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Circular Recovery of Organic Waste from Mining Canteens for the Production of Biofertilizers: Life Cycle Assessment and Circularity Indicators in High-Andean Regions

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Abstract

The management of organic waste in high-altitude mining poses a distinctive circularity challenge: waste is generated at sites decoupled from agricultural systems, while the same operations are legally required to revegetate the land they disturb. This study provides, to the best of our knowledge, the first primary-data environmental characterization of a real system that valorizes dining-facility organic waste from a high-altitude mining unit in northern Peru into a solid biofertilizer and a liquid biol, both applied in situ for land reclamation. Unlike methanogenic digesters, the system operates under a lactic (acidogenic) fermentation regime inoculated with effective microorganisms and does not recover biogas. A cradle-to-gate life cycle assessment (ISO 14040/14044) with Monte Carlo uncertainty propagation was combined with a well-established family of five circular economy indicators, adapted to the non-energy-recovery case by redefining the Energy Self-Sufficiency Ratio (ESSR) and the Decarbonization Circularity Indicator (DCI). The principal contribution is methodological: the framework is extended to a circularity archetype that previous, biogas-centered formulations could not represent, showing that a system can close its material and nutrient loops robustly ($WVI = 0.97$) while the energy loop is absent by design ($ESSR = 0$). The climate result is conditional and is a first-order greenhouse-gas (GHG) screening balance, not a physical carbon-sequestration claim: under the upper-bound assumption of full fertilizer substitution, the avoided fertilizer credit outweighs non-methane process emissions only below a narrow fugitive-methane threshold (≈ 0.32 kg CH_4 per ton), a margin that narrows further once agronomic equivalence is discounted. The measured product acidity suggests that this condition is plausible, but, because methane was not measured directly, the low-emission interpretation is presented as a hypothesis requiring confirmation rather than as a demonstrated result. The environmental burden is driven by material and electricity inputs—chiefly the polypropylene containers and grid electricity—rather than by the biological process, which broadens the set of improvement priorities beyond methane management to include capital-good reuse and electricity decarbonization, without implying that methane can be neglected.



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1. Introduction

The valorization of organic waste constitutes one of the operational pillars of the circular economy, and its implementation becomes particularly challenging when waste streams are generated in locations decoupled from the agricultural or industrial systems capable of utilizing them. High-altitude mining exemplifies this disconnect in a particularly pronounced manner. The integration of circular economy principles into mining activities has received increasing attention [1–3], including a growing body of recent work (2023–2026) addressing waste valorization and industrial symbiosis within the sector; yet the focus has been overwhelmingly directed toward mineral flows and the valorization of tailings and metallurgical wastes, whereas the organic waste streams generated in situ—particularly food waste from the dining facilities serving large workforces at remote mining sites—have scarcely been considered as circular resources. At the same time, mining operations are required to reclaim and revegetate the land they disturb, and field evidence has consistently demonstrated that organic amendments, including compost, digestate, and biosolids, effectively rehabilitate degraded mine soils by restoring soil organic carbon, nitrogen, and phosphorus while promoting vegetation establishment [4–13]. The coexistence, within the same industrial boundary, of an underutilized organic waste stream and a simultaneous demand for soil amendments creates the opportunity for an internally closed nutrient cycle that remains largely unexplored and lacks quantitative environmental characterization in the current literature.

Anaerobic conversion of organic waste is a well-established pathway for closing nutrient cycles, and its environmental performance has been extensively assessed through life cycle assessment (LCA), most commonly in comparison with composting [14–17], with increasing attention devoted to digestate as a biofertilizer and to nutrient circularity [18–20]. However, the vast majority of these assessments are built upon an assumption that is rarely made explicit: the recovery and energetic valorization of biogas. Recent reviews of anaerobic digestion LCA confirm that energy recovery through biogas remains the predominant framing of these studies, so that the environmental case for anaerobic treatment is typically constructed around avoided fossil energy [17,19,21]. Carbon balances are dominated by methane capture and its substitution of fossil-derived energy, while the principal environmental credit arises from avoided emissions within the energy system [21]. This framework accurately represents engineered digesters operating under controlled mesophilic or thermophilic conditions, but it implicitly assumes that anaerobic digestion proceeds to the methanogenic stage and that biogas is recovered as a valuable product [22,23].

The quantification of the degree to which a process approaches a circular configuration has motivated the development of dedicated circular economy indicators [24–26]. Molina-Moreno et al. [27] proposed the first family of such indicators for the anaerobic treatment of swine wastewater, defining them around the three recovered resources—water, biofertilizer, and biogas—and the conversion of waste into a technological nutrient. This family of indicators has subsequently been reformulated and aligned with the ISO 59020:2024 framework for measuring circularity performance [28,29]. Across these formulations, however, a structural feature has remained largely unquestioned: they all assume biogas recovery and incorporate energy self-sufficiency and energy-based decarbonization as central dimensions of circularity. Concretely, the limitation is that two of the five indicators are defined only through recovered biogas: the Energy Self-Sufficiency Ratio equates on-site

generation with the electricity obtainable from recovered methane, and the Decarbonization Circularity Indicator embeds an avoided-energy credit in its numerator. In a system that does not reach methanogenesis, both terms are structurally undefined, which is precisely the limitation that motivates the redefinition proposed here. Consequently, systems that do not recover biogas cannot be meaningfully evaluated under these indicators as currently defined. Either the energy-related indicators collapse to trivial values or, more problematically, the framework is applied selectively by omitting the energy dimension altogether, thereby concealing rather than revealing the system's actual degree of circularity.

There exists, however, a significant class of decentralized systems—common throughout Latin America and the Andean highlands—in which organic waste undergoes anaerobic fermentation to produce liquid and solid biofertilizers (locally known as *biol* and *bioabono*) without completing the methanogenic stage or recovering biogas [30,31]. In these systems, carbon is not directed toward an energy carrier but is instead stabilized, under acidogenic conditions, as volatile fatty acids and biogenic CO₂ [32–35]; consequently, the value proposition is agronomic rather than energetic. Acidogenic and lactic fermentation regimes are, in fact, well-documented beyond the mining context—for example, in the acidogenic fermentation of food waste for volatile fatty acid recovery and in lactic acid production from municipal and agro-industrial residues [32–34]—which situates the present case study within a broader and established scientific line rather than as an isolated configuration. Research on low-cost anaerobic digestion in the Andean region has documented precisely this operating regime—characterized by limited biogas production due to low temperatures, prolonged hydraulic retention times, and the agricultural reuse of digestate—in tubular manure digesters implemented in high-altitude rural areas [30,36], including case studies from Cajamarca, Peru [37]. Nevertheless, this body of literature has remained focused on rural energy production. Neither the industrial mining context, the use of food waste as substrate, the acidogenic pathway without energy recovery, nor the final application of the resulting biofertilizers in mine land reclamation has been environmentally characterized as an integrated system.

This omission is not inconsequential, because anaerobic systems without energy recovery are commonly assumed—rather than empirically demonstrated—to exhibit inferior environmental performance, particularly by being regarded as significant sources of fugitive-methane emissions [15,38]. Whether this assumption is valid depends on the actual progression of methanogenesis, which is governed by operating conditions such as suboptimal temperatures, short hydraulic retention times, and the presence of rapidly fermentable co-substrates. In decentralized Andean systems, these conditions shift the metabolic equilibrium toward acid accumulation and methanogen inhibition. To the best of the authors' knowledge, no published framework has adapted circular economy indicators to the limiting case in which the energy loop is absent, nor has the circularity of a non-biogas fermentative system been assessed using primary operational data and physicochemical characterization of its products.

The specific objectives of this study are therefore threefold: (i) to adapt the established family of circular economy indicators to the limiting case of anaerobic valorization without biogas recovery; (ii) to characterize, on the basis of primary operational data, the environmental performance of a real mining dining-facility valorization system through a cradle-to-gate LCA with Monte Carlo uncertainty propagation; and (iii) to test whether fugitive methane is, as commonly assumed, the dominant environmental liability of such systems. This study adopts the established family of circular economy indicators—as a methodological foundation rather than as an original contribution—to pursue these objectives, taking advantage of the unusual availability of all the information required for their application: a real-world system that valorizes organic waste generated by the

dining facility of a high-altitude mining operation, supported by a primary mass balance, physicochemical characterization of its products (biol and bioabono), and documented in situ application for the reclamation of land disturbed by the mining activity itself. The anticipated contributions, distinct from the objectives above, are as follows. First, it extends an established family of circular economy indicators to the limiting case of anaerobic systems without biogas recovery by redefining the energy self-sufficiency and decarbonization indicators so that the absence of the energy loop is explicitly represented rather than omitted. Second, it provides, to the best of the authors' knowledge and supported by the literature review above, the first environmental characterization, based on primary data, of the valorization of mining dining-facility waste for in situ land reclamation in the high Andes—an internal circular loop intrinsic to mining operations—through a cradle-to-gate life cycle assessment with uncertainty propagation by Monte Carlo simulation. Third, it reports a counterintuitive finding: the measured acidity of the products ($\text{pH} \approx 3.7$) is consistent with incomplete methanogenesis, which suggests that fugitive methane—commonly assumed to be the dominant environmental liability of systems without biogas recovery—is likely low, whereas the environmental burden of the system is instead associated with its material and electricity inputs. The broader generalization of these findings to other system configurations is explicitly left for future research requiring replication with measured data from additional case studies.

2. Materials and Methods

The study is structured to follow the four phases of the LCA methodology defined in ISO 14040 [39] (Figure 1 of that standard): goal and scope definition and life cycle inventory analysis are reported in this Materials and Methods section (Sections 2.2–2.4); life cycle impact assessment and the circularity indicators are reported in the Results (Section 3); and the interpretation phase is developed in the Discussion (Section 4).

