepted" the technology. Thus, regulation is required as far as acceptable technology standards are concerned. NIR-sensors for nutrient detection in liquid manure are often combined with other techniques, such as cultivation or incorporation of the manure into the soil, so that losses via volatilisation of nitrogen (ammonia), can be minimised. In addition, NIRS optimises nutrient supply according to plant needs and site-specific conditions and reduces N-surplus which leads to lower nitrate leaching into the groundwater, consequently contributing to improve water quality. Finally, potential positive effects on yield via increased nutrient use efficiency is expected, and negative effects of overfertilisation of crops can be minimised, such as fungal diseases, reduced steadiness of cereals.
- ***RL5.LL40 'Insect breeding as an alternative protein source on solid agro-residues (manure and plant wastes)'*** It has been scored as 24% HPBE, 35% PBE, 12% IE, 24% PHE and 6% HPHE. This new market can create new job positions, and training is required since employees need to know how the insect facility works in practice. Furthermore, insects are a new product that will be used to feed animals, therefore, understanding the product and animal's needs is essential. After the worker has established the correct amount and diet for the insects, it requires the same time as any other feeding activity. Currently, processing manure with insects and using them for feed or food is illegal. Therefore, new regulations are needed before it can be implemented. Insect protein and fat is produced in the process, which can replace imported protein which is often not produced in a sustainable way. On the other hand,

harmful gasses such as NH_3 are emitted, representing a source of damage for workers, and some black soldier fly producers get complaints about odour. In addition, for the cleaning of crates and the insect rearing facility, extra water is needed, contributing to higher consumption and insect breeding facilities require a high amount of energy. Phosphorus and nitrate are recaptured in the larvae contributing to reducing P and NO_3 leaching, and the technology can contribute to producing feed for animals. Finally, insects represent an innovative food and feed source rich in high quality protein as well as other beneficial nutritional ingredients such as fat, minerals, and vitamins. Despite traditional knowledge about insects and their harvest in the wild, for the industrial mass production of safe insects and insect products for consumption and for processing into food and feed, the development of rearing, harvest as well as post-harvest technologies is required.

- ***RL12.LL41 'Floating wetland plants grown on liquid agro-residues as a new source of proteins'*** It has been scored as 24% HPBE, 41% PBE, 29% IE, 6% PHE and 0% HPHE. This new market can create new job positions, but it does not require high-level skills from workers involved. Training workers is required to know the cultivation of duckweed and how to use the end-product. The technology can contribute to new regulations since the final product will be used as animal feed. Proteins are of high need in Europe due to an intensive livestock production. Thus, this technology can reduce the import of animal feed from countries with high social and environmental impact. In addition, nutrients (nitrates and phosphorus) will be recovered from water, improving its quality, and since water recirculates during the process, water consumption will be reduced. Energy consumption will be decreased once the requires less energy than protein crops already produced or importing soybean as feed. Finally, a higher protein production per hectare is a sustainable option compared with alternative for imported soybean meal, and a valorisation of useful nutrients in liquid agro-residues with a high efficiency is also very relevant with this technology.
- ***RL4.LL49 'Nitrogen and phosphorus recovery from pig manure via struvite crystallisation and design of struvite based tailor-made fertilisers'*** It has been scored as 35% HPBE, 53% PBE, 6% IE, 6% PHE and 0% HPHE. In case a crystallisation plant is installed at a farm or anaerobic digestion plant, new workers will be needed for the operation and maintenance of the plant. Operation and maintenance of the plant requires only unskilled personnel, although plant personnel require prior training in its handling. Farmers can save some labour since the plant can run alongside other farm operations or the anaerobic digestion plant. Currently, struvite can only be used in a few countries. Therefore, the adoption of this technology will contribute to the development of new legislation. The N present in digestate or manure is efficiently recovered in the form of struvite, reducing ammonia emissions and odour, contributing to a healthier environment for workers and the local community. By removing most of the N and P from the digestate, the final effluent will be found to have lower nutrient concentrations and could therefore be suitable for fertigation, contributing to reducing water consumption for irrigation. Energy consumption in the handling, transport, and management of digestate is avoided, and contributes to reduce GHG emissions. Struvite has been found to be a good fertiliser and provides essential nutrients such as magnesium, nitrogen and phosphorus for agriculture and horticulture. Another factor that supports the use of struvite as a fertiliser is

its low concentration in heavy metals compared to the phosphate rock usually used in the manufacture of synthetic fertilisers.

- ***RL4.LL55 'Manure processing and replacing mineral fertilisers – The Netherlands'*** It has been scored as 29% HPBE, 35% PBE, 29% IE, 6% PHE and 0% HPHE. Processing manure and replacing mineral fertilisers requires high-level skilled workers to provide a good product and enough nutrients for the crop. Thus, the technology can contribute to creating new job positions and to attracting high-level skills workers to agriculture. Regulation for manure application should be improved to be fairer compared to inorganic fertilisers. In addition, the use of organic fertilisers will reduce the carbon footprint because the production of mineral fertilisers is avoided. On the other hand, the application of organic fertilisers could increase NH_3 emissions if no other actions are taken to prevent this. Finally, the underlying principle of the project is to use animal manure more efficiently, by processing the nitrogen, phosphorus and organic matter rich components of manure into separate products, that can be applied more efficiently, contributing to a science purpose and new knowledge.

According to the ILO (2021), insufficient labour inspections and a lack of hazard training are causes of accidents and incidents in agriculture. Thus, promoting training and development for workers and creating new regulations is essential if accidents and incidents are to be avoided following the implementation of the technologies.

