Life cycle assessment indicators of urban wastewater and sewage sludge treatment

Elvira Buonocore[a], Salvatore Mellino[a,1], Giuseppe De Angelis[b], Gengyuan Liu[b,c], Sergio Ulgiati[a,b,c]

[a] *Department of Science and Technology, Parthenope University of Naples, Centro Direzionale-Isola C4, 80143 Napoli, Italy*

[b] *State Key Joint Laboratory of Environment Simulation and Pollution Control, School of Environment, Beijing Normal University, Beijing 100875, China*

[c] *Beijing Engineering Research Center for Watershed Environmental Restoration & Integrated Ecological Regulation, Beijing 100875, China*

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Abstract

The world is facing a water quality crisis resulting from continuous population growth, urbanization, land use change, industrialization, unsustainable water use practices and wastewater management strategies, among others. In this context, wastewater treatment (WWT) facilities are of vital significance for urban systems. Wastewater management clearly plays a central role in achieving future water security in a world where water stress is expected to increase. Life cycle assessment (LCA) can be used as a tool to evaluate the environmental impacts associated to WWTPs and improvement options. In this study, LCA is applied to compare the environmental performance of different scenarios for wastewater and sludge disposal in a WWT plant located in Southern Italy. The first scenario (BAU, Business As Usual) is based on the present sludge management performed within and outside the case-study plant: after mechanical treatment, dewatered sludge is transported by truck to a landfill for final disposal, while treated water is released to a river. The second scenario (B) assumes a partially circular pattern, with anaerobic fermentation

[1] Salvatore Mellino, Corresponding Author, salvatore.mellino@gmail.com

of sludge to biogas, biogas use for electricity and heat cogeneration, integrated by additional thermal energy from previously recovered waste cooking oil (WCO), electricity and heat feedback to upstream WWT steps (including sludge drying), and final disposal of dried sludge to landfill and water to river. The third scenario (C) suggests an improved circular pattern with gasification of the dried sludge to further support heat and electricity production (with very small delivery of residues to landfill). The fourth scenario (D) builds on the third scenario in that the volume of treated wastewater is not discharged into local rivers but is partially used for fertirrigation of *Salix Alba* fields, whose biomass is further used for electricity generation. In doing so, the water P and N content decreases and so does the water eutrophication potential. Finally, a renewable scenario (E) is built assuming that the electricity demand of the WWT plant is met by a green electricity mix, for comparison with previous options. The most impacted categories in all scenarios result to be Freshwater Eutrophication Potential (FEP) and Human Toxicity Potential (HTP). Increased circularity through recycling in scenarios B and C reduces the process contribution to some environmental impact categories such as Global Warming Potential (GWP) and Fossil Depletion Potential (FDP), but does not provide significant improvement to FEP. Fertirrigation in scenario D lowers FEP by about 60% compared to the BAU scenario. Furthermore, HTP is lowered by almost 53%. Finally, other options are discussed that could be also explored in future studies to evaluate if and to what extent they could further improve the overall performance of the WWT plant.

1. Introduction

Global population is expected to exceed nine billion people by 2050. Population increases are expected to further increase water usage and wastewater production (Corcoran et al., 2010). The world is facing a water quality crisis resulting from continuous population growth, urbanization, land use change, industrialization, food production practices, increased living standards, unsustainable water use practices and wastewater management strategies. Wastewater has a direct impact on the biological diversity of aquatic ecosystems and its inappropriate management is capable of disrupting the fundamental integrity of life support systems, on which a wide range of sectors, from urban development to food production and industry, depend (UNWATER, 2016).

In this context, wastewater treatment (WWT) facilities are of vital significance for urban systems. It has been acknowledged that wastewater management clearly plays a central role in achieving future water security in a world where water stress is likely to further increase (OECD, 2012). While being crucial for pollutants removal and reusable water supply, WWT consumes resources and triggers environmental emissions during a plant lifetime (Shao et al., 2014). Urban wastewater management requires large material, energy, economic and technological investments for the construction and operation of treatment plants. Energy consumption in WWT plants and the related greenhouse gas (GHG) emissions are also steadily increasing due to strict treatment requirements.

A crucial aspect of WWT is represented by the management of sewage sludge. Sludge is an unavoidable by-product of WWT and may hold many toxic substances such as pathogens, heavy metals and organic contaminants, which can cause serious environmental pollution. The management of this by-product is still a challenge especially in developing countries, due to the lack of clear regulation, lack of a methodology for selecting a suitable sludge management system and high investment and operation cost for refurbishing (upgrading) old WWT facilities.

Given the need to achieve long-term sustainability, the objectives of urban water systems need to go beyond the protection of public health and receiving bodies, and also focus on strategies to reduce the impacts on natural resources, to optimize the use of energy and water and reduce waste generation. Urban systems should adopt innovative approaches to wastewater management to maximize the recovery of useful materials and/or energy and minimize emissions releases. It is critical for decision-makers to adopt appropriate urban development strategies so that cities can move toward sustainable development (Dong et al., 2016).

A step ahead toward more sustainable procedures requires the identification of management routes capable of maximizing recycle and recovery benefits through low energy impact systems and development of operational systems appropriate to local circumstances (Spinosa et al., 2011). The optimization of system processes, the upgrade to more efficient technologies, and the improvement of energy management, and energy generation within the WWT plants (i.e., sludge digestion with biogas production and reuse, sludge gasification for syngas generation and use) are possible ways to lower energy consumption and environmental impacts as well as to achieve energy selfsufficiency.

Nonetheless, it is crucial to evaluate the effectiveness of such implemented options in terms of reduction of resources consumption, waste, and emissions. Indicators of efficiency and environmental performance are fundamental to marking progress toward more sustainable patterns of human development (Brown et al., 2012).

Life cycle assessment (LCA) is a valuable tool that can be used to evaluate the environmental impacts associated to WWT plants (Guest et al., 2009). LCA investigates the environmental impacts of systems or products from cradle to grave throughout the full life cycle, from the withdrawal, refining and supply of materials and fuels, through the production and operation of the investigated objects, to their final disposal or recycling (Rebitzer et al., 2004). LCA provides a comprehensive set of environmental indicators that can be interpreted as impact indicators at global and local level. This set of indicators offers information about potential or realized effects of human activities on environmental phenomena of concern thus resulting crucial for assessing the environmental sustainability of human-driven systems (McBride et al., 2011).

Several LCA studies were conducted to assess the environmental impacts caused by WWT systems as reviewed in Corominas et al. (2013). Published LCA studies about WWT plants deal with the energy consumption, GHG emissions of existing plants as well as the potential energy and GHG emission benefits that can be achieved by introducing new alternative technologies (Corominas et al., 2013). LCA has been often applied to evaluate different types of conventional WWT plants (Hospido et al., 2004; Pasqualino et al., 2009; Rodriguez-Garcia et al., 2012). Other LCA studies explored non-conventional WWT techniques (i.e., constructed wetlands, biological filters and sand filtration systems) and new designs of WWT as feasible alternatives with lower environmental impacts compared to conventional technologies (Machado et al., 2007; Yildirim and Topkaya, 2012; Bisinella de Faria et al., 2015).

Several studies were more focused on the LCA of sludge management strategies. Some studies compared alternative options for sludge treatment inside the WWT plants (Hospido et al., 2005; Peregrina et al., 2006; Cao and Pawlowski, 2013) while others suggested and compared alternatives for sludge management also outside the WWT plants (Houillon and Jolliet, 2005; Valderrama et al., 2013).

In this paper LCA is used to compare the environmental performance of different scenarios for sludge management in a WWT plant located in the municipality of Nocera Superiore, in the province of Salerno, Southern Italy. The different scenarios aim at decreasing the amount of sludge disposed of in landfill as well as at increasing the energy efficiency of the different process steps via increased recycling of still usable waste resources. The alternatives chosen have been selected according to the potentialities of the investigated WWT plant and those of the area where the WWTP evaluated is located.

2. Material and methods

2.1. *The methodological framework*

This study was performed by applying the Life Cycle Assessment method (LCA) to empirical foreground data collected on-field by the authors. LCA is a tool for assessing the potential environmental impacts and resources used throughout a product's lifecycle, from raw material acquisition, via production and use phases, to waste management, from the so-called "cradle-to-grave" perspective (ISO 14040, 2006; ILCD, 2012). All activities and processes result in environmental impacts due to consumption of resources, emissions of substances into the natural environment, and other environmental exchanges. LCA allows technology comparisons in terms of environmental burden, providing valuable insights about the environmental performance of different technologies across categories through the development of life cycle indicators. Although developments of the tool continue to be achieved, International Standards of the ISO 14000 series provide a consensus framework for standardized LCAs (ISO 14040, 2006; ISO 14044, 2006). The ILCD Handbook (ILCD, 2012), stemming from the ISO 14040-44 standards, confirms the importance and the role of LCA as a decision-supporting tool in contexts ranging from product development to policy making. The Handbook provides clear and goal-specific methodological recommendations, specific terminology and nomenclature, an accurate verification and review frame other supporting documents and tools. LCA provides a large set of environmental indicators. Currently, a large number of indicators can be found in the LCA literature. They refer to different types of indicator: life cycle impact indicators (LCI), midpoint life cycle impact assessment (LCIA) indicators, and endpoint LCIA indicators. Selected LCI flows can be useful in tracking the quantity of flows, e.g. the use of secondary energy throughout the product's life cycle.

These are not direct impact indicators, but they can be useful for the interpretation phase of any LCA study. Midpoint LCIA indicators (or potential indicators) make it possible to characterize different environmental problems, such as climate change, ozone depletion, photochemical ozone formation, acidification, eutrophication and resource depletion. End-point LCIA indicators refer to actual damage categories, such as damage to resources, damage to human health, and damage to the ecosystem.