2.1. Case Study and System Description

The system under study operates in a high-altitude mining unit in northern Peru and valorizes the organic waste generated in the operation's dining facility. The waste is processed in 1 m^3 hermetically sealed polypropylene containers, arranged in a rotating batch system of approximately thirty units: each container is filled with daily collected waste, inoculated with effective microorganisms (EMs) and molasses, and sealed for a retention time of approximately thirty days, after which the products are harvested. Molasses is added as a readily fermentable carbon and energy source that activates the effective microorganism consortium—chiefly the lactic acid bacteria—promoting a rapid onset of acidification and driving the pH down early in the batch, thereby steering the process toward the intended acidogenic regime. The effective microorganism inoculum was selected, rather than native microbial consortia or conventional methanogenic seed sludge, because it is a standardized, commercially available and reproducible consortium (lactic acid bacteria, photosynthetic bacteria and yeasts) widely used in Andean agricultural practice; it is intended to steer the process toward a lactic-dominant acidogenic regime and to limit batch-to-batch variability relative to uncontrolled native consortia or a methanogenic inoculum. We note, however, that the microbial community, lactate and volatile fatty acid profiles were not measured; the acidogenic outcome is inferred from the final-product acidity rather than demonstrated metabolically. Regarding its origin and composition, the inoculum was an effective microorganism preparation obtained from the Instituto Nacional de Innovación Agraria (INIA), the Peruvian National Agrarian Research Institute of the Ministry of Agrarian Development and Irrigation (MIDAGRI), which develops and supplies microbial inoculants for agricultural use. It corresponds to the EM-type consortium concept originally developed by Higa

at the University of the Ryukyus (Okinawa, Japan) in the early 1980s [40]: a liquid mixed culture maintained under acidic, largely anaerobic conditions and dominated by three functional groups—lactic acid bacteria (*Lactobacillus* spp.), phototrophic purple non-sulfur bacteria (*Rhodopseudomonas* spp., *Rhodobacter* spp.) and yeasts (*Saccharomyces* spp.)—with lactic acid bacteria predominating. The composition is reported as specified by the supplier and as documented in the EM literature; the strain-level composition and cell counts of the specific batch used were not independently determined in this study. The process comprises ten sequential stages, from waste collection and homogenization (shredding), through filling and inoculation, anaerobic fermentation, and product recovery, to final application in the field for the reclamation of land disturbed by mining activities.

Regarding process control, the operation records the physicochemical state of the final products at harvest—pH, electrical conductivity, moisture and macronutrient content (Section 3.2)—across three batches operated with different inoculum doses. Time-resolved in-process monitoring of temperature, pH evolution and oxygen availability during fermentation was not performed; the anaerobic condition is maintained by the hermetic sealing of the containers, and the acidogenic outcome is verified *a posteriori* through the measured product acidity. The absence of continuous in-process monitoring is acknowledged as a limitation and identified as a direct improvement for future campaigns (Section 4).

Unlike a methanogenic digester, the system operates under an acidogenic regime: the measured pH of the final products (≈ 3.7) is incompatible with methanogenic activity, which is inhibited below $\text{pH} \sim 6$ [41,42]. Consequently, methanogenesis does not proceed to completion, biogas is not recovered, and carbon is stabilized as volatile fatty acids and biogenic CO_2 . This behavior is consistent with low-cost anaerobic digestion systems reported in the Andean highlands, where biogas production is limited by low temperatures and the effluent is reused for agricultural purposes [30,36]. The fermentation yields two valorized products—a solid bioabono, which contains most of the mass fraction, and a liquid biol—both applied in situ as organic amendments for revegetation, thereby closing an internal nutrient loop within the mining operation.

2.2. Objective, Scope, and Functional Unit

The study was conducted in accordance with the principles and requirements of the life cycle assessment (LCA) methodology established in ISO 14040 and ISO 14044 [39,43]. The objective is to quantify the environmental and circularity performance of the described valorization system. Since the primary function of the system is waste management, the functional unit (FU) is defined as the treatment of one metric ton (1000 kg) of mining dining organic waste, rather than a quantity of product; this choice avoids assigning to the biofertilizer the full environmental burdens of a process whose stated purpose is waste management. A product-based functional unit (e.g., per kg of biofertilizer) was deliberately not adopted because it would frame a waste-management operation as a fertilizer-production process and would attribute to the product burdens that are incurred to treat the waste, regardless of product yield; the waste-based unit therefore reflects the driving function of the system. Because the fertilizer credit is tracked separately (Sections 3.3 and 3.4) rather than embedded in the functional unit, the qualitative conclusions of the study are not sensitive to this choice.

The system boundary is cradle-to-gate and includes waste collection and homogenization, inoculation, fermentation in containers, and the recovery of biofertilizer products (bioabono and biol), together with the associated material and energy inputs. The field application of the products—including any organic cover layer used at the time of land reclamation (Section 4.2)—lies beyond the cradle-to-gate boundary and is therefore not part of the inventory.

The system is multifunctional: in addition to waste treatment, it provides a fertilization service. This multifunctionality is addressed through system expansion with substitution [43]: the nutrient content of the bioabono and biol displaces mineral fertilizer production, and the avoided impact is credited to the system. The relevance of a mineral fertilizer counterfactual in this specific setting deserves justification, because the products are applied to mine-disturbed land rather than to conventional cropland. Under the Peruvian mine closure regulatory framework—Law 28090 (Ley que regula el Cierre de Minas) and its regulation, Supreme Decree 033-2005-EM (Reglamento para el Cierre de Minas) [44,45]—mine operators are legally required to submit a Mine Closure Plan that provides for the rehabilitation and revegetation of the areas disturbed by mining, and the establishment of vegetative cover on degraded, nutrient-poor high-Andean substrates routinely relies on mineral NPK fertilization to correct the nutrient deficit. In the counterfactual without the valorization system, this fertilization demand would be met with conventional mineral fertilizer; the biofertilizer therefore displaces a mineral input that would otherwise be applied, which is what legitimizes the substitution credit. The substitution is quantified on a nutrient-mass-equivalent basis; that is, it assumes that the nitrogen, phosphorus and potassium supplied by the biofertilizer displace an equal mass of mineral nutrient. This is a first-order assumption: the plant availability of organic nitrogen in the first application cycle is generally lower than that of synthetic nitrogen, so the substitution credit reported here should be interpreted as an upper bound, and its potential influence on the estimated credit is discussed in Section 4.2. To maintain traceability of each term, substitution is explicitly quantified in the carbon balance (Section 3.4) and in the fertilizer substitution rate, while the multi-category characterization profile (Section 3.7) is reported on a gross cradle-to-gate production basis, avoiding double counting.

Foreground data are primary, obtained from operational records of the mining facility. Background data (polypropylene production, grid electricity, molasses, and substituted mineral fertilizers) were taken from the ecoinvent database [46]. The ecoinvent v3 database (cut-off system model) was used. Peru-specific datasets were adopted where available—most importantly for grid electricity—while for the remaining background processes (polypropylene, molasses, mineral fertilizers) global or rest-of-world (RoW) proxy datasets were used, as region-specific Peruvian inventories were not available; this is noted as a source of background-data uncertainty and is reflected in the pedigree-based data quality scoring (Section 2.7). Modeling was carried out in SimaPro using the ReCiPe 2016 midpoint method, hierarchical perspective (H), across its eighteen impact categories [47].

One methodological clarification is required regarding climate change. The SimaPro model of the process was built upon a generic anaerobic digestion template that includes a biogas-handling node. This node's direct emissions are not representative of the acidogenic system studied here, in which no biogas is produced. Consequently, the climate change category is not taken from the multi-category SimaPro run; instead, it is assessed through a system-specific carbon balance constructed from the primary inventory (Section 3.4). The remaining seventeen categories, whose burdens originate in the material and energy inputs rather than in the biogas node (Section 3.7), are taken directly from the SimaPro characterization. This split is made explicit here so that the dual treatment of climate change is transparent to the reader.

2.3. Feedstock Origin and Characterization

The treated waste corresponds to the organic fraction generated in the dining facility of the mining unit, collected in the district of Hualgáyoc (Cajamarca region, Peru) during 2024. It constitutes the sole organic input to the system, with no co-substrates, and exhibits a moisture content of 70% by weight—i.e., an approximate total solids fraction of 30%.

Table 1 summarizes its characterization.

Table 1. Characterization of the organic dining waste used as feedstock (referenced to the functional unit). n.d.: not experimentally determined.

Parameter	Value
Waste type	Dining organic waste (100%)
Origin and year	Hualgáyoc, Cajamarca, Peru (2024)
Mass per functional unit	1000 kg
Moisture content	70%
Total solids (derived)	≈30%
Volatile solids	n.d. (literature value; Section 2.5)
C/N ratio	n.d.

The volatile solids content of the substrate was not analytically determined. Since this parameter governs the estimation of fugitive methane (Section 2.5), a representative value for dining waste reported in the literature was adopted, and its influence was assessed through sensitivity analysis (Section 2.7). The adopted value corresponds to the upper part of the range reported for canteen and food waste, in which volatile solids typically represent about 85–95% of total solids; selecting a high VS fraction is deliberately conservative for the carbon balance, because it maximizes the estimated methane-generation potential and therefore does not understate the fugitive-methane liability. Referred to the functional unit, this corresponds to $m_{SV} \approx 276$ kg VS (300 kg total solids $\times$ 0.92), the value used in the fugitive-methane scenarios of Section 3.4. Its experimental determination is identified as a direct improvement to strengthen the system's carbon balance.

An additional source of feedstock variability, beyond the analytical parameters above, is the day-to-day variation in the dining menu: the composition of the organic waste is not strictly constant, and this introduces variability in total and volatile solids and in the C/N ratio. This effect is partly captured by operating and characterizing three separate batches (Section 3.2); its residual influence is discussed together with the other uncertainty sources in Sections 3.3 and 3.6.

2.4. Life Cycle Inventory

The inventory was constructed from the process mass and energy balance, referenced to the functional unit. The foreground data correspond to three operating batches carried out under the same protocol with different inoculum doses; reported flows are representative per-cycle values referred to the functional unit rather than a single production campaign, which improves the representativeness of the inventory. Table 2 summarizes the main input and output flows. The gaseous stream fraction is explicitly addressed in Section 2.5, rather than adopting the aggregated allocation reported in the original operational record.

As detailed in the note to Table 2, polypropylene containers are accounted for by prorating their mass (approximately 45 kg per unit) over their service life (approximately 30 cycles), yielding 1.5 kg per functional unit. The produced biofertilizer substitutes mineral fertilizer according to its nutrient content, and this credit is incorporated via system expansion. The nutrient composition of the products is determined through physicochemical characterization and reported in the Section 3.

Table 2. Life cycle inventory of the valorization system, referenced to the functional unit (1000 kg of treated organic dining waste). Foreground data are primary. The polypropylene container is a durable capital good; its mass is prorated over its service life (see note below the table).