In addition, the precise application of nitrogen according to plant demand, the avoidance of nitrogen pollution from the environment and the use of organic fertilisers will help to reduce the consumption of mineral fertilisers, decreasing the pressure on natural resources. Improving nutrient efficiency by increasing the use of organic fertilisers and reducing losses will contribute to a decrease in the import of mineral fertilisers, consequently reducing potential social impacts in the value chain.

It is also suggested that novel technologies can create new workplace opportunities in rural communities. Again, the way in which the technology will impact this indicator should be assessed in a specific analysis because it will depend on the working conditions. For instance, a company could train an employee, not necessarily with high-level skills, to work with the technology or they can hire another worker with experience in that technology. More research and greater maturity of the technologies are needed before an effect on this indicator can be documented.

With regards to working time, labour-saving technologies (i.e., precision fertilisation and adoption of other machinery) are in demand due to the complex, highly variable environment in agriculture, and can lead to increased productivity and quality of agricultural output, and reduced dependence on non-skilled human labour, as well as improved environmental control (Gallardo and Sauer 2018). For instance, the time, effort, and energy use in a small family homestead differs significantly from that on a large commercial livestock farm. Thus, full-time employees work a little under 35 hours and part-time workers typically work around 20 hours a week. However, farmers who own their own businesses usually work much more than 44 hours a week (Bureau of Labor Statistics, 2021). Umstätter et al. (2018) claim that the working hours per person have tended to remain stable with technological progress since the resulting reduction in working time is being used for other activities. Thus, it is hard to make predictions for the indicator on working time since it depends greatly on the farm and work conditions.

Finally, it could be argued that there is some overlap between social and environmental indicators; however, their inclusion is deemed important in a social assessment because it results in greater focus on the social consequences of environmental damage, while in environmental LCA the focus is on quantifying the damage.

3.3.1.4 Complementary quantitative sLCA adapting environmental inventory and costs

In this section, we provide a complementary social assessment considering required inputs (e.g., materials, water, electricity and machinery) and costs associated to the application of the technology, using social flows retrieved from the social database Product Social Impact Life Cycle Assessment (PSILCA) (Maister et al., 2020). In that sense, the technology RL4.LL20, 'Low temperature ammonium-stripping using vacuum' (led by IRTA, Spain) was analysed. The technology removes nitrogen from pig slurry using vacuum stripping and the final products are an ammonia salt solution that can be reused as a fertiliser, and organic fertiliser with less nitrogen content, which in turn improves further management of nutrients, and facilitates final disposal of the treated slurry.

3.3.1.4.1 PSILCA database

The PSILCA database is a global, consistent database, suitable to assess social impacts of products, along product life cycles, providing generic information on social aspects in country-sector combinations and commodities that can be used for screening purposes to identify high-risk regions (Maister et al., 2020; UNEP, 2020). In PSILCA, the social flows (i.e., sector and country-specific data) are obtained from international institutions (e.g., World Bank, OECD) and attributed to the selected product systems and indicators. Using PSILCA, it is possible 'to measure' how externalities (e.g. corruption, child labour, trade unionism) affect or can be affected by the product being assessed (Kono et al., 2018; Martin & Herlaar et al., 2021). PSILCA version 3, the version applied in the current deliverable, uses the multi-regional input/output database EORA, 2019 version, which covers the entire world economy. As with EORA, PSILCA uses money flows to link processes providing social impacts for around 15,000 sectors in 189 countries (Maister et al., 2020).

In the current study, 31 qualitative and quantitative indicators from PSILCA were used to calculate the social impacts of a novel technology for nutrient recovery in agriculture and to identify potential social hotspots in the product systems (Maister et al., 2020). The indicators address stakeholders such as workers, local community, society and value chain actors. 30 indicators represent potential risks (negative impacts) and 1 indicator 'Contribution of the sector to economic development' represents an opportunity (positive impact).

A cut-off of 1E-05 was applied in the impact analysis, which is the maximum detail in the version 'starter' of PSILCA (Maister et al., 2020). The results included all the sectors up to the fifth level of upstream processes, which is sufficient for the current study and no further modification was made to the product systems or indicator values provided by PSILCA.

The impact assessment was performed in the free software openLCA using the Social Impacts Weighting method from PSILCA, applying characterisation factors to each indicator according to the risks or opportunities created (Table 37). The assignment of risk and opportunity levels was based on international conventions and standards, labour laws, expert opinions, and the literature (Maister et al., 2020). 'Medium risk hours' (med risk hours) is the unit that represents the total risk involved in producing the output.

The total impact of the product is provided by the sum of all risks minus the potential opportunity of the product developed.

3.3.1.5 Goal and scope

The functional unit of the system is 1 m^3 of treated slurry, and the impacts were assessed from cradle-to-gate. A 10-year life span was used for machinery; a 20-year life span for the concrete pit.

Inventory data was collected in the field. Costs of the technology and social flows from the PSILCA database (Maister et al., 2020) were used to estimate the social impacts caused by the technology. The methodology adapted from the study of Serreli et al. (2021), in which the inputs to the system were used in PSILCA as economic values. It is important to highlight that we assessed the social impacts of producing and using the technology in a country-level since PSILCA provides sector and country-level data. As did in Werker et al. (2019), some social impact categories were excluded of the social assessment since they are covered in the environmental LCA of the technology.

Table 37. sLCA: Characterisation factors for the Social Impacts Weighting method in PSILCA

Retrieved from Maister et al. (2020)

Nature of indicator	Level	Factor
Risk	Very low	0.01
	Low	0.1
	Medium	1
	High	10
	Very high	100
	No risk	0
Risk/Opportunity	No data	0.1
Opportunity	Low	0.1
	Medium	1
	High	10
	No opportunity	0

3.3.1.6 Inventory

Most of the inventory (amounts and costs) was collected directly with the technology developer. When it was impossible to obtain the primary cost data, we adopted the same strategy as described in Serrelli et al. (2021), searching prices in well-known sources. It is important to highlight that PSILCA works on dollar, thus costs in euro (year base 2019) were converted to dollar (year base 2019).

