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Ex ante Life Cycle Assessment and Environmental Cost-Benefit Analysis of an anaerobic digester in Italy

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ABSTRACT

The end-use valorization of food waste (FW) and biowaste is currently being focused on biofuels and bioproducts production through different technologies. This study evaluated the implementation of a new eco-industrial system in Italy that incorporates a micro-scale anaerobic digestion (mAD) and a Solid-State Fermentation (SSF) unit to produce renewable energy (e.g., electricity and heat) from AD biogas and high-quality bio-based products (e.g., bio-pesticides) from digestate. Three scenarios (S0, S1, S2) were modeled. S0 and S1 included only a solid-liquid separation of digestate through a centrifuge, assuming a different fate for the solid fraction (composting in S0 and application on farmland in S1). S2 integrated SSF and reverse osmosis technologies for the treatment and valorization of digestate with nutrient recovery. The Life Cycle Assessment (LCA) and Environmental Cost-Benefit Analysis (eCBA) methodologies were applied to assess the environmental performances and economic feasibility of the project. The pilot system showed solid environmental performances, especially for S1 and S2, in the five impact categories considered. According to LCA results, the eCBA gives a positive outcome for S2. While the financial and economic analysis showed positive Net Present Values for S2, the project's profitability was not achieved for S0 and S1. If AD plants are implemented at a smaller scale they would represent a favorable investment for the local community; particularly when considering the benefits of nutrient recovery through a complete post-treatment of digestate. The valorization of organic residues could be better supported through introducing alternative market-based policy tools, as well as removing regulatory barriers and encouraging the implementation of financial schemes to support small-scale renewable production systems and the enhancement of market-based instruments for credits certification from renewable energy production.

1. Introduction

1.1. Food systems and circular approaches

The food supply chain is one of the most resource-demanding activities at the global level. Population growth and increasing urbanization rates are forcing modern industrial economies to accelerate their dependence on industrialized food production systems to satisfy an increasing food demand (Osei-Owusu et al., 2019). Moreover, these systems are based on unsustainable input of non-renewable resources, such as mineral fertilizers and chemical pesticides that contribute to affecting both the environment and human health (Sattari

Abbreviations: AD, Anaerobic digestion; CAP, Common Agricultural Policy; CAPEX, Capital costs; CBA, Cost-Benefit Analysis; CBE, Circular bio-economy; Cd, Cadmium; CHP, Combined heat and power; DECISIVE, DECentralIzed management Scheme for Innovative Valorization of urban biowastE; eCBA, Environmental Cost-Benefit Analysis; ENPV, Economic Net Present Value; FE, Freshwater eutrophication; FNPV, Financial Net Present Value; FRS, Fossil resource scarcity; FU, Functional Unit; FW, Food waste; GHG, Greenhouse gas; GW, Global warming; HT, Human toxicity; LCA, Life Cycle Assessment; mAD, Micro-scale anaerobic digestion; MAP, Monoammonium phosphate; NPV, Net Present Value; OPEX, Operational costs; P, Phosphorus; SSF, Solid-State Fermentation; TA, Terrestrial acidification; VAT, Value Added Tax

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et al., 2016).

At the same time, the food supply chain is characterized by another key issue regarding food waste (FW) generation and management. Indeed, the Food and Agricultural Organisation (FAO) (2015) estimated that approximately one-third of global food production is wasted with environmental, social, and economic implications (Teigiserova et al., 2020). In this regard, the European Commission through the Waste Framework Directive (2008/98/EC), later amended by the Directive 2018/851/EC within the 2018 Circular Economy Package, established a legislative framework for the handling of waste in the EU Community. As highlighted by Ng et al. (2019), in view of achieving the sustainable management of natural resources and maximum environmental and socio-economic benefits, it is expressively recommended to transform the current waste management approaches from a linear "take-make-dispose" model to a circular economy model (Ellen MacArthur Foundation, 2014; European Parliament, 2017).

Different methods already exist for the sustainable management of food production systems that can reduce external inputs, promote the circular use of resources and support multiple ecosystem services (e.g., nutrient cycling, climate regulation) (FAO, IFAD, UNICEF, WFP and WHO, 2015). To this purpose, several technologies can transform FW and food residues into value-added products (Tian et al., 2021). The anaerobic digestion (AD) technology is considered one of the most cost-effective biological treatments of organic waste (Ranieri et al., 2018). It converts biowaste into biogas, the latter consisting of methane (CH_4) and carbon dioxide (CO_2). The biogas can also be directly burned through a combined heat and power (CHP) system and generate renewable energy (electricity and/or heat). The by-product of AD, that is rich in nutrients and organic matter, is known as digestate, and can be used on agricultural land to replace mineral fertilizers and prevent the depletion of resources such as phosphorous and potassium (Adam et al., 2018). The digestate also contributes to carbon sequestration as indigestible organic matter can be incorporated into the agricultural soils (Vaneeckhaute et al., 2013). Within the circular economy framework, different technologies are also available for the treatment and valorization of the digestate, such as solid-liquid separation through centrifuge system (Adam et al., 2018) and Solid-State Fermentation (SSF) unit (Cerda et al., 2019). While the first case represents a partial treatment process aiming to separate the digestate into individual fractions, the SSF is a fermentation process that allows the use of solid substrates for generating valuable bio-products (e.g., hydrolytic enzymes, biofuels, bio-surfactants, aromas and bio-pesticides) (Rodríguez et al., 2019).

The possibility of implementing AD and SSF in small-scale digesters and at various geographical locations makes these technologies suitable for optimizing local food production systems like urban agriculture (Thiriet et al., 2020). In particular, the multi-functionality of the AD technology facilitates provision of local ecosystem services able to support human well-being and contributes to creating local circular bioeconomy (CBE) systems (Cong and Thomsen, 2021). For example, supporting (i.e., nutrient cycling) and regulating services (i.e., climate regulation) may be obtained from the use of digestate as fertilizer and the production of renewable energy from biogas.

Non-technological barriers often prevent a more widespread diffusion of these systems, such as low public acceptance; Not in my backyard syndrome; fragmentation among different stakeholders (ISAAC, 2018); lack of knowledge and awareness, and political propaganda.

1.2. The DECISIVE project for the valorization of food waste

The Horizon2020 project DECISIVE (A DECentralIzed management Scheme for Innovative Valorization of urban biowastE) (www.decisive2020.eu), seeks to demonstrate and assess the performance of a decentralized valorization network for biowaste management that incorporates a mAD, a CHP Stirling engine, a centrifuge, and a SSF unit to produce renewable energy and high-quality bio-based products (e.g., bio-pesticides) at the local level. Moving towards a decentralized biowaste management offers key advantages compared to the conventional centralized system as a decrease of transport requirement, a potential increase of community involvement and an opportunity for strengthening local nutrient and energy loops (Thiriet et al., 2020; Walker et al., 2017).

This study presents an ex-ante sustainability assessment of the DECISIVE eco-industrial system currently developing in San Dorligo della Valle (municipality of Dolina; province of Trieste, Italy) served by the A&T (2000) SpA company (https://aet2000.it/). In particular, the municipality of Dolina and A&T (2000) are the two main stakeholders in charge of the administration and waste management system in the municipal area, respectively.

An environmental and techno-economic analysis was performed to identify potential benefits generated for the local community. For this purpose, the Life Cycle Assessment (LCA) and the environmental Cost-Benefit Analysis (eCBA) methodologies were applied. The returns in terms of renewable energy, nutrients and ecosystem services can provide many social and environmental benefits in addition to the economic outputs, even if most of them are not properly compensated by the current market setting (ING, 2015).

To support the potential benefits associated with DECISIVE, as well as to underpin its implementation and potential scaling-up of the system to the municipality, a supportive institutional regulatory environment is essential (Cong and Thomsen, 2021; Angouria-Tsorochidou et al., 2021a). In this study, we conducted a preliminary assessment of the institutional policy framework for efficient management of organic waste as a measure of the readiness, predisposition and potential capabilities of the municipality of Dolina and the waste management operators to implement, scale-up and maintain the DECISIVE system. For this purpose, the study identified and provided potential policy levers within the regulatory framework required to support local CBE systems (Bugge et al., 2019).

