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1 **Techno-economic and Life Cycle Assessment of an Integrated**

2 **Hydrothermal Carbonization System for Sewage Sludge**

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10 **Highlights**

- 11 • Comparative analysis of anaerobic digestion (AD) and hydrothermal
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- 13 • Estimated improvement of +14.0% energy efficiency for the HTC-integrated
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- 15 • Calculated treatment costs per tonne of sludge of 66.2 € (AD) and 94.3 €
16 (HTC+AD).
- 17 • Generally reduced environmental impacts for HTC+AD option.
- 18 • Environmental performance of HTC+AD sensitive to the source of substituted
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20

21 **Abstract**

22 The need of more effective solutions for sewage sludge management is twofold: i) to
23 alleviate the overall operating costs for wastewater treatment and ii) to ensure proper
24 utilization/disposal considering environmental legislation. A comprehensive solution
25 integrating hydrothermal carbonization and anaerobic digestion to treat sewage sludge
26 was analyzed and compared with that of standalone anaerobic digestion. In order to
27 evaluate its performance, a process simulation model in Aspen Plus® served as the main
28 data source of the required input-output inventories to perform both techno-economic
29 and life cycle assessment, which follow conventional methodological standards. It was
30 observed that the integrated strategy generally reduces the environmental impacts
31 compared to the standalone configuration due to the recovery of a hydrochar that could
32 replace fossil fuels. In contrast, the standalone option only recovers a digestate that can
33 generate even a higher impact than its counterfactual alternative. The integration of
34 hydrothermal treatments showed an increase of 14 % in the gross energy efficiency of
35 the anaerobic digestion. However, economic concerns make the approach to require
36 further optimization, since the estimated cost for the HTC option were 42 % higher in
37 comparison to the conventional anaerobic digestion alternative. Although there is a
38 changing regulatory framework around sewage sludge, especially on nutrient recovery,
39 hydrothermal treatments have been proven as a potentially sustainable route for sewage
40 sludge treatment.

41

42 **Keywords**

43 Hydrothermal carbonization, Anaerobic Digestion, Aspen Plus, Techno-economic
44 Analysis, Life Cycle Assessment.

45 **Nomenclature**

46	<i>AD</i>	<i>Anaerobic Digestion</i>
47	<i>CAPEX</i>	<i>Capital Expenditure</i>
48	<i>CHP</i>	<i>Combined Heat and Power</i>
49	<i>COD</i>	<i>Chemical Oxygen Demand</i>
50	<i>d.b.</i>	<i>Dry Basis</i>
51	<i>GEE</i>	<i>Gross Energy Efficiency</i>
52	<i>HPS</i>	<i>High Pressure Steam</i>
53	<i>HTC</i>	<i>Hydrothermal Carbonization</i>
54	<i>HTCLP</i>	<i>Hydrothermal Carbonization Liquid Product</i>
55	<i>LPS</i>	<i>Low Pressure Steam</i>
56	<i>OPEX</i>	<i>Operational Expenditure</i>
57	<i>PSS</i>	<i>Primary Sewage Sludge</i>
58	<i>Sc_AD</i>	<i>Standalone Anaerobic Digestion scenario</i>
59	<i>Sc_HTC+AD</i>	<i>Hydrothermal carbonization and anaerobic digestion integrated</i>
60		<i>scenario</i>
61	<i>SS</i>	<i>Sewage Sludge</i>
62	<i>SSS</i>	<i>Secondary Sewage Sludge</i>
63	<i>TIC</i>	<i>Total Investment Cost</i>
64	<i>TOC</i>	<i>Total Organic Carbon</i>
65	<i>WWTP</i>	<i>Wastewater Treatment Plant</i>

1. Introduction

The management of vast amounts of sewage sludge (SS) is a major concern derived from the operation of wastewater treatment plants (WWTPs), making up 50-60% of their total operating costs. Specifically, the European Commission reported an estimation of 11.5 Mt, on a dry basis (d.b.), of produced SS in the EU27 (Samolada and Zabaniotou, 2014) and a 13.0 Mt prospect for 2020 (Raheem et al., 2018). Given its generation rate and nature, a convenient treatment of SS is necessary to avoid undesirable environmental effects, as it contains nutrients which can lead to eutrophication, hazardous substances (heavy metals, pathogens and persistent organic pollutants) and presents high biological activity (Wang et al., 2019). In regard of this, there is an active quest towards minimization of gross generation and effective management solutions for SS. Traditional fates have been landfilling, incineration and application on agricultural soils. Within the EU, SS management is currently regulated by various Directives (for a complete description, see Hudcová et al. (2019) and Kacprzak et al. (2017)), whose guiding threads are i) landfilling minimization, ii) limitation of incineration-derived emissions, iii) proper stabilization for a safe disposal and iv) reuse enhancement. However, lax legislation in the EU, especially in the field of agricultural application, has traditionally allowed Member States to set their own limitations when it comes to concentrations of pollutants (e.g. heavy metals or phenols) and pathogens (Kacprzak et al., 2017). As a result, notable differences arise between countries respecting their preferred management practices so far (Hudcová et al., 2019). For instance, in 2016, Ireland utilized almost 80.0 % of its reported sludge disposal in agriculture, while Germany and the Netherlands respectively incinerated ~60.0 % and ~90.0 %. In Spain, the last available data are for 2012 at the time of the present document (“Eurostat - Sewage sludge production and disposal”, 2019); in that year the

distribution was: 72.0 % agriculture, 4.00 % incineration, 14.0 % landfilling and 10.0
 % other applications, such as non-energy applications e.g. through the fabrication of
 additives for the cement industry (Chang et al., 2020) and construction materials
 (Gálvez-Martos, 2020), although these are limited by specific pollutants regulations.
 Regarding economic aspects, reported costs for SS treatment via conventional
 agricultural disposal, and combustion are respectively of 160 €·t⁻¹ (d.b) and 330 €·t⁻¹
 (d.b) (Wang et al., 2019).
 Novel technologies for energy recovery from SS emerge as better options from an
 environmental viewpoint, and hence, an increased interest for them exist within the EU
 framework (Tsybina and Wuensch, 2018). Amongst thermal technologies, pyrolysis and
 gasification are two promising alternatives but still involve high expenses related to
 feedstock moisture reduction and further product upgrading. Anaerobic digestion is
 considered the most appropriate valorization technique regarding costs, but it suffers
 from poor efficiency and long processing time (Raheem et al., 2018). Another route is
 that of hydrothermal technologies, such as hydrothermal carbonization (HTC). HTC is
 an induced coalification process in which the feedstock is dehydrated in order to
 decrease its oxygen to carbon and hydrogen to carbon ratios. It is carried out at
 temperatures of 180-250 °C and autogenous saturated pressure (10-50 bar) in an
 aqueous medium (40-90 % moisture), so the necessity for a previous drying step is
 avoided (Wang et al., 2019). In this process, biomass is converted into a lignite-like
 carbonaceous solid known as *hydrochar*, which presents improved dewaterability and
 hence, better stability and sanitation characteristics regarding disposal when compared
 with raw SS. Furthermore, the hydrochar presents good properties to be used for
 phosphorous recovery or as solid fuel in small scale heating systems (Song et al., 2019).
 Together with hydrochar, a liquid product (HTCLP) containing a diverse range of

chemical compounds is also obtained. HTCLP presents high chemical oxygen demand (COD) values, which makes it unsuitable for direct disposal. Although HTCLP could be the main contributor to the environmental impacts generated by the HTC of SS (Berge et al., 2015), it is particularly interesting from an energy recovery perspective via AD (Villamil et al., 2020, 2018).

In the present work, we compare an integrated technological pathway combining HTC and AD for SS valorization with the conventional approach based on AD. Process simulation models were developed to collect inventory data (input-output) from mass and energy balances, which were further utilized to evaluate energy efficiency, and the economic and environmental performance. In particular, we performed a techno-economic assessment including capital expenditures (CAPEX), operational expenditures (OPEX) and treatment cost estimation, as well as a comparative life cycle assessment (LCA).

2. Material and methods

2.1. Description of technology scenarios

In this work, we address a comparison between two technological scenarios for SS management. SS feedstock was assumed to be composed of a mixture of primary sewage sludge (PSS) and secondary sewage sludge (SSS) with a 3:2 mass ratio. PSS is the sludge resulting from the primary (physical) treatment in WWTPs while SSS comes from the secondary treatment (biological). Both scenarios were contemplated as to achieve (or approach) energy self-sufficiency, so heat integration and internal consumption of produced electricity were considered. A description of these scenarios is given hereafter:

- Standalone Anaerobic Digestion scenario (Sc_AD): base scenario consisting in co-feeding both primary sewage sludge (PSS) and secondary sewage sludge (SSS) to produce biogas, which is utterly cleaned and combusted to generate steam and electricity.
- Hydrothermal carbonization and anaerobic digestion integrated scenario (Sc_HTC+AD): alternative layout consisting in treating SSS in a HTC reactor to produce hydrochar. The HTC liquid product (HTCLP) is then mixed with PSS to produce biogas, which is cleaned and combusted with air. Flue gases from combustion are used to produce heat and electricity, as in Sc_AD, but also to dry the hydrochar up to the required final moisture content. The election of SSS to enter the HTC stage alone is based on its composition. SSS presents a more homogeneous composition when compared to PSS, which besides organic matter, also contains heavier compounds i.e. metals.

Block diagrams and the system boundaries for both scenarios are presented in Figure 1.

The composition and calorific value of treated SS is included in Table 1. The main technical specifications are summarized in Table 2.

[Figure 1. Block diagrams for studied technology scenarios and system boundaries.

Dotted lines represent avoided production. Top: Sc_AD; Bottom: Sc_HTC+AD]

[Table 1. Characterization of solid feedstock.]

[Table 2. Process specifications.]

In Sc_AD, PSS and SSS are mixed prior to entering a mesophilic AD stage (1.00 bar, 35.0 °C) (Villamil et al., 2018), where biogas is produced. Since both feedstocks pose a moisture content of 85.0 %, there is no need for an extra addition of water, neither for

162 pumping (Stemann et al., 2013) nor to run AD itself (Angelonidi and Smith, 2015). A
163 digestate stream containing non-digested material (6.00 % w/w), together with water
164 and a small amount of dissolved gases also leaves the stage. The utilization of biogas
165 requires previous conditioning to avoid further corrosion and emission issues in a CHP
166 unit (Bak et al., 2019). Considering this, NH₃ is cleaned from biogas (98.0 % retention)
167 in a scrubber unit where water is injected in the top. Some H₂S is retained as well
168 (21.0%). Column bottoms composed by utilized water plus retained pollutants are
169 headed to wastewater treatment. The clean biogas (58.0 % CH₄, 41.0 % CO₂ v/v) is
170 compressed and combusted with air (20.0 % volumetric excess) in a gas turbine, which
171 operates at 10.0 bar and 1,200 °C, to generate electricity. Heat from the exhaust gas
172 leaving the turbine (1.50 bar and 846 °C) is then recovered in a high pressure steam
173 (HPS) generator to produce steam at 40.0 bar and 250 °C. The exhaust gas is finally
174 emitted to the atmosphere at 85.0 °C.

