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Comparative Carbon Footprint Analysis of Sludge Management Pathways in Isolated Regions

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Abstract

In isolated areas, wastewater reuse is a key solution to water scarcity, enabling the completion of the integral water cycle. However, managing sludge from treatment plants in these regions poses significant environmental and economic challenges, particularly due to limited land availability. This study presents a cradle-to-gate comparative carbon footprint analysis of various sludge management pathways, ranging from traditional systems to advanced thermochemical conversion processes. The regional assessment reveals a significantly higher carbon footprint in Fuerteventura (23.0 kg $CO_{2,eq}$/capita · year) compared to Tenerife (13.2 kg $CO_{2,eq}$/capita · year). Centralized thermochemical processing shows the greatest decarbonization potential under the studied conditions; specifically, pyrolysis maximizes the reduction to 54% and 40% for Tenerife and Fuerteventura, respectively. This behavior is due to the carbon footprint recovery associated with pyrolysis byproducts. However, these findings are based solely on carbon footprint considerations and are subject to the technical and operational feasibility of thermochemical processing. These results provide a strategic framework for decarbonizing wastewater treatment plants in similar regions, identifying the most efficient pathways toward achieving carbon neutrality in the sludge line.

Keywords: life cycle assessment; water-energy nexus; membrane bioreactor; conventional activated sludge process; pyrolysis; incineration

1. Introduction

Water scarcity is a growing global challenge linked to economic development and climate change. In this context, the issue is exacerbated in isolated regions due to their geographical conditions [1,2]. Furthermore, when these areas are subject to arid climates with erratic rainfall and significant demographic pressure from tourism, the problem becomes even more acute, necessitating the search for new water sources [1]. To address this, many coastal regions, such as islands, have turned to seawater desalination, with reverse osmosis being the most widely adopted technology despite its high energy demand [2,3]. However, this approach does not support a circular economy model, making wastewater treatment and reuse essential. In this context, membrane bioreactor technology (MBR) produces a higher quality effluent than conventional activated sludge process (CASP), allowing for direct reuse [4], albeit at the cost of higher energy consumption due to membrane fouling [5,6]. This scenario of unconventional water sources clearly illustrates the water-energy nexus.

Achieving effective management of the entire water cycle requires addressing the water-energy nexus, as the water sector is estimated to consume approximately 4% of the

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world's electricity [7]. In this context, various strategies have been implemented, such as incorporating renewable energy into the design of desalination plants [1]. Emerging low-energy membrane technologies such as gravity-driven membrane bioreactors (GDM) have also been proposed as sustainable alternatives for isolated settings, given their reduced energy requirements, minimal sludge production, and low operating costs [8]. Furthermore, another key strategy involves reorienting wastewater treatment plants to become energy-positive [9], aiming to transform them from energy consumers into energy and resource recovery centers [10].

Among the strategies considered to achieve this transformation is the valorization of sludge, given that wastewater contains chemical energy that exceeds treatment requirements by a factor of 10 [10]. In this context, processes such as anaerobic digestion can offset between 25% and 50% of the energy demand in a CASP system [10]. Furthermore, incineration is recognized as a mature technology capable of reducing volume, destroying pathogens, and generating heat and electricity [11]. Another alternative for volume reduction and pathogen elimination is pyrolysis, which yields byproducts such as biochar, tar, and syngas [12]. Ultimately, these thermochemical processes enable effective sludge management while providing improved environmental performance [13].

Despite their advantages, thermochemical processes present environmental and technical limitations that must be addressed. Anaerobic digestion, for instance, has a relatively low recovery rate, as only 45–50% of the volatile solids are converted into biogas [14]. In the case of incineration, the process requires significant energy consumption and the use of external fuels for drying and combustion; thus, while it mitigates the impacts associated with landfilling, it introduces new environmental burdens [12], and its climate change benefits diminish as energy systems transition towards renewable sources, since its primary advantage relies on energy recovery, which provides decreasing carbon credits in increasingly decarbonized grids [13]. Similarly, pyrolysis is an energy-intensive process whose efficiency depends strictly on operating parameters, particularly temperature, which determines the distribution of by-products (char, tar, and syngas) and their associated recovery potential [12]. While incineration achieves significant volume reduction, its climate change benefits are outweighed by pyrolysis in increasingly decarbonized energy systems, where thermochemical processes can achieve net-negative CO_{2eq} emissions through material recovery and carbon fixation in biochar [13]; anaerobic digestion, by contrast, recovers less than 25% of carbon as biogas and offers limited overall environmental benefit compared to thermochemical alternatives [11,13]. A robust strategy for evaluating the benefits and drawbacks of these thermochemical processes compared to conventional treatment is Life Cycle Assessment (LCA) [11]. Specifically, determining the carbon footprint allows for the assessment of direct emissions and energy consumption, enabling a comprehensive analysis to define the optimal strategies for sewage sludge treatment and management [14].

Carbon footprint analysis has been widely employed in several previous studies to evaluate the integrated water cycle. For example, a comprehensive study was conducted on the Scottish islands to assess the entire water cycle, from water treatment to wastewater management, identifying energy consumption and direct emissions from landfills and septic tanks as key areas of concern [7].

Simultaneously, different wastewater treatment plant configurations have been evaluated by analyzing the impact of nutrient removal [14]. This study highlights the need to select appropriate configurations to minimize the environmental footprint and identifies sludge management and final disposal as the primary strategies for decarbonizing the process. Other studies have addressed sludge treatments aimed at volume reduction [15] or have examined thermochemical processes for byproduct recovery [12]. These studies highlight that while sludge management aims to reduce volume or minimize landfill disposal, such processes often result in additional electricity consumption, thereby increasing the environmental footprint. Furthermore, they identify dewatering parameters and the recovery of energy from byproducts as key elements of the process. However, comparative research is still missing regarding conventional systems and MBRs coupled with thermochemical processes for sludge recovery in isolated regions. This study addresses this gap by simultaneously evaluating six sludge management strategies across two secondary treatment technologies (CASP and MBR) in two islands with contrasting energy mixes, where dependence on imported fossil fuels and limited land availability for renewable energy deployment represent key constraints for decarbonization.

In this context, this manuscript proposes a carbon footprint analysis of wastewater treatment processes using CASP and MBR technologies coupled with six sludge management treatment lines, ranging from conventional landfill disposal to advanced thermochemical processes. The specific objectives are to identify optimal decarbonization pathways for sludge management in isolated regions and to quantify the influence of the local energy mix on the carbon footprint of each evaluated scenario, aiming to integrate the full water cycle into the circular economy.

2. Materials and Methods

2.1. Scope Definition and Scenario Analysis

This study analyzes the carbon footprint of various sewage sludge treatment and management strategies with the aim of identifying decarbonization pathways. The carbon footprint was determined using the methodology set out in the ISO 14040 series standards [16–18]. The functional unit adopted in this study is 1 m^3 of treated wastewater, which serves as the basis for normalizing all carbon footprint contributions. Study on Life Cycle Assessment (LCA) are based on four stages: defining the goal and scope, conducting an emissions inventory, quantifying the impact, and interpreting the results. While the objective of this paper has already been described, the study area has not yet been defined as an archetype of an isolated region. In this case, the Canary Islands (specifically Tenerife and Fuerteventura) have been selected. These volcanic islands present significant challenges regarding the water-energy nexus and are recognized by the European Union as Outermost Regions (ORs).

The Canary Islands are experiencing water scarcity due to erratic rainfall, aquifer overexploitation, population growth, and tourism [1]. Conventional water sources are scarce; islands such as Lanzarote and Fuerteventura rely on seawater desalination to meet demand [2]. In this context, wastewater reclamation is crucial for mitigating the problem; however, these processes are energy-intensive and generate sludge. Currently, most of this sludge in the Canary Islands is disposed of in landfills. Furthermore, the island's electricity generation infrastructure relies on imported fossil fuels, with renewable energy penetration varying between islands [19]. Consequently, the Canary Islands serve as a case study for analyzing the decarbonization of sludge treatment in isolated regions.

The characteristics of the influent effects sludge production and, consequently, the results of the study. The physicochemical characteristics of the feedwater were obtained from official sources provided by the Canary Islands' water authorities and represent the mean properties of wastewater throughout 2024 [20]. Table 1 shows the characteristics of the wastewater used in this study.

This study adopts a partial cradle-to-gate approach and considers three types of emissions: direct emissions from wastewater treatment and sludge management; indirect emissions associated with energy and chemicals consumption; and recovered emissions resulting from the production of electricity, heat, or by-products. Pretreatment, primary treatment, and tertiary treatment processes were excluded from the system boundary, as they are either common to all evaluated scenarios or have a marginal contribution to the overall carbon footprint relative to secondary treatment and sludge management stages. This scope was selected as it captures the most energy-intensive and environmentally significant stages of the water cycle in isolated regions, consistent with previous carbon footprint studies on wastewater and sludge management [12–14]. The study evaluates two water treatment lines: a conventional activated sludge process (CASP) and a membrane bioreactor (MBR) system. For the sludge stream, six treatment trains were considered, comprising combinations of gravity thickening, anaerobic digestion, dewatering, thermal drying, incineration, pyrolysis, and landfill disposal. Figure 1 shows a flow diagram of each process, along with the inputs accounted for (e.g., electricity, heat, and chemicals), direct emissions, and recovered emissions (marked in green in Figure 1).