3.3.1.7 Impact assessment

The total impact of the technology is 6.37 medium risk hours per 1 m^3 of treated slurry. 65% of total impact is concentrated in four impact categories: 27% in 'fair salary - Workers' (1.76), 22% in 'freedom of association and collective bargaining - Workers' (1.42), 9% in 'corruption – Value chain actors' (0.58), and 7% 'value added (total) – Value Chain Actors' (0.48) (Figure 40)

Main processes responsible for those impacts in 'fair salary' were related to the high risk of living wage, meaning that the workers are not paid well enough, in the flows 'manufacture of machinery and equipment' - representing all equipment used in the technology - and 'computer and related services' - used to represent plant automation. The impact caused in 'freedom of association and collective bargaining' is also mainly due to 'manufacture of machinery and equipment' and 'computer and related services', in this case, the very high risk is in the trade union density, that is, workers are not well involved in trade unions which could represent a benefit for them. The impact category 'Corruption' is represented by the subcategories 'active involvement of enterprises in corruption and bribery' and 'public sector corruption'. In the first, the processes 'construction' – used to represent the infrastructure – and 'other land transport; transport via pipelines' have a very high risk; in the second, the processes 'manufacture of machinery and equipment' and 'computer and related services' is the one with a high risk of corruption. Finally, the very high risk in the processes 'metal products' and 'metallurgy products' were mainly contributing to the impact category 'value added (total)'. Lowest impacts (in medium risk hours) were found in 'men in the sectoral labour force - Workers' (6.89E-04), 'fatal accidents – Workers' (2.90E-04) and 'frequency of forced labour – Workers' (2.14E-04).

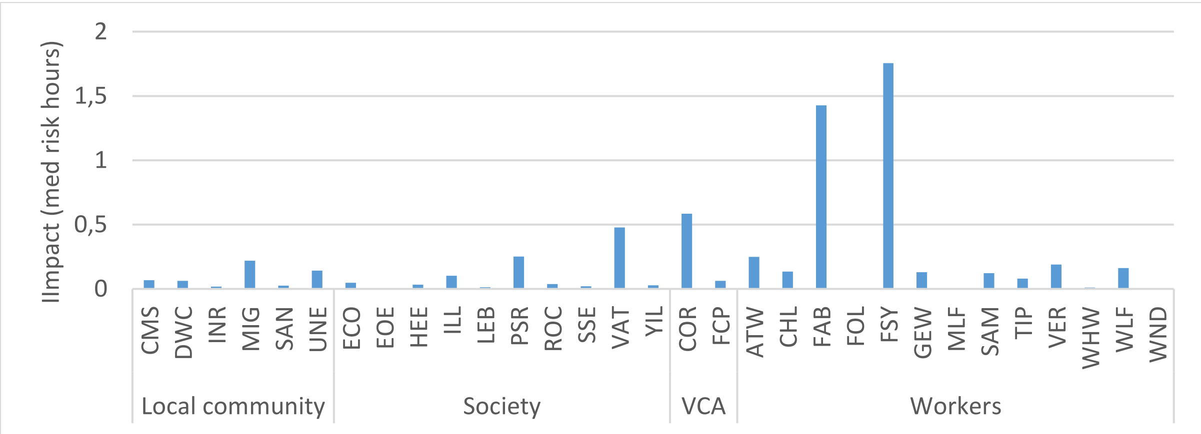


Figure 40. sLCA: Assessment results -2

Overview of results for stakeholder group and indicators for the novel technology for ammonia recovery from livestock

Legend: ATW: accidents at work. CHL: child labour. CMS: Certified environmental management system. COR: public sector corruption. DWC: drinking water coverage. ECO: contribution of the sector to economic development. EOE: expenditures on education. FAB: freedom of association and collective bargaining. FCP: fair competition. FOL: forced labour. FSY: fair salary. GEW: gender wage gap. HEE: health expenditure. ILL: illiteracy. IMS: international migrant stock. INR: indigenous rights. LEB: Life expectance at birth. MIG: migration. MLF: men in the sectoral labour force. POL: pollution. PSR: promoting social responsibility. ROC: risk of conflicts. SAM: safety measures. SAN: sanitation coverage. SSE: social security expenditures. TIP: trafficking in persons. UNE: unemployment. VAT: value added (total). VER: violations of employment laws and regulations. WHW: weekly hours of work per employee. WLF: women in the sectoral labour force. WND: workers affected by natural disasters. YIL: youth illiteracy. VCA : Value chain actors

Although most of the impacts are due to the activity in the main country (Spain), various processes around the world can also contribute to the total impact of the product. Thus, for instance, even if there is a very low risk of child labour in Spain, there is a contribution in the total impact of the product due to the 'coal and lignite products' processed in South Africa, and to the 'mining and quarrying (energy)' process located in Russia (Figure 41). Those impacts are called non-domestic impacts and can only be seen when considering the whole life cycle of the product.