Category	Flow	Quantity (Unit)
Inputs	Organic dining waste	1000 kg
	Water (container washing)	330 L
	Water (inoculation)	18 L
	Effective microorganism (EM) inoculum	1 L
	Molasses	1 L
	Electricity (homogenization, 3 HP motor)	5–9 kWh
	Fuel (internal transport)	0.3 gal
	Polypropylene container (prorated)	1.5 kg
Outputs	Solid biofertilizer (bioabono)	805 kg
	Liquid biofertilizer (biol)	~165 L
	Washing wastewater	330 L
	Leachate/reject stream (within the 3% loss)	≤30 kg
	Fugitive CH ₄ /biogenic CO ₂	estimated (Section 2.5)

Prorated polypropylene: each 1 m³ container has a mass of approximately 45 kg and a service life of approximately 30 fermentation cycles (≈2.5 years at one 30-day cycle per month). Since one functional unit corresponds to one container batch, the prorated container mass per functional unit is $45/30 = 1.5$ kg. This value feeds both the carbon balance (Section 3.4) and the multi-category characterization profile (Section 3.7). The leachate/reject stream is not an additional loss on top of the 3% gaseous-plus-leachate figure of Section 3.1, but a component of it, reported on a feedstock basis; the container-washing water (330 L) enters and leaves the boundary as a separate stream and is excluded from the product mass balance.

2.5. Estimation of Fugitive-Methane Emissions

Since the system vents the gas stream upon container opening, fugitive methane must be estimated rather than directly measured. The fugitive mass is calculated according to Equation (1):

$$m_{\text{CH}_4, \text{fug}} = m_{\text{SV}} B_0 f_{\text{conv}} (1 - \kappa) \rho_{\text{CH}_4} \quad (1)$$

where m_{SV} is the volatile solids mass of the substrate (kg VS) (Section 2.3), B_0 is the biochemical methane potential of the substrate (m³ CH₄ kg^{−1} VS), i.e., the theoretical maximum yield, and $f_{\text{conv}} \in [0, 1]$ is the fraction of that potential actually realized under the operating conditions (dimensionless). κ is the capture efficiency (dimensionless, with $\kappa = 0$ for a vented system), and ρ_{CH_4} is methane density (≈0.668 kg m^{−3} at ≈20 °C); the resulting fugitive mass $m_{\text{CH}_4, \text{fug}}$ is expressed in kg CH₄ per functional unit. This formulation deliberately separates the biochemical potential B_0 (a literature value for canteen and food waste) from the realized yield $B_0 f_{\text{conv}}$: the two are not interchangeable, and conflating them would overstate emissions in a system that does not reach methanogenesis. The measured product acidity (pH ≈ 3.7) is indicative of strong methanogen inhibition and therefore of a low f_{conv} ; however, because neither the gas stream nor f_{conv} was measured, the realized methane yield is not fixed at a single value but is treated as an unmeasured scenario variable and swept across its admissible range (Section 3.4). The favourable lower-emission boundary ($f_{\text{conv}} \rightarrow 0$) is reported as one end of this range rather than as the operating condition of the system. The climate contribution of this stream is evaluated using the 100-year global warming potential of methane according to IPCC factors [48].

2.6. Circular Economy Indicator Framework Adapted to the Non-Energy-Recovery Case

System circularity is assessed using five indicators from the established family for anaerobic valorization systems [27,28], aligned with ISO 59020:2024 [29]. Three of them—mass recovery, fertilizer substitution, and water loop closure—are applied in their standard

form. The two energy-related indicators—Energy Self-Sufficiency Ratio (ESSR) and Decarbonization Circularity Indicator (DCI)—are redefined to explicitly represent the absence of the energy loop. Figure 1 summarizes schematically how the original family is adapted: the three material-, nutrient- and water-related indicators are applied unchanged, whereas the two energy-linked indicators are redefined to remove their dependence on recovered biogas, making the framework applicable to acidogenic systems without energy recovery.

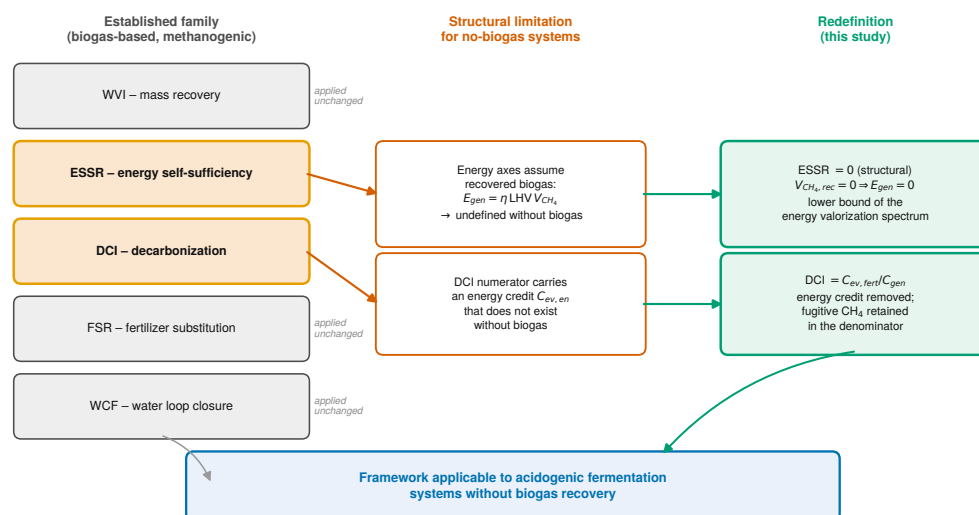


Figure 1. Schematic adaptation of the established five-indicator circular economy family to the non-energy-recovery (acidogenic, no-biogas) case. Three indicators (WVI, FSR, WCF) are applied in their standard form; the two energy-linked indicators (ESSR, DCI, highlighted) are redefined to remove their structural dependence on recovered biogas, extending the framework to systems that do not reach methanogenesis.

The methodological contribution is twofold and is deliberately minimal and targeted. First, the framework is extended so that the zero-biogas limit becomes representable and interpretable rather than structurally undefined: ESSR and the energy term of the DCI are no longer expressed only through recovered methane, so that the absence of the energy loop is reported as an informative structural value ($\text{ESSR} = 0$) instead of collapsing the indicator or silently dropping the energy dimension. Second, and with genuine analytical content, the DCI numerator is reformulated with an explicit first-year agronomic-equivalence coefficient α_k , the fugitive-methane term is introduced into the denominator, and the realized methane yield is separated from the biochemical potential ($B_0 f_{\text{conv}}$ vs. B_0); these three elements, together with the decarbonization decision map (Section 3.6), constitute the substantive extension over the prior scalar formulations. The three indicators that quantify material, nutrient and water circularity retain their original definitions, so that comparability with the prior studies from which the framework derives is preserved; the intervention is confined to the two indicators whose original formulation is inseparable from biogas recovery. This localizes the contribution—the framework is extended, not replaced—and clarifies that the resulting $\text{ESSR} = 0$ and the energy-free DCI are structural properties of the acidogenic configuration rather than the consequence of applying an ill-suited metric. Each of the five indicators is defined below.

2.6.1. Waste Valorization Index (WVI)

This index quantifies the mass fraction of waste converted into valorized products, as shown in Equation (2):

$$\text{WVI} = \frac{m_{\text{bioabono}} + m_{\text{biol}}}{m_{\text{in}}} \quad (2)$$

where m_{in} is the treated waste mass (FU). Consistent with the non-energy-recovery regime, the vented gaseous stream and process losses are excluded from the numerator; $WVI \in [0, 1]$.

2.6.2. Energy Self-Sufficiency Ratio (ESSR)—Redefined

The energy performance of the system is assessed through the Energy Self-Sufficiency Ratio (ESSR), Equation (3), which expresses the balance between locally generated and externally required energy. For the present system, this indicator is reformulated to explicitly account for the absence of biogas recovery. It is defined as

$$ESSR = \frac{E_{gen}}{E_{dem}}, \quad E_{gen} = \eta_{el} LHV_{CH_4} V_{CH_4,rec} \quad (3)$$

In general configurations with biogas recovery, E_{gen} represents on-site electricity generation from recovered biogas. In the acidogenic fermentation system studied here, biogas is not recovered ($V_{CH_4,rec} = 0$), and therefore $E_{gen} = 0$ and $ESSR = 0$. This value is not a missing parameter but the lower bound of the energy valorization spectrum: the energy loop is open, and the process electricity demand E_{dem} is fully supplied by imported grid electricity.

2.6.3. Decarbonization Circularity Indicator (DCI)—Redefined

In its general form, the DCI relates avoided emissions to generated emissions, as shown in Equation (4):

$$DCI = \frac{C_{ev,en} + C_{ev,fert} + C_{ev,alm}}{C_{gen}} \quad (4)$$

Without energy recovery, the avoided energy credit is zero ($C_{ev,en} = 0$), and the indicator reduces to Equation (5):

$$DCI = \frac{C_{ev,fert}}{C_{gen}} = \frac{\sum_k \alpha_k m_k EF_k}{C_{el} + C_{mat} + C_{CH_4} + C_{tr}} \quad (5)$$

where $k \in \{N, P_2O_5, K_2O\}$, m_k is the mass of substituted nutrient—obtained from the physicochemical characterization of the products (Results section)—and EF_k is the emission factor of the corresponding mineral fertilizer. The coefficient $\alpha_k \in (0, 1]$ is the first-year agronomic-equivalence factor of nutrient k , i.e., the fraction of the organic nutrient that is plant-available and therefore genuinely displaces mineral fertilizer in the reclamation year; $\alpha_k = 1$ denotes full mass-for-mass substitution and is the upper bound, whereas organic nitrogen in particular mineralizes gradually and typically has $\alpha_N < 1$ in the first season. Results are reported for the $\alpha_k = 1$ upper bound and for a realistic discounted case (Section 3.4). C_{el} , C_{mat} , and C_{tr} represent electricity, materials and auxiliary inputs (prorated polypropylene and molasses), and transport, respectively, while $C_{CH_4} = GWP_{CH_4} \cdot m_{CH_4,fug}$ accounts for fugitive-methane emissions (Section 2.5). The redefinition therefore consists of removing the non-existent energy credit from the numerator and explicitly incorporating actual methane emissions into the denominator. Because the numerator depends on the agronomic-equivalence coefficients and the denominator on the estimated fugitive methane, the DCI is jointly sensitive to α_k , to the realized methane yield $B_0 f_{conv}$, and to the methane emission factor; these dependencies are quantified explicitly in Section 3.4 rather than left implicit.

2.6.4. Fertilizer Substitution Rate (FSR)

This index expresses the mass of mineral nutrients displaced by the biofertilizer, as shown in Equation (6), for nitrogen:

$$\text{FSR}_N = m_{\text{bioabono}} w_{\text{DM}} c_N^{\text{DM}} + V_{\text{biol}} c_N^{\text{biol}} \quad (6)$$

where w_{DM} is the dry matter fraction of the bioabono, c_N^{DM} its nitrogen content on a dry basis, and c_N^{biol} the nitrogen concentration in the liquid fraction; analogous expressions apply for P_2O_5 and K_2O .