2.2. *Research goals*

The present situation and legislation in Europe, the state of the art of the available technologies, the development of circular economy concepts, and the achieved maturity of the LCA method, make the implementation of new and sustainable strategies for wastewater and sludge management urgent, crucial and feasible, within a Circular Economy and Technology framework. With this aim, the WWTP of Nocera Superiore located in Southern Italy was chosen as case-study. This choice was driven by the characteristics of this WWTP and the sludge management currently applied inside and outside the plant. Nocera Superiore WWTP is a modern and centralized plant exploiting the most wide-spread technology in Europe, i.e. the advanced activated-sludge process. On the other hand, the operation of such systems is cost and energy intensive, mainly due to the aeration and sludge treatment associated processes. Additionally, nevertheless the sludge treatment line includes dynamic thickening, belt press dewatering, anaerobic digestion with biogas recovery and heat drying, the sludge treatment in this WWTP is poorly performed. This is due to the fact that anaerobic digestion and heat drying treatment steps are presently not in operation for technical and administrative reasons. The resulting wet sludge cannot be disposed of within Campania Region due to environmental concerns and specific legislation. For this reason, the wet sludge is transported to Puglia Region for disposal in a controlled sanitary landfill (the average transportation distance is 200 km), causing further environmental and economic costs. For these reasons, Nocera Superiore WWTP is particularly suited to be chosen as case-study, where evaluating the possible environmental benefits due the implementation of new wastewater and sludge management strategies aimed at decreasing the amount of waste disposed of as well as at increasing the energy efficiency of the different process steps via increased recycling of still usable waste resources.

Therefore, this study is conducted to answer the following research questions:

- What are the environmental impacts of the "Business-As-Usual (BAU) scenario"?
  - What are the system's hotspots?
  - Are there circular (reuse, recycling) alternatives to BAU?
  - If so, are they feasible and less impacting? What are their costs and benefits?

In light of this, the main research objectives are:

- To evaluate the environmental performance of the business-asusual (BAU) scenario of Nocera Superiore WWTP. This scenario is based on the actual wastewater and sludge management currently performed inside and outside the case-study WWTP.
- To compare the environmental performance of the BAU scenario with alternative scenarios for wastewater and sludge disposal. In fact, once the "hotspots" throughout the entire Life Cycle are identified, a consequential LCA can be carried out in order to evaluate possible environmental benefits provided by the proposed alternative scenarios.
- To propose improved management strategies to reduce the environmental impacts associated to wastewater treatment and sludge management.

2.3. *Scope definition*

2.3.1. *Description of the investigated scenarios*

Four scenarios for wastewater and sewage sludge treatment are considered in this study (Fig. 1a–d). The first scenario (scenario A, business-as-usual, hereafter BAU) is based on the WWT processes actually performed in the WWT plant of Nocera Superiore: after mechanical treatment, dewatered sludge is transported by truck to a landfill for final disposal, while treated water is released to a river. The second scenario (scenario B) assumes the anaerobic digestion of sewage sludge with biogas recovery within the WWT plant and its use for cogeneration of heat and electricity. As the investigated WWTP is already equipped with a two-stage mesophilic digester, the mesophilic fermentation of sludge was chosen as the technology to be evaluated. Anaerobic digestion consists of a series of biological processes in which microorganisms break down biodegradable material in the absence of oxygen. In such a process biogas is produced in anaerobic tanks where sludge is mixed and maintained at a temperature of 30–40 °C, in order to optimize bacterial activity (Jungubluth et al., 2007). The recovered biogas is then used for heat and electricity cogeneration. Electricity is fedback to the WWT process, in order to lower the huge demand for grid power. Heat is used for downstream thermal drying of digestate, in order to lower its mass and make transportation less energy expensive. Moreover, while wet sludge cannot be disposed of in Campania region, dry sludge disposal in local landfills is allowed. As a consequence, transport distance to landfill decreases to 30% of the distance in scenario BAU. Thermal drying of digestate also benefits from the use of heat from WCO collected from restaurants, hotels and agro-food industry in Campania Region. WCO is collected and transported to a treatment plant where it is mechanically pre-treated to lower the content of solid waste by means of decantation and centrifugation. The purified WCO can be directly burnt to produce energy or used as a useful feedstock for biodiesel production (Ripa et al., 2014).

In this study the recovered WCO is combusted for heating purpose, i.e. thermal drying of digestate. The amount of used WCO is assumed to be a fraction of the total WCO collected in Campania Region calculated according to the population equivalents (PE) of the WWT plant (300,000 PE). WCO covers 15% of the total energy demand of thermal drying of digestate, while about 55% is covered by the use of biogas from anaerobic digestion. The residual energy demand (about 30%) is supposed to be met by the use of purchased methane.

The third scenario (scenario C) suggests a furtherly circular pattern: the sludge is dried and the residual mass is gasified. Syngas is added to previously produced biogas for heat and power cogeneration. The heat and electricity generated are fedback to the WWT plant. Heat produced from syngas is used for the thermal drying of sludge. This feedback of heat avoids the use of the methane required in scenario B. In so doing, thermal drying of digestate is totally performed by utilizing heat produced within the WWT plant. The feedback of electricity further lowers the demand for grid power of BAU scenario. A very small residual fraction of digestate is landfilled.

The fourth scenario (scenario D) is drawn on the same assumptions as scenario C except for the final disposal of wastewater. In all previous scenarios, treated wastewater is released to a nearby river, with discharge within the law limits. Scenario D is based, instead, on a pioneering bioenergy production system investigated in Sweden by Buonocore et al. (2012), integrating wastewater treatment and willow (*Salix Viminalis*) farming. The Mediterranean climate of Campania Region is suitable for willow production. Furthermore, as pointed out by Fahd et al. (2012), by combining statistical data about available land in Campania Region in 1985 with data regarding the agricultural and polluted land and the urbanised areas in 2006, there are about 150,000 ha of marginal land abandoned or set aside since they do not provide enough income to the farmer. As a consequence, their hypothetical use for an energy-oriented system linked to the WWT would not compete with food production.

In this context, scenario D assumes that almost 50% of the treated wastewater volume is not discharged into surface waters, but it is used for irrigating willow cropped on marginal land (about 1150 ha) in the same area where the WWT plant is located. The amount of wastewater needed is estimated by taking into account land availability, nutrients content in wastewater, and water and nutrients requirements of willow during the growth season (May–October) (Guidi et al., 2008). The residual amount of treated wastewater is assumed to be released to a nearby river.

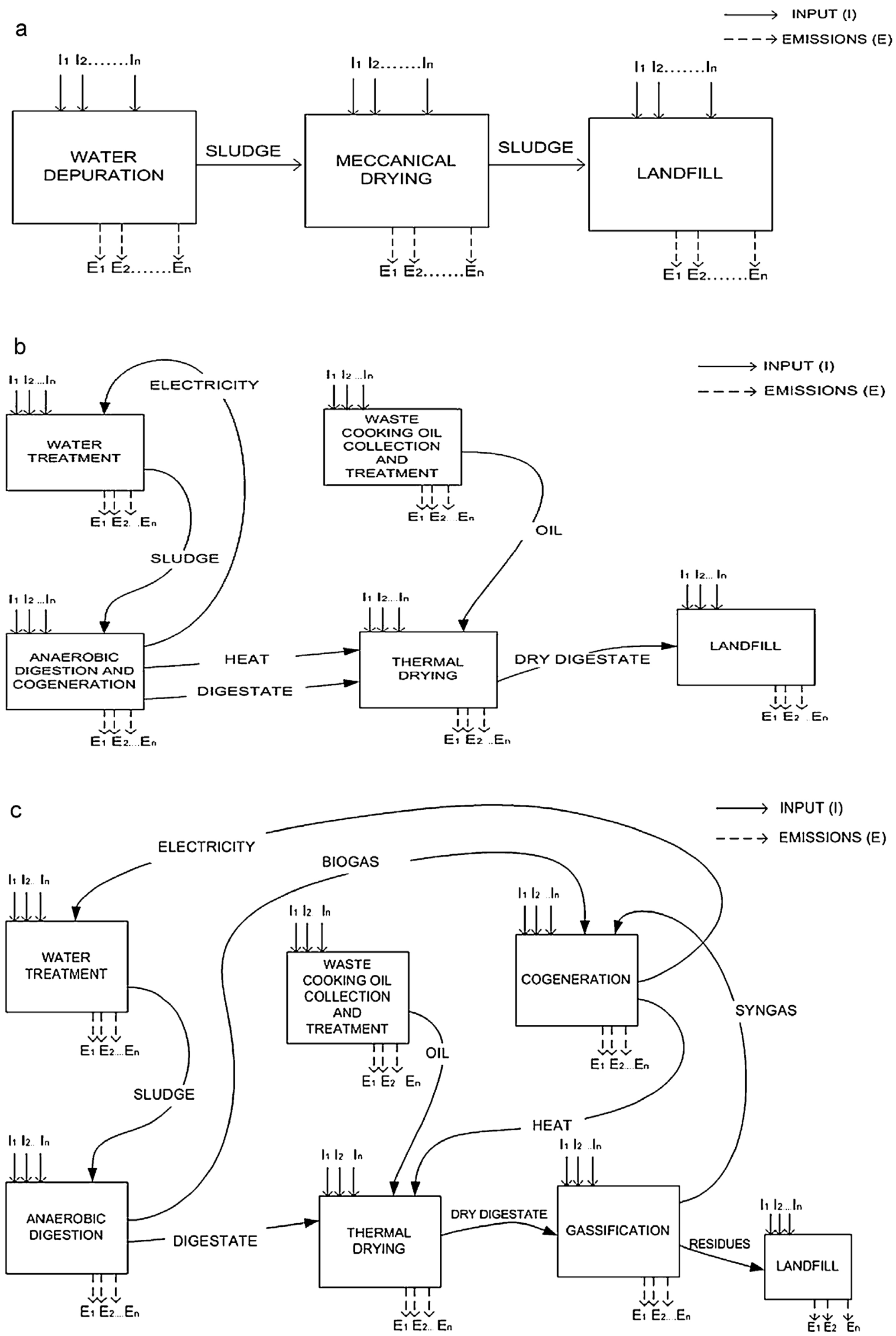

Fig. 1. Flow diagrams of scenarios investigated for sewage sludge treatment: (a) scenario A, business as usual – BAU; (b) scenario B, (c) scenario C; (d) scenario D. Scenarios B–D differ from scenario A according to increased implementation of circular patterns (recycling of still usable energy content of sludge or external waste resources).

