Environmental accounting of closed-loop maize production scenarios: manure as fertilizer and inclusion of catch crops

Erica Montemayor[a*], August Bonmatí[a], Marta Torrellas[a], Francesc Camps[b], Carlos Ortiz[c], Francesc Domingo[b], Victor Riau[a], Assumpció Antón[a]

[a] Institute of Research and Technology for Food and Agriculture-IRTA, Torre Marimon, 08140, Caldes de Montbui, Barcelona, Spain

[b] Fundació Mas Badia, 17134 - La Tallada d'Empordà, Girona, Spain

[c] Ministry of Agriculture, Livestock, Fisheries and Food, 191 Rovira Roure, E-25198 Lleida, Spain

[*]Corresponding author: Tel.: +34 688 900 043
E-mail address: erica.montemayor@irta.cat

Declarations of interest: none

Abstract

The agri-food sector has moved towards a more linear production economy, partly caused by worldwide food demand. One clear example is the intensification of livestock production, with consequent manure-management and feed-production challenges, the effects of which have led to large environmental problems. Currently, efforts are being made to move the agricultural sector towards closed-loop alternatives. To ensure high environmental performance of these alternatives, realistic quantification of environmental impacts is needed. Thus, using Life Cycle Assessment (LCA) tools, we analyzed the environmental profile of six closed-loop maize scenarios focusing on different combinations of mineral fertilizer, digested organic fertilizer (digestate) from a manure co-digestion biogas plant, and rotation with (or without) catch crops (*CCs*) as a strategy to prevent nitrate leaching to groundwater and as a co-substrate in the biogas plant.

Results demonstrated that replacing a large portion of the mineral fertilizers with digestate could help offset much of the total potential impact of global warming (by 25-35 %), resource depletion (by 94-96%), photochemical ozone formation (by 17-22 %), ozone depletion (by 96-

99%) or even avoid it entirely as in freshwater eutrophication. However, digestate production and application contributed greatly to acidification (51%) and particulate matter (51-52%) categories, with minor differences depending on the species of *CC* used. An optimal combination of both digestate and mineral fertilizers is recommended. The incorporation of *CC*s in a maize rotation can reduce freshwater eutrophication impacts but increase global warming potential. Conclusions were drawn suggesting better management strategies to decrease environmental impacts of maize production.

Keywords: Life Cycle Assessment (LCA), maize, catch crop rotation, manure fertilization, emissions, anaerobic co-digestion

1 Introduction

In Europe and other developed countries, animal production is intensifying due to growing livestock populations, shrinking farm numbers, and consequently higher livestock density. For example, Spain's pig population increased by 14% from 2006 to 2017, while its total number of pig farms declined by 55% from 2005 to 2013 (Eurostat, 2018), concentrating most of the population in seven Spanish provinces – Lleida, Huesca, Zaragoza, Barcelona, Murcia, Segovia and Badajoz. Lleida and Barcelona are located within the larger region of Catalonia, which holds 25% of the total pig population (MAPA, 2016). In 2016, livestock production in Catalonia represented approximately 62% of Catalonia's total agricultural income, of which pigs, poultry, cows and crops represented 37%, 10%, 6%, and 34%, respectively, making livestock the main agronomic activity in the region (MAPA, 2016). Areas with high livestock density produce a large amount of excess livestock manure with high nutrient contents, due not only to farm intensification but also the lack of nearby agricultural land on which to apply it.

Improper management and lack of technological resources to treat manure have led to several environmental problems, including (1) excess nutrients and pathogens in the soil when manure is over-applied or illegally dumped onto cropland as fertilizer (Gagliardi and Karns, 2000); (2) elevated nitrate (NO_3^-) concentrations in local drinking water supplies (ACA, 2016); (3) eutrophication of water bodies that has led to the death of fauna (Camargo and Alonso, 2006); (4) ammonia (NH_3), particulate and odor emissions; (5) emission of greenhouse gases (GHG) such as methane (CH_4) from storage ponds; (6) accumulation of phosphorus (P) and heavy metals (copper, zinc) in soils and (7) leaching of micro-pollutants such as antibiotics from manure-based fertilizers (Thorsten et al., 2003).

These problems represent a clear example of a linear production economy within the agri-food sector. To address the problem of linearity, this study puts into practice the European Commission's circular economy action plan (EC, 2015) by valorizing slurry and crop residues through waste-to-product synergy between livestock farms and crop fields, and complies with the Nitrates Directive (EC, 1991) by using catch crops (*CCs*) to absorb nitrates. This synergy could also help build a collaborative community of farmers and industry professionals and add value to the main crop. In this study, the slurry was valorized by recovering its nutrients in the form of fertilizers and biogas through anaerobic digestion. The feasibility of biogas as an energy source is marked by its manageability, storability, its equivalence to natural gas (if purified to biomethane) and the ability to continuously operate the plant producing it.

To ensure high environmental performance of alternatives, realistic quantification of environmental impacts is needed. Thus, this study performed a life cycle assessment (LCA) with the aim to (i) identify the hotspots in circular maize production using its associated environmental impacts, (ii) compare the environmental performance of different closed-loop maize scenarios to conventional ones, and (iii) suggest improvements that may decrease the environmental impacts. Conventional scenarios were considered as those that use only mineral fertilizers as opposed to digested organic, manure-based fertilizers (digestates). Among other methods (Bockstaller et al., 2009; Lebacq et al., 2013), LCA methodology was adopted for this assessment since it is the most comprehensive approach that uses multi-criteria analysis and a perspective of the entire value chain. It is also a standardized approach that follows ISO standards (ISO-14040, 2006). To our knowledge, this is the first LCA that has analyzed the implementation of *CC* rotation and manure treatment in crop production compared to conventional scenarios, across multiple impact categories. Like the present study, Bacenetti et al. (2016) have compared similar fertilization strategies for maize, but they excluded: *CC* rotations, NH_3 emissions from production and storage of digestate in a biogas plant, electricity and fertilizer credit, and biogas plant infrastructure. They also used background datasets for mineral fertilizer production instead of regionalized datasets, which is used in the present study. Similarly, an LCA of wheat with integrated grass/clover rotation and digestate application was performed (Tidåker et al., 2014), but only analyzed digestate fertilizers and included only a few impact categories.

Maize was chosen as the main crop for this LCA study due to its widespread cultivation in Spain and importance as livestock fodder; maize covers 10% of irrigated land in Spain and contributes 6.5% of total maize production in the European Union (EU) (Eurostat, 2016). This study analyzed six different maize production scenarios with integrated *CC* rotation and different fertilization systems as a strategy for reducing N leaching after fertilizing the main crop, collecting environmental data on emissions, addressing manure management challenges and ultimately closing the energy and nutrient loops in the feed-livestock system. Following the LCA perspective, this study analyzed not only N-derived environmental impacts but also the other midpoint impact categories recommended by the International Reference Life Cycle Data System (ILCD) (EC-JRC, 2010).

2 Goal and scope

2.1 Objectives of the study

The objective of the study was the environmental assessment of closed-loop alternatives for maize production, comparing digested manure to mineral fertilizers and the inclusion of *CC*s in the rotation. The main processes affecting environmental impacts were identified, and several fertilization scenarios were compared to identify those with reduced emissions. The study was developed in the context of the *Futur Agrari* Life project (LIFE+12 ENV/ES/000647; 2013-2018). *Futur Agrari* puts into practice manure management and treatment technologies in areas of Catalonia with a high concentration of livestock farming. Results of the project will help decrease high nutrient contents of agricultural soil while seeking compatibility with farming and forestry development. By evaluating different fertilization strategies and integrated *CC* rotation options, results of this study will help identify the most effective components that reduce excess nutrients in the soils. Outcomes of the study may be useful to support decisions about agricultural practices for farmers, advisors, and stakeholders.

2.2 System description

The system under study was closed-loop livestock production. The closed-loop consisted of activities from manure management at a biogas plant, digestate application to a maize *CC* rotation, use of the maize harvest in livestock feed and use of the *CC* harvest in anaerobic co-digestion.

An environmental assessment was performed using LCA methodology, following ISO 14040 standards (ISO-14040, 2006). LCA goes hand in hand with circular economy, a business model based on restorative use of non-renewable resources and product cycles (Mc Donough and Braungart, 2002).

