Nutri2Cycle

D1.7 Modelling of nutrient flows at farm scale

Deliverable: Modelling of nutrient flows at farm scale

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Abbreviations

AD	Anaerobic Digestion
CAP	Common Agricultural Policy
CAPRI	Common Agricultural Policy Regional Impact
CH_4	Methane
CNP	Carbon Nitrogen Phosphorus
CO_2	Carbon Dioxide
EF	Emission Factor
GHG	Greenhouse Gas
IPCC	Intergovernmental Panel on Climate Change
LCA	Life Cycle Assessment
LL	Long List
LSU	Livestock Units
N	Nitrogen
NH_3	Ammonia
N2C	Nutri2Cycle
N_2O	Nitrous Oxide
NO_x	Nitrogen Oxides
NO_3	Nitrate
NUTS	Nomenclature Des Unités Territoriales Statistiques
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
SON	Soil Organic Nitrogen
OM	Organic Matter
P_2O_5	Phosphorus Pentoxide
P	Phosphorus
RENURE	Recovered Nitrogen from Manure
TAN	Total Ammoniacal Nitrogen

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Glossary

ALFAM2 model: The ALFAM2 model is a semi-empirical (semi-mechanistic) dynamic model that predicts ammonia emission from field-applied slurry in response slurry properties, management, and weather. The figure below shows the general structure of the model.

Anaerobic digestion: A series of biological processes in which microorganisms break down biodegradable material in the absence of oxygen and produce biogas.

ANIMO model: The ANIMO model 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.

Bio-based fertilisers: Organic fertilisers produced from organic residues following some treatment. This would suggest that animal manure is a bio-based fertiliser only following a treatment of the raw manure. Furthermore, bio-based fertilisers may also comprise inorganic materials, e.g., after thermal treatment of organic waste leading to a carbon free ash product.

CAPRI model: The CAPRI modelling system is a global agro-economic model, initiated in 1999, designed for assessing economic and environmental impacts on agriculture at regional level.

Daisy model: Daisy is a mechanistic simulation model of the physical and biological processes in an agricultural field.

Digestate: The material remaining after the anaerobic digestion of a biodegradable feedstock. Anaerobic digestion produces two main products: digestate and biogas.

Life cycle assessment: Life cycle assessment or LCA is a methodology for assessing environmental impacts associated with all the stages of the life cycle of a commercial product, process, or service.

MITERRA-EUROPE: Is a deterministic and static N cycling model which calculates N emissions on an annual basis, using N emission factors and N leaching fractions. A carbon module is also included, which calculates soil organic carbon changes using the default IPCC methodology.

Mitigation technology: Encompasses technologies and practices that can lead to a reduction in greenhouse gas (GHG) emissions or increase the capacity of carbon sinks to absorb GHGs from the atmosphere.

Nutrient cycling: The continued movement and use (with possible temporary accumulations) of nutrients between different compartments (soil, plants, animals, humans, water, air) and trophic levels in the biosphere.

Nutrient uptake: The amount of each nutrient required for the crop to complete its life cycle at a given yield level.

NUTS classification: The nomenclature of territorial units for statistics (Nomenclature des Unités territoriales statistiques – NUTS) is a geographical system, according to which the territory of the European Union is divided into hierarchical levels.

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Nutrient use efficiency: The ratio of the nutrient in desired output (e.g., crop product) divided by the total nutrient input of a system (field, farm, technological unit, region), expressed in kg kg^{-1} or in %.

RothC model: The Rothamsted carbon model (RothC) is a model to assess soil matter turnover, which can indirectly indicate the state of degradation.

Precision fertilisation: A strategy to adapt the fertiliser use based on the crop nutrient requirement. It combines GPS, proximal or remote sensors, and computers on agricultural machinery and tractors in order to observe, measure and respond to spatial and temporal variation in crop nutrient requirements.

Slurry acidification: Acid (e.g., sulphuric acid, lactic acid) is added to slurry to reduce the PH from about 7 to between 5 and 6. When this value is lowered, the ammonia does not evaporate, but instead stays within the slurry. This allows the ammonia to contribute to the slurry but not emit into the atmosphere.

SWAP model: The SWAP model simulates transport of water, solutes and heat in the vadose zone in interaction with vegetation development.

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Executive summary

Models are needed to assess the technologies that have been developed and/or tested in the Nutri2Cycle project as measurements of emissions were in many cases not available. Field-scale models (Daisy model and SWAP-ANIMO) and European scale models (CAPRI and MITERRA-Europe) were used for the quantification of emissions. However, a farm scale model was lacking at the start of the project, while such a model can provide relevant insights into the flows of nutrients and emissions on the farm and provide the linkage between feed, housing, manure storage and field emissions and the nutrient flows. Although many farm-level models have been developed, we did not identify a model that could be used on farms throughout Europe, for different farming systems and focussed on CNP flows and emissions. Therefore, we decided to build a simple farm level model to calculate nutrient flows and emissions that can be used for multiple farming systems in EU member states. The model structure is based on the existing MITERRA-Europe model, which calculates nutrient flows and emissions at the regional level (NUTS2). The farm model, named MITERRA-Farm, was then used to assess the environmental impact of selected Nutri2Cycle technologies for different farms.

MITERRA-Farm downscales MITERRA-Europe to the farm level, by replacing regional-level input data with more detailed farm-level data, while keeping the national/regional emission factors and leaching fractions. In this way, MITERRA-Farm can use a uniform approach to model farms in different regions, and in the meantime accounting for regional differences. To better model some dynamic processes that are strongly influenced by external factors such as climate and input material compositions, MITERRA-Farm also integrates two dynamic sub-models, the ALFAM2 model to simulate NH_3 volatilization following field application of slurry and the RothCN model for soil organic C/N turnover. ALFAM2 is an ammonia emission model, which uses a semi-empirical dynamic approach for predicting ammonia volatilization from field-applied slurry. In the new RothCN model, we extend the RothC model to comprise soil organic nitrogen dynamics as well. We adopted a simple approach to include SON turnover in the simulation by assigning a C:N ratio to each organic compartments. Considering that plant and manure materials have very different C:N ratios, we further divided organic inputs to plant material and manure input.

For the modelling, we used two sets of farm data. We selected five dairy farms from the Dutch project Cows and Opportunities network and five pig farms with arable land from Belgium. For modelling the Nutri2Cycle technologies, we used the Dutch farms for the dairy related technologies and the Belgian farms for pig and arable related technologies. The following Nutri2Cycle technologies were selected for the modelling:

- Slurry acidification to reduce NH_3 volatilization from animal husbandry
- Adapted stable construction for separated collection of solid manure and urine in pig housing
- Small/Farm-scale anaerobic digestion
- N fixation with grass-clover
- Precision farming coping with heterogeneous qualities of organic fertilizers

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Slurry acidification significantly reduced ammonia emissions during storage (-76%) and resulted in less NH_3 volatilization following field application of slurry (about -20%). Due to a higher amount of TAN in acidified slurry, and less NH_3 loss after application, there was a higher N content remaining in the soil after application of acidified slurry as compared to unacidified slurry, which increased the potential for N leaching.

Adapted stable construction to separate urine and faeces reduced and NH_3 emissions during housing and storage. However, there was a side effect of increasing N_2O emissions during storage from the solid fraction. The changes in total gaseous N emissions following field application of urine and faeces to replace slurry/manure were minor, with a reduction in NH_3 volatilization but an increase in N_2O and NO_x emissions.

Farm-scale anaerobic digestion reduced the CH_4 and NH_3 emissions during the storage phase, and the biogas use can further reduce the environmental impact of anaerobic digestion. The difference in N emissions after field application and from soils between digestate and untreated manure was different for N_2O and NH_3. Although N_2O and NO_x emissions from digestate application were lower, the total emissions of those gases were minor compared to the more abundant emissions of NH_3.

The inclusion of N-fixating crops was modelled by replacing grassland with grass-clover leys. N emissions to the atmosphere were reduced in grass-clover leys because of the avoidance of mineral N fertilizers. However, soil denitrification and leaching increased in grass-clover leys compared to grassland. This effect was the result of increased N input to soil by N fixation and clover residues in the model calculation. The results suggest that in these farms, fertilization may be further lowered with grass-clover leys to reduce N losses without compromising yields.

The modelling results on the adoption of precision fertilization were inconsistent. In some farms, there was a clear effect by precision fertilization on reducing N emissions to the atmosphere as well as surface and groundwater. However, in other farms, N losses increased significantly with precision fertilization. These inconsistencies might be related to missing or incorrect information on the fertilization data in the input data, e.g. we did not have information on the N content of the applied slurries.

In this Deliverable also, the results from Nutri2Cycle solution LL71 'soil organic carbon practices for Dutch arable farms' are presented. As for this solution the results were not obtained by experimental work, but through modelling by the RothC model, as measuring changes in soil carbon stocks are difficult to measure, as it takes a long time, and the additional amount of carbon is small compared to the large existing stock. The following SOC practices were selected: use of cover crops; increase share of cereals in crop rotation; substitution of slurry by solid manure; and incorporation of straw. The results showed that the potential of the SOC practices is quite variable amongst regions and amongst farms. For cover crops and substitution of slurry by solid manure the variation amongst regions is limited, but for increasing the share of cereals in the rotation and incorporation of straw large regional differences are observed, as cropping patterns are often also regionally determined. The results show that farm-specific advice regarding SOC practices is needed.

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The results of the simulations with the MITERRA-Farm model for the selected technologies show the added value for modelling C, N and P flows at the farm level, as the modelling provides insight into the variability amongst farms. The benefit of the model is that it uses actual input data that in most cases is available with farmers, which allows potentially widespread use. The development of the model is not concluded and is actively progressing within the Horizon Europe NutriBudget project, which can be considered as a follow-up of the Nutri2Cycle project. For that project, the model will be extended to simulate all major nutrients and heavy metals and additional soil processes such as P leaching. The MITERRA-Farm model will be used as the core model in the so-called NutriPlatform, a decision-support tool to determine which agronomic or policy measures to implement to reach the desired state regarding agronomic and environmental targets.

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1. Introduction

For the assessment of the technologies that have been developed and/or tested in the Nutri2Cycle project models are needed as for many solutions no emission measurements are available. On the one hand, detailed process-based models at the field scale were used to assess the effects on nitrogen emission to the air and leaching to ground and surface water, i.e., the Daisy model (Abrahamsen & Hansen, 2000) and the SWAP/ANIMO model (Groenendijk et al., 2014), and on the other hand, assessment was made at national and European scale using the CAPRI and MITERRA-Europe model (Deliverable 4.2, Rieder et al., 2023).

However, the field scale models are not able to simulate the effects of technologies related to stable or manure storage and the regional scale models often lack the granularity to model the specific processes and the interactions between emissions from manure storage and field application. These models therefore require many simplifications, as detailed farm specific data is mostly not available at regional scale. A farm scale model can provide insights into the flows of nutrients on the farm and provide the linkage between feed, housing, manure storage and field emissions and flows.

At the start of the Nutri2Cycle project, an inventory was made of farm-level models that could potentially be used for the assessment of selected Nutri2Cycle solutions. Although many farm-level models have been developed, we did not identify a model that could be used on farms throughout Europe, for different farming systems and focussed on CNP flows and emissions. The identified models are either national models linked to national databases, developed for a specific livestock sector (e.g., the Dairywise model (Schils et al., 2007) for dairy farms) or only focus on specific emissions, like most carbon footprint models such as the Cool Farm Tool[1]. Therefore, we decided to build a simple farm level model to calculate nutrient flows and emissions that could be used for multiple farming systems in all EU countries. The model structure is based on the existing MITERRA-Europe model, which calculates nutrient flows and emissions in the NUTS2 regions. The farm model, named MITERRA-Farm, has been used to assess the environmental impact of selected Nutri2Cycle technologies for different farms.

This report describes the development and application of the MITERRA-Farm model. In Chapter 1, the model structure and the underlying calculations and sub-models are described. The data set of farm level data to test the model is described in Chapter 2. Next, we applied the model for selected Nutri2Cycle solutions within three farming systems (Chapter 3). In this Deliverable, we also include the results of Nutri2Cycle solution LL71 'soil organic carbon practices on Dutch farms', which is based on RothC modelling for a group of about 50 Dutch arable farms (Chapter 4). In Chapter 5, we discuss the limitations and perspective of the MITERRA-farm model and provide some conclusions.

[1] https://coolfarm.org/

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2. Description of MITERRA-Farm model

The MITERRA-Farm model shares the common basic principles and algorithms with the MITERRA-Europe model. MITERRA-Europe is a deterministic model using emission factors and leaching fractions to estimate emissions from and nutrient flows in agriculture in the EU at Member State and regional levels (NUTS-2) (Velthof et al., 2009; Lesschen et al., 2011). The use of MITERRA-Europe to model carbon, nitrogen and phosphorus (CNP) flows for different farming types at the regional level is reported in the Nutri2Cycle project Deliverable 1.5 *Mapping and characterization of CNP flows and their stoichiometry in main farming systems in Europe* (Duan et al., 2022).

MITERRA-Farm downscales MITERRA-Europe to the farm level by replacing regional-level input data with more detailed farm-level data while keeping the national/regional emission factors and leaching fractions. In this way, MITERRA-Farm can use a uniform approach to model farms in different regions, and in the meantime accounting for regional differences. To better model some dynamic processes that are strongly influenced by external factors such as climate and input material compositions, MITERRA-Farm also integrates dynamic sub-models to simulate NH_3 volatilization following field application of slurry and soil organic C/N turnover.

2.1 Model structure

Figure 1 illustrates the schematic structure of the MITERRA-Farm model. The model tracks nutrient (N and P) flows and calculates balances within the boundary of a single farm. The model simulates a farm consisting of two sectors: the livestock sector in which manure production and associated emissions are accounted for, and the crop sector where crop production and emissions from soil are tracked. MITERRA-Farm is also capable of modelling animal farms with no arable land or arable farm without animals.

In the livestock sector, MITERRA-Farm estimates manure production based on the number of animals and excretion fractions per animal type. Both solid and liquid forms of manure are modelled. The split of solid-liquid fractions is based on the stable system, or determined by national level factors in case no farm specific data is available. Manure and slurry from housed animals are stored or processed before they are applied as fertilizer to arable crops, whereas excretion by grazing animals is directly spread to grassland. The atmospheric emissions during housing, storage, and grazing are calculated separately for solid and liquid manure forms. The default excretion and emission factors are national specific values that are derived from MITERRA-Europe. This means that especially the livestock related emissions from housing and manure storage are often proportional to the number of livestock on a farm, as information on specific housing and manure management systems is not included in the model.

