clean technologies

MDPI

Article

Life Cycle Environmental Assessment of a Demonstration-Scale OFMSW Biorefinery Producing Advanced Biofuels

Konstantinos Passadis [1], Giannis Pachakis [1] and Dimitris Malamis [2,*]

[1] Unit of Environmental Science and Technology, School of Chemical Engineering, National Technical University of Athens, 9 Iroon Polytechniou, Zografos, 15772 Athens, Greece; kpassadis@mail.ntua.gr (K.P.); gpachakis@mail.ntua.gr (G.P.)

[2] Department of Civil and Environmental Engineering, College of Engineering, Design & Physical Sciences, Brunel University of London, London UB8 3PH, UK

* Correspondence: dimitris.malamis@brunel.ac.uk

Abstract

Biorefineries that convert the organic fraction of municipal solid waste (OFMSW) into advanced biofuels can integrate waste management with renewable energy production. However, their environmental performance remains insufficiently characterised owing to a scarcity of life cycle assessment (LCA) studies based on real operational data. This study presents a gate-to-gate LCA of a demonstration biorefinery processing source-separated food waste into bio-oils, bioethanol, and biogas. The ReCiPe 2016 Midpoint (H) method was applied across 18 impact categories, with system expansion crediting the displacement of rapeseed oil, maize-derived ethanol, and marginal biogas-derived electricity. The net global warming potential (GWP) was 68.5 kg CO_2 eq per tonne of wet OFMSW (69% reduction from gross), placing the biorefinery 83–93% below landfilling, 63% below incineration with CHP, and above standalone anaerobic digestion systems that lack the energy-intensive drying and enzymatic hydrolysis steps of the present configuration. Bio-oil and bioethanol achieved net-negative GWP per kilogram of product (−0.89 and −0.66 kg CO_2 eq, respectively), whilst eleven of eighteen categories achieved net savings under system expansion. Enzyme production dominated the bioethanol environmental profile (37–94% across categories), whilst drying dominated bio-oil (46–93%). Monte Carlo simulation confirmed that the sign of the net impact stayed unchanged across the entire 95% confidence interval (the interval did not span zero) for 17 of 18 categories.

Keywords: biorefinery; OFMSW; life cycle assessment; advanced biofuels; SimaPro; environmental hotspots; system expansion

Academic Editor: Sabino De Gisi

Received: 30 April 2026
Revised: 9 June 2026
Accepted: 15 June 2026
Published: 17 August 2026

1. Introduction

Municipal solid waste management remains one of the most pressing environmental challenges in the European Union. According to Eurostat, approximately 225 million tonnes of municipal solid waste are generated annually across the EU member states, of which roughly one-third consists of biowaste [1,2]. At a global scale, an estimated 1.3 billion tonnes of food produced for human consumption is lost or wasted each year across the supply chain, wasting the land, water, and energy embedded in its production [3]. The organic fraction of municipal solid waste (OFMSW), comprising food waste, garden waste, and other putrescible materials, is a widely available feedstock suited to circular economy valorisation [4,5]. Unlike purpose-grown energy crops, OFMSW does not compete with food production for arable land, and its diversion from landfill, a practice increasingly

restricted under the Landfill Directive (Council Directive 1999/31/EC, as amended by Directive (EU) 2018/850) [6], simultaneously addresses waste management obligations and resource recovery objectives [7].

The conversion of biowaste into biofuels and bio-based products aligns circular economy and bioeconomy principles [8,9]. By treating waste streams as feedstocks rather than disposal liabilities, such pathways close material loops whilst generating renewable energy carriers that can displace fossil fuels [10]. Energy-driven biorefineries processing OFMSW treat a problematic waste stream while producing renewable fuels that contribute to decarbonisation targets [4,11]. OFMSW biorefineries therefore fall under both waste and energy policy, two domains that share explicit legislative links within the EU regulatory framework.

Several EU directives and regulations now support or mandate biowaste valorisation through advanced conversion pathways. The revised Waste Framework Directive (2018/851) requires all member states to implement source-separation collection of biowaste and encourages its treatment through methods that yield the highest environmental benefit [12]. The recast Renewable Energy Directive (RED III, 2023/2413), amending Directive (EU) 2018/2001 (the RED II recast) [13], sets a binding target requiring that advanced biofuels derived from feedstocks listed in Annex IX Part A, which includes the biowaste fraction of municipal solid waste, together with renewable fuels of non-biological origin (RFNBOs), contribute a combined minimum share of 5.5% of the energy supplied in the transport sector by 2030, with a 1% sub-target for RFNBOs [14]. These sectoral directives are reinforced by the EU Taxonomy Regulation (2020/852), which establishes technical screening criteria for environmentally sustainable economic activities [15], the European Green Deal, which targets climate neutrality by 2050 [16], and the Circular Economy Action Plan, which emphasises waste-to-resource pathways for sustainable production and consumption [17]. The Commission Recommendation on the Environmental Footprint method (2021/2279) promotes LCA-based approaches for substantiating environmental claims, reinforcing the case for LCA evaluation of biorefinery technologies [18]. Best Available Techniques (BAT) reference frameworks for waste treatment provide the operational counterpart to these strategic instruments [19]. Recent life cycle assessments of OFMSW to biofuel biorefinery scenarios illustrate this evaluation in practice and show that the environmental outcome depends strongly on process configuration and on methodological choices such as the avoided products assumed [20,21].

A biorefinery integrates multiple conversion technologies within a single facility to produce fuels, chemicals, materials, and energy from biomass [8,9]. Such multi-product configurations increase the value recovered from the feedstock by cascading it through successive processing stages, each targeting a different fraction of the organic matter. The environmental and process performance of such configurations is sensitive to feedstock composition, which varies with source-separation efficiency and seasonal factors [22]. A demonstration plant at the National Technical University of Athens (NTUA) implements this design. The facility processes source-separated household food waste through four sequential process units a rotary drum drying, a hexane-based solvent extraction, an SSF bioconversion, and a plug flow reactor AD to produce three energy products: bio-oils, bioethanol, and biogas [23,24]. Passadis et al. [23] published a companion cost analysis and Energy return on investment (EROI) assessment of this plant, reporting production costs of 0.44–1.46 €/kg for the liquid biofuels and an overall biorefinery EROI of 0.5–2.0 depending on the drying technology employed, with the drying stage identified as the dominant cost and energy driver. Whereas that companion study addressed the techno-economic and energy performance of the plant, its environmental performance has not been assessed. More broadly, few life cycle assessments of multi-product OFMSW biorefineries

are based on primary operational data from an operating demonstration plant; most rely on simulated or literature-derived inventories and report a single product or a single impact category. The present study addresses this gap with a multi-product, 18-category LCA built on measured demonstration-scale data, and by quantifying how the choice of avoided product governs the net result.

2. Materials and Methods

2.1. Goal and Scope Definition

This study assessed the environmental impacts of a demonstration-scale biorefinery that processes OFMSW into three energy products: bio-oils, bioethanol, and biogas. The assessment was conducted in accordance with the ISO 14040 [25] and ISO 14044 [26] standards for LCA. A summary of the goal and scope parameters is provided in Table 1.

Table 1. Goal and scope summary of the LCA study.

Parameter	Description
Goal	Assess the environmental impacts of a demonstration-scale OFMSW biorefinery producing bio-oils, bioethanol, and biogas
Standards	ISO 14040:2006 and ISO 14044:2006
Functional unit	1 tonne (1000 kg) of wet source-separated food waste (75.9% moisture)
System boundary	Gate-to-gate: reception of wet food waste to production of three energy co-products
Process units	P1: Drying; P2: Bio-oil extraction; P3: Bioconversion (SSF); P4: Anaerobic digestion
Product yields per FU	Bio-oils: 23.04 kg; Bioethanol: 26.66 kg; Biogas: 60 kg (~50 m^3)
LCIA method	ReCiPe 2016 Midpoint (H) V1.12/World (2010) H, 18 impact categories
Software and database	SimaPro 10.3.0.1; ecoinvent 3.10 (PRé Sustainability, Amersfoort, The Netherlands)
System expansion	Bio-oils → rapeseed oil; Bioethanol → maize-derived ethanol; Biogas CHP electricity → marginal biogas electricity
Sensitivity analysis	±20% on SEC, enzyme dosage; one-at-a-time + 2^2 factorial
Uncertainty analysis	Monte Carlo, 5000 iterations, 95% CI (background data only)

Avoided products were selected for the closest product-for-product equivalence to the biorefinery outputs using available ecoinvent datasets. Bio-oil is credited against crude rapeseed oil (the demonstration lipid requires further refining before it could substitute biodiesel); bioethanol against maize ethanol (the standard first-generation reference); and biogas CHP electricity against marginal biogas electricity, a deliberately conservative choice that avoids overstating the credit. See Section 2.5.

The functional unit (FU) was defined as the processing of 1 tonne (1000 kg) of wet source-separated food waste with an initial moisture content of 75.9%. This FU reflects the as-received condition of the feedstock at the biorefinery gate and is consistent with the companion cost analysis and energy assessment [23].

The system boundaries were defined as gate-to-gate [27], encompassing all conversion processes from the reception of wet food waste at the biorefinery to the production of the three energy co-products. The system comprised four process units: (P1) drying, (P2) bio-oil extraction, (P3) bioconversion to bioethanol, and (P4) AD with Combined Heat and Power (CHP) generation. The system boundaries and process flows are illustrated in Figure 1. Upstream waste collection and transport to the plant, downstream fuel distribution, capital goods, and infrastructure were excluded from the system boundaries, consistent with the companion study [23]. The fate of the digestate produced by AD was not explicitly

modelled as a separate end-of-life scenario; however, the ecoinvent process for biowaste treatment by anaerobic digestion (RoW) used to model the AD unit inherently includes default assumptions for digestate handling as part of the ecoinvent system model. The environmental burdens associated with digestate management reported in the results therefore reflect these embedded ecoinvent assumptions rather than site-specific digestate management practices. Within this dataset the fermented residue (digestate) is de-watered and stabilised by post-composting (aerobic maturation), with the associated rotting and conditioning emissions included in the inventory; under the cut-off system model no credit is taken for the resulting compost as a fertiliser or soil amendment, and direct field application of raw digestate is not separately represented.

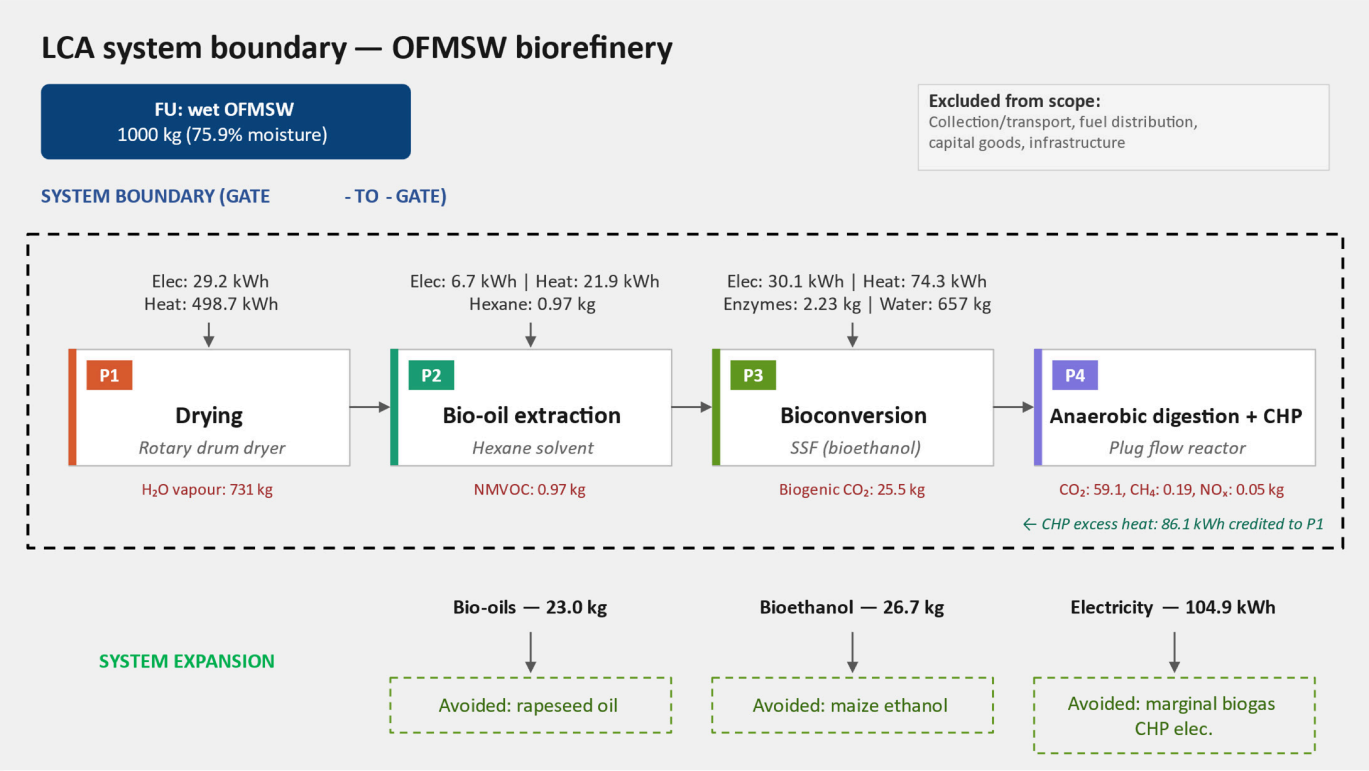

Figure 1. System boundary diagram of the CIRCforBIO demonstration-scale OFMSW biorefinery showing the four process units (P1–P4), material flows, and system expansion credits.

The LCA was performed using SimaPro 10.3.0.1 software. Background data were sourced from the ecoinvent 3.10 database using the cut-off by classification system model [28]. Life cycle impact assessment was carried out using the ReCiPe 2016 Mid-point (H) V1.12/World (2010) H method [29]. Long-term emissions were excluded from the assessment. In Figure 1 the system's boundaries are presented.

2.2. System Description

The biorefinery analysed in this study is a demonstration-scale facility established in the premises of NTUA at the Lavrion Technological and Cultural Park in Greece. The plant processes source-separated household food waste collected from the Municipality of Vari-Voula-Vouliagmeni (VVV) in Attica, Greece. A detailed description of the plant configuration, equipment specifications, and operational data is provided in Passadis et al. [23]; the following subsections summarise the four process units as modelled in the LCA. Because the present LCA draws on the same operational dataset as that companion techno-economic and EROI assessment [23], the environmental and economic analyses of the plant rest on a common inventory basis and can be read together.