This study was an attributional LCA that aimed to describe relevant physical input and output flows of the system, more than its use in decision-making. According to the decision-context classification of the ILCD Handbook (EC-JRC, 2010), this study would be considered a C1 situation: description of an existing system, accounting for interactions it has with other systems (i.e. crediting existing avoided burdens from electricity and mineral fertilizers for the scenarios using digestate).

2.3 Functional unit

The functional unit (FU) quantifies the performance of a product system, providing a reference to which the inputs, outputs and results are related. As in most agricultural LCA studies, a mass FU was chosen. Since the main function of the system was the production of maize, 1 t of harvested maize dry matter (DM) was selected as the FU.

2.4 System boundary

The system boundary was considered from raw material extraction to the farm gate. Processes and flows considered in the life cycle inventory (LCI) included inputs and outputs in maize and *CC* production (water, seeds, digestate and mineral fertilizers, herbicides, transport of materials, machinery use, emissions from fertilizer application), as well as inputs and outputs in the processing of livestock manure at the biogas plant (biogas plant structure, transport of materials and co-substrate, electricity consumption and emissions from feedstock storage, liquid fraction (LF) and solid fraction (SF) storage) (Figure 1). All co-products of the biogas plant (electricity, LF and SF) could replace products with the same function or service. In this case, the LF and SF from biogas production were considered as organic fertilizers, thus recovering the nutrients in manure. Part of the electricity generated was used to operate a combined heat and power (CHP) engine, and the surplus was discharged to the national electricity grid (exported electricity). Electricity consumption for plant operations was bought from the electrical grid. Part of the thermal energy produced from biogas combustion was recovered to heat anaerobic reactors, and the rest was dissipated to the air as waste heat. This thermal energy was not a co-product since it was not recovered to be used in other systems outside of the biogas plant. To solve the allocation problem, the substitution method was followed, and the system boundary was expanded to include the average electricity mix from the national grid (replaced by the net electricity exported from the biogas plant) and mineral fertilizer production (replaced by the LF and SF of the digestate).

Material disposal was also included, but not recycling processes, following the cut-off allocation procedure of Ekvall and Tillman (1997). The lifespan of each component was considered in the LCA. Livestock production, as well as commercialization of maize and organic fertilizers from the biogas plant, were excluded from the system since the aim of this study was to identify ways to improve maize production.

The biogas plant stage included the plant structure (materials and transport to the plant), electricity consumption, co-substrate transport to the plant and emissions from the manure storage compartments: influent storage, LF storage, and SF storage. A flow diagram for the production system is shown in Figure 1.

SimaPro software v. 8.5 (PréConsultants, 2017) was used for the LCA, performing only the obligatory phases of classification and characterization and without performing normalization or weighting phases (ISO-14040, 2006).

2.5 Identification of life cycle impact assessment methods to be applied

The indicators and impact categories for the LCA were selected from the ILCD Midpoint+ method (EC-JRC, 2012): global warming potential (GWP, kg CO_2 eq), ozone depletion potential (ODP, kg CFC-11 eq), particulate matter potential (PMP, kg PM 2.5 eq), photochemical ozone formation potential (POP, kg NMVOC eq), air acidification potential (AAP, molc H^+ eq), freshwater eutrophication potential (FEP, kg P eq), marine eutrophication potential (MEP, kg N eq), land use potential (LUP, kg C deficit) and mineral, fossil and renewable resource depletion potential (RDP, kg Sb eq). They were selected because of their relevance to agricultural production and energy-related processes. RDP and GWP are important indicators related to energy consumption. Emissions related to agricultural inputs, such as mineral fertilizers, are major contributors to GWP because of the energy consumed in the manufacturing process. NH_3, NO_3 and nitrous oxide (N_2O) emissions from N fertilizer application are principal contributors to AAP, FEP, MEP, and GWP, respectively. Moreover, NH_3 and CH_4 emissions from manure storage are important contributors to AAP and GWP as well. Emission contribution to POP and PMP may have significant consequences on human health, ecosystems and the crops in question.

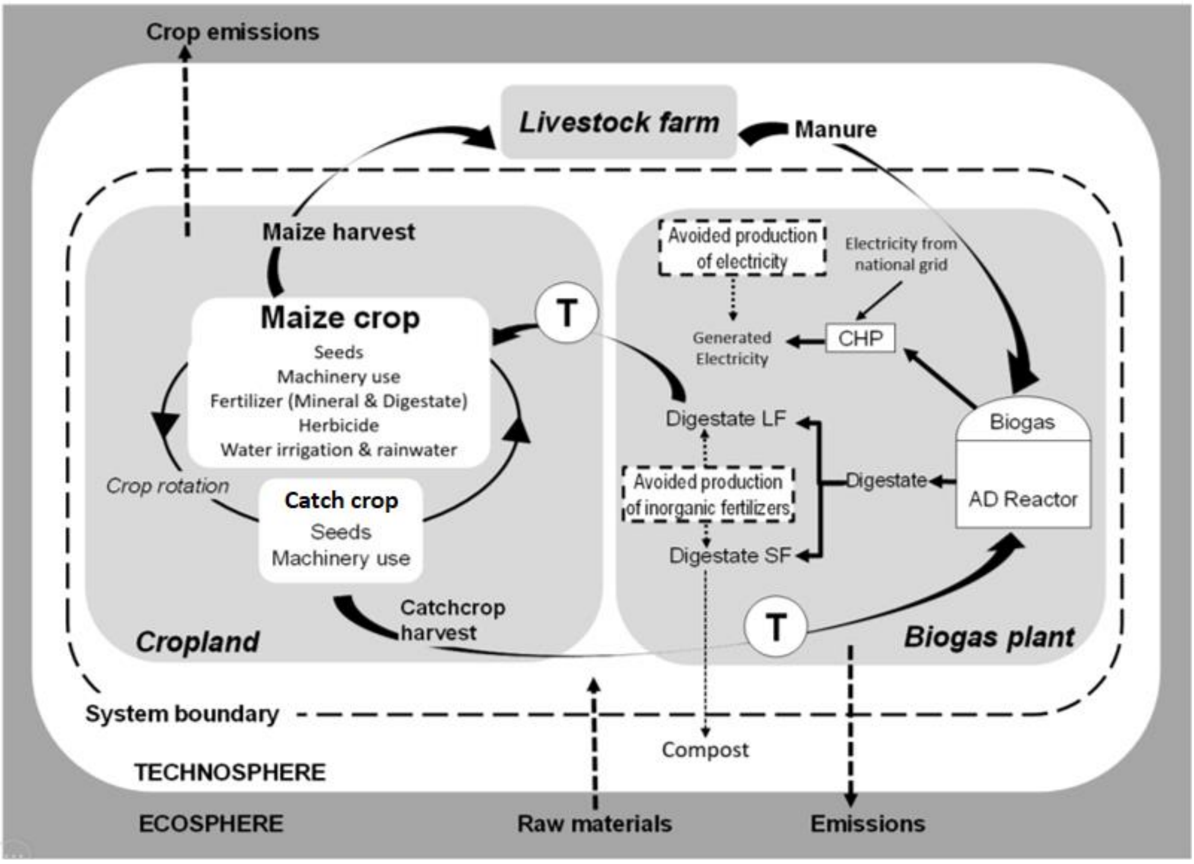

Figure 1. System flow diagram for the production of 1 t of maize. Crop emissions: emissions due to organic and mineral fertilizer application to the maize crop; Seeds: seed production and transport; Machinery use: machinery and diesel production, and emissions from machinery use. T: transport, AD: anaerobic digestion, CHP: combined heat and power engine

2.6 Scenarios

Six scenarios for maize production were considered that differed in two aspects: type of fertilization (digestate fertilization, *DF* vs. mineral fertilization, *MF*) and presence and species of *CC* (black oat (*Avena strigosa*), Italian ryegrass (*Lolium multiflorum*), or forage rapeseed (*Brassica napus*). The six scenarios were as follows: maize with *DF* and black oat, ryegrass or forage rapeseed *CC* rotation (*SC1*, *SC2* or *SC3*, respectively), maize with *DF* but without *CCs*

(*SC4*), maize with *MF* but without *CCs* (*SC5*), and maize with *MF* and black oat *CC* rotation (*SC6*). To compare *CC* scenarios fairly and because most farmers in the area do not use *DF*, *SC6* was considered the baseline scenario. Scenarios SC1-3 (*DF*+*CC*s) were considered as feasible alternatives to reduce NO_3^- leaching. Scenarios SC1-4 and SC6 were performed on an experimental field in Girona, Catalonia, Spain. *SC5,* a virtual scenario, is useful to compare to SC6 since it uses *MF* but without *CC*s to emphasize the possible efficacy of *CC*s, and is also a common practice in the area. *SC5* was assumed to have the same inputs and outputs as *SC6* except for all processes related to *CC*s (including NO_3^- absorption). Table 1 summarizes the main differences among the six treatments, and Table 2 shows their estimated fertilizer application emissions.