175 In Sc_HTC+AD, SSS enters the HTC stage, which operates at 20.0 bar and 208 °C
176 (Villamil et al., 2018). Regarding the dry matter to moisture ratio, mechanically
177 dewatered SS (80-85% moisture content), as in this case, has been reported to be
178 suitable to undergo the HTC process without needing any additional water supply
179 (Wang et al., 2019). The products stream from HTC is subsequently depressurized in
180 two flash tanks (6.24 bar and 1.98 bar). Streams obtained from depressurization are
181 used to preheat the sludge inlet to the HTC reactor up to 142 °C and to produce a certain
182 amount of low pressure steam (LPS) at 2.33 bar and 128 °C. After that, the solid product
183 is separated from the liquid phase by means of a filter press. The obtained hydrochar at
184 this point has a moisture content of 30.0 %. HTCLP is mixed with PSS and fed to the
185 AD stage, which occurs as for Sc_AD. In this case, the exhaust gas is used to dry the

hydrochar to a final moisture content of 8.00 %. The exhaust gas is emitted to the atmosphere at 53.0 °C.

2.2. Process modeling and simulation

The described scenarios were simulated in Aspen Plus® V10, with the aim of obtaining relevant process information for the techno-economic and life cycle inventory. The treatment capacity for the simulated plants was set from real sludge production data in *La China* wastewater treatment facility, in Madrid (Spain) (SICE, 2019). The total amounts of PSS and SSS entering the system in both scenarios were respectively 4,530 and 2,970 t·year⁻¹ (d.b.).

The reader is addressed to the Supplementary Material to check complete simulation screenshots from Aspen Plus® and a sequential description of process modeling steps. A *user-friendly* process modeling flowchart is presented in Figure 2.

[Figure 2. Simulation flowchart excerpts. A: Hydrothermal Carbonization.

B: Anaerobic Digestion. C: Biogas cleaning and combustion. D: Hydrochar drying.]

2.3. Techno-economic analysis framework

A spreadsheet-based model was developed to gauge both scenarios as regards to their economic performance. Specifically, we estimated the contribution of the capital (CAPEX) and operational (OPEX) expenditure impacts on wastewater sludge management. Here, it is worth to note that the process was approached as an environmental management service, rather than focusing on hydrochar production as a profit-making activity. This perspective seems to be a more reasonable option, especially if we compare reported prices for hydrochar, which range 100 – 160 €·t⁻¹ (Lucian and Fiori, 2017; Saba et al., 2019; Zeymer et al., 2017), with that of fossil coal

(45.0 €·t⁻¹) (Markets Insider, 2019). Also, wastewater and SS management projects have widely been identified as non-profitable if not publically subsidized, even when revenues from energy generation are considered (Fersi et al., 2015). Because of this, project evaluation through a discounted cash flow analysis was decided to be out of the scope of this work.

Overall, methodology described in Towler and Sinnott (2013) served as the primary guideline for this task. The study corresponds to a Class 4 estimate (± 30.0 % accuracy), according to the classification by the Association for the Advancement of Cost Estimating International (*AACE I8R-97*, 2005). Cost data were all updated to February 2019. We first estimated the individual equipment purchase cost by using cost correlations or capacity ratioing (Williams' rule), based on design parameters obtained from process simulation. All equipment was assumed to be constructed in stainless steel (304), because of the presence of acids and corrosive materials such as H₂S, so a material factor was applied when necessary. Then, we utilized the Lang factorial method to estimate the Total Investment Cost (TIC). Fixed operational costs were calculated as percentages of the CAPEX and revenues from products sales deducted. Finally, assuming a 20-year plant lifespan, we estimated the average treatment cost per tonne of treated sludge. The main assumptions adopted to perform the TEA are compiled in Table 3.

[Table 3. Parameters and assumptions considered in the techno-economic assessment.]

2.4. Life cycle assessment framework

2.4.1 Goal and scope

The goal of the LCA is to compare the environmental performance of the HTC and AD integrated scenario (Sc_HTC+AD) with the standalone AD system (Sc_AD). The

functional unit is the treatment and disposal of 1,000 kg wet mixed sludge consisting of 60.4 % PSS and 39.6 % SSS with the composition showed in Table 1. The system boundary starts when the dewatered mixed sludge entered into the treatment process and includes the treatment itself, the utilization of the biogas, the transport and application of digestate to land, and the transport and utilization of hydrochar in domestic heating stoves (Figure 1). Therefore, the environmental burdens associated with the life cycle stages before the treatment system were not included.

Since the LCA addressed in this work is intended to compare the environmental performance of two scenarios, an attributional modelling framework was adopted with average data used. We applied the substitution approach so that the by-products obtained from SS treatment (i.e. energy from biogas, energy from hydrochar, and digestate) were assumed to avoid the production of the corresponding market products and their environmental burdens (Figure 1). A 100-year time horizon was considered and therefore only emissions to the environment occurring during the first 100 years were considered.

Given the availability of data, nine of the impact categories recommended by the International Reference Life Cycle Data System (ILCD) were selected for the assessment: global warming, acidification, terrestrial eutrophication, freshwater eutrophication, marine eutrophication, photochemical ozone formation, respiratory inorganics, human toxicity cancer effects, and human toxicity non-cancer effects. The life cycle impact assessment (LCIA) methods used are those recommended by ILCD.

2.4.2 Life cycle inventory

The LCIA data for background processes (e.g. electricity and raw materials) were taken from ecoinvent v3.6 using the cut-off system model. The life cycle inventory (LCI) of

the foreground system was modelled with data provided by simulation and completed with data from the literature. The main parameters used for LCI modelling are summarized in Table 4 and detailed in the following paragraphs. The comprehensive LCI of each scenario is provided in Supplementary Material (Tables S4-S18).

[Table 4. Summary of the main parameters used for life cycle inventory modelling].

2.4.2.1 Sc_AD

Fugitive emissions from the AD stage were assumed to equal 3.00 % of the biogas produced (Yoshida et al., 2014). The remaining biogas is sent to the scrubber unit, where 1.00 % of the input biogas is also lost as fugitive emissions (Petersson and Wellinger, 2009). Biogas cleaning column bottoms are to be treated prior discharge. The treatment process data were obtained from the ecoinvent v3.6 database. Energy and water consumption for AD operation, energy production, and the composition of the exhaust gas produced from biogas combustion are given by the simulation results. The composition of the exhaust gas was completed with data from Nielsen et al. (2010) regarding CH₄ and CO emissions. The surplus of electricity feed into the grid was assumed to avoid an equivalent amount of electricity produced by the Spanish electricity production mix. The surplus of HPS sold was assumed to avoid the average market for steam production in industrial applications.

The mass and composition of the digestate produced was calculated by subtracting the mass of C, N, and S degraded during AD from the composition of the input feedstock (Table 1). Leaching of metals during AD is usually assumed negligible and therefore all metals contained in the feedstock are transferred into the digestate (Boldrin et al., 2011). Based on this mass balance, the metal content of the digestate obtained in Sc_AD and Sc_HTC+AD is below the Spanish legislation limits for the application of biological

281 treated SS to agricultural land (Table S3). In consequence, we assumed that the
 282 digestate is applied to agricultural land after transported 50.0 km.
 283 Emissions of C, N, and P following digestate application were modelled by means of
 284 emission factors. The emission factors describe how much of the applied element is
 285 transferred to water and air during the 100-year time horizon. Emission factors for N
 286 were taken from Bruun et al. (2016), corresponding to average data for a sandy loam
 287 soil located in a low precipitation region: 3.00 % of the N applied is emitted as N₂O,
 288 6.90 % is emitted as NH₃, 37.5 % is emitted to groundwater as nitrate, and 14.0 % is
 289 emitted to surface water bodies as nitrate. The C balance was assumed as follow:
 290 0.05 % of the C applied is emitted as methane, 93.5 % as biogenic CO₂ (Yoshida et al.,
 291 2018), and the remaining 6.00 % remains sequestered in soil after 100 years (Bruun et
 292 al., 2016). Finally, 3.53 % of the P applied is emitted to surface water bodies based on
 293 the median value presented by Kronvang et al. (2005). All the metals contained in the
 294 digestate are also applied to land.
 295 The application of digestate to land was assumed to avoid the production and
 296 application of N, P, and K mineral fertilizers. The rate of mineral fertilizer that is
 297 avoided is determined by the nutrients content in the digestate (N, P, and K) and the
 298 plant-availability of the digestate compared to the plant-availability of the mineral
 299 fertilizer (Hansen et al., 2006). The plant-availability of the nutrients contained in
 300 digestate was assumed to be 24.5 % for N (Bruun et al., 2016), 73.0 % for P (Yoshida et
 301 al., 2018), and 100 % for K (Bruun et al., 2016). The overall nutrients mass balance
 302 shows that 22.7% of the N, 73% of the P, and 100% of the K contained in the input
 303 feedstock is absorbed by plants in Sc_AD. The plant-availability of the N contained in
 304 the avoided mineral fertilizers is 67.0 % of the N applied (Yoshida et al., 2018), while
 305 for P and K the same dynamic as for digestate was assumed (Brockmann et al., 2018).

The avoided environmental impacts associated with the application of mineral fertilizers to land were modelled as follows. It was assumed that 2.00 % of the N contained in a mineral fertilizer would be emitted as N₂O, 10% would be emitted to groundwater as nitrate, and 4.00 % would be emitted to surface water bodies as nitrate (Yoshida et al., 2018). Furthermore, the metal content of mineral fertilizers was calculated with average data from McBride and Spiers (2001). Since the application of metals to land can be relevant to the human toxicity category, it should be noted that the USEtox method considers a linear relationship between toxicity potential and amount of metals added. This implies that any potential beneficial effect of adding crop nutrients like zinc and copper is not accounted for (ten Hoeve et al., 2019).

2.4.2.2 Sc_HTC+AD

The same assumptions from the Sc_AD apply to the Sc_HTC+AD. The composition of the digestate was obtained by a mass balance between the chemical elements in the PSS and HTCLP and the mass of C, N, and S degraded during AD. In this case, the overall nutrients mass balance shows that 18.9% of the N, 48.2% of the P, and 84.9% of the K contained in the input feedstock is absorbed by plants in Sc_HTC+AD. The lower nutrients recovery rate is because a portion of nutrients ends up in the hydrochar. Approximately 15.4% of the N, 34.0% of the P, and 15.1% of the K is transferred into the hydrochar during HTC.

The hydrochar produced is transported 20.0 km and combusted in a domestic heating stove with heat conversion efficiency of 70.0 % of the LHV. Heat production was 14.1 MJ·kg⁻¹ hydrochar. Because of the lack of measured data associated with emissions from hydrochar combustion, the inventory was adapted from Owsianiak et al. (2016). They reported measured emissions for hydrochar combustion in a pilot-scale stove: 38.0 mg CO·kg⁻¹ hydrochar (d.b.), 3.00 g NO_x·kg⁻¹ hydrochar (d.b.),

0.120 g PM_{10-2.5}·kg⁻¹ hydrochar (d.b.), and 73.0 mg PM_{2.5}·kg⁻¹ hydrochar (d.b.). For other waste-specific emissions like heavy metals, we applied the transfer coefficients for SS incineration retrieved from the ecoinvent database, as has been previously used in the literature (Berge et al., 2011). We also used the information from Owsianiak et al. (2016) to model the impacts associated with hydrochar ash disposal. Finally, the utilization of hydrochar for heat production in stove avoids the production, combustion, and ash disposal associated with hard coal briquettes. The substitution ratio is 1MJ:1 MJ. The LCIA data was obtained from ecoinvent database v3.6.