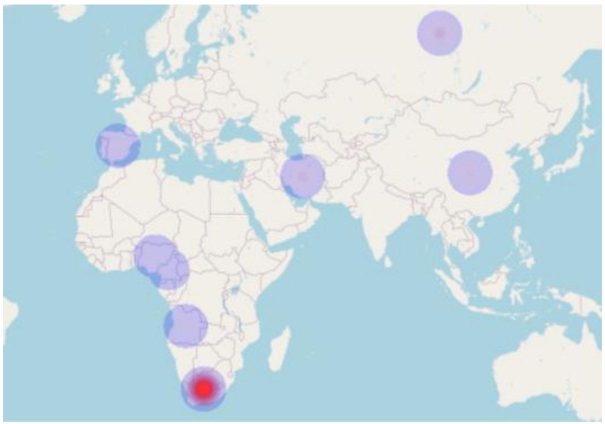


Figure 41. sLCA: Assessment results – 3

Domestic and non-domestic impacts (blue - red circles) in the impact category 'child labour' of the novel technology for ammonia developed in Spain

The social impacts assessed in the current deliverable for the 'Low-temperature ammonium-stripping' technology are related to the inputs required to develop the technology, that is why we considered impacts from the cradle-to-gate. The impacts related to the performance of the technology in a bigger system, that means, when the technology achieves a higher level of adaptation in the society must be also assessed, using specific data and adding more social indicators. Thus, the whole life cycle of the livestock system can be better checked for social impacts that is caused along it.

3.3.2 Conclusion

The aim of the present study was to select and test indicators in order to perform a sLCA of novel Nutri2Cycle technologies / solutions to be applied in agro-food systems for improved nutrient recovery and recycling. A set of indicators enables the assessment of potential social hotspots and opportunities related to novel technologies applied in agriculture to recover nutrients and improve of nutrient efficiency.

Through the questionnaire and expert knowledge, examples of potential impacts of the technologies included the need for highly skilled workers, attracting a highly qualified labour force to agriculture, increasing training and employee development, improving the efficiency of the technologies. Some technologies also helped reducing accidents at work. Moreover, they will impulse the need for new regulations to deal with organic fertilisers more effectively. In addition, novel ways to properly deal with manure can result in a reduction in odour and other gases in the local community. Finally, many technologies can also contribute to new knowledge and scientific research to improve agriculture. Other indicators, such as new jobs or a reduction in extra hours at the farms, were site-dependent and varied depending on the company or farmer behaviour.

The inclusion of novel technologies may introduce new sources of damage, for instance, when using acids or working with heavy machinery, although these risks are controllable.

Qualitative assessments for prospective studies in sLCA may be a starting point for predicting the potential benefits and harmful effects of novel technologies. Finally, we would like to encourage that social assessment is included in case studies assessing the sustainability of agricultural technology, which will help making more complete assessment of the sustainability. Furthermore, it will help improving the databases and methods for such assessments.

For future work, also depending on the maturity of the technologies, wherever possible a full sLCA of technology, either standalone or in the context in which is applied, should be undertaken, in order to provide quantitative ranges for each indicator, as it was provided for 'Low temperature ammonium-stripping using vacuum'. In addition, advancing on weighting social and environmental indicators in simultaneous assessments is essential to compare or aggregate results from the two dimensions.

3.3.3 References

Albaum, G., 1997. The Likert scale revisited: An alternate version. *Int. J. Mark. Res*. 39, 331–348. https://doi.org/10.1177/147078539703900202

Andrade, E.P. Bonmati, A., Esteller, L.J., Vallejo, A.A., 2022. Assessment of social aspects across Europe resulting from the insertion of technologies for nutrient recovery and recycling in agriculture. *Sustainable Production and Consumption*, 31, 52-66, https://doi.org/10.1016/j.spc.2022.01.025

Arcese, G., Lucchetti, M.C., Massa, I., Valente, C., 2018. State of the art in S-LCA: integrating literature review and automatic text analysis. *Int. J. Life Cycle Assess*. 23, 394–405. https://doi.org/10.1007/s11367-016-1082-0

Arvidsson, R., 2019. On the use of ordinal scoring scales in social life cycle assessment. *Int. J. Life Cycle Assess*. 24, 604–606. https://doi.org/10.1007/s11367-018-1557-2

Brundtland, G.H, 1987. The Brundtland report: 'Our common future.' Med. War 4, 17–25. https://doi.org/10.1080/07488008808408783

Bureau of Labor Statistics, 2021. U.S. Department of Labor, Occupational Outlook Handbook, Agricultural Workers. https://www.bls.gov/ooh/farming-fishing-and-forestry/agricultural-workers.htm

Chen, W., Holden, N.M., 2017. Social life cycle assessment of average Irish dairy farm. *Int. J. Life Cycle Assess.* 22, 1459–1472. https://doi.org/10.1007/s11367-016-1250-2

Darnhofer, I., Fairweather, J., Moller, H., 2010. Assessing a farm's sustainability: Insights from resilience thinking. *Int. J. Agric. Sustain*. 8, 186–198. https://doi.org/10.3763/ijas.2010.0480

Franze, J., Ciroth, A., 2011. A comparison of cut roses from Ecuador and the Netherlands. Int. *J. Life Cycle Assess.* 16, 366–379. https://doi.org/10.1007/s11367-011-0266-x

Gallardo, R.K., Sauer, J., 2018. Adoption of Labor-Saving Technologies in Agriculture. Annu. Rev. Resour. Econ. 10, 185–206. https://doi.org/10.1146/annurev-resource-100517-023018

Hauschild, M.Z., Dreyer, L.C., Jørgensen, A., 2008. Assessing social impacts in a life cycle perspective-Lessons learned. CIRP Ann. - Manuf. Technol. 57, 21–24. https://doi.org/10.1016/j.cirp.2008.03.002

ILO, 2021. Agriculture: a hazardous work. International Labour Organization. https://www.ilo.org/safework/areasofwork/hazardous-work/WCMS_110188/lang--en/index.htm

Kono, J., Ostermeyer, Y., Wallbaum, H., 2018. Trade-Off between the Social and Environmental Performance of Green Concrete: The Case of 6 Countries. *Sustainability* 10. https://doi.org/10.3390/su10072309

Kühnen, M., Hahn, R., 2017. Indicators in Social Life Cycle Assessment: A Review of Frameworks, Theories, and Empirical Experience. *J. Ind. Ecol*. 21, 1547–1565. https://doi.org/10.1111/jiec.12663

Maister, K., Di Noi, C., Ciroth, A., Srocka, M., 2020. PSILCA v.3.

