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# Coupling economic and GHG emission accounting models to evaluate the sustainability of biogas policies

Andrea Bartoli[a], Lorie Hamelin[a,b], Stelios Rozakis[a,d,*], Magdalena Borzęcka[a], Miguel Brandão[a,c]

[a] Institute of Soil Science and Plant Cultivation, Department of Bioeconomy and System Analysis, Czartoryskich 8 Str. 24-100 Pulawy, Poland
[b] Department of Engineering of Biological Systems and Processes (LISBP), National Institute of Applied Sciences (INSA), INRA UMR792 and CNRS UMR5504, Federal University of Toulouse, 135 Avenue de Rangueil F-31077 Toulouse, France
[c] KTH Royal Institute of Technology, Industrial Ecology, S-100 44 Stockholm, Sweden
[d] Technical University of Crete, School of Environmental Engineering, 73100 Chania, Greece

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ABSTRACT

The aim of this study is to evaluate and quantify the impacts of different biogas and related policies on the agricultural sector as well as their performance in terms of climate change mitigation and associated costs. To do so we coupled the partial equilibrium approach simulating the market clearing process with the perspective of Life Cycle Assessment of GHG applying it to the well-documented Lombardy case. Results show that the recent Italian biogas policy – prompting manure utilization and reducing the average subsidy per kWh – effectively increased the environmental sustainability of the system, which only now seems able to counteract global warming. Synergies are observed when the recent Common Agricultural Policy greening reform is simultaneously considered by the model.

## 1. Introduction

Sustainable Development Goals comprise 17 objectives adopted by the UN General Assembly in 2015 agreed by the 194 UN Members States aiming at (1) ending poverty and hunger, (2) mitigating the effects of climate change and (3) ensuring prosperous, fulfilling and peaceful lives for all. World Biogas Association estimates that biogas is among the most effective industry in this respect as it can contribute to the achievement of 9 out of 17 SD goals. Recently, the European Commission has moved towards incorporating these objectives into existing EU policies. For instance, in order to fulfill the agricultural SDG number 2 to "End hunger, achieve food security and improved nutrition, and promote sustainable agriculture", the Commission relies on the ongoing revisions of the EU's Common Agricultural Policy and the Fertilisers Regulation. Furthermore, in the Italian context the most important effects would concern: objective 9, namely "Industry, innovation & infrastructure", creating value chains enhancing innovative technological and organizational arrangements; objective 13, namely "Climate action" as it drives to GHG emission reductions when complying to the iLUC norms; avoid deforestation in pursuit of biofuels); and last but not least, objective 15, namely "Life on land" thank to recirculation of nutrients with AD and digestate bio-fertiliser substituting for fossil fertilizers promoting bioeconomy.

Recovery of biogas from agricultural residues, like manure and straw is an acknowledged cost-effective greenhouse gases (GHG) mitigation technology for the agricultural sector, as reflected by the policies established throughout Europe [1]. However, some of these policies, especially the early ones focusing on the biogas output, resulted in the expansion of energy crops dedicated to biogas production [2]. For example, in Italy, the land allocated to maize silage grown for biogas production increased from less than 0.5% of the total utilised agricultural area (UAA) in 2007 to more than 10% at the end of 2012 in the entire country, reaching as much as 18% in the Lombardy region [3]. This, in turn, spurred concerns over the land rent prices and the overall sustainability of biogas production [4], so that the need for a new generation of biogas policies with an increased focus on the sustainability of the feedstock input is apparent.

During the first period of biogas subsidization in Italy (2008–2012), there was no distinction in plant size and feedstock volume used for biogas production: all plants up to 1 MWe rated power were entitled to receive support in the form of a feed-in tariff of 0.28 euro $kW\,h^{-1}$ for 15 years (Table 1). From January 2013, a revised energy policy scheme was introduced, aiming to reduce the level of subsidies for installations built from 2013 onwards by favouring small size plants (lower than 300 kWe; Table 1) along with an enhanced utilization of manure in the feedstock input blend [5]. High manure recovery rate and the reuse of

* Corresponding author at: Technical University of Crete, School of Environmental Engineering, 73100 Chania, Greece.
*E-mail address:* srozakis@isc.tuc.gr (S. Rozakis).

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**Table 1**
Policy changes in the agricultural biogas incentive system in Italy.
Source: Adapted from Chinese et al. [11]

| Policy intervention parameters | *Pre* 2013 policy (Law 99/23 July 2009) | *Post* 2013 policy (Decree 6 July 2012) | | |
|---|---|---|---|---|
| **Incentive value** | Feed in tariff for plants up 999 kWe (280 € $MWh^{-1}$) | **Size class** | **Energy crops** (€ MWh) | **Animal by-products based** (€ MWh) |
| | | 1–300 kWe | 180 | 236 |
| | Green Certificate for plants > 1000 kWe (223 € $MW h^{-1}$ ; average 2011–13) | 301–600 kWe | 160 | 206 |
| | | 601–1000 kWe | 140 | 178 |
| | | 1001–5000 kWe | 104 | 125 |
| **Substrate based tariff differentiation** | None | Different tariffs depend on the ratio between energy crops and by-products (e.g. manure): when lower than 30% the plants receive the incentive for energy crops, otherwise it receives the incentive for by-products. | | |
| **Time horizon** | 15 Years | 20 Years | | |

biogas slurry (digestate) become thereafter key factors to maintain investment profitability of biogas facilities in Italy [6].

A targeted literature review on biogas in Italy reveals that the research interest in this field raised in parallel with the setting of the biogas incentive schemes. A search using as keywords “Ital*”, “biogas”, “policy”, “price*” and “feed-in tariff” resulted in 26 publications after excluding 18 purely technical papers. Only one paper was published before 2012 [7]. This reflects that i) the incentive scheme in force from 2008 (*pre* 2013 policy) was set without specific ex-ante research concerning the rebounded impacts of this policy on land use change (LUC), GHGs savings and feedstock price; ii) academic studies have been conducted after the public debate arose around the market distortions related to the *pre* 2013 policy, in parallel with the introduction of the new policy scheme (*post* 2013 policy). In Table 2, the articles are classified according to the year of publication, case study localization and scale of investigation; type of analysis and methodologies applied are detailed in Table 3.

Various studies have highlighted the market distortions occurring in the Po Valley (Northern Italy) under the initial incentive system, from here onwards referred to as the *pre* 2013 policy scheme. Carrosio [8] argued that under this subsidization policy, only one prevalent model of biogas plant (999 kWe) was encouraged, leading to low efficiency in energy use and doubtful environmental outcomes. Similarly, a positive correlation between farmland rental prices and installed power per hectare is highlighted by Demartini et al. [9], who concluded that the *pre* 2013 policy significantly affected land rental prices in areas where the sector was the most developed. Warnings about the potential effects of *pre* 2013 policies in increasing land rent market are raised also by Bartolini et al. [10] for the Italian province of Pisa. Under the restrictive assumption that no more than 10% of the total agricultural area

**Table 2**
List of articles concerning biogas policy analysis in Italy.
*Source: our elaborations on Scopus database*

| [id] [year] | key words | Case study localization | Scale of investigation |
|---|---|---|---|
| [67] (2017) | Biomass, Citrus pulp, Geographic information system, Sicily, Spatial localization. | Sicily | NUTS 2 |
| [68] (2017) | BeWhere model, Biogas supply chain, Biomethane, CHP, Environmental impact, External costs. | Po Valley | NUTS 3 |
| [69] (2017) | Biogas, methane. | Country level | Raster 250 × 250 m |
| [70] (2017) | Bioenergy, Grounded innovation, Local governance, Rural development, Sustainability. | Po Valley (Emilia-Romagna) | NUTS 3 |
| [71] (2017) | Bio-based economy, EU 2020 targets, Impact assessment, Mathematical programming model, Renewable energy, Sustainability. | Tuscany | Farm level |
| [9] (2017) | Biogas, farmland rental price, farmland value, environmental trilemma, bioenergy, social sustainability, agricultural fixed resource, rent, agricultural land price, land use. | Po Valley (Lombardy) | NUTS 3 |
| [72] (2016) | Biogas, Causal processes, Policies Sustainability Triple bottom line. | Po Valley (Emilia-Romagna) | NUTS 3 |
| [73] (2016) | Biomethane supply chain, Economic optimization, MILP, Tradable certificates. | Po Valley (Emilia-Romagna and Friuli Venezia Giulia) | NUTS 3 |
| [12] (2016) | Biogas, Land use, Market simulation Mathematical Programming, Policy analysis. | Po Valley (Lombardy) | NUTS 3 |
| [74] (2016) | Alternative fuel, biogas, LBG, LNG, renewable fuels. | Country level | NUTS 0 |
| [13] (2016) | Biogas, maize, manure, sorghum, co-digestion, cost analysis. | Po Valley | Farm level |
| [75] (2015) | Airport building, Biomass, Building integrated bioenergy, CHP, Economic sustainability, Zero kilometer energy. | Apulia | NUTS 3 |
| [76] (2015) | Biogas supply chain, Biomethane upgrading, MILP, Spatial explicit optimization. | Po Valley | NUTS 3 |
| [77] (2015) | Biogas, Common agricultural policy, Energy production, Farm household model, Mathematical programming model, Real options, Short rotation coppice. | Tuscany | Farm level |
| [78] (2015) | Anaerobic digestion, climate change, Greenhouse Gases, renewable energy, LCA. | Po Valley (Lombardy and Piedmont) | Farm level |
| [79] (2014) | Biogas, Ecological modernization, Organizational models, Repeasantization, Rural development. | Po Valley (Emilia-Romagna and Lombardy) | Farm level |
| [80] (2014) | Bioenergy supply chain, Biogas, Optimization modelling. | Po Valley (Friuli Venezia Giulia) | NUTS 3 |
| [11] (2014) | Agricultural biogas, Bioenergy support schemes, Supply chain optimization. | Po Valley (Friuli Venezia Giulia) | NUTS 3 |
| [81] (2014) | Biogas cooperatives, Community energy, Institutional forces. | South Tyrol | NUTS 3 |
| [82] (2013) | Agro-energetic system, Biogas, Dairy farm, Economic budget, Life cycle assessment (LCA). | Umbria | Farm level |
| [83] (2013) | Biogas electricity, Dairy farming, Integrated environmental assessment, Milk production, Solar electricity. | Po Valley (Emilia-Romagna) | Farm level |
| [8] (2013) | Agricultural biogas, Neo-institutionalism, Isomorphism. | Country level | NUTS 0 |
| [3] (2013) | Biogas, green energy, energy policy, biofuels, Italy, Po Valley, biomethane, sustainability, energy crops. | Po Valley | NUTS 1 |
| [84] (2013) | Animal waste, Biogas, Corn stalk, Feedstock, Straw. | Po Valley (Friuli Venezia Giulia) | NUTS 5 |
| [85] (2012) | Common agricultural policy, Real options, Renewable energy. | Po Valley (Emilia Romagna) | NUTS 3 |
| [7] (2009) | Animal sewage, Biogas, Energy production, Potential resources. | Country Level | NUTS 0 |