Table 1. Mean yields and inputs (2014-2016) for the six fertilization and catch crop scenarios (SC). FM: fresh matter, DM: dry matter, CAN: calcium ammonium nitrate 27%, N/A: not applicable

Parameter	**Unit**	**SC1**	**SC2**	**SC3**	**SC4**	**SC5**	**SC6**
Fertilization	treatment	Digestate[1]	Digestate[1]	Digestate[1]	Digestate[2]	Mineral	Mineral
Catch crop	species	Black oat	Italian ryegrass	Forage rapeseed	none	none	Black oat
Maize yield, FM	t/ha	64.42	62.32	62.49	59.98	66.77	66.77
Maize yield, DM	t/ha	22.49	22.49	21.72	20.75	22.82	22.82
Catch crop yield, FM	t/ha	21.15	28.59	44.44	N/A	N/A	18.03
Catch crop yield, DM	t/ha	5.01	6.14	5.50	N/A	N/A	4.74
Inputs							
Digestate	t/ha	37.50	37.50	37.50	37.50	N/A	N/A
N, digestate	kg N/ha	170	170	170	170	N/A	N/A
N, mineral CAN	kg N/ha	40.5	40.5	40.5	40.5	70	70
N, mineral 15-15-15	kg N/ha	N/A	N/A	N/A	N/A	100	100
N, total	kg N/ha	210.5	210.5	210.5	210.5	170	170
Machinery, fuel consumption	L/ha	170	170	170	117	117	170

[1] Liquid fraction (LF) of digestate from a biogas plant in Vilademuls, Spain, with catch crop co-substrate

[2] LFof digestate from a biogas plant in Vilademuls, Spain, with sewage sludge and industrial food waste co-substrates

Table 2. Estimated emissions from fertilizer application for each scenario (SC). CC: catch crop, N/A: not applicable

Emissions to air	**Unit**	**SC1**	**SC2, SC3**	**SC4**	**SC5**	**SC6**
Fertilization	treatment	Digestate[1]	Digestate[1]	Digestate[2]	Mineral	Mineral
Catch crop	species	Black oat	Italian ryegrass or Forage rapeseed	none	none	Black oat

NH_3	kg/ha	23.62	23.62	23.62	4.49	4.49
NO_x, as NO_2	kg/ha	6.00	6.00	6.00	3.84	3.84
N_2O, direct and indirect	kg/ha	3.54	3.54	3.54	1.81	1.81
PM2.5, from maize crop	kg/ha	1.04	1.04	1.04	1.04	1.04
PM10, from maize crop	kg/ha	20.65	20.65	20.65	20.65	20.65
PM2.5, from catch crop	kg/ha	0.60	0.60	N/A	N/A	0.60
PM10, from catch crop	kg/ha	11.25	11.25	N/A	N/A	11.25
Emissions to groundwater						
NO_3^-	kg/ha	9.57	0.00	139.34	79.69	6.22
Emissions to surface water						
Phosphorus	kg/ha	0.0025	0.0025	0.0025	0.0025	0.0025

[1] Liquid fraction (LF) of digestate from a biogas plant in Vilademuls, Spain, with catch crop co-substrate

[2] LF of digestate from a biogas plant in Vilademuls, Spain, with sewage sludge and industrial food waste co-substrates

2.7 Data collection

Primary data for crop production and management were obtained from Mas Badia Experimental Station in Girona from yearly crop monitoring over the crop rotation period from April 2014 to April 2017, and thus were three-year means. Data referred to crop yield, organic and mineral fertilizer composition and consumption, and water and seed consumption. Nutrient contents of organic fertilizers were obtained yearly by analyzing the LF of digestate samples. Daily climate data were provided by the weather station at the experimental station. Data on the types of machinery used and operation time also came from the experimental station and farmers in the area.

All primary data related to biogas plant structure, management and operation were provided from the biogas plant under study in Vilademuls, Girona, from 2013 to 2016, where monitoring times and emissions from slurry storage were obtained by Torrellas et al. (2018). Lifespans of biogas plant infrastructure components were estimated from the literature (LIFE ES-WAMAR, 2011).

Secondary data were obtained from the ecoinvent database v. 3.5 (Wernet et al., 2016), including those for the manufacture of biogas plant components, tractor, tillage machinery, mineral fertilizers, seeds and herbicide; the electricity production mix and diesel production;

and material transports. The electricity production mix was adapted according to the Spanish electricity mix in 2015 (REE, 2015). The electricity production processes most similar to those in the production system were selected. The mineral fertilizer production mix was also adapted according to the Spanish fertilizer market from 2014 to 2016 (ANFFE, 2017). Data for silage plastic was obtained from literature (Robledo and Martín, 1981). All LCIs for all stages can be found in the Supplementary Material. Data Quality Rating guidelines from the ILCD Handbook (EC-JRC, 2010) were used to rate the quality of the LCI data.

3 Inventory analysis

In the maize production system under study, two main activities were identified: maize *CC* rotation, to produce maize biomass, and the biogas plant, to process cow manure into biogas and digestate to be used as organic fertilizer in maize crop production. In the LCA, flows and processes of the production system were structured into several stages to facilitate compilation of data and interpretation of results.

3.1 Crop rotation

Crop rotation included all activities and inputs to produce maize and *CC* biomass, as well as emissions produced during the cropping period. Maize was the main crop in the rotation, and the *CC* was the secondary crop.

Table 1 shows the main yields and inputs of the six scenarios assessed. Crop techniques were those usually used for maize crops in the area. The soil was first tilled mechanically (Table S5). Subsequent machinery operations for which data were collected include digestate application (SC1 -4), soil preparation for sowing, maize sowing, and application of herbicide and a mineral fertilizer supplement (Table S5). The maize crop was furrow irrigated 5-6 times per season. Maize was sown in April and harvested at the end of summer of the same year; exact dates are shown in the Supplementary Material (Table S1). In scenarios without a *CC*, the soil remained fallow.

Catch crop production was analyzed as a separate stage, which included its own machinery operations, seeds, ensilage, and on-farm transport of fresh *CC* matter a distance of 3 km to the ensilage area and its corresponding particulate matter emissions. The *CC* was sown from September to October, after appropriate soil tillage. No fertilizer, irrigation or herbicide was applied to *CCs*. The *CC* was harvested in March of the following year. After harvest, the *CC* was ensiled for six months and then used as a co-substrate at the biogas plant (Table S6).

3.2 Fertilization treatments

Six fertilization treatments were applied among the scenarios (Table 1). *DF* was applied in *SC1, SC2*, *SC3*, and *SC4*. A mineral fertilizer supplement (calcium ammonium nitrate (CAN) 27%) was also applied during the growing season in Summer to achieve maize production requirements along with the *DF* (Table 1). Calcium ammonium nitrate (27%) and 15-15-15 NPK fertilizer was applied in *SC5* and *SC6* as *MF*.

3.2.1 Digestate fertilization

Digestate was produced using anaerobic digestion and cow slurry from a nearby dairy cow fattening farm and stored as a LF in the biogas plant in Vilademuls, Girona. Digestate fertilizer was transported 2 km from the biogas plant to the maize field by tractor. Digestate fertilizer was applied in March of every year using multiple hoses that hung from the back of a truck a few cm from the ground to reduce NH_3 volatilization, then later incorporated into the soil using a rotary tiller and tractor within 24 h of application. For information about specific characteristics of the digestate, refer to the Supplementary Material (Table S2).

3.2.2 Mineral fertilization

Production of mineral fertilizers was included in the LCA. Datasets for N, P and K mineral fertilizers were regionalized according to Spanish market data from 2014 to 2016 (ANFFE, 2017) using the method described in the French database AGRIBALYSE (Koch and Salou, 2016). The amount of each nutrient was adapted to Spanish market values and as well as the transport of N, P and K fertilizers (LCI Available upon request). It was assumed that all imported mineral fertilizers came from the same sources as those in AGRIBALYSE, via road and ship to Spanish suppliers. Distances were regionalized into t·km.