2. Material and methods

In recent decades, the Life Cycle Assessment (LCA) methodology has been used extensively to evaluate the environmental benefits and drawbacks of waste management, including energy recovery technologies (Astrup et al., 2015). Today, it is considered one of the most applied methods to assess the environmental performance of products, processes, or systems throughout their entire life cycle (Neri et al., 2018). LCA can also find potential policy-making hotspots to improve the performances of a product/service by acting on the most burdensome processes and avoiding shifting the environmental impacts from one life-cycle phase to another. Moreover, the availability of several impact categories allows a multi-objective analysis that can be adapted to the needs and choices of the user and cover global as well as regional and local environmental impacts (Hauschild et al., 2018). The LCA was performed according to the international standards ISO 14040–14044 (International Organization for Standardization (ISO), 2006, 2020) and following the four phases: goal and scope definition, life cycle inventory, life cycle impact assessment and interpretation of results.

A Cost-Benefit Analysis (CBA) was carried out to assess the attractiveness of the project (Molinos-Senante et al., 2010). In particular, an environmental Cost-Benefit Analysis (eCBA) was applied to evaluate the environmental performance of the project in monetary units (Hunkeler et al., 2008; Weidema, 2006). The eCBA evaluates the environmental damages in monetary terms, such as greenhouse gas (GHG) emissions, but also environmental restorative actions, i.e., climate change mitigation and nutrient recovery (Pearce et al., 2006; Global Green Growth Institute, 2015).

In this study, we adopted the framework presented in Hoogmartens et al. (2014) to integrate all three dimensions of sustainability, i.e., financial, environmental and social, and to include external social and environmental costs and benefits.

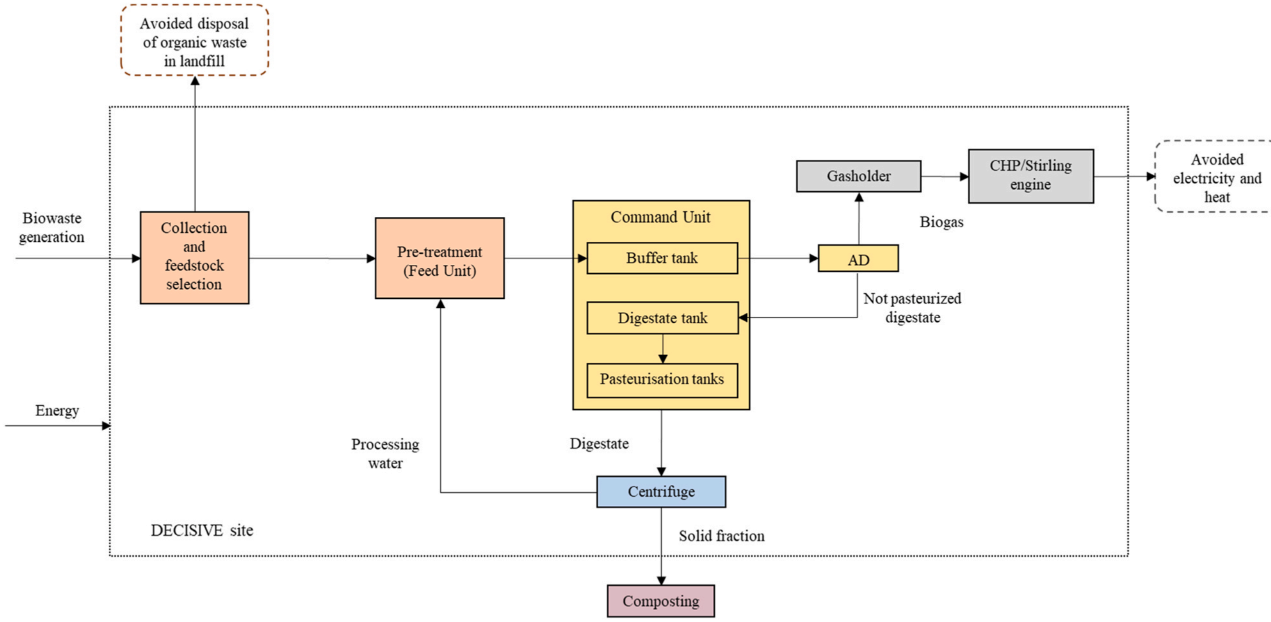

Fig. 1. System visualization with the transferring of solid fraction to composting plant (S0).

2.1. Life Cycle Assessment (LCA)

Fig. 1 shows the current state of the mAD system assessed and the inputs and outputs involved in the process analyzed. The main stages included in the system boundaries are the biowaste collection, pre-treatment and AD.

The municipality of Dolina is located in a heterogeneous territory hosting a protected transboundary natural area with about 374.34 ha of agricultural land. The total biowaste generated from the municipality mainly belongs to households, private companies and school canteens. The biowaste is source-separated from the organic fraction of the municipal solid waste by the residents of Dolina, collected twice per week via a “door-to-door" system and transported to the main treatment plant managed by the A&T (2000) SpA company.

The biowaste generated is collected separately avoiding its disposal on landfill.

Based on the primary data provided by the manufacturing company, the system is a semi-portable technology housed in five 6 m shipping containers for the treatment capacity of 200 t y^{-1} of biowaste. The biowaste is loaded, chopped and mixed in the feed unit (orange boxes in Figs. 1–3) for the pre-treatment step and then delivered to a command unit (yellow box in Figs. 1–3), which is the core of the plant. Here, the pre-pasteurization and pasteurization processes occur in the buffer tank and pasteurization tank, respectively. The feedstock is delivered to the AD unit and converted into biogas and raw digestate, which is further treated in the digestate tank (Fig. 1, Scenario 0 – S0). The biogas is stored in a separated container (the gasholder) and then burned in a CHP engine to generate electricity and heat (grey boxes). Currently, as a stabilization treatment for the raw digestate produced is not available at the A&T2000 plant, it is phase-separated with a centrifuge (light blue box), to produce a solid and liquid fraction. The solid fraction is delivered to a composting plant (pink box) outside the Trieste province (e.g., Codroipo – Udine province), which implies the use of a truck over a distance of 100 km to the final site. While the liquid fraction can be used onsite as processing water (e.g., equal to 50 % of the mass of the feedstock) to ensure a complete mixing process during the pre-treatment step as conventionally occurring in wet AD process (Møller et al., 2009).

Although the direct soil application of digestate from source-separated biowaste is not possible in Italy (Bartocci et al., 2020), a material flow analysis was performed to evaluate the nutrient and micro pollutant content of the digestate and its suitability to be used as organic fertilizer following the new European waste-derived fertilizer regulation (EU, 2019/1009). Two scenarios were modeled to evaluate the impact of different post-treatment practices on the environmental performance of the system (Figs. 2 and 3) and inside the DECISIVE site. The sequence of scenarios represents different configurations of the plant, progressively adding solutions that imply investments but also possible improvements in environmental and economic-financial results. The study will investigate if more mature and equipped plant configurations may be more adequate to take care of waste management and resource recovery towards implementation of circular economy solutions.

In the first scenario (S1) (Fig. 2), the raw digestate is phase-separated with a centrifuge (light blue box). Unlike S0, the solid fraction was assumed to be applied on farmland to substitute for mineral fertilizers. Regarding the liquid fraction, this was recovered and returned to the pre-treatment stage as in S0.

