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ARTICLE

Environmental Assessment of Wastewater Treatment Plants in Developing Countries Using LCA: A Case Study in Perú

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ABSTRACT

Treating municipal wastewater is essential to safeguarding both ecosystem integrity and public health. Although wastewater treatment plants (WWTPs) significantly improve effluent quality, they also incur collateral environmental burdens. In this investigation, a “gate-to-gate" Life Cycle Assessment (LCA) was conducted to analyze the environmental performance of two major WWTPs in Arequipa: La Escalerilla (Plant A, activated sludge) and La Enlozada (Plant B, trickling filters). The analysis was conducted using OpenLCA and the ReCiPe Midpoint (H) 2016 impact assessment method, with a functional unit defined as 1 m³ of treated effluent. Energy consumption emerges as the primary driver for the climate change (GWP100), fossil depletion (FDP), and human toxicity (HTPinf) impact categories, accounting for approximately 75% to 85% of the total effects. Plant A, which requires 0.59 kWh/m³ of electricity, achieves superior nutrient removal reflected in a freshwater eutrophication potential of 1.92×10^{-6} kg P-eq/m³, and exhibits marginally higher CO_2-eq emissions (GWP100) (1.17×10^{-1} kg CO_2-eq/m³). Conversely, Plant B consumes only 0.34 kWh/m³, resulting in a slightly lower GWP100 (1.14×10^{-1} kg CO_2-eq/m³) and a significantly greater reduction in fossil depletion potential

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(FDP) (2.56×10^{-2} kg oil-eq/m³ vs. Plant A's 4.75×10^{-2} kg oil-eq/m³), although it exhibits an elevated eutrophication potential of 4.10×10^{-6} kg P-eq/m³. Both plants meet discharge standards. This study shows that treatment technologies must balance efficiency and sustainability, with energy use being critical. As Peruvian LCA research is scarce, these results offer key insights for future policies.

Keywords: Environmental Impact; OpenLCA; Metodologia Recipe; Wastewater Treatment Plant

1. Introduction

The treatment of waste and wastewater management are important in modern society [1]. In Arequipa (the second most populated city in Peru), the water and sanitation services are important in its development, among them the wastewater treatment.

The growing urbanization and industrialization in the last century has increased the volume of wastewater production [2]. The implementation of a Wastewater Treatment Plant (WWTP) is of utmost importance, ensuring the water quality where it is discharged. However, because of the different processes that compose it, a WWTP also signifies a pollution source, since these generate, directly and indirectly, an environmental impact on their surroundings. The sustainability analysis is necessary in a WWTP as heavy metals or contamination with potentially hazardous elements in urban soils can lead to serious human health repercussions [3]. Pollution of crops and water sources with metals such as lead, mercury, cadmium, arsenic, and chromium can cause neurological disorders, developmental delays, kidney damage, an increased probability of cancer, and respiratory difficulties [4,5].

This is why, since 2016, Life Cycle Analysis (LCA) studies have increased greatly because of the importance that has begun to be given to the impacts generated by wastewater treatment [6]. The International Organization for Standardization defines LCA as a technique that allows the collection, quantification, and evaluation of the environmental damages of any product in its different stages, to make strategic decisions for better environmental performance [7]. Likewise, De Feo et al. stress that its use allows us to evaluate and compare the environmental performance of different systems [8], and Machado et al. assure that LCA is important to achieve WWTP sustainability [9].

Various LCA studies have analysed stages of the treatment process based on established boundaries and units. Depending on the LCA approach, system boundaries are defined, as observed in the work by Mehmeti and Canaj [6], which points out that 40% of the reviewed studies focused solely on the operational stage, excluding infrastructure due to its low impact. For example, Moussavi et al. assessed the operation and construction phases in small-scale plants [10], while Burchart-Korol and Zawartka examined the operation, construction, and dismantling stages of septic tanks [11]. In contrast, Abello-Passteni et al. conducted an LCA of 15 treatment plants in Chile [12], comparing conventional technologies (activated sludge and aerated lagoons) with non-conventional ones (biofilters and vermibiofilters), concluding that the latter are more eco-efficient. Yeo et al. [13], on the other hand, analysed the environmental impact and comprehensive cost-benefit analysis of wastewater treatment. Furthermore, recent studies emphasize LCA's potential to advance sustainable models. Lima et al. [14], for instance, underscore that nutrient and water recovery in treatment systems will be pivotal for transitioning to a circular economy. This study evaluated 12 scenarios in a Brazilian region using the Santiago software, reinforcing the importance of LCA for efficiently comparing alternative scenarios.

In the majority of LCA studies of wastewater treatment, it is very common that energy consumption is the main source that generates a significant impact [15]. Direct gaseous emissions and energy consumption have greater effects on environmental impacts. Sabeen et al. stress that direct emissions are Greenhouse Gases (GHG) emitted by each treatment process within the system boundaries [16], while indirect emissions are related to energy consumption. Kyung et al. concluded that the operation stage contributed approximately 99% of GHG emissions in the LCA, generated in the WWTP secondary process [17].

