Impacts on water quality of producing biogas on pig farms as a function of the associated agricultural practices

Ouarda Baziz, Fabrice Beline, Patrick Durand

Environmental Research Communications

OPEN ACCESS

RECEIVED
1 February 2024

REVISED
26 June 2024

ACCEPTED FOR PUBLICATION
2 July 2024

PUBLISHED
15 July 2024

PAPER

Impacts on water quality of producing biogas on pig farms as a function of the associated agricultural practices

Ouarda Baziz , Fabrice Beline and Patrick Durand
INRAE, Institut Agro Rennes-Angers, SAS, 35000, Rennes, France
E-mail: ouarda.baziz@inrae.fr

Keywords: anaerobic digestion, water quality, TNT2, systemic approach, watershed

Abstract

The aim of this study was to assess positive or negative impacts of anaerobic digestion (AD) on water quality using a systemic approach. To this end, we used the agro-hydrological model Topography-based Nitrogen Transfer and Transformation (TNT2), a spatially explicit model that simulates nitrogen and water flows at the watershed scale on a daily time step. Four scenarios were constructed and analyzed: a baseline before the introduction of AD (S0), AD with adjusted fertilization (S1), AD with unadjusted fertilization (S2), and agroecological AD (S3). The results showed that, when spreading practices were similar and an equivalent amount of effective nitrogen was applied, digested pig slurry generally had a predicted amount of nitrate leaching similar to that of undigested pig slurry. In addition, replacing catch crops with energy cover crops had little impact on water quality. Scenario S3 was the most favorable one for water quality and biogas production, but not for soil organic nitrogen storage and food and feed production. This study's strength is its systemic approach, which considered both environmental and agronomic aspects to assess the scenarios.

1. Introduction

The history of anaerobic digestion (AD) dates back several centuries, with the first records describing combustible gas produced by decomposing organic waste (Abbasi *et al* 2012). AD is a biological process in which organic matter decomposes in an anaerobic environment and produces biogas composed mainly of methane (CH_4) and carbon dioxide (Ahring 2003). This decomposition can produce renewable energy from waste and help contribute to carbon neutrality (Ward *et al* 2008). AD is developing in livestock farms, which produce a large amount of organic waste as manure, slurry, and crop residues. Livestock waste is one of the main concerns regarding the degradation of water quality (Hooda *et al* 2000) since it contains large amounts of nitrogen (N) (Follett and Hatfield 2001) and phosphorus (Knowlton *et al* 2004), which can leach into rivers and groundwater. In addition to producing renewable energy, AD also generates a residue as a by-product called 'digestate', which is composed of minerals and non-decomposed organic matter. Digestate can replace organic and inorganic fertilizers and other soil additives (Westerman *et al* 2012). The AD industry is growing rapidly. Germany is the leading biogas producer in Europe and the world, with more than 9000 digesters (Sesini *et al* 2023), while France ranks third, with more 1400 digesters.

French public policies support AD development in order to meet the national objectives of renewable energy production and carbon neutrality. Indeed, AD plays a crucial role in French energy prospects and ECC and fodder (including grassland) are the main substrates used to fulfil such objective (ADEME 2021). Thereby from 50 to 60 and 14 to 20 Twh/year could be produced by energy catch crops (ECC) and fodder, respectively, through AD in 2050 for a total energy production of AD between 113 and 151 TWh/year from the agriculture sector (Beline *et al* 2023). Initially developed using livestock and food processing waste, AD development is actually largely based on ECC and in a lesser extent on fodder such as maize and grass (Launay *et al* 2022). The introduction of ECC in AD systems is widely used by several countries (Szerencsits *et al* 2016, Strauß *et al* 2019, Riau *et al* 2021, Levavasseur *et al* 2023), mainly to ensure a continuous supply of substrate for biogas units without the need to rely on main crops and without competing with food production (Styles *et al* 2015). Unlike catch crops (CC), they are usually fertilized and sometimes irrigated. Despite their potential for maximizing agronomic and environmental benefits (Launay *et al* 2022), they are sometimes suspected of competing with the main crops and being less effective than CC in preventing NO_3 leaching. AD can also be considered as a technology that adds value to fodder crops (e.g. grassland, legumes (alfalfa) and thus may increase their cultivation, even on farms without ruminants, as predicted by forecasting studies (Couturier *et al* 2016). In Germany, 30 to 40% of biogas plants use grass from permanent grasslands as feedstock for the digester (Elsäßer *et al* 2012). Such changes could improve water quality. For sustainable management of anaerobic digestion, developing other biomasses that do not compete with food crops is crucial (Bedoić *et al* 2019).

However, the development of AD raises questions about environmental externalities and the associated risks of air, soil, and water pollution. While AD is often seen as a solution to improve the management of agricultural waste, particularly livestock waste, by replacing it with a more manageable digestate, the resulting impact of the series of changes to farm management entailed by an AD project remain under debate (Möller 2015).

Several studies have examined impacts of digestate on water quality at the field scale (Pötsch 2004, Möller and Stinner 2009, Svoboda *et al* 2013, Nicholson *et al* 2017). Although their results varied, few differences in the risks of nitrate (NO_3) leaching were observed between different types of livestock waste (whether digested or not) and inorganic fertilizers. However, digestate compositions and characteristics vary widely. Thus, to minimize the risks of NO_3 leaching, it is important to consider the composition of each type of digestate and adapt spreading practices accordingly. Since the characteristics of the inputs used in AD influence the quality of the digestate (Jimenez *et al* 2019), and these inputs depend on farmers' management practices, a comprehensive systemic approach is necessary to assess the environmental risks of AD properly.

There are more reasons advocating for a systemic approach. First, the flux of nitrate at the outlet of a catchment, even a small one, is generally not the sum of the leaching of each field: many processes related to spatial interactions, buffering, transit time, mixing of different aquifers, etc, result in a nitrate retention depending on the agricultural and physiographic context (e.g. Ruiz *et al* 2002, Abbott *et al* 2018, Ehrhardt *et al* 2019, Dupas *et al* 2020). Therefore, if an AD project modifies the distribution of crop rotations in the landscape, the resulting effect will be different that what it observed at field scale. Second, ECC can only be introduced in specific intercropping sequences, when the period of time between the harvest of the preceding crop and the sowing of the new one is long enough (e.g. between wheat and maize), whereas CC implantation is more flexible. Therefore, the proportion of CC and ECC and their resulting effect on nitrate concentrations can only be assessed at the farm or landscape scale. Third, AD implies more circularity in a farming system and different management of the crops, by-products and wastes, which may have consequences on the productivity, feed/food ratio, carbon sequestration, etc of the farm, and it is important to balance the positive or negative effects of AD on water pollution by gain or losses on other functions of the farm system.

Among all the farm systems, indoor pig breeding farms have often been identified as a major source of nitrogen pollution, since they concentrate large fluxes of nitrogen on a proportionally limited area (Sutton *et al* 2011). Indeed, hot spots of nitrogen pollution in the world, either atmospheric or aquatic, are associated to high density of livestock in general and often pig farming (Erisman *et al* 2014, Billen *et al* 2011, de Vries *et al* 2011). Pig farming is often associated with cereal cropping such as maize, wheat and barley fertilized with the manure and used to feeding the animals. This is typically the case in Europe, one of the major pig producing region in the world. Within Europe, Brittany, Western France, which produces 5% of the European pig production on less than 1% of the area, is representative of these hot spots, where AD develops rapidly to improve animal waste management.

This study aimed to analyze the combined effects of changes induced by AD on a farm's operating system or a group of farms. Unlike other studies, this study adopted a systemic approach at the farm scale rather than the field scale and attempted to consider all changes caused by introducing AD that could influence water quality. The underlying hypothesis was that these systemic changes have larger impacts, whether positive or negative, than does changing the types of fertilizers alone.

Given the many management practices available to farmers who invest in AD, the difficulty and cost of setting up a real-world experiment, and the time lag between a change in management practices and observable effects on stream water quality (Guillaumot *et al* 2021, Malik *et al* 2022), we adopted a scenario-based modeling approach. The objectives of this study were to (1) construct contrasting scenarios based on changes generated at the farm level by introduction AD to a pig farm, (2) predict impacts of introducing ECC to the cropping system on N flows and especially water quality, and (3) predict impacts of introducing permanent grassland into the cropping system. The scenarios constructed were simulated using the agro-hydrological model Topography-based Nitrogen Transfer and Transformations (TNT2) (Beaujouan *et al* 2002) along with simple, data-based farm and AD modeling approaches.