In the crop sector, nutrient additions (C, N and P) to the soil from all sources are recorded, including organic and/or mineral fertilizers, atmospheric N depositions, biological N fixation, and incorporation of crop residues. Crop N uptake is estimated retrospectively from crop yield data of the farm, or regional (NUTS2) values in case no yield information is available, combined with crop specific N content in harvested products and residues and a default efficiency coefficient as used in MITERRA-

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Europe (Velthof et al., 2009). NH_3 emission factors come from the GAINS model[2] (Klimont and Brink, 2004) and are national specific, which account for the already taken emission reduction practices. Soil organic C and N turnovers are modelled by the RothCN sub-model. Ammonia volatilization during fertilization is modelled by the ALFAM2 model, and NO_x and N_2O emissions are estimated using emission factors from the IPCC 2019 guidelines. N runoff to surface water and leaching to groundwater are calculated on the basis of soil N surplus (total N input minus crop uptake and other losses) by applying leaching fractions modified by soil and climate factors.

As we now focus on the nutrient flows and losses within the farm, nutrient exports and imports are not yet implemented in the present version of MITERRA-Farm. For example, manure transfer to neighbouring farms or export to a regional treatment facility – these flows are currently not captured by MITERRA-Farm.

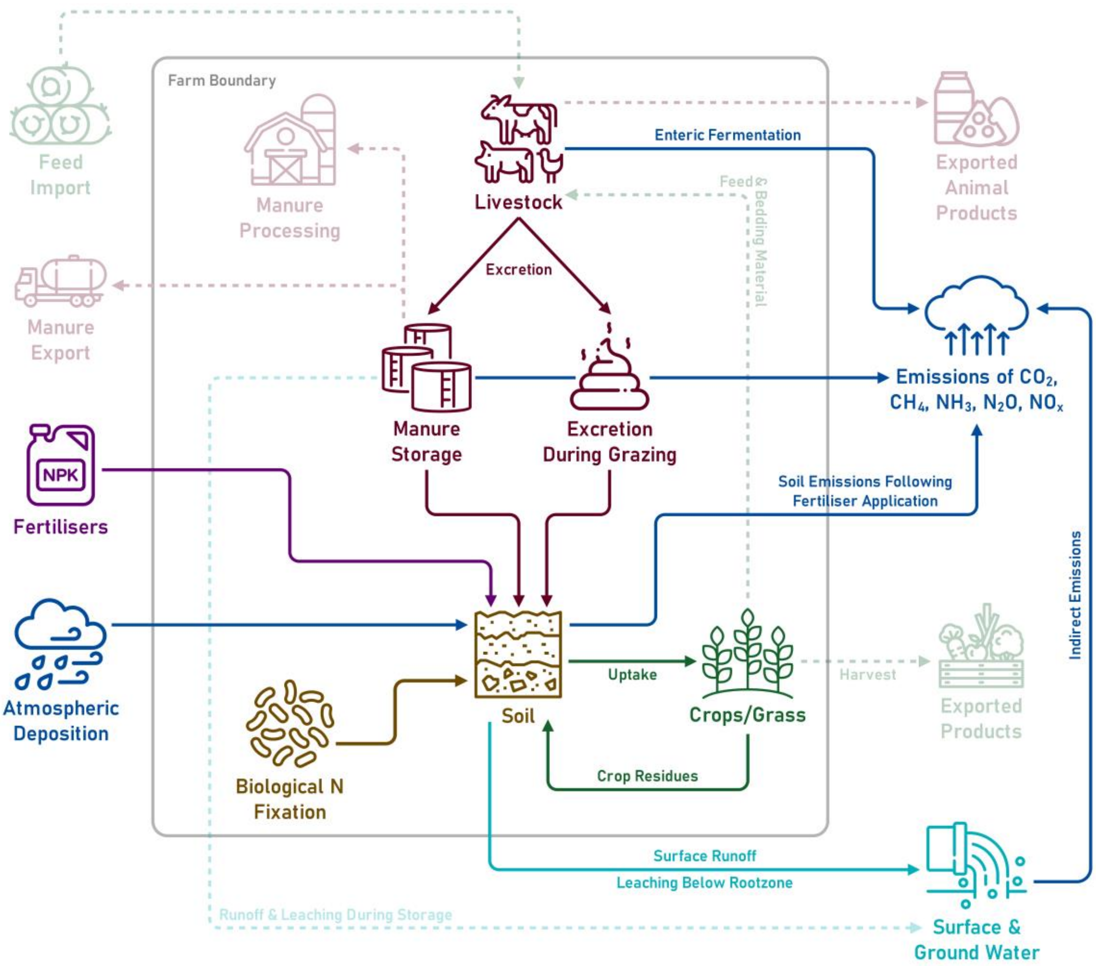

Figure 1: Schematic illustration of the MITERRA-Farm model.

[2] https://gains.iiasa.ac.at/models/

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2.2 The ALFAM2 model

The ALFAM2 model is an ammonia emission model developed by Hafner et al. (2019), which uses a semi-empirical dynamic approach for predicting ammonia volatilization from field-applied slurry. ALFAM2 is fully integrated with MITERRA-Farm as a submodel to estimate NH_3 volatilization of slurry from fertilization and grazing.

ALFAM2 takes into account the infiltration process of slurry in soil after application and splits the applied slurry into two pools: a “fast” pool that is in direct contact with the atmosphere, and a “slow” pool that is less available for emission due to infiltration or other processes. The fast pool can be transferred to the slow pool as a first-order process, and volatilization from each pool follows a first-order kinetics as well. The partitioning fractions and first-order rate constants are determined by a group of modifier parameters, including slurry dry matter, total ammoniacal nitrogen (TAN), slurry pH, application method, incorporation time and depth, air temperature, wind speed, and rainfall rate. The prediction power of ALFAM2 is generally well-established, with a mean error of ca. 12% of applied TAN for 72-hour cumulative emission, although the model also tends to over- or under-estimate emissions for many individual plots. Nonetheless, as national or regional emission factors may be too coarse for farm-level estimation, and farm-specific emissions are difficult to obtain, the ALFAM2 model provides a simple approach to estimate NH_3 volatilization from the field application of slurry.

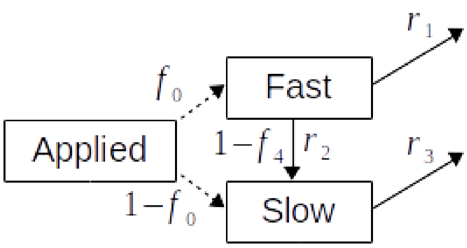

Figure 2: Structure of the ALFAM2 model (Hafner et al., 2019).

In MITERRA-Farm, the slurry application is assumed to be in March or as specified by user input. Livestock excretion produced during grazing is evenly distributed over the grazing months (from April to September). Average temperature, wind speed, and precipitation at the time of slurry application are extracted from regional datasets. Default values for slurry composition (DM, TAN, and pH) are assumed. The final NH_3 volatilization and emission factor are calculated from 72-hour cumulative emission.

2.3 The RothCN model

A new model, RothCN, is developed to simulate the turnover of both soil organic carbon (SOC) and nitrogen (SON) based on the classic Rothamsted Carbon Model (RothC) by Rothamsted Research (https://www.rothamsted.ac.uk/rothamsted-carbon-model-rothc).

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The original RothC model partitions carbon input into two compartments: decomposable plant material (DPM) and resistant plant material (RPM). Each compartment decomposes following a first-order reaction kinetics with its own decay rate and is converted to microbial biomass (BIO) and humus material (HUM), and CO_2 is released during this decomposition (Coleman and Jenkinson, 2014). The decomposition is regulated by environmental factors such as soil type, temperature, moisture content, and plant cover. However, RothC does not model SON turnover.

In the new RothCN model, we adopted a simple approach to include SON turnover in the simulation by assigning a C:N ratio to each organic compartments. Considering that plant and manure materials have very different C:N ratios, we further divided organic inputs to plant material and manure input. Similar to RothC, the plant material input is partitioned into DPM and RPM compartments. For manure input, it is partitioned into decomposable manure material (DMM) and resistant manure material (RMM) compartments, and an additional humified manure material (HMM) compartment to represent the fraction of already decomposed material (Figure 3).

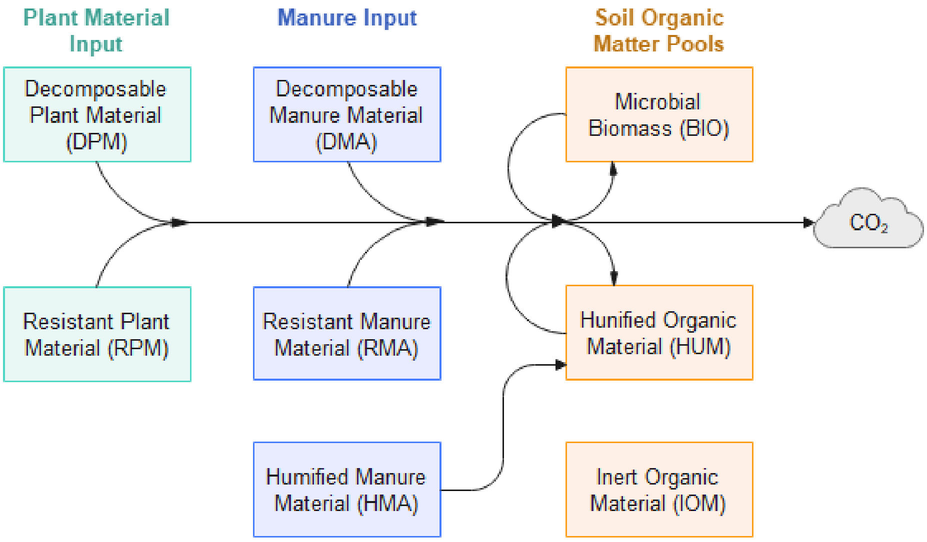

Figure 3: Partitioning and turnover of organic inputs of the RothCN model.

In RothCN, SON turnover is closely coupled with SOC decomposition. During each timestep, the decomposed C from each compartment is calculated per standard RothC algorithm, and N released from decomposition is calculated using compartment-specific C:N ratio. N assimilated into BIO and HUM compartments are also calculated according to the C:N ratio of the respective compartment. Then, the difference between total N release and N assimilation is calculated to determine whether net N mineralization or immobilization should take place. If total N release is greater than N assimilation, then net mineralization takes place, and the N surplus is added to the soil inorganic N pool. If N assimilation requires more N than that is released, then immobilization must take place. In that case, RothCN will first check if the soil inorganic N pool is sufficient for the immobilization requirement. As microorganisms must compete with plants for inorganic N, an availability fraction is

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applied so that immobilization will not deplete the soil of inorganic N. The BIO compartment always has priority to N immobilization over the HUM compartment. If there is not enough inorganic N to immobilize, C assimilation to BIO and HUM compartments will be reduced (deferred) according to the total N available for immobilization and compartment-specific C:N ratios.

2.4 Input data

MITERRA-Farm requires two types of data: a) farm-specific data, including the number of different animals, area and yields of crops, type and amount of fertilizer applications, etc.; and b) optional background data, such as soil characterisation of the fields, and climate of the region where the farm is located. Table 1 summarizes the input data used by MITERRA-Farm.

The farm-specific data captures the major variation in farm typology and management and are essential to model the main nutrient flows in the farm properly. Therefore, farm-specific data are required in most cases. In the event of an incomplete farm dataset, default values from the MITERRA-Europe dataset may be derived for some input entries. For example, if crop yields are not available in the farm dataset, average yields from the country or NUTS2 region will be used as an alternative. However, this will lead to less accurate results.

The background data mainly reflect the regional differences of the farms. In most cases, climate data are not measured at the farm level. Whereas soil characterisation of farm fields is sometimes available, they do not always contain all the required entries by the model. Therefore, we opted to extract these data by default from the MITERRA-Europe datasets at the regional level to avoid critical data gaps. However, a complete background dataset at the farm level allows for a more accurate estimation of farm emissions and nutrient flows when the model is fed with it.

Table 1: Required and optional input data to the MITERRA-Farm model.

Data	Remarks
Livestock	
Number of animals per type	Required.
Crop	
Area per type	Required.
Yields (DM or fresh weight)	Optional. Default yield values for the country may be extracted from MITERRA-Europe datasets if not provided by input.
Fertilization	
Amount per type	Required.
Application time & method	Optional. If not provided, all fertilizers are assumed to be applied in March. Livestock excretion during grazing is spread over the grazing period.
Soil	
Texture, clay content, depth to rock, SOC content	Optional. Default values for the NUTS2 region may be extracted from MITERRA-Europe datasets.
Climate	
Monthly mean temperature, precipitation, wind speed	Optional. Default values for the NUTS2 region may be extracted from MITERRA-Europe datasets.

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2.5 Model output

MITERRA-Farm generates output of the following categories:

Atmospheric emissions

- CH_4 emissions from housing (enteric fermentation) and grazing of animals, and storage of manure.
- NH_3 emissions from housing and grazing of animals, storage of manure, and fertilization.
- N_2O emissions from housing and grazing of animals, storage of manure, and fertilization.
- NO_x emissions from housing and grazing of animals, storage of manure, and fertilization.

Runoff and leaching

- Runoff to N and P to surface water
- Leaching of N to surface and groundwater

Nutrient balances

- N and P balances in animal excretion
- N and P balance in the soil

Soil organic matter

- Long-term SOC and SON turnovers
- Annual SOC and SON balances

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3. Description of farm data

For the simulation of the Nutri2Cycle technologies, we collected a test dataset of farm data. The Nutri2Cycle partners were asked to provide farm data sets for the simulations of the Nutri2Cycle solutions. However, it appeared difficult to obtain the required farm data as most experiments were done at research stations and not at actual farms. While existing data sets from previous projects were often not available due to privacy or data ownership issues. In the end, we obtained two datasets for the testing of the model and the simulations of selected Nutri2Cycle scenarios.

One data set is from the Dutch *Cows and Opportunities* project, a network of 15 dairy farms, of which five farms were selected as example farms for modelling in this study. The other data set is from Belgium, which consists of five dairy farms, five arable farms and five pig farms. From this data set we used the 5 pig farms with arable land for the simulations of solutions for pig farms and arable farms. This last data set was kindly provided by United Experts. For each of the farms we received information on animal numbers, crop areas, and fertilization data. Soil data were not available at field level and instead regional data was used. Climate data was taken from the existing database of MITERRA-Europe. For the Belgian farms, crop yield information was unavailable, and we used the regional NUTS2 data instead. The main properties of these farms are summarised below in Table 2 and 3.