The relationship between operation, GHG emissions, and eutrophication is also significant. Lopes et al. [18], in a Brazilian study, highlight that operational problems at plants directly influence the results of impact assessment, exacerbating effects such as eutrophication. This work also identifies that the lack of comprehensive studies in the region hinders the creation of tools for a robust LCA, limiting informed decision-making.

To model the different systems, the open-source software OpenLCA was used, which is the third most used software with 12% [6], with Sima Pro being the first. Some studies (22%) did not specify the software used, while the database that was chosen was Ecoinvent v3.8, which, with 80%, is the most used.

The scarcity of research in Latin America and the current lack of data emerge in different studies, making them irreproducible. Rebello et al. present the following limitations: (1) the exclusive evaluation of the operational phase; (2) the use of previously used bibliographic data, as not all the necessary data are available [19]. Likewise, Nguyen et al. mention that at that time, no study had thoroughly investigated the estimation methods to obtain GHGs when applying a LCA to evaluate the total impacts [20]. In this regard, Lopes et al. reinforce that these methodological and operational gaps hinder the standardization of environmental assessments in the Latin American context [18]. In this context, with LCA research focused on WWTP in Peru being almost non-existent, this study represents a pioneering effort to set an important benchmark and incentivize further research in this field both in Peru and in Latin America.

This study evaluated the environmental performance of two main WWTPs within the Arequipa region, at The Escalerilla (plant A) and The Enlozada (plant B), to detect the most critical environmental points, and to verify if the Peruvian regulations are met, and to investigate the improvement possibilities in terms of environmental impact.

2. Materials and Methods

We performed the study following the methodology of (ISO 14040:2006, 2006) based on 4 stages [7]:

- Definition of the objective and scope
- Inventory analysis
- Evaluation of the environmental impact
- Interpretation of data

2.1. Definition of Objective and Scope

The study of both plants identifies the environmental impact generated by the effluents during one year and sets a milestone in LCA studies for Peru. The characteristics of both plants are summarized in **Table 1**, where both differ in terms of: (1) the flow rate that each plant treats (which implies a different size for each), (2) different secondary treatment technology, since plant A has an activated sludge system and plant B has a trickling filter system, and (3) the final effluent disposal. Although plant A has a tertiary treatment, this is not used, since the effluent that comes out of the secondary treatment has the optimal parameters to be discharged into an intermittent stream and it complies with the current regulations. Plant B has a discharge-reuse authorization, and as indicated by the Diagnosis of the National Superintendence of Sanitation Services [21], 65% of its effluent is reused for a processing plant in the mining sector, and the rest is discharged into the Chili River for its recharge.

Table 1. Characteristics of Both WWTPs.

PLANT	A	B
Flow (m^3/d)	6,707.04	146,066.11
Treatments	Pre-treatment	Primary
	Secondary (Activated Sludge System)	Secondary (Trickling Filter System)
	-	Tertiary
Final disposal of the effluent	Intermittent river	Seasonal river

The construction and dismantling stages were excluded. The study focused solely on the water line corresponding to the operation stage. The operational phase impacts are much greater than the construction phase for conventional activated sludge treatment systems [22]. The construction stage only generates 4% of the total environmental impact of a WWTP [23]. Nguyen et al. confirmed that the construction phase has an insignificant impact compared to the operational phase [20]. Therefore, this study is a "gate-to-gate" LCA for both WWTPs. The functional unit considered was 1 m³ of wastewater, so the results were standardized to this. The functional unit choice influences the final results [20]. The boundaries and processes of the WWTPs are configured as follows: the orange line represents the current study focus, while the green line refers to the sludge process (**Figure 1**).