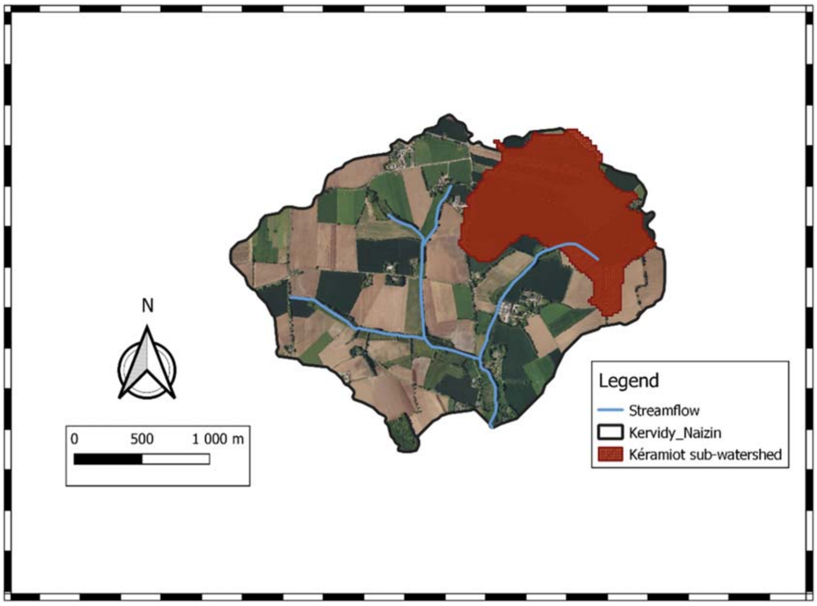

Figure 1. The Kervidy Naizin watershed and the Kéramiot sub-watershed.

2. Materials and methods

2.1. Study site and challenges of the approach

The Kervidy-Naizin watershed in Brittany (figure 1), western France, is associated with AgrHyS, a widely recognized long-term observatory in the fields of agriculture and the environment (Fovet *et al* 2018). It covers 4.9 km^2 and is drained by the Coët-Dan River, a tributary of the Evel River. It is characterized by intensive agriculture that combines mixed cropping (i.e., cereals, maize, grassland, and vegetables) and livestock farming (i.e., cattle, pigs, and poultry) on relatively moderate slopes (5%). Approximately 91% of the area is used for agricultural purposes.

Considered as an outdoor laboratory, the watershed is equipped with water monitoring devices, soil sensors, and other equipment to measure and record environmental parameters. It was chosen in the present study due to its long-term time series of quantitative and qualitative data: streamflow, daily NO_3 concentrations, cropping practices, and characteristics of the climate, soil, topography, and agricultural system.

The watershed has a temperate oceanic climate, with mean annual precipitation of 827 mm and a daily mean temperature of 11.2 °C. Its mean annual specific runoff is 314 mm, with seasonal variability. The dominant soils are clay loams 60–80 cm deep whose drainage depends on the slope. The watershed's elevation ranges from 98 m at the outlet to 140 m. AgrHys provides a large amount of data online (Ozcar-RI 2021), see Gascuel-Odoux *et al* (2018) for a history of the Kervidy-Naizin watershed. One of the main characteristics of these data is the high NO_3 concentration in stream water: ca. 16.5 mg L^{-1} N^{-1}-NO_3.

This study simulated N dynamics at the watershed scale by representing the processes involved in the 'soil-plant' compartment and in the hydrological pathways. The principle was to construct a 'virtual' farm whose fields would cover an entire sub-watershed of the Kervidy-Naizin watershed, and to compare different scenarios with and without AD implementation. The Kéramiot sub-watershed, drained by an upstream reach of the Kervidy, was chosen to represent the farm. Existing data on the farming operations of this sub-watershed were replaced with data from a virtual pig farm based on real soil, crop, and practice characteristics. The farm covered the entire sub-watershed area (i.e., 104 ha) and produced 5363 finishing pigs per year. The number of pigs on the farm was set to meet the regulatory limit of 170 kg spreadable organic N/ha/yr, based on the emission factor of 3.17 kg spreadable N/pig.yr (Légifrance 2016). We used weather data recorded from 1995–2020 at the study site to drive the simulation.

Simulating this system required representing agricultural practices (e.g., crop rotations, fertilization) before and after the introduction of AD. To this end, a survey was conducted of four farms in the municipality of Naizin that recently developed a collective AD project.

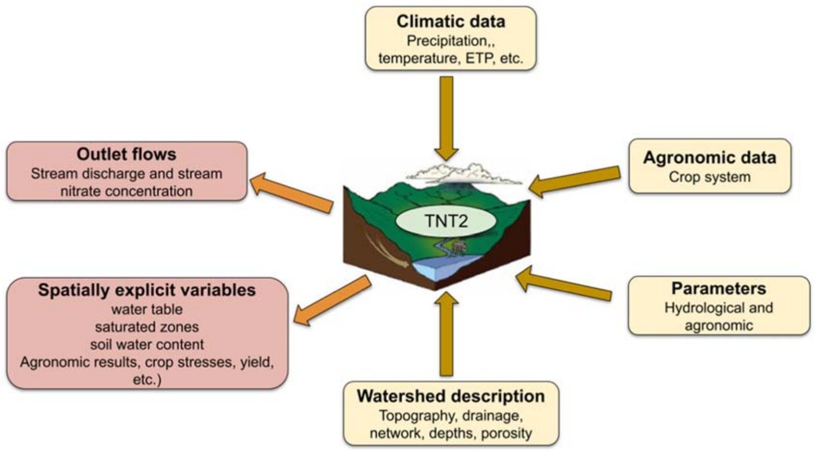

Figure 2. Diagramm of the TNT2 model. ETP : evapotranspiration.

2.2. Presentation of the models

2.2.1. Soil and groundwater nitrogen fluxes modelling by the tool topography nitrogen tranfert and transformations (TNT2)

TNT2 was used to simulate the fate of N in soil-crop-groundwater system scenarios (figure 2). It is a spatially explicit agro-hydrological model (Beaujouan *et al* 2001) that combines a hydrological model inspired by the model TOPMODEL (Beven 1997) with the agronomic crop and soil model STICS (Simulateur mulTIdisciplinaire pour les cultures standard) (Brisson *et al* 1998). As such, it is designed to model plant growth and N transfer and transformations in an agricultural watershed and plant growth based on soil and climate conditions on a daily time step (Beaujouan *et al* 2002). Thus, it also predicts NO_3 leaching, which can percolate below the soil and move in groundwater to the river.

STICS, developed at INRAE in France since 1996, simulates crop growth and soil water and N balances. It follows a mechanistic approach that describes the biophysical processes that govern plant growth and exchanges between crops and their environment (Brisson *et al* 2003). STICS simulates crop growth using daily weather data to drive water and N balances of the soil, as well as interactions between roots and soil. It can predict agricultural variables (e.g., yield, input consumption) and environmental variables (e.g., water dynamics, N losses) to help assess agronomic and environmental performances of crops. It can be adapted to different crops using relevant generic parameters for most crops and options in the model equations to represent the physiology and management of each crop.

The hydrological submodel of TNT2 is based on the assumption of TOPMODEL. It represents darcian saturated flow, a hydraulic gradient that is constant and equal to the topographic slope, hydraulic transmissivity that decreases exponentially as soil depth increases, runoff generated on saturated areas at valley bottoms, and dynamics described by a succession of steady states (Beven and Kirkby 1979). TNT2 uses a fixed grid to represent the watershed, with each cell divided into horizontal slices of varying thickness to distinguish immobile water from mobile water (Beaujouan *et al* 2002). The flows are vertical (downwards or upwards) in the unsaturated zone and lateral in the saturated zone, with multidirectional flows between neighboring cells at lower elevations. The topography of the watershed (calculated from a digital elevation model) is particularly important in the model, since it determines the distribution of surface and groundwater flows (Beven *et al* 2021). TNT2 simulates Hortonian runoff to represent the presence of poorly permeable areas. Hydrological parameters include maximum transmissivity, the exponential decay coefficient of transmissivity, retention and drainage porosities, the thicknesses of each layer considered, and infiltration rate, differentiated by soil type. To connect the two models, the STICS model was slightly simplified and integrated into the TNT2 code using C++ (GNU Genral Public License (GNU GPL)).

We used the parameters that had been calibrated in previous studies of the Kervidy-Naizin watershed (Casal *et al* 2019). Since the present study compared multiple scenarios, no calibration or validation was performed, and we assumed that the many studies that had used TNT2 in similar contexts provided sufficient confidence in the applicability and robustness of the model. The scenarios were simulated for 25 years (1995–2020) using the climate data record for Agrhys observatory.