P1. Drying unit. The first stage of the biorefinery employed a conventional rotary drum dryer with a processing capacity of 1 tonne per day of wet food waste. The drying unit integrates mechanical size reduction (shredding) as part of the feeding process; the electrical energy consumption reported (29.244 kWh per FU) includes the energy demand for both shredding and drying. The food waste, received at an initial moisture content of approximately 76% on a wet basis, was subjected to thermal drying to reduce its moisture content to below 10%. The physicochemical characterisation of the food waste feedstock, as determined from 23 batches totalling approximately 3 tonnes collected from VVV municipality [23], is presented in Table S4. These batches were collected across successive seasons during 2020–2021 to capture the seasonal variability of the source-separated feedstock; the corresponding seasonal characterisation is reported in detail by Tsafara et al. [30], who found that, of the structural components, only the free-glucose content varied significantly between seasons, supporting the use of a single representative feedstock composition in the present assessment. The dryer operated in direct convection mode and was powered by a biomass burner using wood chips sourced from a certified sustainable supplier (represented in the LCA by a generic ecoinvent wood-chip dataset, a conservative choice that takes no credit for the certification), with inlet air temperatures in the range of 350–500 °C. Per functional unit, the drying stage produced 269 kg of dried biomass and evaporated 731 kg of water. Electrical consumption from the Greek medium-voltage grid amounted to 29.244 kWh per FU. The thermal energy requirement was governed by the SEC, defined as the total energy input to the drying unit per unit mass of water evaporated:

$$SEC = \Sigma E / \Sigma m H_2O, \quad (1)$$

where SEC is the specific energy consumption [kWh/kg H_2O], ΣE is the total energy input to the drying unit [kWh], and $\Sigma m H_2O$ is the total mass of water evaporated (kg).

The resulting thermal energy input was 498.73 kWh from the wood chips furnace. The drying process served as a shared preparatory step and did not itself generate a final product. Its energy consumption was therefore allocated between the bio-oil and bioethanol production lines, with 80% assigned to bio-oil extraction (P2) and 20% to bioconversion (P3). The allocation of drying energy between bio-oil extraction and bioethanol production was determined by physical causality. Bio-oil extraction via hexane solvent requires feedstock moisture below 10% (wet basis), necessitating deep drying through the energy-intensive falling-rate period. In contrast, SSF bioconversion operates at 25% solid loading and does not require moisture reduction to below 10%. During the falling-rate drying period, the energy required per unit mass of water evaporated increases non-linearly as the remaining moisture transitions from free water to bound water associated with the solid matrix [31,32]. Consequently, approximately 80% of the total drying energy is consumed in achieving the low moisture content required by extraction, whilst the remaining 20% is attributable to the partial drying sufficient for bioconversion. This physical causality-based allocation differs from the equal (50/50) cost allocation adopted in the companion cost analysis and energy assessment [23], which did not distinguish between the energy demands of different drying endpoints.

P2. Bio-oil extraction. The extraction unit employed an integrated solid–liquid extraction process combined with a closed-loop solvent recovery system. Dried food waste was contacted with hexane as the extraction solvent under ambient conditions. The resulting mixture of hexane and oils (miscella) was separated by thermal distillation, with the vaporised hexane condensed and returned to dedicated storage tanks for reuse in subsequent extraction cycles. The system's thermal energy requirements were met by a centralised steam boiler supplied by a wood chips furnace. The extraction efficiency was 85%, yielding 23.038 kg of bio-oils per FU. This efficiency was determined directly from the measured

input–output mass balance of the demonstration-scale extraction unit, and therefore represents primary operational data rather than an assumed or theoretical value. Energy inputs comprised 6.65 kWh of grid electricity and 21.855 kWh of thermal energy. Hexane consumption was 0.966 kg per FU, which also represented the fugitive non-methane volatile organic compound (NMVOC) emissions to air from solvent losses [33,34]. This assumption represents a worst-case estimate, as a fraction of the make-up hexane may be retained as residual solvent in the extracted meal or bio-oil product rather than being emitted to air. No direct fugitive-emission measurements were available from the demonstration plant, so the full make-up hexane input (0.966 kg per FU) was conservatively assigned to air emissions.

P3. Bioconversion (bioethanol). Bioethanol production was based on SSF, a process combining enzymatic hydrolysis and anaerobic microbial fermentation in a single step [24,35]. The feedstock for the SSF stage was the defatted dried biomass following solvent extraction, which had undergone drying that reduced moisture content and altered the polysaccharide matrix in a manner analogous to the thermal pretreatments described for lignocellulosic feedstocks, thereby facilitating subsequent enzymatic access [36]. The bioconversion unit consisted of a bioreactor with a working volume of approximately 1.5 m^3, equipped with indirect heating and cooling via a double-jacketed wall. The process employed Saccharomyces cerevisiae as the fermentation microorganism, supplemented with commercial enzyme preparations (CellicCTec3 cellulase and Spirizyme Excel HS amylase; both supplied by Novozymes A/S, Bagsværd, Denmark), at a total enzyme dosage of 2.231 kg per FU. The optimised operating conditions comprised a solid loading of 25%, a temperature of 35 °C, and a fermentation time of 18 h. Water input amounted to 657 kg per FU. The process yielded 26.656 kg of bioethanol per FU at a concentration subsequently suitable for recovery by distillation [37], with energy inputs of 30.07 kWh of grid electricity and 74.326 kWh of thermal energy from a wood chips furnace. Fugitive emissions consisted of 25.508 kg of biogenic CO_2 released as fermentation off-gas.

P4. Anaerobic digestion. The stillage remaining after bioethanol recovery (873 kg) was directed to an AD unit consisting of a plug flow reactor (PFR) with a total volume of 5 m^3. The bioreactor operated under mesophilic conditions (approximately 40 °C) with a hydraulic retention time of 12–15 days. The AD process produced 60 kg of biogas per FU (equivalent to approximately 50 m^3), consistent with reported biogas yields from food waste in small-to-mid-scale digesters [38,39]. The biogas was assumed to be utilised in a CHP system operating at an overall efficiency of 86%, with an electrical efficiency of 37% and a thermal efficiency of 49%, consistent with literature values for biogas CHP applications [23,40]. This yielded 104.89 kWh of electricity per FU. The recovered excess thermal energy was considered to be available for use in the drying process. In the LCA model, 86.07 kWh of CHP excess heat was credited against the thermal energy demand of the drying unit, reducing the external wood chips heat requirement accordingly. Freight transport of 50 tkm (lorry 16–32 t, EURO 6) was included to account for internal material handling and logistics associated with the AD feedstock within the facility boundaries. Fugitive emissions from CHP combustion comprised 59.104 kg of biogenic CO_2, 0.19 kg of biogenic CH_4 (representing methane slip of approximately 1% of total methane mass flow), and 0.05245 kg of NO_x. The biogas was modelled as utilised on-site in the CHP unit to cogenerate electricity and heat, consistent with the as-built plant configuration; biomethane upgrading is an alternative valorisation route with a different energy-substitution profile.

2.3. Life Cycle Inventory

The life cycle inventory (LCI) was compiled primarily from operational data obtained during demonstration-scale trials at the biorefinery, complemented by ecoinvent 3.10 background processes for upstream inputs. All mass and energy balances were nor-

malised to the functional unit of 1 tonne of wet food waste. The complete LCI is presented in Table S1.

Foreground data encompassed measured mass flows, energy consumption, material inputs, and product yields for all four process units. Where direct measurements were unavailable or deemed unrepresentative at the demonstration scale, the literature data and engineering assumptions were applied [23]. The generic ecoinvent enzyme production process was used as a proxy for the specific commercial preparations (CellicCTec3 and Spirizyme Excel HS), as manufacturer-specific LCI data for these proprietary formulations are not publicly available. This proxy may over or underestimate the actual environmental footprint of the commercial preparations, and this uncertainty should be considered when interpreting the enzyme impact.

Fugitive emissions to air were modelled for three process units. In P2, NMVOC emissions of 0.966 kg per FU arose from hexane solvent losses during the extraction cycle. In P3, biogenic CO_2 emissions of 25.508 kg per FU were released as fermentation off-gas during the SSF process. In P4, CHP combustion generated 59.104 kg of biogenic CO_2, 0.19 kg of biogenic CH_4 from methane slip, and 0.05245 kg of NO_x. The total biogenic CO_2 emission across the biorefinery amounted to 84.612 kg per FU. Biogenic CO_2 was treated in accordance with the ReCiPe 2016 characterisation factors, which assign a global warming potential of zero to biogenic CO_2 from short-cycle biomass sources [29]; we note that this convention has been contested on the grounds that atmospheric residence before biomass regrowth contributes to warming [41]. All biomass inputs into the system are short-cycle biogenic carbon sources. The associated supply chain emissions, including any land-use related impacts, are captured within the respective ecoinvent background processes. Biogenic methane (CH_4) from CHP methane slip was characterised using the same global warming potential (GWP_{100}) characterisation factor as fossil methane under the ReCiPe 2016 (H) method.

Data quality varied across the inventory. Foreground data for mass flows, product yields, and direct energy consumption were obtained from demonstration-scale trials and are considered of high quality with respect to technological and temporal representativeness. Background data from ecoinvent 3 are geographically representative for European conditions, with the exception of the biowaste treatment and fodder yeast processes, which use rest-of-world (RoW) datasets. The ecoinvent enzyme production process is a generic proxy and may not fully represent the specific commercial preparations used. A complete list of all ecoinvent 3.10 datasets employed in the foreground and background systems is provided in Table S3. Because these RoW datasets represent global-average rather than country-specific conditions, they may not fully reflect Greek or European biowaste-treatment and fermentation practice; the impacts attributed to these processes should therefore be read as approximate, a recognised geographic-representativeness limitation of the present inventory.

2.4. Life Cycle Impact Assessment

The environmental impacts were evaluated using the ReCiPe 2016 Midpoint (H) V1.12 method with the World (2010) H set of characterisation factors [29]. This hierarchist perspective was selected as it represents a consensus time horizon and is widely adopted in waste management LCA studies [42,43]. The assessment covered all 18 midpoint impact categories: global warming potential (GWP, kg CO_2 eq); stratospheric ozone depletion potential (ODP, kg CFC-11 eq); ionising radiation potential (IRP, kBq Co-60 eq); ozone formation—human health (HOFP, kg NO_x eq); fine particulate matter formation potential (PMFP, kg $PM_{2.5}$ eq); ozone formation—terrestrial ecosystems (EOFP, kg NO_x eq); terrestrial acidification potential (TAP, kg SO_2 eq); freshwater eutrophication potential (FEP, kg

P eq); marine eutrophication potential (MEP, kg N eq); terrestrial ecotoxicity potential (TETP, kg 1,4-DCB); freshwater ecotoxicity potential (FETP, kg 1,4-DCB); marine ecotoxicity potential (METP, kg 1,4-DCB); human carcinogenic toxicity (HTPc, kg 1,4-DCB); human non-carcinogenic toxicity (HTPnc, kg 1,4-DCB); land use potential (LOP, m^2a crop eq); mineral resource scarcity (SOP, kg Cu eq); fossil resource scarcity (FFP, kg oil eq); and water consumption potential (WCP, m^3).

Normalisation was performed using World (2010) reference values [44,45] to identify the relative significance of the impact categories at the global scale. Long-term emissions were excluded from the characterisation step. This exclusion follows standard practice in waste management LCA and avoids the introduction of highly uncertain long-term emission estimates (typically exceeding 100-year time horizons) that can disproportionately affect toxicity categories without improving decision-making.

2.5. System Expansion

The multi-functionality of the biorefinery—producing three energy co-products from a single waste feedstock—was addressed through system expansion (avoided burden), in accordance with the ISO 14044 hierarchy for resolving co-product allocation [26,46,47]. Under this approach, the environmental burdens avoided by displacing conventional products were credited to the biorefinery system. The net environmental impact was calculated as:

$$I\_net = I\_gross - I\_avoided, \quad (2)$$

where I_net is the net characterised impact for a given category, I_gross is the gross impact encompassing all burdens of the biorefinery without credits, and I_avoided is the characterised impact of the displaced conventional product.

Three avoided products were defined, with displacement quantities determined on the basis of product equivalence:

1. Bio-oils displacing rapeseed oil. The 23.038 kg of bio-oils produced per FU were credited as displacing an equivalent mass of crude rapeseed oil (solvent-grade, market mix, at regional storage, RER). Rapeseed oil was selected as the avoided product because the extracted lipids serve as a functional substitute for vegetable oil in solvent and chemical applications. This mass-based substitution reflects direct product equivalence rather than energy-equivalent displacement. The demonstration bio-oil is a crude extracted lipid that requires further refining and transesterification before it could serve as biodiesel; at this processing stage it is functionally equivalent to crude vegetable oil rather than to a finished fuel, which makes rapeseed oil the appropriate product-for-product substitute and the dominant vegetable-oil benchmark in the European market, allowing the waste-derived (second-generation) lipid to be compared against a conventional first-generation route. Food-waste derived bio-oil may nonetheless differ from refined rapeseed oil in properties such as oxidative stability, free fatty acid content, and impurity profile, so the substitution is made on a mass and primary-function basis rather than implying identical fuel quality.
2. Bioethanol displacing maize-derived ethanol. The 26.656 kg of bioethanol produced per FU were credited as displacing an equivalent mass of ethanol from maize fermentation (RoW). Maize ethanol was selected as the marginal conventional production route for fuel-grade ethanol, representing the product that OFMSW-derived bioethanol would most plausibly displace in the European market. Maize ethanol is the standard first-generation bioethanol reference, so this choice frames the waste-derived (second-generation) ethanol against the conventional first-generation route; wheat-based ethanol is a regional European alternative but is less established as a benchmark. The

displaced ethanol is modelled using the ecoinvent "ethanol production from maize" (RoW) dataset, listed in full in Table S3.

3. Biogas CHP electricity displacing marginal electricity. The 104.89 kWh of electricity generated by the biogas CHP system was credited as displacing an equivalent quantity of electricity from heat and power co-generation with biogas and gas engine (GR, high voltage). This avoided product was deliberately selected as the marginal technology rather than the average Greek grid mix. Since biogas-derived CHP electricity already possesses a low fossil carbon intensity, this substitution represents the most conservative system expansion choice and yields a smaller credit than displacing grid-average electricity or natural gas combined-cycle generation [47–49].