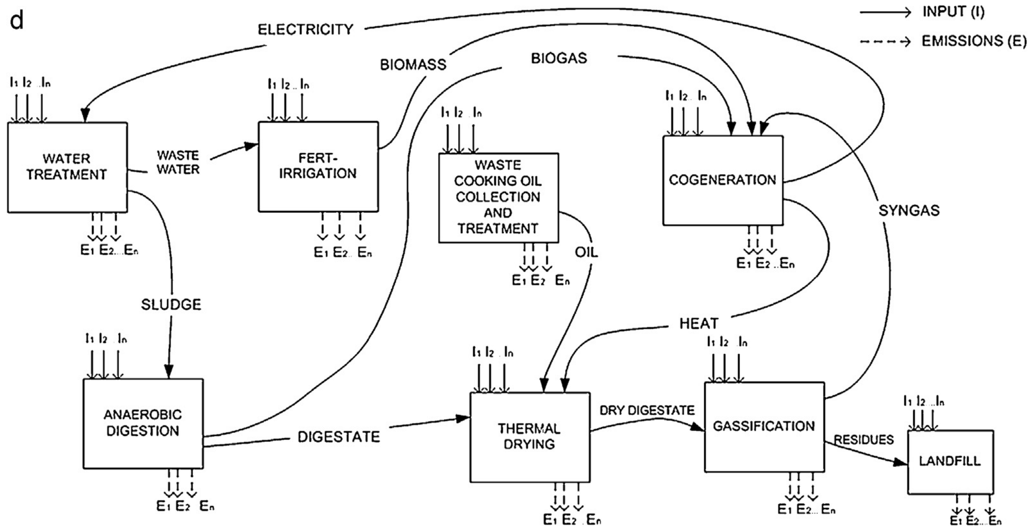

Fig. 1. (*Continued*)

Since willow can uptake 75–95% of nitrogen and phosphorus in wastewater, the annual wastewater load can easily meet the requirements of willow in terms of water and nutrients. Irrigation with nutrient-rich water would promote plant growth, thus resulting in high biomass yield (7.2 t dry mass/ha). Willow is harvested and delivered to a Combined Heat and Power plant for cogeneration of heat and electricity. A fraction of the generated electricity (30%) is supposed to be fedback to the WWT plant thus preventing the demand for grid power. The remaining fraction of electricity generated by the CHP plant (around 70%) would be supplied to local industrial and/or domestic users and it is considered as an avoided burden to the regional system in which the WWT system is embedded. In order to include the benefits provided by the virtuous use of local biomass for electricity generation, the scale of interest is expanded to include the entire regional area. This choice allows to account for the advantages due to the electricity that is not fed back to the plant and that would not be considered in the results if only the plant scale is considered.

Furthermore, scenario D is compared with a renewable scenario (scenario E) assuming that the electricity used in the BAU scenario is met by renewable sources. The assumption is based on data available from the Enel Green Power that is a society of the Enel Group developing and managing energy generation from renewable sources at a global level and present in Europe, Americas, Asia and Africa (http://www.enelgreenpower.com/en-GB/).[1][2] The "Enel Green Power" mix adopted in this scenario is based on the renewable power installed in Italy. It includes 49.9% of hydroelectric, 23.8% of geothermal, 23.7% of wind and 2.6% of solar.

The Ecoinvent 2.2 database is used for relevant background data of anaerobic digestion, syngas production, WCO combustion, power cogeneration and irrigation processes. The ReCiPe midpoint method[3] was chosen among the LCIA methods. The method allowed to assess the environmental impacts in different impact categories: climate change (GWP), fossil depletion (FD), freshwater eutrophication (FEP), human toxicity (HTP), particulate matter formation (PMFP), photochemical oxidant formation (FOFP), and terrestrial acidification (TAP). The method provides characterization factors to quantify the contribution to impact categories and normalization factors to allow a comparison across categories. Normalization is also performed by using the Recipe Midpoint H normalization factors (Sleeswijk et al., 2008).

2.3.2. *System function and functional unit*

The definition of the functional unit (FU) is a crucial issue for LCA studies. In this study the treatment of 1000 m^3 of wastewater was chosen as functional unit. (FU). All materials, emissions, cost, energy consumption, and recovery levels are referred to this amount of treated wastewater.

Furthermore, the investigated WWTP was designed to treat about 3.0 10^7 m^3 per year, while the actual volume of wastewater treated over the considered period was 1.1 10^7 m^3. This is due to the fact that some areas of the Municipalities served by the plant are still not connected to the local sewerage system.

2.3.3. *System boundaries*

This LCA analysis can be defined as an expanded "gate to gate" study, since the perimeter fences of the investigated WWTP were set as the physical system boundaries of the directly analyzed construction and operation phases, while for the

[2] Enel is a multinational manufacturer and distributor of electricity and gas (https://www.enel.it).

[3] http://www.lcia-recipe.net/

processes production of chemicals, electricity, construction materials, waste disposal, transportation, anaerobic digestion, gasification and fertirrigation, the system boundaries were expanded by using case studies from the *Ecoinvent* database and scientific literature. Fig. 2 illustrates the system boundary for the base case model. The system boundary includes as a first step the delivery of wastewater to the plant and as the last step the release of wastewater effluent. In light of this, the performed LCA study covers the actual processes associated to wastewater treatment, including:

- the construction phase and production of construction materials,
- the operation and maintenance (O&M) phase,
- the treatment performed within the WWTP, the transportation and final disposal of sludge, grit and screening waste.

Finally, the decommissioning phase is excluded from this study due to insufficient data pertaining to such a phase.

2.3.4. *Assumptions*

The main assumptions made in this study are listed as follows: *Construction Phase:* according to the performed literature review, a WWTP life-span of 30 years was assumed. The construction of the sewer system within the Municipalities served by the investigated plant was not taken into consideration.

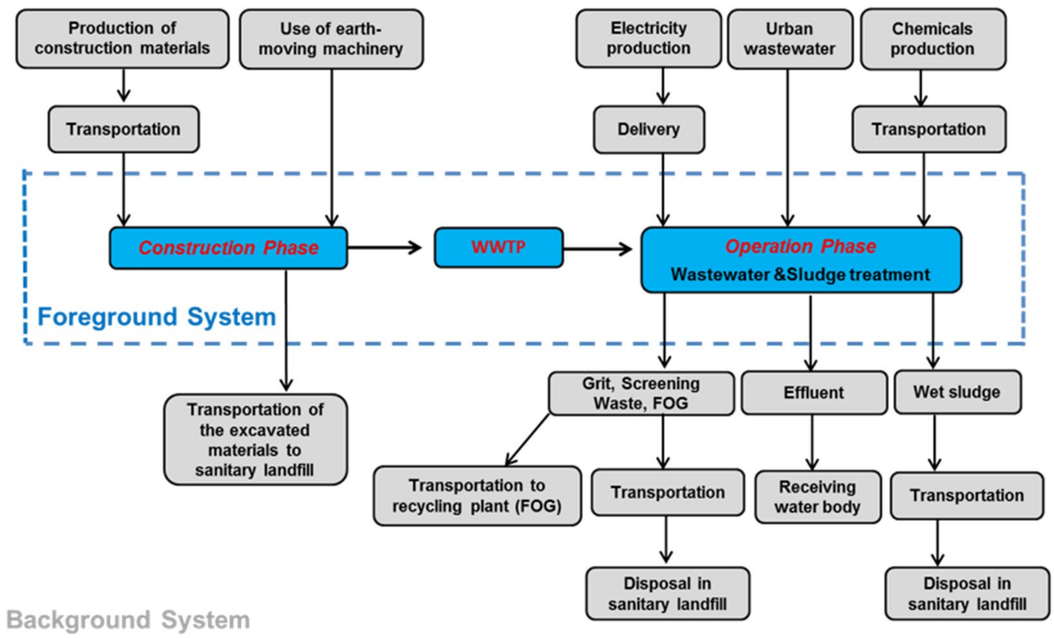

Fig. 2. Physical boundaries of scenario BAU.

Table 1

Total contribution of the four scenarios to selected impact categories. Values are referred to 1000 m^3 of wastewater treated (functional unit).

Impact category	Reference unit	Scenario A	Scenario B	Scenario C	Scenario D
Climate change—GWP100[a]	kg CO_2-Eq	620.64	563.92	404.27	1.93
Fossil depletion—FDP	kg oil-Eq	101.95	92.39	64.95	−43.90
Freshwater eutrophication—FEP	kg P-Eq	0.84	0.83	0.81	
Human toxicity—HTPin[b]	kg 1,4-DCB-*E*q	198.70	160.82	108.76	93.17
Particulate matter formation—PMFP	kg PM10-Eq	0.49	0.40	0.37	0.12
Photochemical oxidant formation—POFP	kg NMVOC	1. 18	0.90	0.77	0.41
Terrestrial acidification—TAP100[a]	kg SO_2-Eq	1.25	1.02	0.98	−0.020

[a] Characterization factors are expressed as potential impact for a time horizon of 100 years.

[b] Characterization factors are expressed as potential impact for an infinite time horizon.

Direct GHG Emissions: according to the IPCC Guidelines for National Greenhouse Gas Inventories, the direct emissions of CO_2 due to wastewater and sludge treatment were not accounted for, since they are considered as of biogenic origin (IPCC, 2007). With regard to the direct emissions of CH_4 and N_2O, the emission factors of 0.0053 g CH_4*g COD_{inflow}*$^{-1}$ and 28 g N_2O N*kg TKN^{-1} were applied, respectively (GWRC, 2011; Daelman et al., 2013).[4]

Production of Paracetic Acid (PAA): as there is no available data on the production of PAA in the *Ecoinvent* database, the production processes of acetic acid (CH_3COOH) and hydrogen peroxide (H_2O_2) were considered, assuming that the production of 1 kg of PAA requires 0.45 kg of CH_3COOH, 0.79 kg of H_2O_2 and 0.28 kg of water.

Production of Polyelectrolyte: since the *Ecoinvent* database has no data about the production of Polyelectrolyte, the production process of acrylonitrile was considered, which is the precursor of the polyelectrolyte used in the investigated WWTP.