2.6.5. Water Closure Factor (WCF)

The WCF quantifies the extent to which water used in the system is internally recirculated, relating the volume of reused water to the total water consumption within the system boundary. It provides a direct measure of water circularity independent of other material and energy flows. It is formally defined in Equation (7):

$$\text{WCF} = \frac{V_{\text{water, reused}}}{V_{\text{water, consumed}}} \quad (7)$$

where $V_{\text{water, reused}}$ is the water reused within the system boundary and $V_{\text{water, consumed}}$ is the total water consumption. In the documented system, the water consumed (330 L of container-washing water and 18 L of inoculation water per functional unit) is not internally recirculated: the washing water leaves the boundary as wastewater (~330 L; Table 2). The reused volume is therefore zero and $\text{WCF} = 0$. As with the energy loop, the water loop is open by design rather than by inefficiency. A hypothetical wash-water recirculation retrofit would raise this value, but it is not part of the documented system and does not alter the base-case result.

2.7. Uncertainty Analysis

Uncertainty in the results was propagated using Monte Carlo simulation. The inventory data quality was translated into probability distributions through the pedigree matrix approach, assigning quality indicators (reliability, completeness, and temporal, geographical, and technological correlation) to each flow [49]; the scoring criteria and the scores assigned to the primary inventory flows are detailed in Tables S1 and S2 of the Supplementary Material. A total of 10,000 iterations were performed, and the mean, median, and 5th and 95th percentiles were reported as a 90% confidence interval for each impact category and circularity indicator. In the present analysis the inventory parameters were treated as statistically independent; correlations among parameters were not modeled. This is a standard simplifying assumption, but it may narrow the reported confidence intervals relative to a fully correlated treatment, and it is therefore acknowledged as a limitation of the uncertainty analysis. A sensitivity analysis was additionally conducted on the most uncertain or assumption-driven parameters—methane yield B_0 and substrate volatile solids (Sections 2.3 and 2.5), and nutrient content—to assess the robustness of the conclusions. All processing is fully reproducible.

3. Results

The results are organized into three blocks: first, indicators directly derivable from primary data—mass valorization and fertilizer substitution—which constitute the most robust findings of the study; second, the carbon balance and decarbonization index, whose outcome is conditional and governed by a single unmeasured parameter; and third, the multi-category impact profile, which requires full SimaPro characterization within the defined system boundary.

3.1. Mass Balance and Product Yield

The treatment of 1000 kg of waste yields 805 kg of solid biofertilizer and approximately 165 L of biol; referred to the feedstock mass, the combined gaseous-plus-leachate loss not recovered as product is therefore about 30 kg (3%; Figure 2). This figure is expressed on a feedstock basis (1000 kg): the container-washing water (330 L) enters and leaves the boundary as a separate wastewater stream and is excluded from the product balance, whereas the minor liquid co-inputs (inoculation water, EM and molasses, ≈ 20 kg) are incorporated into the wet products and are therefore not counted in the feedstock denominator; the WVI is thus referred to the 1000 kg of treated waste rather than to the total wet mass entering the containers. The leachate/reject stream listed in Table 2 is a component of this 3% loss and not an additional term. The Waste Valorization Index (Equation (2)) is therefore:

$$\text{WVI} = \frac{805 + 165}{1000} = 0.97.$$

The numerator sums the solid product mass (805 kg) and the biol volume (165 L); a biol density of $\approx 1 \text{ kg L}^{-1}$ is assumed, consistent with its high moisture content, so that $165 \text{ L} \approx 165 \text{ kg}$. This value indicates that 97% of the input mass is recovered as valorized products. More important than the numerical value is its thermodynamic interpretation: a gaseous loss of only 3% is incompatible with active methanogenesis, which would divert a substantially larger fraction of carbon to the gas phase as CH_4 and CO_2 . It therefore provides independent evidence that the system operates under a stalled acidogenic regime. This gaseous-plus-leachate loss is small, but because the products are wet and not all added and residual flows were independently quantified, it is not used as a strict upper bound on fugitive methane; the methane term is instead treated as an unmeasured scenario variable (Sections 2.5 and 3.4). It should be noted that a high WVI on a wet basis partly reflects water retention in the products (moisture $> 80\%$); while it remains a valid indicator of mass circularity, it should not be interpreted as complete solid conversion.

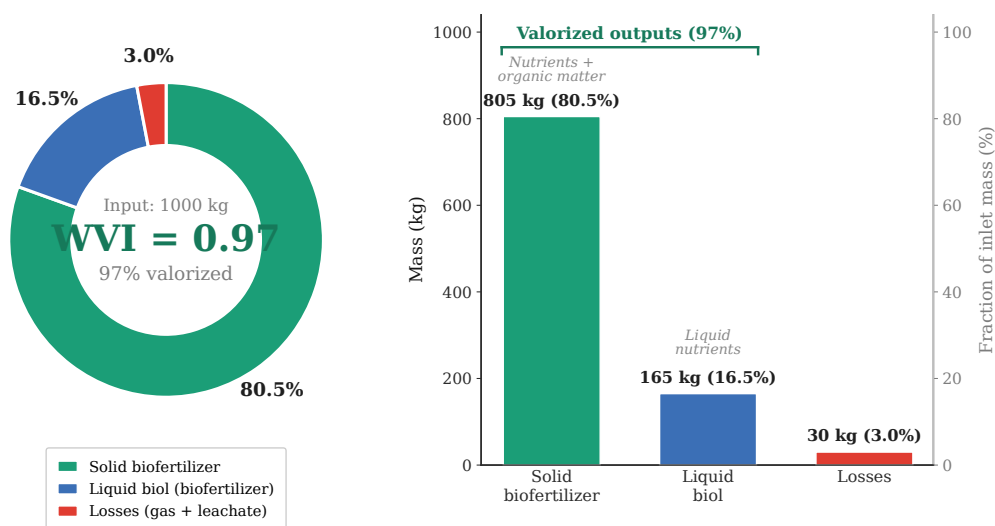


Figure 2. System mass balance per functional unit (1000 kg of waste). Ninety-seven percent of the input mass is recovered as bioabono and biol. The small (3%) gaseous-plus-leachate loss, reported on a feedstock basis, is consistent with suppressed methanogenesis; because the products are wet and not all flows were independently quantified, it is not used as a strict upper bound on fugitive methane.

3.2. Physicochemical Characterization of the Products

Table 3 summarizes the characterization of the bioabono and biol. Both products are strongly acidic ($\text{pH} \approx 3.7$) and saline (electrical conductivity between 9 and 12.5 dS/m). The acidity confirms, at the product level, the acidogenic regime already inferred from

the mass balance: the measured pH is incompatible with methanogenic activity, which is inhibited below $\text{pH} \sim 6$. This condition has a dual implication, agronomic and environmental. Agronomically, the acidity and salinity require subsurface application under an organic soil layer to buffer effects on the rhizosphere, consistent with the in situ reclamation scheme. Environmentally, it supports the hypothesis of low fugitive-methane emissions, which is evaluated in subsequent sections. It should nevertheless be acknowledged that a low bulk pH does not, by itself, guarantee the complete absence of methane formation: localized microenvironments or spatial heterogeneity within the container could sustain limited pockets of methanogenic activity even when the bulk product is strongly acidic. This possibility does not alter the qualitative conclusion of suppressed methanogenesis, but it reinforces the need for direct gas-stream measurement to close the carbon balance conclusively (Section 3.4).

Table 3. Physicochemical and macronutrient characterization of bioabono and biol (range of three experimental runs with different inoculum doses). Nutrient content of bioabono is reported on a dry matter basis (d.b.). Phosphorus and potassium contents are reported as the element (P, K); the oxide equivalents (P_2O_5 , K_2O) used in the FSR (Section 3.3) are obtained with the standard stoichiometric factors 2.29 and 1.205, respectively.

Parameter	Bioabono (Solid)	Biol (Liquid)
pH	3.71–3.81	3.71–3.75
Electrical conductivity (dS/m)	12.2–12.5	9.08–12.1
Moisture (%)	81.6–86.1	—
Total nitrogen	1.93–2.26% (d.b.)	1041.6–1198.4 mg/L
Phosphorus (as P)	0.92–1.07% (d.b.)	330.2–474.9 mg/L
Potassium (as K)	1.31–1.84% (d.b.)	1400–1620 mg/L

3.3. Fertilizer Substitution Capacity (FSR)

Applying Equation (6) to the contents in Table 3, and using a bioabono dry matter fraction of 13.9–18.4% (derived from measured moisture content) together with standard conversions of P and K to their oxide equivalents, the mineral nutrient displacement per functional unit is 2.3–3.5 kg N, 1.2–1.8 kg P_2O_5 , and 1.7–3.1 kg K_2O (Figure 3). Nitrogen is derived almost entirely from the solid fraction; despite its large volume, the contribution of the biol to N and P is marginal due to dilution effects. The uncertainty in these values is dominated by the dry matter fraction and the expression of nitrogen on a dry basis, rather than analytical variability between runs; an additional, second-order contribution arises from day-to-day variability in the dining menu (Section 2.3), which is partly captured by the three-run range; more precise determination of these parameters would directly narrow the range. These results establish nutrient circularity in a robust manner, independent of energy or carbon balance considerations.

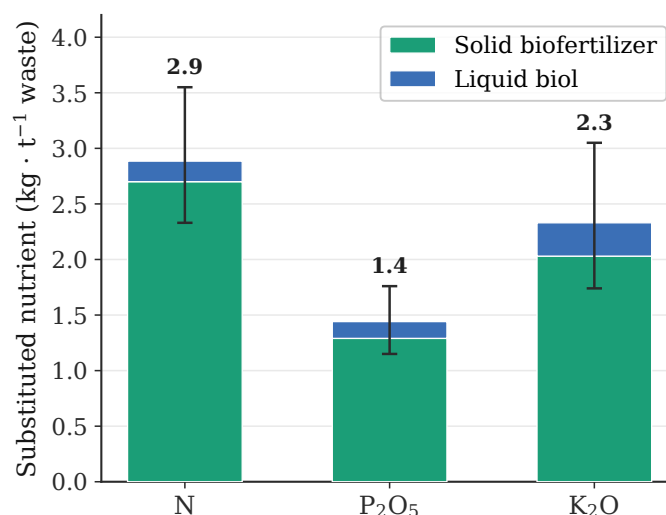


Figure 3. Mineral nutrient displacement induced by the biofertilizer (FSR) per functional unit, disaggregated into the contributions of bioabono and biol. Error bars represent the range associated with the dry matter fraction and the expression of nitrogen on a dry basis; analytical dispersion between experimental runs is secondary.