Throughout the study, results were reported under two complementary system views: "gross" (absolute burdens without credits) and "net" (with system expansion credits). This dual reporting enables transparent evaluation of both the intrinsic environmental performance and the substitution benefits of the system.

2.6. Sensitivity Analysis

A parametric sensitivity analysis was conducted to evaluate the influence of key operational variables on the environmental performance of the biorefinery. Two parameters were selected on the basis of their expected contribution to environmental impacts and their inherent variability at the demonstration scale:

1. SEC of drying. The base-case SEC of 0.8 kWh/kg H_2O was varied over the range 0.64–0.96 kWh/kg H_2O, corresponding to ±20% of the base case.
2. Enzyme usage. The base-case enzyme dosage of 2.231 kg per FU was varied over the range 1.785–2.677 kg (±20%). Enzyme production is known to carry substantial upstream environmental burdens [50–52], and dosage optimisation represents a realistic improvement pathway.

Two complementary designs were employed. First, a one-at-a-time approach varied each parameter individually to its low (−20%) and high (+20%) values whilst the remaining parameter was held at base case. Second, a full 2^2 factorial design evaluated all combinations of the low and high settings for the two parameters, yielding four combined scenarios. Results were expressed as percentage change from the base-case characterised impact for each of the 18 midpoint categories.

2.7. Monte Carlo Uncertainty Analysis

Stochastic uncertainty analysis was performed using Monte Carlo simulation to quantify the propagation of parameter uncertainty through the LCA model [53]. A total of 5000 iterations were executed within SimaPro, with uncertainty propagated through all ecoinvent background processes using the default pedigree matrix uncertainty distributions, an approach widely adopted for stochastic uncertainty analysis in LCA [54]. The 95% confidence interval was adopted as the basis for reporting. The Monte Carlo simulation propagated uncertainty through the ecoinvent background processes only; foreground inventory parameters (SEC, enzyme dosage, CH4 slip) retained their deterministic base-case values during the simulation. Foreground parameter variability was instead addressed through the deterministic sensitivity analysis (Section 2.6). The sensitivity analysis addresses foreground parameter influence; the Monte Carlo analysis addresses background data uncertainty.

For each of the 18 midpoint impact categories, the following summary statistics were computed: mean, median, standard deviation (SD), and the 2.5th and 97.5th percentile

values defining the 95% confidence interval. The coefficient of variation (CV) was calculated to provide a standardised measure of relative dispersion:

$$CV = (SD/|\mu|) \times 100, \quad (3)$$

where CV is the coefficient of variation [%], SD is the standard deviation of the Monte Carlo distribution, and μ is the mean value. The CV enables comparison of uncertainty across impact categories irrespective of their absolute magnitudes, with lower values indicating greater robustness of the LCA results.

3. Results

3.1. Product-Level Environmental Profiles

The characterised environmental impacts per kilogram of each product under both gross and net system views are presented in Table 2.

Table 2. Characterised environmental impacts per kilogram of product under gross and net system views.

Impact Category	Unit	Bio-Oil Gross	Bio-Oil Net	Bioethanol Gross	Bioethanol Net	Biogas Gross	Biogas Net
GWP	kg CO_2 eq	1.154	−0.892	2.300	−0.665	2.214	1.779
ODP	kg CFC-11 eq	2.57×10^{-6}	-2.21×10^{-5}	7.26×10^{-6}	-1.46×10^{-5}	5.52×10^{-6}	4.77×10^{-6}
IRP	kBq Co-60 eq	6.78×10^{-3}	4.39×10^{-3}	1.23×10^{-2}	7.80×10^{-3}	6.95×10^{-4}	6.26×10^{-5}
HOFP	kg NO_x eq	2.01×10^{-2}	1.41×10^{-2}	8.82×10^{-3}	1.16×10^{-3}	2.15×10^{-3}	1.38×10^{-3}
PMFP	kg $PM_{2.5}$ eq	3.70×10^{-3}	-9.38×10^{-4}	5.04×10^{-3}	1.75×10^{-4}	1.07×10^{-3}	4.86×10^{-4}
EOFP	kg NO_x eq	2.51×10^{-2}	1.44×10^{-2}	9.09×10^{-3}	1.11×10^{-3}	2.24×10^{-3}	1.42×10^{-3}
TAP	kg SO_2 eq	7.45×10^{-3}	-2.84×10^{-2}	1.46×10^{-2}	-2.86×10^{-3}	2.72×10^{-3}	1.13×10^{-3}
FEP	kg P eq	1.10×10^{-4}	-5.57×10^{-4}	4.13×10^{-4}	4.25×10^{-6}	3.43×10^{-5}	2.53×10^{-5}
MEP	kg N eq	1.55×10^{-5}	-4.27×10^{-3}	1.67×10^{-3}	-1.10×10^{-3}	1.69×10^{-5}	1.54×10^{-5}
TETP	kg 1,4-DCB	9.314	−9.328	9.366	3.913	3.253	0.465
FETP	kg 1,4-DCB	8.79×10^{-4}	−0.326	2.76×10^{-2}	5.96×10^{-3}	5.39×10^{-4}	1.27×10^{-4}
METP	kg 1,4-DCB	6.88×10^{-3}	−0.276	1.71×10^{-2}	5.26×10^{-3}	2.49×10^{-3}	4.51×10^{-4}
HTPc	kg 1,4-DCB	8.50×10^{-3}	5.45×10^{-3}	1.10×10^{-2}	1.62×10^{-3}	2.23×10^{-3}	1.31×10^{-3}
HTPnc	kg 1,4-DCB	1.306	−2.147	1.744	1.052	0.118	0.051
LOP	m^2a crop eq	2.331	−4.218	3.191	−1.129	0.027	0.004
SOP	kg Cu eq	3.79×10^{-3}	-1.65×10^{-4}	7.76×10^{-3}	-3.06×10^{-3}	1.54×10^{-3}	5.35×10^{-4}
FFP	kg oil eq	0.374	0.070	0.581	−0.058	0.189	0.098
WCP	m^3	7.89×10^{-3}	3.44×10^{-4}	0.143	−0.117	1.19×10^{-3}	4.64×10^{-4}

For some impact categories the net value is larger in magnitude than the corresponding gross value (e.g., FETP). This occurs when the displaced (avoided) product carries a higher characterised burden in that category than the biorefinery process itself, so that the system-expansion credit exceeds the gross burden and the net result becomes strongly negative.

Figure 2 compares the GWP across the three products. Bio-oil and bioethanol achieved net-negative GWP (−0.89 and −0.66 kg CO_2 eq per kg, respectively) after crediting the avoided rapeseed oil and maize-derived ethanol; whereas, biogas retained a net GWP of 1.78 kg CO_2 eq per kg owing to the smaller CHP electricity credit. Detailed process contribution breakdowns for all 18 impact categories are provided in the Supplementary Material (Figures S2–S4).

Drying dominated the bio-oil gross profile (46–93% across categories), enzyme manufacturing dominated bioethanol (37–94%), and digestate disposal dominated biogas (77–99%). Fugitive emissions across the three subsystems are presented in Figure S1. The contrast between environmental and economic hotspots was most pronounced for bioethanol: drying accounts for 49% of production cost [23], but enzyme production accounts for 37–94% of the environmental burden.

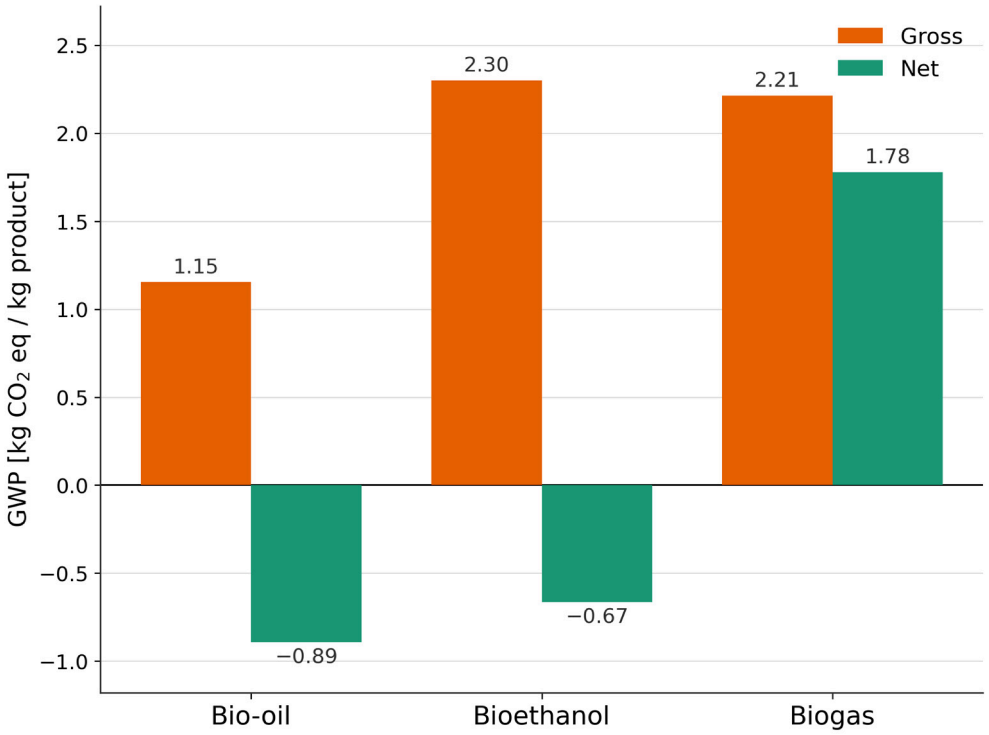

Figure 2. Global warming potential per kilogram of product under gross and net system views.

3.2. System-Level Assessment per Functional Unit

Table 3 presents the characterised impacts for all 18 ReCiPe midpoint categories under both gross and net system views.

The net GWP amounted to 68.5 kg CO_2 eq, representing a 69% reduction from the gross value of 220.7 kg CO_2 eq through system expansion credits. For transparency and comparability, the product yields on a dry-matter basis were: 95 kg bio-oils per tonne dry matter, 122 kg bioethanol per tonne dry matter, and 299 m^3 biogas per tonne dry matter.

The system expansion credits were markedly larger than those observed under the previous fossil fuel displacement approach [23], because the avoided products (rapeseed oil, maize-derived ethanol) carry substantially higher cradle-to-gate environmental footprints than their fossil fuel counterparts (diesel, petrol). As a result, eleven of the eighteen impact categories achieved net environmental savings under the revised system expansion: ODP, TAP, FEP, MEP, TETP, FETP, METP, HTPnc, LOP, SOP, and WCP. The remaining seven categories (GWP, IRP, HOFP, PMFP, EOFP, HTPc, FFP) retained net positive burdens, though all were substantially reduced relative to the gross values.

Figure 3 presents the product-level contributions to the gross and net characterised impacts at the FU level. The contribution profile varied markedly across impact categories, reflecting the distinct hotspot structures of the three products. Biogas was the dominant contributor to gross GWP (60%), driven by digestate disposal. Bioethanol dominated FEP (71%), MEP (97%), and WCP (94%), consistent with the upstream enzyme demands. Bio-oil dominated HOFP (56%) and EOFP (61%), driven by drying combustion and hexane NMVOC emissions.

Figure 4 presents the absolute characterised values under both gross and net views on a symmetric logarithmic scale. The largest gross values were TETP (659 kg 1,4-DCB), GWP (221 kg CO_2 eq), and LOP (140 m^2a crop eq). Under the net view, several categories reversed sign: TETP, FETP, METP, HTPnc, LOP, and WCP shifted from positive to negative, confirming that the system expansion credits exceeded the gross burdens for these pathways.

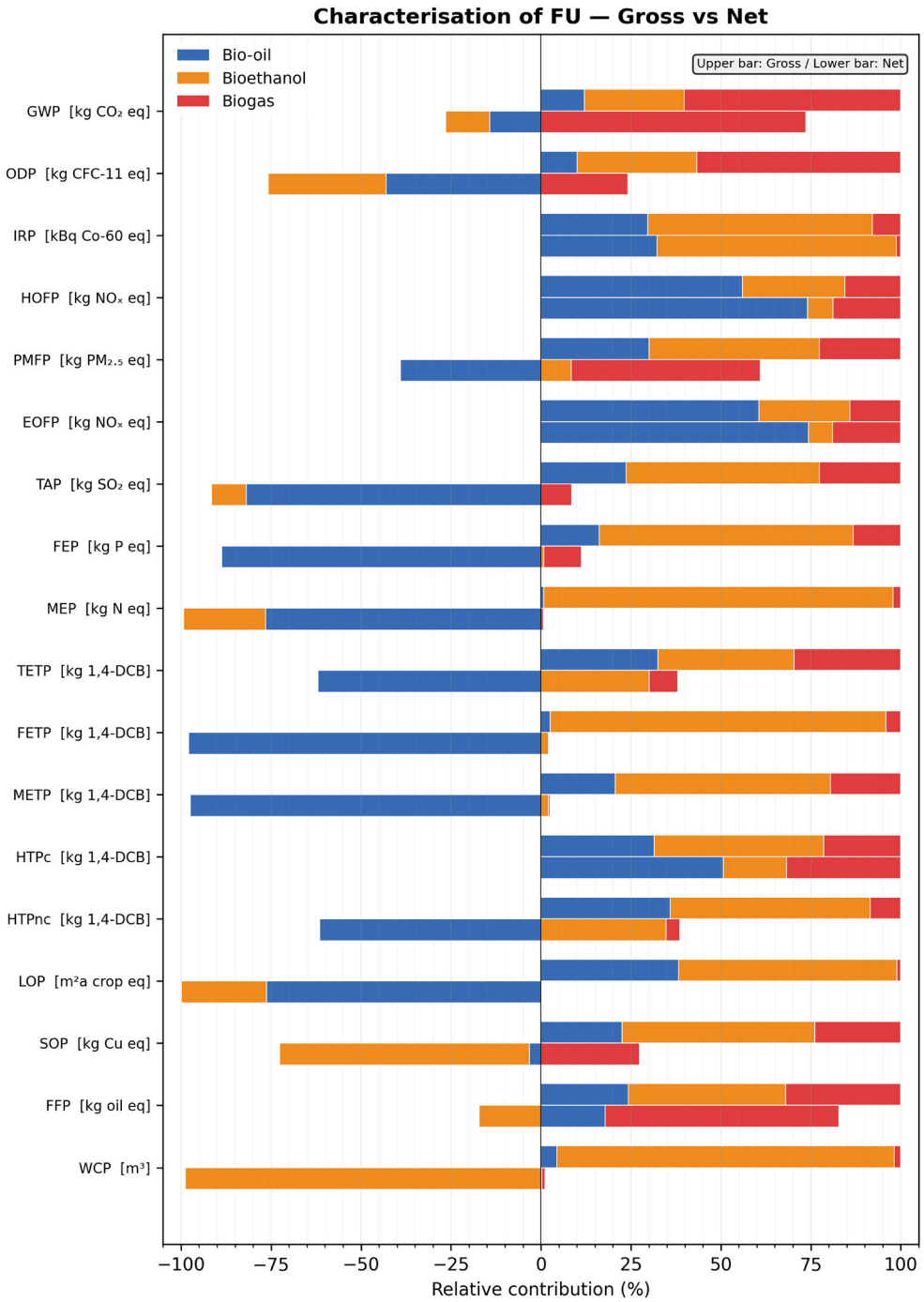

Figure 3. Product-level contributions to gross and net characterised impacts at the functional unit level (1 tonne wet OFMSW).