**Table 3**
Contributions assortment for type of analysis and methodology applied.
*Source: our elaborations on Scopus database*

| | | |
|---|---|---|
| **Type of analysis** | Environmental (GHG) | [13,67,68,73,75,76,80,82,83] |
| | Environmental (LUC) | [3,12,71,77,78] |
| | Economics (feasibility) | [11,12,71,73–77,79,82,85] |
| | Economics (commodities prices) | [9,12] |
| | Scenario | [11–13,68,73,75–77,79,85] |
| | Optimization | [11,12,68,69,71,73,76,77,79,85] |
| | Statistical | [3,7–9,69,70,72,84] |
| | Qualitative | [8,70,72,81,85] |
| **Methodology applied** | Spatial Variability | [3,7,9,11,13,67,68,73,74,76,78,79,84] |
| | Geographical Information System (GIS) | [67,69,84] |
| | Life Cycle Assessment (LCA) | [75,80,82,83] |
| | Mathematical Programming (MP) | [11,12,68,69,71,73,76,77,79,85] |
| | General/Partial Equilibrium | [12] |
| | External Costs | [68,76] |
| | Proceeds and costs comparison | [8,9,74,75,82] |
| | Sample Interview | [8,70,72,81,85] |
| | Regression | [9] |

currently used for the production of maize can be allocated for biogas production and using cultivation costs as a proxy for input price, Chinese et al. [11] modelled the effects of the *pre* and *post* 2013 policies on plant installed capacity (kWe), feedstock mix and profitability. Their results confirmed that the *post* 2013 incentive system would likely lead towards smaller plant size, using more manure than maize in the blend and thereby reducing the competition for maize silage (i.e. the pronounced policy objective). In line with these results, Bartoli et al. [12] quantified the impact of *pre* and *post* 2013 policy schemes on land demand and maize silage price in Lombardy. To overcome the limiting assumptions in Chinese et al. [11] on maize silage price and supply, the authors adopted a sector modelling framework, integrating (by means of Mathematical Programming) the supply-side agricultural sector and the demand-side biogas industry. The model implemented by Bartoli et al. [12] delivered the market-clearing prices (i.e. the price cleared at supply-demand equilibrium) and the optimal quantities of maize silage production under the two policy schemes described above. The output of this model thus allows endogenously estimating the land use changes associated to the policy shift.

Agostini et al. [13] estimated the GHG emissions and the economic feasibility of biogas production from different feedstock in the Po valley. They pointed out that the use of energy crops (mono-digestion of maize or sorghum) not only provides no GHGs savings but, under the current policy scheme, it is also not economically feasible. Finally, Patrizio et al. [14] compared the potential impact of alternative biogas policy schemes in terms of external costs related to electricity or biomethane production. Focusing on Northern Italy, they found small differences between biogas and the equivalent fossil-fuel based pathways. In some cases, biogas resulted in even higher external costs, suggesting that, policies exclusively justified on the external cost internalization would not lead to further development of biogas-based technology.

The most common methodology applied in the reviewed papers is based on the spatial analysis of biomass potential suitable for biogas production (50% of the papers) and the economic feasibility of biogas technologies. In most cases, both land use and feedstock prices are exogenous.

Studies applying Mathematical Programming (10 articles, see Table 3) can overcome these limitations. However, only two of them investigated the rebound effects of biogas policies on commodity prices [9,12]. Moreover, only with the methodology proposed in [12] it is possible to forecast the impacts of different policy options on land use change and feedstock prices at the same time. The econometric approach followed by Demartini et al. [9], is in fact classifiable as ex-post analysis. On the contrary the partial equilibrium framework introduced for Italy by Bartoli et al. [12] renders possible to ex ante simulate the interactions between the agricultural and the industrial-biogas sectors, at the same time overcoming the limits related to the spatial variability analyses.

In this study, biogas feedstock demand and supply models have been articulated in a modelling framework that comprehensively estimates the consequences of biogas policies; not only in terms of feedstock used, biogas produced and land needed, but also including the overall effectiveness in reducing GHG emissions. To calibrate and test this integrated model, we use the well-documented case study of the Italian Lombardy region, where two distinct biogas policies have been subsequently implemented in recent years. The model was further elaborated to accommodate for Common Agricultural Policy developments due to the 2013 reform. Building upon recent literature, this study endeavours to couple the Partial Equilibrium approach with a Life Cycle Assessment (LCA) approach to GHG accounting to assess the economic and environmental effectiveness of the whole biogas system in Lombardy. Given the urgency and importance of climate change [15], the environmental performance is here focussed on that single impact only, although the full flow of substances affected has been modelled, allowing for other impacts to be represented in later work (Supporting information).

We first quantify the differential impact of *pre* and *post* 2013 biogas policies combined with previous and current CAP in terms of energy production, feedstock used for biogas, and changes in land use. Environmental impacts are then estimated in response to the changes triggered by policies at the market clearing (equilibrium) point. Based on this, the full flow of substances (carbon, nitrogen, and phosphorus flows, among others) is calculated for all processes induced and avoided as a consequence of the policies.

Regarding the biogas sector the RES policy materializing in Feed-in prices level and subsidies to investment intends to enhance financial sustainability of RES projects compensating for external costs of the fossil fueled competitors whereas the agricultural policy intends to ensure the viability of biomass providers (farms) and their compliance to environmental norms, in other words, to enhance the environmentally sustainable biomass supply. We demonstrate that the combined outcome of these two policies may result in different optimal level of biogas industry deployment and of biomass clearing price and quantity in both regions under examination. This finding is important for practical policy making and it enriches previous analyses on policy consistency regarding the support to renewable energy and its dependence to related policy changes. Synergies and conflicts and the overall policy consistency is reported in the RES literature as a major factor in factual implementation and sustainability of RES carriers [16]. To our knowledge, this is the first study attempting to simultaneously simulate the actual implications of renewable energy and agricultural policies as well as their environmental consequences, in a life cycle perspective where all induced and avoided processes are accounted for.

The structure of the paper is as follows: in Section 2, we provide an overview of the models and the PE approach used to quantify energy production and land use change under pre and post 2013 policies, data, model characterization and LCA approach. The section describes the methodology used to calculate GHG emissions costs. In Section 3, we illustrate equilibrium displacement effects, quantifying the energy production, direct and indirect land use change and the system's efficiency in terms of GHG emission savings. Section 4 summarizes the main findings, providing policy implications and conclusions.

## 2. Materials and methods

### *2.1. Modelling framework for biogas production: feedstock supply and demand*

The market of feedstock for biogas is simulated by means of parametric optimization of mathematical programming models. Farm businesses select activities (crop production and livestock) maximizing their gross income and biogas industry determines purchase price at the farm gate for biomass so that investment payback is maximized after consideration of the most efficient transport scheme. In terms of models, the former generates the supply curve and the latter the demand curve as shown in Fig. 1. Using a iterative algorithm based on market clearing rules detailed in 2.1.3, supply and demand shaped by the partial equilibrium economic model intersect determining the equilibrium, that is the prevailing price in the market and the quantity exchanged as well. As also shown in Fig. 1, supply curve may shift forward or backward affected by policy changes, the same holds for the demand curve that can accommodate changes in renewable energy policies concerning feed-in prices or investment subsidies or both (in this case moved to the left, Fig. 1).

By combining the economic PE model with LCA, we aimed at quantifying the overall effectiveness of the system in reducing GHG emissions and the social cost of alternative policies. The LCA approach is backed by the integrated model determining environmental consequences in response to the changes in the market. Thus, optimal values provided from both scenarios indicating activity output and input levels are injected in the LCA module to calculate environmental impacts (here climate change). The modelling rationale and structure of individual modules is presented in Sections 2.1.1–2.1.3 below whereas mathematical relationships along with symbols used are specified in the Appendix A.

#### *2.1.1. The agricultural model (supply)*

The agricultural sector in the area under investigation is modelled following the approach proposed in [17,18], to build a bottom-up micro-economic model (MAORIE, Modele Agricole de l'Offre Regionale INRA Economie) identifying individual farms to decision making units in order to simulate the consequences of the CAP reform on arable crops in France. One key requisite for the use of this farm-level model is the availability of specific data (which includes crop yield, variable costs, crop mix distribution) for a representative sample of farms. Farmers' choices are modelled in terms of crop mix and land allocation and validated against the base-year observations [19].

The farm model assumes that farms select an activity plan to maximize their gross margin (Eq. (1)). Vector $c^T$ contains the gross margins for one hectare of selected activities and vector x is the selected areas (ha) of activities. Farming business is also subject to certain constraints