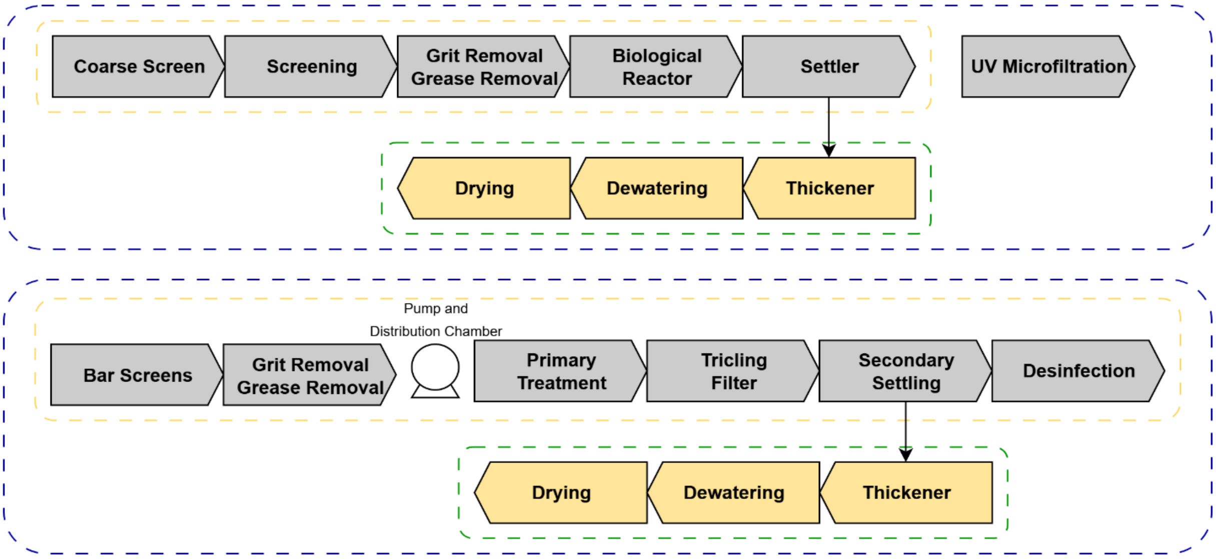

Figure 1. Flow Diagram and Boundaries of Plants (A) and (B).

2.2. Life Cycle Inventory

The companies in charge of the operation of both WWTPs provided the average annual data for 2022. Together with technical visits carried out by us with the help of those responsible for the plants, we obtained the inputs and outputs for each WWTP. Likewise, to satisfy the variable needs, the literature and the same EcoInvent database were taken as references, varying the input and output data for both plants. This base process was taken from E: Water supply; sewerage, waste management and remediation activities in the subfolder 3700: Sewerage from the professional Ecoinvent v3.8 database:

- Electricity consumption data were provided by both plants, with the advantage that Ecoinvent v3.8 provides us with specific electricity generation processes for Peru.
- We assumed the input data regarding total nitrogen (N_T) and total phosphorus (P_T) that enters both plants [24].
- Although the implementation of new and sophisticated technologies has reduced the output concentrations of heavy metals in the WWTPs, we considered them in this study, since 20% of the total heavy metal load from wastewater contributes to the impact on agricultural soils [25]. The input and output data from the plants regarding heavy metals were assumed from [26], which studied the spatial variability of 22 WWTPs in Latin America and the Caribbean. Regarding GHGs, methane (CH_4) and nitrous oxide (N_2O) were estimated for both plants according to IPCC guidelines – 2006 [27].
- Regarding chemical consumption, in the case of plant B, chlorine gas is added at the entrance of the secondary sedimentation tanks and at the entrance of the contact chamber for appropriate pathogen elimination. This data was provided by the same service provider company.
- The deodorization towers of both WWTPs were not taken into account because the chemical products that were added to purify the collected gas, both their production and transport, represented less than 1% of the total impact of all the categories studied [28], so it was considered negligible.

Tables 2 and **3** provide a brief summary of the data evaluation. The plants' different configurations and water treatment capacities allow for a more detailed assessment of their performance.

Table 2. Inputs and Outputs of Wastewater Treatment Plant (A).

Parameters	Influent	Effluent Conc./Gas Emissions/Power Consumption	Unit
Actual Data Collected			
BOD5	870.65	16.75	mg/L
COD	1823.17	65.44	mg/L
TSS	653.08	46.40	mg/L
Energy consumption	-	0.59	KWH/m^3
Estimated Data			
Nt	61	27.6	mg/L
Pt	11	6.6	mg/L
Al	0.24	0.010	mg/L
Cd	0.05	0.026	mg/L
Cu	0.05	0.021	mg/L
Cr	0.03	0.025	mg/L
Pb	0.07	0.032	mg/L
Zn	0.18	0.007	mg/L
CH_4	-	0.0035	Kg/m^3
N_2O	-	0.0000244	Kg/m^3

Table 3. Inputs and Outputs of Wastewater Treatment Plant (B).

Parameters	Influent	Effluent Conc./Gas Emissions/Power Consumption	Unit
Actual Data Collected			
BOD5	428.95	11.51	mg/L
COD	874.63	86.58	mg/L
TSS	372.06	12.99	mg/L
Energy consumption	-	0.34	KWH/m^3
Estimated Data			
Nt	69	20.7	mg/L
Pt	15	2.2	mg/L
Chlorine Gas	0.00509	-	Kg/m^3
NaClO	0.00194	-	Kg/m^3
Al	0.24	0.010	mg/L
Cd	0.05	0.026	mg/L
Cu	0.05	0.021	mg/L
Cr	0.03	0.025	mg/L
Pb	0.07	0.032	mg/L
Zn	0.18	0.007	mg/L
CH_4	-	0.00626	Kg/m^3
N_2O	-	0.000173	Kg/m^3

2.3. Evaluation of Life Cycle Impact

For the evaluation, the results focused on midpoint impacts, which is the most common for this study type and seeks cause-effect results at an early age related to change and environmental quality issues [23].