3. Results

3.1. Process inventory

The main results of the mass and energy balances from process simulation are depicted in Figure 3. For the sake of LCIA, all quantities are referred to the functional unit, as defined in Section **Error! Reference source not found.**

[**Figure 3.** Process simulation mass and energy balance results. Top: Sc_AD; Bottom: Sc_HTC+AD. All quantities are referred per tonne of treated mixed sludge (60.0 % PSS, 40.0 % SSS) per year.]

In Sc_HTC+AD, 26 kg hydrochar·t⁻¹ sludge are obtained, which corresponds to the 40.0 % mass yield (γ_m) from dry SSS that was set for the process. The carbon yield (γ_C) and energy yield (γ_e) for the HTC reaction, as defined in Erlach (2014), are 42.0 % and 54.0 % respectively.

$$\gamma_c = \frac{\text{Carbon retained in hydrochar}}{\text{Carbon in SSS}} \quad \text{Eq. 1}$$

$$\gamma_e = \gamma_m \frac{HHV_{hydrochar}}{HHV_{SSS}} \quad \text{Eq. 2}$$

The net production of biogas in both scenarios is similar (103 Nm³·t⁻¹ sludge in Sc_AD, versus 100 Nm³·t⁻¹ sludge in Sc_HTC+AD), but still significant to derive a slight difference in the amount of required air for combustion and outlet quantities of digestate, exhaust gas and wastewater coming out the biogas cleaning scrubber. Regarding energy generation, either in Sc_AD and Sc_HTC+AD the produced electricity and heat are sufficient to fulfill the internal energy demand from the plant. The amounts of electricity and HPS are virtually the same and the surplus of produced electricity is ~30.0 % in both scenarios. On the other hand, 60.0 % of the generated HPS in Sc_AD is available to be sold compared to only 15.0 % in Sc_AD+HTC. This is mainly due to the heating demand from the HTC stage. The HTC reaction itself is exothermic (Funke and Ziegler, 2011), but in order to preheat the sludge to the reaction temperature, heat recovery from product streams, as shown in Figure 2A, is insufficient. After the last heat exchanger prior to the HTC reactor, the temperature of the sludge stream is still 142 °C, so HPS is utilized to fulfill the extra requirement. Another difference between both scenarios is the production and utilization of LPS in Sc_HTC+AD, which is almost entirely used to satisfy the heat demand from the AD stage. This represents an advantage, since it avoids the usage for such purpose of HPS, which has a superior market price. However, this positive effect is outweighed, as heat retrieved for LPS production comes from the HTC stage, which also consumes 85.0 % of the produced HPS. This means that at the same time the inclusion of HTC stage enables the recovery of more heat from the process, also more heat is required to sustain it.

Differences in water requirements for each scenario also result from steam generation and utilization. Internal steam delivery system is considered to be a closed circuit, with only 5.00 % annual water replacement necessities due to leakage. This explains the lesser water requirement in Sc_HTC+AD compared to Sc_AD (2.16 t water·t⁻¹ sludge vs. 3.34 t water·t⁻¹ sludge), as a smaller amount of steam is available to be sold.

3.2. Energy efficiency

The energetic performance of the studied scenarios was characterized by means of the Gross Energy Efficiency (GEE) indicator, as defined in Cao and Pawłowski, (2012):

$$GEE = \frac{\sum(CV_{products} \cdot W_{products})}{CV_{sludge} \cdot W_{sludge}} \quad \text{Eq. 3}$$

Where *CV* and *W* respectively address the calorific value and the mass amount/flow of either the products or the inlet sludge (d.b.).

Translated into our scenarios:

- {Sc_AD}:

$$GEE = \frac{[MJ \text{ sold electricity}] + [MJ \text{ sold HPS}]}{[MJ \text{ contained in SSS}] + [MJ \text{ contained in PSS}]}$$

- {Sc_HTC+AD}:

$$GEE = \frac{[MJ \text{ produced hydrochar}] + [MJ \text{ sold electricity}] + [MJ \text{ sold HPS}] + [MJ \text{ sold LPS}]}{[MJ \text{ contained in SSS}] + [MJ \text{ contained in PSS}]}$$

Based on this, the calculated GEE for Sc_AD and Sc_HTC+AD were 14.0 % and 28.0 %. This difference may be explained from the fact that in Sc_HTC+AD we have an additional product, the hydrochar, in which more energy is recovered, compared to the correspondent biogas produced from the same amount of SS. The Sc_AD system

produces 378 MJ per tonne of sludge, most of it as heat and around 30% as electricity.

On the other hand, the Sc_HTC+AD produces 164 MJ of electricity and heat, which is less than AD, but 562 MJ of hydrochar, of a more flexible exploitation.

Yet, caution should be applied when interpreting these results, as considered process boundaries and assumptions may have a significant impact on calculation. For instance, as stated in section 2.1, we considered that all the energy demand from the plant is self-supplied. Instead, we could have assumed steam and electricity to be purchased from an external supplier. In such case, the balance would change, as all the produced energy would be available to be sold as a product, at the same time an additional term accounting for externally provided energy should be included in the denominator of Eq. 3.

3.3. TEA results

The economic breakdown for CAPEX and OPEX of both scenarios is presented in Table 5. A detailed list of purchased equipment can be found as well in the Supplementary Material for this work. Sc_HTC+AD presents a 37.0 % increase in CAPEX compared to Sc_AD. Such increase comes from the addition of all the required equipment for the HTC reaction and hydrochar drying sections, as the investment for the other sections is almost equal. Specifically, the HTC reactor and the hydrochar dryer account for 37.0 % and 17.0 % of the additional equipment expense, respectively.

[Table 5. Estimated CAPEX, OPEX and sludge treatment cost for studied scenarios.

Values referred per tonne of treated sludge if no other claim made.]

OPEX is almost entirely contributed by fixed expenses, which make up 99.0 % in both Sc_AD and Sc_HTC+AD (before deducting revenues). This is a logical result for a self-sustained process managing waste, since no expenses for raw materials or utilities exist.

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416 A +41.0 % difference in OPEX is observed in Sc_HTC+AD when compared to Sc_AD.

417 Attending to the calculation procedure exposed in Table 3, this is explained by the
418 higher CAPEX and the necessity of a larger working force due to additional pieces of
419 equipment. The working force has a direct impact on the *Operating labor* category,
420 which simultaneously affects the *Direct salary overhead* and *General plant overhead*
421 (charges to cover corporate overhead functions such as human resources, research and
422 development, information technology, finance, etc.). Deduction for sales revenues only
423 has a moderate effect in the reduction of OPEX values, which accounts for 8.40 % in
424 Sc_AD and 6.60 % in Sc_HTC+AD. In particular, hydrochar sales, despite accounting
425 for 55.0 % of total revenues in Sc_HTC+AD, do not have a significant impact, because
426 a notable reduction in HPS sales occurs at the same time, which in the end cancels the
427 positive effect.

428 We estimated the treatment cost per tonne of sludge for both scenarios (Table 5),
429 assuming a 20-year plant lifespan. The contribution of CAPEX and OPEX to the
430 calculated treatment cost was roughly 30.0 % and 70.0 % in both scenarios. We
431 calculated 66.2 € for Sc_AD and 94.3 € for Sc_HTC+AD, which implies a difference of
432 +42.0 % in Sc_HTC+AD.

433 3.4. Life cycle impact assessment

434 Figure 4 shows the LCIA results for the environmental impact categories addressed in
435 this study. The global warming impact is reduced from 72 kg CO_{2-eq}·t⁻¹ sludge in
436 Sc_AD to 18 kg CO_{2-eq}·t⁻¹ sludge in Sc_HTC+AD (Figure 4a). This reduction is
437 associated with the utilization of the hydrochar in stove for heat production, which
438 avoids the combustion of hard coal briquettes and the subsequent emissions. It is worth
439 noting that the carbon contained in the hydrochar is from biogenic origin and we

440 assumed that biogenic carbon is neutral with respect to the global warming. By contrast,
441 the combustion of hard coal briquettes would release large amounts of fossil CO₂.
442 The application of digestate to land is the main contributor to the global warming
443 impact produced in both scenarios. The global warming impact of digestate application
444 is higher than the avoided impact. This is mainly because of a higher emission of N₂O
445 after digestate application compared to mineral fertilizer application. Methane fugitive
446 emissions from the AD stage and the biogas cleaning scrubber and methane emissions
447 from biogas combustion have also a relevant contribution to the global warming impact,
448 although very similar between the two scenarios.

449 The impact on acidification, respiratory inorganics, and photochemical ozone formation
450 follows the same trend to that of global warming (Figure 4b-c). The reduced impact in
451 Sc_HTC+AD compared to Sc_AD is driven by the avoided impacts associated with
452 hydrochar utilization in a domestic heating stove. In particular, SO₂ and PM_{2.5}
453 emissions from the utilization of hydrochar are considerably lower compared to those
454 from the utilization of hard coal briquettes due to a higher content of S in the latter. The
455 avoided production of the hard coal briquettes further contributes to reduce the impact
456 in these categories. The main contributor to the impact produced in both scenarios is the
457 application of digestate to land because of the emission of NH₃ in the case of
458 acidification and respiratory inorganics and the emission of NO_x in the case of
459 photochemical ozone formation. Biogas combustion is also a relevant contributor to the
460 impact on photochemical ozone formation in both scenarios due to the emission of CO,
461 NO_x, and CH₄.

462 Regarding eutrophication impact categories, digestate application is the largest
463 contributor (Figure 4e-g). Terrestrial eutrophication is related to NH₃ emissions to air,

freshwater eutrophication to P leaching, and marine eutrophication to nitrate leaching.

Therefore, the impact is slightly reduced in Sc_HTC+AD since a lower amount of digestate is applied to land compared to Sc_AD.

Finally, human toxicity is also dominated by digestate application to land (Figure 4h-i). The impact slightly differs from one scenario to another. The impact on human toxicity, cancer effects is dominated by the emission of chromium and mercury to soil, while the impact on human toxicity, non-cancer effects is largely dominated by the emission of zinc.

[Figure 4. Life-cycle impact assessment results. The functional unit is 1,000 kg wet mixed sludge. Values above zero represent environmental impacts produced. Values below zero represent environmental impacts avoided.]

4. Discussion

4.1. The potential role of hydrochar in the management of sewage sludge

The results from the assessment of the integration of HTC with AD generally show a better life cycle environmental performance compared to the standalone AD configuration, but a generally higher costs despite the proven higher energy efficiency. In fact, the LCA results highlight the large influence of the environmental savings of using hydrochar as a substitute for hard coal briquettes in domestic heating systems as well as the environmental impacts caused by the application of digestate to land. This is confirmed by other authors, which also highlight the large environmental benefits associated with energy recovery from hydrochar (Benavente et al., 2017). Overall, no significant differences are observed for the analyzed scenarios regarding mass and energy flows if we attend to the AD section. As stated before, the production of

hydrochar, despite modest, has a notable positive impact on the global energy efficiency. In this sense, HTC integration with AD, as presented in this paper, can be a good option to enhance the limited energy efficiency shown by AD.