Table 3. Characterised environmental impacts per functional unit (1 tonne wet OFMSW) under gross and net system views.

Impact Category	Abbr.	Unit	Gross	Net	Δ (%)
Global warming	GWP	kg CO_2 eq	220.7	68.5	−69
Stratospheric ozone depletion	ODP	kg CFC-11 eq	5.84×10^{-4}	-6.10×10^{-4}	−204
Ionising radiation	IRP	kBq Co-60 eq	0.527	0.313	−41
Ozone formation, human health	HOFP	kg NO_x eq	0.828	0.439	−47
Fine particulate matter	PMFP	kg $PM_{2.5}$ eq	0.283	0.012	−96

Table 3. *Cont.*

Impact Category	Abbr.	Unit	Gross	Net	Δ (%)
Ozone formation, terrestrial ecosystem	EOFP	kg NO_x eq	0.955	0.447	−53
Terrestrial acidification	TAP	kg SO_2 eq	0.723	−0.662	−192
Freshwater eutrophication	FEP	kg P eq	0.0156	−0.0112	−172
Marine eutrophication	MEP	kg N eq	0.0459	−0.127	−377
Terrestrial ecotoxicity	TETP	kg 1,4-DCB	659.4	−82.7	−113
Freshwater ecotoxicity	FETP	kg 1,4-DCB	0.787	−7.35	−1034
Marine ecotoxicity	METP	kg 1,4-DCB	0.765	−6.20	−910
Human carcinogenic toxicity	HTPc	kg 1,4-DCB	0.623	0.247	−60
Human non-carcinogenic toxicity	HTPnc	kg 1,4-DCB	83.6	−18.4	−122
Land use	LOP	m^2a crop eq	140.4	−127.0	−190
Mineral resource scarcity	SOP	kg Cu eq	0.386	−0.053	−114
Fossil resource scarcity	FFP	kg oil eq	35.4	5.92	−83
Water consumption	WCP	m^3	4.06	−3.08	−176

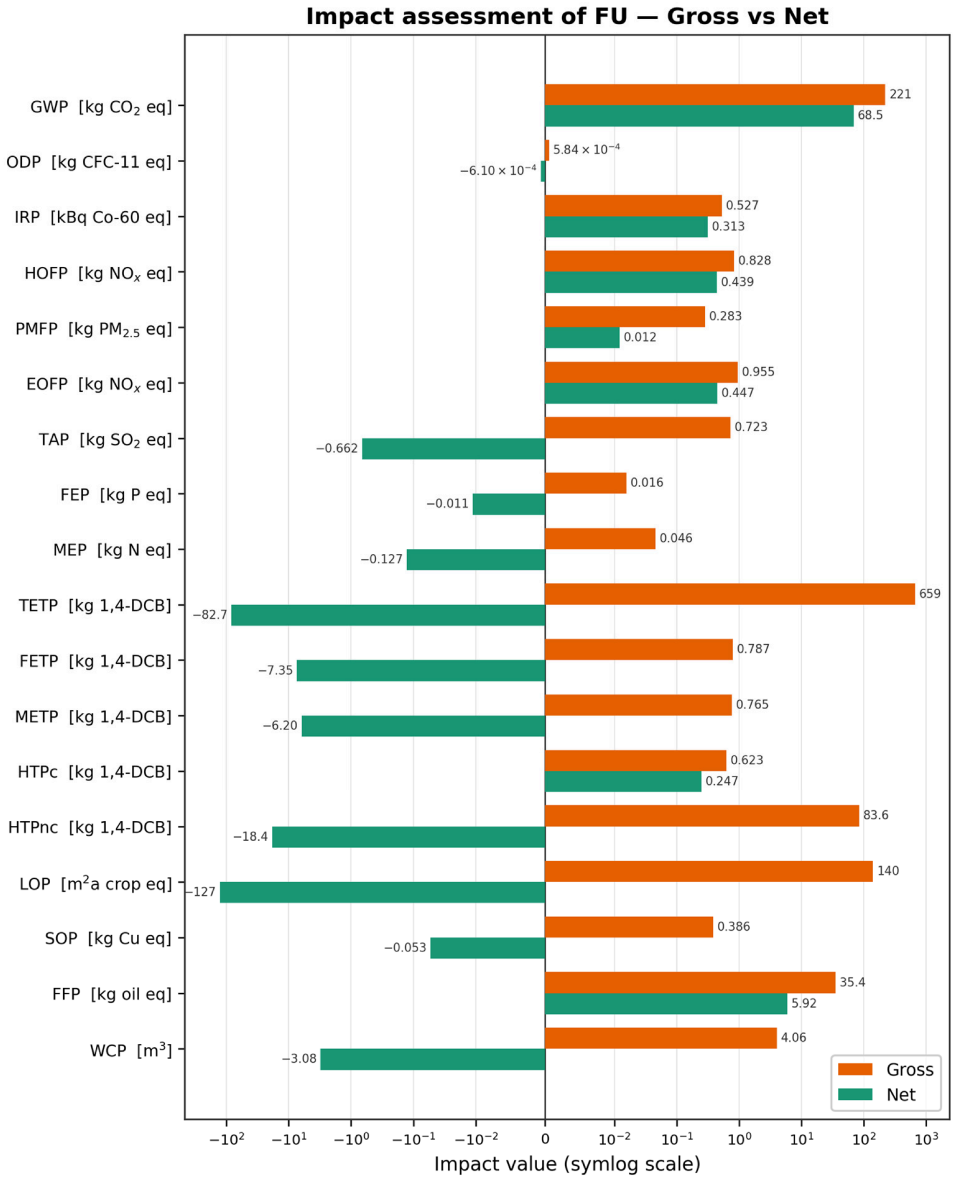

Figure 4. Absolute characterised impact values under gross and net system views at the functional unit level (symmetric logarithmic scale).

Figure 5 presents the normalised impacts under gross and net system views. The five highest-ranking categories under the gross view were HTPc (0.061), EOFP (0.054), TETP (0.043), HOFP (0.040), and FFP (0.036). Under the net view, the ecotoxicity categories (FETP, METP) achieved the largest normalised savings, whilst HTPc (0.024), EOFP (0.025), and HOFP (0.021) remained the most significant positive contributions. GWP ranked lower in the normalised gross view (0.028), indicating that toxicity and ozone formation categories carry greater relative weight when benchmarked against global per capita impacts.

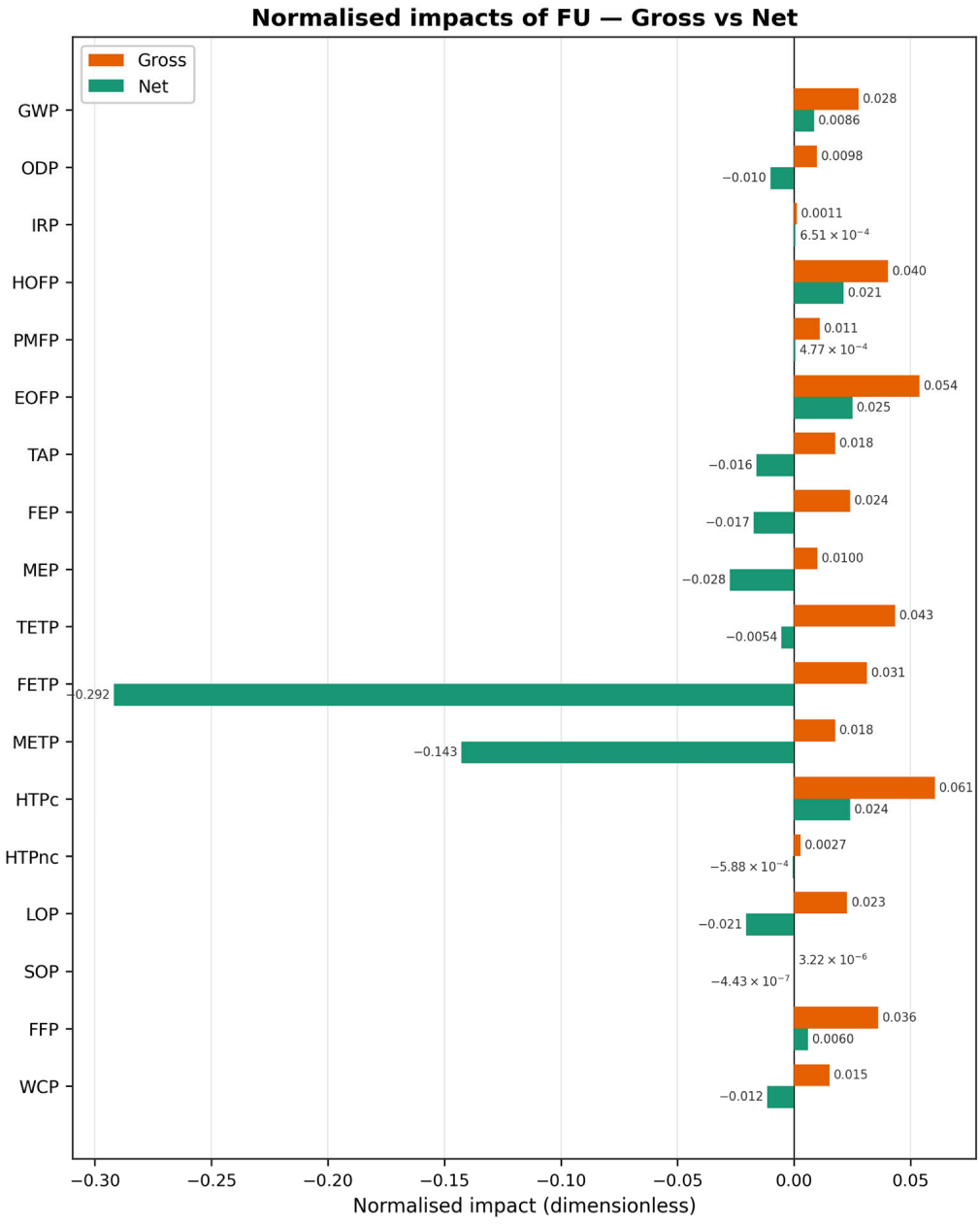

Figure 5. Normalised impacts under gross and net system views using World (2010) reference values.

3.3. Sensitivity Analysis Results

A parametric sensitivity analysis evaluated the influence of SEC (±20%) and enzyme dosage (±20%) on the environmental performance. Figure 6 presents the heatmap of all eight scenarios: four single-parameter variations (S↓, S↑, E↓, E↑) and four combined scenarios (S↓E↓, S↓E↑, S↑E↓, S↑E↑). Product-level tornado charts and heatmaps are provided in the Supplementary Material (Figures S5–S9).

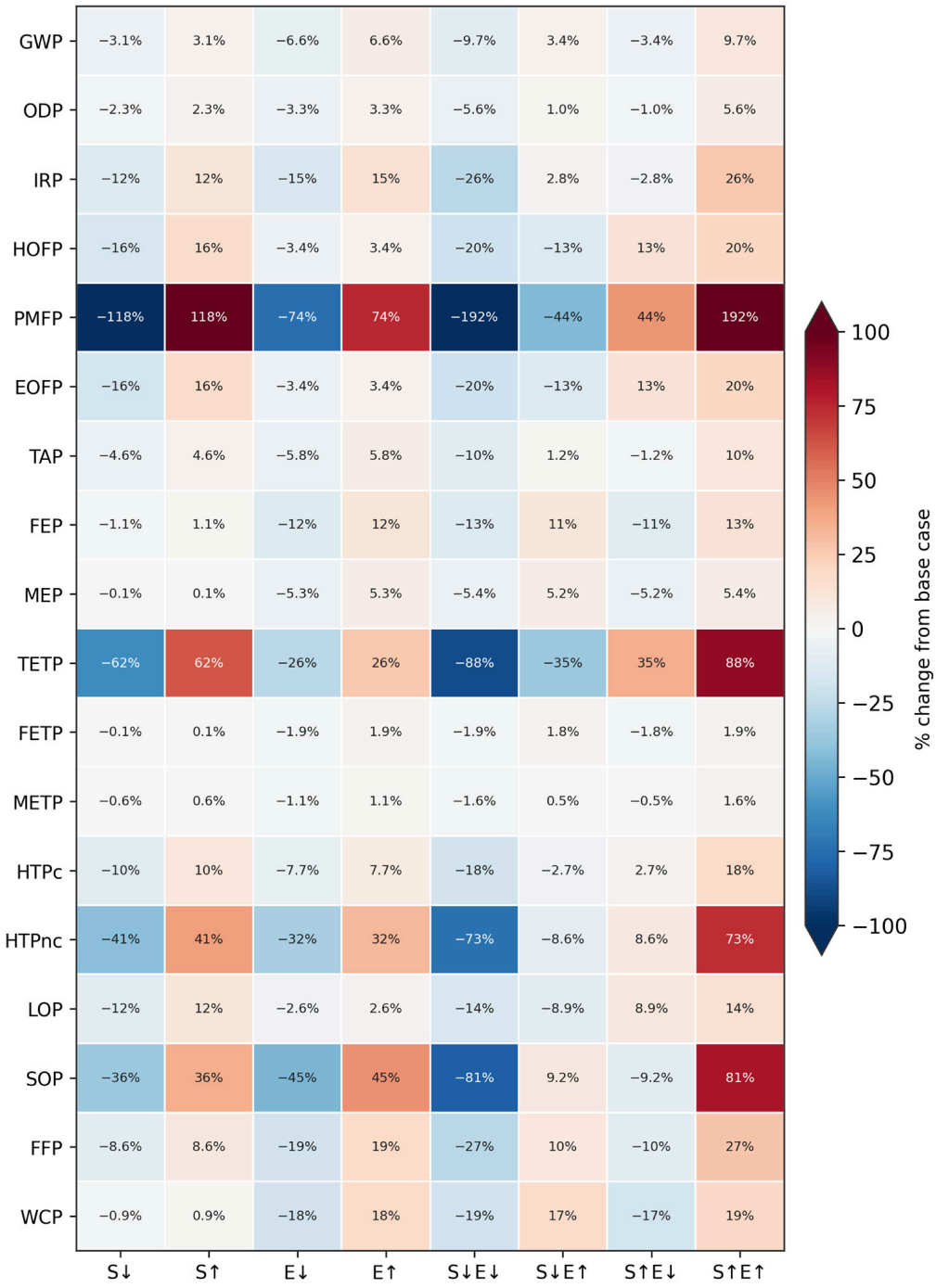

Figure 6. Heatmap of the sensitivity analysis showing percentage change from the base case for single-parameter (S↓, S↑, E↓, E↑) and combined (S↓E↓, S↓E↑, S↑E↓, S↑E↑) scenarios at the functional unit level.