3.2.3 Fertilizer application emissions

Emissions from fertilizer application were estimated following the methodologies described below, where emission factors and models were adapted to climatic characteristics of the area. Final emission estimates are shown in Table 2. The N emissions considered were NH_3, N_2O, N oxide (NO_x) emissions to air and NO_3^- leaching to groundwater. P emissions due to surface water erosion were also considered. A detailed explanation including equations of all fertilizer emissions can be found in the Supplementary Material.

Total NH_3 emissions equaled the sum of NH_3 emissions produced by mineral fertilizers and NH_3 emissions from digestate fertilizers. The former was estimated as a function of the N content of each mineral fertilizer applied (kg N/ha, Table 1) and the N-NH_3 Tier 2 emission factors per kg N of the mineral fertilizer in basic soil ($pH > 7$) (EEA, 2013). NH_3 emissions from digestate were estimated as a function of the amount of N in digestate applied (kg N/ha, Table 1), the Total Ammonia Nitrogen (TAN) per total kg N in digestate (0.59 kg N-NH_3/kg N total), and the N-NH_3 emission factor for cow-manure digestate (0.194 kg N-NH_3/ kg TAN). This last emission factor depended on weather conditions (mean air temperature 9°C, wind speed <1.6m/s and 51.84 mm rainfall, from November-March 2014 – 2016, primary data), machinery used (hoses) and the time between fertilizer deposition and incorporation (<24 h) (Bittman et al., 2014; Soogard et al. 2002).

NO_x emissions were estimated as a function of the conversion factor from N-NO_2 to NO_2, the N content of the digestate applied, and the recommended NO_x emission factor (EEA, 2013) (regardless of whether the fertilizer was organic or mineral) (Table 1).

Total N_2O emissions equalled the sum of direct and indirect N_2O emissions. Direct N_2O emissions were estimated as 1% of the total N from mineralization, mineral fertilizers, digestate and existing crop residues (0.006 kg N/ha) (IPCC, 2006). Indirect N_2O emissions were estimated as a function of emissions from NH_3, NO_x, and N applied, and fertilizer volatilization factors (IPCC, 2006). Since the crops were located in a semi-arid zone (precipitation is less than half of the evaporation), NO_3^- emissions did not contribute to N_2O emissions (IPCC, 2006).

To capture the influence of application time, crop growth stage and weather conditions on potential NO_3^- leaching, total NO_3^- emissions were estimated during seedling development from March to May, jointing, flowering and filling stages from June to August, and the *CC* or fallow period from September to March. The method used to calculate NO_3^- leaching during these four time intervals can be found in the Supplementary Material.

Due to the basic soil and field slopes of <3%, P leaching and phosphate (PO_4^{3-}) runoff were considered irrelevant. However, P emissions due to surface water erosion were relevant and were calculated as a function of the Universal Soil Loss Equation (Bos et al., 2016), the P content of the surface layer of soil, a correction factor that reflected that eroded soil has more P content than the average (1.86) (Wilke and Schaub, 1996), and another correction factor that depends on the distance to the watercourse. Considering streams as watercourses and a mean distance of <100 m to the watercourse, a default value of 15% was used (Prashun, 2006).

Emissions from fertilizer production, except for those related to toxicity, were also included in the fertilizer stage.

3.3 Water consumption

Water consumed to produce the maize crop was rainwater (1958 m^3/ha/yr) and irrigation water (4072 m^3/ha/yr). *CC* consumed only rainwater (2415 m^3/ha/yr).

3.4 Seeds

The seed stage included the production of seeds for maize and *CC*s, and their road transport to the field by a 16–32 t Euro 5 lorry for an estimated distance of 50 km.

3.5 Machinery operations

Machinery operations for maize cultivation included soil preparation, sowing, fertilizer application, herbicide spraying and harvesting (Supplementary Material, Table S5). Operations for *CC* cultivation included soil preparation, sowing, harvesting, 3 km of transport to the harvest storage area and materials and processes related to silage (Tables S5 and S6).

For each agricultural operation, a dataset was built based on field measurements of working time and a 120 HP tractor with a corresponding operation implement. This type of tractor was used on the experimental field and was recommended by farmers in the area. Diesel consumption was estimated using a Tractor-implement tool in Microsoft Excel® (MAGRAMA, 2008) as a function of the machine power, working time and load weight, all provided by farmers in the area (Table S5). Emissions from fuel combustion are dependent on the amount of diesel consumed (Janulevičius et al., 2013), thus were estimated (shown in Table S4) following methodology by Nemecek and Kagi (2007) - the product of the amount of diesel consumed during the agricultural activity and corresponding emission factors (SAEFL, 2000; Nemecek and Kagi, 2007; Rinaldi & Stadler, 2002). Particulate matter emissions to air from tractor operations were estimated (Table 2) as a product of the cultivated area, the number of times the practice was carried out and Tier 2 soil cultivation emission factors for dry climate conditions (EEA, 2013). The specific amount of machinery (Table S5) required for each agricultural process was calculated by multiplying the weight of the machinery by the operation time divided by the lifetime of the machinery (lifetime from Planas de Martí, 2019.; equation and weight from Nemecek and Koch, 2007).

3.6 Herbicide treatments

CAMIX® herbicide was applied at 3.5 kg/ha, and the production of its active ingredients (40% metolachlor and 4% mesotrione) were considered in the assessment.

3.7 Digestate production

The biogas plant located in Vilademuls, Girona, was built to process manure produced on a dairy farm with a mean processing capacity of 36 000 t/yr of cow slurry and agricultural food waste (co-substrates). Biogas plant data were collected for infrastructure components and materials, feedstock, plant energy consumption, electricity generation, CHP efficiency and plant management and operations (Tables S7-S10). Infrastructure waste was transported by a 16-32 t lorry to the waste management plant for a mean distance of 75 km.

NH_3, N_2O, CH_4, and hydrogen sulfide emissions were measured in the field from manure storage compartments using a Lindvall hood. Samples were analyzed in the laboratory to determine gas emission rates (Torrellas et al., 2018). *CC* from the rotation with maize were the co-substrates considered in this study.

3.8 Digestate transport

As mentioned, digestate was transported 2 km from the biogas plant to the field by a tractor.

4 Results and discussion

The life cycle impact assessment estimated environmental impacts of the six maize scenarios per t of maize DM ("

Table 3). Regarding data quality, the reliability and technological, geographical and temporal representativeness of the data was verified, and the data collected had a Data Quality Rating of 1.5 on a scale from 1 to 5, in which 1 is “very good quality”, and 5 is “very poor quality.”

Table 3. Estimated impacts of scenarios with digestate fertilization (DF) or mineral fertilization (MF) and a catch crop (CC) or no CC per t of maize dry matter. Font formatting identifies the largest (**bold**), second largest (wavy underline) and smallest (straight underline) impact for each impact category. Refer to Section 2.5 for the meaning of impact category acronyms.

Impact category	Unit	***SC1***	***SC2***	***SC3***	***SC4***	***SC5***	***SC6***
GWP	kg CO_2 eq	6.5×10^{1}	7.4×10^{1}	7.5×10^{1}	4.5×10^{1}	9.0×10^{1}	**1.0×10^{2}**
ODP	kg CFC-11 eq	-6.1×10^{-8}	3.6×10^{-7}	1.1×10^{-7}	-4.1×10^{-6}	8.7×10^{-6}	**1.0×10^{-5}**
PMP	kg PM2.5 eq	3.6×10^{-1}	3.6×10^{-1}	**3.7×10^{-1}**	2.8×10^{-1}	1.9×10^{-1}	2.6×10^{-1}
POP	kg NMVOC eq	5.0×10^{-1}	5.3×10^{-1}	5.3×10^{-1}	3.5×10^{-1}	5.3×10^{-1}	**6.4×10^{-1}**
AAP	molc H+ eq	7.4×10^{0}	7.5×10^{0}	7.7×10^{0}	**7.8×10^{0}**	1.3×10^{0}	1.4×10^{0}
FEP	kg P eq	-1.4×10^{-2}	-1.1×10^{-2}	-1.3×10^{-2}	-2.0×10^{-2}	2.5×10^{-2}	**2.7×10^{-2}**
MEP	kg N eq	5.0×10^{-1}	5.0×10^{-1}	4.7×10^{-1}	**1.8×10^{0}**	1.0×10^{0}	3.7×10^{-1}
LUP	kg C deficit	4.9×10^{3}	**6.2×10^{3}**	4.6×10^{3}	4.7×10^{3}	4.4×10^{3}	4.8×10^{3}
RDP	kg Sb eq	3.6×10^{-4}	6.9×10^{-4}	3.5×10^{-4}	-1.3×10^{-3}	9.5×10^{-3}	**1.1×10^{-2}**

Production processes contributed to different degrees to the impacts of each scenario (Figure 2).