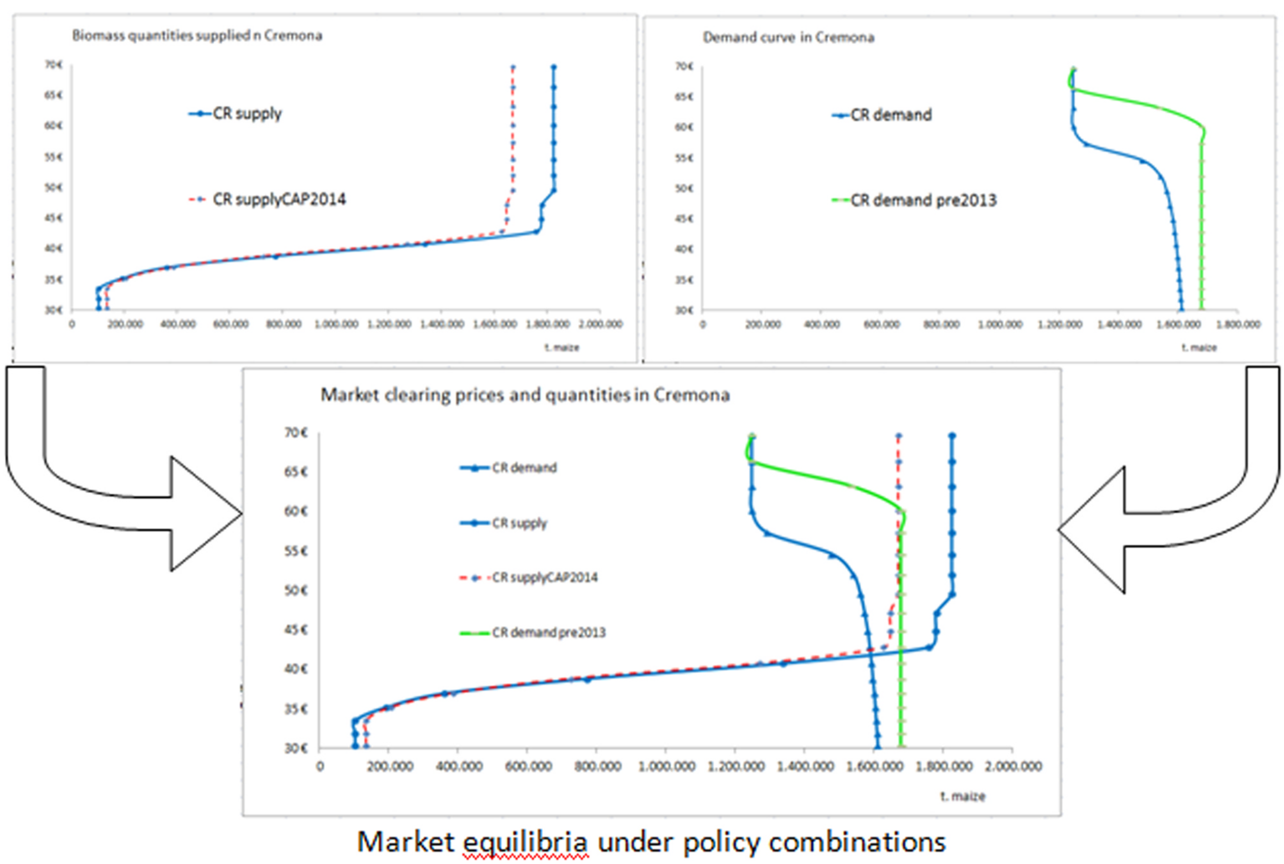


**Fig. 1.** Market clearing process: Demand against Supply of maize silage. Supply curve generated by the agricultural farm based model for different agricultural policy regimes (CAP 2008–CAP 2015) Demand curve generated by biogas conversion and transport model for different Renewable Energy Policy schemes (pre2013 – post 2013).

 

(Eq. (2)) where matrix A contains the resources needed for one unit of an activity and vector b contains the available resource for each constraint. We have modelled the following types of constraints: resource constraints (land, labour, working capital) agronomic, market, and institutional constraints. We provide the detailed algebraic form of the model in the Appendix A (relationships A1–A5). Institutional constraints are adjusted to take into account the reform of CAP 2015–2020 to represent the obligations that farms have to fulfill at the horizon 2020, in order to receive the total amount of the new CAP decoupled payment, i.e. both the 'Basic' and 'Greening' payment (see in Appendix A – relationships A6–A8).

$$\max c^T x \tag{1}$$

$$s.t\, Ax \leq b \tag{2}$$

The output of the agricultural model is the optimal crop mix distribution supplied by farms at each level of predefined vector of exogenous prices. Iterative solution of the model generates the so-called supply response function of biomass, in this case, maize silage for the biogas sector. Alternatively, supply curves can be generated by means of the opportunity cost concept. The model is iteratively solved driven by the increasing quantity of energy crop (*d*) ($q_d^j$) to be produced at each level of price (*j*) triggered by the exogenous demand from the bioenergy system [18]:

$$q_d^j = \sum_{f \in F} \gamma_{d,f} xe_{d,f}^j \tag{3}$$

### 2.1.2. The industrial model: ReSI-M (demand)

To determine the maize demand for biogas production, the ReSI-M (Regionalised Location Information System – Maize) model, developed by Delzeit et al. [20], is applied. This static model simulates, through an iterative maximization of the Return on Investment (ROI), optimal number (n) and type (s) of biogas plants that can be built in each region under investigation (the objective function to maximize is presented in algebraic form in Eq. (A9) in Appendix B). Under a vector of exogenous input prices for maize silage (*w*), the model yields the optimal input demand (*d*) in each region (*c*), as an aggregation of each biogas plant demand:

$$d_c(w) = \sum_s n_{c,s}(w) x_s \tag{4}$$

Where $n_{c,s}$ is the number of plants in region (*c*) and size (*s*) and $x_s$ is the demand of biomass input for each typology of plant assigned to the model. The model specification is detailed in Appendix B.

The biogas input feedstock comprises maize (as energy crop) and manure. Constraint (A10) limits the amount of input maize used to the maximum maize production in the region. Constraints (A11) and (A12) impose equivalence between the amount of feedstock inputs (i.e. maize and manure) transported to and demanded by plants, assuming a maize loss of 8% (in terms of fresh weight) during the transportation process. Regarding losses from silage, we considered that these were negligible, based on the work of Kreuger et al. [21]. Condition (A13) establishes a relation between the input mixture, the biogas produced, and the digestate output. Finally, non-negativity constraints are set in conditions (A14-A16). Considering the distribution and the density of the feedstock in each region under investigation, the objective function (A9) is iteratively maximized, placing the first plant in the region where the lowest transportation costs are found. The ROI for all possible combinations of plant sizes is modelled simultaneously in all regions, and at each model iteration, the biogas plant with the highest ROI is selected. After each iteration, the available feedstock input diminishes and consequently the additional plants must bear higher transportation costs that makes the ROI progressively lower. The iteration process continues until the ROI becomes void or until the input feedstock is exhausted.

### 2.1.3. The biogas partial equilibrium model

Hereby the aforementioned models are articulated, based on Bartoli et al. [11] extending the algorithm by Delzeit et al. [18] by a fully-fledged agricultural module allowing endogenously estimating feedstock used and land use changes related to biogas production. The demand and supply models described above run simultaneously with the same vector of maize silage prices ($p_{maize}$ = {30…70 € $t^{-1}$}). This price range was selected considering the 2013–2014 average maize silage price in Lombardy (approximately 40 € $t^{-1}$).[1] In so doing, on one side, the agricultural model provides the supply curve of maize; on the other side, the maize demand is generated from the industrial-biogas sector. Market clearing prices and quantities are displayed at the equilibrium, when the supply from the agricultural sector (2.1.1) matches the demand from the biogas sector (2.1.2). The outputs of the model at the equilibrium in terms of maize and manure demanded by each class type (size) of biogas plant represent the inputs for the LCA of *pre* and *post* biogas policies applied in Italy. Recent literature that assesses the environmental impact of the introduction of energy crops in the crop mix takes into account the indirect land use change (iLUC [22];). A variety of approaches are proposed, from taking market and policy mechanisms into account to being based on a series of service substitution. Land use changes cannot be a priori considered as simple direct substitution, as a matter of fact they result from changes in crop rotations and overall use of arable land. As land use change is among the most important factors of variability of biofuel impacts, PE models are well-suited to approximate its GHG emissions since crop mix changes are determined at the optimum taking into account competitive crops, cost parameters, and the overall policy context as well [23].

## 2.2. Case study of the Lombardy Region

### 2.2.1. Data and model characterization

Based on data from 2013,[2] one observes, at the end of 2012, 361 biogas plants in the Lombardy region (40% of national total), mainly concentrated in two provinces (from now on called "regions"): Brescia (68 installations) and Cremona (137 installations). Five classes (size) of biogas plants (130, 250, 530, 999 and 2000 kWe), with different energy crop and manure requirements (Table 4), have been then assigned to the industrial-demand model (ReSI-M model described in 2.1.2). To ensure the model tractability, only maize silage is considered as energy crop used to feed the biogas plants of these regions.[3]

The agricultural-supply model (described in 2.1.1) is fed with data input from a sample of ca. 60 farms. These data, i.e. on farm' structure, cost and yield were obtained from the RICA (Rete Italiana di Contabilità Agraria) dataset, the Italian office of FADN (Farm Accountancy Data Network).[4] Yields are provided by crop for sample farms whereas variable costs are aggregate by item and farm. Variable expenses per crop are then estimated for each farm using regional statistics as benchmark to allocate aggregate cost items to individual crops in each farm by means of goal programming. The outputs of the model can be extrapolated to the regional level using weights ($w_f$) also provided by FADN, translating the representativeness of each farm in the whole region. Projecting the outputs of the agricultural model weighted for each simulated farm, maize aggregate supply curves are thus obtained in both regions. In our analysis, the original data correspond to year 2012. Consequently, we apply the above formulation for the year 2012.

[1] Average value obtained from 2014 data of Camere di Commercio, Industria, Artigianato e Agricoltura della Lombardia (Lombardy Chambers of Commerce, Industry, Agriculture and Handicraft).

[2] Eco-biogas project [55].

[3] This assumption is supported by actual data [55] that clearly highlight maize silage as the most representative energy crop used for biogas production in Lombardy (three quarter of the total energy crop mix used in the blend).

[4] A more detailed description of data used and assumptions introduced can be found in Bartoli et al. [12].

**Table 4**
Farm data descriptive statistics based in FADN information.

| | Yield in t/ha | | | | | |
|---|---|---|---|---|---|---|
| | soft wheat | | maize grain | | silage maize | |
| | Cremona | Brescia | Cremona | Brescia | Cremona | Brescia |
| mean | 6,02 | 6,07 | 11,54 | 11,96 | 58,38 | 59,00 |
| max | 7,05 | 7,18 | 13,04 | 13,24 | 66,67 | 66,18 |
| min | 5,02 | 5,00 | 9,23 | 10,75 | 46,15 | 50,00 |
| stdev | 0,59 | 0,46 | 1,01 | 0,57 | 5,28 | 3,93 |
| | variable costs in euro per ha | | | | | |
| | soft wheat | | maize grain | | silage maize | |
| | Cremona | Brescia | Cremona | Brescia | Cremona | Brescia |
| mean | 272,9 | 410,3 | 625,7 | 744,7 | 804,7 | 945,8 |
| stdev | 15,8 | 17,5 | 24,0 | 28,8 | 31,8 | 38,4 |
| | area cultivated by farm in ha | | | | | |
| | soft wheat | | maize grain | | silage maize | |
| | Cremona | Brescia | Cremona | Brescia | Cremona | Brescia |
| mean | 15,63 | 4,64 | 31,08 | 10,88 | 9,00 | 2,86 |
| max | 99,27 | 12,68 | 232,86 | 65,57 | 11,00 | 3,56 |
| min | 2,56 | 1,72 | 2,25 | 1,30 | 8,00 | 1,50 |
| stdev | 28,18 | 3,88 | 52,94 | 12,47 | 1,00 | 1,18 |

Additionally, the land entitlements for each farm are based on the total area cultivated in the year 2012. After calculating the decoupled payment per hectare for each farm, we adjusted accordingly the parameters of decoupled payment per hectare and land entitlements of the agricultural model 2012.