Enzyme dosage was the most influential parameter for the majority of categories. A ±20% variation shifted FETP by ±1.9%, MEP by ±5.2%, WCP by ±17%, and FFP by ±10%. SEC governed the ozone-formation categories (HOFP ±20%, EOFP ±19%) and dominated IRP (±27%) and TETP (±62%) owing to the biomass combustion supply chain.

Categories with net values near zero exhibited amplified percentage sensitivity: PMFP shifted by ±192% and SOP by ±81% under combined parameter variation, reflecting the proximity of these categories to the burden/saving threshold rather than genuine model instability.

The combined effect of simultaneously reducing both SEC and enzyme dosage by 20% yielded improvements of 10–81% across categories, with the largest synergistic effects for SOP (−81%), HTPnc (−73%), and TETP (−68%). This co-optimisation potential has direct practical implications: waste-heat integration for drying combined with reduced enzyme loading would deliver the largest marginal environmental return. An additional one-at-a-time sensitivity on the CHP methane slip, varied across the 0.5–4% range about the 1% base case, is reported in the Supplementary Material (Table S5); it moves the net GWP between approximately 66 and 86 kg CO_2 eq per tonne while leaving the sign of every impact category and the biorefinery's position relative to landfilling and incineration benchmarks unchanged.

3.4. Monte Carlo Uncertainty Analysis Results

The Monte Carlo summary statistics (mean, median, SD, CV, and 95% CI) are reported in Table S2. The per-category coefficient of variation and 95% confidence interval for all 18 categories are additionally tabulated in the main text in Table 4.

Table 4. Monte Carlo uncertainty statistics for the 18 ReCiPe 2016 Midpoint (H) impact categories per functional unit.

Impact Category	Abbr.	Unit	Mean	95% CI	CV (%)
Global warming	GWP	kg CO_2 eq	68.4	[63.2, 73.7]	4.2
Stratospheric ozone depletion	ODP	kg CFC-11 eq	−0.00061	[−0.000637, −0.000583]	2.3
Ionising radiation	IRP	kBq Co-60 eq	4.17	[3.33, 5.03]	10.8
Ozone formation, human health	HOFP	kg NO_x eq	0.438	[0.366, 0.511]	9.7
Fine particulate matter	PMFP	kg $PM_{2.5}$ eq	0.0119	[−0.00642, 0.0304]	82.5
Ozone formation, terrestrial ecosystem	EOFP	kg NO_x eq	0.446	[0.372, 0.52]	9.8
Terrestrial acidification	TAP	kg SO_2 eq	−0.662	[−0.716, −0.608]	4.3
Freshwater eutrophication	FEP	kg P eq	0.0319	[0.0289, 0.035]	5.7
Marine eutrophication	MEP	kg N eq	−0.124	[−0.13, −0.117]	3.2
Terrestrial ecotoxicity	TETP	kg 1,4-DCB	−82.4	[−141, −24.1]	39.2
Freshwater ecotoxicity	FETP	kg 1,4-DCB	−5.03	[−5.57, −4.49]	5.7
Marine ecotoxicity	METP	kg 1,4-DCB	−3.12	[−3.75, −2.49]	10.6
Human carcinogenic toxicity	HTPc	kg 1,4-DCB	13.9	[12.1, 15.7]	7.0
Human non-carcinogenic toxicity	HTPnc	kg 1,4-DCB	38	[19.9, 56.3]	25.2
Land use	LOP	m^2a crop eq	−127	[−142, −112]	6.9
Mineral resource scarcity	SOP	kg Cu eq	−0.0537	[−0.0869, −0.0199]	33.0
Fossil resource scarcity	FFP	kg oil eq	5.9	[4.64, 7.19]	11.9
Water consumption	WCP	m^3	−3.09	[−3.62, −2.55]	10.5

The probability distributions for four representative impact categories (GWP, FFP, WCP, HTPc) are presented in Figure 7. The GWP distribution exhibited a narrow spread (95% CI: 63.2–73.7 kg CO_2 eq, CV = 4.2%), confirming that the climate change result is stable under background data variation. The FFP distribution was centred in positive territory (mean 5.9 kg oil eq, 95% CI: 4.6–7.2), confirming that fossil resource scarcity remains a net burden under the revised system expansion. The WCP distribution lay entirely in negative territory (mean −3.09 m^3, 95% CI: −3.62 to −2.55), confirming a statistically robust water consumption saving. The HTPc distribution (mean 13.9 kg 1,4-DCB, CV = 7.0%) showed moderate uncertainty around the most significant normalised impact category.

The complete set of confidence intervals and CVs is presented in Figure 8. The CVs ranged from 2.3% (ODP) to 82.5% (PMFP), a wider spread than expected from a single uncertainty source. Categories with net values near zero (PMFP, CV = 82.5%) or dominated by highly variable ecotoxicity characterisation factors (TETP, CV = 39%; HTPnc, CV = 25%) exhibited the largest relative uncertainty. By contrast, categories with large posi-

tive or negative net values—GWP (4.2%), ODP (2.3%), MEP (3.2%), TAP (4.3%)—remained tightly constrained.

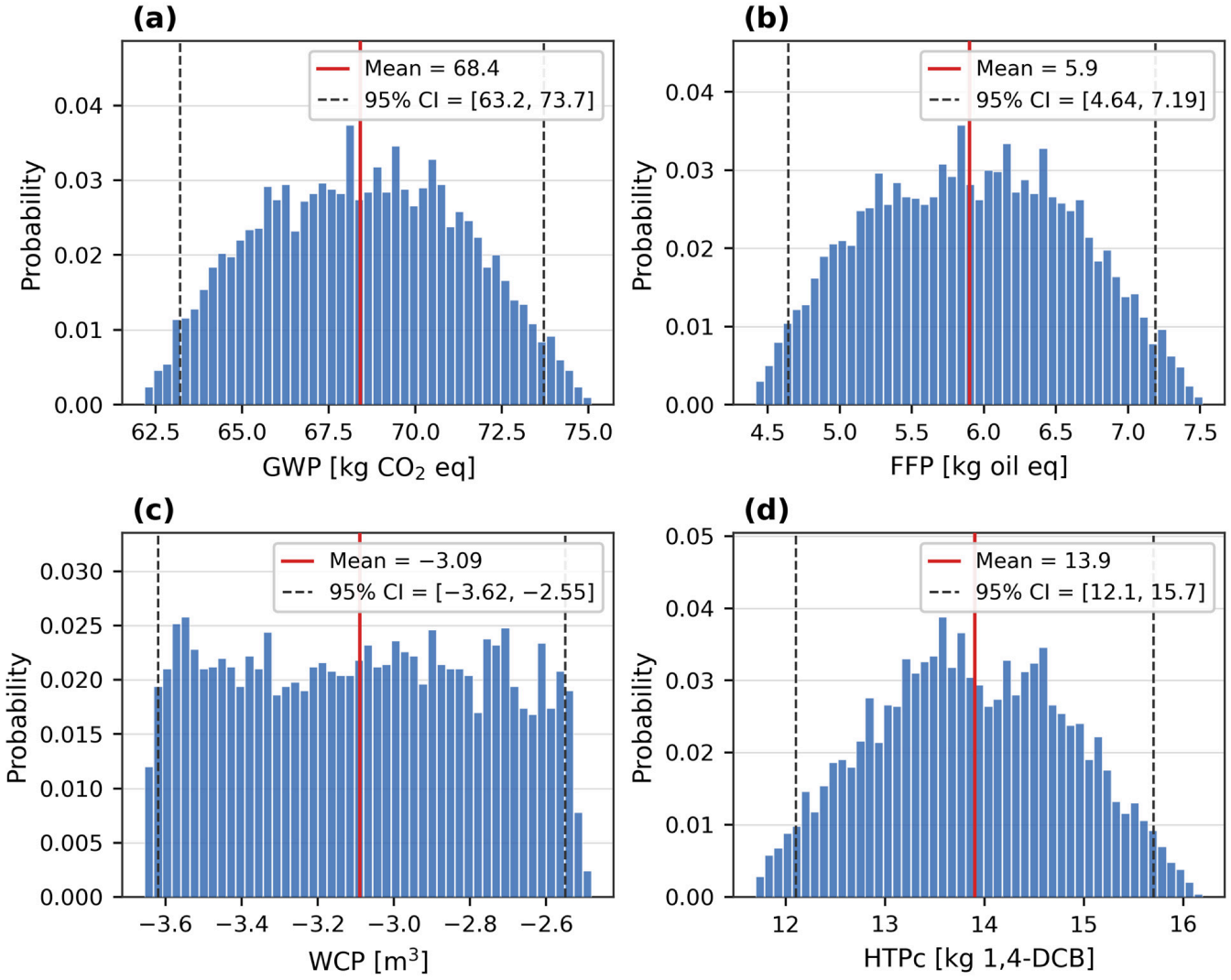

Figure 7. Probability distributions from Monte Carlo simulation (5000 iterations) for four representative impact categories: (**a**) GWP, (**b**) FFP, (**c**) WCP, and (**d**) HTPc.

For 17 of 18 categories the 95% confidence interval did not span zero, so the sign of the net impact (burden or saving) is unchanged across the full interval. The sole exception was PMFP, whose confidence interval [−0.006, 0.030] spanned zero, indicating that the direction of this impact (burden vs. saving) is uncertain under background data variation. For all other categories, including those that reversed sign under system expansion, the Monte Carlo analysis confirms that the sign reversal is statistically robust.

The reported CVs reflect background data uncertainty only. The deterministic sensitivity analysis (Section 3.3) demonstrated that foreground parameters can shift individual categories by up to ±192% for PMFP, far exceeding the Monte Carlo-derived CVs. The full uncertainty envelope spans both the deterministic foreground ranges (Section 3.3) and the stochastic background distributions reported here.

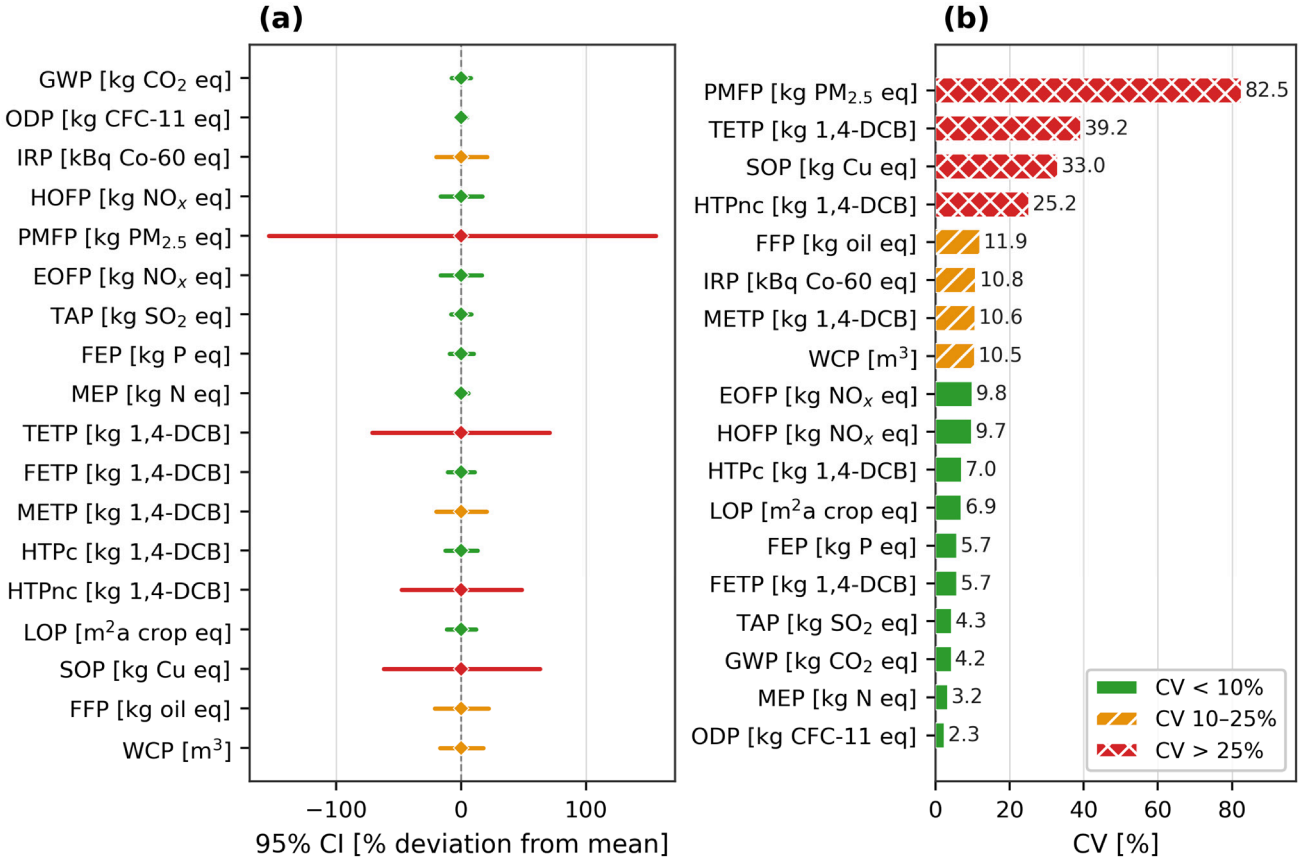

Figure 8. Monte Carlo uncertainty results: (**a**) forest plot showing 95% confidence intervals for all 18 midpoint categories and (**b**) coefficients of variation. In both panels, the colours denote the coefficient of variation (CV): green (CV < 10%), orange (CV 10–25%), and red (CV > 25%).