Martin, M., Herlaar, S., 2021. Environmental and social performance of valorizing waste wool for sweater production. *Sustain. Prod. Consum*. 25, 425–438. https://doi.org/10.1016/j.spc.2020.11.023

Pelletier, N., 2018. Social sustainability assessment of Canadian egg production facilities: Methods, analysis, and recommendations. *Sustain.* 10, 1–17. https://doi.org/10.3390/su10051601

Pereira Andrade, E., Bonmati, A., Jimenez Esteller, L., Anton Vallejo, A. 2022. Assessment of social aspects across Europe resulting from the insertion of technologies for nutrient recovery and recycling in agriculture. *Sustainable Production and Consumption*, 31, 52-66. https://doi.org/10.1016/j.spc.2022.01.025.

Ramirez, P.K.S., Petti, L., Brones, F., Ugaya, C.M.L., 2016. Subcategory assessment method for social life cycle assessment. Part 2: application in Natura's cocoa soap. *Int. J. Life Cycle Assess.* 21, 106–117. https://doi.org/10.1007/s11367-015-0964-x

Serreli, M., Petti, L., Raggi, A., Simboli, A., Iuliano, G., 2021. Social life cycle assessment of an innovative industrial wastewater treatment plant. *Int. J. Life Cycle Assess.* 26, 1878–1899. https://doi.org/10.1007/s11367-021-01942-w

Siebert, A., O'Keeffe, S., Bezama, A., Zeug, W., Thrän, D., 2018. How not to compare apples and oranges: Generate context-specific performance reference points for a social life cycle assessment model. J. Clean. Prod. 198, 587–600. https://doi.org/10.1016/j.jclepro.2018.06.298

Umstätter, C., Stark, R., Schmid, D., Schick, M., 2016. Auswirkung des technischen Fortschritts auf die Arbeitszeit in der Landwirtschaft. Agrar. Schweiz 7, 204–209

UNEP, 2020. Guidelines for Social Life Cycle Assessment of Products and Organizations 2020. Norris, C., Traverso, M., Neugebauer, S., Ekener, E., Schaubroeck, T., Garrido, S., Berger, M., Valdivia, S., Lehmann, A., Finkbeiner, M., Arcese, G., 2020. (eds.). United Nations Environment Programme (UNEP).

Werker, J., Wulf, C., Zapp, P., 2019b. Working conditions in hydrogen production: A social life cycle assessment. *J. Ind. Ecol*. 23. https://doi.org/10.1111/jiec.12840

4 Discussion and overall conclusions

4.1 Life cycle assessment studies

Given the different functional units, system boundaries and technologies which are analysed in the individual life cycle assessments in the current deliverable, it is difficult or even impossible to give general conclusions about which technology performs the best for improving C, N and P recovery and recycling. Such a comparison is actually not even be desirable as the assessed solutions are addressing completely different environmental problems, which is also reflected in the different functional units and system boundaries, and therefore they are not comparable or competing.

The environmental performance of the technologies or solutions assessed in this report is always compared to a baseline, allowing us to quantify whether the technology is benefitting the environment in relative terms. For each environmental impact we can compare which technologies/ solution improve or reduce the impact, and which make no difference or increase/worsen the environmental impact.

However, the selection of baseline scenario is critical. For instance, we may consider that a new technology, e.g. a slurry treatment, results in the recovery of a high-quality fertiliser. The obtained organic fertiliser could be compared with the baseline scenario of synthetic fertilisation, which likely holds better results, because fertiliser manufacturing is very efficient and mature (case for LL#20). If the baseline scenario was raw slurry instead, impact results could depend on the distance between farm and field. We conclude that there is not Black and White solution, but that tailored solutions should be developed to meet specific requirements and the results depends on what the default reference is.

In Figure 42, we have summarised the results of all the eLCA studies. Each technology is evaluated per impact category and in terms of whether the technology performed better, worse, or similar to the baseline. The studies and scenarios and the baselines, against which they are compared to, are summarised in Table 38 (serves as a legend to Figure 42). It should be noted that the total height of columns indicates how many of eLCA studies included that respective impact category. The number of impact categories varies across solutions, with most/all of them including i.e. *acidification*, *climate change* and *eutrophication freshwater*, while only some included *land use*, *resource use of minerals* and *water use*; this will depend on relevance for the particular solution.

The figure shows that no general conclusions can be made with regards to the performance of the suggested solutions tested in the assessments. Each technology results in different environmental impact patterns, with some technologies decreasing the environmental impact potential only for some impact categories and some decreasing it in all categories. Nevertheless, some tendencies can be seen across the different technology assessments.

In terms of *acidification potential* and *climate change potential*, the tested solutions in many cases (14 out of 23 for *acidification potential* and 15 out of 22 for *climate change potential*) reduced the impact potentials. With regards to *non-cancer human toxicity* and *fossil and minerals and metals resource use*, the tested solutions performed worse than their currently established baselines in many cases (in 10 out of 14 for *non-cancer human toxicity* and 10 out of 15 for *fossil and minerals and metals resource use*). This might be because many of the technologies assessed have been developed to mitigate greenhouse gases and to reduce nitrogen losses through recycling. This reduces the impacts related to nutrients and climate change potential. However, in many cases that comes at a cost such as energy consumption for heating, pumping or transportation and use of chemicals or materials such as for example sulphuric acid and polymers. The production of these inputs is often associated with the use of energy resources and toxicity impacts.