There are important additional benefits associated to the production of hydrochar beyond those due to the higher energy efficiency of the process. Hydrochar is a high energy density, transportable solid fuel, which e.g. can be considered renewable in the framework of the European renewable energy directive (Directive (EU) 2018/2001, 2018). This definition of the renewability of the fuel is however based on the greenhouse gases savings, which must be over 80% in the production of heat according to the directive, which ignores other environmental categories. The EU taxonomy (EU-TEG on Sustainable Finance, 2020), a European instrument for the definition of threshold criteria for *sustainable* investments or bonds, establishes anaerobic digestion of sewage sludge as a climate change mitigation technology under two main criteria: (i) electricity, heat or upgraded biogas shall be produced, and (ii) methane leakage needs to be monitored. On other environmental categories, it establishes the need to reduce air pollutants in the combustion of gas, and the fulfillment of national rules regarding the use of digestate. In this policy framework, the combination of HTC with AD has been proven to perform better from the energy efficiency point of view, has a better environmental performance than the sole application of AD, and there are not relevant trade-offs towards other environmental categories. So, it fulfills the criteria of the renewable energy directive on emissions reductions and is an eligible technology according to the EU taxonomy.

Regarding the rules towards land application, the approaches differ in many countries on the criteria for its applicability. In any case, the high impact of the agricultural use of digestate, potentially higher than its counterfactual (i.e., mineral fertilizers) has been

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512 proven ((Yoshida et al., 2018). Therefore, compared with the standalone AD, the
513 integration with HTC reduces the amount of digestate obtained in benefit of hydrochar,
514 which, potentially, can avoid more polluting products. Probably, the produced
515 hydrochar would be mainly composed of recalcitrant or fixed carbon, rather than
516 biodegradable carbon, which reacts in the anaerobic digestion to produce biogas, and
517 therefore, fugitive emissions would be lower. The carbon content in the digestate of the
518 two scenarios would have different reactivity or degradability. This, however, has not
519 been considered due to the lack of data and the associated uncertainty.

520 The main drawback at policy level is the new trend towards higher nutrients recycling,
521 which may produce legal barriers to hydrochar. Many European countries are
522 encouraging and starting to regulate nutrients recycling. In Europe, the Fertilising
523 Products Regulation (Regulation (EU) 2019/1009, 2019) recognizes the potential of
524 phosphate recycling from sewage sludge, although its application remains voluntary;
525 countries such as Germany have started to mandate phosphate recycling from sewage
526 sludge in large wastewater treatment plants and for contents higher than 2% of
527 phosphorous in dry basis. Other countries such as the Netherlands, have established
528 voluntary commitments for the recycling of phosphate, while phosphorous recovery will
529 become obligatory in Switzerland from 2026 (Hermann and Thornton, 2019). In our
530 analysis, a part of the phosphorous contained in the sewage sludge is transferred into the
531 hydrochar (about 34%) and eventually lost or diluted in the combustion ash. Recovery
532 from diluted phosphorous streams can lead to inefficient recycling, and the preferable
533 waste treatment then may require specific incinerators of sewage sludge for its
534 management if the use of AD digestate is not able to recycle a significant amount of
535 phosphorous (Gálvez-Martos, 2020).

4.2. Expectable routes for the implementation of hydrothermal treatments

This work evaluates the potential environmental and economic performance of a hydrothermal carbonization system applied to sewage sludge; however, the implementation of such technology is still far from a large commercial application, since the technology readiness level has not reached maturity. Therefore, in this context, this work shall be read as a preliminary approximation to its real performance. For a deeper study on the potential deployment of the technology for sewage sludge, three aspects need to be considered: (i) the regulatory framework, (ii) its efficiency and economics, and (iii) the prospective market for hydrochar.

The legal framework has already been introduced in the previous section. As observed, there is a clear regulatory trend towards recycling of nutrients in a probable 2025 - 2030 scenario. Consequently, the use of hydrochar as a fuel should allow recycling of nutrients under certain circumstances and the fate of nutrients in the HTC process needs to be better known. Should the recovery of phosphate from hydrochar ashes become regulated, the potential application may favour centralized installations rather than distributed or even domestic use, reducing its potential prospective market.

On the other side, higher costs derived from increased plant complexity arise as the main drawback for the integration option to compete with the standard AD technology. Still, the cost gap might be narrowed by improving the energy yield of HTC, plant size effect. Further improvement in the energy yield of HTC (approximately +20.0 %) could be achieved by optimizing reaction temperature and retention time, as suggested by (Danso-Boateng et al., 2015). Scale should also be a matter of further analysis. Akbari et al. (2019) described the influence of plant capacity on the capital intensity (i.e. the capital investment per unit capacity) for the hydrothermal processing of yard waste. The

1 authors found a rapid reduction in the capital intensity curve as plant capacity was
2 increased (for small plant capacities) until the curve flattens and no further reduction is
3 attained as plant capacity is augmented.
4
5 Furthermore, a trade-off solution between improved energy efficiency and
6 environmental performance on one side, and treatment costs on the other, could be
7 pursued. A metric for this balance could be calculated as the cost of CO₂ avoidance; in
8 our work, the better performance of the HTC_AD in relation to AD has an associated
9 cost of 392 € per tonne of avoided CO₂ (only OPEX considered in this calculation),
10 which is a value exceeding largely the cost of carbon capture in the industry
11 (Roussanaly, 2019). However, this cost is calculated from a well-established sustainable
12 technology such as AD, considered, for instance, in the EU taxonomy for sustainable
13 financing in the field of climate change mitigation. A further reduction of costs might be
14 achieved by the inlet amount of SS that undergoes HTC, which utterly means the
15 inclusion of a certain amount of PSS, rather than only SSS, as considered in this work.

16 **4.3. Sensitivity analysis**

17 Sensitivity analyses were performed in order to evaluate the uncertainty associated with
18 key assumptions (Table 6). On the one hand, the sensitivity of LCA results to N₂O
19 emissions following digestate application to land was tested against a lower emission
20 factor of 1.0 % and a higher one of 4.5 %. On the other hand, the substitution of
21 alternative heat production systems was tested. This includes the substitution of the
22 European average heat market from natural gas, heat from wood pellets used in small
23 boilers, and heat from biogas co-generation plant (Table S19). The results show that a
24 change in the N₂O emission factor does not alter the ranking of the scenarios, and that a
25 higher emission factor implies higher climate benefits for the integrated alternative. In

contrast, the environmental performance of the HTC and AD integrated alternative is highly sensitive to the fuel substituted by the hydrochar. The substitution of fossil fuels, like hard coal briquette and natural gas, is imperative to guarantee the environmental superiority of the integrated alternative over the AD standalone configuration. The substitution of renewable fuels like biogas and wood pellets does not provide climate benefits with respect to the standalone AD.

[Table 6. Sensitivity analysis. Results are given as net difference between Sc_AD and Sc_AD+HTC. A positive value means that the impact in Sc_AD+HTC is lower compared to Sc_AD. The units of the impact categories are those in Figure 4.]

5. Conclusions

The integration of HTC with AD represents an opportunity to improve the energy recovery from sewage sludge when compared to the benchmark process of standalone AD (28 % vs. 14 %). Furthermore, this strategy generally shows a better life cycle environmental performance. For instance, the global warming impact is reduced from 72 to 18 kg CO₂-eq·t⁻¹ sludge when AD is shifted to the integrated configuration. However, a sensitivity analysis showed these savings, along with the reduction in other LCA impact categories (i.e. acidification, respiratory inorganics, and photochemical ozone formation), are strongly dependent on hydrochar utilization assumptions. We found the hydrochar would do derive these positive effects if assumed to replace fossil fuels, but not for the case of other biofuels (e.g. wood pellets or biogas). Also, economic concerns hamper the commercial deployment of the technology, since the calculated treatment cost for the integrated configuration is 42 % higher than conventional anaerobic digestion (94.3 €·t⁻¹ sludge vs 66.2 €·t⁻¹ sludge). The higher

cost can be mostly explained from the increased capital investment (+37 %) required by additional plant complexity, which is not compensated by additional revenues, given the current low market value of hydrochar. Nevertheless, the economy of scale, and the potential improvement of hydrochar yield, along with its consideration as renewable fuel, could improve the overall economic potential of HTC. The changing regulatory framework around sewage sludge on nutrient recovery will probably influence the prospective market of hydrochar produced from hydrothermal treatment of sewage sludge, which, in any case, has been proven as an environmentally-sensible route for sewage sludge treatment.

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1 **802 Tables and figures captions**

2
3
4 **803 Table.** Characterization of solid feedstock.

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8 **804 Table.** Process specifications.

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11 **805 Table.** Parameters and assumptions considered in the techno-economic assessment.

12
13
14 **806 Table 4.** Summary of the main parameters used for life cycle inventory modelling.

15
16
17 **807 Table 5.** Estimated CAPEX, OPEX and sludge treatment cost for studied scenarios.

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20 **808**Values referred per tonne of treated sludge if no other claim made.

21
22
23 **809 Table 6.** Sensitivity analysis. Results are given as net difference between Sc_AD and
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26 **810**Sc_AD+HTC. A positive value means that the impact in Sc_AD+HTC is lower compared
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28 **811**to Sc_AD. The units of the impact categories are those in Figure 4.

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32 **812**

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35 **813 Figure 1.** Block diagrams for studied technology scenarios and system boundaries.

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37 **814**Dotted lines represent avoided production. Top: Sc_AD; Bottom: Sc_HTC+AD.

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40 **815 Figure 2.** Simulation flowchart excerpts. A: Hydrothermal Carbonization. B: Anaerobic
41
42 **816**Digestion. C: Biogas cleaning and combustion. D: Hydrochar drying.

43
44
45 **817 Figure 3.** Process simulation mass and energy balance results. Top: Sc_AD; Bottom:
46
47 **818**Sc_HTC+AD. All quantities are referred per tonne of treated mixed sludge (60.0 %
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49 **819**PSS, 40.0 % SSS) per year.

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53 **820 Figure 4.** Life-cycle impact assessment results. The functional unit is 1,000 kg wet
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55 **821**mixed sludge. Values above zero represent environmental impacts produced. Values
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57 **822**below zero represent environmental impacts avoided.

823 **Tables**

824 **Table 1.** Characterization of solid feedstock.

		PSS	SSS
Proximate Analysis (% d.b.)	Moisture	85.0	85.0
	Fixed Carbon	1.60	12.7
	Volatile Matter	77.5	73.6
	Ash	20.9	13.7
Ultimate Analysis (% d.b.)	Carbon	43.0	41.5
	Hydrogen	6.10	6.00
	Nitrogen	4.20	6.80
	Chlorine	0	0
	Sulfur	0.600	0.700
	Oxygen	25.2	31.3
	Phosphorus	2.03	2.08
	Potassium	0.49	0.74
Metals (mg·kg ⁻¹ d.b.)	Aluminum	9.50·10 ³	1.57·10 ⁴
	Arsenic	2.84	0.110
	Cadmium	0.630	0
	Chromium	37.9	3.74
	Copper	400	0
	Lead	71.4	0.770
	Mercury	0.750	0
	Nickel	16.5	2.78
	Zinc	800	0
HHV (MJ·kg ⁻¹)		17.6	17.6

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Table 2. Process specifications.

