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ORIGINAL ARTICLE

Life cycle assessment of management alternatives for sludge from sewage treatment plants in Chile: does advanced anaerobic digestion improve environmental performance compared to current practices?

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Abstract
Sludge generation is currently one of the most important issues for sewage treatment plants in Chile. In this work, the life cycle environmental impacts of four sludge management scenarios were studied, focusing on the comparison of current practices and advanced anaerobic digestion (AD) using a sequential pre-treatment (PT). The results show that AD scenarios presented lower potential impacts than lime stabilization scenarios in all assessed categories, including climate change, abiotic depletion, acidification, and eutrophication in terrestrial, marine and freshwater ecosystems. The overall environmental performance of advanced digestion was similar to conventional digestion, with the main difference being a decrease in the climate change potential and an increase in the abiotic depletion potential. Acidification and eutrophication categories showed similar performances in both conventional and advanced AD. The effect of PT in the AD scenarios was related to energy recovery, sludge transport requirements and nutrient loads in the sludge and supernatant after digestate dewatering. Considering the results, PT could be a useful strategy to promote sludge valorization and decrease the environmental burdens of sludge management in Chile compared to the current scenario.

Keywords Life cycle assessment · Sludge management · Pre-treatment · Anaerobic digestion

Introduction

During the last decades, sludge management has become a growing issue for wastewater treatment facilities [1]. In Chile, 60% of sewage treatment facilities utilize activated sludge technology [2], whose sludge generation rates surpass those of technologies such as sequential batch reactors and extended aeration systems [3]. Historically, the most utilized sludge disposal strategy in Chile is landfilling, with 55% of the total sludge generated in sewage treatment facilities being disposed either in municipal or industrial waste landfills [4]. Comparatively, beneficial land application only represents 9% of total disposed sludge [4] and, therefore, there is a rising concern about the sustainability of this strategy in the long term.

Considering stabilization alternatives, anaerobic digestion (AD) represents a preferred alternative due to the production of biogas and digestate that can be used as replacements for conventional energy sources and fertilizers, respectively [5]. However, digestion is limited at the hydrolysis step, and

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consequently, long sludge retention times are necessary, and only partial conversion of organic matter is achieved [6].

The implementation of sludge pre-treatment (PT) prior to digestion has been proposed as a method to improve the efficiency of hydrolysis [7, 8]. However, while a wide array of processes have been proposed [9], energy consumption is a barrier to the full-scale implementation of most technologies [10]. In this scenario, ultrasound and thermal hydrolysis at 50–70 °C have been proposed as low-energy consumption alternatives for sludge PT [8, 11]. Both processes have been reported to have significant effects on digestion, as their sequential application can result in increases of up to 50% in the methane yield of sludge [7, 12].

While the main objective of PT is increasing biogas production, factors such as energy, raw material consumption and digestate quality could also be affected [6, 9]. Therefore, the related changes in emissions and other environmental aspects could result in increases in the impacts associated with other steps of sludge management. In this scenario, a suitable tool to estimate these possible trade-offs is Life Cycle Assessment (LCA) [13, 14], a methodology that allows the estimation of the environmental impacts of products, processes and services, potentially including all steps from raw material extraction to waste disposal [15].

Although sludge management and AD have been extensively analyzed using the LCA method [14, 16, 17], to the best of our knowledge, the literature associated with the assessment of PT is limited. In one of the most extensive studies, Carballa et al. [6] concluded that mechanical and chemical PT showed better overall environmental performance than other assessed technologies. Mills et al. [18] reported that thermal hydrolysis improved the environmental and economic performance of sludge digestion, while Gianico et al. [19] reported that the energetic cost of implementing primary sludge wet oxidation and secondary sludge thermal PT surpassed the benefits of combined heat and power (CHP) generation from biogas, even though the environmental performance of the upgraded facility was similar to that of the conventional plant.

The objective of this study is to compare the life-cycle environmental performance of advanced digestion (i.e., including PT) with current management practices used in Chile. The purpose is to provide insight into the potential consequences or benefits in related environmental impacts associated with PT implementation in the national context, evaluating a potentially feasible technology consisting of the sequential application of ultrasound and thermal hydrolysis at 55 °C [12], which has not previously been assessed using LCA.