4. Discussion

4.1. Product-Level Performance

Bio-oil and bioethanol achieved net-negative GWP, driven by the large cradle-to-gate footprints of their conventional substitutes (rapeseed oil and maize-derived ethanol). Biogas retained a net GWP burden because the marginal biogas-derived CHP electricity carries a low fossil carbon intensity [48], yielding a conservative credit. For bio-oil, the environmental and economic hotspots aligned: drying dominated both cost [23] and environmental burden. For bioethanol, they diverged: drying governed cost while enzyme production governed environmental impact.

This enzyme–drying inversion extends, to the OFMSW biorefinery context, the environmental dominance of enzyme production previously observed in lignocellulosic ethanol systems [51,52]. Drying consumes thermal energy from biomass combustion (low fossil carbon intensity), whereas enzyme manufacturing embeds upstream industrial impacts from chemical synthesis, fermentation energy, and agricultural land use [50,55].

4.2. System-Level Environmental Performance

At the functional-unit level, the net GWP represents a 69% reduction from the gross value (Table 3). Eleven of eighteen impact categories achieved net savings under system expansion, with ecotoxicity and eutrophication categories showing the largest relative reductions. The magnitude of these credits reflects the high cradle-to-gate environmental footprints of the displaced agricultural products (rapeseed oil, maize ethanol) compared with fossil fuel alternatives (diesel, petrol) that are conventionally assumed in biorefinery LCA studies.

The normalised impact assessment (Figure 5) identified HTPc, EOFP, and TETP as the most significant categories under the gross view, indicating that toxicity and ozone formation carry greater relative weight than climate change when benchmarked against global per capita impacts. Under the net view, FETP (−0.292) and METP (−0.143) achieved the largest normalised savings, driven by the ecotoxicity-intensive rapeseed oil and maize ethanol supply chains avoided.

Sensitivity analysis confirmed that enzyme dosage is the most influential foreground parameter for the majority of categories, while SEC dominates the ozone-formation and ionising radiation categories (Figure 6). The combined reduction in both parameters by 20% yields improvements of 10–81% across categories. Categories with net values near the burden/saving threshold (PMFP, SOP) showed amplified sensitivity (±81–192%), reflecting their proximity to zero rather than model instability—their 95% confidence intervals from Monte Carlo analysis confirm the sign of the net result.

Monte Carlo simulation (5000 iterations) confirmed statistical robustness for 17 of 18 categories: the 95% confidence interval lay entirely on one side of zero. The sole exception was PMFP, whose interval [−0.006, 0.030] spanned zero (Figure 7). For the key policy-relevant categories—GWP (CV = 4.2%, 95% CI: 63–74 kg CO_2 eq) and FFP (CV = 11.9%, 95% CI: 4.6–7.2 kg oil eq)—the results are tightly constrained.

4.3. Benchmarking Against Waste Management Alternatives

Table 5 positions the system-level GWP of the present biorefinery (gross: 221; net: 69 kg CO_2 eq/t) against conventional OFMSW treatment routes published since 2010. The net GWP of 69 kg CO_2 eq/t is 83% lower than landfilling in California (~400 kg CO_2 eq/t [56]) and 93% lower than in South Korea (1010 kg CO_2 eq/t [57]). Against incineration with CHP, the result is 63% lower than Evangelisti et al. [58] (186 kg CO_2 eq/t for food waste in London), though that study applied cradle-to-grave boundaries. Reviews of waste management LCA confirm that thermal WTE remains the dominant alternative to landfill in benchmarking studies [49,59,60]. Because the studies compiled in Table 4 differ in system boundary (gate-to-gate, gate-to-grave, and cradle-to-grave), functional unit, and credit basis, these benchmark values are not strictly like-for-like; the comparison should be read as indicative of broad magnitude rather than a controlled ranking, and the Scope column is provided so that each value can be interpreted against its own boundary.

Table 5. Global warming potential of OFMSW/food waste treatment routes benchmarked against the present biorefinery. All values in kg CO_2 eq per tonne of waste processed.

Treatment	Country	Scope	kg CO_2 eq/t	Key Credits	Study
Multi-product biorefinery	Greece	G2G, SE	Gross: 221 Net: 69	Rapeseed oil, maize ethanol, marginal biogas CHP elec.	This study
Integrated dry AD + composting + biomethane	Italy	G2G, SE	−112	Biomethane (transport), compost, $(NH_4)_2SO_4$	[61]
Advanced AD (18 configurations)	Italy	G2G	+28	No credits	[62]
Dry AD (RNG upgrading)	USA (CA)	C2Gr, SE	−36 to −2	NG/diesel displacement	[56]
AD (household food waste)	UK	C2Gr, SE	−39	Grid elec., fertiliser	[63]
Incineration-CHP	UK	C2Gr, SE	+186	Grid elec., heat	[58]
Dry AD + digestate incin.	China	G2Gr, SE	−195	Energy, digestate util.	[64]
AD (CHP/vehicle fuel)	Denmark	C2Gr, SE	−375 to +111	Coal elec., vehicle fuel	[65]
Composting (direct em.)	Review	G2G	+119	—	[66]
Landfill	USA (CA)	C2Gr	~400	—	[56]
Landfill	South Korea	C2Gr	+1010	—	[57]

G2G = gate-to-gate; G2Gr = gate-to-grave; C2Gr = cradle-to-grave; SE = system expansion.

Standalone AD systems with system expansion frequently achieve net-negative GWP: −39 for food waste AD in the UK [63], −36 to −2 for dry AD with RNG upgrading in California [56], and –195 for dry AD with digestate incineration [64]. Comparable food-waste AD configurations in centralised and decentralised plants have reported similar magnitudes [67,68]. State-of-the-art reviews of AD biorefinery LCAs confirm that energy and substitution credits are the principal performance levers [69]. The present biorefinery does not achieve net-negative GWP at the system level because it includes energy-intensive unit operations (thermal drying and enzyme-intensive SSF) that AD-only configurations do not require. The trade-off is the production of three distinct energy carriers rather than a single biogas stream.

The most structurally comparable benchmark is Le Pera et al. [61], who reported 112 kg CO_2 eq/t for an integrated dry AD composting plant in southern Italy under gate-to-gate boundaries with system expansion. The 181 kg CO_2 eq/t gap traces to three factors: (i) the biomethane-for-transport credit displaces high-carbon diesel, yielding a larger credit than the marginal biogas CHP electricity assumed here [40]; (ii) enzyme-intensive SSF and hexane extraction, contributing 37–94% of characterised impacts in the present study, have no analogue in a conventional AD–composting configuration [66]; and (iii) thermal drying adds process burdens absent from their system [31].

Møller et al. [65] showed that the choice of displaced energy source, fugitive CH_4 rate, and digestate N_2O assumptions alone can shift AD results by ~500 kg CO_2 eq/t (–375 to +111). Bernstad and la Cour Jansen [70] concluded from a review of 25 comparative LCAs that GWP differences across food waste management studies are driven predominantly by methodological choices—system boundaries, substituted energy sources, and composting emission factors—rather than inherent technology performance. Subsequent OFMSW biorefinery LCAs report similarly wide ranges depending on substitution and boundary assumptions [62,71–76]. The present study confirms this pattern: changing the avoided product from fossil fuels to agricultural substitutes shifted the net GWP from 167 to 69 kg CO_2 eq/t, a 59% reduction from a single methodological decision.

Benchmarking beyond global warming is constrained by reporting practice: most of the studies in Table 5 report GWP (and sometimes energy or resource indicators) as the headline result, and the few that cover further categories use different impact-assessment methods, so the comparison is necessarily partial and largely directional. The most commensurable benchmark is Slorach et al. [60], who assessed household-food-waste anaerobic digestion with the same ReCiPe midpoint (H) family per tonne of food waste; Table 6 compares the two systems across the six non-climate categories they report. Slorach et al. achieved net savings in 13 of 19 categories but net burdens in terrestrial acidification and marine eutrophication, which they attribute to ammonia volatilisation and nitrate leaching from applying the digestate to land; whereas, the present biorefinery returns net savings in all six categories. The sign reversal in acidification and marine eutrophication has two causes: the ecoinvent dataset stabilises the digestate by post-composting rather than applying it raw to land (Section 2.1), so the digestate-borne ammonia and nitrate emissions that dominate Slorach's burdens are absent from the present inventory; and the displaced agricultural products (rapeseed oil, maize ethanol) carry high acidification and eutrophication footprints that enlarge the system-expansion credit. Were the digestate instead field-applied, these two categories would move towards the burdens reported by Slorach et al., consistent with the digestate-fate caveat in Section 2.1. The three ecotoxicity categories agree in direction across both studies—net savings driven by avoided production (grid electricity and mineral fertiliser for Slorach et al.; agricultural substitutes here)—and are larger in the present system. Evangelisti et al. [55] report acidification and nutrient enrichment on the CML basis but no ecotoxicity category, so only the direction is

comparable, and the remaining benchmarks report GWP as the headline indicator. This multi-category view reinforces the normalised assessment (Section 3.2): the toxicity and ozone-formation categories, rather than climate change, carry the greatest relative weight, underlining the value of reporting the full 18-category profile rather than GWP alone.

Table 6. Multi-category comparison of the present biorefinery with the most commensurable benchmark, Slorach et al. [60] (household food-waste anaerobic digestion; ReCiPe 2016 Midpoint (H); per tonne of food waste), for the six non-climate categories they report. Positive values are net burdens, negative values net savings; present-study values are the deterministic net results from Table 3. The biorefinery returns net savings in every category, reversing the terrestrial-acidification and marine-eutrophication burdens that Slorach et al. attribute to land application of the digestate.

Impact Category	Unit	Slorach et al. [60]	This Study
Terrestrial acidification	kg SO_2 eq	+7.6	−0.662
Freshwater eutrophication	kg P eq	−0.014	−0.0112
Marine eutrophication	kg N eq	+0.72	−0.127
Freshwater ecotoxicity	kg 1,4-DCB	−3.3	−7.35
Marine ecotoxicity	kg 1,4-DCB	−3.0	−6.20
Terrestrial ecotoxicity	kg 1,4-DCB	−0.005	−82.7

4.4. Process Optimisation Priorities

Three priorities for environmental improvement follow from these results. First, enzyme sourcing offers the greatest potential: on-site enzyme production [52,77], reduced dosage, and sustainable enzyme-manufacturing feedstocks [55] could reduce impacts across FETP, MEP, WCP, and IRP. Recent OFMSW bioethanol LCAs corroborate enzymes as the principal environmental lever [78–81]. Second, waste heat integration from co-located industry would address both the environmental and economic hotspot for bio-oil whilst reducing ozone formation precursors. In a study conducted about the impacts of upstream logistical planning, it is shown that decentralised waste-to-energy configurations reduce transport-related impacts by 33–45%, suggesting that spatial planning should complement process-level improvements [82]. Third, improving hexane recovery rates or transitioning to greener solvents would mitigate NMVOC fugitive emissions (0.966 kg per FU) and their contribution to photochemical ozone formation; analogous lessons from biorefinery configurations producing activated carbon and levulinic acid reinforce the value of solvent-management strategies [83]. Process-water reuse strategies further offer a route to reduce water consumption potential in fermentation-heavy biorefineries [84]. In addition, future research should prioritise obtaining primary life cycle inventory data for the specific commercial enzyme preparations used (CellicCTec3 and Spirizyme Excel HS), which would improve the accuracy of the bioethanol environmental profile that the generic ecoinvent enzyme proxy currently constrains. Finally, the demonstration unit processes 1 t/day; scaling up to a commercial 50–100 t/day facility would be expected to lower the specific energy consumption of drying and the per-unit burdens of ancillary operations through economies of scale and tighter heat integration, so the demonstration-scale impacts reported here are likely to represent an upper bound relative to a mature commercial plant. Because digestate post-treatment dominates the biogas environmental profile (77–99% across categories), the fate of the digestate is a decisive lever for the anaerobic-digestion line. Higher-value utilisation—direct use as a fertiliser substitute or soil amendment, nutrient recovery (for example as ammonium sulphate or struvite), or valorisation into materials—would displace mineral fertilisers and convert part of this burden into an environmental credit, representing an important pathway to further reduce the biorefinery's impact. Such routes fall outside the present gate-to-gate boundary and merit dedicated assessment.

The system expansion results depend on the choice of avoided product. The large credits observed here reflect the high environmental footprint of agricultural production routes (rapeseed oil, maize ethanol) and would differ under alternative substitution assumptions (e.g., fossil fuel displacement) [85,86]. Net environmental benefit is statistically supported only for categories whose 95% confidence interval lies entirely in negative territory in 10 of the 11 sign-reversed categories.

5. Conclusions

This study quantified the environmental performance of a demonstration-scale OFMSW biorefinery co-producing bio-oil, bioethanol, and biogas, through a gate-to-gate life cycle assessment built on primary operational data. The system achieved a net global warming potential of 68.5 kg CO_2 eq per tonne of wet OFMSW (gross 220.7 before substitution credits), placing it 83–93% below landfilling (400–1010 kg CO_2 eq/t) and 63% below incineration with Combined Heat and Power (186 kg CO_2 eq/t). It did not reach the net-negative GWP of standalone anaerobic digestion (−39 to −195 kg CO_2 eq/t): the thermal drying and enzymatic hydrolysis needed to make liquid fuels impose burdens that biogas-only systems avoid. This is the central trade-off of producing three marketable energy carriers rather than one.

Under system expansion against agricultural substitutes (rapeseed oil, maize ethanol) and marginal biogas-CHP electricity, the biorefinery returned net savings in 11 of 18 ReCiPe midpoint categories (the largest in the ecotoxicity and eutrophication categories that the displaced agricultural supply chains dominate) while seven retained net burdens. Monte Carlo analysis (5000 iterations) confirmed the result is sign-stable; the 95% confidence interval lay wholly on one side of zero for 17 of 18 categories (GWP tightly constrained at CV = 4.2%), the sole exception being particulate-matter formation (PMFP). The sign of the benefit is thus secure; its magnitude is set by how the plant is operated.