Unit equipment	Feature	{Sc_AD}	{Sc_HTC+AD}
HTC Reactor	T (°C)	—	208
	P (bar)	—	20.0
	Residence time (h)	—	1.00
	Hydrochar Yield (% w/w) ^a	—	40.3
	Hydrochar composition	—	
	Proximate Analysis (% d.b.)	—	
	Moisture	—	8.00
	Fixed carbon	—	14.8
	Volatile matter	—	65.2
	Ash	—	20.1
	Ultimate Analysis (% d.b.)	—	
	Carbon	—	43.0
	Hydrogen	—	5.80
	Nitrogen	—	4.60
	Chlorine	—	0
	Sulfur	—	0.200
	Oxygen	—	26.4
	Hydrochar HHV (MJ·kg ⁻¹)	—	21.6
	Total organic carbon (g/l) ^b	—	42.5
Flash 1	T (°C)	—	158
	P (bar)	—	6.24
Flash 2	T (°C)	—	120
	P (bar)	—	1.98
Solids separator	Efficiency (%)	—	100
	Elect. Consumption (kWh) ^c	—	60.0 ^d
AD Reactor	T (°C)	35	35.0
	P (bar)	1	1.00
	Residence time (days)	21	21.0
Scrubber	Number of stages	3	3
	Cond. press. (bar)	5	5
	Column press. drop (bar)	0.1	0.100
Gas Turbine	T (°C) ^e	1,200	1,200
	P (bar) ^e	10	10.0
	Discharge P (bar)	1.50	1.50
Hydrochar dryer	T (°C)	—	53.0
	Final moisture content (% w/w)	—	8.00

^a From dry SSS feed.

^b In liquid phase product.

^c Per tonne of solid feed.

^d (Huber Technology Spain, 2019).

^e Combustion chamber.

Table 3. Parameters and assumptions considered in the techno-economic assessment.

Parameter	Value/Comment
Base year	2019 (February) CEPCI = 617.0 EUR/USD = 0.8810 ^a
Currency	EUR
Plant lifespan	20 years
Plant location	Madrid (Spain) Location Factor = 1.14 ^b
Plant construction material	316 stainless steel Material Factor (f_m) = 1.3 ^c
Capital cost estimation	Lang (Fluids-Solids process) ^c
ISBL	$ISBL = C_E + C_{er} + C_p + C_i + C_{el} + C_c + C_s + C_l$
Equipment purchase cost (C_E)	Sum of individual purchased equip.
Equipment erection	$f_{er} = 0.5$ $C_{er} = C_E (f_{er}/f_m)$
Piping	$f_p = 0.6$ $C_p = C_E f_p$
Instrumentation and control	$f_i = 0.3$ $C_i = C_E (f_i/f_m)$
Electrical works	$f_{el} = 0.2$ $C_{el} = C_E (f_{el}/f_m)$
Civil works	$f_c = 0.3$ $C_c = C_E (f_c/f_m)$
Structures and buildings	$f_s = 0.2$ $C_s = C_E (f_s/f_m)$
Lagging and paint	$f_l = 0.1$ $C_l = C_E (f_l/f_m)$
OSBL	$f_{OSBL} = 0.4$ $OSBL = ISBL f_{OSBL}$
Design and Engineering ($D\&E$)	$f_{D\&E} = 0.25$ $C_{D\&E} = ISBL f_{D\&E}$
Contingency (X)	$f_x = 0.1$ $C_x = ISBL f_x$
Total Investment Cost (TIC)	$TIC = ISBL + OSBL + D\&E + X$
Fixed production costs estimation ^c	
Operating labor (OL)	
Supervision	25% of OL
Direct salary overhead	60% of ($OL + Supervision$)
Maintenance	5% of $ISBL$
Taxes & insurance	2% of $ISBL$
Rent of land & buildings	2% of ($ISBL + OSBL$)
General plant overhead	65% of ($OL + Supervision$)
Allocated environmental charges	1% of ($ISBL + OSBL$)
Fees & royalties	1% of TIC
Capital charges	1% of TIC
Sales & marketing	—
Variable production costs estimation	
Utilities (price)	
Electricity	138.90 €·MWh ⁻¹ ^d
Water	0.49 €·t ⁻¹ ^c

Table 3 (Continued). Parameters and assumptions considered in the techno-economic assessment.

Parameter	Value/Comment
Selling price of byproducts	
Hydrochar	103.08 €·t ⁻¹ ^e
Electricity	57.29 €·MWh ⁻¹ ^f
Low Pressure Steam	14.81 €·t ⁻¹ ^g
High Pressure Steam	16.75 €·t ⁻¹ ^g

^a (PACIFIC Exchange Rate Service, 2019).

^b No specific factor for Spain was found in the literature. We utilized the one reported for Italy in Towler and Sinnott (2013), considering the similar socio-economic context of both countries.

^c (Towler and Sinnott, 2013). More details on CAPEX calculation by the Lang Method can be found in the Supplementary Material.

^d (Eurostat, 2019).

^e (Saba et al., 2019).

^f (OMIE, 2019).

^g (Pérez-Uresti et al., 2019).

Table 4. Summary of the main parameters used for life cycle inventory modelling

Parameter	Value	Unit
Anaerobic digestion		
Fugitive emissions	3.00 ^a	% of biogas
Biogas cleaning scrubber		
Fugitive emissions	1.00 ^b	% of biogas
Digestate application to land		
Digestate transportation	50.0	km
N ₂ O emissions following digestate application	3.00 ^c	% of N applied
NH ₃ emissions following digestate application	6.90 ^c	% of N applied
Nitrate emissions to ground-water following digestate application	37.5 ^c	% of N applied
Nitrate emissions to surface water following digestate application	14.0 ^c	% of N applied
P emissions to surface water following application	3.53 ^d	% of P applied
CH ₄ emissions following digestate application	0.05 ^e	% of C applied
C sequestered in soil after 100 years following digestate application	6.00 ^c	% of C applied
Plant-availability of N contained in digestate	24.5 ^c	% of N applied
Plant-availability of P contained in digestate	73.0 ^e	% of P applied
Plant-availability of K contained in digestate	100 ^c	% of K applied
Avoided mineral fertilizers production and application to land		
Mineral fertilizers transportation	50.0	km
N ₂ O emissions following mineral fertilizers application	2.00 ^e	% of N applied
Nitrate emissions to ground-water following mineral fertilizers application	10.0 ^e	% of N applied
Nitrate emissions to surface water following mineral fertilizers application	4.00 ^e	% of N applied
Plant-availability of N contained in mineral fertilizer	67.0 ^e	% of N applied
Plant-availability of P contained in mineral fertilizer	73.0	% of P applied
Plant-availability of K contained in mineral fertilizer	100	% of K applied
Hydrochar combustion in domestic stove		
Hydrochar transportation	20.0	km
Heat conversion efficiency	70.0	% of LHV

Source: ^a Yoshida et al. (2014), ^b Petersson and Wellinger (2009), ^c Bruun et al. (2016), ^d Kronvang et al. (2005), ^e Yoshida et al. (2018)

Table 5. Estimated CAPEX, OPEX and sludge treatment cost for studied scenarios.

Values referred per tonne of treated sludge if no other claim made.

	{Sc_AD} (€·t ⁻¹)	{Sc_HTC+AD} (€·t ⁻¹)
CAPEX (capital intensity)	378	516
ISBL	199	273
Purchased equipment	62.3	85.3
OSBL	79.8	109.2
Design & Engineering	69.8	95.5
Contingency	27.9	38.2
OPEX (annual)	47.3	68.5
Variable	0.270	0.220
Utilities	0.160	0.110
Consumables	0.100	0.120
Fixed	51.4	73.1
Operating labor	7.65	11.5
Supervision	1.91	2.87
Direct salary overhead	5.73	8.60
Maintenance	9.97	13.7
Taxes & insurance	3.99	5.46
Rent of land & buildings	5.58	7.64
General plant overhead	6.21	9.32
Allocated environmental charges	2.79	3.82
Fees & royalties	3.77	5.16
Capital charges	3.77	5.16
Revenues (Sales)	4.33	4.86
Hydrochar		2.68
Low Pressure Steam		0.0400
High Pressure Steam	2.62	0.600
Electricity	1.71	1.54
Cost of sludge treatment ^b	66.2	94.3
CAPEX ^c	(29 %) 18,900,000	(27 %) 25,700,000
OPEX ^c	(71 %) 47,300,000	(73 %) 68,500,000

^b Cost over 20-year plant lifespan. $Treatment\ Cost = \frac{CAPEX + OPEX_{over\ 20-years}}{Treated\ sludge_{over\ 20-years}}$

^c Total amount over 20-year plant lifespan.

Table 6. Sensitivity analysis. Results are given as net difference between Sc_AD and Sc_AD+HTC. A positive value means that the impact in Sc_AD+HTC is lower compared to Sc_AD. The units of the impact categories are those in Figure 4.

Parameter changed	GW	AC	ING	POF	TE	FE	ME
Base Case	54	0.57	1.77E-06	1.81E-01	1.21	2.15E-02	0.55
N ₂ O emission following digestate application: 1% of applied N	43	0.56	1.76E-06	1.73E-01	1.18	2.15E-02	0.55
N ₂ O emission following digestate application: 4.5% of applied N	63	0.57	1.78E-06	1.87E-01	1.24	2.15E-02	0.56
Substitution of average market for heat from natural gas	11	0.12	-3.88E-06	-1.31E-01	0.76	1.84E-02	0.51
Substitution of heat from wood pellets	-13	0.16	2.28E-06	-7.71E-02	0.99	1.88E-02	0.53
Substitution of heat from biogas co-generation plant	-14	0.20	-3.24E-06	-1.52E-01	1.14	1.83E-02	0.51

GW: global warming; AC: acidification; ING: respiratory effects, inorganics; POF: photochemical ozone formation; TE: terrestrial eutrophication; FE: freshwater eutrophication; ME: marine eutrophication

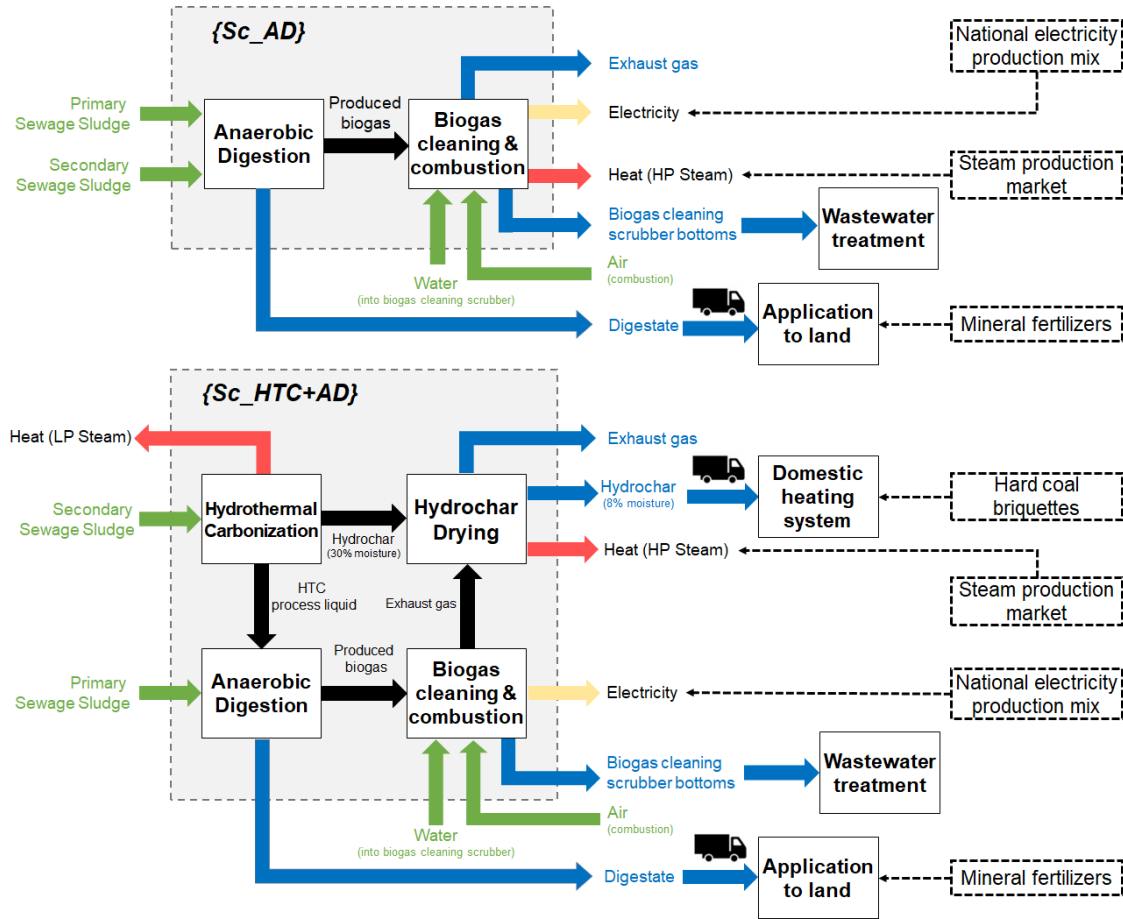
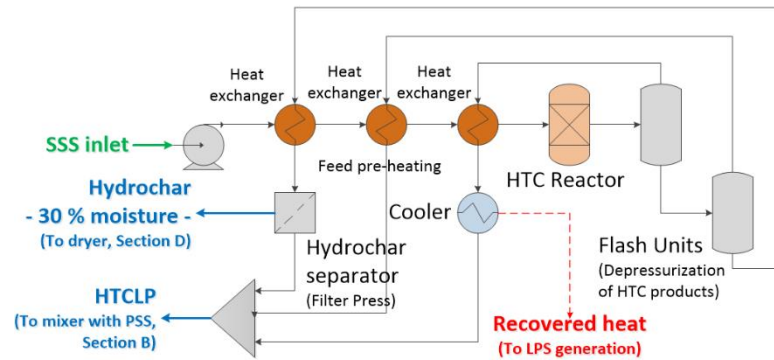