Materials and methods

Goal, scope and life cycle inventory (LCI) data sources

The aim of this work is to assess the environmental impacts of sludge management scenarios, focusing on the comparison between advanced AD and common practices in Chile, including chemical (i.e., alkalinization) and biological (i.e., conventional AD) stabilization. Four scenarios were established, where 0a and 0b represent the most common and, therefore, the "business-as-usual" scenarios:

- *Scenario 1* (S_1) AD and agricultural sludge application.
- *Scenario 2* (S_2) AD including sequential ultrasound and thermal PT and agricultural sludge application.
- *Scenario 0a* (S_{0a}) Chemical stabilization using lime and landfilling without electricity recovery from landfill gas.
- *Scenario 0b* (S_{0b}) Chemical stabilization using lime and landfilling with electricity recovery from landfill gas.

The functional unit (FU) for waste treatment systems can be defined in terms of the products obtained (energy, materials) [20] or treated waste [17, 21, 22]. As sludge valorization through energy or material recovery is not common across all assessed scenarios, a treatment perspective was selected, and the stabilization of 1 ton of mixed sludge (dry basis) was chosen as the FU [16–18]. All selected scenarios were divided into sub-systems, with the aim of identifying the parameters that influence environmental performance, allowing the proposal of improvement alternatives when possible (Fig. 1).

Data for the Life Cycle Inventory (LCI) of AD were obtained from the Concepción wastewater treatment plant (WWTP) in central–southern Chile [12] for a period of 1 year; these data were complemented with laboratory and literature data when required. The influence of PT on digestion performance was assessed in laboratory during a period of ~ 90 days using two semi-continuous 10-L digesters, one fed with raw sludge and the other with pretreated sludge. Further details of the laboratory assays can be found in the Supplementary Information. Scenarios 0a and 0b were mainly assessed based on LCI models [23]. Infrastructure was not included in the assessment, as previous reports state its low contribution to environmental loads, especially in large-scale facilities [24, 25]. Inputs during the operational stage (i.e., polymers, material for construction of landfill ditches) were included. Detailed information pertaining to the parameters used to calculate the LCI is provided in the Supplementary Information (Tables SI1 and SI2).