Table 1. Characteristics of wastewater.

Parameter	Units	Value [20]
BDO_5	mg/L	548
COD	mg/L	1044
TN	mg N/L	85
SS	mg/L	345
Conductivity	µS/cm	1984

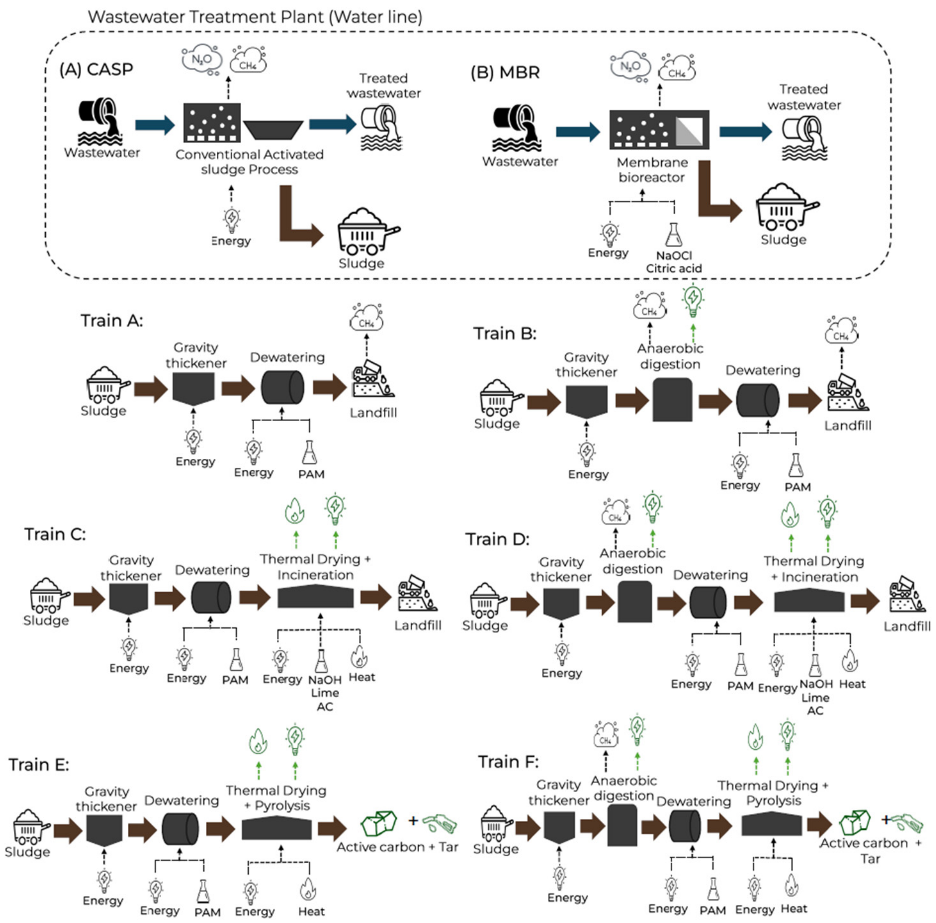

Figure 1. Diagrams showing flows of the different treatment processes studied.

Direct emissions were accounted for using the IPCC methodology [21]. Consequently, direct CO_2 emissions were not considered, as they are of biogenic origin. However, CH_4 and N_2O emissions from biological processes in water treatment, as well as methane emissions from landfill disposal and potential leaks from anaerobic digesters, have been included.

Indirect emissions from energy production require an analysis of energy consumption at each stage. For the wastewater treatment line, only secondary treatment was analyzed (see Figure 1), as it accounts for more than 60% of total consumption; pretreatment and tertiary treatment were considered negligible in this context [5,9]. Regarding sludge treatment lines, the energy consumption of each process was considered individually (see Figure 1), disregarding the energy required for pumping or material transfer between stages. Notably, when sludge transport by truck was evaluated, direct emissions from that stage were also included. Regarding indirect emissions from chemical use, all chemicals commonly used across the various stages were considered. These include membrane cleaning agents for ultrafiltration, polyelectrolytes, ferric chloride, and air pollutant control reagents for the incineration stage. Figure 1 details the specific chemicals used in each stage.

Finally, recovered energy includes electricity generated from biogas during the anaerobic digestion stage, as well as electricity and heat produced through incineration. Additionally, the pyrolysis stage accounts for heat and energy recovery from syngas, the production of activated carbon (AC) from the char, and the use of pyrolytic oil (tar) as a substitute for fuel oil.

2.2. Inventory Analysis and Impact Assessment

The emissions inventory and impact assessment were conducted using the standard IPCC methodology [21], complemented by findings from previous studies. This study incorporates both field data and results reported in specialized literature.

The carbon footprint of the wastewater treatment processes (MBR and CASP) accounts for direct emissions as well as those associated with energy and chemical consumption. These footprints were estimated using literature data and standard design parameters, which are considered representative of normal operating conditions for MBR and CASP technologies. While local variations in influent characteristics may exist, the selected parameters (Table 2) align with established technical specifications and expected performance for municipal wastewater treatment.

Table 2. Design parameters of processes.

Parameters	Units	Value	Source
Biokinetic parameters and coefficients			
Non-biodegradable particulate COD fraction ($f_{XU,DQO}$)	--	0.04	[22]
Non-biodegradable soluble COD fraction ($f_{SU,DQO}$)	--	0.12	[22]
Fraction of volatile solids in the suspension and in the feed (f_{SSVI})	--	0.80	[22]
COD-to-MLVSS ratio (f_{DV})	g COD/g MLVSS	1.48	[22]
Residual endogenous fraction (f_d)	--	0.15	[22]
Cellular yield (Y)	g MLVSS/g bCOD	0.4	[22]
Endogenous degradation coefficient (k_d)	d-1	0.14	[22]
Design conditions for CASP			
Percentage of COD removal in the primary	%	37.5	[22]
$SEC_{recirculation\ pump}$	kWh/m^3	0.009	[14]
$SEC_{second\ settling}$	kWh/m^3	0.0077	[14]
SAEturbines	kgO_2/kWh	1.5	[22]
HRT	h	7.75	[22]
SRT	d	8	[22]
COD effluent	mg/L	90	--

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Table 2. *Cont.*

Parameters	Units	Value	Source
Design conditions for MBR			
Percentage of COD removal in the primary	%	37.5	[22]
$SAE_{coarse\ bubble}$	kgO_2/kWh	1.05	[22]
HRT	h	7.85	[4]
SRT	d	15	[4]
SAD_m	m^3/hm^2	0.3	[23]
J	L/hm^2	20	[23]
ε_B	--	0.7	[23]
ε_M	--	0.9	[23]
P_1	kPa	101.32	[23]
P_2	kPa	135.64	[23]
NaOCl specific consumption	Kg/m^3	0.012	[20]
Citric Acid specific consumption	Kg/m^3	0.006	[20]
COD effluent	mg/L	60	--
Thickening			
Energy consumption	kWh/t DM	5	[14]
Dewatering			
Energy consumption	kWh/t DM	50	[14]
PAM specific consumption	Kg/t DM	3.5	[15]
Anaerobic Digester			
SRT	d	18	[14]
Volatile material destruction rate	%	45	[14]
Energy consumption	kWh/t DM	4.08	[24]
Methane production rate	$Nm^3CH_4/kg\ VM$	0.25	[25]
Methane proportion in biogas	%	73.1	[26]
Ratio of methane losses in the digester	$Nm^3CH_4/m^3biogas$	0.012	[27]
Heating Value	kWh/Nm^3CH_4	10	[28]
Engine efficiency	%	30	[28]
$FeCl_3$ specific consumption	Kg/m^3	0.011	[20]
Transport			
Diesel fuel consumption for trucks	$L/tn_{wet}\cdot km$	0.16	[12]
Thermal Drying			
Volatile material destruction rate	%	7.6	[12]
Electrical Energy consumption	kWh/t DM	0.11	[12]
Heat consumption	MJ/Kg DM	1.8	[12]
Incineration			
Electrical Energy consumption	kWh/t DM	0.42	[12]
Heat consumption	MJ/Kg DM	7.8	[12]
NaOH specific consumption	Kg/t DM	50.9	[12]
Lime specific consumption	Kg/t DM	5.2	[12]
Active Carbon specific consumption	Kg/t DM	1	[12]
Electrical energy recovery	%	10.8	[12]
Heat energy recovery	%	49.8	[12]
Pyrolysis			
Electrical Energy consumption	kWh/t DM	2.89	[12]
Energy in sewage sludge	MJ/kg DM	22.1	[12]
Transfer coefficient of energy in the sludge transferred into Tar	%	42.28	[12]
Transfer coefficient of energy in the sludge transferred into Sin-gas	%	28.57	[12]
Carbon active production	%	10.69	[12]

For the CASP line, energy consumption was determined by accounting for the energy demand of biological aeration, secondary sedimentation, and internal recirculation pumping. For the latter two, standard data from specialized literature were used [14]. The oxygen uptake rate and standard aeration efficiency of the diffusers were used to calculate the air energy consumption required for the biological process. The equations and reference data for these calculations were sourced from [22]. In accordance with the literature, other energy contributions have been considered negligible [9].