Table 1. Characteristics of the inputs for the anaerobic digester in the simulated scenarios. All data came frome the MéthaSim database. BMP : biochemical methane potiential, ECC : energy cover crop, NH_4 : ammonium content.

Naming inputs	BMP (m^3 CH_4/t Organic matter)	Dry matter (kg/t)	Organic matter (kg/t)	Total N (kg N/t)	NH_4 (kgN/t)
Pig slurry	323	44.3	33.2	3.7	2.22
Maize stalks	294	793	750.9	8.7	0.2
Cereal straw	261	873	739.4	4.7	1.23
Green rye (ECC)	389	197	181.04	4.3	1.38
Grass	303	158	140	4.8	0.01

2.2.2. Modelling of nitrogen fluxes from animals and the effluents management

Because pigs had a strong influence on the farm's N dynamics, N input and output flows related to pig production were modelled using reference data used by the French administration (Légifrance 2016) to estimate N flows at the farm level. For the pig system, N enters through the purchase of post-weaning piglets and leaves through pig excretions, gaseous emissions, and the sale of fattened pigs. The piglets have a mean liveweight of 31 kg upon arrival (Légifrance 2016) and are sold at 118 kg. To calculate the corresponding N input and output flows from animals, the following equation was used from Dourmad *et al* (2016): number of pigs/yr $\times$ 0.02456 $\times$ liveweight.

N excretion was estimated assuming that each pig excreted 4.56 kg N/pig.yr (CORPEN 2004), while gaseous N emissions were calculated as excreted N minus spreadable N. Spreadable N amount comes from (Légifrance 2016), and is equal to 3.17 kg N/pig.yr. These N losses correspond mainly to ammonia (NH_3) emissions from building and storage. We assumed that gaseous N emissions did not differ among scenarios (i.e., regardless of the presence of AD), assuming that AD did not influence gaseous emissions from buildings and influenced those from storage only slightly. Indeed, raw livestock waste is stored for a shorter period with AD, which decreases associated gaseous emissions, than without AD. However, emission after AD due to the increase in the ammonium content and pH are higher, which results in an unclear trend in gaseous N emissions for the entire chain. Both livestock waste and digestate could be covered during storage, which would reduce their gaseous emissions.

The amount of N in the feed was calculated as the N retained plus N excreted by the pigs.

2.2.3. Modelling of anaerobic digestion

Predicting biogas production was of paramount importance since it assessed the energy-production potential of each AD scenario. The biogas production in each scenario was calculated using data from the French MéthaSim database (IFIP 2021), which provided the characteristics of each input (table 1). The biogaz production was assumed to be 80% of biochemical methane potential.

The amount of digestate (t fresh matter) was calculated by summing all of the inputs and then allocated to the crops on the farm. N enters the digester in organic form and exits mainly in the form of reduced inorganic N (NH_3). The total amount of N that enters AD equals the amount that exits, but the content of inorganic N increases after AD, while that of organic N decreases (Quideau *et al* 2013). The mineral content in the digestate was estimated at 75% for scenarios.

2.2.4. Description of the scenarios

Four scenarios were constructed to explore different options possible for a pig farm when adopting AD. We considered here changes that have already been observed in real farms and strived to make the scenarios as comparable as possible with respect to dimensions not directly related to AD. In particular, the total area of the farm and the number of pigs produced were kept constant. The four scenarios considered were: a baseline before the introduction of AD (S0), AD with adjusted fertilization (S1), AD with unadjusted fertilization (S2), and agroecological AD (S3) (table 2).

2.2.4.1. Baseline scenario without anaerobic digestion (S0)

The baseline scenario (S0) was based on current data from pig farms without AD (figure 3). The cropping system consists of crop rotations used to produce pig feed (35% grain maize, 30% wheat, and 10% barley, overall) or sold as cash crops (10% potatoes, 10% green beans, and 5% rapeseed, overall). In compliance with current regulations in Brittany, CC are grown between the main crops (ryegrass between winter and spring crops, mustard between winter crops), not fertilized, and buried at the end of their cycle. The residues of the main crops (including stalks) are also buried, and the straw is sold. The main crops are fertilized with pig slurry (at 66% for mineral content) and inorganic fertilizers up to the regulatory limits of total N load to the soil of ca. 210 kg ha^{-1} yr^{-1} and organic N load of 170 kg/ha/yr. All excess slurry is exported.

Table 2. Summary of the four scenarios (Scen.) simulated. Cover crop coverage of the sub-watershed studied includes catch crops (CC) (ryegrass and mustard) and an energy cover crop (ECC) (green rye). The ammonia (NH_3) emission factors (EF) are those used for buried or unburied slurry or digestate.

Sce.	Definition	Cover crop coverage	Fertilization	NH_3 EF		Description
				Buried	Unburied	
S0	No anaerobic digestion	58% in CC	Pig slurry + inorganic fertilizers	5%	17%	Based on current practices (Naizin farm). All excess slurry was exported.
S1	Anaerobic digestion and adjusted fertilization	14% in CC + 42% in ECC	Raw digestate + inorganic fertilizers	13%	25%	Incorporation of ECC. Organic fertilization replaced with digestate (up to the limit of 170 kg N/ha). All excess digestate was exported.
S2	Anaerobic digestion and non-adjusted fertilization	14% in CC + 42% in ECC	Raw digestate + inorganic fertilizers	13%	25%	Incorporation of ECC. Organic fertilization replaced with digestate (exceeds the limit of 170 kg N/ha). All excess digestate was exported
S3	Agroecological scenario	12% in CC + 40% in ECC	Raw digestate + inorganic fertilizers	13%	25%	Incorporation of ECC. Introduction of 10% of permanent grassland into the cropping system. All excess digestate was exported

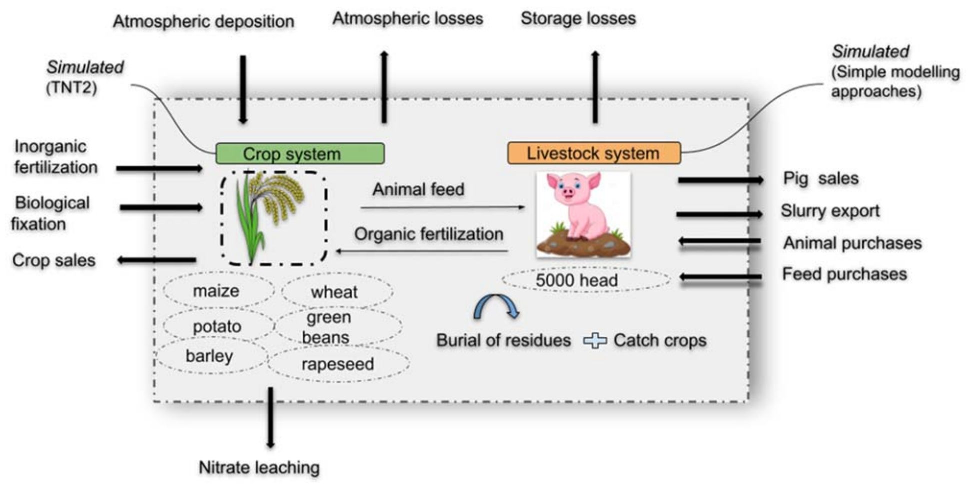

Figure 3. Nitrogen flows considered in scenario S0.

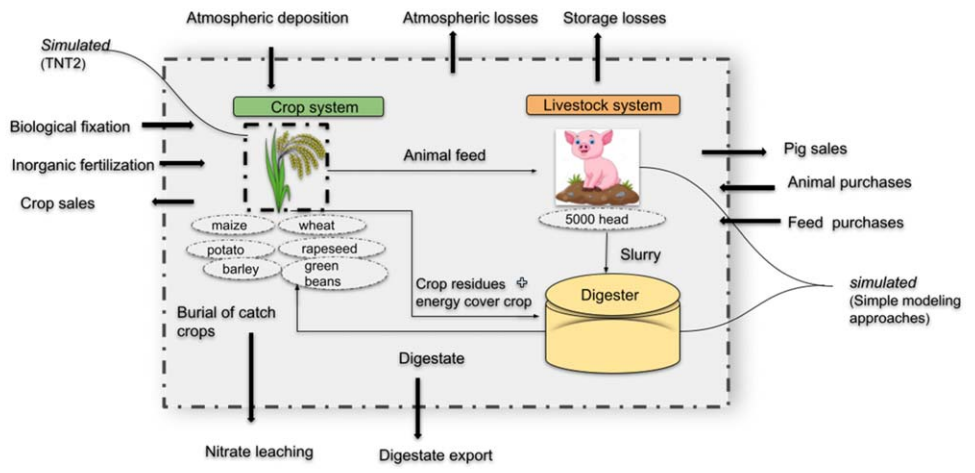

Figure 4. Nitrogen flows considered in scenario S1 and S2.