Open LCA was chosen as it is an open-use program, which the work by Mehmeti and Canaj indicates is the third most used [6], with Sima Pro being the first, however, in 22% of the studies in their review, the software used was not specified.

The methodology considers the following impact categories:

- Climate change.
- Stratospheric ozone depletion.
- Ionizing radiation.
- Fine particle formation.
- Photochemical ozone formation.
- Terrestrial acidification.
- Freshwater eutrophication.
- Marine eutrophication.
- Toxicity.
- Water use.
- Land use.
- Scarcity of mineral resources.
- Scarcity of fossil resources.

No uncertainty or sensitivity analyses were undertaken within the scope of this study since, in a developing country context like Peru's, neither the statistical datasets nor the complete regional characterisation factors needed to establish robust probability distributions are available. Although ISO 14044 advises assessing the reliability of LCA outcomes via quantitative uncertainty methods, in this first application of LCA to Peruvian wastewater treatment plants, we have chosen to concentrate our discussion on the environmental impacts of WWTP operation under developing country conditions.

3. Results and Discussion

As noted above, this study does not aim to perform a direct side-by-side comparison between the two WWTPs, given their differing scales and operational conditions. Instead, it independently characterises each facility's environmental impact profile using Life Cycle Assessment methodology.

The LCIA results per functional unit for both plants are summarized in **Table 4**. Global warming potential (GWP_{100}) amounts to 0.117 kg CO_2-eq per m³ for Plant A (activated sludge) and 0.114 kg CO_2-eq per m³ for Plant B (trickling filter). Fossil depletion potential (FDP) is 0.0475 kg oil-eq per m³ for Plant A and 0.0256 kg oil-eq per m³ for Plant B. The human toxicity, terrestrial acidification, and photochemical ozone formation categories differ by less than 15 % between the two systems.

Regarding eutrophication, the freshwater eutrophication potential (FEP) is 1.92×10^{-6} kg P-eq per m³ in Plant A versus 4.10×10^{-6} kg P-eq per m³ in Plant B, more than double, reflecting its lower phosphorus removal efficiency. For marine eutrophication potential (MEP), Plant A records 6.10×10^{-5} kg N-eq per m³ compared to 4.05×10^{-5} kg N-eq per m³ in Plant B.

Table 4. Life Cycle Impact Assessment Results Calculated for Plants A and B.

Impact Category	Units	Plant A	Plant B
Climate change - GWP100	kg CO_2-Eq	1.17E-01	1.14E-01
Fossil depletion - FDP	kg oil-Eq	4.75E-02	2.56E-02
Human toxicity – HTPinf	kg 1,4-DCB-Eq	1.66E-02	1.63E-02
Agricultural land occupation - ALOP	m2a	5.93E-03	3.27E-03
Metal depletion - MDP	kg Fe-Eq	1.48E-03	1.41E-03
Freshwater ecotoxicity - FETPinf	kg 1,4-DCB-Eq	6.74E-04	1.38E-03
Ionizing radiation - IRP_HE	kg U235-Eq	6.41E-04	6.80E-04
Water depletion - WDP	m^3	3.83E-04	4.98E-04
Marine ecotoxicity - METPinf	kg 1,4-DCB-Eq	3.72E-04	2.23E-04
Photochemical oxidant formation – POFP	kg NMVOC	2.14E-04	2.11E-04

Table 4. *Cont.*

Impact Category	Units	Plant A	Plant B
Urban land occupation - ULOP	m2a	1.76E-04	1.32E-04
Terrestrial acidification - TAP100	kg SO_2-Eq	1.64E-04	1.22E-04
Particulate matter formation - PMFP	kg PM10-Eq	6.17E-05	5.55E-05
Marine eutrophication – MEP	kg N-Eq	6.10E-05	4.05E-05
Natural land transformation – NLTP	m^2	1.71E-05	9.76E-06
Terrestrial ecotoxicity - TETPinf	kg 1,4-DCB-Eq	6.94E-06	5.40E-06
Freshwater eutrophication - FEP	kg P-Eq	1.92E-06	4.10E-06
Ozone layer depletion – ODPinf	kg CFC-11-Eq	8.34E-09	9.69E-09

3.1. Significant Impact Categories

Among the assessed indicators, three impact categories emerge as the primary drivers of environmental pressure:

- Climate change (GWP_{100})
- Fossil depletion (FDP)
- Human toxicity (HTPinf)

In all three cases, the dominant driver is the electricity consumed during the operational phase, a phenomenon repeatedly documented in the literature for conventional WWTPs [17]. When normalized to the functional unit, the inter-plant difference is subtle for GWP_{100} (–2 %), more pronounced for FDP (–46 %), and attenuated for HTPinf (–3 %).