The results were not univocal; no scenario outperformed the others in all of the categories analyzed. Scenario *SC6* had the highest impacts per t of maize DM in five of the nine impact categories (GWP, ODP, POP, FEP, and RDP) but low impacts for AAP and MEP (”

Table 3). *SC5* had the second highest impacts in six of the nine categories (GWP, ODP, POP, FEP, MEP, and RDP) but the lowest impacts in three categories (PMP, AAP, and LUP). Scenarios *SC1*, *SC2* and *SC3* had the highest impacts for PMP, and *DF* scenarios generally had the highest impacts for AAP. Scenario *SC4* had the lowest impacts in five of the nine impact categories (GWP, ODP, POP, FEP, and RDP) but the highest impacts for MEP and AAP. Each impact category was then analyzed independently to compare the scenarios and the contributions of their production stages.

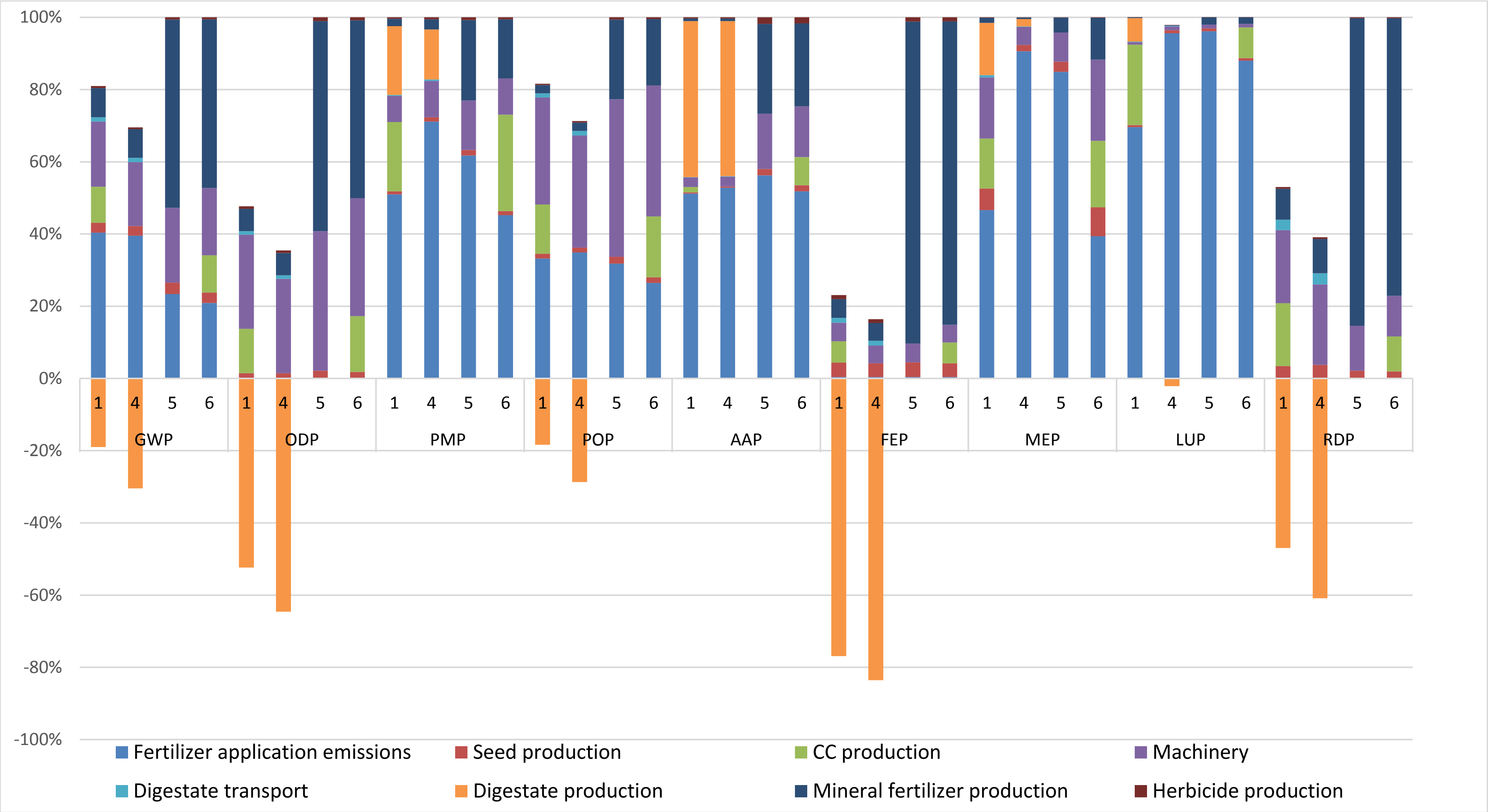

Figure 2. Contribution (%) of production processes to total environmental impacts for scenarios 1, 4, 5 and 6 (the baseline scenario) for Spanish maize production with integrated catch crop (CC) rotation and biogas generation. [Refer to Section 2.5 for the meaning of impact category acronyms]. Due to similarities between contributions in scenarios with digestate fertilization and *CCs*, SC2 and SC3 were omitted.

4.1 Global warming potential (GWP)

Among the *DF* scenarios, *SC4* had the lowest GWP, followed by *SC1*, *SC2*, and *SC3* (4.5, 6.5, 7.4, and 7.5 × 10^1 kg CO_2 eq, respectively). The machinery used in catch crop cultivation in *SC1*, *SC2*, and *SC3* can contribute between 10-12% to the total GWP. Thus, an absence of *CC*s can reduce GWP by this same amount (Figure 2). The main emissions that contributed to machinery use were GHGs such as carbon dioxide (CO_2) and N_2O. The use of more efficient machinery would help reduce GHG emissions.

Digestate production in the biogas plant avoided total GWP impacts by 12 - 30% (Figure 2) across all *DF* scenarios due to avoided use of the electricity mix, showing that biogas plants can help achieve the EU's decarbonization goal. Additionally, and overall, *DF* scenarios *(SC1, 2, 3, 4)* had 35%, 26%, 25% and 55% lower total GWP, respectively, than the baseline *SC6*. These results agree with those of Bacenetti et al. (2016), whose *DF* scenarios had lower GWP than *MF* scenarios.

SC6 had the highest GWP (1.0 × 10^2 CO_2 eq, Table 3), due mainly to mineral fertilizer production (47% of its total GWP), of which N fertilizers contributed the most (36% of its total GWP), compared to PO_4^{3-} (4%) and K (4%) fertilizers.

Other significant contributors to total GWP included N_2O emissions from fertilizer application, especially in *DF* scenarios (40-43%), and CO_2 from the combustion of fossil fuels in agricultural machinery (18-19%) (Figure 2). Recall, however, that we used a fixed emission factor of total N applied to estimate N_2O emissions. The true emission factor depends on the specific type of fertilizer, the soil, and the climate. Nkoa (2014) pointed out that due to a lower content of easily degradable carbon (C) in digested feedstock, N_2O emissions from digested feedstock are generally lower than those from undigested feedstock, which could decrease the value we used.

4.2 Freshwater eutrophication potential (FEP)

Scenario *SC6* had the highest FEP (2.7×10^{-2} kg P eq), followed by *SC5* (2.5×10^{-2} kg P eq) (”

Table 3). This was due to indirect effects of electricity use by machinery in upstream manufacturing processes, specifically, PO_4^{3-} emissions to water from landfill disposal of coal mining waste and the corresponding 17% of coal-generated electricity in the Spanish electricity mix (average from 2014 to 2017; REE, 2017).