The Italian government has opted for the partial convergence scheme for direct payments between 2015 and 2019. Focusing on arable farming region, the average entitlement value per hectare for the period 2015–19 equals 330 euro/ha. The decoupled payment value of 2015 was decreased by 15% comparing with the previous period, because of the transfer of economic resources to the Second Pillar of CAP. Additionally, each hectare receiving the decoupled area payment in the year 2015 can claim the new CAP land entitlement. If a farm's Initial value of decoupled payment is lower than 90% of average region entitlement value per hectare (330 euro/ha), then this Initial value will rise by 33% of the difference between Initial value and 90% of average entitlement value of the region, reaching at least the 60% of the average region entitlement value per hectare until 2019. In all these cases, the convergence process is linear towards 2020, thus, farms loss or gain a fixed amount each year. Concerning the amount of manure available for biogas production, data are geographically localised at provincial level and have been taken from the Decision Support System ValorE [24] as set out in [12].

This case-study is meant to investigate the effects of the *pre* and *post* 2013 policies on energy production, land use change and climate-change mitigation efficiency. To do so, we consider 2012 as the reference year, testing what would have happened if the incentive system would not have changed (*pre* 2013 policy; here referred to as “Scenario 1”) or if it would have changed as it actually has (*post* 2013 policy; here referred to as “Scenario 2”). The amount of maize silage demanded to feed the plants built prior to January 1st, 2013 is therefore considered as unavailable for the industrial model. Four cases are thus modelled with the PE model, namely Brescia-1, Brescia-2, Cremona-1, Cremona-2, deriving at the optimum the additional number of biogas plants of each size being built, the amount of manure and maize demanded for each, and the changes in land use at the regional level. Subsequently, these elements outcome of the PE model are directed as inputs to feed the environmental assessment model, in order to quantify the climate change impact of each policy. Under Scenario_1 there is no distinction in plant size and feedstock used for all plants up to 1 MWe, that are entitled to receive support accordingly to the feed-in tariff of 0.28 euro $kWh^{-1}$ (*pre* 2013 polices, Table 1). Conversely, under Scenario_2, small plants (lower than 300 kWe) using more than 70% of manure or by-products in the bland are fostered, although the incentives have been reduced for all plant sizes (*post* 2013 policy, Table 1).

### *2.3. Assessing the GHG consequences of biogas policies*

#### *2.3.1. The environmental assessment model*

The environmental assessment is based on a Life Cycle Assessment approach, a recognized approach to quantify the environmental consequences of decisions (investment, policy, etc.) in a holistic perspective [25]. In this study, LCA is applied considering the requirements of the ISO standards [26,27] and strives to reflect all induced and avoided activities reacting to the decision under investigation. All input and output flows were related to the production of 1 kWh of electricity (kWhe) from biogas. The study focuses on the biogas plants built after 2013. Considering that the feed-in tariff of the studied policies will be valid for 20 years (Table 1), the time scope of this LCA is 2013–2033, i.e. the data are chosen in order to be representative for this period. The geographical scope is the Lombardy region, i.e. the inventory data for biomass (e.g. composition, biochemical methane potential) and technology (e.g. efficiencies for the combustion engine, electricity and heat consumed by the anaerobic digestion process itself, etc.) is specific for that region. The impact assessment (i.e. how substance flows are aggregated and related, through a given metric, to an impact category) follows the updated ILCD 2011 Midpoint recommended methods [28] and was facilitated by the LCA software SimaPro v.8.3.0. Background (or generic) LCA data are based on the Ecoinvent v3.3 database [29], while foreground (or system-specific) data are adapted from Italian/Lombardy-specific data from the Italian biogas sector (e.g. [6,12,30–34]), as mentioned above.[5]

#### *2.3.2. System boundary*

The system boundary considered is illustrated in Fig. 2, showing Scenario 1 (pre-2013 policy) in Brescia (Brescia-1). Only processes that would react to a change in demand for biogas-based electricity are included in the LCA. For all scenarios, the system boundary thus starts with dairy slurry ex-housing (i.e. slurry as it leaves the temporal storage in animal houses). This means that any environmental impacts occurring prior to that stage are not taken into account (i.e. those related to animal and milk production), as these prior processes are not affected by an additional demand for biogas-based electricity. Manure is subsequently transported and fed to a biogas plant, along with maize silage. The total amounts of maize and manure that feed each type of biogas plant is a direct outcome of the PE model, as shown in Table 4. The biogas production considered in this study is based on a so-called wet anaerobic digestion process, consisting of a completely stirred main digester whose input is limited under 10% dry matter (DM). It is considered that the production is operated under mesophilic conditions, and that the biogas produced is constituted of 65% methane and 35% carbon dioxide ($CO_2$), with a density of 1.158 kg $Nm^{-3}$ and a lower heating value (LHV) of 22.88 MJ N $m^{-3}$ [35]. Fugitive losses corresponding to 1% of the $CH_4$ content in the biogas produced were assumed. This may be seen as representing “best available plants” rather than “average plants”. The International Energy Agency (IEA) [36] recently reported overall leaks varying from 0.2% to 13.7% of the $CH_4$ produced for upgrading biogas plants (in Germany), while measurements at 10 Danish biogas plants revealed losses varying from 0% to 10% (average 4.3%), which were however reduced to 0.1–4.4% (average 0.8%) after minor repairs [37]. Yet, there are still huge uncertainties related to the measurements of these fugitive losses [36].

[5] Based on an average of Italian data, the composition of the dairy cow slurry as it is immediately prior to its input in the digester is, per tonne of manure: 91.9 kg DM, 72.1 kg VS, 3.8 kg nitrogen (N), 0.73 kg phosphorus (P), 2.92 kg potassium (K), 40.5 kg carbon (C). For maize silage, the composition considered is, per (wet) tonne of maize: 358 kg DM, 315 kg VS, 4.2 kg N, 0.81 kg P, 3.95 kg K, 161 kg C. See Supplementary material for further details.

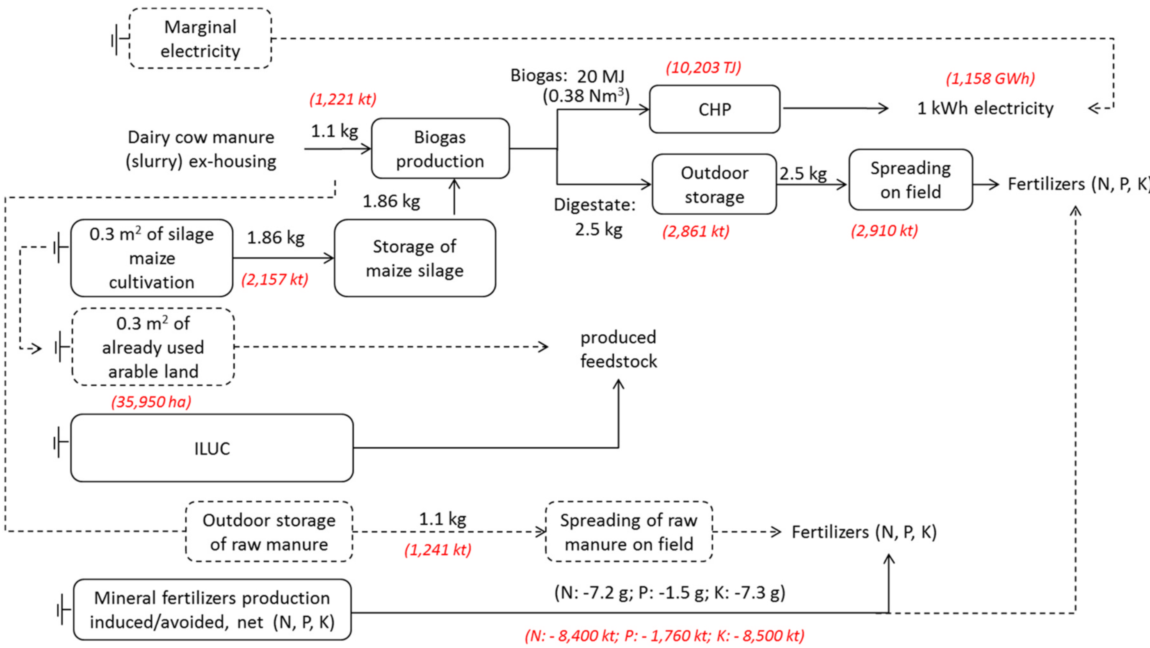


**Fig. 2.** System boundary considered for the LCA model, example for Brescia-1 (pre-2013 policy). Full lines illustrate induced processes while dotted lines show avoided processes. Flows in red are for the whole system (for 1 y), while flows in black are for 1 kWh electricity (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article).
Source: Adapted from Hamelin et al. [35].

This is for example reflected by the extensive campaign on measuring the leakages from the Linköping biogas (upgrading) plant in Sweden [38] where total losses varying between 0.64% and 3% were measured for the same plant (the variation is only due to the team and method to perform the measurements, and to some extent the precise measurement time in the day). For the present case, it should be noted that although "optimistic", this figure applies equally to all cases and therefore does not affect the conclusions to be drawn from the comparisons.

The biogas is considered to be burned in an internal combustion engine with electrical efficiency of 32% and 41% for the 130 $kWh_e$ and 999 $kWh_e$ plants, respectively [30]. The emissions related to this incomplete combustions were estimated based on [39,40] Consistently with the PE model, internal electricity consumption corresponding to 8% of the net electricity production is considered, as well as no heat recovery from the biogas engine [12]. The biochemical methane potential (BMP) of maize and dairy cow manure were considered as 400 $Nm^3$ $CH_4$ $t^{-1}$ volatile solids (VS) and 210 N $m^3CH_4$ $t^{-1}$ VS, respectively, based on the median of Italian data (and a biodegradability of 90% and 45%, respectively). The amount and composition of the produced digestate is calculated from the mass balance (i.e. total input minus gas losses, considering no water losses). The digestate is considered to be stored in a concrete storage tank without any cover (other than the natural crust naturally forming on the surface). Emissions of carbon (methane, carbon dioxide) and nitrogen (ammonia, direct and indirect nitrous oxide as well as nitrogen monoxide) were estimated on the basis of the algorithms presented in [35]. No losses to soil and water were considered from the concrete tank. The digestate is applied on land and used as a fertilizer. Depending on the scenarios, it contains about 30–50 kg of carbon per wet tonne (7–12% DM, and ca. 40% of that is carbon); based on [35], it is here considered that 70% of that carbon ends up emitted as biogenic carbon dioxide ($CO_2$), the rest contributing to the soil carbon pool. A similar procedure was employed in [41–43]. Methane emissions from digestate application were considered negligible (as under aerobic conditions), ammonia emissions were estimated as 16% of the applied N [44], while nitrous oxide ($N_2O$) emissions were estimated on the basis of the algorithms suggested by the IPCC [45]. Losses of nitrogen to water were estimated as 41% of the applied N, and losses of P as 0.03 kg per tonne of applied digestate [46].