Table 2: Summary of main properties of pig/arable farms used in the modelling.

	BE_Fa1	BE_Fa2	BE_Fa3	BE_Fa4	BE_Fa5
Number of animals	5962 pigs	1037 pigs	4320 pigs	2936 pigs	5014 pigs
Area (ha)	40.2	25.1	235.7	13.0	24.7
Crops	Fodder maize, grass, potato	Fodder maize, maize, grass, vegetables	Sugar beet, winter wheat, winter barley maize, potato, vegetables	Grass	Fodder maize, grass, potato
Fertilization	Pig manure, mineral N	Pig manure, mineral N	Cattle manure, pig manure, compost, mineral N	Pig manure, digestate, mineral N	Pig manure, digestate, mineral N

Table 3: Summary of main properties of dairy farms used in the modelling.

	NL_Fa1	NL_Fa2	NL_Fa3	NL_Fa4	NL_Fa5
Number of animals	120 dairy cows 58 young cattle	131 dairy cows 92 young cattle	80 dairy cows 42 young cattle	140 dairy cows 43 young cattle	192 dairy cows 96 young cattle
Area (ha)	43.3	76.3	55.0	63.5	39.5
Crops	Fodder maize, grass	Fodder maize, grass, natural grass	Fodder maize, grass	Fodder maize, grass, natural grass	Grass
Fertilization	Cattle slurry, mineral N	Cattle slurry, mineral N	Cattle slurry, mineral N	Cattle slurry, mineral N	Mineral N

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4. Modelling of Nutri2Cycle solutions

In the Nutri2Cycle project, a selection of the long list (LL) solutions has been analysed by one or more methods, including environmental modelling (Deliverable 3.4; Beyers et al., 2022b; Deliverable 4.2; Rieder et al., 2023), lifecycle assessment (Deliverable 3.4; Beyers et al., 2022b), cost-benefit analysis (Deliverable 3.3; Bamelis et al., 2022) and socio-economic modelling (Deliverable 3.4; Beyers et al., 2022b). For the testing of the MITERRA-Farm model, we selected five solutions. For the selection, we used the following criteria: i) applicability for different farming systems (in this case pig farms, dairy farms and arable farms), ii) for livestock farms, the solutions should preferably have impacts on both the housing/manure storage emissions as well as field emissions, and iii) preferably the solutions should from the priority list (see Deliverable 3.2). This resulted in the following solutions:

- Slurry acidification to reduce NH_3 volatilization from animal husbandry (LL18)
- Adapted stable construction for separated collection of solid manure and urine in pig housing (LL24)
- Small/Farm-scale anaerobic digestion (LL10)
- N fixation with grass-clover (related to LL45)
- Precision farming coping with heterogeneous qualities of organic fertilizers (LL30)

4.1 Solutions for pig farms

4.1.1 Slurry acidification to reduce NH_3 volatilization from animal husbandry

Slurry acidification is a technique using acid to lower slurry pH and reduce NH_3 emissions. Acid may be added to slurry in-house, during storage, or when slurry is field applied. Acidification during storage achieves the best balance between cost and mitigation effect.

Beyers et al. (2022a) performed a life-cycle assessment (LCA) on the environmental impact of slurry acidification during storage in Denmark, the Netherlands, and Spain. They reported a significant reduction in NH_3 and CH_4 emissions, but an increase in N_2O emissions. The effect factors derived from this LCA forms the basis for the modelling of slurry acidification by MITERRA-Farm. Field-level modelling also showed a slight increase in N leaching with acidified slurry compared to unacidified slurry (Beyers et al., 2022a), possibly due a higher N content and less volatilization after field application.

In this study, we modelled slurry acidification during storage. Therefore, there is no change in emissions from housing. An increase or reduction factor was applied to NH_3, CH_4, and N_2O emissions during storage according to Beyers et al. (2022a), and the total N content post-storage was adjusted accordingly. At field application phase, NH_3 volatilization was modelled by ALFAM2, and the effect of acidification was modelled by increasing slurry TAN (as a result of reduced loss during storage) and lowering slurry pH from 7.5 to 5.5. N content remained after volatilization loss was added to soil, and the effect on N runoff and leaching was modelled using standard MITERRA-Farm procedures.

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

Table 4: Modelling approach for slurry acidification during storage in MITERRA-Farm.

Process	Modelling approach
Housing emissions	
NH_3, N_2O, CH_4	Same EFs as unacidified slurry.
Storage emissions	
NH_3	Reduced by 76% compared to unacidified slurry.
N_2O	Increased by 12% compared to unacidified slurry.
CH_4	Reduced by 96% compared to unacidified slurry.
Fertilisation	
Manure replacement	Unacidified slurry is replaced by acidified slurry by the same volume.
Fertiliser composition	Acidified slurry is assumed to have a 3% higher TAN content due to lower NH_3 emissions from storage.
Field application emissions	
NH_3	Reduction modelled by ALFAM2 by changing slurry composition and pH (6.0).
Other gases	Same as unacidified slurry.
Crop production	Acidified slurry does not affect crop yields when applied at the same rate.
Runoff and leaching	N content after emission loss during storage and field application are adjusted. Leaching loss can then be modelled by standard MITERRA-Farm process.

Modelled results on emissions from housing and storage, field application, and soils are shown in Figure 4 to Figure 6. As acidification was performed during storage, it had no effect on emissions during housing. Ammonia emissions during storage were significantly reduced, but the reduction in other gases was minor (Figure 4).

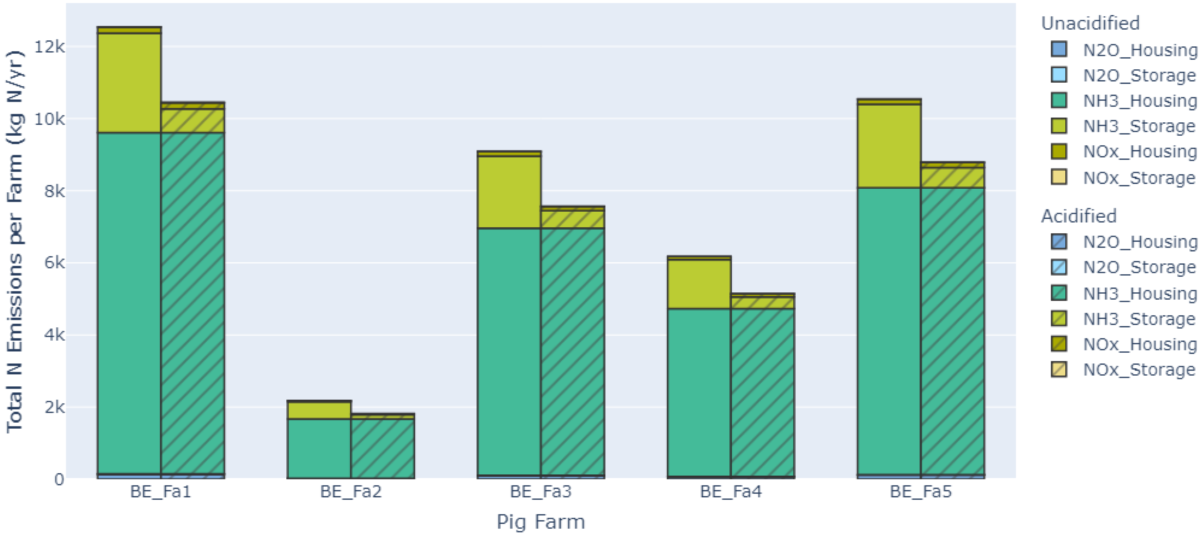

Figure 4: Total N emissions at farm level to the atmosphere from unacidified and acidified pig manure management.

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The acidified slurry also had less NH_3 volatilization following field application of slurry (Figure 5). However, at farm level, the reduction effect depended on the fraction of N from slurry in the total N application, as NH_3 volatilization from mineral fertilizers was not affected by slurry acidification. Due to a higher amount of TAN in acidified slurry, and less NH_3 loss after application, there was a higher N content remaining in the soil after application of acidified slurry as compared to unacidified slurry, and therefore a higher potential for N leaching (Figure 6). This is consistent with field-level modelling by Daisy (Beyers et al., 2022), even though this leaching potential is only marginal.

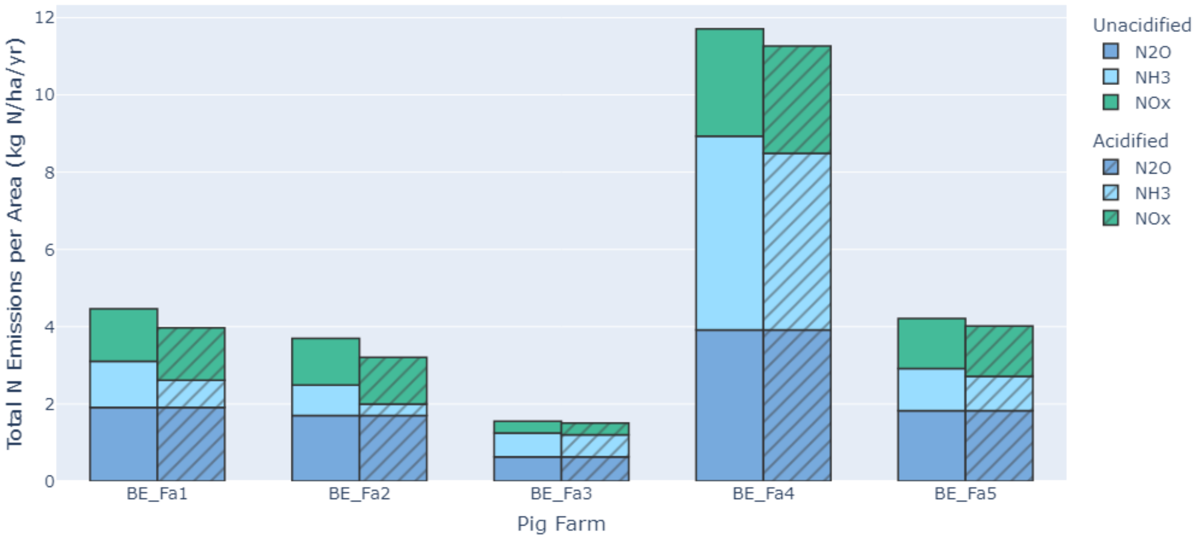

Figure 5: Total N emissions at farm level to the atmosphere from field application of unacidified and acidified pig slurry/manure. Emissions include those from both organic and mineral fertilizers.

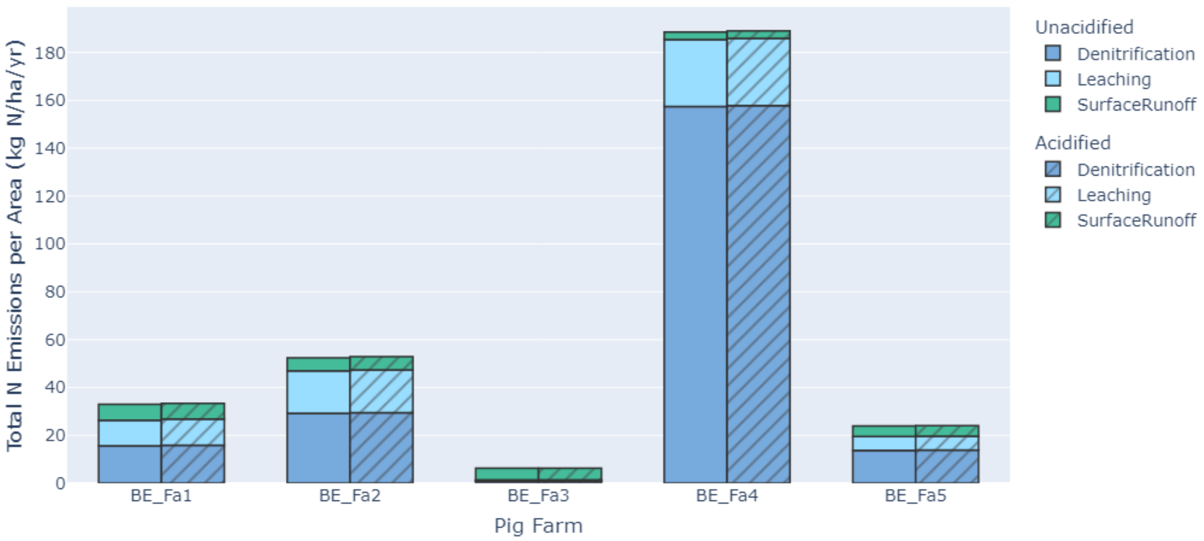

Figure 6: Total N emissions at farm level to surface and groundwater from soil after application of unacidified and acidified pig slurry. Emissions include those from both organic and mineral fertilizers.

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The effect of slurry acidification at farm level modelled by MITERRA-Farm are generally in agreement with other studies. In a review of animal slurry acidification, Acidification of animal slurry has proved to be an efficient solution to minimize NH_3 emissions in-house, during storage, and after soil application, Fangueiro et al. (2015) reported that acidification reduced NH_3 emissions by 50–88% during storage and 40–80% at field application. Our modelled field NH_3 reduction was lower than their reported range in some cases, because farmers usually apply other N fertilisers such as mineral N in addition to acidified slurry. MITERRA-Farm calculates the overall emissions of all fertilisers applied, and therefore can provide a better estimation of farm-level emissions.

4.1.2 Adapted stable construction for separated collection of solid manure and urine in pig housing

Solution LL24 describes the construction of a separation cellar beneath the floor of the stable which enables the separation of urine and solid manure. By reducing the contact of urine to solid manure, the hydrolysis of urea in the urine can be reduced, thereby reducing NH_3 emissions from the stable. The separated urine can be used as a replacement for NK fertilisers, and solid manure may be used as input material to biogas digestors or as an organic fertiliser. The modelling approach is based on the life-cycle assessment on the separation of pig faeces and urine performed by De Vries et al. (2013). The partition fractions for excretion N to urine and faeces are 0.4 and 0.6, respectively. The emission factors for faeces, urine and reference manure during housing, storage, and field application are obtained from De Vries et al. (2013). It is assumed that both urine and faeces are applied to the field with a mixing rate of 0.4 and 0.6, respectively.

Table 5: Modelling approach for separated collection of solid manure and urine in pig housing in MITERRA-Farm.