For this scenario, an NH_3 emission factor at spreading of 5% or 17% was used when the slurry was buried or not buried, respectively.

2.2.4.2. Scenario with anaerobic digestion and adjusted fertilization (S1)

In scenario S1, a digester was introduced into the farm (figure 4). The cropping system is similar to S0 as much as possible, but when possible, the CC are replaced with the ECC (i.e., green rye, commonly grown as an ECC in Brittany). Replacing CC with an ECC aims to avoid competition with main crops, but the growth cycle of certain ones, such as maize, still needed to be shortened to insure a sufficient growing time for the ECC. The remaining CC (during short periods between crops) were still buried. The residues of the main crops (including stalks) are also buried and the straw is sold.The crop residues, ECC, and pig manure are fed into the digester. The main crops are fertilized with raw liquid digestate and inorganic fertilizers. The ECC was fertilized with raw liquid digestate at ca. 100 kg N/ha. To meet the regulatory limits and have the same total N load to the soil as in S0, fertilization of the main crops was reduced accordingly. For this and the other AD scenarios (S2 and S3), an NH_3 emission factor of 13% or 25% was used when the digestate was buried or not buried, respectively.

2.2.4.3. Scenario with anaerobic digestion and unadjusted fertilization (S2)

In scenario S2, the system was similar to that in S1, but fertilization of the main crops was not reduced, which increased the total N load to the soil (figure 4). The objective was to simulate a relatively common situation in which the farmer does not consider the increased N load caused by fertilizing ECC. All excess digestate was exported.

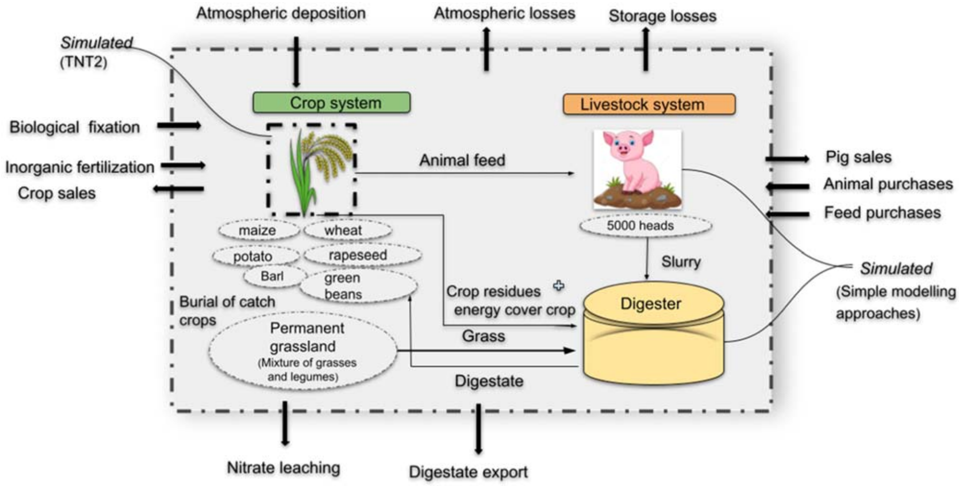

Figure 5. Nitrogen flows considered in scenario S3.

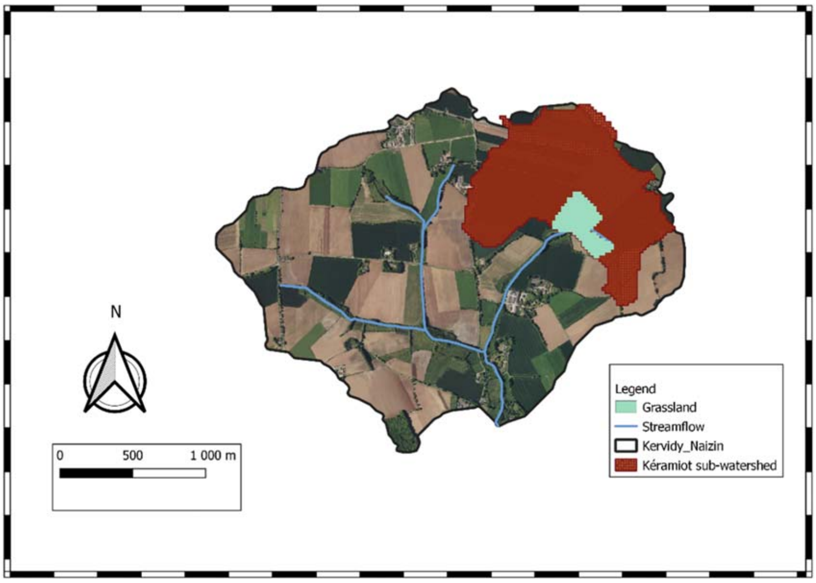

Figure 6. Location of grassland plot in Kéramiot-sub-watershed.

2.2.4.4. Agroecological scenario S3

Agroecology is an approach to agriculture that aims to develop ecologically sustainable and socially equitable agricultural production systems (Wezel *et al* 2009). It integrates agricultural practices that minimize the use of chemicals and promote biodiversity while improving soil quality and long-term productivity (Altieri 2018). Among these practices, permanent grassland can be introduced into a farm with AD and its biomass fed into the digester. Thus, the grassland is used as an ECC (McEniry and O'Kiely 2013), as in other bio-energetic systems (Prochnow *et al* 2009).

In scenario S3 (figure 5), ca. 10% of the watershed area near the stream was converted into permanent grassland composed of a mixture of ryegrass and clover. This grassland was not fertilized and was mown four times per year (i.e., May, June, July, and October).

The location (figure 6) near the stream was chosen because of its potential to intercept N leached from upslope (Casal *et al* 2019).

To summarize differences among the scenarios, we calculated seven performance indices: N-use efficiency of the farm (Gross NUE) and cropping system (Soil NUE), proportion of the total N input lost to water, (Nw/Nin) or the atmosphere (Natm/Nin), proportion of the total N input transformed into food products

Table 3. Nitrogen (N)-related performance indices calculated to assess the scenarios. All variables are in kg N/ha/yr except for Biog (m^3/yr).

Index	Description and equation
Gross NUE	N-use efficiency of the farm: $= (Ncrops + Npigs + Nwaste)/(Nfert + Natm + Npp + Npfeed)$
Soil NUE	N-use efficiency of the cropping system: $= (Ncrops + Nafeed + Nec)/(Nfert + Norg + Natm + Nbf)$
Nw/Nin	Proportion of the total N input lost to water: $= Nno3/(Nfert + Natm + Nbf + Npp + Npfeed)$
Natm/Nin	Proportion of the total N input lost to the atmosphere: $= (Nnh3 + Ndenit)/(Nfert + Natm + Nbf + Npp + Npfeed)$
Nexpfood/Nin	Proportion of the total N input transformed into food products: $= (Ncrops + Npigs + Nwaste)/(Nfert + Natm + Npp + Npfeed)$
Biog/Nin	Biogas production per kg of N input: $= Biog/(Nfert + Natm + Nbf + Npp + Npfeed)$
N org soil	Organic soil nitrogen conservation: $= 0$ if $\Delta N > 0$: $= -\Delta N/(Ncrop + Nafeed + Nec)$ if $\Delta N < 0$

(Nexpfood/Nin), biogas production per kg of N input (Biog/Nin), and (N org soil) organic soil conservation (table 3).

Variable abbreviations: Ncrops: nitrogen (N) exported by harvested crops, Nwaste: N in pig waste, Nafeed: N in the grain harvested to produce feed for the farm, Nec: N in the biomass harvested for the digester, Npigs: N in sold pigs, Npp: N in purchased piglets, Npfeed: N in purchased feed, Nno3: N load as nitrate at the outlet, Nnh3: N load as ammonia emitted in the fields, Ndenit: denitrification load, Natm: atmospheric N deposition, Nbf: biological N fixation, ΔN: change in soil organic N content, Biog: biogas production.

3. Results

3.1. Impact of scenarios on water quality at the outlet

After 25 simulated years, cumulative NO_3 losses at the outlet of scenarios S1 and S2 were 1.89% and 12.2% larger, respectively, than those of S0, while those of S3 were 15.1% lower (figure 7). Differences among the scenarios became visible after five years, and seemed to stabilize after 20 years. The average mean annual flow for the last five years for the four scenarios are, respectively, 30 Kg N/ha, 31 Kg N/ha, 35 Kg N/ha and 25 Kg N/ha.