A similar methodology was applied to the MBR emissions inventory. In this case, membrane scouring aeration was included in the energy consumption calculations, using expressions and characteristic values proposed in the literature [23], as shown in Table 2. The energy consumption of the permeate pump was neglected, as it is widely accepted that MBR energy demand is predominantly driven by biological aeration and membrane scouring [5]. During continuous MBR operation, chemical consumption arises from membrane cleaning processes, specifically using citric acid and sodium hypochlorite. Specific consumption rates for these chemicals were derived from official data published by the operators of this type of wastewater treatment plant on the island of Tenerife [20] (see Table 2).

In this manuscript, specific sludge production is evaluated based on the daily mass generated and the influent flow rate, as the latter serves as the functional unit for the LCA. Specific sludge production, expressed in kg VSS/m^3, is calculated as the sum of the endogenous residue, heterotrophic biomass production, and the contribution of non-biodegradable volatile solids from the influent stream. These values were determined using the parameters listed in Table 2 and the expressions proposed in the relevant literature [22].

For the thickening stage, only energy consumption was accounted for based on literature data, as direct emissions from this process are considered negligible [14]. In the case of dewatering, indirect emissions from chemical and energy use were included, using parameters sourced from specialized literature (see Table 2). The analysis assumes a dry matter (DM) content of 20% in the sludge cake following dewatering.

During anaerobic digestion, a 45% volatile solids reduction (VSR) was assumed, while the inert material content remains constant. Furthermore, standard methane yield values proposed in the literature were employed (see Table 2). The use of ferric chloride as an additive in the digestion process has been considered. Additionally, it is assumed that the resulting biogas is recovered to generate energy via an internal combustion engine. The design parameters for this stage are detailed in Table 2.

The design methodology proposed in the specialized literature was followed for thermal drying [12]. Table 2 summarizes energy and heat consumption, as well as the volatile matter reduction. It should be noted that heat consumption was calculated based on fuel oil combustion. Regarding the incinerator, a fluidized bed system was adopted for this study. Table 2 shows the heat and energy requirements, alongside the recovery rates for heat and electricity from the flue gases. It should be noted that these gases are treated using cyclones, electrostatic precipitators, and activated carbon [12]. The specific consumption of the required chemicals is shown in Table 2. All design parameters were sourced from technical literature.

Regarding the pyrolysis stage, specific design criteria for temperatures exceeding 600 °C were adopted. Table 2 summarizes the design parameters, which were sourced from the technical literature [12]. Notably, the process includes the recovery of char as activated carbon (AC), the use of pyrolytic oil (tar) as a fuel oil substitute, and the valorization of syngas for electricity generation. Finally, for the treatment trains involving landfill disposal, only direct emissions (primarily methane) were accounted for.

The carbon footprint of each scenario is determined by adding together the individual contributions. These are calculated by multiplying their relative consumption per functional unit by their emission factor. Emission factors (EFs) for the study were sourced from local grid reports and established literature. The values used are summarized in Table 3, which represent the most accurate available data for the region and the evaluated technologies.

While these factors are subject to inherent uncertainties due to temporal variations in energy mixes and operational fluctuations, their representativeness is further addressed through the sensitivity analysis described in Section 2.3. The electricity emission factor assumed in Table 3 (0.550 kg $CO_{2,eq}$/kWh) represents the average carbon intensity of the Canary Islands' electricity grid. The electricity mix in the Canary Islands relies heavily on fossil fuels; according to official sources [29], approximately 80% of total electricity generation comes from non-renewable sources, primarily combined-cycle power plants (42%) and diesel engines (21%). Renewable energy, mainly wind (15%) and solar (4%), completes the mix. This regional average is used as a baseline for the general inventory.

Table 3. Emission factors and data sources for the life cycle inventory.

Contribution	Unit (u)	EF ($kgCO_2$/u)	Source
	Energy consumption		
Electricity	kWh	0.550	[29]
	Chemicals reagents		
NaOCl	kg	0.300	[30]
Citric acid	kg	4.7	[31]
PAM	kg	1.5	[15]
$FeCl_3$	kg	2.71	[15]
NaOCl	kg	1.121	[32]
Lime	kg	0.68	[15]
Carbon active	kg	0.9	[33]
Fuel-oil	MJ	0.074	[34]
	Direct emissions		
K_{rem}	kg COD/kg MS	0.95 [1]	[21]
$EF_{CH4,bio\ process}$	kg COD	0.0075 [2]	[21]
$EF_{N2O,\ bio\ process}$	kg N	0.0016 [3]	[21]
$EF_{CH4,landfill}$	Kg MS	0.0606 [2]	[15]
	Transport		
EF_{diesel}	L	2.7	[11]
	Global Warming Potential		
GWP	Kg N_2O	265	[21]
GWP	Kg CH_4	28	[21]

[1] The unit is kg COD/kg MS. [2] The units for these parameters are $kgCH_4$/u. [3] The unit for this parameter is kgN-NO_2.

To estimate the carbon footprint associated with sludge transport, it was assumed that incineration and pyrolysis facilities are centralized, reflecting the isolated nature of the study area. In these scenarios, sludge is transported immediately after the dewatering stage. The carbon footprint was determined by considering the dry matter (DM) content, specific diesel consumption, and the corresponding EF. These values, extracted from specialized literature, are presented in Tables 2 and 3. The distance and duration of sludge transport routes between wastewater treatment plants and solid waste treatment centers or landfills were determined using the Google Maps platform (v. 26.18.01.905972481). Across the evaluated scenarios, transport distances ranged from 5 to 108 km, with an average distance of approximately 51 km. Notably, fuel consumption calculations account for the empty return trip of the vehicle to ensure a comprehensive assessment.

Regarding energy recovery, only electricity generation from anaerobic digestion was considered, given the specific energy mix of the Canary Islands (see Table 3). A consistent methodology was applied to the recovery of electrical energy from the incineration and pyrolysis stages. Furthermore, the heat recovered from incineration is assumed to directly offset fuel oil consumption, similar to the energy recovery achieved by reusing pyrolytic oil (tar). Finally, concerning the use of char as activated carbon (AC), it is estimated that the avoided carbon footprint is equivalent to the emissions generated during the production of commercial AC.

2.3. Sensitivity Analysis

Some of the parameters used in this study were sourced from specialized literature. However, due to their theoretical and generalized nature, these values may introduce uncertainties that could significantly influence the results.

Therefore, a sensitivity analysis was conducted to evaluate the impact of key operational variables on the overall carbon footprint of the different treatment trains. This analysis focused on three specific groups of parameters: (i) specific energy consumption; (ii) variables associated with the recovered (avoided) footprint; and (iii) direct emissions originating from the sludge treatment line.

Within the first group, the specific energy consumption of all technologies involved in sludge management and treatment was evaluated. The second group focused on variables influencing the recovered footprint, including biogas-to-electricity conversion efficiency, the recovery rates for heat and electricity from incineration, the electrical recovery from syngas, and the heat recovery potential from the combustion of both pyrolytic oil (tar) and activated carbon. Finally, the third group addressed direct emissions and operational inefficiencies, specifically analyzing biogas leakage in the anaerobic digester, the methane emission factor (EF) at the landfill, and the specific heat demand for both thermal drying and incineration processes.

The sensitivity analysis was conducted by systematically varying key operational and design parameters within a ±50% range. This variation interval was selected based on a conservative approach to account for data uncertainty and is consistent with the ranges employed in previous similar studies [11,35,36]. The resulting impact on the carbon footprint was evaluated following the aforementioned procedures to identify the variables that most significantly influence the environmental performance of the treatment systems.

3. Results and Discussion

3.1. Carbon Footprint Analysis for Different Sludge Treatment Processes

The treatment and management of sewage sludge were evaluated for two secondary treatment technologies: CASP and MBR, and six sludge treatment trains. This section analyzes the carbon footprint of these processes to identify the most environmentally viable options and establish a basis for decarbonization strategies within a specific regional context. While Figure 1 illustrates the flow diagram of inputs and outputs within the carbon footprint scope, Table 4 summarizes the specific technologies incorporated into each treatment train.

Table 4. Sludge treatment and management technologies used in each treatment train.