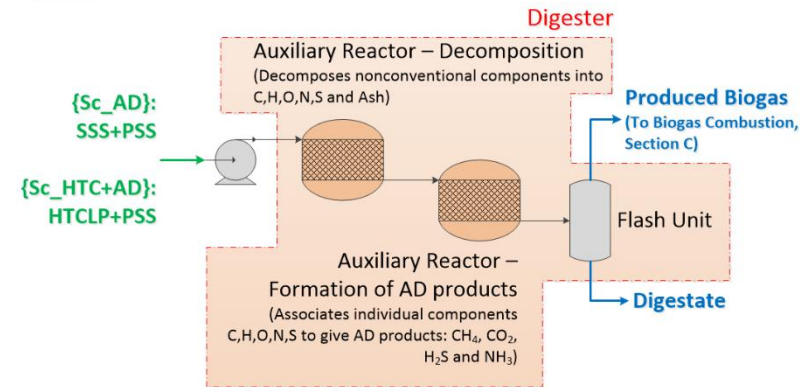
Figure 1. Block diagrams for studied technology scenarios and system boundaries.

Dotted lines represent avoided production. Top: Sc_AD; Bottom: Sc_HTC+AD.

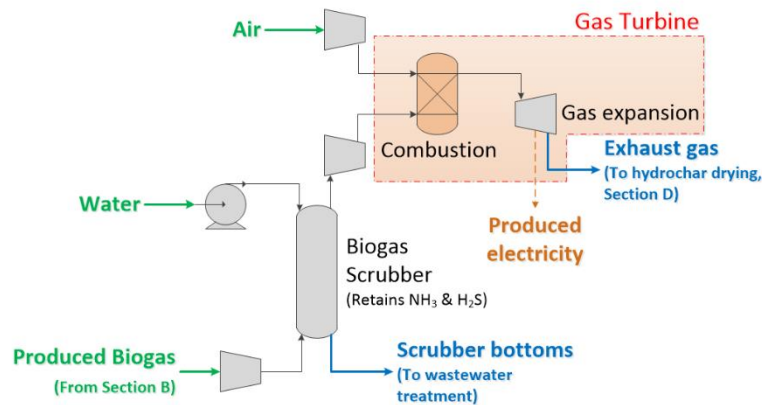
A Hydrothermal Carbonization Stage



B Anaerobic Digestion Stage



C Biogas Cleaning & Combustion Stage



D Hydrochar Drying Stage

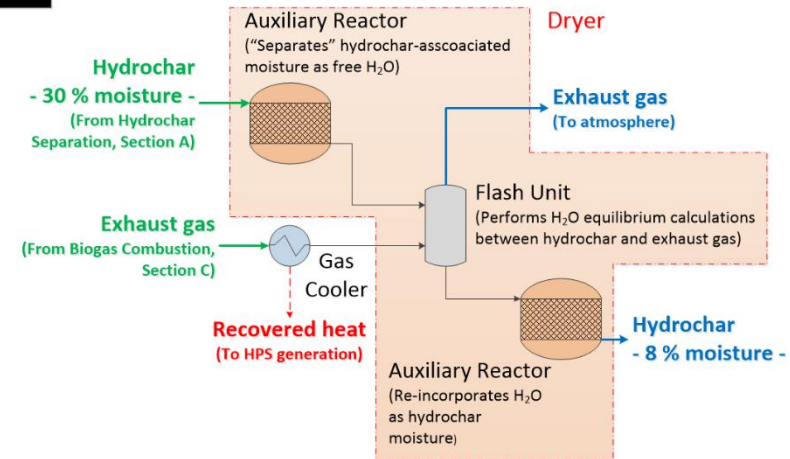


Figure 2. Simulation flowchart excerpts. A: Hydrothermal Carbonization. B: Anaerobic Digestion. C: Biogas cleaning and combustion. D: Hydrochar drying.

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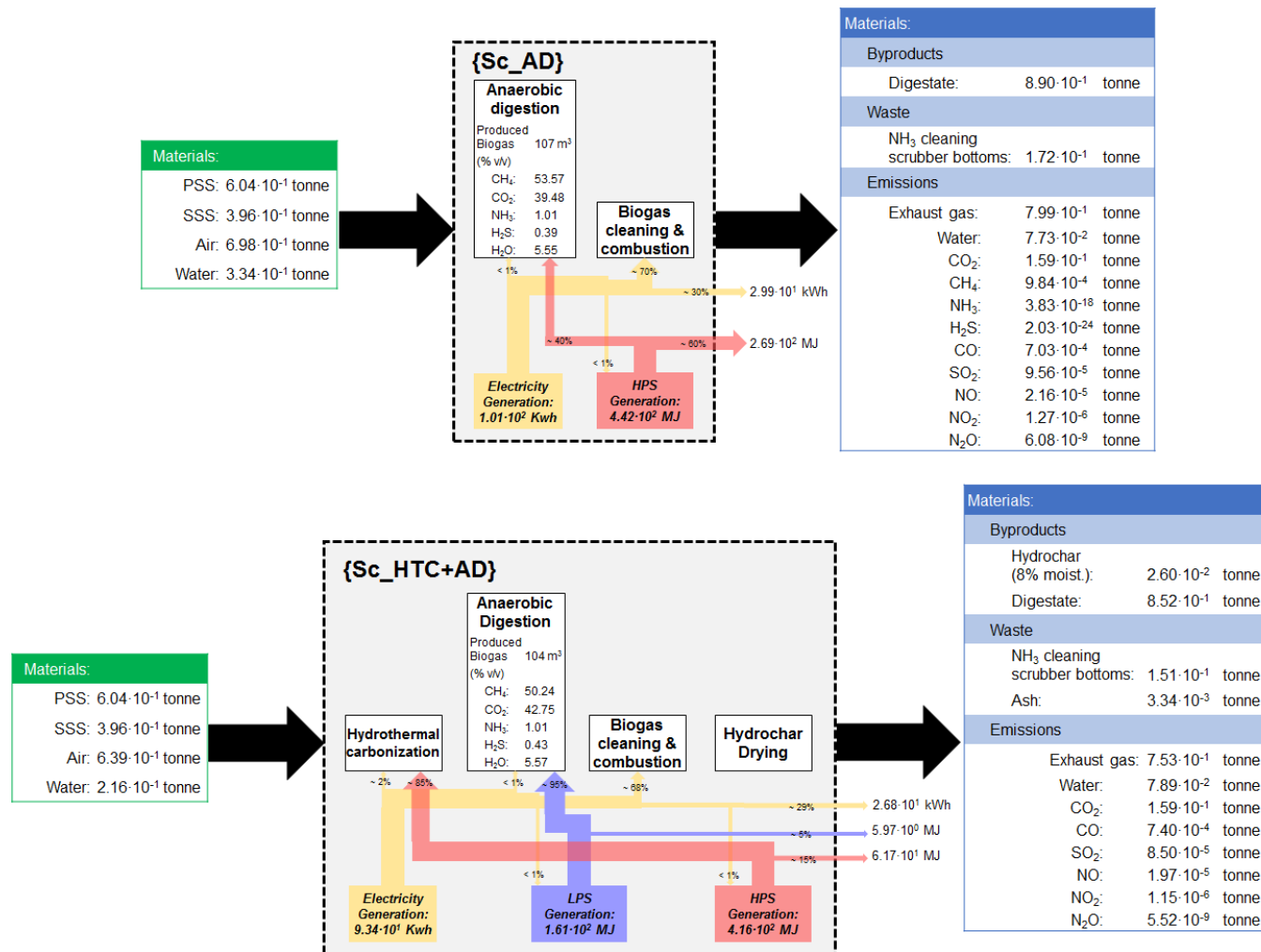


Figure 3. Process simulation mass and energy balance results. Top: Sc_AD; Bottom: Sc_HTC+AD. All quantities are referred per tonne of treated mixed sludge (60.0 % PSS, 40.0 % SSS) per year.