3.4. Carbon Balance and Decarbonization Index (DCI)

The system carbon balance, evaluated as a first-order screening based on the primary inventory and literature emission factors (Table 4), reveals an asymmetric and fragile structure. The credit from avoided mineral fertilizer production amounts to approximately 18 kg CO₂-eq per ton of waste treated, of which 83% is driven by nitrogen substitution. Process emissions—electricity, internal transport, and prorated polypropylene containers, together with molasses—sum to approximately 9.4 kg CO₂-eq, with polypropylene (≈ 2.8 kg CO₂-eq from 1.5 kg prorated per cycle) and transport being the dominant contributors (Figure 4). The electricity term corresponds to the representative per-cycle homogenization energy (≈ 7 kWh, the midpoint of the metered range) and the Peruvian grid emission factor, rather than to an extreme of the range; no single input is therefore held at a boundary value that would flatter the balance.

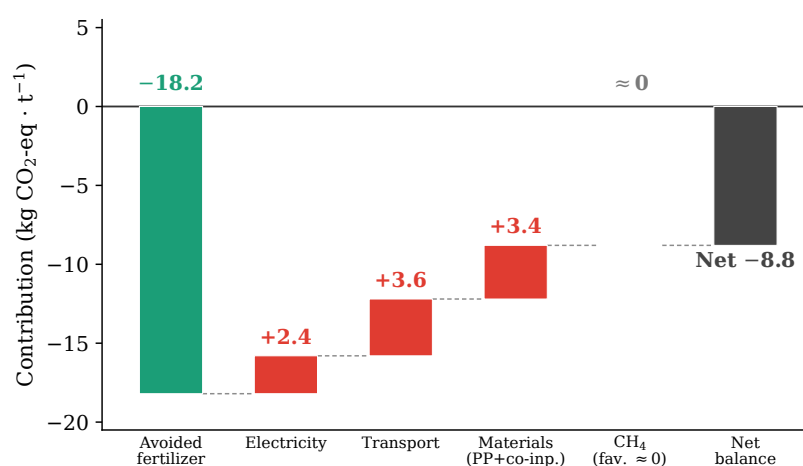


Figure 4. First-order carbon balance per functional unit, under the favourable boundary scenario ($\alpha_k = 1$ and negligible realized methane). Under this upper bound the avoided fertilizer credit (-18.2 kg CO₂-eq, 83% attributable to nitrogen) exceeds process emissions—electricity, transport, and prorated polypropylene containers together with molasses—resulting in a net balance of -8.8 kg CO₂-eq; a fugitive emission of only 0.32 kg CH₄ would be sufficient to neutralize the balance.

Table 4. First-order GHG screening terms per functional unit. The avoided fertilizer term corresponds to the $\alpha_k = 1$ upper-bound substitution scenario. Emission factors are taken from the literature and cross-checked with the ecoinvent database (via SimaPro). The polypropylene emission factor is assumed to be approximately 1.9 kg CO₂-eq/kg (ecoinvent).

Term	Symbol	kg CO ₂ -eq/t	Basis
Avoided fertilizer	$C_{\text{ev,fert}}$	≈18.2	NPK × EF (N: 83%)
Electricity	C_{el}	≈2.4	≈7 kWh (midpoint), Peru grid
Internal transport	C_{tr}	≈3.6	0.3 gal diesel
Materials and co-inputs	C_{mat}	≈3.4	1.5 kg PP (prorated) + 1 L molasses
Fugitive methane	C_{CH_4}	0 to ~1500	pivotal term (Section 3.4)

The result of the decarbonization index, Equation (5), depends jointly on the agronomic-equivalence coefficients and on fugitive methane. Under the upper-bound assumption of full mass-for-mass substitution ($\alpha_k = 1$), the break-even point (DCI = 1) is obtained at

$$m_{\text{CH}_4}^* = \frac{\sum_k \alpha_k m_k \text{EF}_k - C_{\text{process}}}{\text{GWP}_{\text{CH}_4}} \xrightarrow{\alpha_k=1} \frac{18.2 - 9.4}{27} \approx 0.32 \text{ kg CH}_4/\text{t},$$

where $C_{\text{process}} = C_{\text{el}} + C_{\text{tr}} + C_{\text{mat}} \approx 9.4$ kg CO₂-eq. In other words, even in the most favourable substitution scenario, a fugitive emission exceeding approximately 0.32 kg CH₄ per ton is sufficient to shift the system from a potential net GHG benefit to a net GHG burden (Figure 5). This threshold is itself an upper bound, because it assumes $\alpha_k = 1$. Agronomic equivalence is, however, generally lower: organic nitrogen mineralizes gradually, and its first-season plant-available fraction is commonly around one half. Applying an illustrative coefficient $\alpha_N \approx 0.5$ to the dominant nitrogen term (which supplies 83% of the credit), while retaining $\alpha_P = \alpha_K = 1$, reduces the avoided fertilizer credit from 18.2 to ≈10.7 kg CO₂-eq, lowers the break-even methane to ≈0.05 kg CH₄/t, and reduces the zero-methane DCI from ≈1.9 to ≈1.1; a uniform discount ($\alpha_k \approx 0.5$ for all nutrients) would erase the benefit altogether (DCI < 1 even at zero methane). The sign of the climate result is therefore contingent on an agronomic-equivalence factor that was not measured, which we regard as a first-order limitation on equal footing with the unmeasured methane. Table 5 shows the sensitivity of the DCI to fugitive methane under the $\alpha_k = 1$ upper bound.

Table 5. Sensitivity of the decarbonization index to fugitive methane. Values are reported under the $\alpha_k = 1$ upper-bound substitution scenario. The realized methane yield $Y = B_0 f_{\text{conv}}$ (m³ CH₄/kg VS) is applied to Equation (1) with $m_{\text{SV}} = 276$ kg, $\kappa = 0$ and $\rho_{\text{CH}_4} = 0.668$ kg m^{−3}; $C_{\text{process}} \approx 9.4$ kg CO₂-eq and $\text{GWP}_{\text{CH}_4} = 27$. Methane values are scenarios, not measurements.

Scenario	$Y = B_0 f_{\text{conv}}$	m_{CH_4} (kg/t)	DCI
Favourable lower-emission boundary	≈0	≈0	≈1.9
Trace methanogenesis	0.001	0.18	≈1.3
Moderate methanogenesis	0.02	3.7	≈0.17
High-conversion scenario (not representative)	0.30	55.3	≈0.012

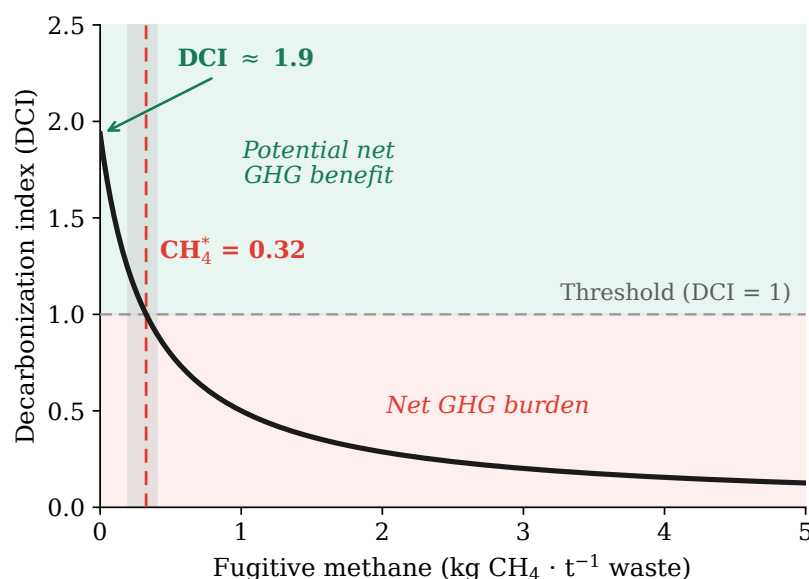


Figure 5. Sensitivity of the decarbonization index to fugitive methane. Under the $\alpha_k = 1$ upper bound, the system shows a potential net GHG benefit ($DCI > 1$) only below approximately 0.32 kg CH_4 per ton (a realistic $\alpha_N < 1$ lowers this threshold, Section 3.4); the shaded band reflects uncertainty in the break-even point (0.20–0.40 kg) arising from emission factors. The favourable boundary condition ($\alpha_k = 1$, realized $CH_4 \approx 0$) places the system in a GHG-beneficial regime, albeit with a narrow margin that vanishes once agronomic equivalence is discounted.

The interpretation is unambiguous: the system is decarbonizing only under a regime of strongly suppressed methanogenesis, which both the measured pH 3.7 and the 3% gaseous loss suggest as plausible, but neither of which demonstrates. The avoided fertilizer credit is structurally insufficient to offset more than a fraction of a kilogram of methane emissions. Consequently, a claim of net decarbonization cannot be established without direct measurement of the gas stream; the two independent lines of evidence available here are necessary but not sufficient conditions. This is the central limitation of the study and, simultaneously, its most actionable finding: it identifies fugitive CH_4 measurement as the decisive variable that separates a truly decarbonizing circular system from one that only closes mass and nutrient loops.

3.5. Circularity Indicator Synthesis

Table 6 integrates the five indicators. The system robustly closes the mass (WVI) and nutrient loops (FSR); the energy loop is absent by design ($ESSR = 0$), as expected for a non-biogas acidogenic fermentation system; the carbon loop (DCI) is conditional on fugitive-methane emissions; and the water loop is likewise open by design ($WCF = 0$), as the documented system does not recirculate water.

Table 6. Synthesis of system circularity indicators and their evidential status.