Contribution	Thickening	Anaerobic Digestion	Dewatering	Thermal Drying	Incineration	Pyrolysis	Landfill
Train A	x		x				x
Train B	x	x	x				x
Train C	x		x	x	x		x
Train D	x	x	x	x	x		x
Train E	x		x	x		x	
Train F	x	x	x	x		x	

The design of each treatment train follows a logical sequence of technologies aimed at sludge valorization from a thermochemical perspective. Train A represents the conventional sludge treatment line for small and medium-sized wastewater treatment plants (WWTPs), which are common in isolated regions such as islands [7,11,15]. In this train, the sludge is first thickened by gravity and subsequently dewatered using centrifuges before its final disposal in a landfill. Trains B, D, and F incorporate anaerobic digestion (AD) between the thickening and dewatering stages. AD is widely implemented in medium- and large-scale WWTPs as it enables biogas production and recovery, while also facilitating overall sludge management [14,15,37]. Treatment trains C, D, E, and F require thermal drying after dewatering to increase the dryness from 20% to 70–90%, ensuring the proper operation of subsequent stages [12]. Trains C and D include an incineration stage for energy and heat recovery. Finally, the resulting ash is disposed of in a landfill. This technology offers management advantages by confining contaminants within the ash and significantly reducing the volume of material destined for final disposal [13,15]. The most advanced configurations, Trains E and F, incorporate sludge pyrolysis to produce both energy (from syngas and tar used as fuel oil substitutes) and biochar, which can be further processed into activated carbon. This technology positions sludge as a resource for valorization, as it not only reduces and stabilizes the waste but also transforms it into energy and high-value products. However, its industrial adoption remains limited [11,12].

Figure 2 shows the carbon footprint, expressed in kg $CO_{2,eq}$ per cubic meter of treated wastewater, for each treatment train across the two secondary processes studied. Additionally, it illustrates the relative contributions of the different categories within the LCA scope, as well as the carbon footprint recovery through thermochemical processes. As observed, the global carbon footprint, which accounts for both emissions and recovery credits, decreases as the sludge treatment process increases in complexity, regardless of the secondary treatment employed. However, the direct carbon footprint, excluding recovery through byproducts, is higher in trains E and F compared to those incorporating incineration (C and D), irrespective of the secondary technology used (Figure 2A,B). These results are consistent with previous literature, which emphasizes the necessity of including these recoveries to obtain a comprehensive perspective of the treatment process [11,12]. Furthermore, the robustness of these reductions against data uncertainty is confirmed by the sensitivity analysis (Section 3.2), ensuring that the observed trends remain stable despite potential variations in the emission factors.

As can be seen in Figure 2, the global carbon footprint is higher when MBR is used as a secondary treatment, regardless of the treatment train selected for the sludge line. Specifically, the global carbon footprint is approximately 1.3 times higher for MBR compared to CASP in the simpler trains (Trains A and B). Furthermore, this difference widens as more complex treatment trains are analyzed, reaching 1.8 times when Train F is used. The increase in the carbon footprint attributable to the secondary treatment is due to several factors. The primary factor contributing to the increase in the carbon footprint is the indirect emissions resulting from energy consumption by the WWTP's water line. This value is approximately 2.2 times higher when using MBR compared to CASP. Furthermore, in both technologies, energy consumption plays a significant role, accounting for 28.8 $\pm$ 3% and 48.1 $\pm$ 2% of the total positive contributions to the carbon footprint for CASP and MBR, respectively. These results are consistent with the literature, which indicates that the higher specific energy consumption of MBRs due to membrane aeration for fouling control leads to a larger carbon footprint [38]. This higher energy demand is fundamentally linked to the intensive aeration required not only for biological processes but also for membrane scouring to mitigate fouling [5]. However, a trade-off exists between this energy footprint and the superior effluent quality; the membrane barrier ensures total retention of suspended solids and pathogens, yielding high-quality water suitable for direct reuse in water-scarce regions [6]. Consequently, the selection of MBR implies prioritizing water recovery and a reduced physical footprint over the minimization of energy-related emissions. Furthermore, the consumption of chemicals in the water line for fouling control in the MBR contributes to an increase in the global footprint regardless of the sludge treatment process used (Figure 2B). Although this contribution is smaller, it represents 4.3 ± 0.2% of the total positive emissions.

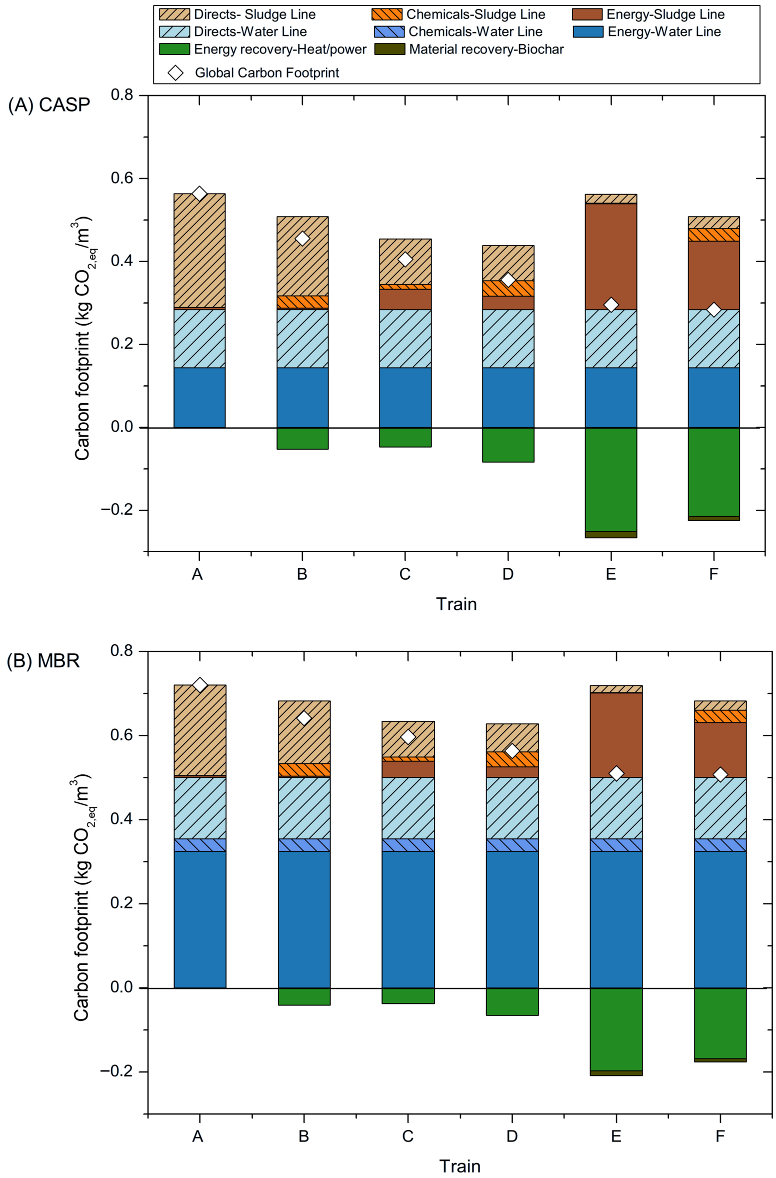

Figure 2. Comparative carbon footprint of wastewater treatment lines ((**A**): CASP, (**B**): MBR) with different sludge management scenarios (Trains A–F).

Finally, the last factor contributing to the increase in the carbon footprint when using an MBR is the typical design SRT for this technology. MBRs typically operate at an SRT of 15 days, compared to 8 days for CASPs [4,22]. This higher SRT results in aged biomass which increases direct emissions from the water line according to the IPCC methodology [21]. This aged biomass exhibits higher endogenous respiration and may also promote N_2O emissions [15]. Although this increase is very minor, representing only 0.007 kg $CO_{2,eq}/m^3$, the SRT has a significant impact on the amount of sludge produced. Conversely, the lower SRT in CASP systems leads to a higher mass of sludge compared to MBRs, which leads to a larger carbon footprint in the sludge line, but also to higher recovery credits. Train A serves as a baseline to evaluate the effect of SRT on sludge management. As shown in Figure 2A,B, when CASP is used, the carbon footprint derived from direct emissions associated with the sludge line reaches 0.274 kg $CO_{2,eq}/m^3$, compared to 0.215 kg $CO_{2,eq}/m^3$ for MBR. This results in a value approximately 1.3 times higher due to the greater sludge production in the CASP. However, the global footprint of the MBR is 0.720 kg $CO_{2,eq}/m^3$, compared to 0.564 kg $CO_{2,eq}/m^3$ when CASP is used. These values fall within the range reported by other authors for similar plants in other regions [7,14,15,39]. It is important to note that despite the increased environmental footprint resulting from sludge management in CASP, the higher energy consumption of MBR systems means that the latter technology does not yield a lower global carbon footprint across all treatment trains assessed (see Figure 2A,B). Clearly, in terms of effluent quality, MBR systems offer numerous benefits, making them a superior technology for wastewater treatment in contexts where high-quality reuse or discharge is required [4,5].