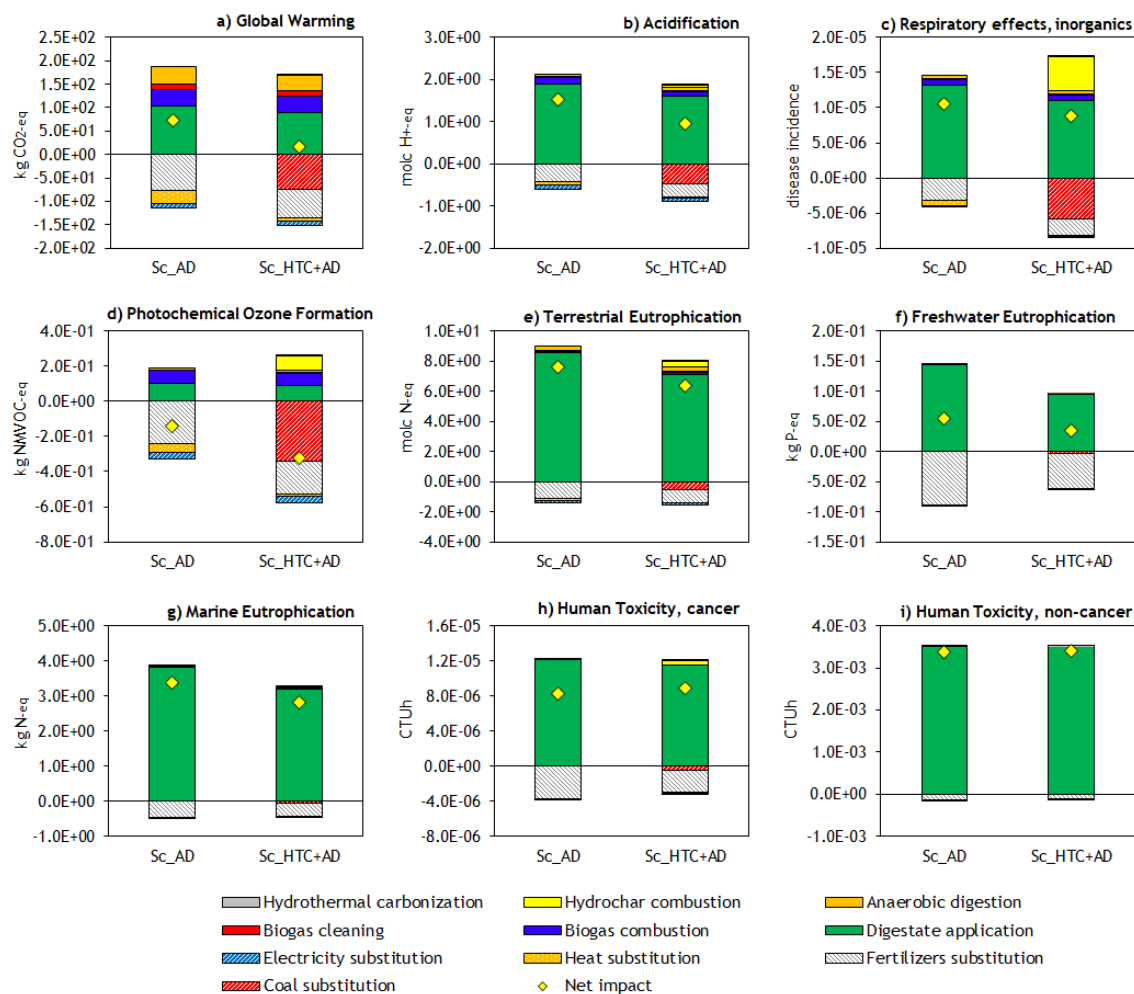
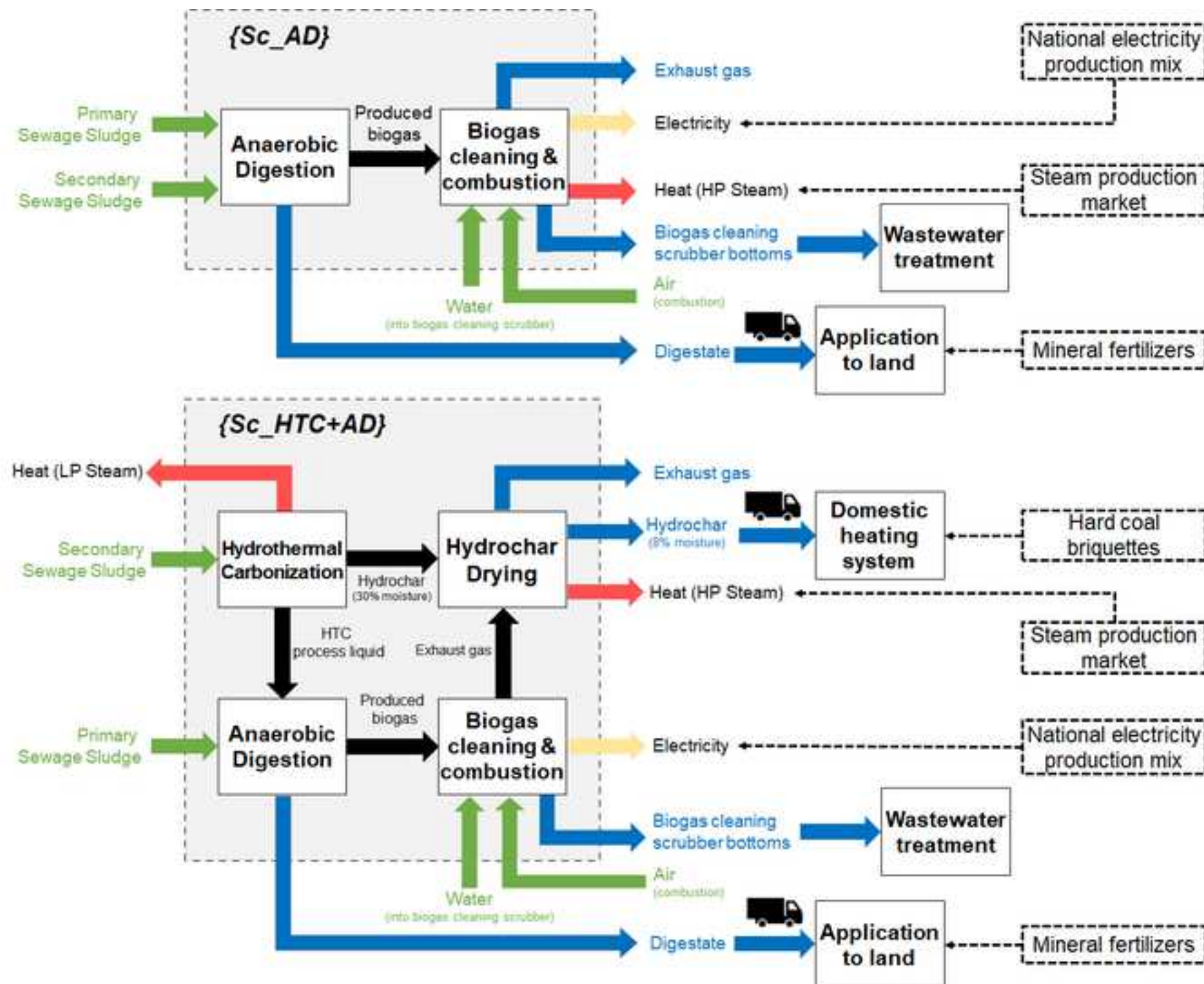
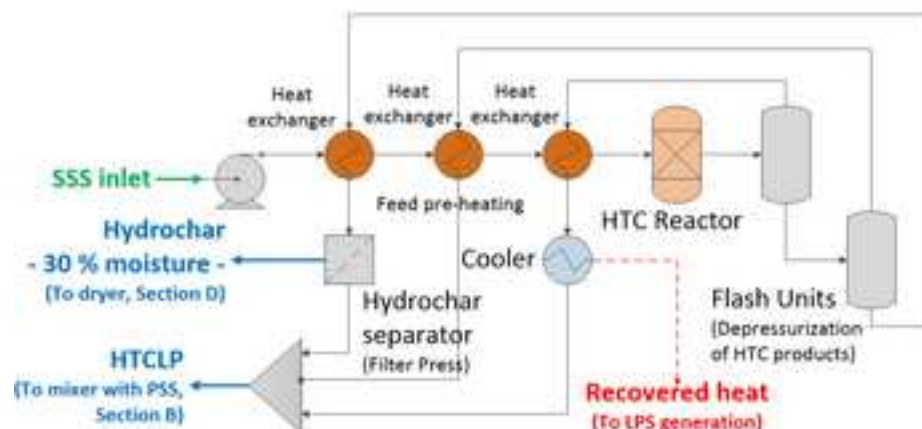


Figure 4. Life-cycle impact assessment results. The functional unit is 1,000 kg wet mixed sludge. Values above zero represent environmental impacts produced. Values below zero represent environmental impacts avoided.

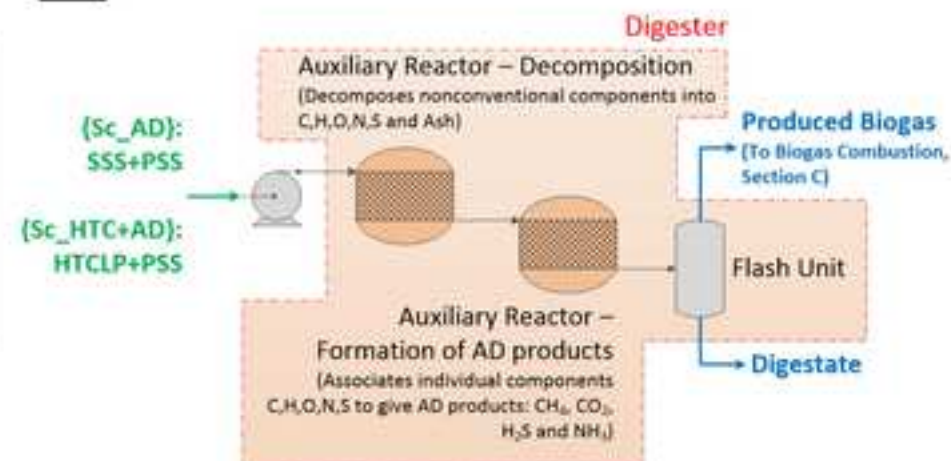
Figure 1



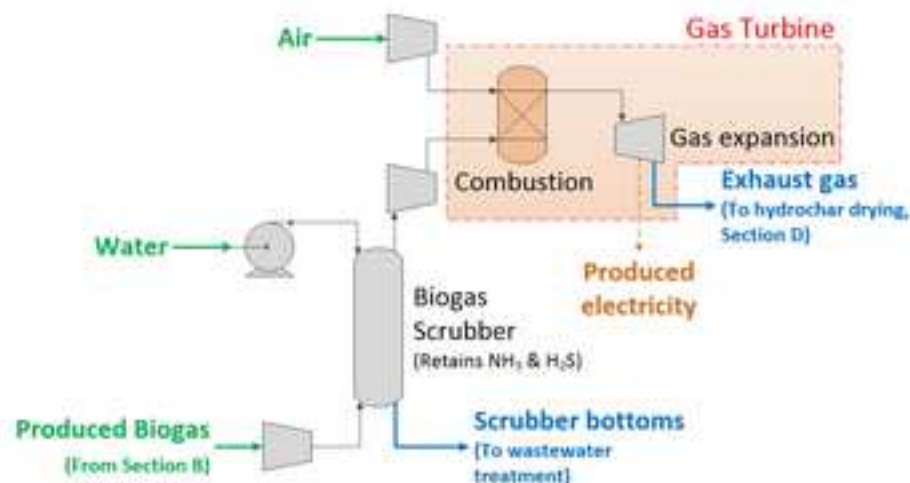
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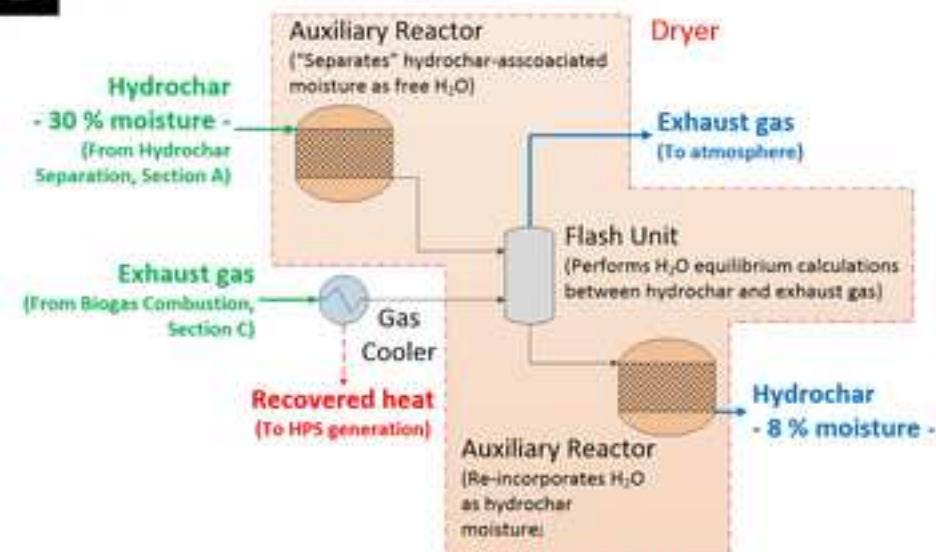