As illustrated in Fig. 2, the LCA model reflects the fact that some inputs are constrained (e.g. land for maize), or cannot be produced on demand due (e.g. manure). The use of these constrained inputs involves that another use is prevented; in other words these constrained resources are diverted from their initial use due to the increased demand in biogas-based electricity. For manure, if not used for biogas, it is considered that it would have been conventionally stored and applied on land, without any additional treatment. This reference manure management is thus avoided as manure is diverted to biogas plants. The counterfactual for maize is detailed in Section 2.3.3. Fig. 2 illustrates that the output products of the system (biogas-based electricity and the digestate as a source of organic fertilization) substitute for their marginal equivalents, which are thus avoided. The marginal electricity considered is based on the data from Ecoinvent v3.3 for Italian electricity, while the marginal mineral fertilizers considered are ammonium nitrate, diammonium phosphate and potassium chloride, based on Hamelin et al. [35]. The avoided net mineral nitrogen, phosphorus and potassium are calculated based on the digestate content in N, P, and K minus the raw manure content in N, P, and K. It is acknowledged that this leads to a slight overestimation, since part of the organic nutrients are often applied in excess under European conditions [41]. The $CO_2$ uptake by silage maize has been considered as presented in [43].

### 2.3.3. Land use change

As mentioned above, the basic output of the PE model is the change in land allocation at the farm level, resulting from the implementation of the two biogas policies. We observe that the shift between crops occurs between grain maize (it covers more than 50% of the UAA in the sample) and maize silage, since the latter became more profitable despite similar production costs. In this specific case, the demand for additional silage maize triggered by the biogas policies is thus considered as displacing grain maize only. Furthermore, the cultivation of silage maize was assumed to involve similar flows to and from the environment as the cultivation of grain maize (more irrigation may be applied, but it is assumed negligible in the Po Valley unlike in Southern

Europe, where irrigation savings are important for silage maize due to shorter harvesting period).[6] As a result, the direct land use changes (i.e. emissions from cultivating maize silage minus emissions from cultivating grain maize) are considered to be null. However, iLUC is included, i.e. the decrease in supply of grain maize in the Lombardy region is considered to cause an increase in agricultural prices, which then provides incentives to increase the production elsewhere. The environmental consequences of iLUC have here been modelled based on Schmidt et al. [47], considering an iLUC factor of 1.26 t $CO_2$-eq. $ha^{-1}$ demanded $y^{-1}$ (productivity-weighted hectare). In the model used to derive this generic iLUC factor, the extra crop production is considered to be supplied through both increased yields (intensification) and land conversion to cropland (expansion), using time-series data. A third mechanism, i.e. the decrease in food supply, is not considered in [47], under the premise that the long-term consequences of decision involve full elasticity of supply.

### 2.4. Economic analysis

To evaluate the efficiency of *pre* and *post* 2013 policies in terms of GHG emission reductions and policy cost, the environmental and economic analyses are combined in a four-step procedure. Adopting the methodology outlined in several peer-reviewed articles [13,48–54] we first evaluate the policy mitigation costs borne by the government under both policies investigated. The policy mitigation costs (*PMC)* are estimated as a function of the accounting framework (see Section 3.3) and represent the costs associated to the reduction of a certain amount of GHG emissions through the introduction of an alternative system in comparison to a reference fossil-fuel system [53] per € spent. They feature therefore the increase in cost per unit decrease in GHG emissions, as compared with the fossil fuel reference system [49], and are given by:

$$PMC = \frac{P_b - P_r}{GHG_r - GHG_b} \tag{5}$$

where $P_b$ denotes the feed-in tariff established by the government for biogas production and $P_r$ is the standard market price for electricity mix in Lombardy, our reference system (0.072 € $kWh^{-1}$ [55]).[7] $GHG_r$ *and* $GHG_b$ are, respectively, the GHG emissions resulting from the (Italian) electricity mix (150 g $CO_{2\ eq}$ $MJ^{-1}$ [13],) and from the biogas system. As already pointed out by [48,49,52], if the biogas system presents higher emissions than the reference system ($\Delta GHG < 0$) no GHG mitigation costs can be calculated.

Subsequently, following the Directive 2009/28/EC on the promotion of the use of energy from renewable sources [56], the effectiveness of the biogas system in GHG savings is evaluated relative to its fossil counterpart:

$$SAVING = \frac{GHG_r - GHG_b}{GHG_r} \tag{6}$$

Knowing the total energy production at the equilibrium ($TP_b$, MWh), in the third step we can estimate, in both regions, for both scenarios, the total expense borne by the government ($TP_b$ *$P_b$). Subtracting from this expense the cost corresponding to the same amount of energy at the market price of the reference fossil fuels system ($TP_b$ * $P_r$), we are able to estimate the overall biogas support cost. Subsequently, following the approach proposed by Patrizio et al. [14], we take into account also the External Cost of Carbon (ECC)[8] for the GHGs emitted by the system, considering it as an external cost factor related to the $CO_2$ emission trading due to bioenergy production. Knowing the total emissions the system at the equilibrium ($TE_b$, t $CO_{2eq}$ $y^{-1}$, positive or negative in comparison with the reference counterpart) and the external costs associated with the emissions of each biogas policy pathway (€ t $CO_{2eq}^{-1}$), the total costs *TC* of the two policies can be finally estimated and compared as follows:

$$TC = TP_b(P_b - P_r) + (TE_b{*}ECC) \tag{7}$$

According to Patrizio et al. [14], the ECC value is a function of the year of emission: 26 € $tCO_{2eq}^{-1}$ for 2010–2019, 32 € $tCO_{2eq}^{-1}$ for 2020–2029 and 40 € $tCO_{2eq}^{-1}$ for 2030–2039. As the feed-in tariffs are currently established for twenty years (Table 1), the value of 40 € $tCO_{2eq}^{-1}$ is applied for this calculation.

In the fourth and last step, the total regional costs (*TC*) are divided by the total amount of energy produced $TP_b$. Doing so enables the comparison of the two incentive polices, in terms of additional cost per MWh of electricity produced from biogas relative to the Italian electricity mix (€ MW $h^{-1}$).

## 3. Results and discussion

Scenario 1 and 2 yield, for both regions under investigation, market clearing quantities and prices (Section 2.1.3). In Fig. 3 are reported results obtained under Scenario 1. Supply and demand in Cremona (red line) and Brescia (blue line) are estimated according the current CAP (2014) and are therefore limited by the greening constraints (Section 2.1.1). To do a comparison we reported also the supplies and demands estimated in both region under the previous CAP (2012, grey lines), as presented in [12]. As concern the supply side, we can observe a reduction (shift of the supplies curves towards the *y-axis*) due to the introduction of CAP 2014 that avoid the cultivation of maize silage as monoculture.[9] This is reflected, in turn, also on the demand side, that is limited to the maximum amount of maize silage simulated by the agricultural model. Since under this scenario the model is limited by maize silage unavailability (as we can see up to 55–60 euro per ton the demand is totally inelastic in both regions) also the demand shifted towards the *y-axis*. The market clearing prices are therefore overlapping to those estimated in [12] under the previous CAP despite the quantities of maize demanded at equilibrium are lowered and, therefore, also the number of forecasted plants.

As shown in Table 5, under Scenario 1 the majority of biogas plants simulated by the PE model are those of large size (999 kWe), highlighting these as the most profitable size of biogas plants under the *pre* 2013 policy. This result is in line with the actual trend described in Section 2.2.1 and is confirmed also in the analyses carried out by Carrosio [8] and Demartini et al. [9]. The difference in terms of number of plants between Brescia and Cremona for this scenario is related to two factors: first, there is a large regional difference in the amount of maize unavailable (i.e. already used to feed the biogas plants built before 2013: 68 already existing in Brescia and 137 in Cremona). As almost saturated by already built biogas plants, the prospects for expansion of the biogas sector in Cremona are therefore limited.[10] Second,

[6] Donati et al. [86], focusing on the Italian province of Parma (Po Valley), investigated water demand for energy corps through a PMP (Positive Mathematical Programming) model. Authors underlined the importance to prevent a non-efficient water allocation for energy crops cultivation in areas jeopardized by the risk of drought, even if for the Po Valley context the availability of water is relatively high and also in this case is not considered as a constraint for the model.

[7] With reference to the Northern Italy area the average annual zonal price recorded during 2011 and 2012 was respectively equal to 0.070 and 0.074 € $kWh^{-1}$, for an average value of 0.072 € $kWh^{-1}$.

[8] The External Cost of Carbon is a monetary indicator of the global damage caused by one additional tonne of carbon emitted into the atmosphere in relation to the net present value of climate change impacts over the next years.

[9] This effect is displayed also in the first part of the supply curve in Brescia: under CAP 2012 there is no supply of maize silage since the model is allowed to simulate other crops more profitable for this level of prices. The greening constraints introduced with CAP 2014 avoid monoculture and therefore maize silage is simulated since is some farms is the second most profitable corp.

[10] To feed these plants it is estimated that 529,952 t $y^{-1}$ of maize silage were

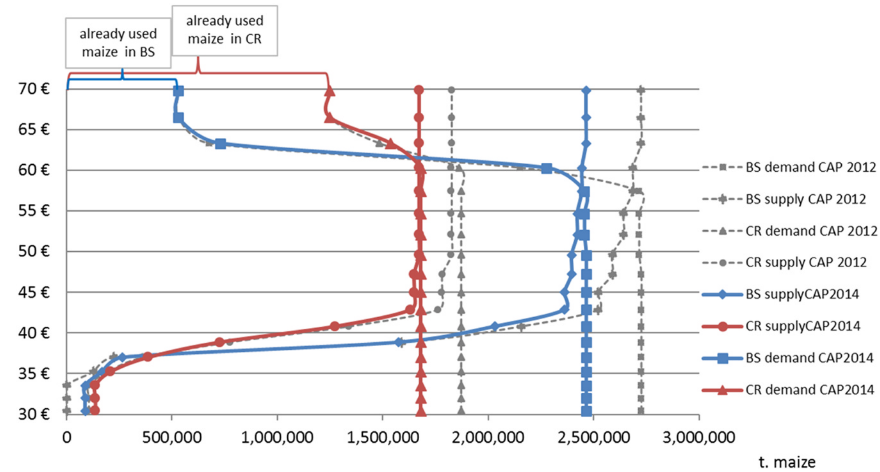


**Fig. 3.** Scenario 1: Market clearing prices and quantities in Brescia (BS) and Cremona (CR).
Source: Authors elaborations on the results of the PE model described in Section 2.1.3

**Table 5**
Maize and manure requirements for the classes of plants identified (obs: year: 2012).
Source: Adapted from Bartoli et al. [12]

| **Power** (kWe) | **130** | **250** | **530** | **999** | **2000** |
|---|---|---|---|---|---|
| **Maize Silage** (t ww $y^{-1}$) | 1000 | 4000 | 10,000 | 18,000 | 33,000 |
| **Manure** (t ww $y^{-1}$) | 10,000 | 12,000 | 13,000 | 9000 | 24,000 |
| **Digestate** (t ww $y^{-1}$) | 10,680 | 18,162 | 17,621 | 29,708 | 44,760 |

the maximum supply of maize silage simulated by the agricultural model under the CAP 2014 constraints is considerably higher in Brescia (2464,425 t) than in Cremona (1680,149 t). This reflects that under a policy not restricting energy crops and favouring a relatively high electrical output, more land is made available for (new) maize silage cultivation in areas where fewer biogas plants are already competing for the resource (ca. 32,000 ha of land is being turned to maize silage in Brescia in comparison to less than 7000 ha in Cremona, see Table 5).