The introduction of anaerobic digestion (AD) in Train B results in a significant reduction in direct emissions from the sludge line compared to Train A (see Figure 2A,B). This reduces the footprint associated with this category by 0.084 and 0.066 kg $CO_{2,eq}/m^3$ for CASP and MBR, respectively. Furthermore, AD allows for the recovery of carbon credits through biogas utilization. Consequently, the global footprint reduction is more pronounced in the CASP, reaching approximately 19% compared to about 11% when MBR is used. This behavior is attributable to the loss of volatile biomass during AD and to biogas production, both of which depend on the amount of feedstock and ultimately reduce the amount of sludge disposed of in the landfill. On the other hand, although its contribution is very small, the introduction of AD leads to an increase in the carbon footprint associated with the chemical category in the sludge line (approximately 0.03 kg $CO_{2,eq}/m^3$). This is due to the dosage of ferric chloride for hydrogen sulfide removal [13]. This behavior is also observed in the other treatment trains when comparing those with and without anaerobic digestion, specifically Trains C/D, and Trains E/F (see Figure 2A,B). However, the introduction of AD combined with other thermochemical processes, such as incineration or pyrolysis, has differing effects.

When AD is combined with incineration, the global carbon footprint is affected, as shown by comparing scenarios C and D (Figure 2). The global carbon footprint is reduced by approximately 0.05 kg $CO_{2,eq}/m^3$ for both CASP and MBR. This limited reduction occurs because, although incorporating AD reduces the amount of sludge subjected to thermal drying and incineration and thus leads to a smaller footprint from energy and heat consumption, the increase in chemical consumption and the slight rise in recovery credits result in a marginal reduction in the global footprint. On the other hand, gross emissions (positive impacts) are very similar for both trains, regardless of the secondary treatment used (see Figure 2A,B). Specifically, these values are 0.453 and 0.439 kg $CO_{2,eq}/m^3$ for Trains C and D when CASP is used, and 0.634 and 0.629 kg $CO_{2,eq}/m^3$ for the same trains when MBR is used.

On the other hand, when pyrolysis is incorporated and the combination with AD is analyzed, specifically when Trains E and F are compared (see Figure 2A,B), a unique behavior emerges. Although the global footprint remains practically constant at around 0.209 ± 0.001 and 0.509 ± 0.002 kg $CO_{2,eq}/m^3$ for CASP and MBR, respectively, the amount of footprint credits recovered is significantly lower when AD is incorporated in Train F. These results are consistent with the literature, since in pyrolysis 100% of the carbon fed into the pyrolysis process is retained in its byproducts, compared to approximately 25% recovered in digestion [11]. In absolute terms, the recovered credits amount to -0.266 and -0.224 kg $CO_{2,eq}/m^3$ for Trains E and F when using a CASP system, and -0.209 and -0.179 kg $CO_{2,eq}/m^3$ for the same trains with an MBR. Regarding the credits recovered through heat and energy, approximately 60% is due to syngas utilization when AD is not used (Train E). However, in the most comprehensive configuration (Train F), AD contributes only about 25% of the credits, thereby limiting the significant capacity of pyrolysis for carbon recovery when AD is implemented [13]. Furthermore, the destruction of volatile matter during AD minimizes the production of pyrolysis byproducts, which also results in a smaller amount of recovered carbon credits (see Figure 2A,B). This phenomenon highlights a clear competition for carbon between the biological and thermal stages. In Train F, the anaerobic microorganisms divert a significant fraction of the organic carbon toward methane production, which is a less efficient pathway for carbon recovery compared to pyrolysis. Consequently, the resulting digestate has a lower carbon content and reduced calorific value, which limits the yield of syngas and biochar during the subsequent thermal stage. This explains why the synergistic combination of AD and pyrolysis (Train F) does not necessarily outperform standalone pyrolysis (Train E) in terms of total carbon credits recovered.

Despite these reductions in recovery, introducing AD offers a specific advantage in terms of the gross carbon footprint (before credits). By processing a smaller amount of sludge, the pyrolysis stage in Train F consumes less energy, resulting in a lower indirect carbon footprint from imported electricity. Nevertheless, this benefit is partially offset in Train F because direct emissions from the sludge line increase due to the inherent risk of methane leaks during AD, which have been accounted for in this LCA (see Figure 2A,B).

Compared to the baseline study (Train A), the introduction of thermochemical processes significantly reduces the global footprint. When pyrolysis is used (Trains E and F), this reduction reaches approximately 48.52 $\pm$ 0.01% and 29.34 $\pm$ 0.01% for CASP and MBR, respectively. In the case of incineration (Trains C and D), the reduction is also substantial, with values of 32.49 $\pm$ 0.06% for CASP and 19.42 $\pm$ 0.03% for MBR. Furthermore, although CO_2 is not accounted for because it is biogenic, pyrolysis sequesters carbon in the char, whereas incineration releases it into the atmosphere as CO_2 [11]. The results of this study align with previous research establishing pyrolysis as a key decarbonization strategy [12]. Regarding the operational conditions and practical implementation of pyrolysis, the results in this study are based on a thermal range exceeding 600 °C. This temperature was specifically selected to maximize the production of pyrolysis gas, as literature indicates that at these levels, approximately 51% of the volatile solids are converted into gas compared to only 33% at lower temperatures [12]. Although energy consumption is higher at these elevated temperatures, it is balanced by the increased energy recovery from both gas and tar. In terms of real-world feasibility, while these emerging thermal technologies involve higher technical complexity, they have already been validated at full scale and show significant market potential for sustainable sludge management. Clearly, for isolated areas, the implementation of centralized pyrolysis or incineration systems is necessary to ensure the project is economically viable. However, the results of this study highlight that the introduction of AD for small-to-medium-sized plants, combined with a centralized pyrolysis unit, is a sound decarbonization strategy. Nevertheless, the practical deployment of such integrated systems must be balanced against technical and economic limitations. These include the high initial capital investment required for thermochemical facilities and the need for specialized technical personnel to manage complex energy recovery loops. Furthermore, further actions are required to reduce the impact resulting from the high energy demand of these processes, ensuring that the carbon benefits are not offset by excessive operational costs.

3.2. Sensibility Analysis

Some of the parameters used in the LCA are based on literature or typical values that may differ from those encountered in real-world scenarios. To address this, a sensitivity analysis was conducted by modifying various design and operational parameters of the sludge line by ±50% to assess their impact on the global environmental footprint. The studied variables were classified into three groups: (1) parameters associated with direct emissions from the sludge line; (2) emissions related to specific energy consumption (SEC); and (3) parameters related to footprint recovery credits. For the sensitivity analysis, Scenarios B, D, and F were selected for both secondary treatments, as they represent the highest level of complexity across the three stages of sludge recovery: anaerobic digestion, incineration and pyrolysis. Furthermore, the results obtained from these scenarios can be readily extrapolated to the remaining treatment trains.

Figure 3 shows the sensitivity analysis of the four parameters influencing direct emissions from the sludge line: the emission factor for sludge deposited in landfills ($EF_{landfill}$), the percentage of methane leakage from biogas, and the direct emissions from heat input during both thermal drying and incineration. For the latter two, the use of fuel oil as the energy source was assumed. As shown in Figure 3, across all studied parameters, CASP technology exhibits greater sensitivity regarding the global carbon footprint. This behavior is due to the higher sludge production associated with this system compared to the MBR. Furthermore, in MBR systems, the high energy consumption of the water line buffers the relative impact of changes in direct emissions from the sludge line.

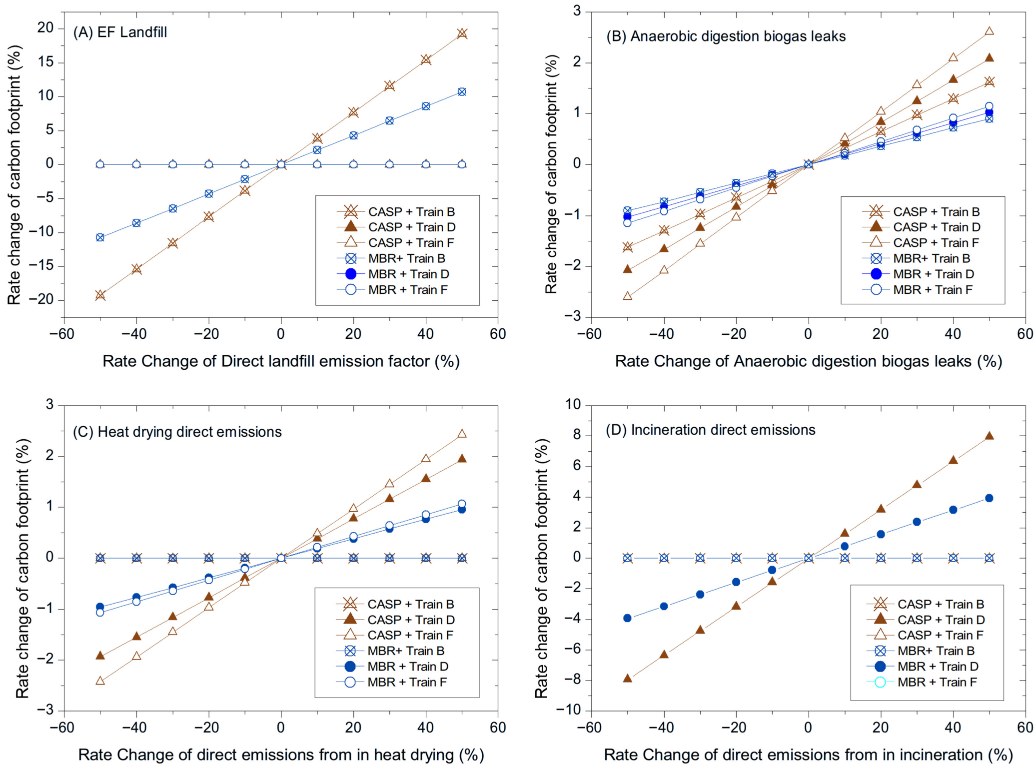

Figure 3. Comparative sensitivity analysis of direct emission factors and gas leakages on the overall process footprint using CASP and MBR for treatment trains B, D and F. (**A**) Emission factor relative to landfill, (**B**) methane leaks in AD, (**C**) heat drying direct emissions and (**D**) incineration direct emissions.