Waste Disposal: in order to evaluate the environmental burdens related to sludge disposal, the *Ecoinvent* process "*disposal, urban solid waste, 22,9% water, to sanitary landfill*" was chosen. The process was conveniently modified by including the environmental impacts due to the use of sludge handling equipment within sanitary landfill, assuming a diesel consumption of 1.16 L*ton handled sludge^{-1}. [60]. With regards to grit disposal, the process "*disposal, inert waste, 5% water, to inert material landfill*" was considered, while the processes "*disposal, plastics, mixture, 15,3% water, to sanitary*

[4] TNK: Total Kjehldahl Nitrogen is defined as the sum of organically bound nitrogen and ammonia, while total nitrogen is the combination of organic nitrogen and inorganic nitrogen (NH_3, NO_3, NO_2).

landfill" and "*disposal, paper, 11,2% water, to sanitary landfill*" were used to estimate the environmental burdens caused by the disposal of screening waste, assuming that such a waste is only composed by paper and plastic.

Sludge Gasification (scenarios C and D): in the absence of sewage sludge gasification-specific data in the *Ecoinvent* database, the process "*synthetic gas, at fixed bed gasifier (wood)*" was chosen to estimate the environmental impacts due to the gasification of digested sludge, assuming a syngas production of 1.92 Nm^3*kg dried $sludge^{-1}$ and a Low Heat Value of 5.2 MJ*m^{-3}.

2.3.5. *Data quality*

Data quality details are provided in this section. Regarding the temporal coverage, foreground data on the operation phase refer to the period November 2012–October 2013. For the background data, preference was given to the latest representative data.

In terms of geographical coverage, foreground data refer to the area directly involved by the investigated process, while background data refer to European and Italian case-studies (whenever possible) or geographical areas with similar climatic conditions.

Completeness is guaranteed since all the flows which could be realistically investigated within the constraints imposed by available data and current knowledge were considered.

Whenever foreground data were not available, the study drew upon background data as representative as possible (from a technological, temporal and geographical point of view) of the investigated processes. Finally, all the made assumptions, data sources and applied calculation models are clearly identified, in order to allow an independent practitioner to reproduce the results of this LCA study and any conclusions or recommendations drawn. The calculation procedures and raw data processing are described in the supplementary material.

3. Results

The life-cycle contribution per 1000 m^3 of wastewater (FU) to selected impact categories in the investigated scenarios is displayed in Table 1. The characterized impacts of the scenarios are shown as percentages in Fig. 3, where the potential improvements achievable in scenarios B–D are compared to the results of scenario A (put conventionally at 100%).

Results show that the contributions to the chosen impact categories decrease in all the scenarios when compared with the BAU scenario. Scenarios B and C reduce the contribution to the Global Warming Potential (GWP) by 9% and 35% respectively, while the contribution to Fossil Depletion Potential (FDP) is lowered by 9% and 36%. The contribution to other impact categories, as Human Toxicity Potential (HTP), Particulate Matter Formation Potential (PMFP) and Terrestrial Acidification Potential (TAP), is also lower compared to the BAU scenario (Fig. 3).

The Freshwater Eutrophication Potential (FEP) does not substantially change in scenarios B and C while it results 53% lower in scenario D compared to the BAU scenario.

Scenario D is also capable of reducing HTP by almost 60%. All the other categories also benefit from this scenario (Fig. 3).

Fig. 4 shows the normalized impacts of the four scenarios. The most impacted category results to be the FEP in all the scenarios. The second most impacted category is HTP. Still, scenario D results the most valuable option for reducing the contribution to both these impact categories.

The characterized impacts of scenario D are also compared to those of the renewable scenario—scenario E (put conventionally at 100% in Fig. 5). Results show that scenario D has higher potential for abating the contribution to environmental impacts categories even when a green mix is assumed to be used within the WWT plant. The comparison between normalized impacts of scenario D and the renewable scenario (Fig. 6) confirms that the scenario D would be more capable of reducing the impacts in most of the selected categories (i.e., FEP and HTP) compared to the choice of an electric green mix for powering the WWT plant (that is, anyway, a better choice than the BAU scenario).

4. Discussion

4.1. *Environmental performance of alternative scenarios*

The investigated scenarios are oriented towards achieving the energy selfsufficiency of the investigated WWTP, decreasing other impacts not directly involving energy supply, and at the same time, sensibly reducing the amount of waste to be transported and disposed of.

Energy production from sewage sludge (i.e. biogas and syngas production) is an important energy source, capable to sensibly reduce plant's dependency on fossil resources, thus mitigating its energy-related environmental burdens. To this end, the combined application of anaerobic digestion, dehydration and gasification has proved to be one of the most promising technologies in terms of both energy recovery and sludge mass reduction (Lacroix et al., 2014; Cao and Pawlowski, 2013). The latter gain is also noteworthy, as the delivery and disposal of sludge have resulted to be among the most important contributions to the environmental profile of WWTPs (Corominas et al., 2013).

Also the reuse of recovered WCO within the WWTP represents an additional step towards closing the local resource circle by linking the treatment of different kinds of municipal wastes, i.e. wastewater, its by-products and waste cooking oil generated from households and restaurants (of course, WCO inclusion requires a boundary expansion to also account for the WCO collection and treatment). In the last scenario, additional interesting benefits are coupled with the possibility to reuse wastewater for irrigation of energy crop fields, in order to provide biomass for energy purpose. This solution manages not only to further minimize plant's dependency on fossil fuels, but also to sensibly reduce the volume of treated wastewater to be discharged in receiving water bodies.

Among the investigated scenarios, the best environmental performance was achieved in scenario D that represents a circular pattern where a) sludge is not disposed in landfill but further processed to generate biogas and syngas and b) the volume of treated wastewater is not completely discharged into surface waters but partially reused for fertirrigating willow fields for biomass production and electricity generation.

The negative value for the fossil depletion and the terrestrial acidification categories resulting for scenario D are due to the avoided impacts associated to the cogeneration of heat and power (CHP) from willow biomass fertirrigation by means of nutrients in wastewater. The energy generated is much greater than the power demand of the WWT system so that the avoided impact refers to the avoided use of grid electricity by the territorial system. The latter, in fact, benefits from the surplus electricity and heat cogenerated by using willow biomass.

The contribution associated to the avoided impact needs to be carefully interpreted. The avoided impact was calculated by subtracting the environmental burden of the amount of electricity generated by the Italian mix that would not be used by the regional system. In this case, the avoided impact mainly depends on the specific electric mix (Italian electricity mix in 2013, www.terna.it). Of course, if this mix changes, results would change as well. Furthermore, these benefits can only be included if the boundary of scenario D is extended to the broader regional scale to account for fossil energy replacement.

The use of wastewater for irrigation and fertilization of willow cropped land allows non-negligible energy savings and contributes to renewable energy generation, but is only feasible if land is available at short distance from the WWT plant. This requires that WWT plant designs are made considering this option into account since the very beginning.

As a result of this "towards zero-emission" oriented production pattern, where waste generated by a process can be used and upgraded as input to support another process, the overall generation of waste and emissions decreases significantly. Such a perspective should represent a valuable option for a sustainable management of wastewater and sewage sludge.

The FEP resulted the most impacted category in all the scenarios. This finding is due to the high content in nitrogen and phosphorus in wastewater mainly deriving from human and agro-industry waste. Wastewater discharges are understood to make a significant contribution to the problems of eutrophication and scenario D seemed to be a valuable option for reducing the nutrient pollution of surface waters. The abatement of the eutrophication impacts in scenario D is due to the utilization of wastewater for growing willow crops that avoids the discharge of nutrients rich treated water into the river. However, the amount of wastewater supposed to be used for fertirrigating willow fields only amounted to about 50% of the total annual volume generated by the WWT plant.

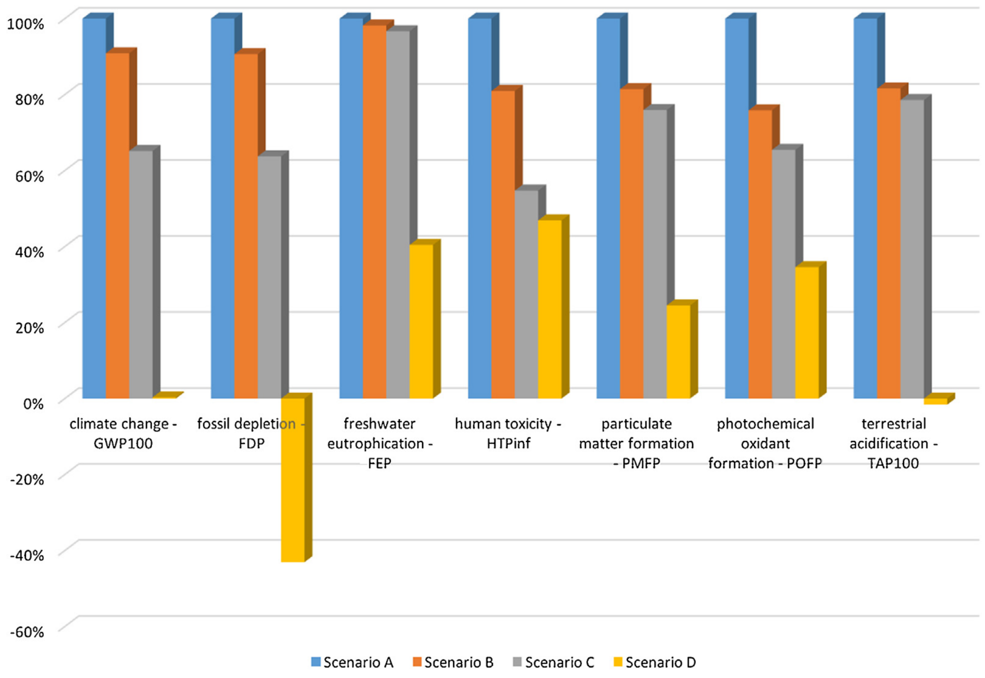

Fig. 3. Characterized impacts of the four scenarios (percentage values; data from Table 1).