Process	**Modelling approach**
Housing emissions	
NH_3	Reduced by 52% compared to unseparated manure.
N_2O	Increased by 1489% compared to unseparated manure.
NO_x	Reduced by 98% compared to unseparated manure.
CH_4	Reduced by 93% compared to unseparated manure.
Storage emissions	
NH_3, N_2O, NO_x	Same as unseparated manure.
CH_4	Reduced by 92% for urine. The same as unseparated manure for faeces.
Fertilisation	
Manure replacement	Manure/slurry is replaced by separated urine (40%) and faeces (60%).
Fertiliser composition	TAN content in urine and faeces are 126% and 73% of TAN in unseparated manure, respectively.
Field application emissions	
NH_3	Volatilisation modelled by ALFAM2.
N_2O	Increased by 50% for urine. The same as unseparated manure for faeces.
Crop production	Crop yields not affected.
Runoff and leaching	Modelled by standard MITERRA-Farm process.

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Adapted stable construction to separate urine and faeces reduced CH_4 (results not shown) and NH_3 emissions (Figure 7) during housing. However, there was a side effect of increasing N_2O emissions (Figure 7). The total N emissions to the atmosphere from housing were reduced with adapted stable. From the perspective of lowering N loss, adapted stable effectively preserves more nutrients in the system. However, as N_2O is a potent greenhouse gas, the increased N_2O emissions may offset some of the benefits.

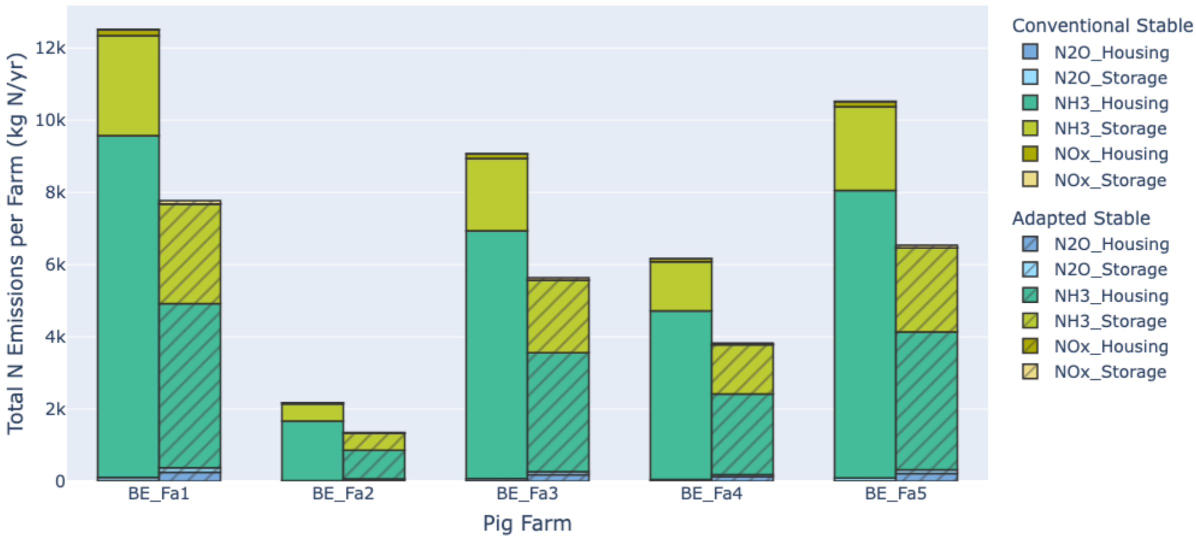

Figure 7: Total N emissions at farm level to the atmosphere from conventional and adapted stables.

The changes in total gaseous N emissions following field application of urine and faeces to replace slurry/manure was minor, with a reduction in NH_3 volatilization but an increase in N_2O and NO_x emissions (Figure 8). There was also a slightly higher potential for denitrification in the soil with urine/faeces application than unseparated slurry, although the difference is only marginal (Figure 9).

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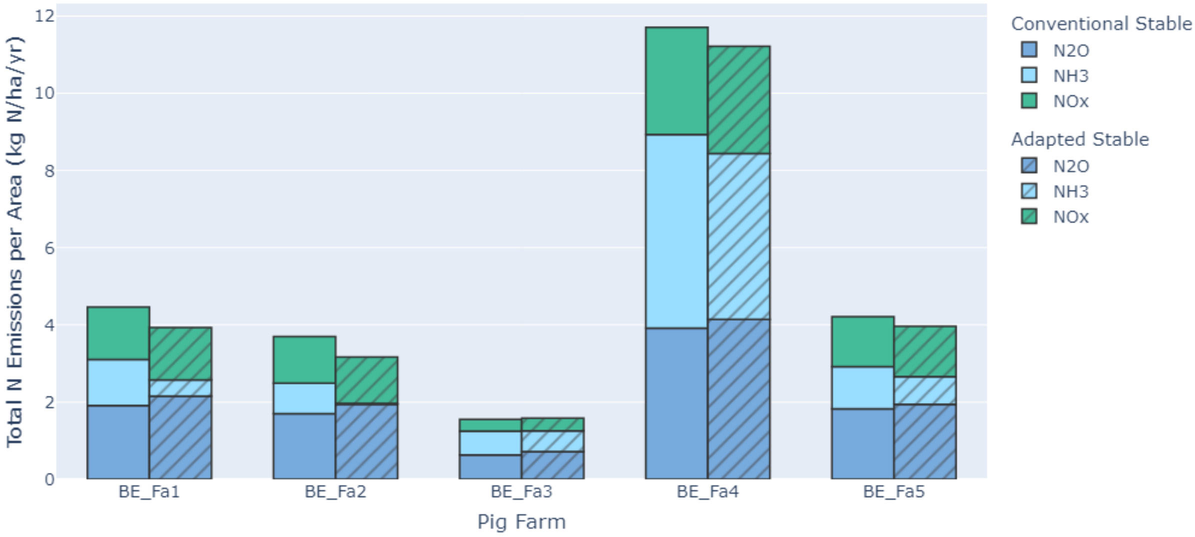

Figure 8: Total N emissions at farm level to the atmosphere from field application of unseparated pig slurry/manure (from conventional stable) and separated pig urine/faeces (from adapted stable).

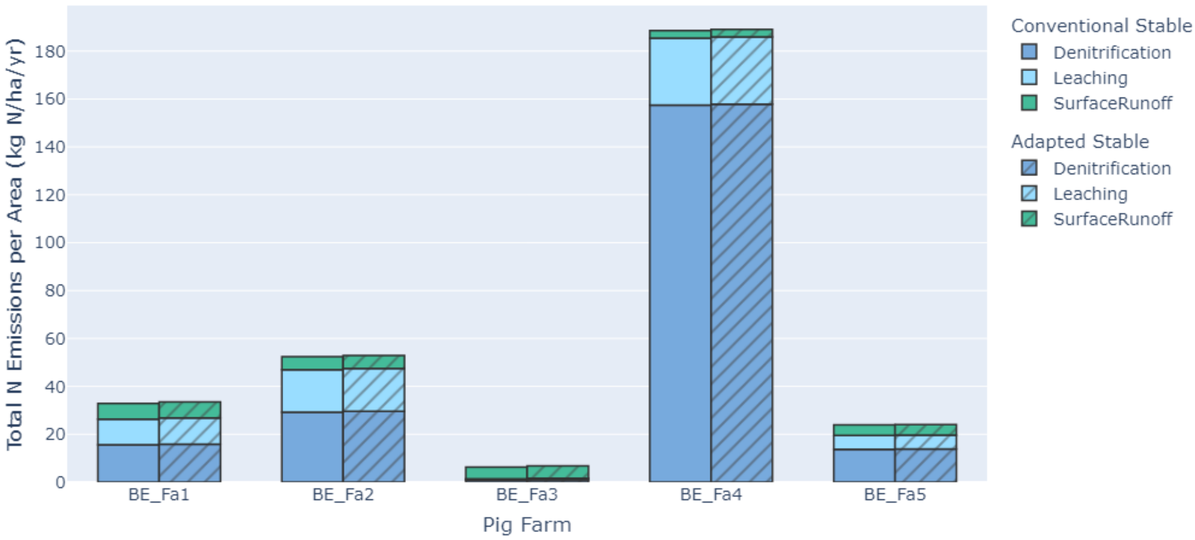

Figure 9: Total N emissions at farm level to surface and groundwater from soil after application of unseparated pig slurry/manure (from conventional stable) and separated pig urine/faeces (from adapted stable).

Several studies have evaluated the effect of source separation of pig urine and faeces (e.g. De Vries et al., 2013; Vu et al., 2016; Dennehy et al., 2017). Separation of pig manure generally decreased NH_3 and CH_4 emissions from housing and storage, but the effect on N_2O emissions differed with an increase of emission from the solid fraction but a decrease from the liquid fraction. Overall, our modelling showed that manure separation decreased atmospheric N emissions from the housing, storage, as well as field application of pig manure, which is consistent with experiment (Vu et al., 2016) and life cycle assessment results (De Vries et al., 2013). It is also reported that the solid fraction has a higher

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methane yield potential than unseparated pig manure (Vu et al., 2016), suggesting that the separation technique may be combined with anaerobic digestion to achieve better economic benefits.

4.2 Solutions for dairy farms

4.2.1 Small/Farm-scale anaerobic digestion

Nutri2Cycle solution LL10 proposed the construction of on-farm anaerobic digesters to process manure and possible crop residues on-site. The inclusion of an anaerobic digestor alters the nutrient flows and emissions in the farm. Firstly, manure is stored for a shorter time waiting for digestion, therefore storage emissions are reduced. Secondly, biogas is produced during anaerobic digestion, which is mostly collected but also with some losses. Thirdly, the end-product of anaerobic digestion, the digestate, has a different composition from manure, and therefore different emissions during storage and field application.

Manure may be added to the digestor shortly after it is collected (fresh manure digestion) or stored for some time before digestion. The storage time can vary depending on the treatment capacity and retention time of the digester. The digestate may also be stored for a period depending on fertiliser spreading regulations. Studies have reported a reduction of 58-78.5% in CH_4 emissions (Amon et al., 2006; Miranda et al., 2015; Vergote et al., 2019). Other gaseous emissions are also affected, with a reduction in NH_3 during storage but an increase in N_2O emissions (Amon et al., 2006).

During anaerobic digestion, most of the biogas CH_4 will be collected and burnt in a boiler for heat and electricity generation. However, a small fraction of CH_4 may be lost via leakage. Vergote et al. (2019) estimate that a total of ca. 4.4% of biogas CH_4 production is lost during anaerobic digestion, however, the variability can be large (Hrad et al., 2022).

The emissions during field application of digestate are also different from untreated manure. Amon et al. (2006) reported that NH_3 emissions of digestate increased by 18% as compared to untreated manure, whereas N_2O emissions decreased by 29%. There was no report on the effect of digestate application on NO_x emissions, but the same reduction factor as that of N_2O was assumed for NO_x, as some NO_x form (nitric oxide, NO) is produced during denitrification, the same process that also produces N_2O.

The solution has been modelled using the approaches and effect factors shown in Table 6.

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Table 6: Modelling approach for on-farm anaerobic digestion in MITERRA-Farm.

Process	Modelling approach
Housing emissions	No change.
Storage emissions	
CH_4	Reduced by 70% compared to no digester.
NH_3	Reduced by 75% compared to no digester.
N_2O	Increased by 41% compared to no digester.
Anaerobic digestion	
Biogas CH_4 production	Methane production fraction is 185 m^3 CH_4 $tonne^{-1}$ volatile solids added. All housing excretion produced is digested (no limitation on treatment capacity).
Biogas CH_4 emissions	4.4% of produced biogas CH_4 is assumed to be lost.
Other gaseous emissions	Not accounted for (assumed to be zero).
Fertilisation	
Manure replacement	Manure is replaced by digestate up to 170 kg N/ha.
Fertiliser composition	Digestate composition defined according to N2C Deliverable 2.6 report.
Field application emissions	
NH_3	Emission modelled by ALFAM2 using digestate composition.
N_2O & NO_x	Reduced by 29% compared to manure.
Crop production	Manure replacement by digestate is assumed to have no effect on crop yields.
Runoff and leaching	Modelled by standard MITERRA-Farm process.

Figure 10 shows the CH_4 production during anaerobic digestion versus CH_4 emissions from housing and manure/digestate storage. Methane emissions from housing, mainly during enteric fermentation, are a major source of CH_4 emissions from dairy farms, which was unaffected by anaerobic digestion. Biogas CH_4 produced from anaerobic digestion was collected for heat and electricity generation, but a small fraction (4.4% in our modelling) may also be lost to the atmosphere. The loss may be further reduced to around 1% with modern digesters. The digestate had lower CH_4 and NH_3 emissions than manure during the storage phase, which further lowered the environmental impact of anaerobic digestion.

The difference in N emissions after field application and from soils between digestate and untreated manure was not so obvious. Although N_2O and NO_x emissions from digestate application were lower, the total emissions of those gases were minor compared to the more abundant emissions of NH_3. The model did not show any difference in surface runoff and leaching in the soil, as is the case with MITERRA-Farm algorithms when the total N input and crop uptake were the same. A process-based model, e.g., Daisy or SWAP/ANIMO, is better suited for modelling dynamic soil processes such as leaching.

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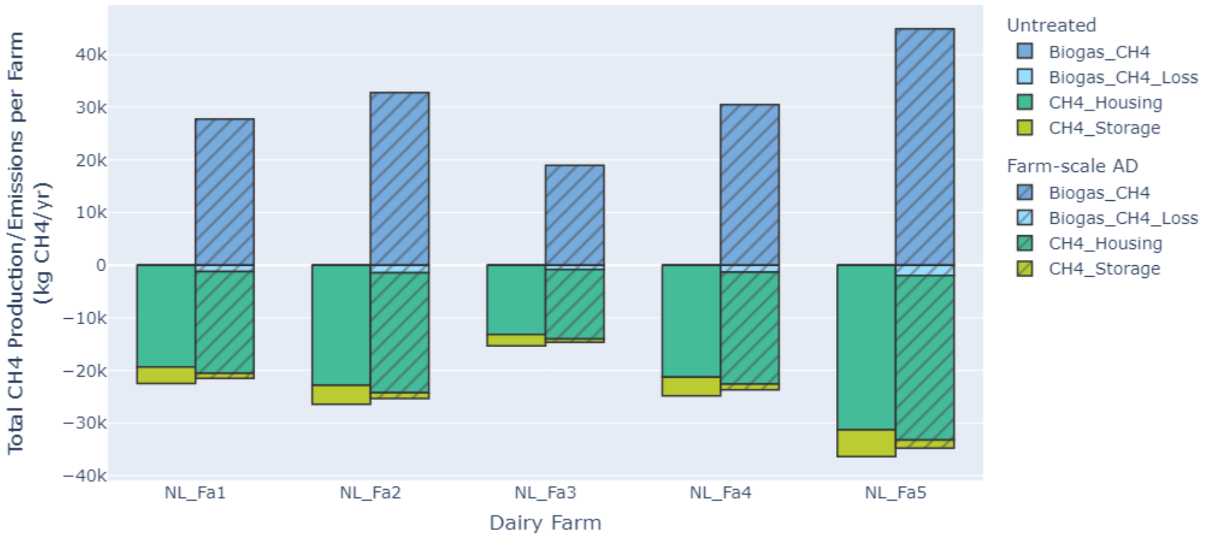

Figure 10: Total biogas CH_4 production at farm level from anaerobic digestion and CH_4 emissions. Negative values indicate emissions. CH_4 emissions from housing are the enteric fermentation related emissions.