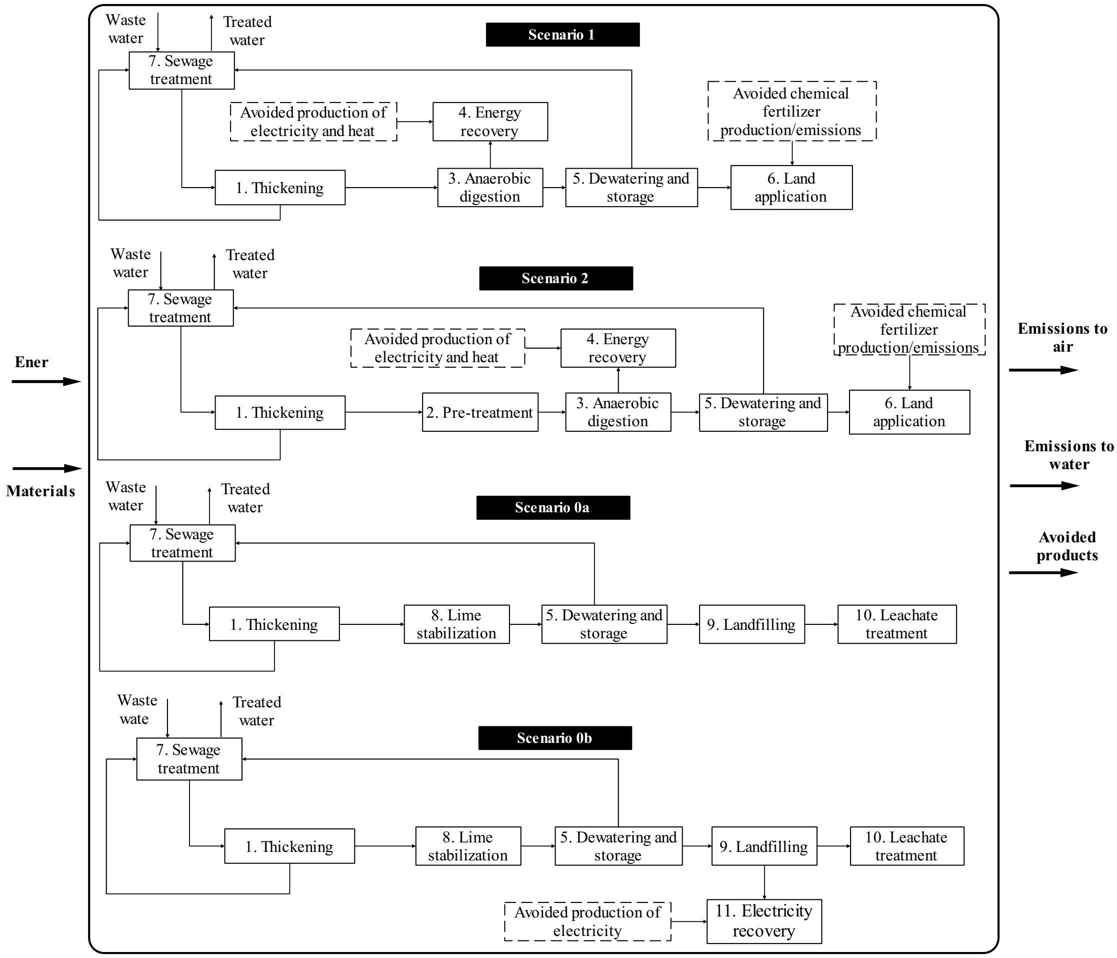

Fig. 1 Process sub-systems and boundaries. Dashed lines represent avoided products (energy or fertilizers)

Biological stabilization sub-systems (scenarios 1 and 2)

The sub-systems in S_1 and S_2 include the in-plant stabilization operations using AD and the transport and application of sludge to soil as replacements of commercial fertilizers. PT prior to digestion was included in S_2.

Sub-system 1: thickening

Prior to AD, sludge from wastewater treatment is thickened to reduce its volume. An efficiency of 90% in terms of solids recovery was assumed. Chemical consumption (polyacrylamide) was estimated based on the WWTP operation, and its production was calculated based on the manufacture of acrylonitrile, one of the raw materials used in acrylamide production [16]. The supernatant originating from thickeners was assumed to be re-circulated to sewage treatment.

Sub-system 2: pre-treatment

In S_2, thickened sludge was subjected to the sequential application of ultrasound and thermal treatment at 55 °C. The PT was tested in laboratory using a Hielscher UP200ST device with a specific energy corresponding to the disruption threshold for sludge under laboratory conditions (2000 kJ/kg total solids; TS) [26]. As industrial-scale devices have lower energy consumption and higher disruption efficiencies than laboratory devices [27], the specific energy consumption of ultrasound was estimated based on information provided by Ultrawaves GmbH (Hamburg, Germany). Estimation was based on the industrial-device power, treatment flow and the average solid concentration of sludge coming from the assessed WWTP, which gives a value of 274 kJ/kgTS, in agreement with previous reports (see Table SI1). Thermal treatment of sludge after ultrasound was performed in a Gerdhardt Thermoshake incubator with a retention time of 8 h. Heat consumption was estimated as the sum of the heat required to increase the sludge temperature and the losses during the process, estimated assuming the use of a cylindrical reactor with heat transfer coefficients corresponding to those of an anaerobic digester (Table SI1) [28]. Stirring electricity consumption was estimated based on specific needs per reactor volume, according to sludge retention times. Heat recovery from the output of the PT to the incoming sludge was considered through the use of a sludge-to-sludge heat exchanger [29].

Sub-system 3: anaerobic digestion

Thermal energy needs for AD were estimated based on the same criteria used for sub-system 2. Biogas production was estimated based on data from the WWTP and modified in S_2 according to the increase observed in laboratory due to PT. CH_4 and CO_2 concentrations in the biogas were measured in laboratory, while N_2, H_2S and NH_3 concentrations were obtained from the literature [30]. A 5% loss of total biogas produced was assumed [31], of which 50% was assumed to be emitted directly and the other 50% to exit in the digestate and be emitted during its storage [32].

Sub-system 4: electricity and heat recovery

Biogas produced during digestion is used in a CHP system. Energy generation was calculated based on the calorific value of CH_4 [33]. Emissions of NOx, CO_2, CO, CH_4 and N_2O to air due to biogas combustion were estimated based on emission factors [34, 35]. Electricity generated was assumed to replace electricity supplied by the Interconnected Central Grid of Chile, while heat was assumed to be utilized in its totality to replace heat generated from natural gas. The influence of this last supposition on the results was assessed through a sensitivity analysis, detailed in Sect. "Life Cycle Impact Assessment (LCIA)".

Sub-system 5: dewatering and storage

Dewatering was assumed to be performed in a centrifuge using polyacrylamide. The mass balance in S_1 was estimated based on WWTP data, while the effect of PT on water recovery was evaluated in laboratory and included in the mass balance for S_2. Dewatered sludge and supernatant were characterized, and the latter was assumed to be re-circulated to sewage treatment. After dewatering, sludge is stored in a roofed field open to the atmosphere, where a 5% loss of water due to evaporation was assumed based on information provided by WWTP operators. Emissions during this step correspond to those detailed under Sect. "Sub-system 3: anaerobic digestion".