Among the *DF* scenarios, *SC4* had the lowest FEP (-2.0×10^{-2} kg P eq), because the absence of *CC* led to less machinery use, and thus lower PO_4^{3-} emissions to water from disposal of coal mining waste. *DF* scenarios had negative FEP values, due mainly to avoided electricity use and mineral fertilizer production (these scenarios generated its own electricity and digestate fertilizer in the biogas plant nearby) (Figure 2). Unlike in the present study, Bacenetti et al. (2016) found that fertilizer application rather than mineral fertilizer production, contributed the most to FEP. This difference is primarily due to the different local conditions (e.g., soil or climate), practices (fertilizer incorporated after three days using a broad sprayer) and dataset processes they used for fertilizers. This highlights the importance of using more appropriate site-specific LCIs for mineral fertilizer production. *DF+CC* scenarios had similar FEP since they had the same life cycle stages and inputs; thus, differences among them were due to differences in *CC* and maize yields.

4.3 Marine eutrophication potential (MEP)

CC scenarios had lower MEP than those that did not use *CC*s. The MEP of *DF+CC* scenarios varied little. Compared to the mean MEP for *DF+CC* scenarios, *SC4* MEP can be reduced by 73% if *CC*s are used. Likewise, between the *MF* scenarios, using *CC*s can reduce MEP by 63% (**Table 3**). Previous studies also conclude that catch or cover crops can reduce eutrophication potential by absorbing eutrophying compounds such as NO_3^- from the soil (Kim and Dale, 2005; Prechsl et al., 2017).

Scenarios without *CC*s (*SC4* and *SC5*) had higher per-hectare NO_3^- leaching (139.34 and 79.69 kg NO_3^-/ha, respectively) than scenarios with *CCs* (9.57, 0, 0 and 6.22 kg NO_3^-/ha for *SC1*, *SC2*, *SC3,* and *SC6*, respectively) (Table 2). NO_3^- leaching is one of the primary concerns of manure application (Nkoa, 2014); hence, using a *CC* can help decrease it.

NO_3^- leaching depends on when, how and how many fertilizers are applied, thus, an optimal combination of organic and mineral fertilizers is recommended and application by injection or hose deposition during growing stages is also advised (*Furtur Agrari*, 2018). For example, *SC4* had higher nitrate emissions to groundwater than *SC5*, showing that the amount of digestate added was too high, contributing to higher MEP impacts. Therefore, a sensitivity analysis was conducted to demonstrate how a reduction in digestate quantity could reduce its corresponding environmental impacts. The analysis indicated that reducing the amount of *DF* in *SC1* by 10% or 20% would reduce MEP by approximately 10% or 14%, respectively (Table 3, Table 4). Only *SC1* was analyzed because *SC2* and *SC3* had similar results and trends, and *SC1* can be compared to the baseline scenario (*SC6*) since both used an oat *CC*. Reduction of up to 20% was chosen since it would provide similar fertilizer application (kg N/ha) as in the conventional scenarios (*SC5* and *SC6*).

That being said, this study brings to light important trade-offs that need to be addressed: *CC*s can reduce MEP, but increase GWP and PMP. These impact trade-offs make it more difficult to decide which environmental impact is more important (e.g., MEP or GWP) and furthermore exacerbate the ability to decide which scenario/practice performs better. One may consider global impacts such as GWP more critical than local impacts such as MEP, as often occurs in LCIA studies, but we wanted to focus this study on local impacts caused by the large amount of excess livestock manure. However, ideally, international weighting consensus factors for impact categories need to be established first in order to avoid value-selection bias.

Table 4. Impacts before and after a 10% or 20% reduction in the amount of digestate fertilizer applied to fields in scenario SC1 per t of maize dry matter. Refer to Section 2.5 for the meaning of impact category acronyms.

Impact category	**Unit**	***No reduction SC1***	***10% DF reduction SC1***	***20% DF reduction SC1***
GWP	kg CO_2 eq	6.5×10^{1}	6.1×10^{1}	5.6×10^{1}
ODP	kg CFC-11 eq	-6.1×10^{-8}	5.1×10^{-8}	5.1×10^{-8}
PMP	kg PM2.5 eq	3.6×10^{-1}	3.5×10^{-1}	3.4×10^{-1}
POP	kg NMVOC eq	5.0×10^{-1}	4.8×10^{-1}	4.5×10^{-1}
AAP	molc H+ eq	7.4×10^{0}	6.7×10^{0}	6.3×10^{0}
FEP	kg P eq	-1.4×10^{-2}	-1.4×10^{-2}	-1.4×10^{-2}
MEP	kg N eq	5.0×10^{-1}	4.5×10^{-1}	4.3×10^{-1}
LUP	kg C deficit	4.9×10^{3}	4.9×10^{3}	4.9×10^{3}
RDP	kg Sb eq	3.6×10^{-4}	4.1×10^{-4}	4.1×10^{-4}

4.4 Mineral, fossil and non-renewable resource depletion potential (RDP)

MF scenarios (*SC5* and *SC6*) had the highest RDP (9.5×10^{-3} and 1.1×10^{-2} kg Sb eq, respectively) (”

Table 3). *SC6* had higher RDP than *SC5* due to greater resource use during *CC* cultivation and silage, where oat production contributed 10% to the total RDP (Table 3).

Use of mineral fertilizers contributed 77% and 85% to the total RDP for *SC6* and *SC5*, respectively. Other contributors in these latter scenarios included machinery use (11-12% of total RDP) and seed production (2%). In contrast, machinery (20%) and *CC* production (17%) contributed the most to the *DF+CC* scenarios (Figure 2). Total RDP can be reduced by 94-96% when some of the mineral fertilizers is replaced by digestate when comparing *DF+CC* scenarios to *SC6* since digestate production would avoid resource use during mineral fertilizer production (Table 3). Previous LCAs of crop rotation and biogas production also showed that replacing mineral fertilizer with digestate and diesel with biogas reduces the use of fossil fuels substantially (Tidåker et al., 2014).

4.5 Air acidification potential (AAP)

DF scenarios clearly had higher AAP (7.4 - 7.8 molc H+ eq) than *MF* scenarios (1.3 - 1.4 molc H+ eq) ("

Table 3). This was due in part to (i) larger amounts of NH_3 emitted per kg of N fertilizer from digestate (0.67 kg NH_3/kg N) than from 15-15-15 and CAN fertilizers (0.037 and 0.022 kg NH_3/kg N, respectively) and (ii) higher NH_3 volatilization during fertilizer production and storage in DF scenarios compared to MF (43-44% and 23-25% of total AAP, respectively). Previous studies also concluded that AAP increases when N mineral fertilizer is replaced by digestate (Björnsson et al., 2013; Lijó et al., 2014). During anaerobic digestion, large fractions of C compounds are converted to CH_4 and CO_2, which are collected as biogas. As a result, the proportion of C in biogas residues decreases while that of N increases in the form of NH_4^+. High pH and NH_4^+ concentrations, in turn, increase NH_3 emissions (Nkoa, 2014).

The substance that was the second greatest contributor to total AAP was nitrogen dioxide (NO_2) (3% in *DF* and 14-15% in *MF*), mainly from machinery. Like for FEP, *DF+CC* scenarios had similar AAP, with small differences due to variability in *CC* and maize yields.

Improving biogas plant management, application techniques and applying an optimal combination of *DF* and *MF* would decrease AAP. The best available techniques to reduce AAP include applying fertilizer near the ground with hoses (as in this study) or through injection, and incorporating the fertilizer within 24 h after application. Since digestate has higher NH_3 emissions per kg N than mineral fertilizers, reducing the amount of *DF* applied by 20% in *SC1* can reduce AAP by 15% compared to that in the baseline scenario (Table 4). Additionally, the volume and nutrient concentration of slurry can be reduced at the origin by adjusting the diets and amount of water given to livestock (*Futur agrari*, 2018). Although *DF* scenarios may have higher AAP, providing cropping systems with N from multiple sources such as manure can help accumulate soil organic C (SOC) (Jensen et al., 2012). However, practical implementation of SOC indicators in LCIA is currently very limited (Oberholzer et al., 2012, Teixeira et al., 2016) and should be included in future LCAs related to organic fertilizers.