Indicator	Symbol	Value	Status
Mass valorization	WVI	0.97	Calculated (primary data)
Energy self-sufficiency	ESSR	0	Structural (no biogas recovery)
Fertilizer substitution (N)	FSR_N	2.3–3.5 kg/t	Calculated
Fertilizer substitution (P_2O_5)	FSR_P	1.2–1.8 kg/t	Calculated
Fertilizer substitution (K_2O)	FSR_K	1.7–3.1 kg/t	Calculated
Decarbonization	DCI	conditional (~ 0.01 –1.9)	CH_4 -sensitive
Water loop closure	WCF	0	Structural (no water reuse)

Figure 6 synthesizes the joint behavior of the indicators as a circularity signature and contrasts it with that of a methanogenic biogas-based system. The comparison reveals two distinct archetypes. The acidogenic system studied achieves the highest mass closure (WVI 0.97 vs. 0.91), but collapses both the energy (ESSR = 0) and water (WCF = 0) axes to the centre, whereas the biogas system distributes performance more evenly across the four loops, with energy self-sufficiency and net energy surplus. The carbon closure of the acidogenic system, even under its most favourable boundary scenario, is lower and conditional, compared to the robust closure of the energy-recovering system. This signature operationalizes the concept of circularity beyond biogas: the system is not energetically inferior due to inefficiency, but rather represents a circularity archetype whose value lies in material and nutrient loop closure, with a carbon balance governed by methane suppression rather than energy recovery.

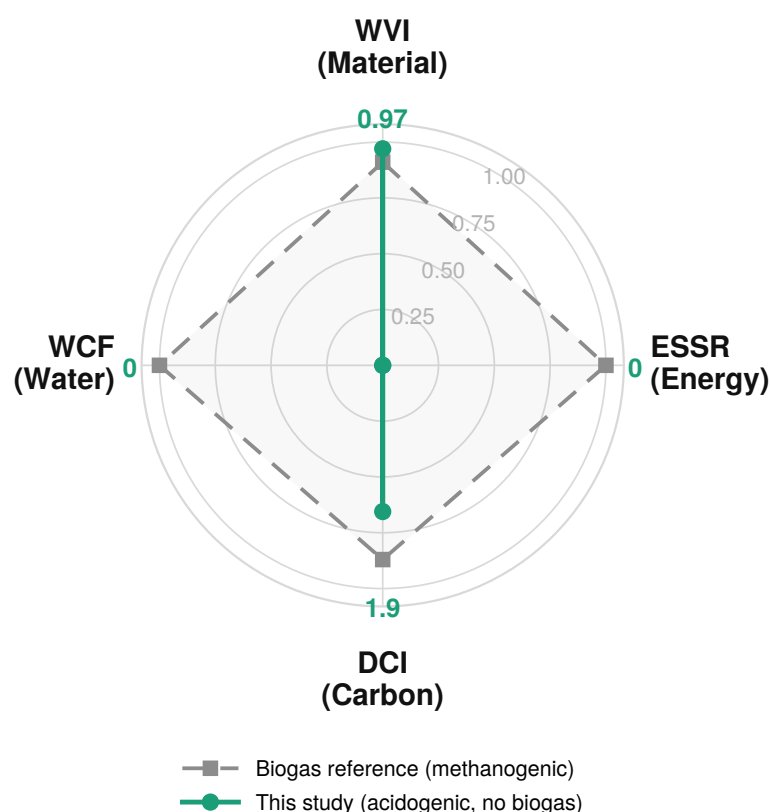


Figure 6. Circularity signature of the acidogenic system (this study) compared with a reference methanogenic biogas recovery system [28]. Each axis represents the normalized closure of a circularity loop in [0,1]: WVI is used directly; ESSR and WCF are truncated at unity (values > 1 indicate surplus); and carbon closure is defined as $DCI / (1 + DCI)$, where 0.50 corresponds to neutrality ($DCI = 1$). Carbon closure for the acidogenic system is conditional on fugitive methane (range 0.50–0.66); for the acidogenic system the documented inventory does not recirculate water, so its water closure is $WCF = 0$. Together with $ESSR = 0$, both the energy and water axes collapse to the centre, so the acidogenic signature reduces to the material–carbon axis while the biogas reference spans all four loops. FSR, being dimensional.

3.6. Probabilistic Uncertainty and Global Sensitivity Analysis

To move beyond interval-based interpretation, the indicators were re-evaluated through Monte Carlo simulation (10,000 iterations) using distributions informed by the pedigree matrix: uniform variability over the measured analytical ranges for moisture and NPK contents; low-dispersion lognormal distributions (GSD 1.05–1.10) for primary mass flows; and higher-dispersion lognormal distributions (GSD 1.2–1.3) for background

emission factors. Figure 7 shows the resulting FSR distributions, with medians and 90% confidence intervals of 2.9 [2.5–3.4] kg N, 1.4 [1.2–1.7] kg P₂O₅, and 2.3 [1.9–2.8] kg K₂O per ton. These intervals, narrower than the deterministic ranges in Section 3.3, confirm the robustness of nutrient circularity.

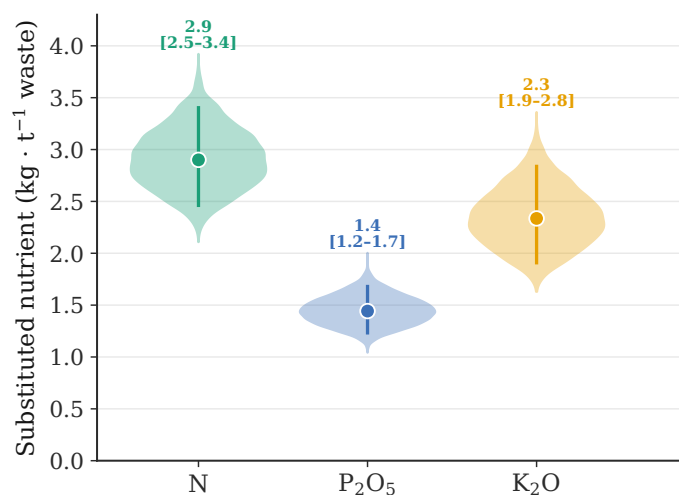


Figure 7. Probability distributions of FSR obtained via Monte Carlo simulation (10,000 iterations), showing median values and 90% confidence intervals for each nutrient. The distributions replace deterministic ranges and reflect the propagation of measured variability and data quality as defined by the pedigree matrix.

Carbon balance uncertainty was propagated analogously, including uncertainty in the polypropylene carbon factor. Under the favourable boundary scenario ($\alpha_k = 1$, negligible realized methane), the net result is -10.0 [-17.5 ; -4.3] kg CO₂-eq per ton, and the system shows a potential net GHG benefit in 99.9% of simulations; the corresponding decarbonization index is 2.0 [1.4–2.9]. This probability is conditional on the two decisive assumptions held at their favourable values ($\alpha_k = 1$ and negligible realized methane) and must not be read as the probability that the real system is climate-beneficial, although with a margin above neutrality that is narrower than in a biogas recovery system.

Global sensitivity was quantified using standardized rank regression coefficients (SRRC; Figure 8). The variance of the FSR is governed by the dry matter fraction (SRRC = +0.77), followed by the bioabono mass and nitrogen content; the contribution of biol is negligible, confirming that precise measurement of moisture and dry-basis nitrogen is the direct pathway to narrowing uncertainty ($R^2 = 0.97$). For the DCI, among parameters other than methane, the dominant driver is the emission factor of avoided mineral nitrogen fertilizer, followed by the grid electricity emission factor and the polypropylene carbon factor of the containers. This shifts mitigation levers toward electricity decarbonization and substitution of virgin polypropylene.

Finally, Figure 9 integrates the results into a decision map representing the DCI in the plane defined by fugitive methane and the emission factor of avoided nitrogen fertilizer. The DCI = 1 boundary is nearly vertical, confirming that fugitive methane—rather than uncertainty in the fertilizer credit—determines the sign of the carbon balance. The favourable boundary scenario (pH-suppressed methanogenesis, $\alpha_k = 1$) lies in the benefit region, but at a distance from the boundary (≈ 0.32 kg CH₄) that is too small to be considered secure without direct measurement. The map thus quantitatively defines the single condition under which the system is decarbonizing.

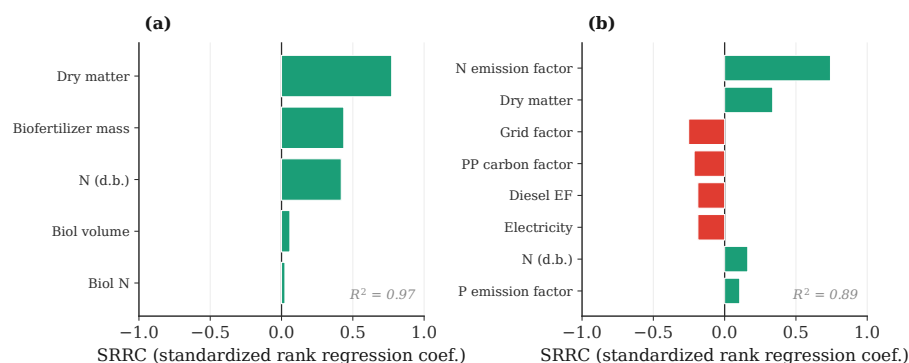


Figure 8. Global sensitivity analysis using standardized rank regression coefficients (SRRCs). (a): drivers of nitrogen FSR variance, dominated by the dry matter fraction. (b): DCI drivers excluding methane, dominated by the emission factor of avoided nitrogen fertilizer, with polypropylene carbon factor as a secondary contributor. The sign indicates direction of influence; R^2 measures the goodness of fit of the rank regression model.

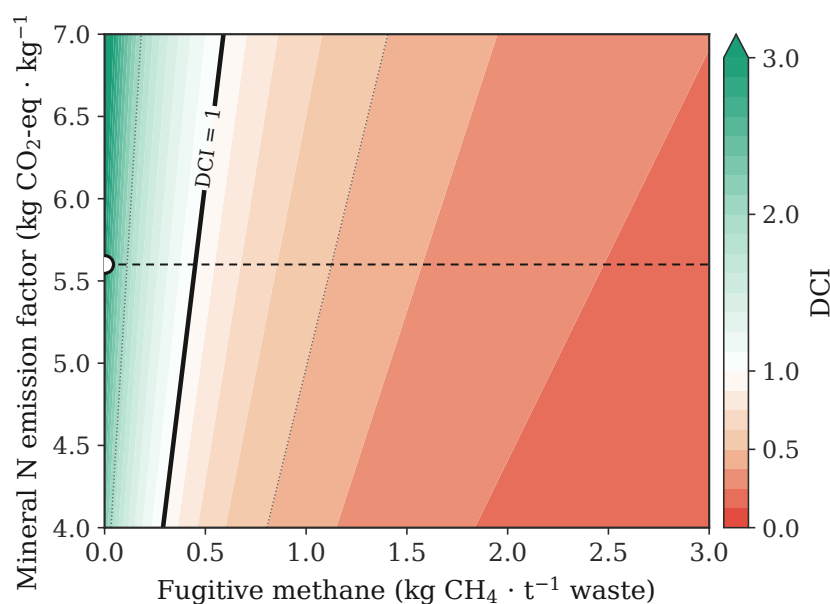


Figure 9. Decarbonization index decision map in the plane of fugitive methane and the emission factor of avoided mineral nitrogen fertilizer, with all other parameters fixed at their central values. The black line represents the DCI = 1 boundary separating a potential net GHG benefit (green) from a net GHG burden (red); the white point indicates the favourable boundary scenario ($\alpha_k = 1$, realized $\text{CH}_4 \approx 0$), not a measured operating condition. The near-vertical boundary highlights methane as the dominant control variable governing the sign of the carbon balance.