Electricity consumption is the principal contributor to both GWP_{100} and FDP, accounting for over 85 % of the impacts in Plant A and approximately 75 % in Plant B, in accordance with their energy demands of 0.59 $kWh{\cdot}m^{-3}$ and 0.34 $kWh{\cdot}m^{-3}$, respectively.

The findings are consistent with earlier research [19], which indicates that the most significant environmental burdens of WWTPs arise from energy consumption and nutrient removal efficiency. Moreover, as reported by Kyung et al. [17], 99 % of greenhouse gas emissions in a WWTP occur during the operational phase, underscoring the critical need to optimize energy use in these facilities.

3.2. Climate Change (GWP_{100})

This impact category ranks among the most significant in our assessment, as it is directly linked to energy consumption and greenhouse gas (GHG) emissions [25]. Accordingly, Plant A exhibits a GWP_{100} of 1.17×10^{-1} kg CO_2-eq/m³, whereas Plant B records a slightly lower value of 1.14×10^{-1} kg CO_2-eq/m³.

Direct CH_4 and N_2O emissions account for less than 5 % of the GWP_{100} in Plant A but rise to nearly 10 % in Plant B, owing to its higher specific emission rates of 6.26×10^{-3} kg CH_4/m³ and 1.73×10^{-4} kg N_2O/m³. This discrepancy stems from Plant B's secondary trickling-filter configuration, which operates with lower electricity demand and thus exhibits moderate GHG emissions, consistent with the findings of Allami et al. [29]. In contrast, Plant A's activated-sludge process entails greater energy consumption, resulting in elevated GHG outputs illustrating that higher electricity use directly translates into increased emissions [17]. Furthermore, a statistical synthesis of 89 LCA case studies by Li et al. demonstrated electricity demands spanning 0.036 to 2.17 kWh m^{-3}, situating our results firmly within this established range [30].

3.3. Fossil Depletion (FDP)

Fossil resource depletion (FDP) refers to the consumption of hydrocarbon-based resources (coal, oil, and natural gas). In this category, Plant A registers 4.75×10^{-2} kg oil-eq/m³, while Plant B reaches 2.56×10^{-2} kg oil-eq/m³, reinforcing the direct link between energy use and environmental footprint. The forced-aeration requirement of the activated-sludge process substantially raises electricity demand and thus FDP which is further exacerbated by Peru's electricity mix, where roughly 40 % of power still derives from thermal plants. Comparable values have been documented in prior studies [17,28], confirming the coherence of our results.

3.4. Human Toxicity (HTPinf)

The Human Toxicity Potential (HTPinf) category quantifies the potential health damage from releasing toxic substances (heavy metals, organic compounds, chlorination by-products, etc.) into air, water, and soil. In our study, this indicator shows similar values for both plants (difference < 15 %).

Electricity generation, approximately 40 % supplied by thermal power plants in Peru's energy mix, accounts for 65 % to 75 % of the HTPinf in both facilities. Specifically, Plant A records an HTPinf of 1.66×10^{-2} kg 1,4-DCB-eq/m³, compared to 1.63×10^{-2} kg 1,4-DCB-eq/m³ for Plant B, corresponding to 12 % higher toxicity in Plant A due to its greater energy consumption (0.59 vs. 0.34 kWh/m³) and the indirect emissions associated with its activated-sludge system.

This energy–toxicity imbalance underscores the need to:

- **Optimize electricity use:** Employ variable-frequency drives on blowers and pumps, and use fuzzy-logic aeration control to match demand to load.
- **Transition to lower-emission sources:** Procure certified renewable energy or install on-site generation (solar, wind) to reduce the thermal share of the mix.

Implementing these measures would significantly reduce HTPinf without compromising treatment efficiency.

3.5. Freshwater Ecotoxicity (FETPinf)

Freshwater ecotoxicity is an impact category that quantifies the toxic potential of chemical substances released into the environment that subsequently affect aquatic organisms. Concerning this category, Plant A shows an impact of 6.74×10^{-3} kg 1,4-DCB eq, in contrast to Plant B, which registers an impact of 1.38×10^{-2} kg 1,4-DCB eq. This difference arises from the discharge of aqueous contaminants, with residual chlorine being a key factor contributing to freshwater ecotoxicity. It should be emphasised that Plant A displays a lower value due to its lack of a chlorination system, as its effluent is discharged into a non-flowing river. In contrast, Plant B does utilise a chlorination system.

3.6. Freshwater Eutrophication - FEP

Freshwater eutrophication potential (FEP) is the increase in nutrient concentrations, chiefly nitrogen and phosphorus, in freshwater bodies, leading to excessive algal blooms, diminished dissolved oxygen, and altered biodiversity. In this category, Plant A records 1.92×10^{-6} kg P-eq/m³, while Plant B reaches 4.10×10^{-6} kg P-eq/m³. These results align with prior research showing that activated-sludge systems deliver superior nutrient-removal performance despite higher energy demands [6], whereas trickling-filter technologies, though more energy-efficient and lower in GHG emissions, exhibit reduced phosphorus removal capacity.