Three in-process levers govern that magnitude. Enzyme supply dominates the bioethanol line, driving 37–94% of its characterised impact and ranking as the most influential foreground parameter, even though drying—not enzymes—accounts for 49% of its production cost; this cost–environment inversion means cost-led optimisation alone would miss the principal environmental driver. Reducing enzyme dosage and specific energy consumption by 20% each improved impacts by 10–81% across categories, with on-site enzyme production a further route. For bio-oil, integrating waste heat from co-located industry would cut the drying energy that is simultaneously its cost and environmental hotspot. For biogas, CHP methane slip is the decisive climate lever; across the realistic 0.5–4% range the net GWP varies between 65.7 and 85.5 kg CO_2 eq/t—at least 54% below incineration even at the upper bound—so disciplined methane management preserves the climate advantage. Improving hexane recovery, or moving to greener solvents, would further curb the 0.966 kg of fugitive NMVOC released per functional unit.

These benefits are plausible, but conditional. The system-expansion credits hinge on the avoided product: substituting agricultural rather than fossil routes shifted the net GWP from 167 to 68.5 kg CO_2 eq/t, a 59% change from a single methodological choice—so both gross and net results should be reported. The inventory uses rest-of-world average data, and digestate fate is decisive for the anaerobic-digestion line, where post-treatment dominates 77–99% of its profile and higher-value utilisation would convert part of that burden into a fertiliser-substitution credit. Because the demonstration unit processes only 1 t/day, scale-up to a commercial 50–100 t/day facility would lower drying and ancillary burdens through tighter heat integration, making the impacts reported here an upper bound on a mature plant. Taken together, the findings support the inclusion of OFMSW-derived

advanced biofuels under Annex IX Part A of the recast Renewable Energy Directive (RED III) and its combined 5.5% advanced-biofuel target for transport by 2030.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/cleantechnol8040134/s1, Figure S1: Fugitive air emissions across the three biorefinery subsystems per functional unit; Figures S2–S4: Process-contribution breakdowns for all 18 impact categories under the gross and net system views; Figures S5–S9: Product-level tornado charts and sensitivity heatmaps for the specific-energy-consumption and enzyme-dosage variations; Table S1: Complete life cycle inventory (LCI) per functional unit; Table S2: Monte Carlo summary statistics (mean, median, standard deviation, coefficient of variation, and 95% confidence interval) for the 18 impact categories; Table S3: List of ecoinvent 3.10 datasets used in the foreground and background systems; Table S4: Physicochemical characterisation of the food-waste feedstock; Table S5: One-at-a-time sensitivity of the combined heat and power methane slip on the net global warming potential.

Author Contributions: Conceptualization, K.P. and D.M.; methodology, K.P. and G.P.; software, K.P. and G.P.; validation, K.P., G.P. and D.M.; formal analysis, K.P. and G.P.; investigation, K.P.; resources, D.M.; data curation, K.P.; writing—original draft preparation, K.P. and G.P.; writing—review and editing, D.M.; visualisation, G.P.; supervision, D.M.; project administration, D.M.; funding acquisition, D.M. All authors have read and agreed to the published version of the manuscript.

Funding: This work was funded by the European Union's Horizon 2020 research and innovation program under grant agreement No. 101084405 (CRONUS).

Data Availability Statement: The original contributions presented in this study are included in the article/Supplementary Material. Further inquiries can be directed to the corresponding authors.

Conflicts of Interest: The authors declare no conflicts of interest.

Abbreviations

The following abbreviations are used in this manuscript:

AD	Anaerobic digestion
BAT	Best Available Techniques
CHP	Combined Heat and Power
CI	Confidence interval
CV	Coefficient of variation
EOFP	Ozone formation, terrestrial ecosystems
EROI	Energy return on investment
FEP	Freshwater eutrophication
FETP	Freshwater ecotoxicity
FFP	Fossil resource scarcity
FU	Functional unit
GWP	Global warming potential
HOFP	Ozone formation, human health
HTPc	Human carcinogenic toxicity
HTPnc	Human non-carcinogenic toxicity
IRP	Ionising radiation
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
LOP	Land use
MEP	Marine eutrophication
METP	Marine ecotoxicity
NMVOC	Non-methane volatile organic compounds
NTUA	National Technical University of Athens

ODP	Stratospheric ozone depletion
OFMSW	Organic fraction of municipal solid waste
PMFP	Fine particulate matter formation
ReCiPe	Life cycle impact assessment method (ReCiPe 2016 Midpoint, Hierarchist)
RED	Renewable Energy Directive
RFNBO	Renewable fuel of non-biological origin
RNG	Renewable natural gas
RoW	Rest-of-World (ecoinvent geographic scope)
SD	Standard deviation
SE	System expansion
SEC	Specific energy consumption
SOP	Mineral resource scarcity
SSF	Simultaneous saccharification and fermentation
TAP	Terrestrial acidification
TETP	Terrestrial ecotoxicity
VVV	Vari-Voula-Vouliagmeni (municipality, Attica, Greece)
WCP	Water consumption

References

1. Eurostat Municipal Waste Statistics. Available online: https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Municipal_waste_statistics (accessed on 1 March 2026).
2. EEA. *Bio-Waste in Europe-Turning Challenges into Opportunities*; European Environment Agency (EEA): Central Copenhagen, Denmark, 2020.
3. Gustavsson, J.; Cederberg, C.; Sonesson, U.; van Otterdijk, R.; Meybeck, A. *Global Food Losses and Food Waste: Extent, Causes and Prevention*; FAO: Rome, Italy, 2011.
4. Molina-Penate, E.; Artola, A.; Sanchez, A. Organic Municipal Waste as Feedstock for Biorefineries: Bioconversion Technologies Integration and Challenges. *Rev. Environ. Sci. Biotechnol.* **2022**, *21*, 247–267. [CrossRef]
5. Alibardi, L.; Astrup, T.F.; Asunis, F.; Clarke, W.P.; De Gioannis, G.; Dessi, P.; Lens, P.N.L.; Lavagnolo, M.C.; Lombardi, L.; Muntoni, A.; et al. Organic Waste Biorefineries: Looking towards Implementation. *Waste Manag.* **2020**, *114*, 274–286. [CrossRef] [PubMed]
6. Council of the European Union. (Ed.) *Council Directive 1999/31/EC of 26 April 1999 on the Landfill of Waste*; Council of the European Union: Brussels, Belgium, 1999; Volume L 182, p. 19.
7. Pham, T.P.T.; Kaushik, R.; Parshetti, G.K.; Mahmood, R.; Balasubramanian, R. Food Waste-to-Energy Conversion Technologies: Current Status and Future Directions. *Waste Manag.* **2015**, *38*, 399–408. [CrossRef] [PubMed]
8. Cherubini, F. The biorefinery concept: Using Biomass Instead of Oil for Producing Energy and Chemicals. *Energy Convers. Manag.* **2010**, *51*, 1412–1421. [CrossRef]
9. Ubando, A.T.; Felix, C.B.; Chen, W.-H. Biorefineries in Circular Bioeconomy: A Comprehensive Review. *Bioresour. Technol.* **2020**, *299*, 122585. [CrossRef] [PubMed]
10. Chertow, M.R. Industrial Symbiosis: Literature and Taxonomy. *Annu. Rev. Energy Environ.* **2000**, *25*, 313–337. [CrossRef]
11. Nizami, A.S.; Rehan, M.; Waqas, M.; Naqvi, M.; Ouda, O.K.M.; Shahzad, K.; Miandad, R.; Khan, M.Z.; Syamsiro, M.; Ismail, I.M.I.; et al. Waste Biorefineries: Enabling Circular Economies in Developing Countries. *Bioresour. Technol.* **2017**, *241*, 1101–1117. [CrossRef] [PubMed]
12. European Parliament and Council (Ed.) *Directive (EU) 2018/851 of 30 May 2018 Amending Directive 2008/98/EC on Waste*; European Parliament and Council: Brussels, Belgium, 2018; Volume L 150, p. 140.
13. European Parliament and Council (Ed.) *Directive (EU) 2018/2001 of 11 December 2018 on the Promotion of the Use of Energy from Renewable Sources (Recast)*; European Parliament and Council: Brussels, Belgium, 2018; Volume L 328, p. 209.
14. European Parliament and Council (Ed.) *Directive (EU) 2023/2413 of 18 October 2023 Amending Directive (EU) 2018/2001 as Regards the Promotion of Energy from Renewable Sources*; European Parliament and Council: Brussels, Belgium, 2023; Volume L 2023/2413.
15. European Parliament and Council (Ed.) *Regulation (EU) 2020/852 of 18 June 2020 on the Establishment of a Framework to Facilitate Sustainable Investment*; European Parliament and Council: Brussels, Belgium, 2020; Volume L 198, p. 43.
16. European Commission (Ed.) *Communication—The European Green Deal*; European Commission: Brussels, Belgium, 2019.
17. European Commission (Ed.) *Communication—A New Circular Economy Action Plan: For a Cleaner and More Competitive Europe*; European Commission: Brussels, Belgium, 2020.

18. European Commission (Ed.) *Commission Recommendation (EU) 2021/2279 of 15 December 2021 on the Use of the Environmental Footprint Methods*; European Commission: Brussels, Belgium, 2021; Volume L 471, p. 396.
19. Pinasseau, A.; Zerger, B.; Roth, J.; Canova, M.; Roudier, S. *Best Available Techniques (BAT) Reference Document for Waste Treatment*; European Commission Joint Research Centre: Brussels, Belgium, 2018.
20. Lin, H.; Zhao, J.; Abdulwahab, A.; Mouldi, A.; Loukil, H.; Lin, Z. Development of a Comprehensive Environmental Guidance and Sustainability Assessment for Different Organic Fraction of Municipal Solid Waste-to-Biofuel Scenarios based on Life Cycle Analysis. *Energy* **2024**, *304*, 132073. [CrossRef]
21. Khandelwal, N.; Kumari, S.; Poduval, S.; Behera, S.K.; Kumar, A.; Gedam, V.V. Life-Cycle Assessment of Three Biorefinery Pathways across Different Generations. *Sci. Rep.* **2025**, *15*, 13135. [CrossRef] [PubMed]
22. Alibardi, L.; Cossu, R. Composition Variability of the Organic Fraction of Municipal Solid Waste and Effects on Hydrogen and Methane Production Potentials. *Waste Manag.* **2015**, *36*, 147–155. [CrossRef] [PubMed]
23. Passadis, K.; Asimakopoulos, D.; Malamis, D. Evaluation of the Economic and Energy Performance of Advanced Biofuels from OFMSW in a Demonstration-Scale Biorefinery. *Biomass Bioenergy* **2025**, *203*, 108279. [CrossRef]
24. Passadis, K.; Christianides, D.; Malamis, D.; Barampouti, E.M.; Mai, S. Valorisation of Source-Separated Food Waste to Bioethanol: Pilot-Scale Demonstration. *Biomass Convers. Biorefin.* **2022**, *12*, 4599–4609. [CrossRef]
25. *ISO 14040:2006*; Environmental Management—Life Cycle Assessment—Principles and Framework. International Organization for Standardization: Geneva, Switzerland, 2006.
26. *ISO 14044:2006*; Environmental Management—Life Cycle Assessment—Requirements and Guidelines. International Organization for Standardization: Geneva, Switzerland, 2006.
27. Jimenez-Gonzalez, C.; Kim, S.; Overcash, M.R. Methodology for Developing Gate-to-Gate Life Cycle Inventory Information. *Int. J. Life Cycle Assess.* **2000**, *5*, 153–159. [CrossRef]
28. Wernet, G.; Bauer, C.; Steubing, B.; Reinhard, J.; Moreno-Ruiz, E.; Weidema, B. The Ecoinvent Database Version 3 (Part I): Overview and Methodology. *Int. J. Life Cycle Assess.* **2016**, *21*, 1218–1230. [CrossRef]
29. Huijbregts, M.A.J.; Steinmann, Z.J.N.; Elshout, P.M.F.; Stam, G.; Verones, F.; Vieira, M.; Zijp, M.; Hollander, A.; van Zelm, R. ReCiPe2016: A Harmonised Life Cycle Impact Assessment Method at Midpoint and Endpoint Level. *Int. J. Life Cycle Assess.* **2017**, *22*, 138–147. [CrossRef]
30. Tsafara, P.; Passadis, K.; Christianides, D.; Chatziangelakis, E.; Bousoulas, I.; Malamis, D.; Mai, S.; Barampouti, E.M.; Moustakas, K. Advanced Bioethanol Production from Source-Separated Bio-Waste in Pilot Scale. *Sustainability* **2022**, *14*, 12127. [CrossRef]
31. Deymi-Dashtebayaz, S.E. A Comprehensive Review of Food Waste Dryers and Their Energy Supply Methods. *Waste Biomass Valorization* **2024**, *15*, 3883–3912. [CrossRef]
32. Mujumdar, A.S. *Handbook of Industrial Drying*; CRC Press: Boca Raton, FL, USA, 2014.
33. Cheng, M.-H.; Sekhon, J.J.K.; Rosentrater, K.A.; Wang, T.; Jung, S.; Johnson, L.A. Environmental Impact Assessment of Soybean Oil Production: Extruding-Expelling Process, Hexane Extraction and Aqueous Extraction. *Food Bioprod. Process.* **2018**, *108*, 58–68. [CrossRef]
34. U.S. Environmental Protection Agency. Vegetable Oil Processing. In *Compilation of Air Pollutant Emission Factors (AP-42), Volume I: Stationary Point and Area Sources*, 5th ed.; U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards: Research Triangle Park, NC, USA, 1995; Section 9.11.1.
35. Olofsson, K.; Bertilsson, M.; Liden, G. A Short Review on SSF—An Interesting Process Option for Ethanol Production from Lignocellulosic Feedstocks. *Biotechnol. Biofuels* **2008**, *1*, 7. [CrossRef] [PubMed]
36. Galbe, M.; Zacchi, G. Pretreatment of Lignocellulosic Materials for Efficient Bioethanol Production. In *Advances in Biochemical Engineering/Biotechnology*; Springer: Singapore, 2007; Volume 108, pp. 41–65.
37. Torli, M.; Geer, L.; Kontogeorgis, G.M.; Fosbol, P.L. Benchmarking of Separation Methods for Bioethanol (<5 Wt%) Recovery. *Ind. Eng. Chem. Res.* **2021**, *60*, 5924–5944. [CrossRef]
38. Weiland, P. Biogas production: Current State and Perspectives. *Appl. Microbiol. Biotechnol.* **2010**, *85*, 849–860. [CrossRef] [PubMed]
39. Rajendran, K.; Aslanzadeh, S.; Taherzadeh, M.J. Household Biogas Digesters—A Review. *Energies* **2012**, *5*, 2911–2942. [CrossRef]
40. Poeschl, M.; Ward, S.; Owende, P. Environmental Impacts of Biogas Deployment—Part II: Life Cycle Assessment of Multiple Production and Utilization Pathways. *J. Clean. Prod.* **2012**, *24*, 184–201. [CrossRef]
41. Cherubini, F.; Peters, G.P.; Berntsen, T.; Stromman, A.H.; Hertwich, E. CO_2 Emissions from Biomass Combustion for Bioenergy: Atmospheric Decay and Contribution to Global Warming. *GCB Bioenergy* **2011**, *3*, 413–426. [CrossRef]
42. Astrup, T.F.; Pivnenko, K.; Eriksen, M.K.; Boldrin, A. Life Cycle Assessment of Waste Management: Are We Addressing the Key Challenges Ahead of Us? *J. Ind. Ecol.* **2018**, *22*, 1000–1004. [CrossRef]
43. Bisinella, V.; Schmidt, S.; Varling, A.S.; Laner, D.; Christensen, T.H. Waste LCA and the Future. *Waste Manag.* **2024**, *174*, 53–75. [CrossRef] [PubMed]
44. Crenna, E.; Secchi, M.; Benini, L.; Sala, S. Global Environmental Impacts: Data Sources and Methodological Choices for Calculating Normalization Factors for LCA. *Int. J. Life Cycle Assess.* **2019**, *24*, 1851–1877. [CrossRef]