Since the water flux at the outlet is similar for the four scenarios, these differences are also visible for the concentrations (figure 8). The average concentration for the last five years for the four scenarios (S0, S1, S2 and S3) are, respectively 46 g m^{-2}, 46 g m^{-2}, 53 g m^{-2} and 36 g m^{-2}.

3.2. Impact of scenarios on total nitrogen flow at the farm level

Annual N inflows to and outflows from the soil for the scenarios were represented using Sankey diagrams (figure 9). The N inflows included atmospheric deposition, biological N fixation, inorganic fertilization, and organic fertilization (pig slurry in S0 and digestate in S1, S2, and S3). Inorganic fertilization was relatively similar among the scenarios. By construction, organic fertilization was highest in S2 and lowest in S3, while biological N fixation was the highest in S3 due to the grassland legumes. Denitrification in S0 was slightly higher than that in S1 and S3 and nearly equivalent to that in S2 (ca. 18, 16, 15, and 19 kg N/ha/yr, respectively). NH_3 volatilization from the fields was highest in S2, followed by S0, S1, and S3 (ca. 21, 18, 17, and 15 kg N/ha/yr, respectively). NH_3 losses in the building and during storage were the same for all four scenarios (74 kg N/ha/yr) because they had the same number of pigs, and we assumed that emissions from slurry and digestate did not differ during storage.

Organic N and carbon were stored in the soil in S0, while soil organic matter decreased in the other three scenarios, but less so in S3 than in S1 and S2. The assumption that stalks are left in the field in S0 but fed to the digester in the other scenarios explains the increase in N export in harvested cereals in the latter. Specifically, grain production decreased slightly in S2 but more in S3 and S1. In addition, S3 produced 5% more biogas than did S1 and S2, which produced similar amounts.

The dotted frame represent the boundaries of pig farming, and the arrows represent input, output, and incoming flows into the system

The profiles of the indices of the scenarios (figure 10) illustrated the main features of each scenario. S0 produced food without mining N from the soil, but lost more NO_3 and had a low NUE. S1 had higher Soil NUE but performed poorly in terms of soil N mining and production, and emitted more pollutants than S0 due to

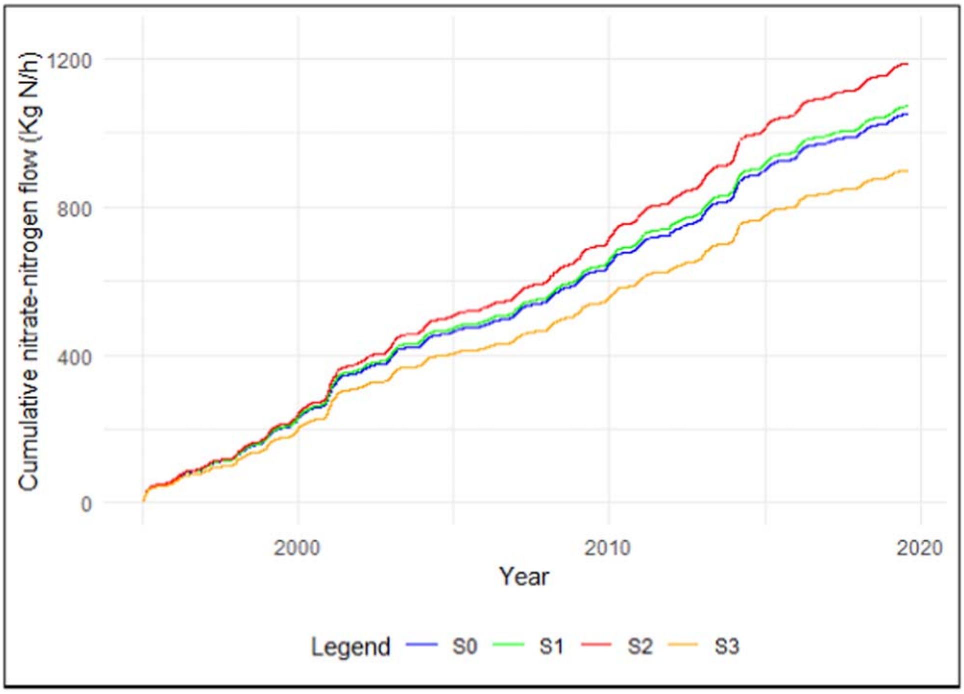

Figure 7. Cumulative nitrogen flow at the outlet of the sub-watershed predicted by TNT2 from 1995–2020.

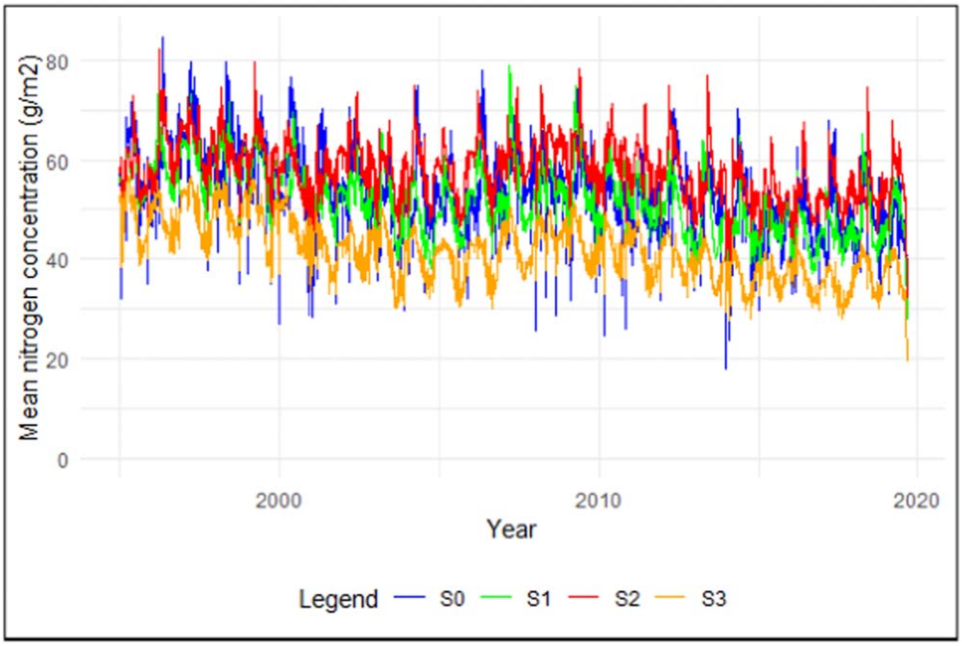

Figure 8. Mean nitrogen concentration at the outlet of the sub-watershed predicted by TNT2 from 1995–2020.

increased NH_3 volatilization. S2 was the most efficient in terms of production (i.e., biogas and food) but also the most polluting. S3 had good performances for NUE, environmental losses and biogas production, but at the expense of less food and feed production.

4. Discussion

Modeling with TNT2 allowed us to compare a set of realistic changes induced by the introduction of biogas production on a pig farm and predict its long-term effects on NO_3 losses at the watershed scale. Comparing scenarios without changing TNT2's parameters limited uncertainty in the predictions, but epistemic

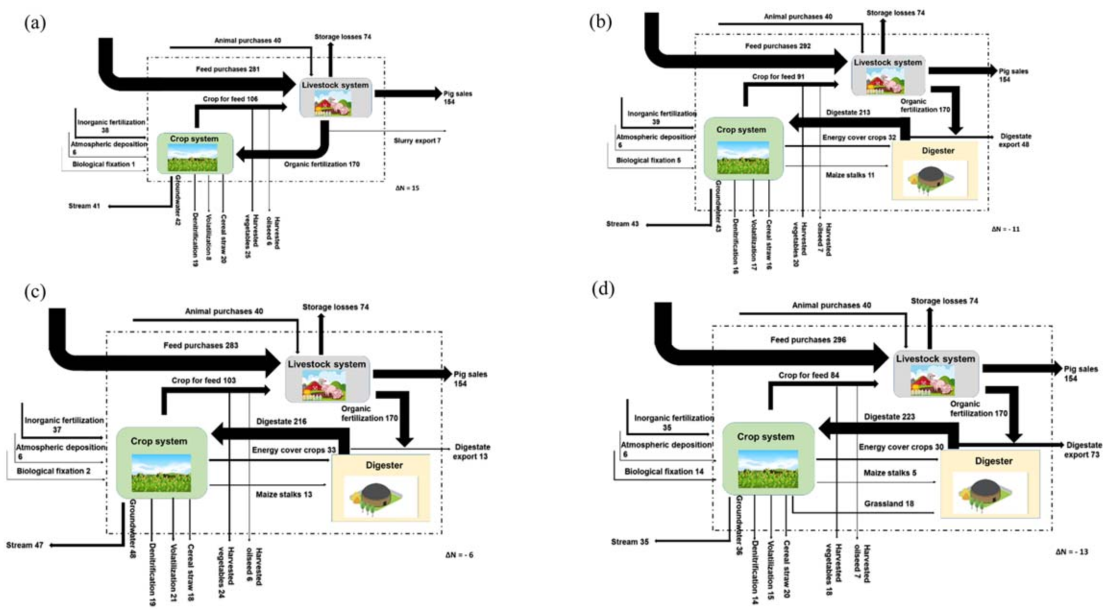

Figure 9. Sankey diagrams of annual nitrogen dynamics for scenarios (a) S0, (b) S1, (c) S2, and (d) S3.