3.7. Midpoint Multi-Category Impact Profile (ReCiPe 2016)

The life cycle impact profile across the eighteen ReCiPe 2016 midpoint categories (H) was obtained from the SimaPro characterization using the ecoinvent database. Consistent with the treatment of multifunctionality (Section 2.2), the profile is reported on a gross production basis (cradle-to-gate), while the fertilizer substitution credit is explicitly and transparently accounted for in the carbon balance (Section 3.4) and in the fertilizer substitution rate. Table 7 presents the results, linearly scaled to the functional unit of 1000 kg.

Table 7. Life cycle impact profile across the eighteen ReCiPe 2016 (H) midpoint categories, per functional unit (1000 kg of treated waste), on a gross production basis (cradle-to-gate). Characterization results from SimaPro/ecoinvent, linearly scaled from a 970 kg run.

Impact category	Unit	Value
Climate change	kg CO ₂ -eq	−8.8 ($\alpha_k = 1$; see Table 4) ^a
Stratospheric ozone depletion	kg CFC-11-eq	3.54×10^{-6}
Ionizing radiation	kBq Co-60-eq	0.124
Photochemical ozone formation, human health	kg NO _x -eq	0.0137
Fine particulate matter formation	kg PM2.5-eq	0.00678
Photochemical ozone formation, terrestrial ecosystems	kg NO _x -eq	0.0144
Terrestrial acidification	kg SO ₂ -eq	0.0192
Freshwater eutrophication	kg P-eq	0.00150
Marine eutrophication	kg N-eq	0.000242
Terrestrial ecotoxicity	kg 1,4-DCB	12.2
Freshwater ecotoxicity	kg 1,4-DCB	0.154
Marine ecotoxicity	kg 1,4-DCB	0.202
Human toxicity (carcinogenic)	kg 1,4-DCB	0.369
Human toxicity (non-carcinogenic)	kg 1,4-DCB	2.62
Land use	m ² a crop-eq	0.222
Mineral resource scarcity	kg Cu-eq	0.0132
Fossil resource scarcity	kg oil-eq	4.77
Water consumption	m ³	1.32

^a Climate change is reported here as the net result of the system-specific carbon balance (−8.8 kg CO₂-eq per ton under the $\alpha_k = 1$ favourable boundary; a full term-by-term breakdown can be found in Table 4 and Section 3.4). It is not taken from the multi-category SimaPro run, whose value (66.5 kg CO₂-eq, 970 kg base) is governed by the biogas-handling node of the reference AD template (Figure 10, panel a) and does not represent the acidogenic regime of the system analyzed in this study.

On a production basis, environmental burdens are concentrated in a limited set of categories—fossil resource scarcity (4.77 kg oil-eq), terrestrial ecotoxicity (12.2 kg 1,4-DCB), and water consumption (1.32 m³)—whereas acidification, eutrophication, and particulate matter formation remain comparatively low. Climate change is treated separately (Section 3.4), since the multi-category run result is governed by biogas handling in the original modeled configuration and does not represent the acidogenic system analyzed here, whose specific carbon balance—driven by the avoided fertilizer credit—is reported therein.

The contribution analysis across the most relevant impact categories reveals the origin of these burdens (Figure 10). Rather than a single category, five categories are shown as panels (a)–(e), which promotes to the main text the multi-category profile previously provided as Supplementary Material.

The structure is unequivocal and supports the central finding of the study: in the most relevant environmental impact categories—terrestrial acidification, water consumption, and, analogously, eutrophication and resource depletion—the burden is almost entirely driven by material and energy inputs (notably polypropylene containers and grid electricity), while the biological process contributes negligibly. Climate change is the only category dominated by direct process emissions, and that contribution corresponds to biogas handling in the original configuration; in the acidogenic system without biogas (Section 3.4), this contribution vanishes and the balance is governed by the avoided fertilizer credit. Consequently, the main environmental improvement levers are virgin-polypropylene substitution and grid-electricity decarbonization, in addition to (not instead of) fugitive-methane control.

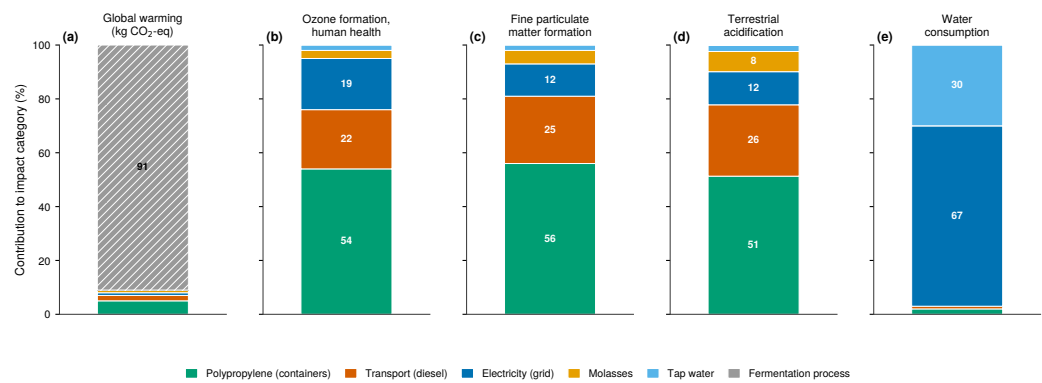


Figure 10. Contribution analysis of ReCiPe 2016 (SimaPro/ecoinvent) characterization results across five midpoint impact categories per functional unit: (a) climate change (global warming), (b) ozone formation, human health, (c) fine particulate matter formation, (d) terrestrial acidification, and (e) water consumption. In panels (b)–(e) the burden is driven almost entirely by material and energy inputs—prorated polypropylene containers, internal diesel transport, and grid electricity—while the fermentation process contributes negligibly. Panel (a) is the only category dominated by the process node (≈91%, shown hatched); this corresponds to the biogas-handling node of the reference AD template and does not represent the acidogenic system, which is why climate change is instead assessed through the system-specific carbon balance (Section 3.4).

4. Discussion

4.1. Positioning of the Circularity Framework Relative to Prior Work

The five-indicator framework used in this study originates from a consolidated methodological line, conceptually initiated by Molina-Moreno et al. [27] for circular economy indicators in anaerobic digestion, and later formalized and aligned with ISO 59020:2024 [29] in the covered lagoon system of Iturralde Carrera et al. [28]. Table 8 positions the present study relative to these antecedents, and Figure 6 provides a graphical comparison of performance signatures.

Table 8. Positioning of the circularity indicator framework relative to the closest prior studies. n.f.: not formalized in equations.

Study	System and Substrate	Regime	Formal Indicators	Distinctive Contribution
Molina-Moreno et al. [27]	Swine anaerobic digestion	Methanogenic	3 (water, biofertilizer, biogas)	Conceptual origin of the framework; all three indicators assume biogas recovery
Cândido et al. [50] (SISTRATES)	Swine-manure treatment with biogas recovery	Methanogenic	n.f.	Applied case without formal equation framework
Iturralde Carrera et al. [28]	Covered lagoon, industrial swine (Cuba)	Methanogenic	5: ESSR = 1.71; WVI = 0.91; DCI = 6.7; FSR = 16.3 t N/year; WCF = 1.30	Formalizes the five indicators and aligns them with ISO 59020; energy-surplus system
This study	Mining cafeteria waste, lactic fermentation (EM inoculum)	Acidogenic (no biogas)	5, with ESSR and DCI redefined: WVI = 0.97; ESSR = 0; DCI ≈ 1.9 (cond.); FSR = 2.9 kg N/t; WCF = 0	Extends the framework to the acidogenic limit without energy recovery; redefines ESSR (=0 structural) and DCI (no energy credit)

The comparison shows that the present work does not merely apply the framework to an additional case, but extends it to a regime not covered by prior studies. All three antecedents are methanogenic systems and construct their energy-related indicators around biogas recovery: the biogas indicator in Molina-Moreno et al., and the ESSR and energy component of the DCI in Iturralde Carrera et al., are not meaningful in systems without usable biogas production. The methodological contribution of this study is, accordingly, twofold (Section 2.6): the extension of the framework so that ESSR and the DCI are well-defined at the zero-biogas limit, and the substantive reformulation of the DCI numerator through the agronomic-equivalence coefficient α_k , the introduction of the fugitive-methane term in its denominator, and the separation of realized yield from biochemical potential. This makes the framework applicable to acidogenic fermentation systems that could not be characterized under previous formulations.

This distinction has a mechanistic basis rather than an operational limitation. The system under study employs effective microorganism (EM) technology—comprising lactic acid bacteria, photosynthetic bacteria, and yeasts—which steers the process toward lactic fermentation rather than methanogenesis. The resulting acidic pH (≈ 3.7) structurally inhibits methanogens, so that the absence of biogas—and therefore $\text{ESSR} = 0$ —is a design attribute of the system rather than a performance deficiency. This interpretation, grounded in inoculum behavior, is consistent with the near-complete retention of mass in the condensed phase (Section 3.1), which is incompatible with the carbon transfer to the gas phase characteristic of methanogenic digestion.

A further methodological point concerns the treatment of the polypropylene containers as capital goods. The contribution analysis (Figure 10) shows that these containers are the single largest source of impact in most categories, which is significant because LCA studies frequently omit capital goods on the assumption that their contribution is negligible once amortized over a long service life. Evidence indicates, however, that for certain processes—notably anaerobic digestion and drying—the impact of capital goods cannot be excluded without distorting the result [51,52]. The present system is a clear instance of this: the containers are durable assets reused over many fermentation cycles rather than operational consumables, and their prorated burden therefore depends directly on the number of cycles achieved per container. This has a direct design implication: extending container service life—increasing the number of cycles over which the fixed manufacturing burden is amortized—reduces the prorated impact per functional unit proportionally, and is thus a more effective mitigation lever than any intervention on the biological process itself. Substituting virgin polypropylene with recycled material acts on the same term from the material-intensity side. This reframes the environmental optimization of the system as a capital-good management problem rather than a process-emission problem.