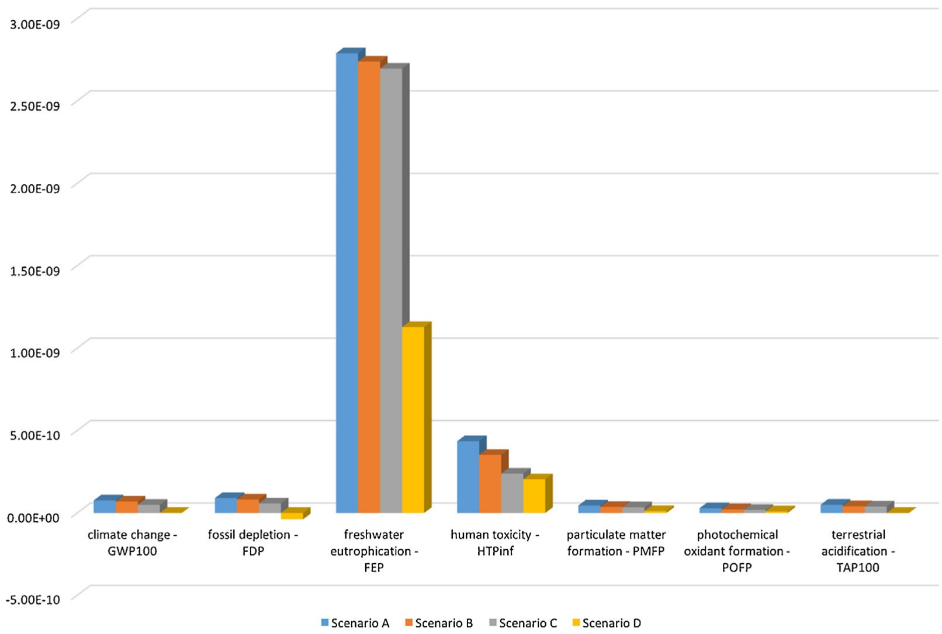

Fig. 4. Normalized impacts of the four scenarios to impact categories. Normalization factors from Sleeswijk et al. (2008).

HTP was the second impacted category. The high contribution to human toxicity is associated to sludge disposal in landfill. The contribution to the HTP decreases from scenarios A to D since their circularity allows the recycling of sludge within the

WWT plant thus reducing the amount landfilled. Scenario D resulted to be the best option also in abating the HTP burden. However, the advantage due to the generation of electricity from local fertirrigated willow biomass is partially offset by wood combustion that also contributes to HTP. In order to overcome this last problem, scenario D could be complemented by the use of an appropriate fraction of wood biomass for platform chemicals instead of combustion. Fiorentino et al. (2014) demonstrated that if wood biomass is processed in a bio-refinery context, by selecting appropriate feedstock and technology suitable for the utilized raw materials, bio-based products actually generate higher environmental and economic benefits than an energy-oriented pattern. Such alternative was also not integrated in this study (as it would require an optimization of the wood fraction allocated to CHP and the wood fraction allocated to the chemical route) but it would certainly be interesting for future studies to explore the potential mitigation of environmental impacts that could be achieved over this new pattern.

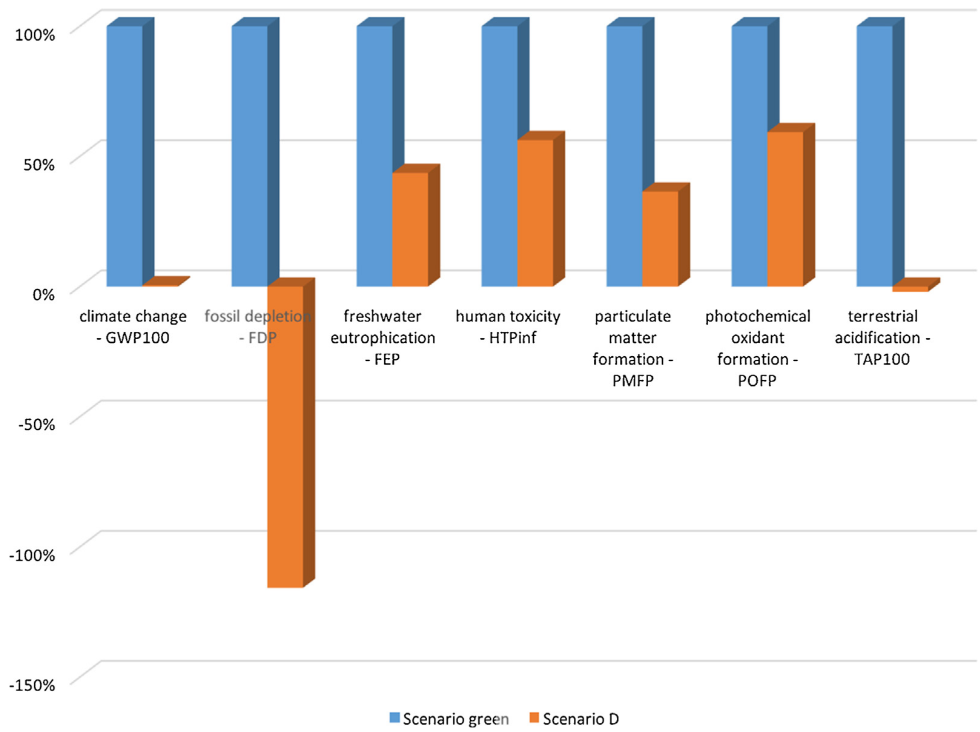

Fig. 5. Comparison between characterized impacts of the renewable scenario and scenario D.

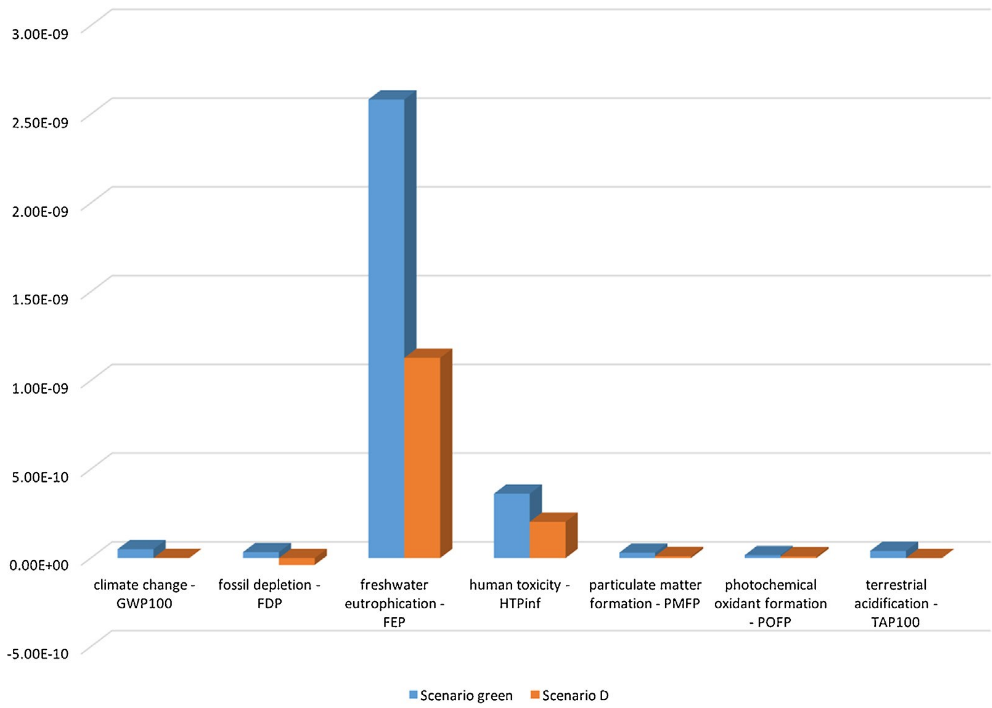

Fig. 6. Comparison between normalized impacts of the renewable scenario and scenario D.

Finally, a green electricity mix supplied by the national electric company ENEL is supposed to be used within the WWT plant (scenario E). Conventional electricity generation is a significant source of greenhouse gas emissions. The emissions from conventional electricity generation contribute to a number of serious environmental problems, including acid rain, fine particulate pollution, and climate change (EPA, 2010). Green power generates less pollution than conventional power and produces no net increase in greenhouse gas emissions, helping protect human health and the environment.

In this study the adoption of a renewable electricity mix was an important option, although not capable of significantly reducing the impact to eutrophication and human toxicity as does scenario D.

4.2. *LCIA indicators for WWT systems*

The use of LCA allowed the calculation of environmental impact indicators. The outcomes of this study can be regarded as robust, effective and appropriate since they appear to be capable of (i) relating pollutants emissions to their environmental effects, (ii) describing the environmental impacts caused by wastewater and sludge treatments with sufficient accuracy and (iii) capturing the differences in the environmental performance due to the implementation of the proposed treatment technologies. Furthermore, through the investigated case study also the main limits of LCA indicators applied to wastewater treatment systems can be addressed,

i.e. the quantification and meaning of the Eutrophication Potential, Global Warming Potential and Toxicity Potential indicators. High uncertainties are still associated with them, mainly due to incomplete regionalization of impact assessment methods, to relevant substances still not being characterized in LCA database and to uncertainty on the inventory part (Roux et al., 2010; Zang et al., 2015). In fact, neither Eutrophication nor Toxicity potential indicators can be fully assessed by LCA methods by means of the present characterization models, as they do not appear capable of capturing the local conditions as well as taking all the substances affecting the above impact categories into proper account (Roux et al., 2010; Zang et al., 2015). LCA indicators on Eutrophication provide useful insights on potential impacts but do not allow to infer the actual modification of the eutrophication level of a specific river due to wastewater discharges, simply because of the fact that the river quality upstream of the discharge point is not taken into account. Being eutrophication highly site-specific, more efforts should be put in developing and implementing new characterization models capable of taking local factors such as recipient ecosystem quality and sensitivity to nutrients emissions and emission source location into consideration (Zang et al., 2015). With regards to toxicity indicators, pathogens as well as the so-called Pharmaceuticals & Personal Care Products (PPCP) that are discharged into receiving water bodies are not currently characterized in LCA neither for human toxicity nor ecotoxicity, thus leading to underestimate such impacts (Roux et al., 2010). Toxicity-related impact categories have recently received increasing attention by means of efforts for the development of more accurate characterization models for toxicity, which should be integrated in current LCA software and databases (Zang et al., 2015). The aforementioned shortcomings are of particular significance for WWTP-related LCA studies, since, historically, the collection and treatment of wastewater has been performed to protect human health and prevent ecosystem eutrophication. Concerning Global Warming Potential, it is crucial to calculate more accurately the direct GHG emissions, since N_2O, CH_4 and fossil CO_2 emissions from WWTPs are understood to make a significant contribution to the plants' carbon footprint, most often exceeding the contribution of indirect GHG emissions associated with electricity use, so far considered the main source of GWP impact (IPCC, 2007; GWRC, 2011; Zang et al., 2015). Some Authors have concluded that direct N_2O emissions are the largest contributor to a plant's carbon footprint by far (GWRC, 2011; Rodriguez-Garcia et al., 2012; Daelman et al., 2013), pointing out that the use of a generic emission factor to estimate such emissions from an individual WWTP is inadequate, being emissions affected by a large number of local parameters. The non-negligible uncertainty associated with N_2O emissions estimate is mainly due to the difficulty of identifying the prominent mechanisms of nitrification-denitrification processes and factors influencing such emissions (Crutzen et al., 2007; GWRC, 2011; Zang et al., 2015). With regard to fossil CO_2 emissions, some authors have pointed out that up to 20% of the biodegradable carbon present in wastewater may be of fossil origin (mainly related to detergents, cosmetics and pharmaceuticals) which is still not accounted for in Life Cycle Inventories, thus leading to underestimate the associated impacts expressed by the indicators (Rodriguez-Garcia et al., 2012; Zang et al., 2015). Furthermore, as methane is understood to make a non-negligible contribution to the overall GHG emissions from WWTPs, it is of vital importance to develop calculation models allowing to accurately estimate direct emissions from both the water and sludge treatment lines (GWRC, 2011; Bao et al., 2015). Progress in the LCA methodology capable of better addressing these still unaccounted for aspects might radically change the LCIA results of wastewater treatment systems, thus helping develop new and more comprehensive LCA indicators, that are more representative of the real environmental impacts caused by WWTPs.