In the second scenario (S2) (Fig. 3), the work of Rodríguez et al. (2019) regarding the valorization of biowaste digestate via SSF was considered for the post-treatment of the solid fraction. The SSF process (green box) is carried out in absence - or near absence – of free water on a substrate that possesses sufficient moisture to support the growth of microorganisms (Thomas et al., 2013). In particular, the *Bacillus thuringiensis* was used as inoculum, while the mass input was composed of solid digestate (47 %), raw biowaste (28 %) and wood chips (26 %), used as a bulking agents, according to DECISIVE (2017b, 2017c). Moreover, additional processing through the reverse osmosis technology (green box) was hypothesized for treating the liquid fraction after the centrifuge. Reverse osmosis is a membrane-based treatment producing a fraction of nutrients concentrate (reverse osmosis retentate) and processing water (reverse osmosis permeate) (Adam et al., 2018). Following Adam et al. (2018), the study assumed that the retentate can be applied on farmland substituting mineral fertilizers and the permeate returns to the pre-treatment unit to facilitate feedstock mixing. However, in this case to achieve the right consistency of the substrate an additional quantity of water was added (Fig. 3). Moreover, the present study calculated the partition percentages of nutrients between retentate and permeate fractions according to the mass flow diagram presented by Adam et al. (2018) for pilot 1.

For S1 and S2, the avoided production of mineral fertilizer and related emissions savings were modeled considering the substitution of

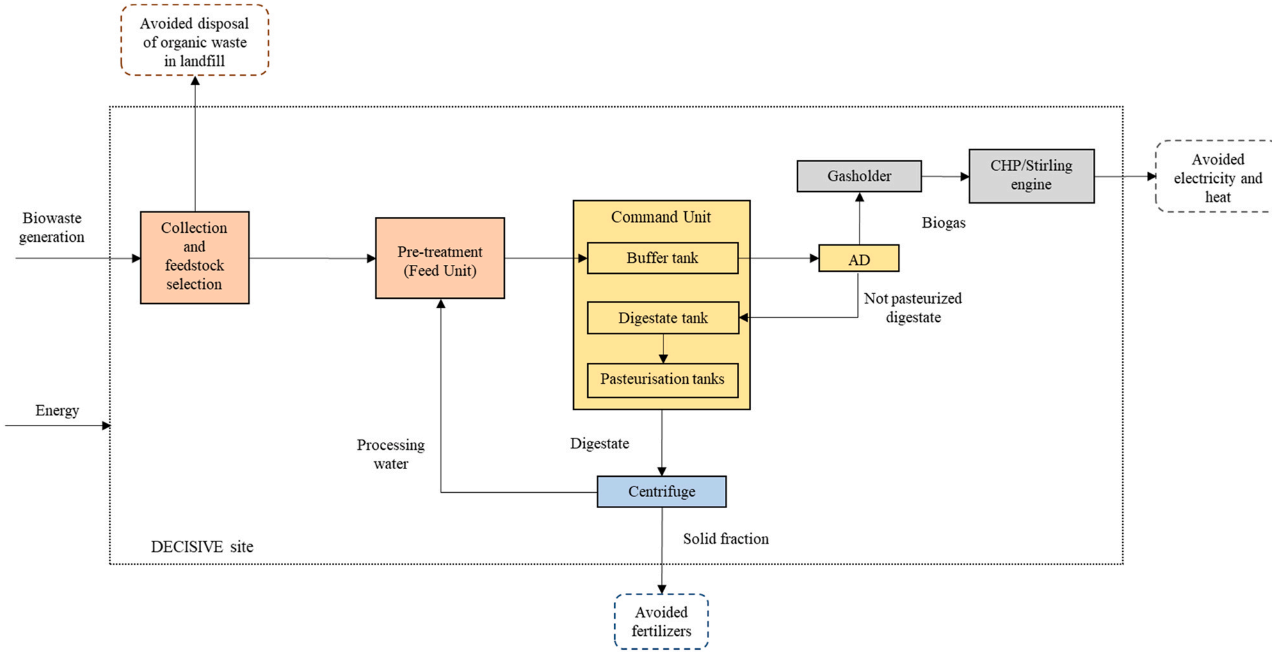

Fig. 2. System visualization with the application of solid fraction of digestate as organic fertilizer (S1).

phosphorus (P) content in monoammonium phosphate (MAP - $NH_4H_2PO_4$) fertilizer with the content in the digestate produced (solid fraction in S1 and retentate fraction in S2) (Niero et al., 2014). According to Ascher et al. (2020) it was assumed that the nutrient content of the digestate was the same as the one in the feedstock. In this regard, the data relating to the chemical composition of the feedstock was provided directly by A&T company. Moreover, within scenarios S1 and S2, the study hypothesized that the bio fertilizer generated from the post-treatment practices of digestate was used locally (application of solid digestate and retentate) and within 10 km of transportation distance (Angouria-Tsorochidou et al., 2021b). Even though the biowaste is source-separated, according to Knoop et al. (2018), Kupper et al. (2014) digestate may contain pollutants such as heavy metals, so before it can be applied to agricultural land, it needs to meet legal thresholds. Attention was focused on Cadmium (Cd), as the frequent application of mineral P rich fertilizer is mainly responsible for it (Marini et al., 2020).

Following a “cradle to cradle” approach, we modeled the DECISIVE pilot plant and a functional unit (FU) of 1-ton wet weight (t ww) of biowaste treated was used for the LCA. Data for the foreground system was obtained from A&T2000 company or collected from the scientific literature and technical reports (Table S1 in Supplementary materials). Background data were obtained from the Ecoinvent database version 3.6. Based on the quantity of biowaste treated by the DECISIVE pilot plant per year, we modeled the collection stage by considering an average total distance of 77 km per collection cycle traveled by two conventional municipal trucks and then allocated for the FU.

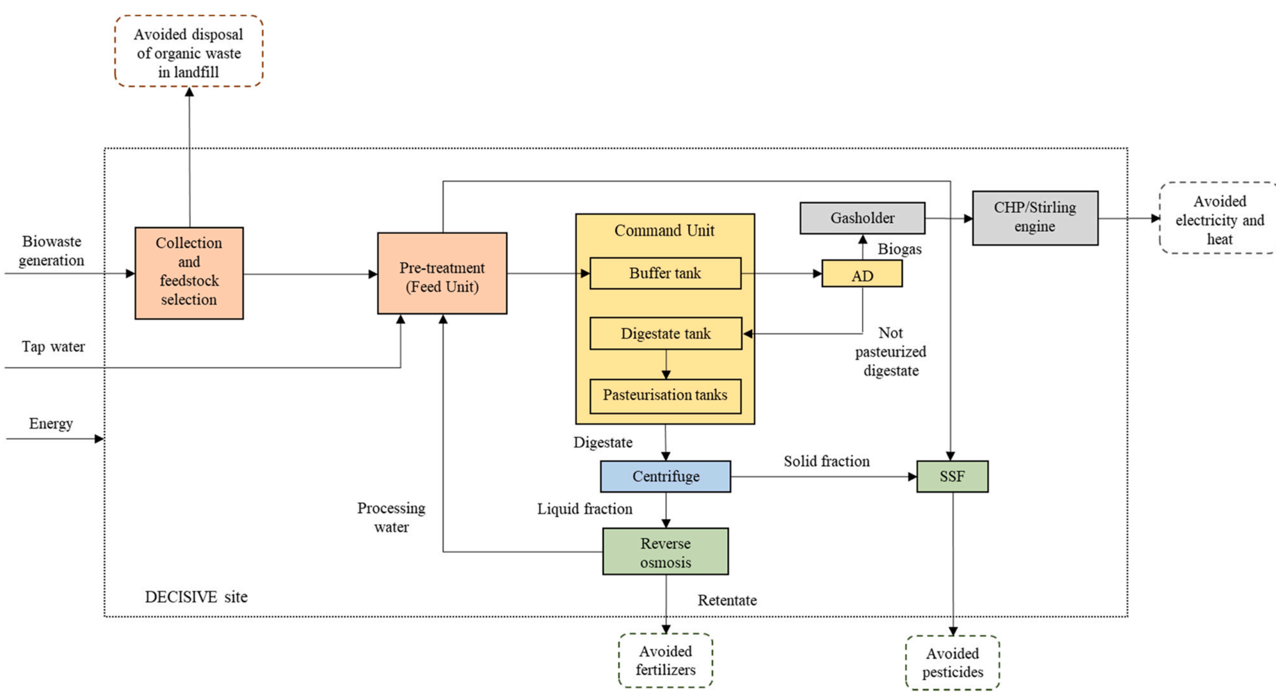

Fig. 3. System visualization with additional processing of the liquid fraction and integration of SSF unit (S2).