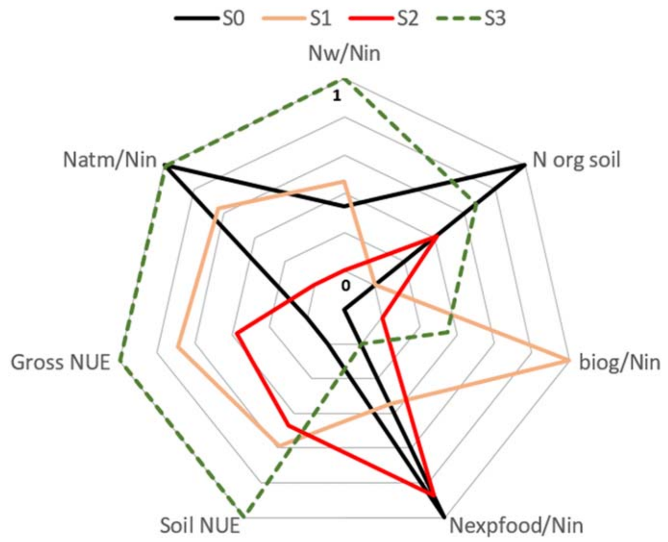

Figure 10. Radar representation of the performance indices of the scenarios. The indices are normalized by their range for the sake of clarity, i.e., (max-value)/(max-min) for Natm, Nw, Norg soil, and 1-(max-value)/(max-min) so that values closer to the edge are better. See table 3 for definitions of the indices.

uncertainties due to the model structure and equations remained (Walker *et al* 2003) and could be assessed only by using an ensemble modeling approach (i.e., using different models with the same inputs), which lay beyond the scope of this study. Moreover, the specific soil and climate contexts and characteristics of the simulated farm call for caution when generalizing the conclusions to other locations. Indeed, pig farming systems are very diverse and the purpose of this study was not to represent this diversity. The farm used here is fairly typical for Western Europe, where pig farming is associated with grain production providing part of the feeding of the pigs. Having adopted a process-based, systemic approach provides useful insights into the main factors that influence the environmental impacts of agricultural biogas production, but the magnitude of these impacts will be specific of the context. Therefore, we are confident that the results confirmed the initial hypothesis that the impact of AD on water quality depends mainly on the changes in practices induced by the need to feed the digester (Möller 2015).

Scenarios S0 and S1 had similar predicted NO_3 losses at the outlet, which suggests that replacing slurry with digestate does not increase NO_3 leaching. This result is consistent with those of previous studies: when spreading practices are similar and an equivalent amount of effective N is applied, digested pig slurry generally has a risk of leaching similar to that of undigested pig slurry (Möller and Stinner 2009, Nicholson *et al* 2017).

Furthermore, replacing CC with ECC, even when fertilized, did not influence leaching greatly when the total amount of N applied per year was similar. It is possible that exporting crop residues rather than incorporating them into the soil compensates for the increased risk of NO_3 leaching due to fertilization. In addition, in S1, fertilization of the main crops was reduced, which can decrease the amount of N left in the soil after harvest (Malone *et al* 2018). In comparison, in S2, in which fertilization of the main crops was not reduced, the amount of NO_3 leaching increased, meaning that NO_3 leaching tends to increase as theamount by which the regulatory limit is exceeded increases. This highlights the importance of managing the entire crop rotation optimally (Launay *et al* 2022), particularly its N fertilization.

Specialization and intensification of livestock production systems have changed land use, and the area of permanent grassland has decreased in many regions, either because they are converted to annual crop successions or abandoned and becam fallow land. Given the environmental benefits of permanent grassland, producing energy from its biomass can be one way to reverse this trend (Khalsa *et al* 2014). Indeed, feeding grass from grassland into on-farm digesters is expected to become more common in northwestern Europe (McEniry and O'Kiely 2013). In agreement with many other studies (e.g., Jankowska-Huflejt (2006) and Malik *et al* (2022)), introducing extensively managed grassland in S3 decreased NO_3 losses effectively.

Denitrification was highest in S0 and S2, likely due to the larger total N load to the soil for S2 (Mulvaney *et al* 1997) and the plowing of the CC in late winter for S0. The denitrification rate is controlled mainly by the amount of NO_3 in soils in wet conditions (Friedl *et al* 2016). Detailed studies of reduction processes in wet soils in Brittany (Jaffrézic 1997) show that anoxic conditions prevail when the soils are still wet and drainage slows down (i.e., usually in late February-March). Some processes, such as N20 emissions due to nitrification of ammonium applied, are not considered in this version of the model. However, considering that this process represent usually less than 1% of the nitrogen applied (Charles *et al* 2017) and that ammonium content in digestate and slurry are similar, this overlooked process is not likely to bias the results.

Our results suggest a loss of soil organic matter in the soil under anaerobic digestion scenarios, contrary to several studies (Launay *et al* 2022, Levavasseur *et al* 2023). This divergence could be attributed to several factors. First, we used the same decomposition parameters for digestate as for pig slurry, although its residual organic matter is probably more resistant. Second, the maize stalks are not buried but rather exported to the digester, which makes a significant difference in the organic input to the soil. This result should therefore be considered with caution.

Field NH_3 volatilization depends mainly on the amounts and types of fertilizers applied and the spreading technique (Launay *et al* 2022), which are the factors used in TNT2 to estimate it. Fertilization with digestate was assumed to have a higher volatilization rate (S1, S2, and S3) than fertilization with slurry (S0) due to digestate's higher proportion of reduced inorganic N and higher pH (Möller 2015). Several studies have shown that fertilization with digestates has higher NH_3 volatilization than does fertilization with raw livestock waste (Möller and Stinner 2009, Ni *et al* 2012, Crolla *et al* 2013, Lili *et al* 2016, Nicholson *et al* 2017). However, we used relatively low volatilization factors as compared to these references, considering that volatilization could be reduced through phase separation of the digestate (Svehla *et al* 2020) and/or less-polluting application techniques, such as injection (Riva *et al* 2016, Maris *et al* 2021).

We chose to limit the changes to a small set of practices to facilitate identification of the main controlling processes. Introducing AD likely induces a wider range of changes, and their effects on farm finances modify the priorities and investment capacities of farmers, leading to more radical changes in the production system.

The index profiles highlight the importance of using multiple criteria to assess the performances of a given system. In particular, they show the limits of using NUE as the only criterion: Soil NUE for S3 was highest when including digestate export, but the lowest when considering food and feed production only (Nexpfood/Nin); in contrast, S2 had higher NUE than S0 but emitted much larger amounts of pollutants.

5. Conclusion and perspectives

Biogas production has become a full-fledged agricultural practice, but its environmental impacts, both positive and negative, are still under debate. The study clearly demonstrated that changes in practices induced by AD can increase or decrease water quality. The cropping system associated with AD is the most influential factor: the crop rotation and management of fertilization and residues influence the overall sustainability of the system. It is possible to design systems that increase biogas and digestate production while maintaining water quality, but at the expense of food and feed production.

The four scenarios were tested through modeling, which enabled exploration of multiple options and increased understanding of the impacts of digester on water quality in a pig farming system. Other performances of the scenarios tested (e.g., other ecosystem services, biotechnical feasibility, profitability) remain to be assessed. Based on this information, policy makers could design measures to promote the most sustainable practices. TNT2 will be used to test other scenarios, including those for dairy or crop farms, with the final aim of upscaling simulations to a larger watershed.