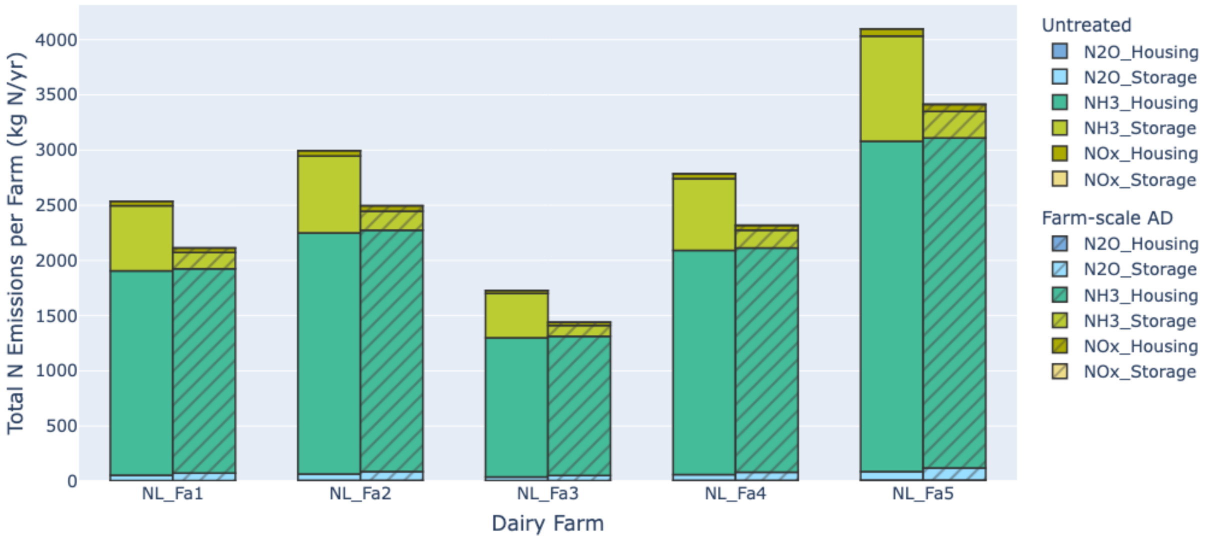

Figure 11: Total N emissions at farm level to the atmosphere from management of untreated manure and manure digested by farm-scale AD.

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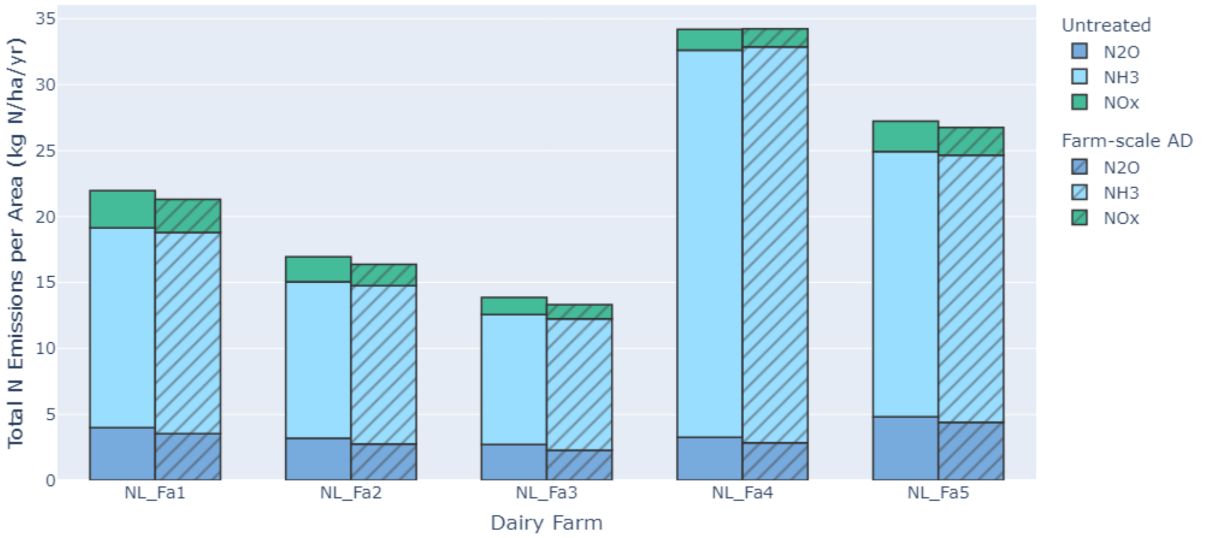

Figure 12: Total N emissions at farm level to the atmosphere from field application of untreated manure and digestate. Emissions include those from both organic and mineral fertilizers.

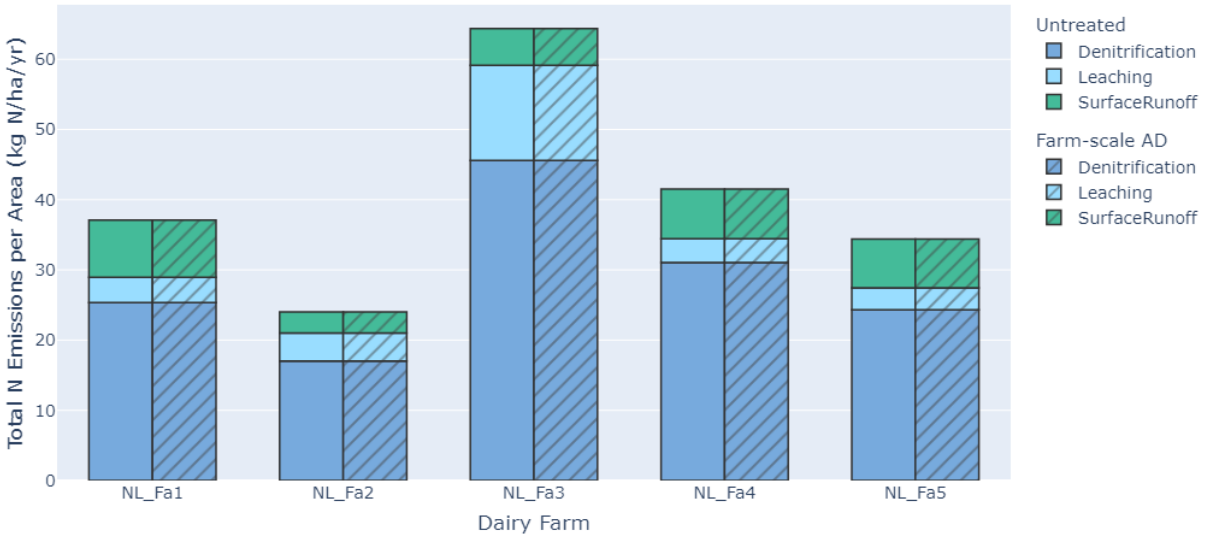

Figure 13: Total N emissions at farm level to surface and groundwater from soil after application of untreated manure and anaerobic digestate. Emissions include those from both organic and mineral fertilizers.

Farm-scale anaerobic digestion is beneficial to produce energy, bio-fertiliser, and mitigate emissions. It showed a clear effect in reducing CH_4 emissions from storage, and N_2O emissions during field application. In a life-cycle assessment of an anaerobic digestor in a commercial dairy farm, Scott and Blanchard (2021) concluded that CH_4 and N_2O emissions were reduced by 18–28% and 47–77%, respectively. They also noted that the airtightness of the system is essential, as 3.6% fugitive CH_4 emissions may be sufficient to offset the overall mitigation advantage of greenhouse gas emissions. In our modelling, a 4.4% biogas CH_4 loss was assumed according to Vergote et al. (2019), and there was still a net reduction of CH_4 emissions, although the difference to the reference scenario is already small. With modern digestors with a lower leakage, the CH_4 emissions can be further reduced.

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4.2.2 N fixation with grass-clover

Nutri2Cycle solution LL45 *INPULSE – Innovating Towards the use of Spanish Legumes in Animal Feed*, proposed mechanisms to promote the cultivation and use of legumes for animal feed in Spain. This solution can be used for closing the nutrient loop in the farm by using locally produced protein as feed and reducing N fertilisation by biological N fixation of legumes.

In the current version of MITERRA-Farm, flows related to animal feed have not been implemented, so it is not possible to model the replacement of imported feed by locally produced feed, but the replacement will not significantly affect the animal and manure related emissions. The introduction of legume crops was simulated for dairy farms, by replacing temporary grass with a grass-clover ley. The field-level model Daisy was used to simulate crop production and biological N fixation by grass-clover leys. The Daisy model was set up based on two baseline scenarios of dairy farms in North-western Europe, ATN-Dairy and CTW-Dairy, as described in Nutri2Cycle report Deliverable 1.5 (Duan et al., 2021). Results from Daisy simulations on crop yields, fixed N, crop residue incorporation, etc., were used as input to MITERRA-Farm.

Table 7: Modelling approach for N fixation with grass-clover in MITERRA-Farm.

Process	**Modelling approach**
Grass-clover yields	Grass-clover yields in NW Europe (10.5 tonne DM/ha/yr) were simulated by field-level Daisy on baseline scenarios ATN-Dairy and CTW-Dairy.
Biological N fixation	Amount of N fixed by clover (on average 165 kg N/ha/yr) was derived from field-level simulation by Daisy. Fixed N is assumed to be taken up by crop immediately, so it is not added to the soil, and there is no loss via atmospheric emissions.
Fertilization	With grass-clover, only cattle slurry was applied. Mineral fertilizer was avoided.
Runoff and leaching	Only N from fertilisation was counted towards leaching. Fixed N was excluded from leaching calculations.

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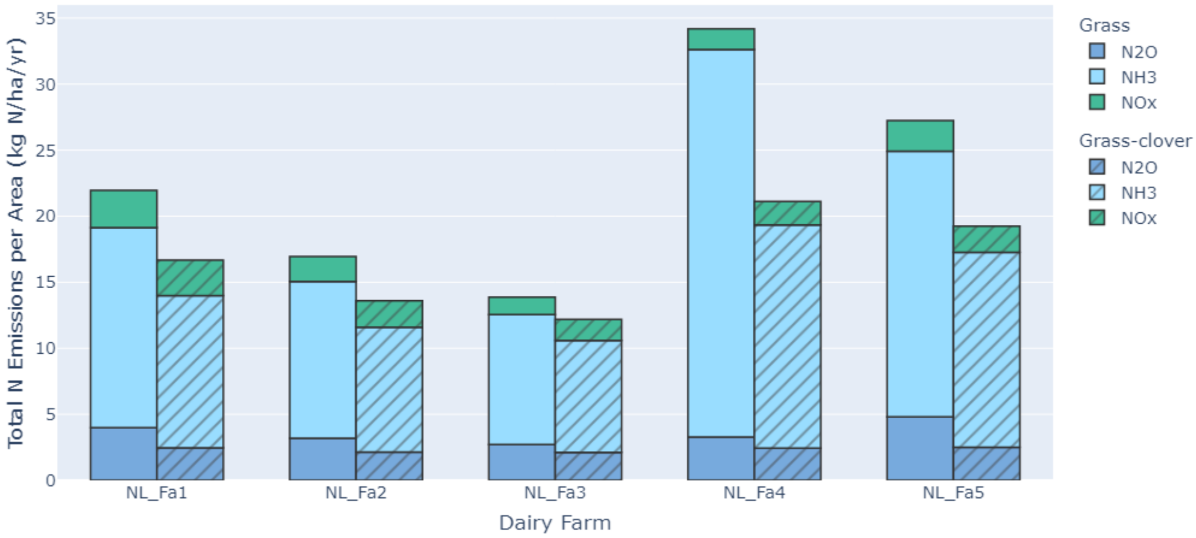

Figure 14: Total N emissions at farm level to the atmosphere from grassland and grass-clover leys. Emissions include those from both organic and mineral fertilizers.

The inclusion of N-fixating crop was modelled by replacing grassland with grass-clover leys. N emissions to the atmosphere were reduced in grass-clover leys because of the avoidance of mineral N fertilizers (Figure 14). However, soil denitrification and leaching increased in grass-clover leys compared to grassland (Figure 15). In several cases, the farmers reported relatively low mineral N fertiliser input, and our assumed N fixation rate was higher than those of the mineral N fertilisers. In the model calculation, this led to an increased soil N surplus and a higher potential for denitrification. N leaching was especially high for farm NL_Fa3, as this was the only farm located on sandy soils, whereas the others are located on clay soils. Probably this farm already had some clover in the baseline, which was not taken into account in the modelling.

The effect of inclusion of N-fixating crops such as clover into the farming system is more complex than what's modelled by MITERRA-Farm. Under excess fertilization, grass would often outgrow clover, thereby reducing bio-fixation of N. As a mostly static model, MITERRA-Farm is not able to capture these dynamic changes in crop growth and production. Nevertheless, the increased N surplus in the results suggest that fertilization may be further lowered by reducing manure application with grass-clover leys to further reduce N losses without compromising yields.