Based on a previous study of DECISIVE (2018) and according to the biogas production yield of 100–150 m^3 t ww^{-1} (Manfredi and Pant, 2011), this study calculated the volume of the biogas produced as 133 N m^3 t ww^{-1} (moisture content of the substrate 49 %, data from laboratory tests), which is the sum of the total volume of CH_4 (86.71 N m^3 t ww^{-1}) and CO_2 (46.69 N m^3 t ww^{-1}): the main components. As the specific data on methane content in the produced biogas was lacking, this study adopted 65 % CH_4, according to Møller et al. (2009) and Curry and Pillay (2012). Although biogas also contains variable amounts of other trace gases (e.g., H_2S, N_2) (Manfredi and Pant, 2011), we assumed negligible for the purpose of the study. The biogas fugitive emissions were assumed from literature to be about 1%, in agreement with Edwards et al. (2017). Regarding the energy production, through the combustion of the biogas, we assumed that the CHP unit had an 80 % efficiency, where the electricity and heat obtained were 20 % and 80 %, respectively (Møller et al., 2009).

Regarding the quantity and quality of digestate produced, the study of Angouria-Tsorochidou et al. (2022) was consulted to evaluate the composition of the liquid and solid fractions achievable using the specific partitioning factors.

Based on Møller et al. (2009), Maulini-Duran et al. (2015) and Khoshnevisan et al. (2018), the system's emissions in the main stage of the process (i.e., anaerobic digestion, combustion of biogas, SSF operation and solid digestate/retentate application) were evaluated (see Table S2 in Supplementary material).

The Life Cycle Inventories were prepared in Microsoft Excel® to store and model the collected data. Regarding the Life Cycle Impact Assessment, the software SimaPro 9.1 (PRé Sustainability, 2021) was used, and the impact assessment was performed with ReCiPe 2016 (H) at midpoint level to evaluate the following impact categories: global warming (GW), terrestrial acidification (TA), freshwater eutrophication (FE), fossil resource scarcity (FRS) and human toxicity (HT). ReCiPe method was chosen as it is one of the most widely used methods for the impact assessment in literature (Ascher et al., 2020; Bacenetti et al., 2016; Slorach et al., 2019). Direct emissions of N_2O and CH_4 from process stages as anaerobic digestion or combustion of biogas contribute to global warming, as well as the application of mineral fertilizers to agricultural soils that are recognized as major drivers of nitrous oxide (N_2O). For this reason, their substitution might contribute to improving environmental performances in terms of GHG emissions, but also in terms of soil acidification, as they would mainly reduce ammonia (NH_3) emissions, and freshwater eutrophication due to nitrate (NO_3^-) and phosphates enrichment. Furthermore, the substitution of chemical pesticides reduces atmospheric organic contamination (Bøckman and Olfs, 1998) and heavy metals concentration (e.g., Cadmium, Cd) in soil, contributing to reducing human health impacts. Finally, the production of renewable energy sources such as biogas can significantly contribute to reducing the depletion of fossil fuel resources (Hasler, 2017).

2.2. Environmental Cost-Benefit Analysis (eCBA)

The financial and environmental benefits of the DECISIVE system were calculated along with the associated operating and capital costs necessary to maximize resource efficiency and minimize waste production within the framework of economic and social sustainability (Hislop and Hill, 2011). The Net Present Value (NPV) was calculated to evaluate all future net cash flows over the entire life of the project discounted to the present time, following Eq. (1) (EC (European Commission), 2017):

$$NPV = \sum_{t=0}^{n} \frac{S_t}{(1+r)^t} \tag{1}$$

where S_t is the balance of cash flow (difference between revenues and costs) at time *t*, over the project operation period (*n*). The discount factor $(1+r)^{-t}$ allows to compare the flow of cost and revenues streams over longer periods and to determine the feasibility of the project's; i.e. whether the net present revenues exceeds the net present costs (NPV > 0) (Global Green Growth Institute, 2015; Arena et al., 2020). Following Eq. (1), a positive NPV generally indicates an expected profit and the possibility to move forward with the project assessed.

The financial performance through the financial analysis of the systems (mAD and SSF) was assessed for an operational period of 20 years and expressed as the financial NPV (FNPV), assuming a social discount rate of 6 % in agreement with Ascher et al. (2020). Furthermore, the study performed the economic analysis through the economic NPV (ENPV) to consider social and environmental externalities for the overall evaluation of the performance of the DECISIVE project. Within the ENPV, in agreement with EU regulation a social discount rate of 3 % was applied together with appropriate conversion factors (European Commission, 2015).

Table S3 in Supplementary materials shows the capital costs (CAPEX), such as the initial investment for the plant construction, and the operational costs (OPEX) which cover the maintenance for running of the plant, collecting and transporting biowaste, and the labor associated with these main process stages. Furthermore, Table S3 shows the conversion factors to convert market prices into shadow prices and correct the market distortions (e.g., fiscal corrections) in the economic analysis (European Commission, 2014).

For both FNPV and ENPV, the benefits included the revenues from the potential sale of bio-based products (e.g., organic fertilizer, biopesticides) (S1 and S2) (Table S3). As currently in Italy, there are no government incentives or subsidies directed toward the running AD or compost production, they were not considered in the analysis.

Regarding the fiscal corrections of bio-based products prices, after determining their exact value, these were withdrawn from the cash flows in agreement with the EU guidelines (European Commission, 2014). In particular, the Value Added Tax (VAT) payments were subtracted in the economic analysis, considering 4% for fertilizers (L.1984/748) and 10 % for bio-pesticides (D.P.R. 1972/633, Table A). In the ENPV, externalities associated with climate change were monetarized based on the system's net GW potential impact category that accounts for the embedded carbon footprints of the system (CO_2eq. trading). The price rate of 25 €/t CO_2eq was retrieved from Sendeco2 in 2019 (Sendeco2, 2021). Negative net carbon footprints were considered as positive externalities and included in the eCBA as revenues, while positive net carbon footprints were considered as negative externalities and included in the eCBA as costs.

Finally, the study carried out a sensitivity analysis to assess the robustness of the CBA results. For S0, S1 and S2 different scenarios were defined with the variation of the characteristic parameters, such as the social discount rate; the selling price and/or production quantity of biopesticides. The selling price of electricity produced was also considered as a parameter influencing output results.

2.3. Institutional policy framework

The institutional policy framework describes the public policies and regulatory instruments that affect the waste management system and its implementation (Turcott Cervantes et al., 2021). We collected information on the institutional functioning as well as policies, laws, regulatory instruments and economic tools adopted at various institutional levels (e.g. national, regional and local) from scientific publications retrieved from Web research tools such as Science Direct, Scopus and Google Scholar®. Other official documentation (e.g. documents and reports released by EU agencies or public and private administrations) was retrieved by screening institutional public web pages (e.g. Dolina Municipality, Italian Ministry of the Ecological Transaction, European Commission, etc.). Moreover, additional information on the existing policy instruments was obtained in collaboration with A&T2000 company.