Although this study is based on a single situation, the farming system modelled here includes the most typical features of pig breeding farms in the major temperate regions of production: high density of animals, large amount of manure to manage, winter and spring crops in rotation. Consequently, if the magnitudes of fluxes presented here cannot be simply extrapolated, the main conclusions and the approach could be easily transposed to other regions. Indeed, the present study shows that environmental assessment of agricultural AD requires a comprehensive and systemic approach and is an invitation to generalize this type of approach to other contexts.

Acknowledgments

Ouarda Baziz PhD is funded by the INRAE research department Aqua and GRDF company (MethaEau project). The authors thank all the colleagues that provided advices to this work, and particularly (Gruau Gerard, Safya Menasseri-Aubry, Romain Girault, Vincent Jean-Baptiste, Armelle Damiano, Laura Toulet, Nouraya Akkal-Corfini, Olivier Godinot, Melynda Hassouna and Aurélie Wilfart). We also thank Michelle and Michael Corson for the thorough and helpful proofreading of the manuscript.

Data availability statement

The data that support the findings of this study are openly available at the following URL/DOI: https://doi.org/https://geosas.fr/web/?page_id=103.

ORCID iDs

Ouarda Baziz https://orcid.org/0009-0007-9316-9022
Fabrice Beline https://orcid.org/0000-0003-1885-1842
Patrick Durand https://orcid.org/0000-0002-0984-693X

References

Abbasi T, Tauseef S M and Abbasi S A 2012 A Brief History of Anaerobic Digestion and "Biogas" *Biogas Energy* vol 2 ed Springer (SpringerBriefs in Environmental Science) p 11

Abbott B W *et al* 2018 Unexpected spatial stability of water chemistry in headwater stream networks *Ecology Letters* **21** 296–308

ADEME 2021 Transition(s) 2050. Choisir maintenant. Agir pour le climat *Rapport* 685 (https://librairie.ademe.fr/recherche-et-innovation/5072-prospective-transitions-2050-rapport.html) (accessed 22 May 2024)

Ahring B K 2003 Perspectives for anaerobic digestion *Biomethanation I* (*Advances in Biochemical Engineering/Biotechnology*) 81 ed B K Ahring *et al* (Springer) pp 1–30

Altieri M A 2018 *Agroecology: The Science Of Sustainable Agriculture* 2nd edn, ed M A Altieri (CRC Press) (https://doi.org/10.1201/9780429495465)

Beaujouan V, Durand P and Ruiz L 2001 Modelling the effect of the spatial distribution of agricultural practices on nitrogen fluxes in rural catchments *Ecol. Model.* **137** 93–105

Beaujouan V, Durand P, Ruiz L, Aurousseau P and Cotteret G 2002 A hydrological model dedicated to topography-based simulation of nitrogen transfer and transformation: rationale and application to the geomorphology– denitrification relationship *Hydrol. Process.* **16** 493–507

Bedoić R, Čuček L, Ćosić B, Krajnc D, Smoljanić G, Kravanja Z, Ljubas D, Pukšec T and Duić N 2019 Green biomass to biogas – A study on anaerobic digestion of residue grass *Journal of Cleaner Production* **213** 700–9

Beline F *et al* 2023 (https://hal.inrae.fr/hal-04093266v1)

Beven K J 1997 *Distributed Hydrological Modelling: Applications of the Topmodel Concept* (Wiley)

Beven K J and Kirkby M J 1979 A physically based, variable contributing area model of basin hydrology / Un modèle à base physique de zone d'appel variable de l'hydrologie du bassin versant *Hydrol. Sci. Bull.* **24** 43–69

Beven K J, Kirkby M J, Freer J E and Lamb R 2021 A history of TOPMODEL *Hydrol. Earth Syst. Sci.* **25** 527–49

Billen G *et al* 2011 Nitrogen Flows From European Regional Watersheds to Coastal Marine Waters *The European Nitrogen Assesment. Souces, Effects and Policy Perspectives* ed MA Sutton *et al* (Cambridge University Press) (https://doi.org/10.1017/CBO9780511976988.016)

Brisson N *et al* 2003 An overview of the crop model stics *Eur. J. Agron., Modelling Cropping Systems: Science, Software and Applications* **18** 309–32

Brisson N *et al* 1998 STICS: a generic model for the simulation of crops and their water and nitrogen balances. I. theory and parameterization applied to wheat and corn *Agronomie* **18** 311–46

Casal L, Durand P, Akkal-Corfini N, Benhamou C, Laurent F, Salmon-Monviola J, Ferrant S, Probst A, Probst J-L and Vertès F 2019 Reduction of stream nitrate concentrations by land management in contrasted landscapes *Nutr. Cycl. Agroecosystems* **114** 1–17