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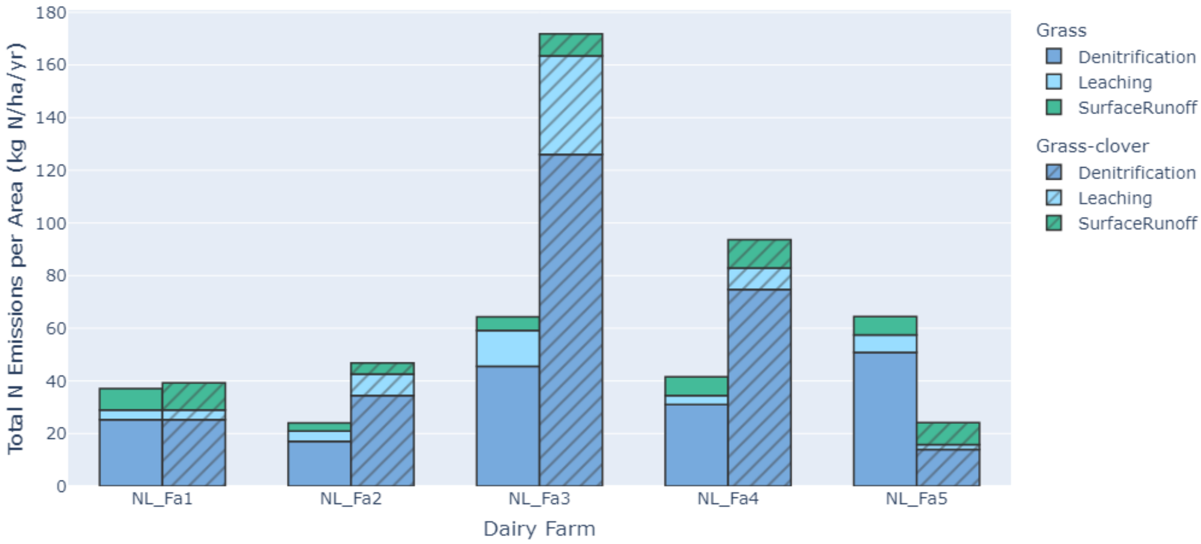

Figure 15: Total N emissions to surface and groundwater from soil with grassland and grass-clover leys.

4.3 Solutions for arable farms

4.3.1 Precision farming coping with heterogeneous qualities of organic fertilizers

In LL30, near infrared sensor (NIRS) is installed on slurry tankers, and it analyses the nutrient content in the slurry on-the-fly during slurry application. Therefore, the volume of the slurry applied can be dynamically adjusted to ensure a precise and uniform application rate throughout the field.

The use of NIRS and dynamic slurry application rates cannot be readily implemented in MITERRA-Farm. We adopted an alternative approach by matching the total N application rate with the crop N demand to model the effect of precision slurry application. This simulates the practice in precision farming when slurry measurement is combined with remote sensing technologies to supply crops precisely what they need.

Crop N demand was calculated by multiplying an efficiency factor and the total N content in crop harvests and residues. For grassland, the efficiency factor is 1; for other crops, the factor is 1.1 for cereal crops or 1.25 for arable crops (Velthof et al., 2009). This accounts for N losses to dead leaves, root exudates, etc. N content in crop harvests was calculated from crop dry matter yields using an N content index, and N in crop residues are estimated using a harvest/residue index. Precision slurry application has no impact on animal housing and manure storage, so all emissions from housing and storage are unchanged.

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Table 8: Modelling approach for precision slurry application in MITERRA-Farm.

Process	**Modelling approach**
Fertilization	Plant available N matches crop N demand. Slurry is applied to a maximum of 170 kg N/ha, and any remaining is supplemented with mineral N. A mineral N equivalence factor is applied to slurry to estimate plant available slurry N.
Field application emissions	Same emission factors as reference used for all gases.
Crop production	Assumed no effect on crop yields.
Runoff and leaching	Modelled by standard MITERRA-Farm process.

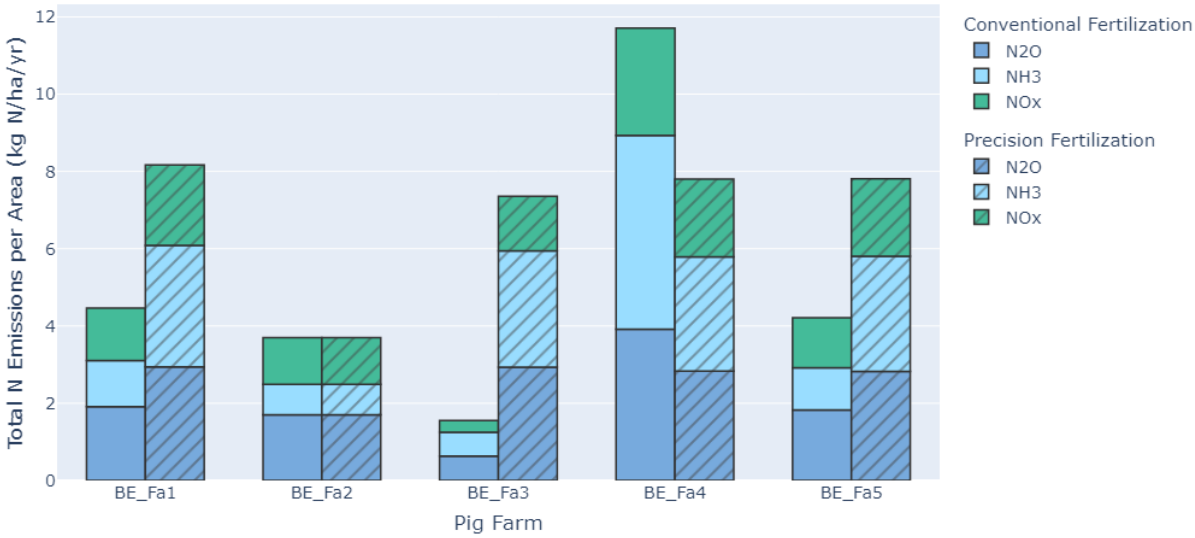

Figure 16: Total N emissions at farm level to the atmosphere from conventional and precision fertilization.

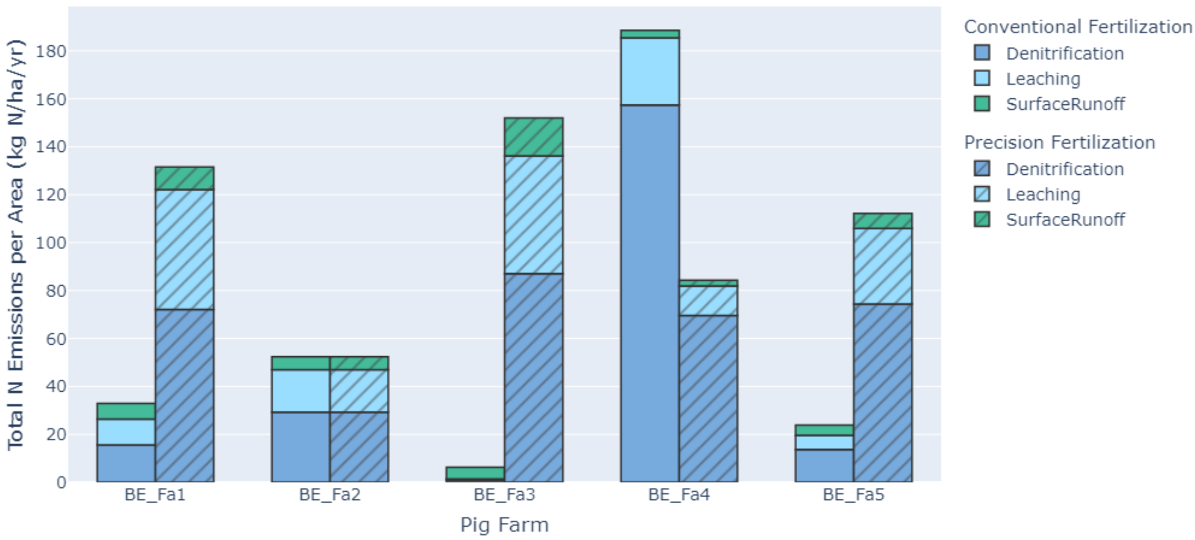

Figure 17: Total N emissions at farm level to surface and groundwater from soil with conventional and precision fertilization.

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Modelling results on the adoption of precision fertilization are inconsistent (Figure 16 and Figure 17). In some farms (e.g., BE_Fa4), there was a clear effect by precision fertilization on reducing N emissions to the atmosphere as well as surface and groundwater. However, in other farms (e.g., BE_Fa1, BE_Fa3 and BE_Fa5), N losses increased significantly with precision fertilization, which defies the concept of precision fertilization.

These inconsistencies might be related to missing or incorrect information on the fertilization and crop yields data in model input. The farm dataset to feed to model did not have information on the nutrient content of the slurries applied. Therefore, the model applied default values based on average slurry content reported by various studies. However, this may lead to a discrepancy in slurry N applied in the actual farm, and the application rate assumed by the model. For example, in farm BE_Fa3, the volume of slurry applied was lower than in the other farms, which resulted in a much lower N emission than other farms. It is possible that BE_Fa3 used a thicker slurry with richer nutrient content to meet the crop requirement, and the emissions in reality would be much higher. However, the model assumed a lower nutrient content in the slurry, resulting in low slurry N application rates as well as low emissions. It might also be that the crop N uptake is overestimated for that farm, as the current N surplus is unrealistically low.

Moreover, the static and empirical nature of the MITERRA-Farm model will also not capture all effects related to the precision farming technology. The MITERRA-Farm model does not simulate crop growth and yield response to N fertilization. Crop uptake of N was derived from crop yields, which was reported by farmers. Therefore, if the model assumed an incorrect slurry N application rate, it would not be reflected in crop yields and crop N uptake. This would lead to a second discrepancy in harvested N content and fertilizer N supply.

The use of a process-based model to simulate part of the farm nutrient flows, e.g., crop N uptake, may be helpful to avoid those discrepancies. However, process-based models often have even more detailed data requirements than MITERRA-Farm. Indeed, a farm dataset with more complete entries is necessary to close the gaps between reality and model simulation. However, such a dataset is difficult to obtain from existing databases in this study, either due to privacy reasons or lack of direct contact with farms.

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5. Modelling of soil organic carbon practices

5.1 Introduction

In 2018, the Dutch government started a programme to stimulate the uptake of practices that increase the soil organic matter content in soils. This programme, so called Smart Land Use Programme, is contributing to both the sustainable soil strategy and the climate targets of the national climate agreement. In this agreement, a specific target for soil carbon sequestration has been included of 0.5 Mton CO_2 additional sequestration per year by 2030[3]. As part of the programme pilots started for the arable sector with practices on improved crop rotation, minimum tillage, and additional application of organic matter (e.g., compost). For the dairy sector the practices maintaining permanent grassland, maize sown in strips in grassland and species rich grassland are being tested. Eight different networks of farmers (four arable and four dairy), each including about 15 farms, are testing the different soil carbon practices.

The underlying principle is that increasing soil organic matter can both contribute to climate change mitigation and improve soil quality and the recycling of nutrients. Many of the practices are not new, but so far not widely applied by farmers, partly due to lack of knowledge, and partly because of barriers, e.g., investment costs or legislation. Within the Smart land use programme all practices and pilots have been assessed following the same measurement protocol and soil samples have been analysed in the same laboratory, to ensure good comparability and replicability.

As changes in soil carbon stocks are taking a long time and only constitute a small contribution of additional carbon compared to the large existing stock, it is difficult to measure the changes in a short period of a few years. Therefore, modelling in combination with long-term experiments is required to assess the effects. The focus was on arable farms as these often have more possibilities for adapting their management to increase soil carbon stocks. In this Chapter, the results of the modelling of soil carbon practices on Dutch arable farms are presented, which are the results for Nutri2Cycle solution LL71.

5.2 Approach

We focussed on arable farms from the Smart Land Use programme for which we simulated the effects of selected practices on the soil carbon stock using the RothC model as included in MITERRA-Farm. Data from farms of four arable networks were collected within each network of 10-15 farms. The following farm data were collected for the soil carbon modelling:

- Crop areas and crop rotation
- Crop yields and crop residue management
- Application of organic fertilizers (manure, compost)
- Soil management (type of tillage, use of cover crops)
- Soil data (soil organic matter content, clay content)

[3] https://www.klimaatakkoord.nl/

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Climate data (monthly temperature, precipitation, and evapotranspiration) were already available for the different regions in the Netherlands. The data were structured in such a way that for each farm, two simulations were made. One simulation with the current crop and soil management (baseline), and one simulation with the selected SOC practices. For each scenario, a simulation was made for the period 2020-2050, taking the crop rotations into account. The main output of the modelling is the predicted changes in soil organic carbon stocks for each of the measures, or a combination of multiple measures will be used on the farm. As multiple farms are included, this provides insights into the variability of the potential C sequestration.

We selected the following soil carbon practices:

- Use of cover crops
- Increase share of cereals in crop rotation
- Substitution of slurry by solid manure
- Incorporation of straw

For the current use of cover crops, we had information about which cover crop was used after which main crop. For the simulation of the additional potential, we allocated a cover crop after a main crop that was not followed yet by a cover crop and if that would still fit within the crop rotation. After some crops that are harvested late, e.g., sugar beet, it is often not possible to grow a cover crop anymore. For the additional cover crops we used an average carbon input based on the cover crops that were applied in the baseline. The C input of the cover crops was partitioned over the RPM and DPM pools according to the default allocation of crop residues in RothC (Lesschen et al., 2021).

As most arable farms in the Netherlands have a very intensive crop rotation with a high share of root crops (e.g., potato and sugar beet), the increase of crops with less soil disturbance and higher carbon inputs, such as cereals, will improve the soil carbon balance. For simulating this practice, we increased the share of cereals, or other crops with high carbon input, to 50% of the crop rotation following Lesschen et al. (2021). This was done by replacing for one or more years an intensive crop (potato, sugar beet or onions) by a cereal crop. The choice for the cereal crop is based on the existing crop rotation, and if no cereals are grown, we assumed it was winter wheat.

Most of the manure in the Netherlands is slurry and for arable farms this is mostly pig slurry. The organic matter content of pig slurry is relatively low compared to the nitrogen and phosphorus content. A shift from slurry to solid manure would increase the carbon input to the soil and therefore, contribute to an improved SOC balance. For the simulations the amount of slurry was substituted by solid manure while keeping the total N input at the same level.

The last practice is the incorporation of straw in the soil. For some farmers, this is a standard practice to maintain the soil quality, whereas other farmers harvest most of the straw to sell it. For the simulations, we assumed that all farmers would not harvest the straw and incorporate it in the soil, therefore increasing the carbon input.

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5.3 Results

The results of the simulations are presented for all individual farms (Figures 18-21) as well as in boxplot graphs for the four regions (Figure 22). The results represent the average SOC balance over the entire crop area of the farm. In the current situation (baseline), most arable farms have a negative SOC balance; only the farms in the Zeeland region (*aze*) have a positive balance. The main reason is the relatively low organic carbon content of the soil. In contrast, in the Veenkoloniën (*ave*), the SOC content is high due to the remaining parts of former peat layers, which results in strong negative SOC balances.