In line with the analysis carried out by Chinese et al. [11], the results obtained under Scenario 2 confirmed the effectiveness of the *post* 2013 policy: selected plants are smaller (being essentially only 130 kWe plants) and, both in Brescia and Cremona, the amount of manure used for biogas production strongly increases in comparison with Scenario 1 (Table 5). A decrease in maize use (and thus land converted to maize silage) is also observed in both regions (Fig. 4). This result can be in large part explained by the lower quantity of maize silage needed for small plants to operate (1000 t $y^{-1}$ rather than 18,000 t $y^{-1}$ for 999 kWe; see Table 4). Consequently, under Scenario 2, the increase of plants is not limited by maize availability, but by the loss of profitability due to the reduction of incentives. This is clearly observable in Fig. 4, where market clearing prices and quantities obtained under Scenario 2 are reported. Since the number of simulated plants is not affected by maize availability, the demand curves are overlapped to those estimated under the CAP 2012. However, before the profitability of biogas plants reaches a level close to zero (last simulated plant before the model stops the iteration process), a considerable number of 130 kWe plants are forecasted in both regions (327 in Brescia and 341 in Cremona; Table 5). This figure is in line with the results carried out by Ragazzoni [57] that quantified, for the whole of Lombardy, the possibility to build 1052 small plants (99 kWe) fed exclusively by manure.

The overall amount of biogas produced is highly influenced by the type of biogas policy applied in Brescia, where a 3-fold decrease is observed for Scenario 2, in comparison to Scenario 1. Interestingly, the opposite is observed in Cremona, where a 23% increase in biogas produced is observed in Scenario 2 (in comparison to Scenario 1; Table 5). This, again, reflects that the amount of maize used as feedstock to biogas plants was highly reacting under Scenario 1 in Brescia, but not in Cremona. Given that much more biogas can be produced out of maize than out of manure [35], the difference in overall maize used is proportional to the difference in biogas produced. In Cremona, however, the overall amount of electricity produced is lower in Scenario 2 in spite of the increase in biogas produced. The non-proportional relation between biogas and electricity production reflects the shift towards smaller (and lower electrical efficiency) plants that Scenario 2 triggers. Nevertheless, we can observe that in areas in which the competition for land between biogas plants and other agri-food supply chains is remarkable and the possibilities of expansions for the biogas sector are therefore limited, the introduction of the new policy scheme not only lowered the demand of land for maize silage, but also allows more electricity production.

The different maize/manure ratio observed for the plants simulated under Scenario 2 is of paramount importance also for the resulting GHG emissions. Fig. 5 shows the LCA results for all policy scenarios and regions, both per $kWh_e$ produced (a) and for the region over a year (b). The net impact for a given alternative is obtained by subtracting the avoided impacts (i.e. the negative values in Fig. 5) from the induced impacts (positive values). The negative values can be seen as the impact of the counterfactual, i.e. of not producing biogas. Fig. 5a shows the emission intensity for producing biogas-based electricity (kg $CO_{2eq}$ per kWh of electricity produced) under the different policies and regions. If under Scenario 1 this ratio was between null and -0.1 for both regions under pre and post 2013 CAP (i.e. the overall impact of the system was in practical terms the same for climate change than not producing biogas at all), under Scenario 2 values are clearly negative ($-0.7$ to $-0.8$ kg $CO_{2eq}$ kW $h^{-1}$). Consequently, only under this policy scheme (*post* 2013 policy) the system can be considered effective in GHGs reduction. This is more evident when we consider the overall emission of the system ($tCO_{2eq}$ $y^{-1}$) at the equilibrium: in Fig. 5b, it is clear that the climate change impact resulting from a lower amount of biogas-electricity (purple bars) with the policy of Scenario 2 is compensated by the fact that more manure is diverted to the digesters (pink and grey bars). This reflects the environmental benefit associated with using manure for biogas instead of managing it conventionally, where its $CH_4$ is released to the atmosphere instead of being captured and used to produce renewable electricity. Considering that $CH_4$ has a global warming potential ca. 30 times higher than that of $CO_2$ ([58]; 100 y horizon),

(*footnote continued*)
used in Brescia and 1248,266 t $y^{-1}$ in Cremona [12].

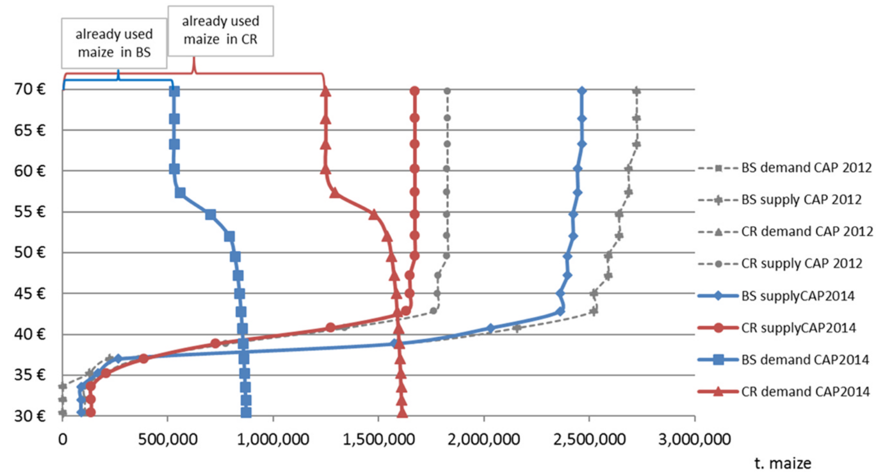


**Fig. 4.** Scenario 2: Market clearing prices and quantities in Brescia (BS) and Cremona (CR).
Source: Authors elaborations on the results of the PE model described in Section 2.1.3

avoiding these releases has an important impact. Similarly, digesting manure instead of managing it conventionally leads to an overall decrease in nitrous oxide [35], a GHG with a global warming impact ca. 300 times that of $CO_2$ ([58]; 100 y horizon). These results are consistent with Agostini et al. [59] who investigated the environmental impacts of three biogas systems based on manure, maize and sorghum in the Po Valley area. In this study the use of manure as main feedstock during anaerobic digestion is indicated as main driver in GHG savings, thanks to the avoided emissions from the traditional storage and management of slurry as organic fertilizer. Regarding the utilization of energy crops as feedstock, the authors pointed out that only a limited fraction should be employed during the anaerobic digestion, in order to avoid losing the benefits of manure digestion [59]. Similar findings have been obtained also in previous studies (e.g [2,35].), where it was shown that producing biogas with maize as a key co-feedstock to manure is worse for climate change than not producing biogas at all. Regarding maize as a biogas feedstock, emissions released during the silage process, albeit representing less than 12% of overall environmental impact related to silage maize production [60] are not accounted for in this exercise. However, this simplification is applied equally to both biogas policy (and regions). Adding the silage emissions would affect the policy scenario 1 (pre-2013) more than the scenario 2 (as the results show much more silage maize in pre-2013 context), and thus would make the GHG gains (from policy post-2013) even greater.

Although ILUC appears negligible under both scenarios (less than 2% of the GHG gross emissions, in average around 1%), it must be highlighted that the absolute magnitude of its climate change impact is highly dependent upon the methodology used to calculate the GHG releases associated with the additional demand for arable land. A modest iLUC factor was used (1.26 t $CO_{2\ eq}$ pw-ha$^{-1}$ y$^{-1}$) as mentioned in Section 2.3.3. then to observe the sensitivity of the outcome it was increased at 4.1 t $CO_{2\ eq}$ pw-ha$^{-1}$ y$^{-1}$suggested by Tonini et al. [61] still resulting in marginal contribution of iLuc to the GHG emissions. Fig. 5 includes all factors with iLUC set at 4.1 t $CO_{2\ eq}$ pw-ha$^{-1}$ y$^{-1}$ illustrating the important contribution to global warming from digestate (and avoided raw manure) application; this reflects that essentially all the

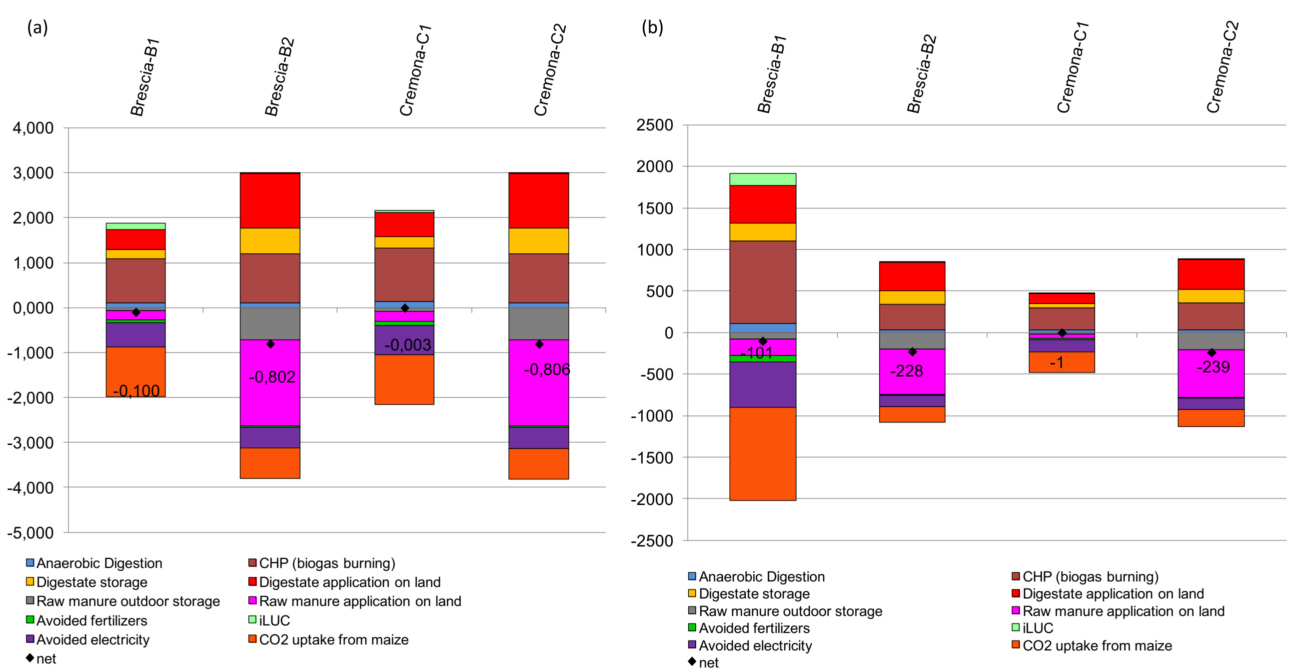


**Fig. 5.** LCA results for global warming (100 y horizon kg $CO_2$.eq$^{-1}$) per functional unit (1 kWh el) (a) and for the whole region for pre-2013 CAP policy kt$CO_2$.eq$^{-1}$ over a year (b).