4.6 Particulate matter potential (PMP) and photochemical ozone formation potential (POP)

PMP and POP depended strongly on machinery use in all scenarios; thus, *CC* scenarios had higher PMP and POP due to higher PM<2.5µm, PM<10µm, NO_2 and SO_2 emissions from combustion of fossil fuels by machinery. For example, *CC* production contributed ~20% and 27% of total PMP in *DF+CC* scenarios and SC6, respectively, in addition to ~14% and 17% of total POP in *DF+CC* and SC6, respectively (Figure 2).

PMP also depended on fertilizer application (51-71%, from NH_3 and NO_x emissions), LF digestate production (15-19% in *DF* scenarios, from NH_3 and CH_4 emissions), and mineral fertilizer production (16% and 22% in SC6 and *SC5*, respectively; 2-3% in *DF+CC* scenarios) (Figure 2). Thus, *DF* scenarios had higher PMP than *MF* scenarios due to NH_3 and CH_4 emissions from LF digestate storage. If the amount of digestate were reduced by 10% or 20%, total PMP could be reduced by 3% or 6%, respectively (Table 4).

PMP can also be reduced by using or replacing machinery engines with newer, cleaner ones (e.g., stage III or IV engines) or retrofitting machinery with diesel particulate filters. These devices trap particulate matter before it is emitted. They can be fitted to almost any type of diesel vehicle and equipment (Client Earth, 2013).

Contributions of NH_3 and NO_x emissions to PMP and POP can also be reduced by changing the method of fertilizer application. By incorporating the fertilizer within hours of application instead of days, or even better, injecting it into the soil, less NH_3 and NO_x could be emitted to the air (Bacenetti et al., 2016).

Comparing the influence of fertilization treatment on POP, *DF* decreased POP by 22% comparing *SC1* to that of *SC6* due to avoided nitric acid, NH_3, and sulfuric acid production, which are used to produce N mineral fertilizers.

4.7 Ozone depletion potential (ODP)

Ozone depletion potential depended strongly on electricity use and thus its production. Specifically, ODP of *SC2* and *SC3* were 96% and 99% lower than those of *MF* scenarios, respectively, or negative in the case of *SC1* and *SC4* (meaning ODP impacts were avoided), due to the substitution of the electricity mix (Table 3Figure 2). This highlights that environmental impacts of electricity production in Spain can be reduced substantially if renewable energy, such as from biogas plants, were used more often.

4.8 Land use potential (LUP)

Among *DF+CC* scenarios, *SC2* had the highest LUP, followed by *SC1* and *SC6* (6.2, 4.9, 4.8 × 10^3 kg C deficit, respectively). This was due to the influence of combined *CC* and maize DM yields, which was the highest for *SC2* (28.63 kg/ha maize + rye), followed by those of *SC1* (27.50 kg/ha maize + oat) and *SC6* (27.50 kg/ha maize + oat) (Table 1). *SC1* had slightly higher LUP than *SC6*, due to the land transformation required to build the biogas plant in the former, since their crop yields were similar.

LUP can help highlight alternatives that may have lower land-use impacts. However, LUP is site-specific and depends on the geography of each area. The LCIs chosen were based on European averages and not necessarily specific to Spain nor the experimental region of Girona. Thus, the estimates for LUP are even more uncertain than those for other impact categories.

5 Conclusion

This LCA study estimated environmental impacts of six maize production scenarios in Girona, Spain, and suggested better management strategies and alternatives to incorporate closed-loop production, showing the efficacy of LCA as a useful tool to improve agronomic practices from an environmental viewpoint.

The results clearly demonstrated that using more organic digestate fertilizers than mineral fertilizers can help offset a large part of the environmental impacts related to the following impact categories: (i) GWP and FEP, mainly due to generation of renewable energy and avoided use of electricity from fossil energy; and (ii) RDP, POP and ODP, due to replacing mineral fertilizer with digestate, leading to a reduction in impacts of mineral fertilizer production. However, due to its high ammonium content, digestate from cow manure can greatly influence AAP; in particular, storage time and application method are aspects to consider. Thus, to reduce acidification, it is recommended to reduce the amount of digestate by 10-20% and incorporate fertilizers within 24 h of application.

Generating on-site biogas electricity as well as using machinery that is more efficient can reduce GWP. Furthermore, replacing mineral fertilizers with digestate can decrease RDP. Catch crops have also played a vital role in reducing nutrient leaching. Thus, these recommended techniques can shift livestock manure from a problem to an advantage while creating a circular economy with value-added crops.

Declarations of interest

None

Acknowledgments

This work forms a part of the Futur Agrari research project, “Farms for the future: Innovation for sustainable manure management from farm to soil” (LIFE12 ENV/ES/000647), co-funded by the European Commission’s Life^{+} Program. The authors would like to thank the “CERCA Programme/Generalitat de Catalunya” for their support, and the editor, and reviewers for their valuable suggestions to improve this article.

Supplementary Material

Supplementary material related to this article can be found in the online version, at doi: https://doi.org/10.1016/j.resconrec.2019.03.013.

References

ACA, Catalan Water Agency, Technical Report: Evaluation of the Problems Caused by Excess Nitrates of Agricultural Origin in the Bodies of Groundwater in Catalonia. Version 3b, 2016, Catalonia: Department of Water Quality Control.

ANFFE, National Association of Fertilizer Manufacturers: Sector Information http://www.anffe.com, 2017, (Accessed 17 December 2017).

Bacenetti J., Lovarelli D., Fiala M. Mechanisation of organic fertilizer spreading, choice of fertilizer and crop residue management as solutions for maize environmental impact mitigation. *Eur. J. Agron.* **79**, 2016, 107 – 118. https://doi.org/10.1016/j.eja.2016.05.015

Bittman S., Dedina M., Howard C.M., Oenema O., Sutton M.A., (eds). *Options for Ammonia Mitigation: Guidance from the UNECE Task Force on Reactive Nitrogen*, 2014, Edinburgh, UK: Centre for Ecology and Hydrology.

Björnsson L., Prade T., Lantz M., Börjesson P., Svensson S-E., and Eriksson H, Impact of Biogas Energy Crops on Greenhouse Gas Emissions, Soil Organic Matter and Food Crop Production - a Case Study on Farm Level, 2013, The Swedish Knowledge Centre for Renewable Transportation Fuels.

Bockstaller C., Guichard L., Keichinger O., Girardin P., Galan M.B. Galliard G., Comparison of methods to assess the sustainability of agricultural systems. A review. *Agron. Sustain. Dev*. **29** (1), 2009, 223-235.

Bos U., Horn R., Beck T., Lindner J.P., Fischer M., LANCA® - Characterisation Factors for Life Cycle Impact Assessment, Version 2.0. ISBN: 978-3-8396-0953-8, Fraunhofer Verlag, Stuttgart.

Camargo J.A., Alonso A., Ecological and toxicological effects of inorganic nitrogen pollution in aquatic ecosystems: A global assessment. *Environmental International*. **32**, 2006, 831 – 849. doi:10.1016/j.envint.2006.05.002

Client Earth, Reducing Particulate Matter Emissions from Diesel Vehicles and Equipment. https://www.healthyair.org.uk/documents/2013/10/black-carbon-retrofit-guidance.pdf/ 2013 , (Accessed 20 February 2018).

EC (91/676/EEC) of 12 December, Concerning the Protection of Waters Against Pollution Caused by Nitrates from Agricultural Sources. https://eur-lex.europa.eu, 1991, (Accessed 10 January 2018).

EC (COM/2015/0614) of 2 December, Communication from the Commission to the European Parliament, The Council, The European Economic and Social Committee and the Committee of the Regions Closing the loop - An EU action plan for the Circular Economy. https://eur-lex.europa.eu/, 2015, (Accessed 10 January 2018).

EC-JRC, International Reference Life Cycle Data System (ILCD) Handbook. General Guide for Life Cycle Assessment Detailed Guidance, 2010, European Commission. Joint Research Centre. Institute for Environment and Sustainability.

EC-JRC, European Commission, Joint Research Centre, Institute for Environment and Sustainability. Characterisation Factors of the ILCD Recommended Life Cycle Impact Assessment Methods. Database and Supporting Information. First edition, 2012, Publications Office of the European Union; Luxembourg, EUR 25167.

EEA. EMEP/EEA Air Pollutant Emission Inventory Guidebook 2013. Technical Guidance to Prepare National Emission Inventories EEA Technical Report No 12/2013, 2013a, European Environmental Agency.