Because eutrophication is a primary concern in wastewater treatment, rigorous monitoring and control are essential to protect receiving water bodies. The scientific literature underscores the significance of this impact for WWTPs [31–34], and our findings concur with the study by Garfí et al. [31], validating the link between treatment technology and eutrophication-potential reduction. Notably, since FEP is highly sensitive to phosphorus-removal efficiency, a mere 5 % improvement in Plant B's phosphorus removal could yield nearly a 25 % decrease in its FEP.

3.7. Comparison of Environmental Performance with Regional and Latin American WWTPs

Contextualizing the environmental performance of Arequipa's WWTPs proves challenging due to the scarcity of comparable LCA studies in Peru and South America [6,26]. Nevertheless, our findings align with global and regional trends: energy consumption remains the dominant environmental impact driver in conventional WWTPs [17,19]. GWP100 and FDP values for both plants fall within ranges reported in developing countries with mixed energy grids [14,15,18,28]. Nutrient removal efficiency and FEP also conform to known technological characteristics [6,33]. Despite limited direct comparisons, this study sets a valuable benchmark for future research in Peru and Latin America, highlighting the need for standardized data to inform regional environmental policies [12,26].

4. Conclusions

This study underscores the importance of selecting wastewater-treatment technologies that balance energy efficiency and nutrient-removal effectiveness. Plant A's activated-sludge system achieves superior phosphorus elimination lowering freshwater eutrophication potential (FEP) to 1.92×10^{-6} kg P-eq/m³ versus 4.10×10^{-6} kg P-eq/m³ for Plant B's trickling filters but this comes at the cost of higher energy consumption and correspondingly greater impacts on climate change (GWP_{100}) and fossil-resource depletion (FDP). Accordingly, it is critical to implement measures that both optimize electrical demand and maintain high nutrient-removal rates, thereby promoting truly sustainable water-resource management.

In both plants, the electricity mix exerts the most substantial environmental influence, contributing up to 70 % of impacts in specific categories, as reported in numerous previous studies. A clear trade-off emerges: high-energy systems like activated sludge deliver better nutrient removal yet generate more greenhouse gases, whereas low-energy technologies such as trickling filters curb GHG emissions but exacerbate eutrophication potential.

Although both facilities meet their Maximum Permissible Limits [35], regulatory compliance alone does not guarantee environmental sustainability. Strategies to further reduce WWTP footprints should include improving energy efficiency, procuring renewable electricity, and valorizing biogas produced in the sludge line.

Critically, this work reveals a fundamental limitation in Peru's LCA applicability: the absence of region-specific datasets (e.g., statistical inventories, characterization factors) precluded robust uncertainty and sensitivity analyses. Future research must prioritize developing these resources to establish reliable probability distributions and enhance decision-making for Peruvian WWTPs.

Given Arequipa's semi-arid climate, any enhancements in energy management substantially curb fossil-fuel depletion. Thus, while nutrient removal remains a core objective of wastewater treatment, initiatives to reduce energy demand and switch to cleaner electricity sources can yield significant fossil-fuel savings without compromising effluent quality.

Finally, although human toxicity potential (HTPinf) may not rank highest in absolute terms, its cumulative and localized health effects make it a critical concern. We therefore recommend integrating renewable-energy strategies, optimizing chemical-dosing protocols, and enhancing heavy-metal monitoring in future operational improvements and follow-up studies.

Author Contributions

Conceptualization, D.C.; methodology, D.C., H.C. and I.Y.; formal analysis, D.C., H.C. and I.Y.; investigation, D.C., H.C. and I.Y.; data curation, D.C., H.C. and I.Y.; writing—original draft preparation, D.C.; visualization, D.C.; supervision, L.R. All authors have read and agreed to the published version of the manuscript.

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Institutional Review Board Statement

Not applicable.

Informed Consent Statement

Not applicable.

Data Availability Statement

The datasets generated and analyzed during this study are derived from two primary sources: (1) operational data (e.g., energy consumption, effluent quality parameters) provided by the wastewater treatment plant operators under confidentiality agreements, and (2) supplementary data from the Ecoinvent v3.8 database and peer-reviewed literature (referenced in the article). Due to confidentiality restrictions, raw operational data are not publicly available. However, aggregated results, OpenLCA models, and methodological details are available upon reasonable request from the corresponding author (dcastro@unsa.edu.pe) or through collaboration with the participating institutions. Publicly accessible data, including Ecoinvent inputs and literature-derived assumptions, are fully cited within the manuscript. Researchers interested in replicating this study may contact the authors for guidance on accessing non-sensitive components of the dataset.

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Conflicts of Interest

The authors declare no conflict of interest. The funders had no role in the design of the study; in the collection, analyses, or interpretation of data; in the writing of the manuscript; or in the decision to publish the results.