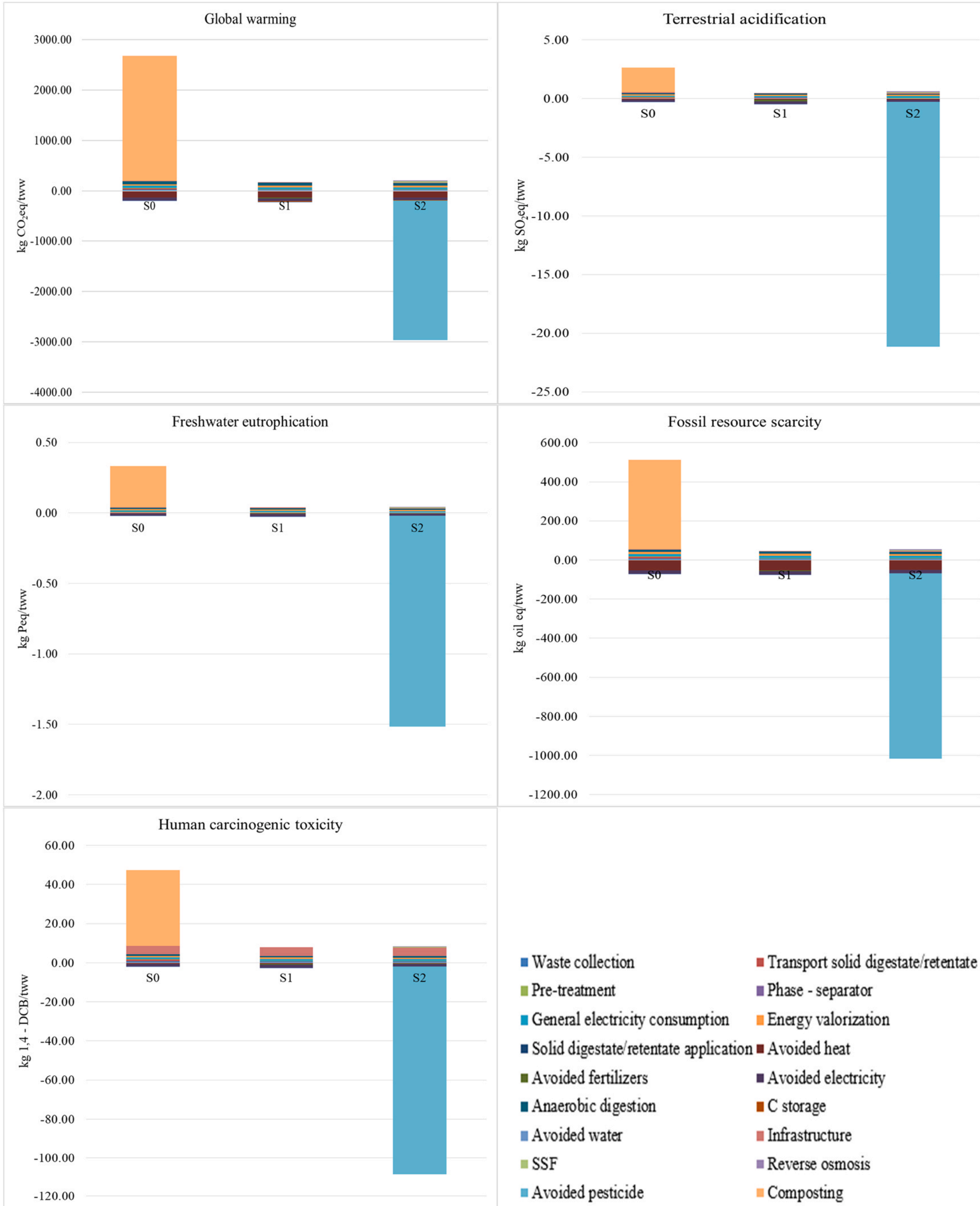

Fig. 4. LCA results for all scenario analyzed.

3. Results and discussion

3.1. Life Cycle Assessment

Fig. 4 shows the LCA results for all scenarios analyzed, with particular attention to the impact categories considered (GW, TA, FE, FRS and HT). Table S4 in Supplementary materials shows the results for the overall impact categories considered in the ReCiPe 2016 method.

The pilot system in Dolina showed solid environmental performances in all five impact categories considered particularly for S1 and S2. Concerning the GW, the energy production through biogas combustion and the avoided burdens of pesticide production contributed to net-negative impacts in S1 and S2: − 46.10 kg CO_2eq t ww^{-1} and − 2767 kg CO_2eq t ww^{-1} respectively. In S0, results showed net-positive impacts due to the emissions from the treatment of digestate solid fraction at the composting plant, which were responsible for 93 % of the GW potential (2484 kg CO_2eq t ww^{-1}) mainly attributable to the heat use (background data from Ecoinvent dataset). On the other hand, in S0 the avoided electricity and heat generation (−69 and −31 kg CO_2eq t ww^{-1}), indicated potential environmental credits obtainable due to decreasing use of fossil fuels for local energy production. For S2, the avoided emissions associated with the production of bio-pesticide through the application of SSF unit presented the highest contribution to net-negative impact for total GW (− 2779 kg CO_2eq t ww^{-1}). The credits from avoided inorganic pesticides emissions could exceed the processed emissions and outweigh the impact from energy used, resulting in net negative GW. Regarding the GHG emissions, the potential savings associated with the substitution of MAP was common within S1

(− 12 kg CO_2eq t ww^{-1}) and S2 (− 0.80 kg CO_2eq t ww^{-1}). In S2, the limited net savings in GHG emissions were mainly caused by the lower quantity of P available in the retentate to replace MAP fertilizer. In this regard, the material flow analysis of nutrient content of the digestate confirmed its suitability to be used as organic fertilizer. In particular, the total amount of P in the solid fraction of digestate and retentate was calculated to be 3.49 and 0.20 g kg ww^{-1}, respectively (details in Supplementary material). This was in line with the regulatory limits established by the EU fertilizer Regulation (4.36 g kg ww^{-1}) (EU, 2019/1009). Regarding the content of contaminants such as heavy metals, the concentration of Cd in the solid fraction of digestate and retentate was calculated as 0.14 and 0.75 mg·kg dry $matter^{-1}$, respectively, in agreement with the EU regulation limit of 1.5 mg·kg dry $matter^{-1}$ (EU, 2019/1009) (details in Supplementary material).

In S1 and S2, the net savings in GHG emissions were in line with the results of Slorach et al. (2019) and Ascher et al. (2020) studies. The former study considered the life cycle environmental implications of recovering energy and material resources from FW comparing four treatment methods (anaerobic digestion, in-vessel composting, incineration and landfill). AD showed the lowest environmental impacts including net-negative GW potential (− 31.6 kg CO_2eq t^{-1} of food waste) mainly due to the displacement of grid electricity. Ascher et al. (2020) conducted an LCA of a community-based food waste treatment system in the UK through a small-scale wet AD and evaluated a GW potential value of − 92.27 kg CO_2eq t^{-1} of food waste mainly attributed to the avoided emissions resulting from energy displacement (− 90 kg CO_2eq t^{-1} due to electricity displacement), and the use of digestate as a fertilizer.

Regarding the positive contribution to GHG emissions, in S0 and S1, the energy consumption for the mAD system operations and the CHP Stirling engine and control units contributed to the total GW by 125 kg CO_2eq t ww^{-1} due to emissions caused by electricity consumption. In S2 these emissions contributed by 138 kg CO_2eq t ww^{-1}, considering the additional energy demand for reverse osmosis. The contribution of electricity consumption for the phase-separator through centrifuge was instead negligible within all scenarios (S0 and S1: 1.17 kg CO_2eq t ww^{-1}, S2:1.13 kg CO_2eq t ww^{-1}) while for the operation of reverse osmosis system in S2 the electricity consumption was responsible for 9% of the emissions in terms of GW potential. In all scenarios the AD process was responsible for a negligible percentage of the emissions with a GW average value of 55 kg CO_2eq t ww^{-1}. In S2 the SSF process accounted for 9 % of emissions contributing to the total GW by 19 kg CO_2eq t ww^{-1}.

Finally, the GHG emissions related to the use of infrastructure materials and the waste collection stage contributed respectively around 7 % and 12 % on the total impact in S1 and S2 and were negligible in S0.