It is rather surprising that for the eutrophication categories, i.e. *marine eutrophication*, *freshwater eutrophication* and *terrestrial eutrophication* it appears that the technologies are increasing and decreasing the impacts in about equally many cases. This may because the eutrophication impacts are caused by upstream or downstream processes, which may have been ignored or forgotten when developing the technology.

It should be noted that each environmental LCA study counts differently in Figure 42, as each study includes at least one technology (as an alternative to the baseline), but in many cases, several different technology *scenarios*. In cases where the scenarios are similar, this can lead to an over-representation of a study in one compartment of the impact category. For example, the technology of slurry acidification LL#18 is represented by three scenarios (one each for Denmark, the Netherlands and Spain) meaning that this technology weighs three times higher in this analysis although it is very much the same technology. In other studies, scenarios can represent completely different technologies, and hence it is not as such an over-representation.

It is also important to note that in the analysis performed above, each scenario and each impact category is given equal weight. Further, it cannot be seen how intense a decrease or increase in an impact category was. It can therefore be that while it seems to be a balance between benefits and drawbacks, it might well be that all benefits are huge, while all drawbacks are minor. This may give a bias in Figure 42.

In spite of these potential biases, we still think that this representation helps interpretation of the overall results. The individual environmental / life cycle assessments were never selected to be a random sample of technologies or solutions and the current summary chapter should not be seen in isolation of the results of the individual studies.

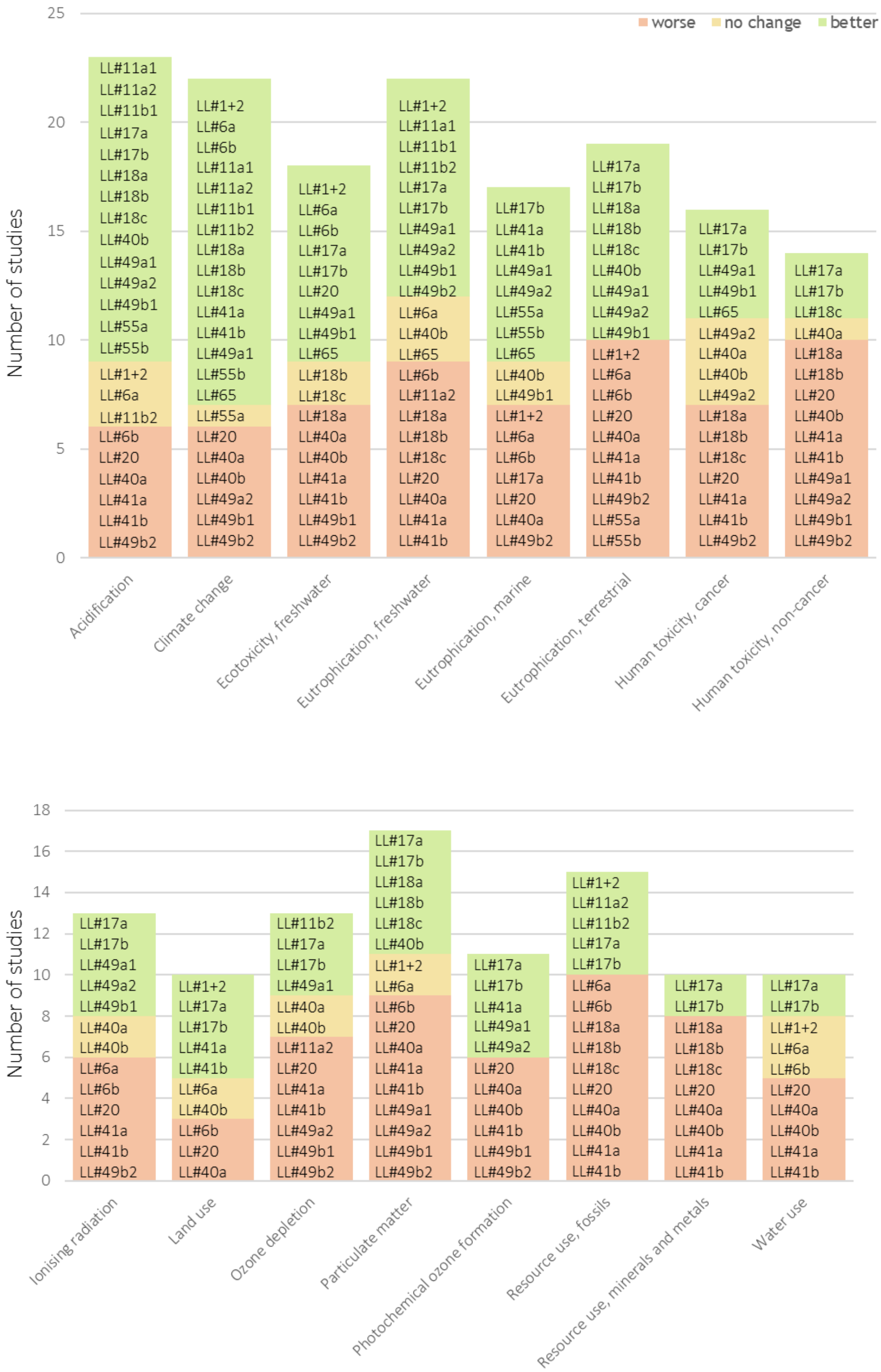


Figure 42. Compiled relative LCA results of all longlist solution studies and their scenarios

Table 38. Overview of longlist technologies, the different scenarios analysed by solution, and the baseline against, which they were compared. Legend to Figure 42