5. Conclusions

Life cycle assessment, used in this study, allowed to compare the environmental performance of different scenarios for wastewater and sludge management, characterized by different degrees of recycling within the plant as well as at larger regional scale.

Results showed that the most desirable option would be a circular pattern where a) sludge is processed to generate biogas and syngas to be further combusted for the generation of electricity and heat, b) collected and refined waste cooking oil from the surrounding area is used as additional heat source, and c) wastewater is used to fertirrigate wood crops for bioenergy purposes.

The circularity adopted in this scenario decreases the overall environmental impacts of the WWT plant, allows the plant to be totally energy self-sufficient and contributes to (although small) renewable energy generation.

Treated wastewater supports biomass fertirrigation that can be used together with other bio-wastes (such as waste cooking oil) to produce energy, nulling plant's power requirement and even creating additional income through the sale of surplus energy to the local grid.

It is evident from the investigated case study that new and improved processes and technology are capable of generating opportunities for impact reduction in WWT plants, but each option needs to be carefully evaluated over the entire life cycle, according to the particular context in which the WWT plant is located.

Further improvements of the wastewater and sludge management could be implemented by adopting additional circular strategies at larger scale, after careful LCA evaluation. Results clearly show, however, that an improved wastewater treatment plant should not be considered a potential energy source (in spite of the biogas and syngas generation and additional biomass production) but instead a self-sufficient facility providing the much more important water treatment service at low or no energy cost. The biomass energy production becomes a tool for and a co-product of the abatement of water eutrophication potential, requiring a large land availability and occupation for this to happen. When marginal land is available, the WWT plant and its improved circular features may provide additional benefits, which calls for preliminary ecodesign and appropriate siting of the plant within the urbanized area that releases the wastewater and enough rural area to receive the treated water and allow biomass cropping. Finally, although LCA has proved to be a desirable tool to evaluate the environmental impacts of WWT plants, efforts are still needed to investigate some impact categories and to provide LCA indicators more adapted to the specific local context in which the WWT plant is embedded.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecolind.2016. 04.047.

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Supplementary Material (SM) for

Life cycle assessment indicators of urban wastewater and sewage sludge treatment.

Elvira Buonocore[1], Salvatore Mellino[1,*5], Giuseppe De Angelis[2], Gengyuan Liu[2,3], Sergio Ulgiati[1,2,3].

[1] Department of Science and Technology, Parthenope University of Naples, Centro DirezionaleIsola C4, 80143 Napoli, Italy

[2] State Key Joint Laboratory of Environment Simulation and Pollution Control, School of Environment, Beijing Normal University, Beijing 100875, China

[3]Beijing Engineering Research Center for Watershed Environmental Restoration & Integrated Ecological Regulation, Beijing 100875, China

1. Calculation procedures and raw data processing

The Life Cycle Inventory (LCI) is structured according to the two considered main phases of the investigated system product, i.e. construction and operation phase. Accordingly, the data needed to establish the inventory were collected for each phase.

1.1 Data collection

Data were collected independently for the construction and operation phases, thus allowing objective review of individual data sets before their contribution to the overall life cycle results is determined. Data collection was an iterative process, whereby ensuring all necessary life cycle information. Foreground data, provided by Consorzio Nocera Ambiente (local society managing the wastewater treatment plant), concern electricity and process chemicals consumptions, volume and characteristics of influent and effluent as well as the amount of waste to be transported and disposed of for the operation phase. With regards to the construction phase, foreground data refer to the identification and quantification of the construction materials, their transportation to the plant, the amount of excavated earthen materials as well as the land covered by the construction site. On the other hand, background data on production of process chemicals, construction materials, waste disposal, transportation and electricity generation were obtained from the Swiss *Ecoinvent* database.

[5] Corresponding Author, Salvatore Mellino, email: salvatore.mellino@gmail.com

Concerning the Italian electricity mix, it was updated to that of 2013 (www.terna.it). Direct GHG emissions due to wastewater and sludge management within the investigated WWTP were estimated by means of using the emission factors of 0.0053 g CH_4*g $COD_{inflow}*^{-1}$ and 28 g N_2O-N*kg TKN^{-1}, respectively (GWRC, 2011; Daelman et al., 2013). Finally, inventory data for transportation distances to sanitary landfills were calculated based on data obtained from Michelin Street Guide (www.viamichelin.it).

1.2 Data analysis

The collected data were converted to values that relate to the functional unit (1000 m^3 of treated water). The adjusted data were entered into the OpenLca LCA software and modeled into environmental inputs and outputs and then aggregated to result in two inventory tables.

1.2.1 LCI of the Construction Phase

The infrastructure components of the investigated WWTP as well as the materials needed to build them were estimated by consulting the Bill of Quantities of Nocera Superiore Plant. In this document all materials, parts and labor needed to build the plant are itemized. Collecting data for the construction phase was a detailed-oriented and time-consuming task, as it required to scrutinize the bill of quantities of a very complex plant composed of thousands of different components. For the construction phase it was considered:

- *Concrete* and *reinforcing steel* for the construction of reinforced concrete works;
- *Lean concrete* ("poor concrete") for the construction of the foundations;
- *Sand*, *gravel* and *limestone* used in the construction of civil works;
- *Plaster* used for masonry covering;
- *Chromium Steel* for the production of screens, rotostrainer, crane bridges and scrapers (sedimentation tanks), compressors, blowers, dosing pumps, storage tanks of PAA, sluice gates, piping system, screw conveyors, immersed agitators, steelworks, polyelectrolyte dosing units, metal components of the digester, conveyor belts, belt presses and storage silos;
- *Cast iron* for the production of hoist pumps, recirculation pumps and check and gate valves,
- *Aluminum* for the building coating and window fixtures;
- *Synthetic rubber (*EPDM) for the production of air disc diffuser membranes;
- *Polyethylene* for pipes production;
- *Polyvinylchloride* (PVC) for the production of pipes as well as air disc diffuser bodies;

- *PRFV* for pipe production;
- The *Land* occupied by the construction site of the investigated WWTP as well as the environmental impacts due to the use of earth-moving machineries within the site and the transportation of the excavated materials to sanitary landfill.

The LCI results for the construction phase are listed in table 1.

Input	**Flow type**	**Unit**	**Total amount**	***Amount referred to the FU***
Aluminium, production mix, cast alloy, at plant (RER)	Product	kg	38.348,4	***0,1***
Bitumen, at refinery (RER)	Product	kg	146.952	***0,4***
Base plaster, at plant (CH)	Product	kg	2.261.000	***6,6***
Cast iron, at plant (GLO)	Product	kg	1.030.000	***3,0***
Chromium steel 18/8, at plant (RER)	Product	kg	546.075,9	***1,6***
Concrete, normal, at plant (CH)	Product	m^3	37.977	***0,1***
Excavation, hydraulic digger (RER)	Product	m^3	321.314,8	***0,9***
Gravel, crushed, at mine (CH)	Product	kg	20.897.400	***60,8***
Limestone, crushed, washed (CH)	Product	kg	11.355.720	***33,1***
Occupation, construction site	Elementary	m^2*a	424.800	***1,2***
Occupation, industrial area, built up	Elementary	m^2*a	1.335.000	***3,9***
Occupation, industrial area, vegetation	Elementary	m^2*a	789.000	***2,3***
Polyethylene, HDPE, granulate, at plant (RER)	Product	kg	3243,7	***9,44E-03***
Polyvinylchloride, at regional storage (RER)	Product	kg	48.450	***0,1***
Poor concrete, at plant (CH)	Product	m^3	13.100	***3,81E-02***
Portland cement, strength class Z 52.5, at plant (RER)	Product	kg	2.482.699,5	***7,2***
Reinforcing steel, at plant (RER)	Product	kg	4.059.072	***11,8***
Silica sand, at plant (DE)	Product	kg	9.268.450,5	***27,0***
Synthetic rubber, at plant (RER)	Product	kg	8.640	***2,52E-02***
Transformation, from pasture and meadow	Elementary	m^3	70.800	***0,2***
Transformation, to industrial area, built up	Elementary	m^3	44.500	***0,1***
Transformation, to industrial area, vegetation	Elementary	m^3	26.300	***7,66E-02***
Transport, freight, rail (RER)	Prodouct	t*km	2.898.460	***8,4***
Transport, lorry 3.5-16t, fleet average (RER)	Product	t*km	2.476.684	***7,2***
Transport, lorry >16t, fleet average (RER)	Product	t*km	4.077.148,2	***11,9***
Transport, lorry >32t, EURO3	Product	t*km	239.360	***0,7***
Window frame, aluminium, U=1.6 W/m2K, at plant	Product	m^2	954,5	***2,78E-03***
Output				
WWTP	Product	N° items	1	***2,91E-06***

In this table, all the materials used for the construction of the investigated plant, their transportation to the construction site as well as the area occupied by the site are expressed (calculated) in absolute terms as well as related to the chosen Functional Unit. Concerning the latter point, results were obtained by dividing the amount of each material by the total lifetime of the plant, and then by the wastewater treated per year. The values so obtained were then multiplied by 1,000, in order to express the results in relation to the chosen Functional Unit. Finally, it was assumed that the water treated per year (for every year during the WWTP lifetime) is 11,450,610 m^3, which is the volume of treated wastewater by Nocera Superiore plant throughout the investigated year.