Regarding TA and FE categories, for S0 and S1 the total values were 2.36 and 0.03 kg SO_2eq t ww^{-1} and 0.31 and 0.01 kg P eq t ww^{-1}, respectively, while in S2, total TA and FE presented negative values (− 21 kg SO_2eq t ww^{-1}, − 1.47 kg P eq t ww^{-1}) due to the avoided emissions from inorganic pesticides. In S0, the composting treatment of the digestate solid fraction contributed significantly to these impact categories (around 84 %) due to energy use (electricity and heat), while the energy valorization phase through the CHP Stirling engine and the electricity consumption processes contributed to around 4 % within TA and FE. The electricity consumption involved in the phase-separator stage was negligible for both categories (less than 0.5 %). These categories were influenced by the contribution of avoided emissions due to the MAP substitution especially in S1 (− 39 % for TA and − 20 % for FE) and the electricity production (− 46 % for TA and − 72 % for FE). In S2, results for TA and FE followed the same trend seen in the GW impact category, highlighting the important contribution of avoided pesticide production, about − 99 % of the impact.

Regarding the positive contribution, in S2 the energy consumption for the mAD system operations and the CHP Stirling engine and control units contributed to TA and FE for 20 %, caused by electricity consumption.

These results highlighted the relevance of implementing the use of digestate rather than sending it for composting, in line with those of Ascher et al. (2020) and Tian et al. (2021). The former study obtained 0.24 kg SO_2eq t ww^{-1} for AD due to emissions from biogas utilization in the CHP unit, while the latter study evaluated 0.27 kg SO_2eq t ww^{-1} and − 0.01 kg P eq t ww^{-1} for a decentralized FW AD system mainly influenced by the recovered nutrients from digestate utilization and recovered energy.

The impact category FRS contributed to outbalance the impact from conventional heat and electricity generated and displaced from the grid, respectively around − 52 and − 18 kg oil eq t ww^{-1} in all scenarios. This data was in line with the results of Tian et al. (2021) who identified the recovered energy as a contributor in reducing fossil fuel consumption by − 59 kg oil eq t ww^{-1}. However in S0, the energy consumption at the composting plant was responsible for the largest percentage of total emissions (442 kg oil eq t ww^{-1}). The electricity consumption for the different treatment stages and energy valorization through the CHP Stirling engine reached about 10 kg oil eq t ww^{-1} in all scenarios, while the electricity use during the phase-separation with centrifuge was marginal. In S2, the electricity for reverse osmosis operation was nearly 5 kg oil eq t ww^{-1} and the SSF process accounted for 10 % of emissions. Regarding the bio-pesticides credits, they reflected 93 % of emissions avoided, equally to − 949 kg oil eq t ww^{-1}.

The total HT value accounts for 45, 5 and − 100 kg 1,4-DCB t ww^{-1} in S0, S1, S2, respectively, highlighting a progressive improvement in performance as scenarios evolve. In S0, the HT impact category was mainly influenced by the operation of composting plant (mainly electricity use), which accounted for 82 % of the emissions. The avoided emission associated with the energy production instead accounted on average for − 48 % in S0 and − 37 % in S1. The use of infrastructure materials was more relevant in scenarios S1 and S2, accounting for 54 % of the emissions. As shown in Fig. 4, the HT impact category in S2 resulted in negative emissions due to the avoided inorganic pesticide production.

3.2. Environmental Cost-Benefit Analysis: financial and economic NPV

The financial and economic NPV of treating one ton of FW in each scenario is presented in Fig. 5. Fig. 5(a) shows that S0 and S1 performed a net-negative FNPV estimated at − 7782 and − 7200 € t ww^{-1}, respectively. This suggests that the investment is not financially profitable when only the mAD unit is implemented, even after accounting for the benefits obtainable by the replacement of mineral fertilizer with solid digestate (S1). On the contrary, S2 showed a positive NPV equal to 54,710 € t ww^{-1}. This result would suggest the excellent profitability of the DECISIVE system for S2 resulting from the bio-pesticide production revenues, which are the main contributor to the total benefits of the scenario (64,047 € t ww^{-1}).

Results from S2 were in agreement with Badgett and Milbrandt (2021) who demonstrated the economic benefits of several FW management pathways (able to treat between 50,000 to 250,000 t y^{-1}) in the U.S. including AD. Although the study of Badgett and Milbrandt (2021) shows that the economic benefits primarily depend on the local tipping (gate) fee charged by the facility, which was not considered in the present study, the additional revenues from the sale of biofuels or bioproducts can reduce this dependence. In this regard, as emerged in Ascher et al. (2020), which also evaluated the scheme's economic feasibility, in S0 and S1 revenues from the sale of electricity, heat and fertilizer were small (S0: 414 € t ww^{-1}; S1: 457 € t ww^{-1}) if compared to the treatment costs as explained below.

In all scenarios, the capital investment for the collection of biowaste (350 € t ww^{-1}) was mainly related to the purchase of trucks and was negligible compared to CAPEX for the treatment stage (S0 and S1: 3636 € t ww^{-1}; S2: 4011 € t ww^{-1}). This was also true in S0, where we

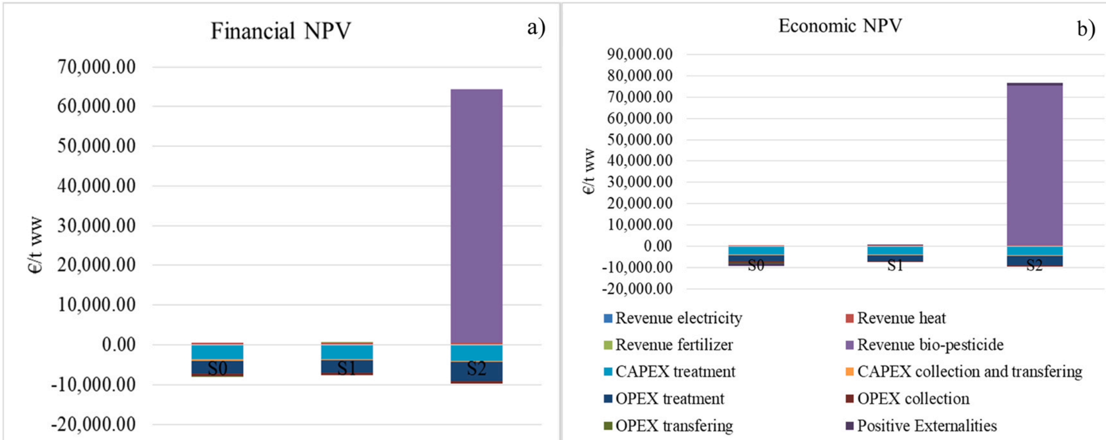

Fig. 5. Results of financial and economic NPV for all scenarios assessed.

considered the additional cost for transferring the digestate to the composting facility through a third truck (175 € t ww^{-1}). In this context, in each scenario assessed, the operational costs for the treatment activities, including maintenance and labor costs, were more relevant (S0 and S1: 3139 € t ww^{-1}; S2: 4842 € t ww^{-1}) compared to the operational costs for the collection stage (533 € t ww^{-1}). The latter included diesel costs for operating the trucks and the costs associated with the system maintenance and labor, which were largely irrelevant. In S0, the delivery of digestate to the composting plant, with the related operational costs (the diesel consumption and labor), was marginal (364 € t ww^{-1}).

Fig. 5(b) shows the results of the ENPV. S0 and S1 presented a net-negative ENPV of − 9180 and − 6868 € t ww^{-1}, respectively. This suggests that the project is not able to generate social benefits. However, when the system integrated the post-treatment of digestate with the production of bioproducts and the positive externalities from avoided GHG emissions, the related revenues ensured social gains in S2, with positive ENPV of about 67,218 € t ww^{-1}.