Figure 11 directly contrasts the three comparable dimensionless indicators of both systems.

The direct comparison with the biogas system of Iturralde Carrera et al. [28]—the only prior study where all five indicators are fully quantified—reveals two opposite circularity archetypes. The acidogenic system achieves higher mass closure (WVI 0.97 vs. 0.91), a direct consequence of lactic fermentation avoiding carbon desorption to the gas phase; however, it collapses the energy loop (ESSR 0 vs. 1.71) and exhibits a lower and conditional carbon closure (DCI ≈ 1.9 vs. 6.7). The comparison of FSR is not directly meaningful due to differing functional units (kg N per ton treated vs. t N per year), and therefore only dimensionless indicators support magnitude-based interpretation. The observed asymmetry is the inverted counterpart of that reported in the biogas system, where mass and water loops are constrained without effluent reuse while energy and carbon loops are robust: here, the energy loop is absent by design, while mass closure is maximized.

Overall, the framework demonstrates its ability to discriminate between technological configurations that a single aggregated index would obscure, and to position each system according to the specific set of loops it actually closes.

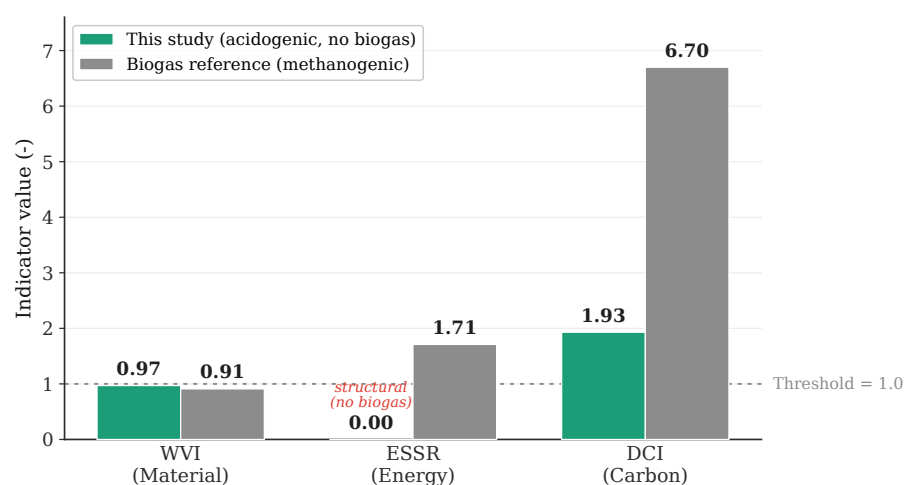


Figure 11. Direct comparison of dimensionless circularity indicators between the acidogenic system of this study and the reference methanogenic biogas recovery system [28]. The acidogenic system shows slightly higher mass closure (WVI), zero energy self-sufficiency (ESSR), and a lower decarbonization index (DCI); FSR is omitted due to differing functional units and lack of direct comparability.

4.2. Comparison of Results with the Literature

The high mass closure obtained (WVI = 0.97; Figure 2) must be interpreted in comparison with the mass performance of the two stabilization pathways against which the system competes. In both aerobic composting and methanogenic digestion, a substantial fraction of the substrate carbon is transferred to the gas phase—as CO₂ in the former [14] and as CH₄+CO₂ in the latter [15]—which reduces the mass yield of the final product. Acidogenic fermentation, by stopping before methanogenesis, preserves carbon in the condensed phase and retains almost the entire mass; however, it should be reiterated that part of this retention corresponds to water (Section 3.1) and should not be confused with a higher solid conversion efficiency. Regarding agronomic quality, the measured nitrogen contents (1.9–2.3 % on a dry basis; Table 3) and the NPK ratio of the products fall within the range reported for digestates and biofertilizers derived from organic waste [18,38], supporting the functional equivalence assumed in the FSR calculation (Figure 7). This functional equivalence is, nonetheless, an approximation: it equates nutrient mass rather than plant-available nutrient. Because the nitrogen in organic amendments mineralizes gradually and is not fully available in the first application cycle, the fertilizer substitution rate and the associated carbon credit should be read as an upper bound on the displaced mineral fertilizer. A more conservative accounting, weighting the organic nitrogen by a first-year availability coefficient, would reduce the nitrogen credit and, given that nitrogen accounts for 83% of the avoided fertilizer term, would narrow the already thin margin of the carbon balance (Section 3.4); it would not, however, affect the mass or nutrient-mass circularity conclusions, which are the most robust findings of the study.

The central finding—that fugitive methane is intrinsically low and not the dominant environmental burden of the system—is consistent with evidence from low-cost digesters in the Andean highlands, where low temperatures, long retention times, and the agricultural reuse of effluents limit biogas production [30,36], including experiences in Cajamarca [37]. This result qualifies the widespread attribution in the literature of a significant fugitive-methane role to systems without energy recovery [15,38]: in the acidogenic regime, methanogenesis is suppressed by acidity (pH ≈ 3.7), shifting the equilibrium toward the

accumulation of volatile fatty acids and CO₂ rather than methane formation, meaning that the assumption of high CH₄ emissions does not apply. This distinction is precisely what makes direct gas measurement—rather than assumption—the key variable separating a decarbonizing system from one that only closes mass and nutrient loops (Figure 5).

The life cycle impact profile across the eighteen ReCiPe 2016 categories, with burdens concentrated in material and energy inputs rather than in the biological process itself (Figure 10), contrasts with the typical pattern of anaerobic digestion LCAs with biogas recovery, where the environmental balance is dominated by methane capture and its substitution of fossil energy, and where the main environmental credit arises from avoided GHG emissions in the energy system [21]. In the absence of this energy loop, environmental improvement opportunities broaden toward replacing virgin polypropylene with recycled material and decarbonizing grid electricity, alongside—rather than instead of—fugitive-methane control—a reorientation of priorities not addressed in the biogas-centered literature.

Finally, the end use of the products—in situ reclamation of land disturbed by the mining operation itself—is consistent with evidence showing that organic amendments can rehabilitate degraded mining soils by restoring organic carbon, nitrogen, and phosphorus and promoting vegetation establishment [4–6]. As noted in Section 2.2, this end use is also what makes the mineral fertilizer counterfactual appropriate: because Peruvian mine closure regulation [44,45] requires revegetation and the establishment of vegetative cover on nutrient-poor substrates conventionally relies on mineral NPK fertilization, the biofertilizer displaces an input that would otherwise be supplied in mineral form. However, the acidity and salinity of the products (Table 3) require subsurface application under an organic cover layer to buffer effects on the rhizosphere, in line with documented reclamation practices and reported applications of organic amendments in mining contexts. This organic cover layer is part of the field-application (use) phase and therefore lies outside the cradle-to-gate boundary adopted here, which is why it does not appear in the inventory (Table 2). In practice, the cover can be provided by the existing topsoil or organic horizon set aside during land stripping, or by the bioabono itself acting as a partial cover; the required cover thickness and mass are site-specific and were not quantified here, as they depend on soil conditions, application depth and local reclamation practice. A full cradle-to-grave assessment incorporating the application phase—including the cover material and its transport—is identified as a natural extension of this work. This chain—mine cafeteria waste transformed into an amendment for the rehabilitation of the same mining site—materializes an internal circular loop that the circular economy literature in mining identifies as a potential opportunity but rarely characterizes with primary data [1,2].

5. Conclusions

This study extended a consolidated family of circular economy indicators to the limiting case of an anaerobic valorization system without biogas recovery, and provided the first environmental characterization based on primary data of the conversion of mining cafeteria waste into biofertilizers for in situ reclamation of high-Andean disturbed land. The main conclusions are as follows.

First, the system achieves a robust closure of mass and nutrient loops, independent of the energy and water balances—both of which are open by design (ESSR = 0, WCF = 0): the Waste Valorization Index reaches WVI = 0.97, and the fertilizer substitution rates amount to 2.9 kg N, 1.4 kg P₂O₅, and 2.3 kg K₂O per ton of treated waste, values confirmed through Monte Carlo simulation. These represent the most solid findings of the study.

Second, the energy loop is absent by design (ESSR = 0). The redefinition of the energy self-sufficiency and decarbonization indicators renders the framework well-defined

at the zero-biogas limit; together with the explicit agronomic-equivalence coefficient α_k , the introduction of the fugitive-methane term into the DCI denominator, and the separation of realized yield from biochemical potential, this constitutes the methodological contribution of this work. It is grounded in a mechanistic basis: the inoculum of effective microorganisms drives the process toward lactic fermentation, inhibiting methanogenesis, such that low methane production is a design attribute of the system rather than a performance deficiency.

Third, the carbon balance is conditional. The avoided mineral fertilizer credit renders the system a potential net GHG benefit only as long as fugitive methane remains below a threshold equilibrium of 0.32 kg CH₄ per ton. The measured acidity (pH $\approx$ 3.7) and the 3% gaseous mass loss support the plausibility of this regime but do not demonstrate it; therefore, a claim of net decarbonization cannot be established without direct measurement of exhaust gas, which is identified as the decisive variable of the system.

Fourth, the life cycle impact profile across eighteen ReCiPe 2016 categories shows that environmental burdens are dominated by material and energy inputs—specifically polypropylene containers and grid electricity—rather than by the biological process itself. Consequently, virgin-polypropylene substitution and electricity decarbonization emerge as improvement levers additional to—not replacing—fugitive-methane control, which remains decisive for the climate result. Because the containers are capital goods amortized over their service life, extending the number of fermentation cycles per container is an equally direct lever, as it lowers the prorated material burden per functional unit.

Fifth, the framework demonstrates that systems without biogas recovery are not energetically inferior due to inefficiency, but rather represent a distinct circularity archetype whose value lies in the closure of material and nutrient loops. Full generalization to other configurations requires replication with measured data from additional systems; direct methane measurement and experimental determination of substrate volatile solids are identified as priority future work.

Supplementary Materials: The following supporting information can be downloaded at: <https://www.mdpi.com/article/10.3390/fermentation12080362/s1>: S1. Inventory data quality assessment (pedigree matrix); Table S1. Pedigree matrix: definition of the data-quality scoring criteria. Table S2. Data-quality assessment of the primary data of the acidogenic fermentation process according to the pedigree matrix. Scale: 1 (best quality) to 5 (worst quality).

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