45. Pizzol, M.; Laurent, A.; Sala, S.; Weidema, B.P.; Verones, F.; Koffler, C. Normalisation and Weighting in Life Cycle Assessment: Quo Vadis? *Int. J. Life Cycle Assess.* **2017**, *22*, 853–866. [CrossRef]
46. Weidema, B.P. Avoiding Co-Product Allocation in Life-Cycle Assessment. *J. Ind. Ecol.* **2000**, *4*, 11–33. [CrossRef]
47. Schrijvers, D.L.; Loubet, P.; Sonnemann, G. Developing a Systematic Framework for Consistent Allocation in LCA. *Int. J. Life Cycle Assess.* **2016**, *21*, 976–993. [CrossRef]
48. Orfanos, N.; Mitzelos, D.; Sagani, A.; Dedoussis, V. Life-Cycle Environmental Performance Assessment of Electricity Generation and Transmission Systems in Greece. *Renew. Energy* **2019**, *139*, 1447–1462. [CrossRef]
49. Astrup, T.F.; Tonini, D.; Turconi, R.; Boldrin, A. Life Cycle Assessment of Thermal Waste-to-Energy Technologies: Review and Recommendations. *Waste Manag.* **2015**, *37*, 104–115. [CrossRef] [PubMed]
50. Jegannathan, K.R.; Nielsen, P.H. Environmental Assessment of Enzyme Use in Industrial Production—A Literature Review. *J. Clean. Prod.* **2013**, *42*, 228–240. [CrossRef]
51. Gilpin, G.S.; Andrae, A.S.G. Comparative Attributional Life Cycle Assessment of European Cellulase Enzyme Production for Use in Second-Generation Lignocellulosic Bioethanol Production. *Int. J. Life Cycle Assess.* **2017**, *22*, 1034–1053. [CrossRef]
52. Nogueira, G.P.; Capaz, R.S.; Franco, T.T.; Dias, M.O.S.; Cavaliero, C.K.N. Enzymes as an Environmental Bottleneck in Cellulosic Ethanol Production: Does on-Site Production Solve It? *J. Clean. Prod.* **2022**, *369*, 133314. [CrossRef]
53. Chiu, S.L.H.; Lo, I.M.C. Identifying Key Process Parameters for Uncertainty Propagation in Environmental Life Cycle Assessment for Sewage Sludge and Food Waste Treatment. *J. Clean. Prod.* **2018**, *174*, 966–976. [CrossRef]
54. Lloyd, S.M.; Ries, R. Characterizing, Propagating, and Analyzing Uncertainty in Life-Cycle Assessment: A Survey of Quantitative Approaches. *J. Ind. Ecol.* **2007**, *11*, 161–179. [CrossRef]
55. Hobusch, M.; Kirtel, O.; Meramo, S.; Sukumara, S.; Welner, D.H. A Life Cycle Assessment of Early-Stage Enzyme Manufacturing Simulations from Sustainable Feedstocks. *Bioresour. Technol.* **2024**, *400*, 130653. [CrossRef] [PubMed]
56. Nordahl, S.L.; Devkota, J.P.; Amirebrahimi, J.; Smith, S.J.; Breunig, H.M.; Preble, C.V.; Satchwell, A.J.; Jin, L.; Brown, N.J.; Kirchstetter, T.W.; et al. Life-Cycle Greenhouse Gas Emissions and Human Health Trade-Offs of Organic Waste Management Strategies. *Environ. Sci. Technol.* **2020**, *54*, 9200–9209. [CrossRef] [PubMed]
57. Kim, M.H.; Kim, J.W. Comparison through a LCA Evaluation Analysis of Food Waste Disposal Options from the Perspective of Global Warming and Resource Recovery. *Sci. Total Environ.* **2010**, *408*, 3998–4006. [CrossRef] [PubMed]
58. Evangelisti, S.; Lettieri, P.; Borello, D.; Clift, R. Life Cycle Assessment of Energy from Waste via Anaerobic Digestion: A UK Case Study. *Waste Manag.* **2014**, *34*, 226–237. [CrossRef] [PubMed]
59. Giugliano, M.; Cernuschi, S.; Grosso, M.; Rigamonti, L. Material and Energy Recovery in Integrated Waste Management Systems: An Evaluation Based on Life Cycle Assessment. *Waste Manag.* **2011**, *31*, 2092–2101. [CrossRef] [PubMed]
60. Demichelis, F.; Piovano, D.; Tommasi, T. Modelling and Optimization of the Best Available Technology for a Waste-to-Energy Plant Coupled with a Thermal Municipal Wastewater Treatment. *Chemosphere* **2022**, *283*, 131193. [CrossRef] [PubMed]
61. Le Pera, A.; Sellaro, M.; Bencivenni, E.; D'Amico, F. Environmental Sustainability of an Integrated Anaerobic Digestion-Composting Treatment of Food Waste: Analysis of an Italian Plant in the Circular Bioeconomy Strategy. *Waste Manag.* **2022**, *139*, 341–351. [CrossRef] [PubMed]
62. Rossi, E.; Pasciucco, F.; Iannelli, R.; Pecorini, I. Environmental Impacts of Dry Anaerobic Biorefineries in a Life Cycle Assessment (LCA) Approach. *J. Clean. Prod.* **2022**, *371*, 133692. [CrossRef]
63. Slorach, P.C.; Jeswani, H.K.; Cuellar-Franca, R.; Azapagic, A. Environmental Sustainability of Anaerobic Digestion of Household Food Waste. *J. Environ. Manag.* **2019**, *236*, 798–814. [CrossRef] [PubMed]
64. Xiao, H.; Zhang, D.; Tang, Z.; Li, K.; Guo, H.; Niu, X.; Yi, L. Comparative Environmental and Economic Life Cycle Assessment of Dry and Wet Anaerobic Digestion for Treating Food Waste and Biogas Digestate. *J. Clean. Prod.* **2022**, *338*, 130674. [CrossRef]
65. Moller, J.; Boldrin, A.; Christensen, T.H. Anaerobic Digestion and Digestate Use: Accounting of Greenhouse Gases and Global Warming Contribution. *Waste Manag. Res.* **2009**, *27*, 813–824. [CrossRef] [PubMed]
66. Boldrin, A.; Andersen, J.K.; Moller, J.; Christensen, T.H.; Favoino, E. Composting and Compost Utilization: Accounting of Greenhouse Gases and Global Warming Contributions. *Waste Manag. Res.* **2009**, *27*, 800–812. [CrossRef] [PubMed]
67. Opatokun, S.A.; Lopez-Sabiron, A.M.; Ferreira, G.; Strezov, V. Life Cycle Analysis of Energy Production from Food Waste through Anaerobic Digestion, Pyrolysis and Integrated Energy System. *Sustainability* **2017**, *9*, 1804. [CrossRef]
68. Tian, H.; Wang, X.; Lim, E.Y.; Lee, J.T.E.; Ee, A.W.L.; Zhang, J.; Tong, Y.W. Life Cycle Assessment of Food Waste to Energy and Resources: Centralized and Decentralized Anaerobic Digestion with Different Downstream Biogas Utilization. *Renew. Sustain. Energy Rev.* **2021**, *150*, 111489. [CrossRef]
69. Martinez-Arce, A.; O'Flaherty, V.; Styles, D. State-of-the-Art in Assessing the Environmental Performance of Anaerobic Digestion Biorefineries. *Resour. Conserv. Recycl.* **2024**, *207*, 107660. [CrossRef]
70. Bernstad, A.; la Cour Jansen, J. Review of Comparative LCAs of Food Waste Management Systems-Current Status and Potential Improvements. *Waste Manag.* **2012**, *32*, 2439–2455. [CrossRef] [PubMed]

71. Ebrahimian, F.; Khoshnevisan, B.; Mohammadi, A.; Karimi, K.; Birkved, M. A Biorefinery Platform to Valorize Organic Fraction of Municipal Solid Waste to Biofuels: An Early Environmental Sustainability Guidance Based on Life Cycle Assessment. *Energy Convers. Manag.* **2023**, *283*, 116905. [CrossRef]
72. Papadaskalopoulou, C.; Sotiropoulos, A.; Novacovic, J.; Barabouti, E.; Mai, S.; Malamis, D.; Kekos, D.; Loizidou, M. Comparative Life Cycle Assessment of a Waste to Ethanol Biorefinery System versus Conventional Waste Management Methods. *Resour. Conserv. Recycl.* **2019**, *149*, 130–139. [CrossRef]
73. Khoshnevisan, B.; Tabatabaei, M.; Tsapekos, P.; Rafiee, S.; Aghbashlo, M.; Lindeneg, S.; Angelidaki, I. Environ-mental Life Cycle Assessment of Different Biorefinery Platforms Valorizing Municipal Solid Waste to Bioen-ergy, Microbial Protein, Lactic and Succinic Acid. *Renew. Sustain. Energy Rev.* **2020**, *117*, 109493. [CrossRef]
74. Ioannidou, S.M.; Lopez-Gomez, J.P.; Venus, J.; Valera, M.A.; Essmann, V.; Alegria-Dallo, I.; Kookos, I.K.; Kou-tinas, A.; Ladakis, D. Techno-Economic Evaluation and Life Cycle Assessment for Sustainable Alternative Bi-orefinery Concepts Using the Organic Fraction of Municipal Solid Waste. *Green Chem.* **2023**, *25*, 4482–4500. [CrossRef]
75. Hou, L.; Basem, A.; Shami, H.O.; Sabri, L.S. Sustainability and Life Cycle Analyzes of Different Biofuel from Municipal Solid Waste Processes. *Int. J. Low-Carbon Technol.* **2024**, *19*, 1610–1622. [CrossRef]
76. Clasen, A.P.; Agostinho, F.; Almeida, C.M.V.B.; Liu, G.; Giannetti, B.F. Advancing towards Circular Economy: Environmental Benefits of an Innovative Biorefinery for Municipal Solid Waste Management. *Sustain. Prod. Consum.* **2024**, *47*, 329–342. [CrossRef]
77. Molina-Penate, E.; Artola, A.; Sanchez, A. Exploring Biorefinery Alternatives for Biowaste Valorization: A Techno-Economic Assessment of Enzymatic Hydrolysis Coupled with Anaerobic Digestion or Solid-State Fermentation for High-Value Bioproducts. *Bioengineered* **2024**, *15*, 2307668. [CrossRef] [PubMed]
78. Musharavati, F.; Ahmad, A.; Javed, M.H.; Sajid, K.; Naqvi, M. Sustainability Assessment of Biofuel and Value-Added Product from Organic Fraction of Municipal Solid Waste. *Environ. Res.* **2024**, *246*, 118121. [CrossRef] [PubMed]
79. Patel, K.; Singh, S.K. Sustainable Waste Management: A Comprehensive Life Cycle Assessment of Bioethanol Production from Agricultural and Municipal Waste. *Environ. Sci. Pollut. Res.* **2024**, *31*, 51431–51446. [CrossRef] [PubMed]
80. Fagundes, V.D.; Machado, E.L.; Schneider, R.C.S.; Colla, L.M. Life Cycle of Bioethanol Production from Blends of Different Food Waste. *Environ. Technol.* **2025**, *46*, 731–751. [CrossRef] [PubMed]
81. Sajid, K.; Javed, M.H.; Al-Muhtaseb, A.H.; Rehan, M.; Khan, M.I.; Musharavati, F.; Anjum, M.W.; Naqvi, M.; Ali, M.; Tabassum, M.R.; et al. Advancing Sustainable Bioethanol Production from Organic Waste by Integrating Life Cycle Modeling. *Energy Nexus* **2025**, *20*, 100563. [CrossRef]
82. Pachakis, G.; Malamis, D.; Mai, S.; Barampouti, E.M. Spatial Influence on Waste-to-Energy Sustainability: A Life Cycle Assessment of RDF Transport and Plant Siting. *Energies* **2025**, *18*, 6238. [CrossRef]
83. Chaparro-Garnica, J.; Guiton, M.; Salinas-Torres, D.; Morallon, E.; Benetto, E.; Cazorla-Amoros, D. Life Cycle Assessment of Biorefinery Technology Producing Activated Carbon and Levulinic Acid. *J. Clean. Prod.* **2022**, *380*, 135098. [CrossRef]
84. Yuan, H.-W.; Tan, L.; Kida, K.; Morimura, S.; Sun, Z.-Y.; Tang, Y.-Q. Potential for Reduced Water Consumption in Biorefining of Lignocellulosic Biomass to Bioethanol and Biogas. *J. Biosci. Bioeng.* **2021**, *131*, 461–468. [CrossRef] [PubMed]
85. Kwon, O.; Son, M.; Kim, J.; Han, J. Carbon-Negative Food Waste-Derived Bioethanol: A Hybrid Model of Life Cycle Assessment and Optimization. *ACS Sustain. Chem. Eng.* **2022**, *10*, 4512–4521. [CrossRef]
86. Wang, Y.; Baral, N.R.; Yang, M.; Scown, C.D. Co-Processing Agricultural Residues and Wet Organic Waste Can Produce Lower-Cost Carbon-Negative Fuels and Bioplastics. *Environ. Sci. Technol.* **2023**, *57*, 2958–2969. [CrossRef] [PubMed]