Figure 3

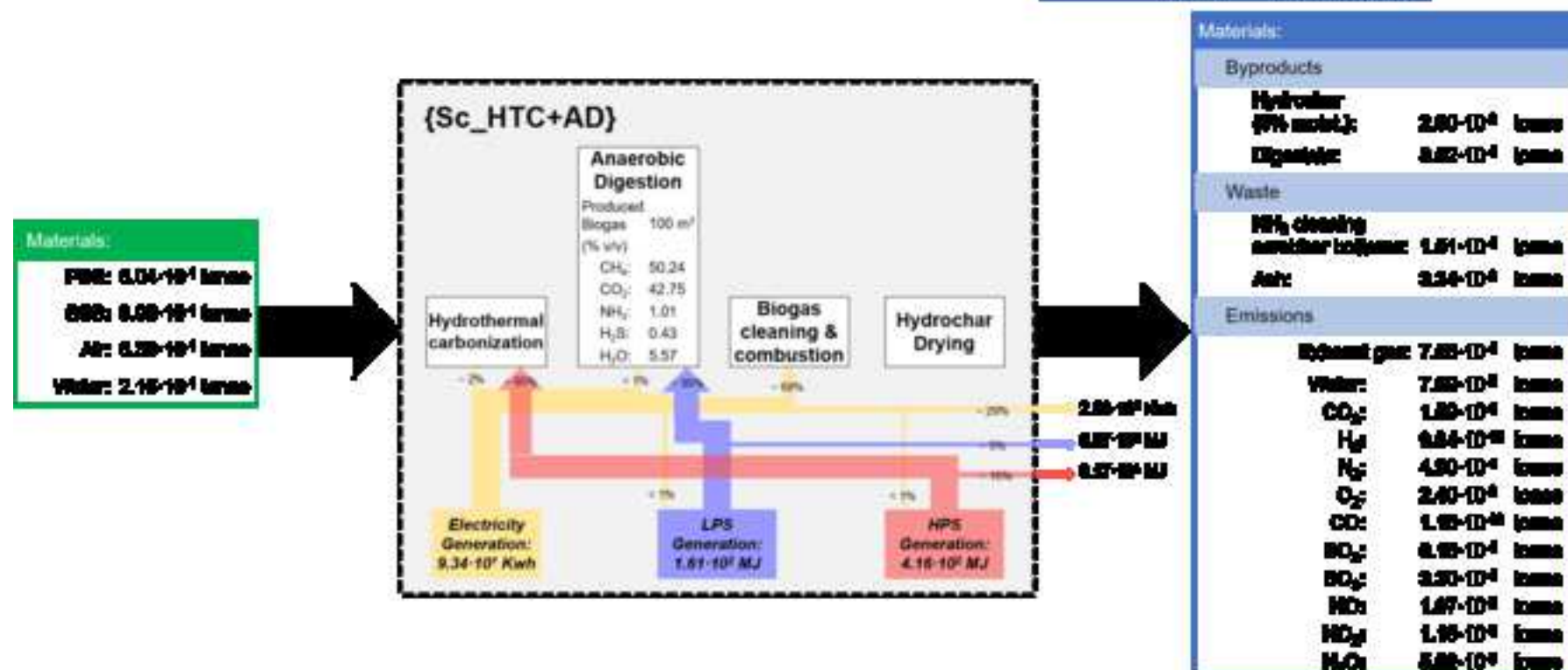
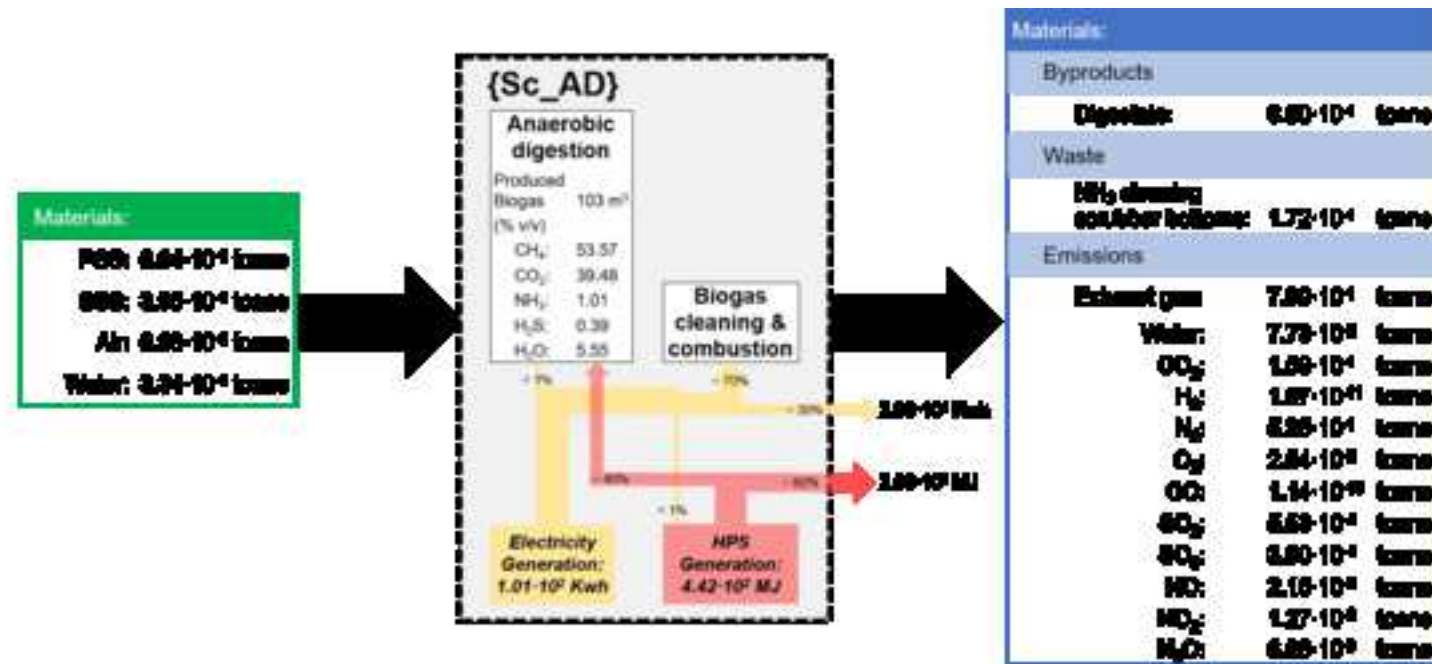
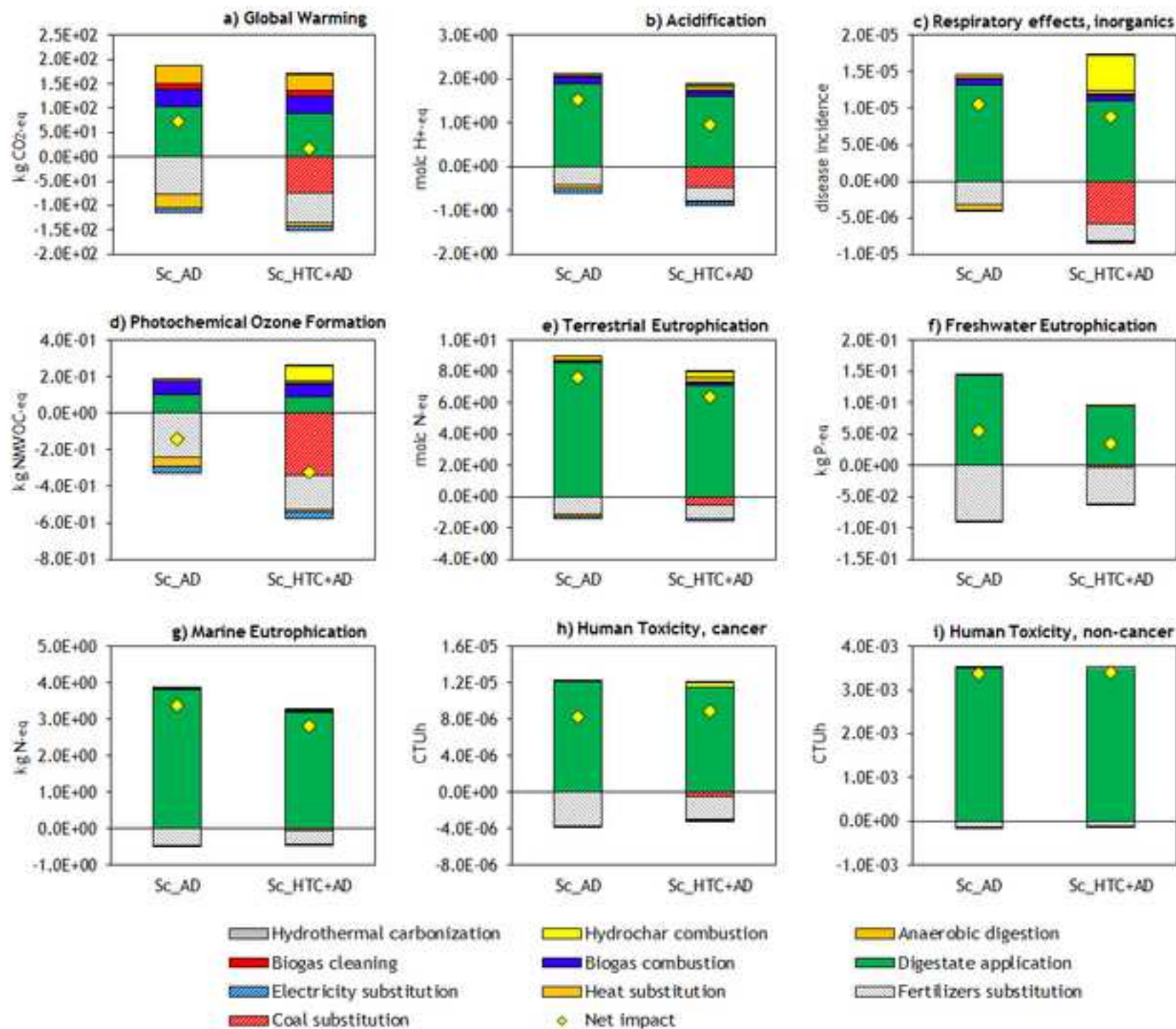


Figure 4



Declaration of interests

☒ 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.

☐ The authors declare the following financial interests/personal relationships which may be considered as potential competing interests:

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1 **Techno-economic and Life Cycle Assessment of an Integrated**

2 **Hydrothermal Carbonization System for Sewage Sludge**

3 **CRedit author statement**

4 **Enrique Medina-Martos:** Conceptualization, Methodology, Formal Analysis,
5 Investigation, Writing -Original Draft, Writing -Review &Editing

6 **Ioan-Robert Istrate:** Methodology, Formal Analysis, Investigation, Writing -Original
7 Draft, Writing -Review &Editing


8 **John A. Villamil:** Conceptualization, Investigation

9 **José-Luis Gálvez-Martos:** Supervision, Writing - Review & Editing, Formal analysis

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12



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