Ekvall, T., Tillman, A., Open-loop recycling, criteria for allocation procedures. *Int. J. Life Cycle Assess*. **2**, 1997, 155-162.

Eurostat, Agricultural Production – Crops. https://ec.europa.eu/eurostat/statistics-explained/index.php/Agricultural_production_-_crops, 2016, (Accessed 19 March 2018)

Eurostat, Agricultural Production – Livestock and Meat; Farm Structure. http://ec.europa.eu/eurostat/data/database, 2018, (Accessed 20 July 2018)

Futur agrari, Farms for the future. LIFE+12 ENV/ES/000647. Coordinator Government of Catalunya, 2013-2018, Department of Agriculture, Livestock, Fisheries and Food.

Gagliardi J.V., Karns J.S., Leaching of Escherichia coli O157:H7 in diverse soils under various agricultural management practices. *App. Environ. Microbiol*. **66** (3), 2000, 877-883.

IPCC, Guidelines for National Greenhouse Gas Inventories, Volume 4: Agriculture, Forestry and Other Land Use; Volume 2: Energy, 2006, IGES, Kanagawa, Japan.

ISO-14040, Environmental Management, Life Cycle Assessment, Principles and Framework, 2006, International Organisation for Standardisation ISO, Geneva, Switzerland.

Januleviĉius A., Juostas A. and Pupinis G., Tractor's engine performance and emission characteristics in the process of ploughing. *Energy Convers. Manage*. **75**, 2013, 498-508. https://doi.org/10.1016/j.enconman. 2013.06.052

Jensen E.S., Peoples M.B., Boddey R.M., Gresshoff P.M., Hauggaard-Nielsen H., Alves B.J.R., Morrison M.J., Legumes for mitigation of climate change and the provision of feedstock for biofuels and biorefineries. A review. *Agron. Sustain. Dev*. **32**, 2012, 329–364.

Kim S., Dale B.E., Life cycle assessment of various cropping systems utilized for producing biofuels: bioethanol and biodiesel. *Biomass Bioenergy*. **29** (6), 2005, 426–439.

Koch P., Salou T., AGRIBALYSE®: Rapport Methodologique – Version 1.3, 2016, Ed ADEME. Angers. France. 332 p. 262-268.

Lebacq T., Baret P., Stilmant, D. Sustainability indicators for livestock farming. A review. *Agron. Sustain. Dev*. **33** (2), 2013, 311-327.

LIFE ES-WAMAR, Demonstration Project of Swine Waste Management set in Aragón, 2011, LIFE06 ENV/E/000044-ES-WAMAR, Sodemasa ed.

Lijó L., González-García S., Bacenetti J., Fiala M., Feijoo G., Lema J., María T., Moreira M., Life Cycle Assessment of electricity production in Italy from anaerobic co-digestion of pig slurry and energy crops. *Renew. Energy*. **68**, 2014, 625-635, https://doi.org/10.1016/j.renene.2014.03.005.

MAGRAMA, Ministry of Agriculture, Food and Environment. Calculation of the costs of using implements and agricultural machines, 2008, http://www.magrama.gob.es/ (Accessed 20 January 2018)

MAPA (Ministry of Agriculture and Fisheries), Department of Agriculture, Livestock and Fisheries. https://www.mapa.gob.es/es/estadistica/temas/estadisticas-agrarias/economia/cuentas-economicas-agricultura/, 2016, (Accessed 23 January 2018).

Mc Donough W., Braungart M., Cradle to Cradle: Remaking the Way We Make Things. ISBN 0-86547-587-3, 2002, Publisher North Point Press, New York.

Nemecek T., Bengoa X., Lansche J., Mouron P., Rossi, V. and Humbert S., Methodological Guidelines for the Life Cycle Inventory of Agricultural Products. Version 2.0, 2014, World Food LCA Database (WFLDB). Quantis and Agroscope, Lausanne and Zurich, Switzerland.

Nemecek T., and Kägi T, Life cycle Inventories of Swiss and European Agricultural Systems. Final report ecoinvent V2.0 No. 15a, 2007, Agroscope Reckenholz-Taenikon Research Station ART, Swiss Centre for Life Cycle Inventories, Zurich and Dübendorf, CH, retrieved from: www.ecoinvent.ch.

Nkoa R., Agricultural benefits and environmental risks of soil fertilization with anaerobic digestates: a review. *Agron. Sustain. Dev*. **34**, 2014, 473–492.

Oberholzer, Freiermuth Knuchel, Weisskopf, Gaillard. A novel method for soil quality in life cycle assessment using several soil indicators. Agronomy for Sustainable Development, *Springer Verlag/EDPSciences*/INRA, 2012, **32** (3), pp.639-649. <10.1007/s13593-011-0072-7>. <hal-00930549>

Planas de Martí, Calcul dels constons d'utilitzacio de les maquines agricoles, 2019, Department of Agroforestal Engineering, University of Lleida, Spain.

Prashun, V., SALCA-Phosphor. Erfassung der PO4-Austräge für die Ökobilanzierung. Agroscope FAL Zürich-Reckenholz, 2006.

Pré Consultants, SimaPro 8.5.2.0, Amersfoort, The Netherlands, 2017

Prechsl U.E., Wittwer R., Van der Heijden M.G.A., Lüscher G., Jeanneret P., Nemecek T., Assessing the environmental impacts of cropping systems and cover crops: life cycle assessment of FAST, a long-term arable farming field experiment. *Agric. Syst*. **157**, 2017, 39–50.

REE, Red Eléctrica de España. http://www.ree.es/en, 2015, (Accessed 15 January 2018).

REE, Red Eléctrica de España. http://www.ree.es/en, 2017, (Accessed 15 January 2018).

Rinaldi, M., and Stalder, E., Trends in the Exhaust Behavior of Agricultural Tractors - New Significantly Cleaner Models. FAT-report no. 577th, 2002, Swiss Federal Research Station for Agricultural Economics and Engineering (FAT), Taenikon, Ettenhausen, Switzerland.

Robledo de Pedro, F., Vicente, L.M., Application of Plastics in Agriculture. Ediciones Mundi-Prensa, Castelló, 37. Madrid. RAYCAR, S.A. Matilde Hernández, 27. Madrid, 1981.

SAEFL, Handbuch Offroad-Datenbank, 2000, Swiss Agency for the Environment, Forests and Landscape (SAEFL). Bern, Switzerland.

Søgaard, H. T., Sommer, S.G., Hutchings, N.J., Huijsmans, J.F.M., Bussink, D.W., Nicholson, F. Ammonia volatilization from field-applied animal slurry — the ALFAM model. *Atmos. Environ.* **36**, 2002, 3309–3319.

Teixeira, Ricardo & Morais, Tiago & Silva, Cláudia & Domingos, Tiago. (2016). Soil organic carbon depletion as a land use indicator: new modelling strategies. Conference Paper: 10th International Conference on LIfe Cycle Assessment of Food, At Dublin, Ireland.

Thorsten, C., Schneider, R.J., Farber, H.A., Skutlarek, D., Meyer, M.T., Goldbach, H.E. Determination of antibiotic residues in manure, soil, and surface waters. *Acta Hydrochim. Hydrobiol.* 2003, **31**: 36-44. https://doi.org/10.1002/aheh.200390014

Tidåker, P., Sundberg, C., Öborn, I., Kätterer, T., Bergkvist, G. Rotational grass/ clover for biogas integrated with grain production – a life cycle perspective. *Agric. Syst*. **129**, 2014, 133–141. http://dx.doi.org/10.1016/j.agsy.2014.05.015.

Torrellas, M., Burgos, L., Tey, L., Noguerol-Arias, J., Riau, V., Palatsi, J., Anton, A., Flotats, X., Bonmati, A., 2018. Different approaches to assess the environmental performance of a cow manure biogas plant. *Atmos. Environ*. **177** 2018, 203-213. https://doi.org/10.1016/j.atmosenv.2018.01.023

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., Weidema, B.A.D.M.Y. The ecoinvent database version 3 (part I): overview and methodology. *Int. J. Life Cycle Assess*. **21**, 2016, 1218-1230.

Wilke, B., Schaub, D. Phosphate accumulation in soil erosion. *Mitt. German Bodenkundl. Gesellsch.* **79**, 1996, 435-438.