The percentage of biogas leakage from the anaerobic digester and the direct emissions associated with thermal drying have a low impact on the overall carbon footprint. These parameters account for approximately ±3% of the total footprint despite variations of 50% in the analyzed factors (Figure 3B,C). Such results stem from the low relative weight of direct emissions from potential biogas leaks and fuel combustion for thermal drying, as in all scenarios, each of these factors represents an individual contribution of less than

0.015 kg $CO_{2,eq}/m^3$. In contrast, the global carbon footprint exhibits higher sensitivity to direct emissions from fuel combustion in the incinerator. However, as shown in Figure 3D, even with a variation of 50% in this parameter, the footprint increase remains below 10% in the worst-case scenario, which corresponds to the CASP + Train D configuration.

Finally, the methane emission factor for sludge disposal in landfills exerts a significant impact on the overall carbon footprint (see Figure 3A). For the CASP + Train B configuration, a 50% variation in this factor leads to a change of approximately 20% in the global footprint. However, when analyzed in absolute terms, this sensitivity corresponds to a variation of approximately 0.09 kg $CO_{2,eq}/m^3$. The baseline emission factor for sludge disposal in landfills was selected based on specialist literature [15], considering the typical anaerobic degradation of organic matter in such environments. Given the high sensitivity of the global footprint to this parameter, the use of a conservative yet representative value is essential to avoid underestimating the environmental impact of non-thermal sludge management routes.

Figure 4 presents the sensitivity analysis results for all parameters related to SEC, covering the six processes involved in the sludge treatment lines studied. Similar to the direct emissions analysis, the configurations combined with CASP exhibit greater sensitivity across all parameters.

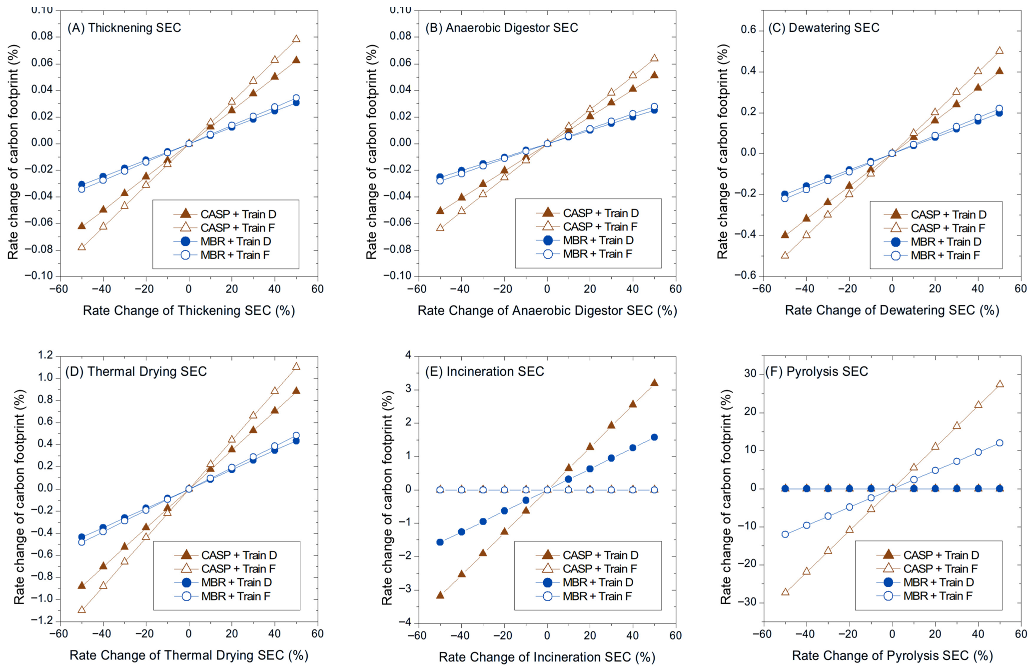

Figure 4. Effect of the specific energy consumption (SEC) of the various sludge management technologies on the overall process footprint using CASP and MBR for treatment trains D and F. (**A**) Thickening SEC, (**B**) Anaerobic Digestion SEC, (**C**) dewatering SEC, (**D**) Thermal drying SEC, (**E**) Incineration SEC, (**F**) Pyrolysis SEC.

As shown in Figure 4A–D, the global carbon footprint displays low sensitivity to the SEC of the thickening, anaerobic digestion, dewatering, and thermal drying processes. This behavior stems from the relatively small contribution of these individual energy requirements compared to the indirect emissions associated with the water line's energy demand or the direct emissions derived from sludge management. In contrast, the incineration and pyrolysis stages exert a more pronounced effect on the overall carbon footprint. Regarding incineration, the global footprint exhibits low-to-moderate sensitivity; despite a 50% variation in the SEC associated with this process, the total carbon footprint fluctuates by less than 4% for the CASP + Train D scenario (see Figure 4E). In contrast, the SEC for pyrolysis shows high sensitivity regarding the overall carbon footprint (see Figure 4F). This sensitivity is further amplified when processing larger amounts of sludges, specifically when using CASP technology. In this case, a ±50% variation in the pyrolysis SEC leads to a change of approximately ±30% in the global footprint. This result is consistent with literature, as it is widely accepted that pyrolysis with low efficiency and high SEC is neither energetically nor environmentally favorable [12,13]. Furthermore, ensuring the proper performance of the pyrolysis system requires an efficient thermal drying stage [11]. This high sensitivity reflects that pyrolysis is a thermally intensive process where energy efficiency is the primary driver of its carbon footprint. Unlike established biological treatments, the environmental viability of pyrolysis depends strictly on maintaining a low SEC through advanced reactor designs and optimal heat recovery. In this context, the environmental performance of the system relies on the strict control of the sludge dewatering and thermal drying stages. Minimizing the moisture content of the feedstock before it enters the reactor is fundamental to reducing the SEC and ensuring the stability of the carbon footprint. However, despite the high relative sensitivity described above, in absolute terms, a ±50% change in the pyrolysis SEC translates to approximately ±0.08 and ±0.06 kg $CO_{2,eq}/m^3$ for the CASP and MBR configurations (Train F), respectively.

Finally, Figure 5 presents the sensitivity analysis conducted to evaluate the influence of parameters related to carbon footprint offset credits. This analysis assessed the performance of the energy recovery systems for biogas, electricity, and heat from the incineration unit, as well as the recovery yields for tar, syngas, and biochar. As in previous cases, all treatment lines using CASP technology show a higher sensitivity to biomass production, which directly influences the carbon footprint. It is important to note that these parameters are inversely proportional; specifically, a reduction in recovery efficiency leads to an increase in the overall carbon footprint.

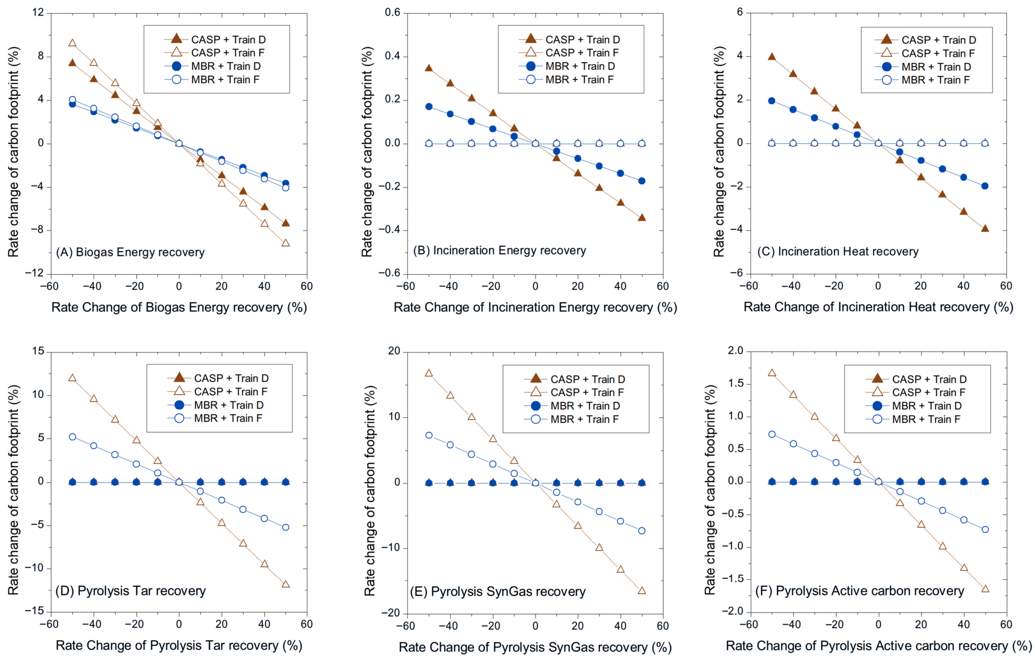

Figure 5. Effect of energy and byproduct recovery efficiency from sludge management on the overall process carbon footprint offset for CASP and MBR treatment trains. (**A**) Biogas energy recovery, (**B**) Incineration energy recovery, (**C**) Incineration heat recovery, (**D**) Pyrolysis tar recovery, (**E**) Pyrolysis syngas recovery and (**F**) Pyrolysis active carbon recovery.