The CAPEX related to the collection stage (310 € t ww^{-1}) in each scenario remained negligible compared to the ones associated with the treatment phase (S0 and S1: 3930 € t ww^{-1}; S2: 4304 € t ww^{-1}). As for the financial analysis, the economic analysis performed for S0 showed that the capital cost associated with the digestate transferred to the composting plant was marginal (155 € t ww^{-1}).

Overall, the revenue prices followed the same trend seen in the financial analysis. The conversion factor applied to electricity and heat was one, as there was no evidence of market failures.

The ENPV associated with S2 highlighted that the revenues from bio-pesticide production represented the main contribution to the total benefits as previously seen for the FNPV. For the economic analysis as well revenues from the sale of electricity, heat and fertilizer were small in S0 and S1 if compared to the treatment costs. In all scenarios, the OPEX for the treatment activities (S0 and S1: 2773 € t ww^{-1}; S2: 4374 € t ww^{-1}) remained more relevant compared to the operational costs associated with the collection phases (469 € tww^{-1}). Regarding the benefits associated with the negative net carbon footprints LCA-based, the results from this study suggested several positive environmental externalities especially for S2 with the following result: 1390 € tww^{-1} respectively. This result was mainly due to the revenues from bio-pesticide. On the other hand, S0 showed negative externalities (− 1235 € t ww^{-1}) associated with its positive net carbon footprint, as no revenues from fertilizer or bio-pesticide were assumed.

The eCBA results indicated that the implementation of the new eco-industrial system as that described in S2 was not only desirable from a financial point of view but also economically viable. In comparison with landfill or incinerator systems, Badgett and Milbrandt (2021) revealed that smaller facilities limited to the management of organic wastes, such as AD, exhibited higher NPV at small scales, but less significant increases in NPV with plant size. This suggests that if AD plants are implemented at a smaller scale they would represent a favorable investment for the local community; particularly when considering the possibility of exploiting the benefits of nutrient recovery through a complete post-treatment of digestate (S2) as well as the production of renewable energy.

3.3. Assessment of the institutional policy frameworks

The Law Decree 116/2020 regarding the waste management has been recently implemented in Italy in agreement with Directive 851/2018 of the EU Circular Economy Package (D.L. 116/2020). According to the D.L. (116/2020), a national plan for waste management is expected to be implemented by 2022 and will define macro-objectives, criteria and guidelines at which the Italian regions and provinces will need to comply with their regional waste management plans (Ministry of the Ecological Transition (MITE), 2010).

Currently, the Regional Prevention Plan (RPP) of Friuli Venezia Giulia (region where Dolina is located) identifies the guidelines for the reorganization of the integrated urban waste management service and defines the overall results to be achieved, by also providing recommendations to the municipalities on which possible economic instrument to adopt, but without establishing any (Regional Prevention Plan (RPP), 2016). Therefore, economic instruments have not been implemented at the regional level. Several of these instruments are also recommended by the D.L. (116/2020) such as the coordination of national taxes on landfills to incentive prevention and recycling; incentives to promote waste prevention and intensify separate collection schemes avoiding to support landfill and incineration; pay-as-you-throw tariff schemes that affect the waste producers based on the actual amount of waste products and provide incentives for source separation of recyclable waste (e.g. a reduction in tax according to the sorting efficiency); fiscal incentives for the food products donations; reduction of subsidies in contrast with the waste hierarchy; support to research and innovation in the advanced recycling technologies, public awareness campaigns on the prevention of the production of organic waste (D.L. 116/2020).

For certain aspects concerning the organic fraction of the municipal solid waste, D.L. (116/2020) is still ongoing an adjustment process to the Directive 851/2018. The following list presents the main modifications of relevance for organic waste management and the support of local CBE systems.

- The organic fraction needs to be source-separated and recycled (including composting or anaerobic digestion) mandatorily in all national territory by the 31st of December 2021
- The organic fraction is defined as recycled only if either composted

in proximity or send to industrial composting or anaerobic digestion plants

- The national fee concerning the management of waste (TARI) can be reduced for households who produce compost from organic waste by promoting self-composting and local or community composting (De Simone, 2016)
- The quality of bio-products from AD has to respect the EU standards for environmental protection (e.g., only the fertilizer products that respect the EU regulation are allowed)
- Within one year of entry into force of this decree (D.L. 116/2020) quality criteria will be established for the source-separation collection and precise criteria to be applied to the quality controls of the collection as well as the recycling plants for the organic fraction

At the municipal level, a specific plan for organic waste management does not exist in Dolina but it is part of the general regulation for solid waste management (San Dorligo della Valle, 2018). The attention of local authorities to prevention policies and awareness is rather scarce and therefore limited performance was registered in terms of the existence of plans and policies for specific waste streams and regarding the existence of proper economic instruments that promote the efficient use of resources and organic waste reduction.

Only in a few regions and provinces in the North of Italy existing mandatory instruments have been implemented such as the Green Public Procurement (GPP), green certifications and the TARI waste management fee (Ronchi et al., 2020). The GPP favors the use of products made with recovered or recoverable materials and the public administration can have a strong capacity of market orientation towards eco-efficient production with less waste production (Damian et al., 2010).

Regarding the recycling of the organic fraction and the use of organic valorization technologies to produce digestate as organic fertilizer, since the quality levels of the organic fraction have not yet been updated, the EU fertilizer regulation 2003/2003 introduced at the national level through the Regulation 75/2010 defines the use of digestate as fertilizer (Ministry of food, agriculture and forests (MIPAAF), 2010). At the national level, when the digestate is placed on the national market as organic fertilizer, it has to respect strict technical parameters described in the national normative 75/2010 including limits in the content of contaminants, such as heavy metals. To obtain this authorization, sampling and analysis procedures requires to be conducted on behalf of the digestate producers and in agreement with the Law Decree 75/2010 (Ministry of food, agriculture and forests (MIPAAF), 2010). However, specific limits for contaminants inside the digestate do not exist in case the digestate is not introduced in the market, and this might represent an issue for the adoption of decentralized waste management systems using micro-biowaste treatment technologies to obtain liquid or solid digestate that can be used as organic fertilizers in local CBE systems.

3.4. Identification of main policy levers to support local circular bioeconomy systems in Italy

The LCA and eCBA analyses identified several hotspots in need of specific policy-making responses to ensure the economic viability as well as the removal or implementation of policy measures that might act as barriers or enablers for the successful implementation, scale-up and maintenance of the DECISIVE system.

First, the energy production through the biogas combustion resulted in net-negative global warming impacts in all scenarios and therefore contributed positively to reducing GHG emissions. Various mechanisms of incentives exist in Italy to support the supply of renewable energy and develop renewable energy sources (Gestore del Servizio Elettrico (GSE), 2020). One important example is the market-based instruments implemented in Italy and UE such as the Italian green certificates and the EU Emission Trading System (EU-ETS) which allow to allocate, buy and sell carbon credits from electricity generated from biogas in the national and EU markets (Miotto, 2014). Another example is the use of feed-in tariff schemes which allowed to significantly expand the biogas sector in Italy over the last decades (Benato and Macor, 2019). While generous financing schemes have been reduced over the last years, the Italian government has tended to implement financing policies toward “more sustainable criteria, switching from financing big plants to encouraging smaller ones, in order to match the availability of farm´s input with the size of the plant” (Pirelli et al., 2021). The bioenergy sector is becoming a key source in the policy agenda of EU countries as it positively contributes to improve energy security by reducing energy dependence from fossil fuel imports from foreign countries and GHG emissions (Benato and Macor, 2019). However, supporting local bioenergy realities can represent a valuable long-term strategy compared to large-scale exploitation of bioenergy, as the latter is already the leading cause of concern for food security, with multiple environmental issues caused by biofuel production in agriculture (Wesseler and Drabik, 2016). Local bioenergy chains might better ensure local employment in rural areas, raising market competitivity and growth for local communities and, possibly, reduced poverty.