Charles A, Rochette P, Whalen J K, Angers D A, Chantigny M H and Bertrand N 2017 Global nitrous oxide emission factors from agricultural soils after addition of organic amendments: a meta-analysis *Agric. Ecosyst. Environ.* **236** 88–98
CORPEN 2004 Circulaire du 19/08/04 relative aux installations classées - élevages de porcs. Utilisation de nouvelles références de rejet. Bulletin Officiel MEDD (https://bulletin-officiel.developpement-durable.gouv.fr/notice?id=Bulletinofficiel-0003856&reqId=e9073348-1776-4084-903d-cb2f9203cad4&pos=1)
Couturier C, Charru M, Doublet S and Pointereau P 2016 *Afterres 2050* 96 (https://afterres2050.solagro.org/wp-content/uploads/2021/06/solagro_afterres2050_version2016_english.pdf)
Crolla A, Kinsley C and Pattey E 2013 Land application of digestate *The Biogas Handbook, Sciences, Production and Applications* (*Woodhead Publishing Series in Energy*) ed A Wellinger *et al* (Woodhead Publishing) pp 302–25
de Vries W, Leip A and Winiwarter W 2011 Geographic variation in terrestrial nitrogen budgets across Europe, ed M A Sutton *et al The European Nitrogen Assessment* (Cambridge University Press)
Dourmad J Y, Levasseur P, Daumer M, Hassouna M, Landrain B, Lemaire N, Loussouarn A, Salaün Y and Espagnol S 2016 *Évaluation Des Rejets D'azote, Phosphore, Potassium, Cuivre Et Zinc Des Porcs.* (RMT Elevages et Environnement) p 26
Dupas R, Ehrhardt S, Musolff A, Fovet O and Durand P 2020 Long-Term Nitrogen Retention and Transit Time Distribution In Agricultural Catchments In Western France *Environmental Research Letters* **15** 115011
Ehrhardt S, Kumar R, Fleckenstein J H, Attinger S and Musolff A 2019 Trajectories of nitrate input and output in three nested catchments along a land use gradient *Hydrol. Earth Syst. Sci.* **23** 3503–24
Elsäßer M, Messner J, Keymer U, Roßberg R and Setzer F 2012 *Biogas from grass. How grassland growths can contribute to producing energy* DLG, German Agricultural Society 19 . (https://dlg.org/fileadmin/downloads/landwirtschaft/themen/publikationen/merkblaetter/dlg-merkblatt_386_e.pdf)
Erisman J W, Van G H, Grizzetti B, Bouraoui F, Powlson D, Sutton M A, Bleeker A and Reis S 2014 *The European Nitrogen Problem in a Global Perspective [WWW Document]* (JRC Publ. Repos) (https://publications.jrc.ec.europa.eu/repository/handle/JRC57902) accessed 24 June 2024
Follett R F and Hatfield J L 2001 Nitrogen in the environment: sources, problems, and management *Sci. World J.* **1** 920–6
Fovet O *et al* 2018 AgrHyS: an observatory of response times in agro-hydro systems *Vadose Zone J.* **17** 180066
Friedl J, Scheer C, Rowlings D W, McIntosh H V, Strazzabosco A, Warner D I and Grace P R 2016 Denitrification losses from an intensively managed sub-tropical pasture—impact of soil moisture on the partitioning of N2 and N2O emissions *Soil Biol. Biochem.* **92** 58–66
Gascuel-Odoux C, Fovet O, Gruau G, Ruiz L and Merot P 2018 Evolution of scientific questions over 50 years in the Kervidy-Naizin catchment: from catchment hydrology to integrated studies of biogeochemical cycles and agroecosystems in a rural landscape *Cuad. Investig. Geográfica* **44** 535–55
Guillaumot L, Marçais J, Vautier C, Guillou A, Vergnaud V, Bouchez C, Dupas R, Durand P, de Dreuzy J-R and Aquilina L 2021 A hillslope-scale aquifer-model to determine past agricultural legacy and future nitrate concentrations in rivers *Sci. Total Environ.* **800** 149216
Hooda P S, Edwards A C, Anderson H A and Miller A 2000 A review of water quality concerns in livestock farming areas *Sci. Total Environ.* **250** 143–67
IFIP 2021 Base de données MéthaSim Available at: (https://ifip.asso.fr/base-de-donnees-methasim/) (accessed 18 July 2023)
Jaffrézic A 1997 Géochimie des eléments métalliques, des nitrates et du carbone organique dissous dans les eaux et les sols hydromorphes. agriculture intensive et qualité des eaux dans les zones humides en bretagne *PhD Thesis* Agrocampus - Ecole nationale supérieure d'agronomie de Rennes. Mémoires de Géosciences Rennes, n° 79, 296 p
Jankowska-Huflejt H 2006 The function of permanent grasslands in water resources protection *J. Water Land Dev.* **10** 55–65
Jimenez J, Patureau D, Grigatti M and Bernet N 2019 Impact de la typologie des digestats sur les nutriments disponibles pour la plante : utilisation d'indicateurs de caractérisation pour prédire la fertilisation minérale *Les 14è RENCONTRES de la fertilisation raisonnée et de l'analyseLe rendez-vous biennal des professionnels de la fertilisation raisonnée (Dijon (France), Nov 2019)*
Khalsa J, Fricke T, Weigelt A and Wachendorf M 2014 Effects of species richness and functional groups on chemical constituents relevant for methane yields from anaerobic digestion: results from a grassland diversity experiment *Grass Forage Sci.* **69** 49–63
Knowlton K F, Radcliffe J S, Novak C L and Emmerson D A 2004 Animal management to reduce phosphorus losses to the environment1 *J. Anim. Sci.* **82** E173–95
Launay C, Houot S, Frédéric S, Girault R, Levavasseur F, Marsac S and Constantin J 2022 Incorporating energy cover crops for biogas production into agricultural systems: benefits and environmental impacts. a review *Agron. Sustain. Dev.* **42** 57
Légifrance 2016 Available at: (https://legifrance.gouv.fr/loda/id/JORFTEXT000033227190)
Levavasseur F *et al* 2023 Energy cover crops for biogas production increase soil organic carbon stocks: A modeling approach *GCB Bioenergy* **15** 224–38
Lili W, Wenzhe L, Zhongjiang W, Zhiwu W, Chao S and Yan L 2016 Effects of digestate application depth on soil nitrogen volatilization and vertical distribution *Int. J. Agric. Biol. Eng.* **9** 101–7
Malik W, Oehler F, Sgro L and Durand P 2022 Nitrogen mitigation scenarios to reduce coastal eutrophication *Agrosystems Geosci. Environ.* **5** 1–17
Malone R W *et al* 2018 Harvesting fertilized rye cover crop: simulated revenue, net energy, and drainage nitrogen loss *Agric. Environ. Lett.* **3** 170041
Maris S C, Capra F, Ardenti F, Boselli R, Pochintesta D, Beone G M, Tabaglio V and Fiorini A 2021 The interaction between types of cover crop residue and digestate application methods affects ammonia volatilization during maize cropping season *J. Environ. Qual.* **50** 504–12
McEniry J and O'Kiely P 2013 Anaerobic methane production from five common grassland species at sequential stages of maturity *Bioresour. Technol.* **127** 143–50
Möller K 2015 Effects of anaerobic digestion on soil carbon and nitrogen turnover, N emissions, and soil biological activity. a review *Agron. Sustain. Dev.* **35** 1021–41
Möller K and Stinner W 2009 Effects of different manuring systems with and without biogas digestion on soil mineral nitrogen content and on gaseous nitrogen losses (ammonia, nitrous oxides) *Eur. J. Agron.* **30** 1–16
Mulvaney R L, Khan S A and Mulvaney C S 1997 Nitrogen fertilizers promote denitrification *Biol. Fertil. Soils* **24** 211–20
Ni K, Pacholski A, Gericke D and Kage H 2012 Analysis of ammonia losses after field application of biogas slurries by an empirical model *J. Plant Nutr. Soil Sci.* **175** 253–64
Nicholson F, Bhogal A, Cardenas L, Chadwick D, Misselbrook T, Rollett A, Taylor M, Thorman R and Williams J 2017 Nitrogen losses to the environment following food-based digestate and compost applications to agricultural land *Environ. Pollut.* **228** 504–16
Ozcar-RI 2021 Available at: (https://ozcar-ri.org/fr/observatoire-agrhys/)

Pötsch E 2004 *Nährstoffgehalt von Gärrückständen aus landwirtschaftlichen Biogasanlagen und deren Einsatz im Dauergrünland - PDF Kostenfreier Download [WWW Document]* URL (https://docplayer.org/79196105-Naehrstoffgehalt-von-gaerrueckstaenden-aus-landwirtschaftlichen-biogasanlagen-und-deren-einsatz-im-dauergruenland.html) accessed 24 October 2024

Prochnow A, Heiermann M, Plöchl M, Linke B, Idler C, Amon T and Hobbs P J 2009 Bioenergy from permanent grassland—a review: 1 *Biogas. Bioresour. Technol.* **100** 4931–44

Quideau P, Morvan T, Guiziou F, Daumer M-L, Pourcher A-M and Béline F 2013 Les effets et conséquences de la méthanisation sur la matière organique et l'azote des lisiers de porc *Sci. Eaux Territ. Numéro* **12** 66–71

Riau V, Burgos L, Camps F, Domingo F, Torrellas M, Antón A and Bonmatí A 2021 Closing nutrient loops in a maize rotation. Catch crops to reduce nutrient leaching and increase biogas production by anaerobic co-digestion with dairy manure *Waste Manag.* **126** 719–27

Riva C, Orzi V, Carozzi M, Acutis M, Boccasile G, Lonati S, Tambone F, D'Imporzano G and Adani F 2016 Short-term experiments in using digestate products as substitutes for mineral (N) fertilizer: agronomic performance, odours, and ammonia emission impacts *Sci. Total Environ.* **547** 206–14

Ruiz L, Abiven S, Durand P, Martin C, Vertes F and Beaujouan V 2002 How nitrate concentration in stream water is affected by agricultural practices in small agricultural catchments? I-Annual nitrogen budgets for six small catchments in western Brittany (France) *Hydrol. Earth Syst. Sci.* **6** 497–505

Sesini M, Creti A and Massol O 2023 25 (https://chaireeconomieduclimat.org/wp-content/uploads/2023/04/WP-2023-07.pdf)

Strauß C, Herrmann C, Weiser C, Kornatz P, Heiermann M, Aurbacher J, Müller J and Vetter A 2019 Can energy cropping for biogas production diversify crop rotations? findings from a multi-site experiment in germany *BioEnergy Res.* **12** 123–36

Sutton M A, Howard C M, Erisman J W, Billen G, Bleeker A, Grennfelt P, van Grinsven H and Grizzetti B 2011 *The European Nitrogen Assessment* (Camb. Univ. Press) p 664

Svehla P, Vargas Caceres L M, Michal P and Tlustos P 2020 Nitrification of the liquid phase of digestate can help with the reduction of nitrogen losses *Environ. Technol. Innov.* **17** 100514

Styles D, Gibbons J, Williams A P, Dauber J, Stichnothe H, Urban B, Chadwick D R and Jones D L 2015 Consequential life cycle assessment of biogas, biofuel and biomass energy options within an arable crop rotation *GCB Bioenergy* **7** 1305–20

Svoboda N, Taube F, Wienforth B, Kluß C, Kage H and Herrmann A 2013 Nitrogen leaching losses after biogas residue application to maize *Soil Tillage Res.* **130** 69–80

Szerencsits M, Weinberger C, Kuderna M, Feichtinger F, Erhart E and Maier S 2016 Biogas from cover crops and field residues: Effects on soil, water, climate and ecological footprint. *International Journal of Environmental and Ecological Engineering* **9** 413–416

Walker W E, Harremoës P, Rotmans J, van der Sluijs J P, van Asselt M B A, Janssen P and Krayer von Krauss M P 2003 Defining uncertainty: a conceptual basis for uncertainty management in model-based decision support *Integr. Assess.* **4** 5–17

Ward A J, Hobbs P J, Holliman P J and Jones D L 2008 Optimisation of the anaerobic digestion of agricultural resources *Bioresour. Technol.* **99** 7928–40

Westerman P R, Hildebrandt F and Gerowitt B 2012 Weed seed survival following ensiling and mesophilic anaerobic digestion in batch reactors *Weed Res.* **52** 286–95

Wezel A, Bellon S, Doré T, Francis C, Vallod D and David C 2009 Agroecology as a science, a movement and a practice. a review *Agron. Sustain. Devices* **29** 503–15