The results indicate that the footprint has very low sensitivity to electricity recovery efficiency in the incinerator (Figure 5B) and to the recovery of biochar as activated carbon

(Figure 5F). This is consistent with their minor contribution to the total footprint previously observed in Figure 2A,B. On the other hand, energy production from biogas and heat recovery during incineration exhibit low-to-moderate impacts (Figure 5A,C). However, in all scenarios, a ±50% variation in process efficiency results in a footprint fluctuation of less than ±10%.

Finally, as with the other groups of parameters analyzed, the pyrolysis stage exerts the greatest impact on the global carbon footprint. As shown in Figure 5D,E, a variation of ±50% in the energy recovery yields from tar and syngas results in footprint changes of approximately ±12% and ±16%, respectively, for the CASP + Train F configuration. It is crucial to highlight the technical synergy between these parameters and the SEC (Figure 4F). An increase in SEC is inherently associated with a lower efficiency of energy recovery via syngas and tar, as more energy is required per unit of energy recovered in these products, thereby worsening the overall carbon footprint [11–13]. These results underscore that while pyrolysis is an environmentally viable solution for sludge recovery, its performance must be strictly monitored to maintain the SEC within acceptable ranges and ensure adequate energy recovery.

In summary, the sensitivity analysis underscores the robustness of the LCA methodology and identifies the MBR as a technology whose global carbon footprint is less sensitive to the design and operational parameters of the sludge treatment line precisely due to its dominant energy demand in the water line. Furthermore, direct emissions from landfilling and the energy efficiency of the pyrolysis stage comprising both SEC and recovery yields are identified as the most critical parameters influencing the overall carbon footprint, necessitating rigorous monitoring and optimization in these management routes.

3.3. Analysis of the Island of Tenerife and Fuerteventura: Decarbonization Strategies

The preceding sections analyzed the individual carbon footprint of different combinations of WWTPs and sludge management strategies, identifying the centralized implementation of thermochemical processes such as pyrolysis or incineration as a potential pathway to decarbonization. However, that analysis did not account for the impact of sludge transportation or the specific local water treatment infrastructure. In this regard, this study proposes a comparative analysis of two isolated regions: Fuerteventura and Tenerife. Both islands share common characteristics such as water stress and high tourism levels. However, they offer the opportunity to analyze key parameters including different energy mixes, water reclaiming structures, and population densities. The energy mix data were obtained from records provided by the public entities responsible for electricity distribution in Spain [29]. Furthermore, the volumes treated at each facility were taken from official sources regarding water distribution and treatment on each island, alongside population figures that account for both permanent residents and the equivalent population derived from tourism [40–42]. The distance and duration of sludge transport routes between wastewater treatment plants and solid waste treatment centers or landfills were determined using the Google Maps platform.

According to official sources, Tenerife has a population equivalent significantly higher than that of Fuerteventura. This difference is reflected in the treatment capacities of the WWTPs, where Tenerife shows a much higher installed capacity. Table 5 shows the annual treated volumes for each island, broken down by secondary treatment and sludge management processes. As can be seen, Fuerteventura has primarily installed CASP systems with conventional sludge treatment and includes only one facility integrating an anaerobic digestion system for sludge reuse (Train B). In contrast, Tenerife shows a higher integration of MBR technology, which represents nearly 43% of its total treated volume. Furthermore, Tenerife has several WWTPs equipped with AD, thus enabling more extensive sludge reuse across the island.

Table 5. Energy mix and wastewater treatment infrastructure and sludge management in Tenerife and Fuerteventura.

Contribution	Tenerife	Fuerteventura
Electricity Generation Mix (%)		
Renewable energies	19	9
Diesel Generator	5	89
Gas Turbine	4	2
Steam Turbine	17	--
Combined Cycle	55	--
Electricity EF ($kgCO_2$, eq/m^3)	0.563	0.628
Wastewater treatment plant capacity (hm^3/year)		
CASP + Train A	4.48	4.72
CASP + Train B	9.04	--
MBR + Train A	4.62	1.08
MBR + Train B	8.85	0.09

In terms of electricity generation, the two islands differ substantially. Table 5 provides a summary of the energy mix where Tenerife shows a more complex generation structure with renewable energy accounting for 19% of its total generation. Regarding non-renewable sources, Tenerife meets its demand primarily through combined-cycle plants (55%) and steam turbines (17%) with the remainder coming from other technologies. In Fuerteventura, the penetration of renewable energy is lower at 9% and energy production is primarily driven by diesel engines. It is important to note that in the Canary Islands, gas turbines do not consume natural gas due to a lack of supply infrastructure [43]. The resulting energy mix shows emission factors for each island that are higher than the Canary Islands average reported in Table 2, with Fuerteventura's factor being 1.11 times higher than Tenerife's as shown in Table 5. This result is logical because certain islands such as El Hierro have high renewable energy penetration that lowers the archipelago's average [43]. Furthermore, the heavy reliance on diesel engines in Fuerteventura significantly increases the environmental impact of its energy production compared to the other islands.

Once the framework for comparative study was established, the next step was to determine the specific carbon footprint per capita per year for each island. Figure 6A shows the carbon footprint from direct and indirect emissions as well as the carbon footprint offset for each island, detailing the partial contributions of each category defined within the scope of the LCA.

As can be seen in Figure 6A, the environmental impact of wastewater treatment and sludge management is significantly different between Fuerteventura and Tenerife. In terms of the global carbon footprint, Tenerife has a footprint of approximately 13.2 kg $CO_{2,eq}$/capita·year, compared to approximately 23.0 kg $CO_{2,eq}$/capita·year in Fuerteventura. This result was expected due to the higher emission factor associated with energy consumption in Fuerteventura. In fact, the increase in the Emission Factor (EF) related to energy consumption means that, despite having a higher percentage of low-energy-demand WWTPs such as CASP, the indirect carbon footprint associated with the water treatment line in Fuerteventura is significantly higher than that of Tenerife, which treats a larger volume using MBR.

The same pattern is observed with the footprint associated with energy consumption in the sludge treatment line. Although the energy demand is very similar on both islands, Fuerteventura's footprint in this category is 2.3 times higher, even though its contribution to the overall footprint is relatively low at around 1%. These results show that the overall footprint is highly influenced by the energy consumption of the water line, with contributions of 40% and 32% for Tenerife and Fuerteventura, respectively.

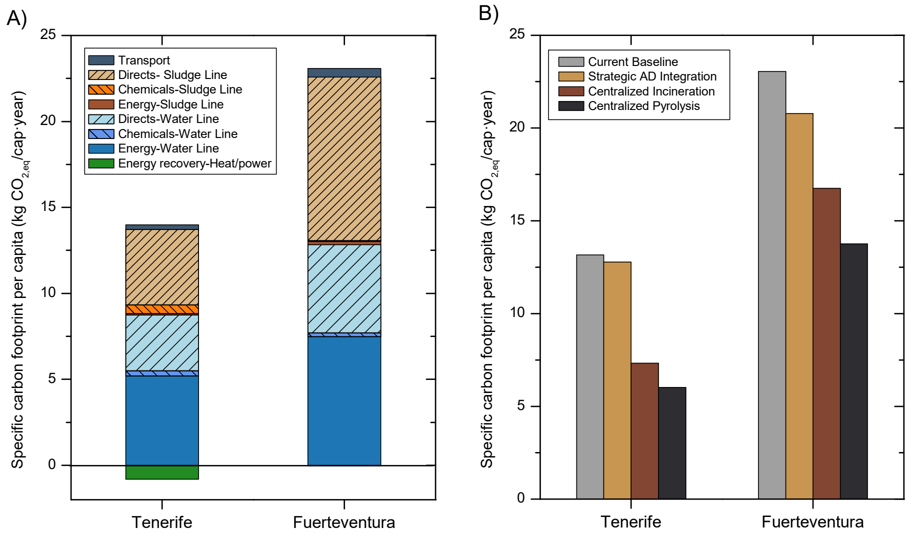

Figure 6. Comparative analysis of the specific per capita carbon footprint in Tenerife and Fuerteventura. (**A**) Current emissions profile by treatment process. (**B**) Net carbon footprint across decarbonization scenarios: Strategic integration of Anaerobic Digestion (WWTP > 2500 m^3/d) and centralized thermal systems (incineration and pyrolysis).