**Table 6**
Key effects resulting from the biogas policy change in the studied regions, over 1 year[a].

| | Brescia - 1 | Brescia - 2 | ΔBS 1–2 | Cremona - 1 | Cremona - 2 | ΔCR 1–2 |
|---|---|---|---|---|---|---|
| **Simulated new plants**[b] | {15 × 130 kWe; 119 × 999 kWe } 106 × 999 kWe | 327 × 130 kWe | {+ 312 × 130kWe; - 119 × 999kWe} + 327 × 130 kWe; - 106 × 999 kWe | {1 × 130 kWe; 32 x 999 kWe} 23 x 999 kWe | 341 x 130 kWe | {+340 x 130 kWe − 32 x 999 kWe } + 341 x 130 kWe −23 x 999 kWe |
| **Maize-to-biogas** (kt ww $y^{-1}$) | {2157} 1980 | 327 | {− 1830} − 1653 | {577} 414 | 341 | {− 236} − 73 |
| **Manure-to-biogas** (kt ww $y^{-1}$) | {1221} 954 | 3270 | {+ 2049} + 2316 | {298} 207 | 3410 | {+ 3112} + 3203 |
| **Biogas produced** (TJ $y^{-1}$) | {10,203} 8958 | 3191 | {− 7012} − 5767 | {2714} 1944 | 3328 | {+ 614} + 1384 |
| **TPb Electricity** (GWh el $y^{-1}$)[c] | {1158} 1020 | 284 | {− 874} − 736 | {309} 221 | 296 | {− 13} + 75 |
| **Land used** (ha) | {35,950} 31,800 | 5450 | {− 30,500} − 26,350 | {9617} 6900 | 5683 | {− 3934} − 1217 |
| **TEb Total LCA GHG** (kt $CO_2$eq $y^{-1}$)[d] | {− 101} − 27 | − 228 | {− 127} − 201 | {− 1} − 62 | − 239 | {− 238} − 23 |

[a] For the whole region, and related to the flows presented in Fig. 5; −1 refers to the pre-2013 policy, while -2 refers to the post-2013 policy (detailed in Table 1).
[b] Results of the partial equilibrium model.
[c] $TP_b$ in (19).
[d] $TE_b$ in (19); an horizon time of 100 y is considered for the GHG metric.

carbon in the digestate/raw manure was considered to be emitted as $CO_2$ in the field (See Supplementary material), and that biogenic $CO_2$ releases were accounted for.

To evaluate the GHG emission-saving efficiency and the costs of *pre* and *post* 2013 policies, we investigated the performance of the system in economic terms. Included in the calculations are the avoided electricity emissions from the Italian electricity grid mix, as is done in LCA (Table 6). From these emissions, we then estimated the policy mitigation cost of the system by applying Eq. (5) (Section 2.4).

According to the literature, biogas technology offers a broad range of potential $CO_2$ mitigation costs, between 95 and 1135 € $tCO_2.eq^{-1}$ [51]. Policy mitigation costs for biogas production in Lombardy are reported in Table 6 so that we observe GHG emission savings in comparison to the reference system. Under Scenario 1, the biogas system presents similar emissions as the reference system and GHG mitigation costs are calculated at values more than double comparing with scenario 2 (biogas post 2013).[11] Arguing in terms of economic surplus,[12] this means that the real objective underlying the pre 2013 policies is not to mitigate GHG but to support agro-energy activity, boosting, therefore, farmer's income.

Mitigation costs associated to the *post* 2013 policies are approximately estimated at 122 € $tCO_{2eq}^{-1}$ in both regions at the low range of literature reported costs [51].

By applying Eq. (6), we evaluated the effectiveness of the biogas system in terms of GHG savings. We observe negligible or even negative savings under Scenario 1 in contrast to Scenario 2 when the biogas system presents a case of GHG mitigation, reaching increases up to 160% in comparison with the reference system (Table 6). Thus, the system can be considered efficient following the target provided by the Directive 2009/28/EC on the promotion of the use of energy from renewable sources [56], savings should be superior to 50% (and even 60% from 2017).

The overall biogas support costs are quantified following Eq. (7). Further including the External Cost of Carbon (positive under Scenario 1 and negative under Scenario 2), the total cost of the *pre* 2013 policy system amounts at 237 M€ $y^{-1}$ in Brescia and 64 M€ $y^{-}$ in Cremona (Table 6). With the introduction of the *post* 2013 policies, we forecast a significantly lower social cost in both regions: 37.5 M€ $y^{-1}$ in Brescia (-200 M€ in comparison with Scenario 1) and 39 M€ $y^{-1}$ in Cremona (-25 M€ in comparison with scenario 1; Table 6). This is due to the average subsidy reduction per kWh of electricity produced (that, in turn, limits the number forecasted of plants) and to the large increase of manure utilization (that, in turn, leads to negative emissions in comparison with the reference system and therefore to a negative social cost of carbon). Finally, the unit costs (€ $MWh^{-1}$) of *pre* and *post* 2013 policy are calculated dividing the overall biogas support cost by the total energy produced in each region under investigation. In so doing, we are able to compare the two incentive polices in terms of additional cost per MWh of electricity produced from biogas instead of from the fossil fuel reference system. The reduction obtained due to the new policy is noticeable in both regions, reaching savings close to 65–75 € MW $h^{-1}$ in both regions depending on CAP scenario (Table 6). This highlighted once again the key role of manure in improving the economic and environmental efficiency of the system (Table 7).

[11] This observation is reported also by Agostini et al. [13] for the biogas plants built in northern Italy fed exclusively with energy crops.

[12] The sum of gross margin generated from the agricultural sector and the profits earned by the industrial-biogas sector can be represented in terms of total agents' surplus. The higher withdrawal price paid for biogas in comparison with the reference system (Italian electricity mix) reflects the higher price paid by the consumer for biogas-based electricity. The loss for the economy caused by the introduction of biogas incentive scheme is therefore represented by the difference between the total budgetary expenses and agents' surplus (agricultural + biogas sectors).

**Table 7**
GHGs from biogas production and costs of the policies, a comparison over 1 year[a].

| | Brescia - 1 | Brescia - 2 | ΔBS 1–2 | Cremona - 1 | Cremona - 2 | ΔCR 1–2 |
|---|---|---|---|---|---|---|
| **Biogas emissions** (kg $CO_{2eq}$ $kWh^{-1}$)[b] | {− 0.10} − 0.03 | {− 0.80} − 0.80 | {− 0.70} − 0.78 | {0} − 0.01 | {− 0.81} − 0.81 | {− 0.80} − 0.80 |
| **Policy Mitigation Costs** (€ t $CO_{2eq}{}^{-1}$) | {256} 290 | {122} 122 | {− 134.2} − 167.4 | {302} 300 | {121.8} 121.8 | {180} − 178.1 |
| **GHGs saving (%)**[c] | {18.7} 5.1 | {+ 160.7} + 160.7 | {142} 155.6 | {− 22.2} 21.5 | {161.5} + 161.5 | {183.65} 183 |
| **Policy Cost** (Mln € $y^{-1}$) | {236.8} 211.1 | {37.5} 37.5 | {− 199.3} − 173.6 | {64.2} 43.5 | {39} 39 | {− 25.2} − 4.5 |
| **Unit cost** (€ $MWh^{-1}$) | {204.5} 207 | {132} 132 | {− 72.6} − 75 | {208} 196.7 | {131.76} 131.76 | {− 76.1} − 65 |

[a] -1 refers to the pre-2013 policy, while -2 refers to the post-2013 policy.
[b] excluding the avoided electricity emissions.
[c] evaluated relating the system's GHGs with the national fossil fuel comparator emissions.

## 4. Conclusion

By combining economic PE modelling with an LCA approach for GHG accounting, we quantified the direct and indirect consequences of biogas policies; not only in terms of the feedstock used, biogas produced and land required, but also in terms of the overall effectiveness in reducing GHG emissions and the related social cost of alternative policies. Our comparison of the two biogas policies in place in Italy over recent years made evident that the new policy scheme introduced in 2013 strongly decreased the GHG emissions per kWh of biogas-electricity produced besides mitigating climate change (i.e. overall reduction in GHG emissions). These findings are consistent with those reported in several studies, which also indicate that policies endeavouring to ensure that biogas-based energy does mitigates climate change typically results in promoting manure utilization for biogas production while limiting the use of land-dependent biomass. However, applying the LCA perspective and considering not only a specific plant-type as a case study but the whole system at the equilibrium, also permitted the accounting of the GHG emitted and saved from all induced and avoided processes that take place within and beyond Italy. By integrating the environmental and economic analyses, we demonstrated that the new policy scheme contributes to the policy objective of reduced GHG emissions by promoting the utilization of manure (instead of energy crops) while reducing the overall policy cost to society. However, the system efficiency in terms of GHG emissions savings is still low in comparison to the targets recently established for renewable energies in Europe. These observations highlight that only by promoting the system towards the exclusive utilization of manure and agricultural by-products will be possible to obtain further improvements in its economic and environmental efficiency. The integrated approach that we have adopted here is valuable, as it comprehensively and consistently estimates the economic and environmental consequences of competing policies. These consequences would not have been fully captured if only a silo approach - that does not integrate these two aspects - was used to capture these effects, which would not robustly support the identification of optimal policy instruments. It is also replicable in other contexts as the data set used relies on public databases and literature data.

In order to minimize the Indirect Land-Use Change (ILUC) impacts, the revised Renewable Energy Directive constrains the contribution of food-based biofuels to fulfill the EU renewable energy target, starting at 7% in 2021 and decreasing progressively to 3.8% in 2030. Thus, it is expected that the use of dedicated energy crops will be increasingly limited, due to sustainability considerations and support directed only to the use of waste and residues [62-65]. In this respect the integrated model could expand to examine the use of agricultural residues such as corn stover and endogeneize manure management to assess feasibility for residual biomass as an input to the biogas activity at a large scale. Emission mitigation policies depend on the management activities and techniques to process manure. Therefore, analytical modelling is required to take into account the manure management alternatives at farm level and to identify costs and benefits as well as environmental impacts of the various manure treatment alternatives for pig and cattle livestock. On this track, potential extension of the present work should take directly in to account also the livestock sector by including it in the agricultural model, in order to allow direct competition for maize also on the displacement of dairy farms. Multi-criteria decision making algorithms can then be used to seek a novel strategy for determining the efficient alternative from economic, environmental and social aspects. The current modelling framework can accommodate such developments.