For cover crops, all farms show an improvement in the SOC balance (Figure 18), ranging from less than 0.05 ton C/ha/year up to 0.25 ton C/ha/year over the entire rotation. This variation is mainly due to the composition of the crop rotation and whether a cover crop can be sown early or not. The variation amongst regions seems higher than the variation amongst farms within the same region. For example, in the Veenkoloniën (*ave*), all farms have a similar increase in SOC balance as the crop rotation in this region is rather similar with a high share of starch potatoes.

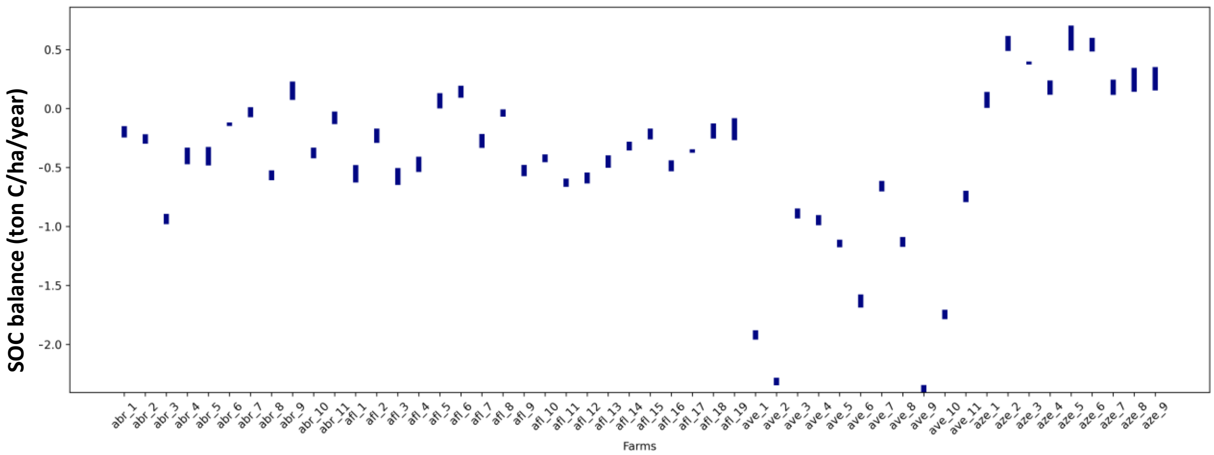

Figure 18: Change in soil carbon balance for arable farms for the practice cover crops. Each bar indicates the change in SOC balance from the baseline (lower value of the bar) to the situation with the practice (upper value of the bar). The first three letters of the farms on the x-axis indicate the respective region.

For the practice of increasing the share of cereals, the results show a very variable pattern (Figure 19). For some farms, the SOC balance is strongly improved (up to 0.37 ton C/ha/year), whereas on other farms even a slightly negative SOC balance is simulated. In those cases, it might be that the cereal crop replaces another crop that had a relatively high carbon input, which might also be linked to the manure application. On average, the improvement of the SOC balance is highest for the Flevoland region (*afl*).

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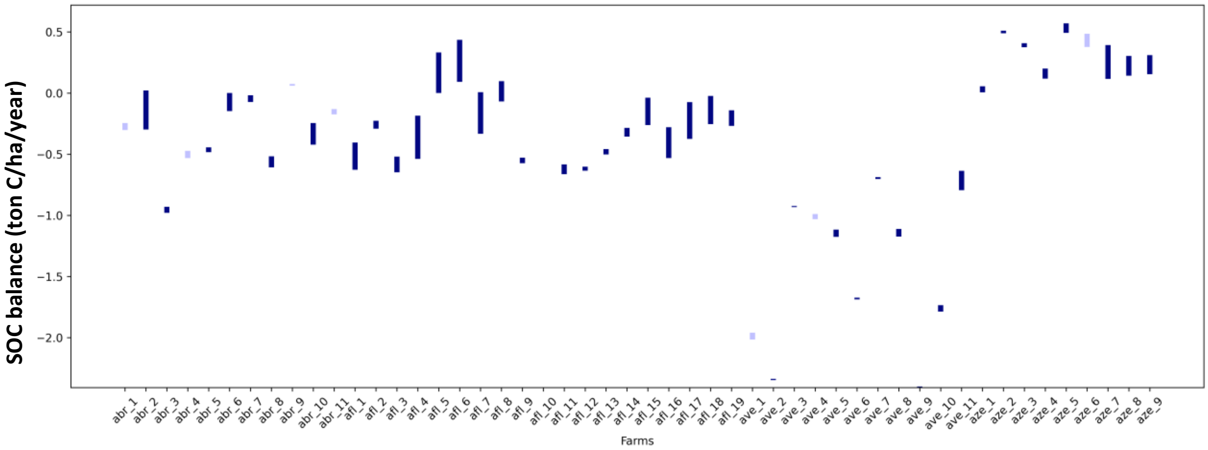

Figure 19: Change in soil carbon balance for arable farms for the practice increasing share of cereals. Each bar indicates the change in SOC balance from the baseline (lower value of the bar) to the situation with the practice (upper value of the bar), for the light blue bars the change is negative. The first three letters of the farms on the x-axis indicate the respective region.

For the substitution of slurry by solid manure, an improvement of the SOC balance is observed for all farms, but the results differ among farms depending on the current type of manure that is applied (Figure 20). Farmers that currently apply pig slurry will see more improvement compared to farmers that apply cattle slurry, as the carbon content of cattle slurry is higher. However, the availability of solid manure is limited, which means that this is not a practice that can be applied at large scale and it will have a trade-off with the practice of incorporating straw, as straw is used in most solid manure management systems. Finally, for the SOC practice incorporation of straw, we see large regional differences (Figure 21) with small effects in Noord-Brabant (*abr*) and Flevoland (*afl*), whereas in the Veenkoloniën (*ave*) and Zeeland (*aze*) there is almost no effect. In these last two regions, the farmers are often incorporating the straw, as in these regions most farms are on clay soil, for which incorporating straw is a good practice for to improve soil structure.

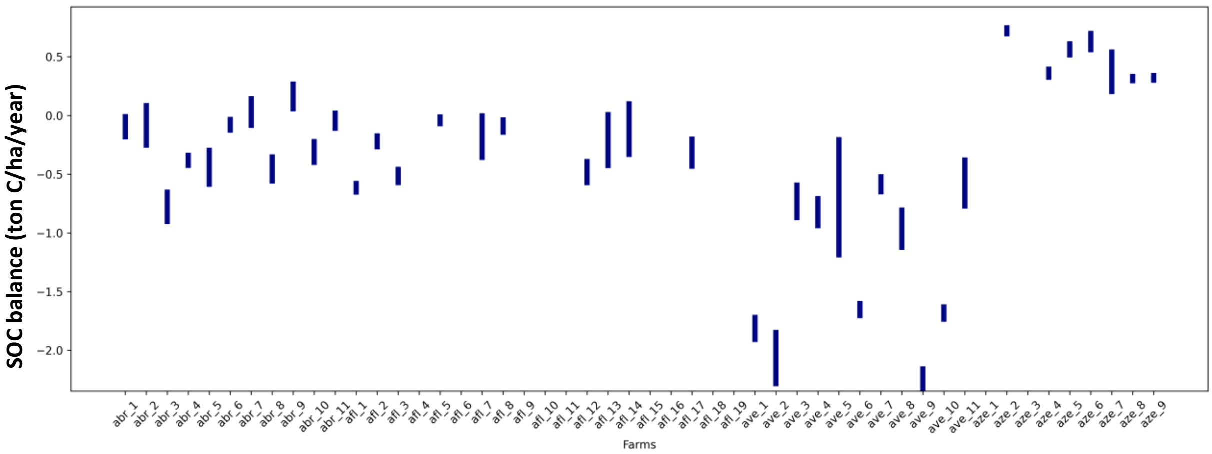

Figure 20: Change in soil carbon balance for arable farms for the practice substitution of slurry to solid manure. Each bar indicates the change in SOC balance from the baseline (lower value of the bar) to the situation with the practice (upper value of the bar). The first three letters of the farms on the x-axis indicate the respective region.

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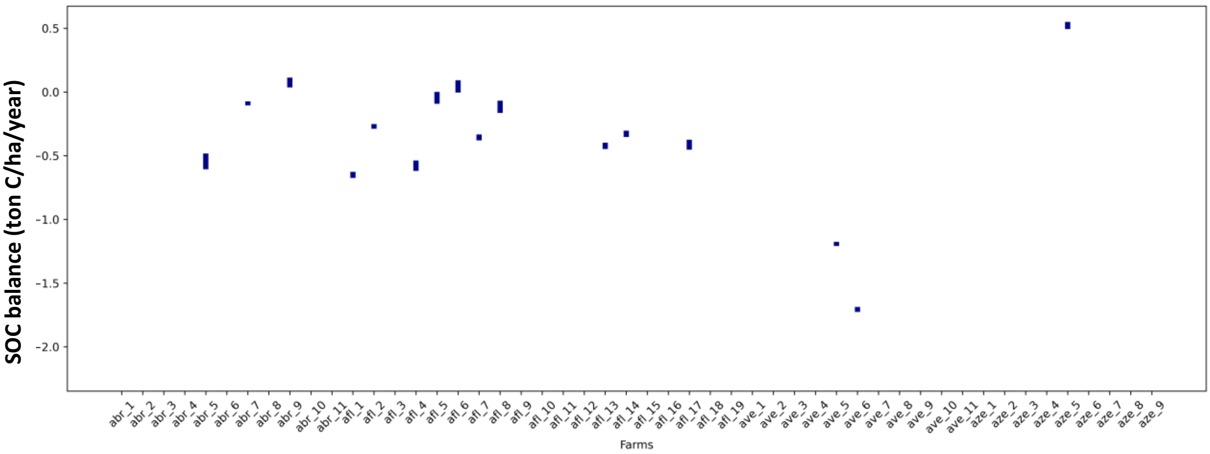

Figure 21: Change in soil carbon balance for arable farms for the practice incorporation of straw. Each bar indicates the change in SOC balance from the baseline (lower value of the bar) to the situation with the practice (upper value of the bar). The first three letters of the farms on the x-axis indicate the respective region.

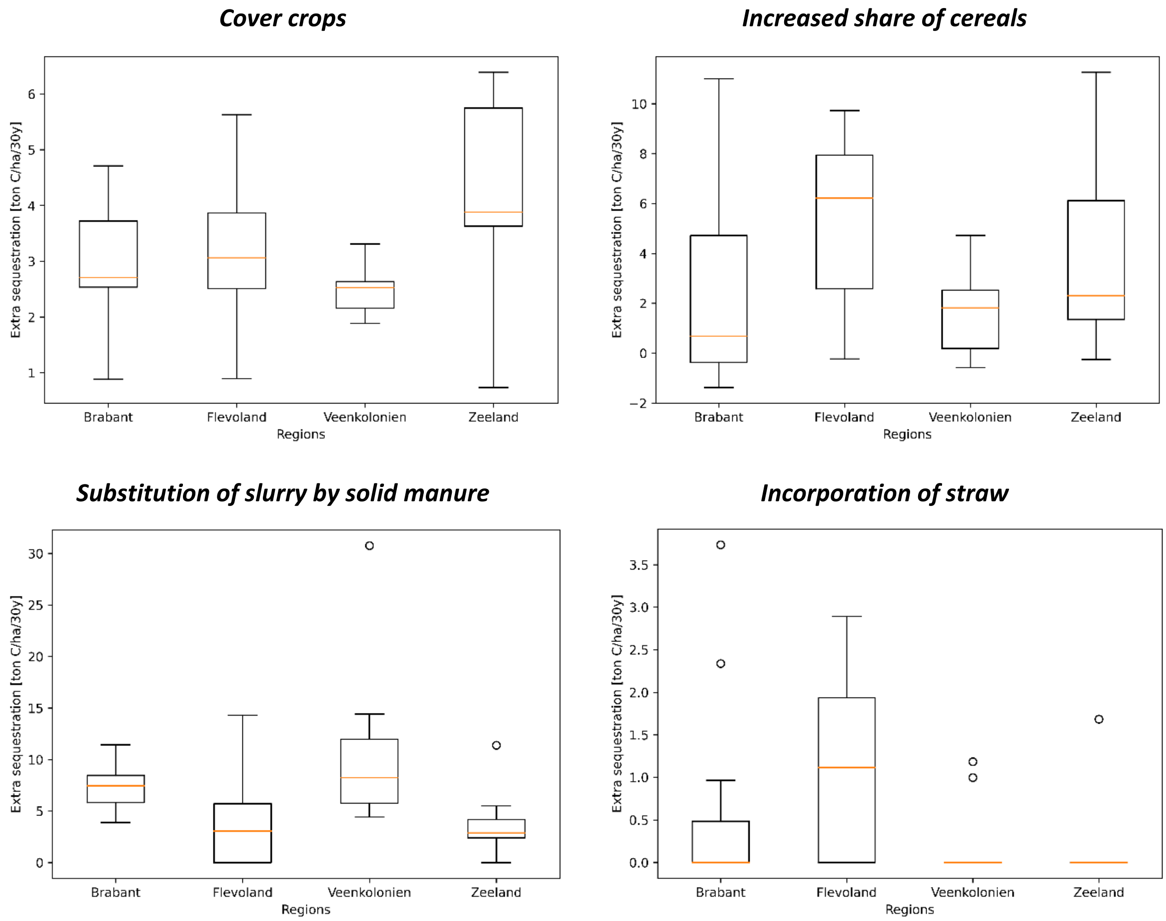

Figure 22: Change in soil carbon balance for the four SOC practices. In the boxplot graph each bar indicates the change in SOC balance from the baseline to the situation with the practice, the orange line is the median value of all simulated farms within the region.

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5.4 Discussion

The results show that the potential of the SOC practices is quite variable amongst regions and amongst farms. For cover crops and substitution of slurry by solid manure the variation amongst regions is limited, but for increasing the share of cereals in the rotation and incorporation of straw large regional differences are observed, as cropping patterns are often also regionally determined. The results show that advising single measures is not very effective and farm-specific advice regarding SOC practices is needed.

In a meta-analysis by Poeplau and Don (2015) an annual increase in soil carbon of 0.32±0.08 ton C ha^{-1} yr^{-1} was found in a mean soil depth of 22cm and during the observed period of up to 54 years. This is slightly higher than the average sequestration rate as modelled by RothC (about 0.1 ton C ha^{-1} yr^{-1}), however, this value is the average over the entire farm area, whereas the value from Poeplau and Don (2015) is based on the area where the cover crops are applied.