Regarding the logistical aspects, both islands rely on a single solid waste treatment facility or landfill. Despite the geographical dispersion of the plants, the environmental impact attributed to sludge transport is low compared to other categories, reaching only 2% for both islands as shown in Figure 6A. This finding justifies the decision in previous sections to exclude transportation from theoretical analysis, as its overall contribution to the carbon footprint is marginal even in fragmented island territories.

As for indirect emissions resulting from the use of chemicals in both the water and sludge treatment lines, Figure 6A shows that their impact is negligible when compared to the other categories. Finally, direct emissions from the water line are similar between the two islands. However, direct emissions from the sludge line in Fuerteventura are very significant, accounting for the largest contribution at 41% of the total footprint (see Figure 6A). This result is consistent with previous findings because CASP technology involves greater sludge production, which in Fuerteventura is deposited almost entirely in a landfill. In contrast, Tenerife treats a larger volume using MBR, which reduces sludge production. Furthermore, 74% of the installed capacity in Tenerife features a treatment train equipped with AD (Train B), which reduces the amount of sludge sent to landfills and enables energy generation through biogas. In this regard, according to current official data, this configuration translates to footprint credits of -0.257 $kgCO_{2,eq}$/capita·year. In this context, the adoption of low-energy treatment technologies such as gravity-driven membrane bioreactors (GDM) could represent an additional decarbonization pathway, particularly for small and medium-sized plants in isolated regions [8,44].

The results point to two clear approaches for developing decarbonization strategies: sludge management and renewable energies. Although MBR technology reduces the amount of sludge produced, the results described in previous sections indicate that the energy consumption of the water treatment line will not offset its associated carbon footprint. Consequently, to mitigate the overall carbon impact, three sludge design strategies are proposed as primary drivers for decarbonization. The first strategy proposes the integration of

AD in all plants with a capacity exceeding 2500 m^3/d. This threshold capacity was defined considering that AD would not be viable for small-capacity plants. The second strategy combines the first with a centralized incineration unit located near the landfill. Similarly, the third scenario incorporates the first but includes a centralized pyrolysis unit instead. Figure 6B shows the overall carbon footprint per capita, taking into account emissions and carbon footprint offsets from energy recovery in the form of electricity and heat as well as byproducts such as activated carbon from char.

In the case of Tenerife, the AD implementation scenario only calls for the installation of two additional digesters, which would allow 91% of the installed capacity at the WWTPs to utilize the sludge produced on-site. Despite the substantial nature of this figure, this strategy only reduces the overall footprint by 3% as shown in Figure 6B. This reduction is more pronounced in Fuerteventura, where the baseline scenario is less favorable due to the current lack of biogas utilization across nearly all plants. Nevertheless, because Fuerteventura relies on a higher number of small-scale treatment plants, the integration of AD is only feasible for 61% of the installed capacity. Even with this technical limitation, a significant reduction of 9.8% in the global footprint is achieved in percentage terms. These results are consistent with previous studies that highlight the necessity of recovering biogas at the treatment plants to reduce emissions from energy consumption [37]. Furthermore, in remote regions, this solution has been proposed as a strategy to mitigate emissions associated with transportation and final disposal in landfills [7]. While this strategy reduces emissions and is likely the most straightforward to implement, significant room for improvement remains. In this regard, installing an incineration or pyrolysis plant could be an alternative that further enhances the environmental performance of the process.

The integration of a centralized thermochemical processing plant appears to be the option that minimizes the carbon footprint as shown in Figure 6B. The integration of incineration reduces the carbon footprint relative to the baseline scenario by approximately 44% for Tenerife and 27% for Fuerteventura. In this case, direct emissions from the sludge line are reduced and carbon offset credits are increased. When pyrolysis is integrated, as expected based on previous results, the carbon footprint reduction is maximized, reaching 54% and 40% in Tenerife and Fuerteventura, respectively. This improvement is due to the utilization of syngas, tar, and char from the pyrolysis process. Thus, the optimal strategy would be to integrate a centralized pyrolysis process for sludge utilization. This approach would reduce pressure on the islands' landfills while simultaneously generating energy and high-value byproducts from waste.

An analysis of the data shown in Figure 6B reveals that the final step toward the complete decarbonization of the process requires a significant increase in the use of renewable energy. The percentage reductions in the carbon footprint as well as their absolute values are much higher in Fuerteventura than in Tenerife. In absolute terms, the best-case scenario featuring the integration of a pyrolysis unit leads to a footprint of 6.0 and 13.8 kg $CO_{2,eq}$/capita·year for Tenerife and Fuerteventura, respectively. As can be seen, the optimized value for Fuerteventura is very similar to the current baseline scenario in Tenerife. This result stems from the fact that while sludge-related strategies can reduce direct emissions from processing or energy production, their impact remains constrained by the overarching energy mix. If the indirect carbon footprint associated with energy consumption continues to depend on a mix heavily reliant on fossil fuels, these strategies can only offer a limited overall improvement. Consequently, these findings highlight the critical necessity of increasing renewable energy penetration across the islands to achieve a deeper and more effective decarbonization of the water and sludge treatment lines, as the regional energy mix remains a key limiting factor that constrains the environmental benefits of advanced sludge management technologies.

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4. Conclusions

The manuscript analyzes the carbon footprint associated with wastewater treatment in isolated areas such as volcanic islands. The study focuses on two secondary treatment processes (CASP and MBR) across six sludge management treatment lines, ranging from conventional systems involving gravity thickening, dewatering, and landfill disposal to thermochemical treatments including pyrolysis, anaerobic digestion, and incineration.

The overall carbon footprint is consistently higher when MBR is used as the secondary treatment, regardless of the sludge treatment line selected. This is primarily due to indirect emissions from the energy consumption of the water treatment line, which is approximately 2.2 times higher in the MBR system compared to the CASP system. Furthermore, this energy consumption represents a dominant factor in the total positive contributions to the carbon footprint, accounting for $28.8 \pm 3\%$ in CASP systems and $48.1 \pm 2\%$ in MBR systems. At the same time, due to the typical design solids retention time (SRT), CASP produces a greater mass of sludge than MBR. Although this generates a larger footprint within the sludge line itself, it also allows for greater recovery credits in subsequent treatment stages. However, this increase in the footprint associated with the sludge line in conventional systems does not exceed the energy related footprint of MBRs, resulting in significantly higher carbon footprints if this secondary treatment is selected.

The integration of thermal processes significantly reduces the global footprint compared to conventional landfill disposal. While incineration (Train C) lowers direct emissions and generates energy credits, the subsequent incorporation of Anaerobic Digestion (Train D) offers marginal additional benefits, as the reduction in sludge volume is largely offset by increased chemical consumption. Pyrolysis (Trains E and F) emerges as the most effective decarbonization pathway. However, the synergy between AD and pyrolysis (Train F) does not yield significant footprint reductions compared to standalone pyrolysis, due to the competition for organic carbon between the biological and thermal stages. The destruction of volatile matter during AD minimizes the production of pyrolysis byproducts (syngas, tar, and char), which reduces the amount of carbon credits recovered compared to direct pyrolysis. The sensitivity analysis underscores the robustness of the LCA methodology, identifying the energy efficiency of the pyrolysis stage and direct landfill emissions as the most critical parameters. Furthermore, it confirms that MBR systems are less sensitive to sludge line variations due to their dominant water line energy demand.

Finally, the study presents a case study on two Spanish volcanic islands (Tenerife and Fuerteventura). The regional assessment reveals a significantly higher carbon footprint in Fuerteventura (23.0 $kgCO_{2,eq}$/capita·year) compared to Tenerife (13.2 $kgCO_{2,eq}$/capita·year), mainly due to the higher emission factor (EF) associated with Fuerteventura's fossil fuel-dependent energy mix. Under the conditions analyzed in this study, centralized thermochemical processing is presented as a highly effective decarbonization strategy. The results indicate that incineration reduces the footprint by 44% in Tenerife and 27% in Fuerteventura, while pyrolysis shows the potential to further extend these reductions to 54% and 40%, respectively. However, even with the reductions achieved through pyrolysis, Fuerteventura's optimized footprint (13.8 $kgCO_{2,eq}$/capita·year) only reaches Tenerife's current baseline level. This highlights that, although sludge management is a significant factor, achieving deep decarbonization in isolated regions is strictly conditioned by the need to increase the penetration of renewable energy.

The findings of this study contribute to the advancement of sustainable development in isolated regions by providing a quantitative framework for decarbonizing the sludge line of wastewater treatment plants. By identifying centralized thermochemical processing as the most effective decarbonization pathway, this work supports the transition of wastewater treatment plants from energy consumers to resource recovery centers, aligned

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with circular economy principles. Furthermore, the results underscore the critical role of renewable energy penetration in maximizing the environmental benefits of advanced sludge management strategies, highlighting the interconnected nature of the water-energy nexus in vulnerable territories. While the results are applied to a specific region, they are easily extrapolated to other isolated regions worldwide.

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