Sub-system 6: land application

After storage, sludge is transported in 16–32-metric ton trucks over a distance of 100 km to the application site, modeled using the Ecoinvent v3 process for transport in freight lorries of that capacity (EURO3, Global, Market).Based on Rodríguez-García et al. [22], it was assumed that sludge replaces the use of diammonium phosphate ($(NH_4)_2HPO_4$) as a source of N and P_2O_5 and ammonium sulfate ($(NH_4)_2SO_4$) as a supplementary N source. The sludge application rate was based on the nutrient requirements of wheat, which represents one of the most common crops in Chile [36].

Non-assimilated N and P were assumed to be emitted to air and to enter water through volatilization and leaching processes. Airborne emissions of NH_3, N_2O and N_2 and water emissions of NO_3^- were estimated based on Brentrup et al. [37], considering local conditions. Emissions of P were estimated assuming that 2.575% of the applied P was leached as PO_4^{-3} [34].

Sub-system 7: sewage treatment

To account for the effect of PT on sewage treatment emissions derived from variations in dewatered supernatant nutrient loads, water treatment was modeled according to Doka [34].

Chemical stabilization sub-systems (scenarios 0a and 0b)

Chemical stabilization (i.e., using lime) scenarios included in-plant operations and transport and disposal of stabilized sludge in landfill sites.

Sub-system 1: thickening

As in S_1 and S_2, sludge from the wastewater treatment was thickened prior to stabilization. The operational parameters used for this sub-system correspond to those used in the digestion scenarios.

Sub-system 8: lime stabilization

After thickening, sludge is subjected to chemical stabilization using lime. Emissions during alkalinization were calculated based on the ammonia concentration and an estimated molecular formula for sludge based on Houillon and Jolliet [38]: $C_{11.7}H_{18.5}O_{6.1}N$. NH_3 emissions were estimated assuming that all free ammonia present in the sludge was emitted directly to the atmosphere.

Sub-system 5: dewatering and storage

Efficiency and electricity/chemical consumption of dewatering were assumed equal to S_1. It was estimated that after storage sludge has 70% humidity [39], and no emissions were considered in this step as it was assumed they occurred during stabilization.

Sub-system 9: landfilling

After storage, sludge is transported to landfill sites for disposal; the same truck capacities and distance used for S_1 and S_2 were considered here. Emissions, landfill gas generation and material inputs were estimated based on Doka [23]. Leachate flow was set at 64 m^3/day, while 40% of the generated gas was assumed to be emitted directly to the atmosphere and the remaining 60% [40] was recovered and used as described in Sect. "Sub-system 11: electricity recovery".

Sub-system 10: leachate treatment

Leachate generated due to sludge disposal was treated according to Doka [34]. Direct emissions and electricity consumption were accounted based on leachate characterization.

Sub-system 11: electricity recovery

Recovered landfill gas is burned as an alternative to avoid direct methane emissions (S_{0a}) or to produce electricity (S_{0b}). Electricity generation in S_{0b} (E; kWh) was determined using Eq. (1), where P_m represents methane production (m^3/day), LCV is the lower heating value of methane (17,657 BTU/m^3), h represents the hours of operation (21 h/day) and CR is the calorific rate (12,000 BTU/kWh) of internal combustion engines [41].

$$E\,(\mathrm{kWh}) = \frac{P_\mathrm{m} \cdot \mathrm{LCV} \cdot h}{\mathrm{CR}}. \tag{1}$$

Life cycle impact assessment (LCIA)

Impact categories were selected based on the correspondence between LCI data and potential environmental impacts and previous reports for sludge management [13, 14]. Moreover, the focus was oriented towards categories related to organic matter and nutrients flows, as those are more likely to be affected by the inclusion of PT in the advanced digestion scenario. Categories selected include climate change, abiotic depletion, acidification and eutrophication (terrestrial, freshwater and marine) impact potentials. Recommendations of the International Reference Life Cycle Data System (ILCD) [42] were followed for the selection of the assessment methodologies. Detailed information can be found in the Supplementary Information (Table SI3).The impact results were calculated by means of SimaPro 8.0.2 software, using a midpoint approach.

Sensitivity analysis

Different types of uncertainty are part of LCA studies, related to inventory parameters, modelling of environmental impacts and scenario choices, among other factors [43]. In this work, we focused on uncertainty derived from assumptions and estimations made during the LCI phase and, therefore, a sensitivity analysis was performed over parameters that showed a significant effect over results. In the base scenario, it was assumed that all carbon present in sewage and the associated emissions were of biological origin. However, the presence of fossil carbon compounds in sewage has been previously reported [44], which can contribute to greenhouse gases (GHG) emissions during treatment and sludge management. Therefore, a sensitivity analysis was performed on the presence of fossil carbon in CO_2 emissions, in the range of 0–30% of total sludge-related CO_2 emissions.