References

[1] Biancardi, A., Colasante, A., D'Adamo, I., 2023. Sustainable education and youth confidence as pillars of future civil society. Scientific Reports. 13(1), 955. DOI: https://doi.org/10.1038/s41598-023-28143-9

[2] Tsangas, M., Papamichael, I., Banti, D., et al., 2023. LCA of municipal wastewater treatment. Chemosphere. 341, 139952. DOI: https://doi.org/10.1016/j.chemosphere.2023.139952

[3] Aslanidis, P.-S.C., Golia, E.E., 2022. Urban Sustainability at Risk Due to Soil Pollution by Heavy Metals—Case Study: Volos, Greece. Land. 11(7), 1016. DOI: https://doi.org/10.3390/land11071016

[4] Bost, M., Houdart, S., Oberli, M., et al., 2016. Dietary copper and human health: Current evidence and unresolved issues. Journal of Trace Elements in Medicine and Biology. 35, 107–115. DOI: https://doi.org/10.1016/j.jtemb.2016.02.006

[5] Chasapis, C.T., Ntoupa, P.-S.A., Spiliopoulou, C.A., et al., 2020. Recent aspects of the effects of zinc on human health. Archives of Toxicology. 94(5), 1443–1460. DOI: https://doi.org/10.1007/s00204-020-02702-9

[6] Mehmeti, A., Canaj, K., 2022. Environmental Assessment of Wastewater Treatment and Reuse for Irrigation: A Mini-Review of LCA Studies. Resources. 11(10), 94. DOI: https://doi.org/10.3390/resources11100094

[7] ISO 14040:2006, 2006. Environmental management — Life cycle assessment — Principles and framework. International Organization for Standardization: Geneva, Switzerland.

[8] De Feo, G., Ferrara, C., Iuliano, G., 2016. Comparative Life Cycle Assessment (LCA) of two on-site small-scale activated sludge total oxidation systems in plastic and vibrated reinforced concrete. Sustainability. 8(3), 212. DOI: https://doi.org/10.3390/su8030212

[9] Machado, A.P., Urbano, L., Brito, A.G., et al., 2007. Life cycle assessment of wastewater treatment options for small and decentralized communities. Water Science and Technology. 56(3), 15–22. DOI: https://doi.org/10.2166/wst.2007.497

[10] Moussavi, S., Thompson, M., Li, S., et al., 2021. Assessment of small mechanical wastewater treatment plants: Relative life cycle environmental impacts of construction and operations. Journal of Environmental Management. 292, 112802. DOI: https://doi.org/10.1016/j.jenvman.2021.112802

[11] Burchart-Korol, D., Zawartka, P., 2019. Environmental life cycle assessment of septic tanks in urban wastewater system – A case study for Poland. Archives of Environmental Protection. 45(4), 68–77. DOI: https://doi.org/10.24425/aep.2019.130243

[12] Abello-Passteni, V., Alvear, E.M., Lira, S., et al., 2020. Evaluación de eco-eficiencia de tecnologías de tratamiento de aguas residuales domésticas en Chile [in Spanish]. Tecnología y Ciencias del Agua. 11(2), 190–228. DOI: https://doi.org/10.24850/j-tyca-2020-02-05

[13] Yeo, J., Chopra, S.S., von Eiff, D., et al., 2022. An integrated techno-economic analysis on wastewater reclamation in Hong Kong: A comprehensive cost – Benefit analysis with life cycle assessment. Journal of Cleaner Production. 357, 131838. DOI: https://doi.org/10.1016/j.jclepro.2022.131838

[14] Lima, P.M., Lopes, T.A.S., Queiroz, L.M., et al., 2022. Resource-oriented sanitation: Identifying appropriate technologies and environmental gains by coupling Santiago software and life cycle assessment in a Brazilian case study. Science of the Total Environment. 837, 155777. DOI: https://doi.org/10.1016/j.scitotenv.2022.155777

[15] Awad, H., Gar Alalm, M., El-Etriby, H.K., 2019. Environmental and cost life cycle assessment of different alternatives for improvement of wastewater treatment plants in developing countries. Science of the Total Environment. 660, 57–68. DOI: https://doi.org/10.1016/j.scitotenv.2018.12.386

[16] Sabeen, A.H., Noor, Z.Z., Ngadi, N., et al., 2018. Quantification of environmental impacts of domestic wastewater treatment using life cycle assessment: A review. Journal of Cleaner Production. 190, 221–233. DOI: https://doi.org/10.1016/j.jclepro.2018.04.053