	TECHNOLOGY	BASELINE
LL#1+2	S2: Ammonium stripping / scrubbing and NH_4NO_3 / $(NH_4)_2SO_4$ as substitute for synthetic N fertilisers	• Treatment of LF via nitrification-denitrification (NDN) • Effluent application to local fields (no fertiliser credits) • Sludge application to fields in P deficient regions
LL#6a	S3: Concentrate from vacuum evaporation/stripping as nutrient-rich organic fertiliser: • Combination of micro-filtration and reverse osmosis + evaporator system is used to concentrate the LF • Concentrate from RO+ evaporator system is considered useful fertiliser, whereas permeate is discharged • Filtrate from micro-filtration unit is transported to non-nutrient surplus regions	
LL#6b	S4: vacuum evaporation without a membrane filtration set-up: • Outputs: concentrate (an NK nitrogen-potassium fertiliser substitute), condensed ammonia water (can be used as a denoxing agent in incineration plants) and process water (that is partly recirculated and partly used as cleaning water)	
LL#11a1	solid manure as cow bedding - manual calculations	using straw as bedding material
LL#11a2	solid manure as cow bedding - LCA calculations	
LL#11b1	digested manure as cow bedding - manual calculations	
LL#11b2	digested manure as cow bedding - LCA calculations	
LL#17a	dairy sludge management with aluminium-precipitated sludge (Al-DPS)	landfilling of dairy sludge & application of mineral fertiliser
LL#17b	dairy sludge management with calcium-precipitated lime-stabilised sludge (Ca-DPS)	
LL#18a	pig slurry handling with acidification under Danish conditions	pig slurry handling without acidification in Denmark
LL#18b	pig slurry handling with acidification under Dutch conditions	pig slurry handling without acidification in the Netherlands
LL#18c	pig slurry handling with acidification under Spanish conditions	pig slurry handling without acidification in Spain
LL#20	low temperature ammonium-stripping using vacuum for fertiliser production & manure treatment	mineral fertiliser production
LL#40a	insect breeding as an alternative protein source on solid agro-residues (plant waste diet)	Plant waste field decomposition & protein production from soy
LL#40b	insect breeding as an alternative protein source on solid agro-residues (manure-based diet)	manure composting & protein production from soy
LL#41a	floating wetland plants grown on liquid agro-residues as a new source of proteins (biological treatment + duckweed pond + field application)	biological treatment + field application & import of protein as soy
LL#41b	floating wetland plants grown on liquid agroresidues as a new source of proteins (biological treatment + duckweed pond (medium residence time)	
LL#49a1	N and P recovery from pig manure via struvite crystallization and design of struvite based tailor-made fertilisers UFP - underfertilised pilot plant	raw pig manure field application
LL#49a2	N and P recovery from pig manure via struvite crystallization and design of struvite based tailor-made fertilisers OFP - overfertilised pilot plant	
LL#49b1	N and P recovery from pig manure via struvite crystallization and design of struvite based tailor-made fertilisers UFP - underfertilised industrial plant	
LL#49b2	N and P recovery from pig manure via struvite crystallization and design of struvite based tailor-made fertilisers	

	TECHNOLOGY	BASELINE
	OFP - overfertilised industrial plant	
LL#55a	Sc-2: Raw digestate is processed into • (i) a solid organic fertiliser (transported to Germany over 300 km) • (ii) RO concentrate (a RENURE fertiliser) as synthetic fertiliser on grassland (25 km transportation) • (iii) a residual organic fertiliser (transported to regions with arable farming within the Netherlands for direct use) • (iv) purified water	Raw digestate is transported over a distance of 250 km to regions in Germany with a demand for organic fertiliser
LL#55b	Sc-3: As Sc-2 but with additional treatment step for solid fraction of digestate to produce a P-fertiliser and a low-P soil improver. The soil improver is used to replace peat in substrate or potting soil.	
LL#65	struvite recovery from municipal wastewater	no struvite recovery but wastewater sludge is used as fuel in clinker production

4.2 LCA vs. DBI – comparison and validation

Each of the technology / solution assessment chapters also contained a comparison of dashboard indicator and life cycle assessment results. We can therefore make an overall comparison of DBI and LCA results, in order to analyse the degree of agreement and tendencies of deviations and reflect on what could cause differences. This is important, because assessment using the DBI facilitates rapid appraisal of a new technology, while LCA typically is time-consuming, data-demanding and costly. Therefore, an overall analysis of the DBI vs. LCA results may provide us with useful learnings for where and how to improve guidelines for DBI assessment.

In Figure 43 a summary of the agreement or disagreement between the DBI indicators as assessed by expert judgement and the same indicators based on the full LCA assessment reported in this deliverable is given. The number in each square represents the number of studies or scenarios with agreement or disagreement between the two types of evaluation. We differentiated between positive, negative and no effects and matched the evaluations accordingly. If both methods resulted in the same assessment (i.e., that they were both positive, negative or no effect), we speak of 'agreement', which is coloured green in Figure 43, if the DBI assessment indicated a better performance, than the LCA, we speak of an 'optimistic expert assessment', coloured red, and if the DBI concluded with less favourable results, than the LCA, we speak of a 'pessimistic expert assessment', coloured blue.