1.2.2 LCI of the Operation Phase

Concerning the Operation phase, the electricity and process chemicals (PAA & polyelectrolyte) consumptions, their transportation to the plant, the volume and characteristics of influent and effluent, the direct GHG emissions as well as the amount of waste to be transported and disposed of were considered. The volume of treated wastewater, the electricity and process chemicals consumptions as well as the wastes produced throughout the investigated year are listed in table 2. The values listed in the table are calculated on monthly and annual basis.

Table 2: Summary of volume of treated wastewater, electricity and process chemicals consumptions, and amount of wastes produced over the investigated period.

Month/Year	Treated Wastewater (m^3)	Electricity (MWh)	PAA (kg)	Polyelectrolyte (kg)	Sludge (ton)	Screening waste (ton)	Grit (ton)	FOG (ton)
Nov-12	626,335.0	345.6	3,375.0	175.0	205.7	0.0	0.0	0.0
Dic-12	769,495.0	413.2	2,975.0	975.0	707.7	0.0	23.0	0.0
Jan-13	713,370.0	402.4	3,250.0	1,525.0	443.0	0.0	11.9	0.0
Feb-13	880,780.0	308.6	2,400.0	125.0	264.8	0.0	0.0	0.0
Mar-13	829,850.0	435.1	3,400.0	1,375.0	919.5	0.0	32.8	0.0
Apr-13	1,079,500.0	421.6	3,475.0	1,350.0	757.8	0.0	0.0	0.0
May-13	882,450.0	421.0	3,950.0	2,000.0	913.1	13.2	49.5	59.0
Jun-13	853,390.0	415.6	3,820.0	2,525.0	683.6	7.0	61.1	0.0
Jul-13	1,157,590.0	447.9	3,720.0	2,150.0	287.2	0.0	0.0	0.0
Aug-13	1,493,430.0	553.9	3,820.0	1,825.0	590.7	0.0	0.0	0.0
Sep-13	1,080,420.0	491.4	3,970.0	2,075.0	836.8	19.2	0.0	0.0
Oct-13	1,084,000.0	458.5	3,245.0	1,750.0	900.6	27.0	30.7	0.0
Tot.	**11,450,610.0**	**5,114.8**	**41,400.0**	**17.850,0**	**7,510.4**	**66.3**	**209.0**	**59.0**
Mean	*954,217.5*	*426.2*	*3,450.0*	*1.487,5*	*625.9*	*5.5*	*17.4*	*4.9*
Std.dev.	*236,718.2*	*62.8*	*455.9*	*749,6*	*265.0*			

The sanitary landfills and the recycling plant where the different types of waste are transported, the estimated transportation distances, the average amount of waste per trip, the number of truck trips and the total waste transport (expressed in mass*distance) calculated for each kind of waste are listed in Table 3.

Table 3: Waste transport.

Destination of waste	Distance (Km)	Sludge (ton)	Grit (ton)	FOG (ton)	Screening waste (ton)	Average amount of waste per trip (ton*trip^{-1})	Number of truck trips	Waste transport (ton*km)
Recycling Plant	36.0	0	0	59.0	0	29.5	2	2,124.7
Landfill 1 (Puglia Region)	185.5	6,902.1	187.9	0	59.3	23.1	309	2,652,356.9
Landfill 2 (Puglia Region)	168.0	348.8	0	0	0	23.1	15	117,203.5
Landfill 3 (Puglia Region)	273.0	259.5	21.1	0	7.0	23.1	12	157,024.1
Total		**7,510.4**	**209.0**	**59.0**	**66.3**		**339**	**2,928,709.3**

Table 4 itemizes the process chemicals used throughout the reference year, the transportation distances from the factories to the dealer and from the dealer to the WWTP, the average amount of goods per trip, the number of trips and the total goods transport for each means of transport.

Table 4 - Transport of process chemicals

Process chemicals	Factory - Dealer distance (km)	Dealer - WWTP distance (km)	Truck average goods load (ton)	Van average goods load (ton)	Number of truck trips	Number of van trips	Truck transport (ton*km)	Van transport (ton*km)
PAA	898	12	10	0.19	1.79	94.2	16,074.2	214.8
Poly	-	12	-	0.19	-	21.9	-	496.8
Total							**16,074.2**	**711.6**

Table 5 lists the estimated direct pollutants emissions released to the receiving water body throughout the investigated period.

Table 5: Direct pollutants emissions in the receiving water body.

Parameter	Unit	Average value	Total (Kg)	Value referred to the FU (kg)
TSS	mg/l	6.2	71,451.8	6.2
COD	mg/l	32.7	398,163.4	348
BOD_5	mg/l	8.5	97,055.4	8.5
Chlorides	mg/l	117.7	1,348,217.7	117.7
Solphates (SO_4)	mg/l	35.7	408,763.9	35.7
Total phosphorus	mg/l	0.7	8,565.0	0.7
Ammonia nitrogen	mg/l	2.6	28,730.1	2.5
Nitrite-nitrogen	µg/l	0.8	9.2	8.0E-04
Nitrate-nitrogen	mg/l	8.6	98,622.9	8.6
Aluminium	mg/l	7.4E-02	847.3	7.4E-02
Cadmium	mg/l	1.0E-03	11.5	1.0E-03
Chromium	mg/l	2.9E-02	332.1	2.9E-02
Chromium IV	mg/l	1.0E-03	11.5	1.0E-03
Iron	mg/l	2.9E-02	332.1	2.9E-02
Manganese	mg/l	1.2E-02	137.4	1.2E-02
Nickel	mg/l	1.1E-02	126.0	1.1E-02
Lead	mg/l	5.4E-02	618.3	5.4E-02
Copper	mg/l	2.4E-02	274.8	2.4E-02
Zinc	mg/l	7.8E-02	893.1	7.8E-02

Finally, the LCI results for the operation phase related to the chosen Functional Unit are summarized in tables 6_a and 6_b, while the methodology employed to estimate the direct GHG emissions is described in the following paragraph.

Tables 6a and b: LCI results for the operation phase.

Input		
Item	Unit	Value
PAA	kg	3,6
Polyelectrolyte	kg	1,6
Electricity	kWh	446,7
WWTP	n°of items	2,91E-06
Transport	Km	11,5
Output		
Sludge disposed of in landfill	kg	655,9
Grit disposed of in landfill	kg	18,3
Screening waste disposed of in landfill	kg	5,8
FOG transported to recycling plant	kg	5,2

Direct emissions		
Direct GHG emissions		
Parameter	Unit	Value
CH_4	kg	1,8
N_2O	kg	0,2
Pollutants emissions into the receiving water body		
COD	kg	34,8
BOD_5	kg	8,5
TNK	kg	3,3
NH_4^+	kg	2,5
NO_3-	kg	8,6
P_{tot}	kg	0,7

1.2.2.1 Estimation of Direct GHG Emissions

WWTPs are also direct sources of CO_2, CH_4 and N_2O, as a result of the biological processes tanking place in both aerobic and anaerobic treatment steps. Direct GHG emissions can be regarded as a nonnegligible contributor to the carbon footprint of WWTPs, that need to be taken into consideration. The different treatment steps which can be involved in direct GHG emissions are shown in figure 1 (GWRC, 2011).

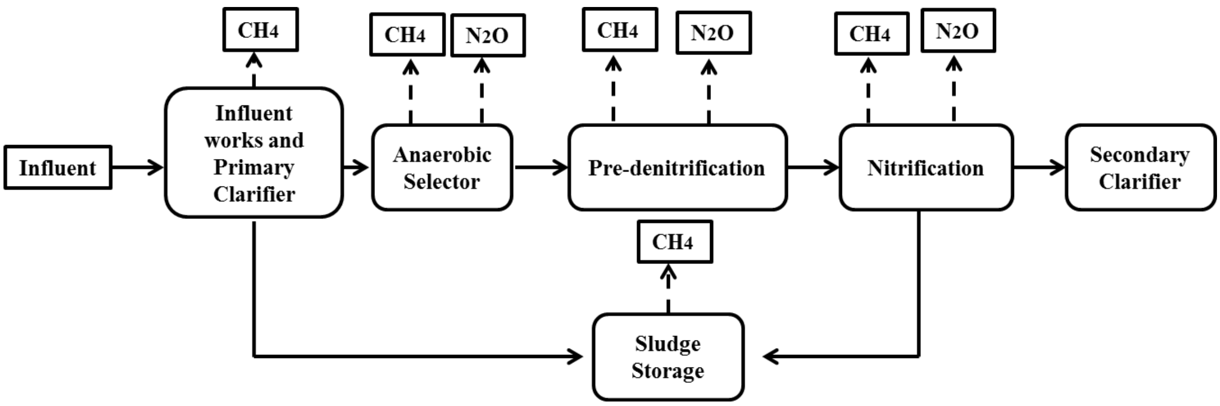

Figure 1: Treatment steps of Nocera Superiore WWTP where GHGs can be emitted.

The GHGs released during wastewater treatment as well as the accounting methods used in the inventory are described separately in the following part of this paragraph.

CO_2 emissions According to the Guidelines for National Greenhouse Gas Inventories (IPCC, 2006), in

this study the direct emissions of CO_2 released during wastewater and sludge treatment are not accounted for, since they are considered as biogenic origin.

CH_4 emissions – in order to estimate the direct methane emissions released by Nocera Superiore WWTP, equation 1 proposed by the IPCC (2006) as well as the emission factors determined during field testing at Papendrecht and Kortenoord WWTPs (IPCC, 2006) were considered.