According to the LCA and eCBA results, it is necessary to provide policy measures that support the implementation of AD in combination with SSF technologies to achieve optimal environmental and economic performance, notably by replacing the use of mineral fertilizers and chemical pesticides.

The current EU regulatory measures to tackle the application of mineral fertilizers and pesticides such as the Nitrate Directive and the EU Common Agricultural Policy (CAP) showed scarce results (Marini et al., 2020). For example, the CAP has so far allocated only 20 % of the subsidies for the period 2021–2027 to small and medium-sized producers for sustainable and quality improvements and reduction of mineral fertilizer application (Gazzani, 2021). Furthermore, large direct subsidies at the national level support the use of nitrogen fertilizers (4 % VAT instead of 22 %) and chemical pesticides (10 % VAT instead of 22 %) and shall be removed. This controversial fiscal mechanism still needs urgent reform to allow the implementation of supportive instruments for organic fertilizers and pesticides which so far have not received any subsidies in Italy (Gazzani, 2021).

In all scenarios evaluated, the capital and operational costs were relevant and can constitute an impediment in the initial implementation of the project for a local community. However, although economic support might be necessary at the initial stage of implementation, alternative policy tools such as payment for ecosystem services (Pizzol et al., 2014; D’Amato et al., 2020; NAAS, 2020; Zandersen et al., 2009) that convert valorized nutrient streams (e.g., biogas, fertilizers and biopesticides) into monetary values might be able to largely compensate these costs over time. Similar mechanisms might also stimulate the appropriate design and adoption of compensatory tools for additional socio-environmental benefits originating from local circular bioeconomy systems that cannot be traded and priced in markets (Cong and Thomsen, 2021).

4. Sensitivity analysis

To evaluate the NPV variations, the sensitivity analysis selected critical variables such as the electricity selling price, the biopesticide selling price, the biopesticide production quantity and the discount rate.

Regarding the selling price of the electricity produced, the sensitivity analysis examined the effect of moving from the subsidized price or incentivized price (0.11 €$\,$kW$\,$h^{-1}) (Gestore del Servizio Elettrico (GSE), 2019a) to the zonal price or non-incentivized price (0.042 €$\,$kW$\,$h^{-1}) (Gestore del Servizio Elettrico (GSE), 2019b). An incentivized price increases the revenue given by the sale of the electricity produced, so it can influence the result of the calculated indicators. In S0 and S1, the absence of subsidy implied a critical variation of the FNPV

(− 1.34 % and − 20 % respectively) which demonstrated the importance of this financial/economic tool for business profitability. Regarding the economic analysis, the variation of the ENPV resulted in − 8 % in S0 and − 21 % in S1. Regarding S2, the variation of the selling price of the electricity produced was not a critical variable for both FNPV and ENVP, as bio-pesticides covered most of the revenues.

Since the selling price of biopesticides (21.28 € kg^{-1}) (Fitoitaly, 2021), played a considerable role in terms of revenues, this also due to the significant quantities produced, they were taken into account as characterizing parameters for the sensitivity analysis. The analysis assumed an expansion of the related market in the next 20 years in Italy. It is therefore expected an increase in competition with a consequent reduction in the selling price of these bio-products. For this reason, we assumed a 50 % price reduction. In S2, our results showed a variation of − 59 % for the FNPV and − 56 % for the ENPV. The difference between FNPV and ENPV was due to the social benefits (positive externalities due to net-negative GHG emissions) included in the economic analysis, which compensated for the reduction of price. Regarding the total quantity of biopesticide produced (around 52 t) through SSF, as the actual market demand of these bio-products is unknown and to avoid that the product remains unsold, the analysis considered it reduced by 30 % in S2. Our results confirmed the importance of this variable as a 30 % reduction in biopesticide production would lead to a − 82 % reduction in FNPV and − 78 % in ENPV.

Finally, the social discount rate used in the financial analysis was reduced from 6 % to 5 %, as in the economic one it was directly defined by the European Union (European Commission, 2014). This parameter turns out to be a very random variable when implementing CBAs and for that reason can affect the performance of the NPV indicator (Almansa and Martínez-Paz, 2011). As expected, a reduction in the social discount rate led to a critical change in the FNPV in all three scenarios (− 1 % for S0, − 27 % for S1 and around 9 % for S2), as the closer the discount rate was to zero, the more important the future period was attributed.

5. Conclusions

The small-scale system for the treatment of biowaste, as that analyzed in Dolina, represents an interesting option to recover the available energy and resources from the urban biowaste, particularly implementing S2. The contribution of avoided emissions thanks to the MAP substitution (S1 and S2) was supported by the digestate suitability to be used as organic fertilizer.

The impact category FRS contributed to outbalance the impact from conventional heat and electricity generated and displaced from the grid, respectively around − 51 and − 18 kg oil eq tww^{-1} in all scenarios.

For S2, the avoided emissions associated with the production of biopesticide through the application of the SSF unit presented the highest contribution to net-negative in all categories analyzed.

Based on results obtained and in agreement with previous studies, the AD system confirmed itself to be a valid option for the treatment of biowaste, playing a significant role in future FW treatment systems. However, the best environmental performances may be achieved by integrating digestate post-treatment systems such as centrifuge and reverse osmosis for nutrient recovery and the SSF to produce high-quality bio-based products (e.g., bio-pesticides). This was in line with the positive assessment given by the eCBA for S2.

The financial analysis showed a positive FNPV for S2 (54,710 € t ww^{-1}), while the project's profitability was unsustainable for S0 and S1 (− 7782 and − 7200 € t ww^{-1}). In addition, when the socio-environmental aspects associated with the accounting of the benefits of the project were included, such as the climate change mitigation effects and the production of bio-products, an improvement in the ENPV was observed in S2 (67,218 € t ww^{-1}). This highlighted how the project created social benefits for the community. In general, the study demonstrated how the progressively add of solutions to the plant configuration, through targeted investments, permits improvements in environmental and economic-financial results.

However, the sensitivity analysis showed the strong dependence of project profitability on bioproduct revenues. This suggested the development of a further analysis of the evolution of the biopesticides market, particularly in the Dolina area. In this regard, it would be interesting to assess how the cost of biopesticide varies as demand changes, as in the present study an average value was assumed for the Italian territory. Since Dolina is in strongly agricultural area, it could be assumed that in the future as the competitive market expands, the price of biopesticide may decrease. The same local market prices for renewable energy produced (mainly electricity) from the biowaste treatment would highly determine the profitability of technologies that generate these products.

The institutional policy framework has been assessed in Dolina to identify potential policy levers to support circular bioeconomy solutions. If consistently implemented, the Law Decree 116/2020 is expected to introduce important modifications to enhance the circularity and valorization of bioresources in the Italian regions, provinces and cities. However, the subsequent implementation of appropriate economic instruments is only recommended and therefore scarcely employed. This is reflected in the Dolina municipality where a specific plan for organic waste management does not yet exist, and proper economic instruments among which the ones proposed by the Law Decree 116/2020 need to be introduced to promote organic waste reduction and the efficient valorization and use of bioresources.

The LCA and eCBA analyses allowed to reveal several policy hotspots for policy intervention to support the DECISIVE implementation in Dolina, as the need to remove environmentally harmful incentives for mineral fertilizers and chemical pesticides. To ensure the economic viability of local BCE in the long run, the possibility of implementing market-based instruments such as payment for ecosystem services would be beneficial for the local communities to receive monetary benefits and support their activities.

CRediT authorship contribution statement

Morena Bruno: Methodology, Investigation, Validation, Writing – original draft, Writing – review & editing. **Michele Marini:** Methodology, Investigation, Validation, Writing – review & editing. **Elisavet Angouria-Tsorochidou:** Conceptualization, Methodology, Validation, Writing – review & editing. **Federico Pulselli:** Methodology, Supervision, Writing – review & editing. **Marianne Thomsen:** Conceptualization, Methodology, Supervision, Writing – review & editing, Funding acquisition.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.clwas.2022.100021.

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