Rock phosphate consumption

8	5	2
0	7	0
0	0	0

Soil quality

3	1	7
0	0	3
0	3	0

Nitrate emissions to water

6	0	2
4	5	2
0	0	0

Natural gas consumption

4	4	4
4	5	2
0	0	0

nutrients recovered

1	3	6
0	0	0
0	0	0

P emissions to water

6	1	5
0	4	2
0	0	1

Oil consumption

4	0	3
5	3	3
2	0	2

Ammonia emissions to air

4	0	4
3	3	2
0	2	2

Particulate matter formation

0	0	0
7	4	5
2	0	0

Electricity consumption

6	0	0
5	1	1
5	2	1

Dinitrogen monoxide emissions to air

4	4	3
5	1	1
0	2	0

Carbon footprint

1	1	12
3	0	3
0	0	0

Water

4	0	2
2	4	4
0	0	1

Methane emissions to air

0	1	4
6	1	4
0	0	2

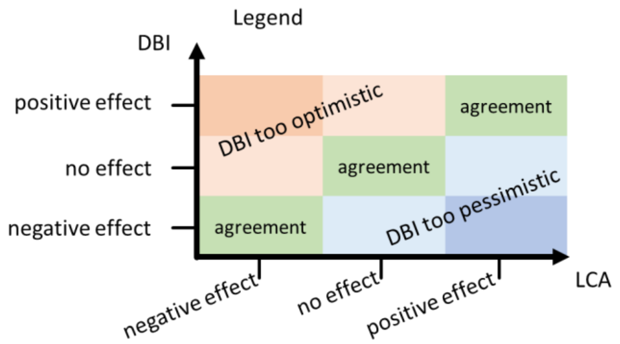


Figure 43. Summary of comparison between DBI and LCA assessment results in the different impact categories.

On the X axis the result of the value of the DBI based on the LCA analysis is indicated as either a negative effect, no effect or a positive effect of the technology. On the Y axis the DBI as assessed by the experts is indicated as either a negative effect, no effect or a positive effect of the technology The values in the squares indicate the number of studies and scenarios that belong to each combination of LCA assessed and expert assessed indicators.

Table 39 gives a summary overview for each impact category of the number times the expert judgement was too optimistic (either a positive effects, where the LCA suggest negative or no effects or no effect, but the LCA suggested negative effects), too pessimistic (either no effect, where the LCA suggested positive effects, or negative effects, where the LCA suggested no or positive effects), or realistic expert judgement, meaning that both methods agreed on suggestion a positive, negative or no effects of implementing the alternative technology compared to the baseline.

The assessment showed that the agreement between DBIs based on expert judgement and the indicators based on the LCA results was less than half (39%), and that in more than one third of cases (43%), the expert assessment of DBI appeared to be too optimistic about the environmental performance of a technology. In about 19% of cases, the expert judgement appeared to be too pessimistic in the assessment of the technology.

Table 39. Summary of DBI vs. LCA results for each impact category and overall

	Optimistic		Realistic		Pessimistic	
	#	%	#	%	#	%
Rock phosphate consumption	13	59%	9	41%	0	0%
Natural gas consumption	12	52%	9	39%	2	9%
Oil consumption	9	41%	8	36%	5	23%
Electricity consumption	11	52%	6	29%	4	19%
Water consumption	6	35%	6	35%	5	29%
Soil quality	4	24%	7	41%	6	35%
Nutrients recovered	4	40%	6	60%	0	0%
Ammonia emissions to air	7	35%	7	35%	6	30%
Dinitrogen monoxide emissions to air	13	65%	4	20%	3	15%
Methane emissions to air	7	39%	5	28%	6	33%
Nitrate emissions to water	10	53%	7	37%	2	11%
P emissions to water	7	37%	9	47%	3	16%
Particulate matter formation	7	39%	6	33%	5	28%
Carbon footprint	5	25%	12	60%	3	15%
Sum & averages	**115**	**43%**	**101**	**39%**	**50**	**19%**

The expert judgement of DBI was particularly over-optimistic about N_2O emissions (65%) and rock phosphate consumption (59%). A good agreement between DBI and LCA was achieved in terms of the carbon footprint of technologies (60%) as well as of nutrient recovery (60%) where equal results were achieved in more than half of the cases. Over-pessimism was detected in a maximum of 35% of cases for soil quality.

One should bear in mind that in the comparative assessment of indicators based on expert judgement and LCA studies we set a benchmark of at least plus/minus 10% change relative to the baseline to speak of a true change of impacts. Setting that benchmark, at 5% or 20% could in some cases perhaps have affected the results.

One major learning from this assessment is that a rapid appraisal or expert interviews might be sufficient for some impact categories (e.g., rock phosphate consumption), but for other categories (e.g., electricity consumption) there is a large risk that a rapid assessment will be misleading. For experts to address these issues it could be advisable to pay attention to the impact categories / indicators with particularly poor agreement and to broaden the perspective to potentially linked processes and secondary effects. Life Cycle Assessment studies facilitate the inclusion of such secondary effects and provides clues about whether achieved benefits in a particular agricultural practice may be nullified through the additional needs of materials or energy required to achieve those benefits. For C, N and P recovery and recycling technologies, such indirect consequences can perhaps be generalised and used for guiding/informing rapid appraisals.

It could also be that those developing, providing and suggesting technology focus on specific benefits such as e.g. reducing N_2O emissions directly resulting from the agricultural practice in question, which is a popular scientific and political topic. Such specific and intensive focus on solving a particular environmental problem can result in an unintentional ignorance of side effects. To overcome such side effects, it might be advisable to include experts in adjacent fields, to ensure that potential pollution swapping effects are avoided.

Another very important difference between the expert judgement and the LCA assessments is that the system boundaries have been well defined in the LCA and all upstream and downstream effects in the background system have been included to the extent possible. In the expert judgements of DBI the system boundaries have not been well defined, but the experts have most likely made a rough system delimitation in their own mind in order to be able to decide what effect to include and the magnitude/direction of these. This means that some of the most obvious up- and downstream effects have been included while some of the more inconspicuous, but perhaps still substantial in terms of some environmental impacts, have been ignored.

In conclusion, the expert assessment of the DBI was able to show some important aspects of the technologies, but it is also clear that important aspects can be missed. Decisions about the implementation of high TRL environmental technologies at a larger scale should always be based on thoroughly conducted LCAs with well-defined system boundaries. However, for initial assessment and prioritization of technologies at a low TRL level, expert judgement of dashboard type indicators may be used, especially if sufficient guidance and information is provided when assessing the more difficult /complex indicators, where up or downstream processes of importance may have an important impact.