In Europe there are few studies published that test the effect of including more cereals in the crop rotation. Götze et al. (2016) compared a monoculture of sugar beets to a rotation with sugar beets and winter wheat in central-east Germany. Crop residues of the wheat crop were removed each year. After forty years SOC stock in the sugar beet – winter wheat rotation was higher than under the monoculture of sugar beets with an average sequestration rate of 0.06 t C ha^{-1} yr^{-1}. Another field experiment of 13 years in central Germany with sugar beet rotations found that the carbon sequestration rate was 0.31 t C ha^{-1} yr^{-1} higher in the top 0-20 centimetres of the soil with a rotation of sugar beet – winter wheat – winter wheat compared to a rotation of sugar beet – winter wheat – silage maize (Grunwald et al., 2021). In this experiment the crop residues of winter wheat were left on the field which could explain the difference in carbon sequestration rates. The RothC simulations showed an average sequestration rate of 0.11 ton C ha^{-1} yr^{-1} for the entire farm area, which seems in line with these two studies.

Substitution of slurry by solid manure had the highest sequestration potential, of an average about 0.17 ton C ha^{-1} yr^{-1} over the entire farm area. Maillard & Angers (2014) evaluated 130 observations from 49 sites across the world and found a linear relationship between the manure input and the change in SOC with an average sequestration rate of 0.42 t C ha^{-1} $year^{-1}$. However, this is based on a a comparison only mineral fertilizer. They found that for every ton manure-C ha^{-1} that was brought into the field, 0.12 t C ha^{-1} was sequestered. A similar result was found in the meta-analysis by Han et al. (2016). This is more or less in line with the RothC simulations, as the average increase in manure C was about 1.9 ton C ha^{-1}.

For incorporation of straw the potential was limited, as many farmers already applied this practice, and the share of cereals is relatively low in the crop rotation of many farmers. A literature review by Lehtinen et al. (2014) reports an average increase in SOC content of 7% ± 1.39% comparing 84 study cases in Europe. For some individual farms this increase is observed, but most Dutch arable farms have lower sequestration rates for this practice.

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A frequently mentioned practice to increase soil carbon stocks is the use of reduced tillage. However, in the experiments on the Dutch Smart Land Use programme no significant effects of reduced tillage on soil carbon stocks were found. Also, in literature contradictory conclusions are found, with more recent publications (e.g., Haddaway et al., 2017) showing that often an increase in soil carbon is found in the topsoil, but a decrease in the subsoil, which would mean that reduced tillage results rather in a redistribution of soil carbon instead of a net sequestration. Therefore, we decided not to include this practice in the modelling of this exercise.

Although the measures are focused on increasing soil carbon, there can be indirect effects on N and P flows. Especially for nitrogen, some effects can be expected, including reduced risk of N leaching and runoff, potential trade-off with additional N_2O emissions. The RothCN module could model the effects on changes in nitrogen mineralisation and accumulation, which will determine the N surplus that can potentially leach. However, the inclusion of nitrogen dynamics into RothC was only done during the last months of the project and we could not use this for this exercise. For future studies the CN interactions will be taken into account.

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6. Limitations and perspectives of MITERRA-Farm

Currently the availability of farm-specific datasets is a major limiting factor to the use of the model for research purposes. In the Nutri2Cycle project, the project partners had difficulties in sharing data sets, because of privacy and data sharing reasons. Therefore, we lacked a good characterisation of farms in the Mediterranean, Nordic and Eastern European regions, as we did not have farm data from these regions in our dataset.

This first version of the MITERRA-Farm model aimed to model all on-farm C, N and P flows and related nitrogen and GHG emissions. The current version of the model starts with the animal excretion and does not account for the feed use. This might be added in the next version, as those types of calculations have been used by the team before in the Cool Farm Tool. That would also allow farmers to simulate changes in the N and P excretion by changing the feed types. MITERRA-Farm models the nutrient flows within a single farm. Therefore, inter-farm exchanges, such as the transfer of surplus manure from one farm to another are not included. Considering the requirement to model the selected and prioritised Nutri2Cycle solutions, some nutrient flows are not implemented in the present version of MITERRA-Farm.

The basic principles and processes of MITERRA-Farm have now been established. However, the development of the model is not concluded and is still actively progressing. This will be continued within the Horizon Europe NutriBudget project, which can be considered as a follow-up of the Nutri2Cycle project. For that project, the model will be extended to simulate all major nutrients and heavy metals and P leaching will be included. The MITERRA-Farm model will be used in that project as the calculation model in the so-called NutriPlatform, which will be a decision-support tool to determine which agronomic or policy measures to implement to reach the desired state regarding agronomic and environmental targets.

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7. Conclusions

This report presented the first version of the MITERRA-Farm model. Although the model is not yet fully developed, it is useful for modelling C, N, and P flows at the farm level and simulating the effect of new technologies. This has been demonstrated by modelling the effect on emissions for five of the Nutri2Cycle solutions. These modelled effects are summarised in Table 9 as the averages of the five farms. For methane the variation of the modelled effects is low, while for N_2O and NO_3 there is a large variation among farms. The five solutions all contribute to closing nutrient loops, but their effects on emissions is not always positive. This highlights the importance of a full farm integrated modelling approach.

Table 9. Modelled effect of Nutri2Cycle solutions relative to the baseline (without application of the solution). Values > 1.0 indicate an increased emission, whereas values < 1.0 indicate a reduction in emission. Values in parentheses are standard deviations.

Solution	CH_4	NH_3	N_2O	NO_3
LL18 Slurry acidification	0.282 (< 0.001)	0.828 (0.002)	1.005 (0.001)	1.005 (0.004)
LL24 Adapted stable	0.491 (< 0.001)	0.597 (0.004)	2.230 (0.343)	1.018 (0.025)
LL10 Anaerobic digestion*	0.956 (0.002)	0.869 (0.023)	0.989 (0.042)	1.003 (0.004)
Grass-clover	1.0 (0.0)	0.928 (0.063)	0.749 (0.065)	1.589 (0.643)
LL30 Precision Farming**	1.0 (0.0)	1.014 (0.028)	*1.529* (1.043)	*4.129* (4.647)

* CH_4 emissions at farm-level for anaerobic digestion also include CH_4 loss from biogas production, but exclude the biogas itself

** For two farms the emissions increased significantly, which is probably due to missing or incorrect information on the fertilization data in the input data, therefore these negative effects are probably not realistic.

The benefit of the MITERRA-Farm model is that it uses actual farm data that would normally be available by farmers, which allows potentially widespread use. Another advantage is that the model is based on the MITERRA-Europe model and makes use of the same kind of input data. In case certain input data would not be available for a farm, it could be replaced by regional data from the MITERRA-Europe database, e.g., crop yields or soil properties. In contrast to the detailed field-scale models, the variability amongst farms is covered much better, as was demonstrated with the modelling of the soil organic carbon practices. The development of the model is not concluded and is actively progressing within the Horizon Europe NutriBudget project.

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Appendix

Modelled N and CH_4 emissions from pig farms

Emissions from livestock housing & manure management (kg N/C yr^{-1})

Treatment	Farm	NH3 Housing	N2O Housing	NOx Housing	NH3 Storage	N2O Storage	NOx Storage	CH4 Enteric	CH4 Storage
Reference	**BE_Fa1**	9464	12.94	167.03	2761	96.31	10.84	8943	26582
	BE_Fa2	1646	2.25	29.05	480	16.75	1.89	1555	4623
	BE_Fa3	6858	9.38	121.03	2001	69.79	7.86	6480	19261
	BE_Fa4	4660	6.37	82.25	1360	47.43	5.34	4404	13090
	BE_Fa5	7959	10.89	140.47	2322	81.00	9.12	7521	22355
Acidification	**BE_Fa1**	9464	12.94	167.03	662	107.87	10.84	8943	1063
	BE_Fa2	1646	2.25	29.05	115	18.76	1.89	1555	184
	BE_Fa3	6858	9.38	121.03	480	78.16	7.86	6480	770
	BE_Fa4	4660	6.37	82.25	326	53.12	5.34	4404	523
	BE_Fa5	7959	10.89	140.47	557	90.72	9.12	7521	894
Adapted Stable	**BE_Fa1**	4543	254.26	3.34	2761	119.05	90.34	626	16799
	BE_Fa2	790	44.22	0.58	480	20.71	15.71	108	2922
	BE_Fa3	3291	184.23	2.42	2001	86.27	65.46	453	12172
	BE_Fa4	2237	125.21	1.65	1359	58.63	44.49	308	8273
	BE_Fa5	3820	213.83	2.81	2322	100.12	75.98	526	14128
Precision Fertilization	**BE_Fa1**	9464	12.94	167.03	2761	96.31	10.84	8943	26582
	BE_Fa2	1646	2.25	29.05	480	16.75	1.89	1555	4623
	BE_Fa3	6858	9.38	121.03	2001	69.79	7.86	6480	19261
	BE_Fa4	4660	6.37	82.25	1360	47.43	5.34	4404	13090
	BE_Fa5	7959	10.89	140.47	2322	81.00	9.12	7521	22355

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Emissions from field following manure application (kg N ha^{-1} yr^{-1})

Treatment	Farm	NH3	N2O	NOx	Surface Runoff	Leaching	Denitrification
Reference	BE_Fa1	1.20	1.91	1.36	6.56	10.79	15.58
	BE_Fa2	0.79	1.70	1.21	5.44	17.73	29.27
	BE_Fa3	0.62	0.63	0.30	4.88	0.51	0.91
	BE_Fa4	5.01	3.92	2.78	3.15	27.95	157.48
	BE_Fa5	1.09	1.83	1.30	4.35	5.83	13.74
Acidification	BE_Fa1	0.70	1.91	1.36	6.58	10.98	15.85
	BE_Fa2	0.30	1.70	1.21	5.45	17.89	29.52
	BE_Fa3	0.57	0.63	0.30	4.88	0.52	0.91
	BE_Fa4	4.57	3.92	2.78	3.16	28.02	157.85
	BE_Fa5	0.89	1.83	1.30	4.35	5.86	13.82
Adapted Stable	BE_Fa1	0.42	2.16	1.36	6.58	10.99	15.87
	BE_Fa2	0.02	1.94	1.21	5.45	17.90	29.54
	BE_Fa3	0.54	0.71	0.33	5.13	0.60	1.06
	BE_Fa4	4.30	4.14	2.78	3.16	28.02	157.89
	BE_Fa5	0.72	1.94	1.30	4.35	5.87	13.84
Precision Fertilization	BE_Fa1	3.15	2.94	2.09	9.43	49.96	72.12
	BE_Fa2	0.79	1.70	1.21	5.44	17.73	29.27
	BE_Fa3	3.02	2.93	1.41	15.82	49.12	87.12
	BE_Fa4	2.96	2.84	2.01	2.38	12.35	69.58
	BE_Fa5	2.98	2.82	2.00	6.23	31.57	74.39

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Modelled N and CH_4 emissions from dairy farms

Emissions from livestock housing & manure management (kg N/C yr^{-1})

Treatment	Farm	NH3 Housing	N2O Housing	NOx Housing	NH3 Storage	N2O Storage	NOx Storage	NH3 Grazing	N2O Grazing	NOx Grazing	CH4 Enteric	CH4 Storage	CH4 Grazing	Biogas CH4	Biogas CH4 Loss
Reference	NL_Fa1	1849	6.29	39.76	589.32	48.54	0.54	535.26	149.86	22.48	19350	3167	152		
	NL_Fa2	2184	7.43	46.96	696.03	57.33	0.65	635.84	176.88	26.53	22832	3611	178		
	NL_Fa3	1261	4.29	27.11	401.77	33.09	0.37	365.33	102.15	15.32	13189	2144	103		
	NL_Fa4	2031	6.91	43.66	647.12	53.31	0.60	584.60	164.67	24.70	21267	3589	168		
	NL_Fa5	2989	10.17	64.26	952.48	78.46	0.88	865.39	242.20	36.33	31273	5108	246		
Farm-scale AD	NL_Fa1	1849	6.29	39.76	147.33	68.45	0.54	535.26	149.86	22.48	19350	950	152	27782	1222
	NL_Fa2	2184	7.43	46.96	174.01	80.84	0.65	635.84	176.88	26.53	22832	1083	178	32810	1443
	NL_Fa3	1261	4.29	27.11	100.44	46.66	0.37	365.33	102.15	15.32	13189	643	103	18940	833
	NL_Fa4	2031	6.91	43.66	161.78	75.16	0.60	584.60	164.67	24.70	21267	1076	168	30508	1342
	NL_Fa5	2989	10.17	64.26	238.12	110.63	0.88	865.39	242.20	36.33	31273	1532	246	44902	1975
Grass-clover	NL_Fa1	1849	6.29	39.76	589.32	48.54	0.54	535.26	149.86	22.48	19350	3167	152		
	NL_Fa2	2184	7.43	46.96	696.03	57.33	0.65	635.84	176.88	26.53	22832	3611	178		
	NL_Fa3	1261	4.29	27.11	401.77	33.09	0.37	365.33	102.15	15.32	13189	2144	103		
	NL_Fa4	2031	6.91	43.66	647.12	53.31	0.60	584.60	164.67	24.70	21267	3589	168		
	NL_Fa5	2989	10.17	64.26	952.48	78.46	0.88	865.39	242.20	36.33	31273	5108	246		

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Emissions from field following manure application (kg N ha^{-1} yr^{-1})

Treatment	Farm	NH3	N2O	NOx	Surface Runoff	Leaching	Denitrification
Reference	NL_Fa1	15.13	4.00	2.84	8.15	3.62	25.33
	NL_Fa2	11.87	3.19	1.88	3.03	3.99	17.01
	NL_Fa3	9.84	2.72	1.30	5.22	13.56	45.58
	NL_Fa4	29.35	3.27	1.57	7.10	3.38	31.04
	NL_Fa5	20.12	4.81	2.31	6.97	6.58	50.86
Farm-scale AD	NL_Fa1	15.26	3.53	2.51	8.16	3.64	25.46
	NL_Fa2	12.00	2.76	1.63	3.03	4.00	17.04
	NL_Fa3	9.97	2.27	1.09	5.23	13.68	45.99
	NL_Fa4	30.01	2.85	1.37	7.10	3.34	30.73
	NL_Fa5	20.25	4.39	2.11	6.98	6.64	51.28
Grass-clover	NL_Fa1	11.55	2.45	2.68	10.36	3.62	25.33
	NL_Fa2	9.45	2.15	1.99	4.24	8.09	34.49
	NL_Fa3	8.46	2.12	1.61	8.33	37.49	126.01
	NL_Fa4	16.90	2.44	1.76	10.79	8.13	74.74
	NL_Fa5	14.76	2.50	1.99	8.42	1.81	13.96

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