IPCC equation 1

CH_4Emissions $_{WWTP}$ = (COD $_{influent}$ – COD $_{effluent}$) * EF Where:

COD $_{influent}$ – COD $_{effluent}$ is the COD removed during wastewater treatment at Nocera Superiore WWTP, that was chosen as activity datum of the investigated process.

EF is the emission factor of CH_4, which is calculated through the IPCC equation 2

FE = B0 * MCF

Where:

Bo is the maximum amount of CH_4 that can be produced per unit of COD in wastewater, expressed as Kg of CH4 * Kg COD^{-1}; The IPCC default value is 0.25 kg CH_4 * kg COD^{-1}.

MCF is the methane correction factor, which indicates the extent to which Bo is released from the investigated WWTP. The default MCF value for aerobic and well managed WWTPs ranges from 0 to 0.1.

FE is, therefore, a function of the maximum amount of CH_4 that can be produced per unit of COD (Bo) and the methane correction factor (MCF) indicating the extent to which Bo is released in the investigated WWTP.

The emission factors found during field testing at Papendrecht and Kortenoord WWTPs (GWRC, 2011), equal to 0.0087 g CH_4*g $COD_{influent}*^{-1}$ e 0.0053 g CH_4*g $COD_{influent}*^{-1}$were also considered. The aforementioned emission factors were selected since both plants take advantage from the same treatment technology exploited by Nocera Superiore WWTP and their sludge treatment lines do not include the anaerobic digestion of sludge. The so calculated CH_4 emissions are listed in table 7.

Table 7: Estimation of direct CH_4 emissions over the investigated period.

Model	Yearly emissions (Kg)
IPCC equation with MCF = 0.1	87,977.9
IPCC equation with MCF= 0.01	8,797.8
GWRC (Papendrecht)	33,870.9
GWRC (Kortenoord)	20,634.0

Results gained by the IPCC equation 1 depend most upon the choice of the MCF value. Since there was no element capable to drive this choice, the use of the IPCC equation was rejected. Therefore, it was decided to apply the emission factor determined at Kortenoord WWTP, since it has the same wastewater treatment schema (A^2/O activated-sludge process) as the investigated plant, with also a similar treatment capacity (100,000 PE). Finally, the emission factors estimated by Daelman et al. (2012; 2013) were not taken into consideration because of the different sludge treatment schema hold by Kralingseveer WWTP, which is equipped with an anaerobic digester with biogas recovery[6].

N_2O emissions – With regard to the estimation of N_2O emissions, the IPCC equation 3 as well as the emission factor proposed by Daelman et al. (2013) were evaluated.

<u>IPCC equation 3</u>

N2Oplant = (P * FIND-COMM * EF)

Where:

P is population served by the investigated WWTP,

$F_{ind-comm}$ is the fraction of industrial and commercial co-discharged protein, equal to 1.25 (default value proposed by the IPCC);

EF is the N_2O emission factor, equal to 3.2 g N_2O*person*$year^{-1}$ (default value proposed by the IPCC).

As it has already been pointed out, the EF proposed by the IPCC is based only on a field study in which the plant was not expressly designed for nitrogen removal. Therefore, its use may lead to underestimate the direct N_2O emissions from WWTPs. In order to overcome this problem, some Countries such as Denmark and USA have developed specific emission factors for their own GHG inventories. In Table 8 the aforementioned EFs as well as the nitrous oxide emission estimated by using the IPCC equation 3 with the considered EFs are reported.

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[6] Nocera Superiore WWTP is also equipped with a two-stage anaerobic digester, which is however not in operation.

Table 8: Direct N_2O emissions calculated by using the IPCC equation 3.

Emission Factors	EF (g N_2O*person*year^{-1})	Yearly emissions (Kg)
IPCC	3.2	259.4
USA*	3.2	259.4
USA**	7	680.8
Denmark	10.8	1,050.5

Note:
USA* EF for WWTPs being equipped with nitrification treatment only
USA** EF for WWTPs being equipped with nitrification and denitrification treatments

Emission factor proposed by Daelman et al. (2013) – The pioneering study carried out by Daelman et al. (2013) was the first long-term, on-line monitoring campaign measuring nitrous oxide and methane emissions from a municipal wastewater treatment plant. For this study, the average N_2O emission factor was 28g N_2O-N*kg $TKN_{influent}^{-1}$, or 2.8% of the incoming nitrogen. As a comparison, the emission factor proposed by the IPCC is 3.2 g N_2O-N*PE^{-1}, which amounts to 0.35 g N_2O-N*kg $TKN_{influent}^{-1}$ for developed countries, characterized by a high intake of protein (Kampschreur et al., 2009). The IPCC emission factor is therefore *eighty times* lower than the EF found in this study. Furthermore, the nitrous oxide emission monitored in this study showed a great seasonal dynamic, which was not fully understood.

The emission factor proposed by Dealman et al. (2013) was applied in this study to estimate the N_2O emissions throughout the investigated period, since it is based on a long term monitoring campaign (which was capable of taking the different operating and environmental conditions faced by the investigated WWTP over time into account) carried out at a WWTP employing a similar waterline treatment schema. As the incoming TKN throughout the investigated period was 78,861.1 kg, the estimated N_2O emission at Nocera Superiore WWTP is 3,469.9 kg.

1.3 Energy and mass balance for the investigated scenarios

The energy and mass balances of the sludge treatment line in scenario B are presented in Figure 2. Values are related to the chosen FU. Heat (red) and electricity (yellow) produced within the plant are shown by large dashes lines.

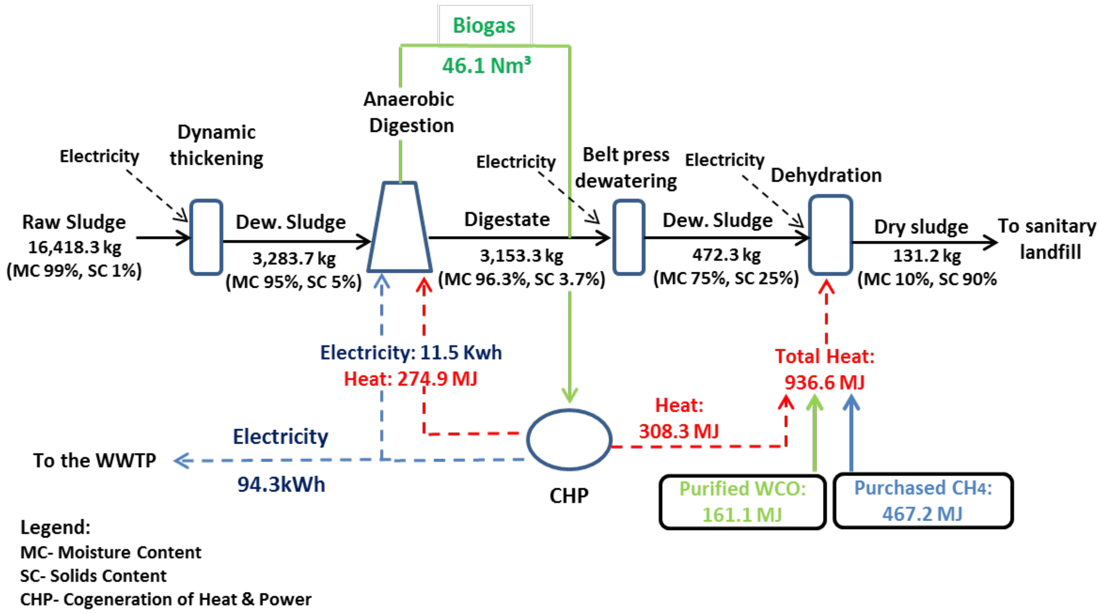

Figure 2: Sludge treatment line process diagram of Scenario B.

Similarly, Figure 3 shows the energy and mass balances of the sludge treatment line in scenario C and D. Values are related to the chosen FU. Heat (red) and electricity (yellow) produced within the plant are shown by large dashes lines.

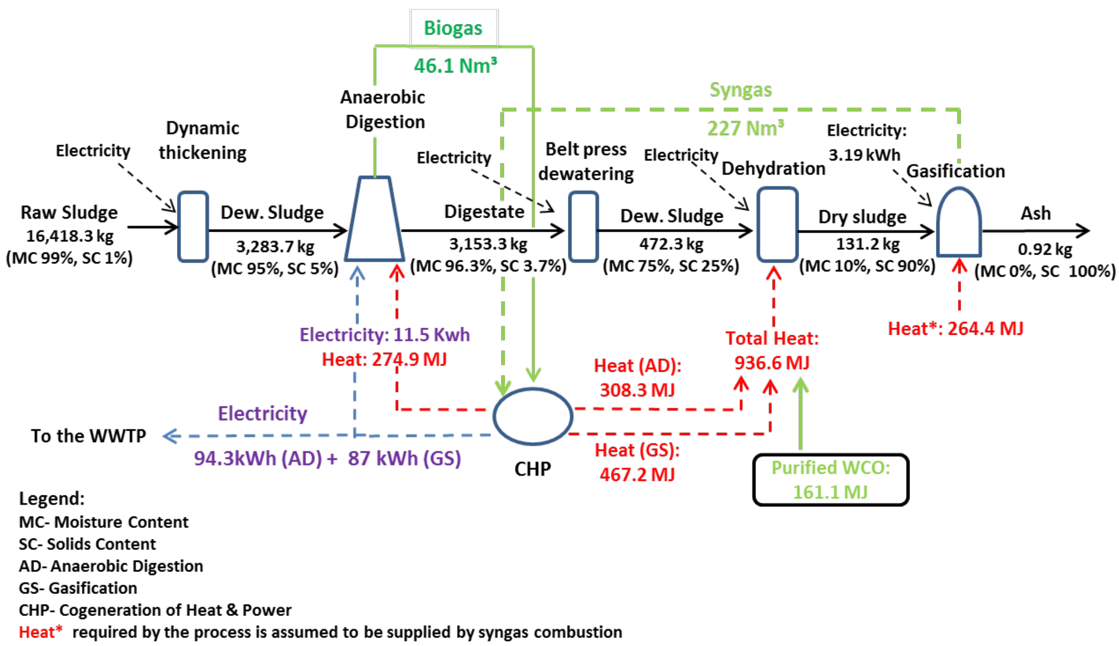

Figure 3: Sludge treatment line process diagram of Scenario C and D.

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