[17] Kyung, D., Jung, D.Y., Lim, S.R., 2020. Estima-

tion of greenhouse gas emissions from an underground wastewater treatment plant. Membrane and Water Treatment. 11(3), 173–177. DOI: https://doi.org/10.12989/mwt.2020.11.3.173
[18] Lopes, T.A.S., Queiroz, L.M., Torres, E.A., et al., 2020. Low complexity wastewater treatment process in developing countries: A LCA approach to evaluate environmental gains. Science of the Total Environment. 720, 137593. DOI: https://doi.org/10.1016/j.scitotenv.2020.137593
[19] Rebello, T.A., Gonçalves, R.F., Calmon, J.L., 2022. Mitigation of environmental impacts in warm-weather wastewater treatment plants using the life cycle assessment tool. International Journal of Environmental Science and Technology. 19(5), 4763–4778. DOI: https://doi.org/10.1007/s13762-021-03430-y
[20] Nguyen, T.K.L., Ngo, H.H., Guo, W.S., et al., 2020. A critical review on life cycle assessment and plant-wide models towards emission control strategies for greenhouse gas from wastewater treatment plants. Journal of Environmental Management. 264, 110440. DOI: https://doi.org/10.1016/j.jenvman.2020.110440
[21] Audit Directorate, 2022. Annual wastewater management report. Ministry of Environment: Lima, Peru [In Spanish].
[22] Corominas, L., Foley, J., Guest, J.S., et al., 2013. Life cycle assessment applied to wastewater treatment: State of the art. Water Research. 47(15), 5480–5492. DOI: https://doi.org/10.1016/j.watres.2013.06.049
[23] Lorenzo-Toja, Y., Vázquez-Rowe, I., Amores, M.J., et al., 2016. Benchmarking wastewater treatment plants under an eco-efficiency perspective. Science of the Total Environment. 566, 468–479. DOI: https://doi.org/10.1016/j.scitotenv.2016.05.110
[24] Tchobanoglous, G., Stensel, H.D., Tsuchihashi, R., et al., 2014. Wastewater engineering: Treatment and resource recovery, 5th ed. McGraw-Hill: New York, NY, USA.
[25] Amann, A., Weber, N., Krampe, J., et al., 2022. Systematic data-driven exploration of Austrian wastewater and sludge treatment - Implications for phosphorus governance, costs and environment. Science of the Total Environment. 846, 157401. DOI: https://doi.org/10.1016/j.scitotenv.2022.157401
[26] Hernández-Padilla, F., Margni, M., Noyola, A., et al., 2017. Assessing wastewater treatment in Latin America and the Caribbean: Enhancing life cycle assessment interpretation by regionalization and impact assessment sensibility. Journal of Cleaner Production. 142, 2140–2153. DOI: https://doi.org/10.1016/j.jclepro.2016.11.068
[27] IPCC, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Institute for Global Environmental Strategies (IGES): Kanagawa, Japan.
[28] Hospido, A., Moreira, M.T., Feijoo, G., 2008. A comparison of municipal wastewater treatment plants for big centres of population in Galicia (Spain). The International Journal of Life Cycle Assessment. 13, 57–64. DOI: https://doi.org/10.1065/lca2007.03.314
[29] Allami, D.M., Sorour, M.T., Moustafa, M., et al., 2023. Life Cycle Assessment of a Domestic Wastewater Treatment Plant Simulated with Alternative Operational Designs. Sustainability. 15(11), 9033. DOI: https://doi.org/10.3390/su15119033
[30] Li, Y., Xu, Y., Fu, Z., et al., 2021. Assessment of energy use and environmental impacts of wastewater treatment plants in the entire life cycle: A system meta-analysis. Environmental Research. 198, 110458. DOI: https://doi.org/10.1016/j.envres.2020.110458
[31] Garfí, M., Flores, L., Ferrer, I., 2017. Life Cycle Assessment of wastewater treatment systems for small communities: Activated sludge, constructed wetlands and high rate algal ponds. Journal of Cleaner Production. 161, 211–219. DOI: https://doi.org/10.1016/j.jclepro.2017.05.116
[32] Banti, D.C., Tsangas, M., Samaras, P., et al., 2020. LCA of a Membrane Bioreactor Compared to Activated Sludge System for Municipal Wastewater Treatment. Membranes. 10(12), 421. DOI: https://doi.org/10.3390/membranes10120421
[33] Postacchini, L., Lamichhane, K.M., Furukawa, D., et al., 2016. Life cycle assessment comparison of activated sludge, trickling filter, and high-rate anaerobic-aerobic digestion (HRAAD). Water Science and Technology. 73(10), 2353–2360. DOI: https://doi.org/10.2166/wst.2016.087
[34] Viotti, P., Tatti, F., Bongirolami, S., et al., 2024. Life Cycle Assessment Methodology Applied to a Wastewater Treatment Plant. Water. 16(8), 1177. DOI: https://doi.org/10.3390/w16081177
[35] Peruvian Ministry of Environment, 2010. Maximum Permissible Limits for Effluents from Domestic or Municipal Wastewater Treatment Plants (Supreme Decree No. 003-2010-MINAM). Ministry of Environment: Lima, Peru. (In Spanish)