Additional investigations could examine the dependency of the results upon some of the key assumptions used in the LCA model (i.e. iLUC coefficient, fugitive methane losses from anaerobic digestion, neglecting silage emissions) and close the loop of integration of economic and environmental models by implementing life cycle activity analysis. Moreover, the eventual yield increases that could be obtained as a result of applying (more) digestate on the field were not considered. In fact, as a result of the digestion process, the digestate contains more inorganic nitrogen than non-digested organic fertilizers, and thus more nitrogen in a form that can be directly used by plants. Future efforts to quantify this effect would further refine the model proposed herein.

Regarding economic environmental interaction the present state of the models determines the environmental consequences in response to the changes in the market. Next step would be to close the loop enabling feedback from the environmental module so that it influences the market clearing solution. As a matter of fact this is the idea of Life Cycle Activity Analysis (LCAA) implying the development of a Consequential Life Cycle Optimization (CLCO) framework that simultaneously optimizes environmental impacts and economic performance. Such method requires global optimization techniques that are beyond the scope of this paper. It is worth to be considered for future research; the environmental impacts of the optimal process designs based on the proposed consequential LCO framework may be significantly lower than those based on the existing attributional LCO framework as demonstrated in [66].

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## Appendix A

The agricultural supply model maximizes profit (A1) under a set of constraints, whose mathematical representation is presented in Eqs. (A2–A5). The indices, parameters and decision variables considered in this model are explained in Box 1.

Objective function:

**Box 1**
Indices, parameters and decision variables of the agricultural model.
*Source: Adapted from Rozakis et al.* [18]

| *Indices/Sets* | | *Parameters* | |
|---|---|---|---|
| $y \in Y$ | non-energy crop index (for sugar beets y = 1) | $\sigma_f$ | farm f total arable area (ha) |
| $d \in D$ | energy crop index ($\lvert D \rvert = m$) | $\sigma_{1,f}$ | maximum amount of land for sugar beet in farm f (ha) |
| $f \in F$ | index for farms | $\pi_v$ | maximum share allowed for crops under agronomic constraint v |
| $v \in V$ | agronomic constraints index | $i_{yv}$ | agronomic constraints dichotomous coefficient = 0 if non-energy crop y is not subject to agronomic constraint v; = 1 otherwise |
| $j \in J$ | index for parametrically imposed prices (only energy crops) | $i_{dv}$ | agronomic constraints dichotomous coefficient = 0 if energy crop y is not subject to agronomic constraint v; = 1 otherwise |
| *Parameters* | | *Decision variables* | |
| $g_{y,f}$ | non-energy crop y gross margin in farm f (€ ha$^{-1}$) | $p_d^j$ | grid j of energy crop d selling price parametrically imposed (€t$^{-1}$) |
| $\gamma_{d,f}$ | energy crop d yield in farm f (t ha$^{-1}$) | $xf_{y,f}^j$ | non-energy crop y area in farm f (ha) under a grid of j exogenous prices |
| $c_{d,f}$ | energy crop d production cost in farm f (€ ha$^{-1}$) | $xe_{df}^j$ | energy crop d area in farm f (ha) under a grid j of parametrically imposed prices |
| $w_f$ | coefficient (weight) to report sample farm arable land to the universe of regional arable land | | |

$$\max \sum_{f \in F} \sum_{y \in Y} g_{y,f} xf_{y,f}^j + \sum_{f \in F} \sum_{d \in D} \left( p_d^j \gamma_{d,f}^j - c_{d,f} \right) xe_{d,f}^j \tag{A1}$$

S.t.

*Land availability:*

$$\sum_{y \in Y} xf_{y,f}^j + \sum_{d \in D} xe_{d,f}^j \le w_f \sigma_f \quad \forall \ f \in F \tag{A2}$$

*Institutional and flexibility constraints:*

$$x_{1,f}^j \le w_f \sigma_{1,f} \ \forall \ f \in F \tag{A3}$$

*Agronomic constraints:*

$$\sum_{y \in Y} i_{y,v} xf_{y,f}^j + \sum_{d \in D} i_{d,v} xe_{d,f}^j \le \pi_v w_f \sigma_f \ \forall \ f \in F \tag{A4}$$

*Non-negativity constraints:*

$$xf_{y,f}^j, \ xe_{d,f}^j \ge 0 \ \forall \ y \in Y \ \forall \ d \in D \ \forall \ f \in F \tag{A5}$$

*Additional constraints for CAP 2014*

Crop Diversification obligation for farms with new CAP land entitlements area > 10 ha:

$$X_n \le 0.75 \quad \text{lg\_land}, n = 1, 2, 3 \ldots N \tag{A6}$$

Farmers should cultivate at least two different crops (set-aside included) and the cropping area of each crop cannot exceed 75% of the new CAP land entitlements area in order to receive the Greening Payment (30% of new CAP decoupled payment).

Ecologic Focus Area obligation for farms with new CAP land entitlements area > 15 ha:

$$0.7 \left[\left[ \sum_{n=1}^{N} Legume_n X_n \right]\right] + X_{st} \ge 0.05 \quad lg\_land \tag{A7}$$

The 70% of the sum of legume crops area[13] plus set-aside area must be at least equal to 5% of the new CAP land entitlements area in order to receive the Greening Payment (30% of new CAP decoupled payment) . Farms with new CAP land entitlements area larger than 15 ha are also obligated to apply the constraint 2.

Crop Diversification obligation for farms with land entitlements area > 30 ha:

$$X_{OPTL1} + X_{OPTL2} \le 0.95 \quad \text{lg\_}land \tag{A8}$$

Farmers should cultivate at least three different crops (set-aside included) and the sum of cropping area of the two largest crops cannot exceed 95% of the new CAP land entitlements area in order to receive the Greening Payment (30% of new CAP decoupled payment). Farms with land entitlements area larger than 30 ha are obligated to apply both the constraints A7 and A8. It should be noted that farmers are not obligated to apply the greening requirements in the organic cropping area.

[13] 1 ha of legume crop corresponds to 0.7 ha in the Ecologic Focus Area.

*Appendix B*

The industrial model consists of the ROI maximization function (8) and constraints (9 − 12), while indices, parameters and decision variables are detailed in Box 2[14].

**Box 2**
Indices, parameters and decision variables of the industrial model (ReSI-M).
*Source: Adapted from Delzeit et al.* [20]

| *Indices / Sets* | | *Parameters* | |
|---|---|---|---|
| $s \in S$ | current plant capacity (size) | $d_s$ | maize needed per plant size (t) |
| $p \in P$ | current input prices (maize) | $dm_s$ | manure needed per plant size (t) |
| $c \in C$ | current region | $dr_s$ | digestate per plant size (t) |
| $k \in K$ | Regions | $s_s$ | maize share (t) on total feedstock (t) (dimensionless parameter) |
| | | fz | output/input coefficient ($m^3$ digestate / t maize) |
| *Parameters* | | fm | output/input coefficient ($m^3$ digestate / t manure) |
| $r_s$ | sum of revenues (€ $y^{-1}$) | | |
| $v_s$ | sum of variables costs (€ $y^{-1}$) | *Decision variables* | |
| $\eta_{sp}$ | per year input costs (maize demand times maize price) | $z_{sc}$ | quantity of maize transported (t) |
| $f_s$ | sum of fixed costs (€ $y^{-1}$) | $y_{sc}$ | quantity of manure transported ($m^3$) |
| $I_s$ | costs for investments (€) | $x_{sc}$ | quantity of digestate transported ($m^3$) |
| $tm_{sck}$ | input (maize) transportation costs (€ $t^{-1}$) | ROI | Return on Investments |
| $tr_{sck}$ | digestate transportation costs (€ $m^{-3}$) | | |
| $tn_{sck}$ | input (manure) transportation costs (€ $m^{-3}$) | | |

Objective function:

$$\max ROI = \sum_{s\ \in S} \sum_{p\ \in P} \frac{r_s - v_s - \eta_{sp} - f_s}{I_s} - \sum_{s\ \in S} \sum_{c \in C} \sum_{k \in K} \left( \frac{tm_{sck}^* \quad z_{sc}}{I_s} + \frac{tr_{sck}^* \quad x_{sc}}{I_s} + \frac{tn_{sck}^* \quad y_{sc}}{I_s} \right) \tag{A9}$$

S.t.

*Input-output constraints:*

$$\sum_{s \in S} z_{sc} \le b_{cp} \forall\ p \in P,\ c \in C \tag{A10}$$

$$\sum_{c \in C} z_{sc} = \sum_{s \in S} d_s{}^* s_s{}^* 1.08 \quad \forall\ s \in S \tag{A11}$$

$$\sum_{c \in C} y_{sc} = \sum_{s \in S} dm_s{}^*(1 - s_s) \forall\ s \in S \tag{A12}$$

$$\sum_{c \in C} x_{sc} = \sum_{s \in S} (z_{sc}{}^*\ f_z + y_{sc}{}^* fm) \tag{A13}$$

*Non-negativity constraints:*

$$zsc \ \ge\ 0c \in C,\ s \in S \tag{A14}$$

$$xsc \ge 0c \in C,\ s \in S \tag{A15}$$

$$ysc \ \ge\ 0c \in C,\ s \in S \tag{A16}$$

$$\pi > 0$$

## 64. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.rser.2019.02.031.

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## Glossary

*ALCA:* Attributional Life Cycle Assessment
*BMP:* biochemical methane potential
*C:* carbon
*CAP:* common agricultural policy
$CH_4$: methane
*CLCA:* consequential Life Cycle Assessment
$CO_2$: carbon dioxide
*dLUC:* direct land use change
*ECC:* External Cost of Carbon
*FADN:* Farm Accountancy Data Network
*GHG:* greenhouse gases
*iLUC:* indirect land use change
*K:* potassium
*LCA:* Life Cycle Assessment
*LCAA:* Life Cycle Activity Analysis
*LCO:* Life Cycle Optimization
*LHV:* lower heating value
*LUC:* land use change
*MAORIE:* Modele Agricole de l'Offre Regionale INRA Economie
*N:* nitrogen
*P:* phosphorus
*PE:* Partial Equilibrium
*PMC:* policy mitigation costs
*PMP:* Positive Mathematical Programming
*ReSI-M:* Regionalised Location Information System – Maize
*RICA:* Rete Italiana di Contabilità Agraria
*ROI:* Return on Investment
*SOC:* Soil Organic Carbon
*TC:* total regional costs
*UAA:* utilised agricultural area
*VS:* volatile solids