Moreover, it was also assumed that all heat generated in the CHP unit in S_1 and S_2 could be used to replace heat generated from fossil sources, particularly natural gas. While this scenario is desirable as heat could be used for digester heating or sludge drying in AD plants, it is not always feasible due to technical or economical limitations. Therefore, a sensitivity analysis was performed on the percentage of heat (0–100%) from biogas burning that could be used to replace natural gas in S_1 and S_2.

Results and discussion

Table 1 summarizes the inventory data for the four scenarios. Broadly, AD scenarios (S_1 and S_2) resulted in higher energy consumption, lower emissions and higher energy replacement compared with the chemical stabilization scenarios. PT inclusion in S_2 resulted in higher electricity and heat consumption, emissions and requirements for the transport and disposal of sludge compared to S_1, while it also resulted in a higher replacement of electricity and heat due to the increase in biogas production.

The environmental impact assessment results are presented in Fig. 2, categorized into the different sub-systems and discussed accordingly.

Table 1 Summary of the inventory data for the four scenarios assessed. All quantities are referred to the functional unit

	S (conventional digestion)	S (advanced digestion)	S (lime and landfilling)	S (lime and landfilling + ER)
Inputs				
Electricity (kWh)	167.0	234.8	97.5	97.5
Heat (kWh)	637.4	649.1	–	–
Polyacrylamide (kg)	9.1	9.1	9.1	9.1
Transport (tkm)	369.4	501.6	333.2	333.2
Lime (kg)	–	–	164.7	164.7
Outputs				
Emissions to air				
CH_4 biogenic (kg)	8.9	10.6	23.7	23.7
CO_2 biogenic (kg)	597.4	712.6	307.7	307.7
H_2S (kg)	0.01	0.01	9.0×10^{-5}	9.0×10^{-5}
NH_3 (kg)	0.72	0.85	14.0	14.0
N_2O (kg)	0.17	0.16	0.04	0.04
NO_x (kg)	0.08	0.10	–	–
Emissions to water				
PO_4^{2-} (kg)	27.2	27.6	28.9	28.9
NO_3^- (kg)	264.1	264.5	262.8	262.8
NH_4^+ (kg)	58.0	58.3	60.0	60.0
NO_2 (kg)	3.4	3.4	3.6	3.6
N_{part} (kg)	2.6	2.6	2.7	2.7
Avoided products				
Energy				
Electricity (kWh)	704.5	839.1	–	64.9
Heat (kWh)	750.4	894.8	–	–
Fertilizers				
N (as ammonium sulfate) (kg)	11.5	11.0	–	–
P_2O_5 (as diammonium phosphate) (kg)	1.6	1.4	–	–

ER: electricity recovery

Climate change potential

The scenarios that included biological stabilization through AD resulted in lower CCP, as observed in Fig. 3 ($S_{0a} > S_{0b} > S_1 > S_2$). The main contributors in S_1 and S_2 were the direct release of CH_4, electricity consumption and chemicals consumed during thickening and dewatering (Fig. 2). While PT resulted in increased energy consumption, emissions and requirements for the transport of sludge, the overall CCP was lower than in S_1 due to the greater replacement of electricity and heat. The observed effect on the transport and spreading of sludge was related to the lower water recovery from the digestate (Table SI1), comparable to what has been reported previously for similar PT processes [9]. Regarding S_{0a} and S_{0b}, the emission of landfill gas was the main contributor to CCP, in accordance with previous reports [17, 40]. GHG emissions could not be compensated by electricity recovery from landfill gas, which resulted in a CCP reduction of approximately 5% in S_{0b} compared to S_{0a}.

Abiotic depletion potential

Regarding ADP, S_1 was the only scenario where a net environmental benefit was observed (Fig. 3; $S_{0a} > S_{0b} > S_2 > S_1$), associated with the replacement of electricity, heat and commercial fertilizers (Fig. 2). In S_2, the replacement of conventional energy sources was higher than in S_1; however, the reduced replacement of fertilizers due to the lower concentration and loads of N and P in sludge (Table SI2) plus the higher consumption of fossil fuels associated with the transport and spreading of sludge resulted in an increased ADP.

ADP in S_{0a} and S_{0b} was mostly associated with sludge landfilling, followed by electricity consumption during thickening/dewatering and lime production. The most significant contributor to ADP during landfilling was the use of fossil fuels for transport. The production of electricity in S_{0b} resulted in a 2% reduction in ADP compared to S_{0a}.

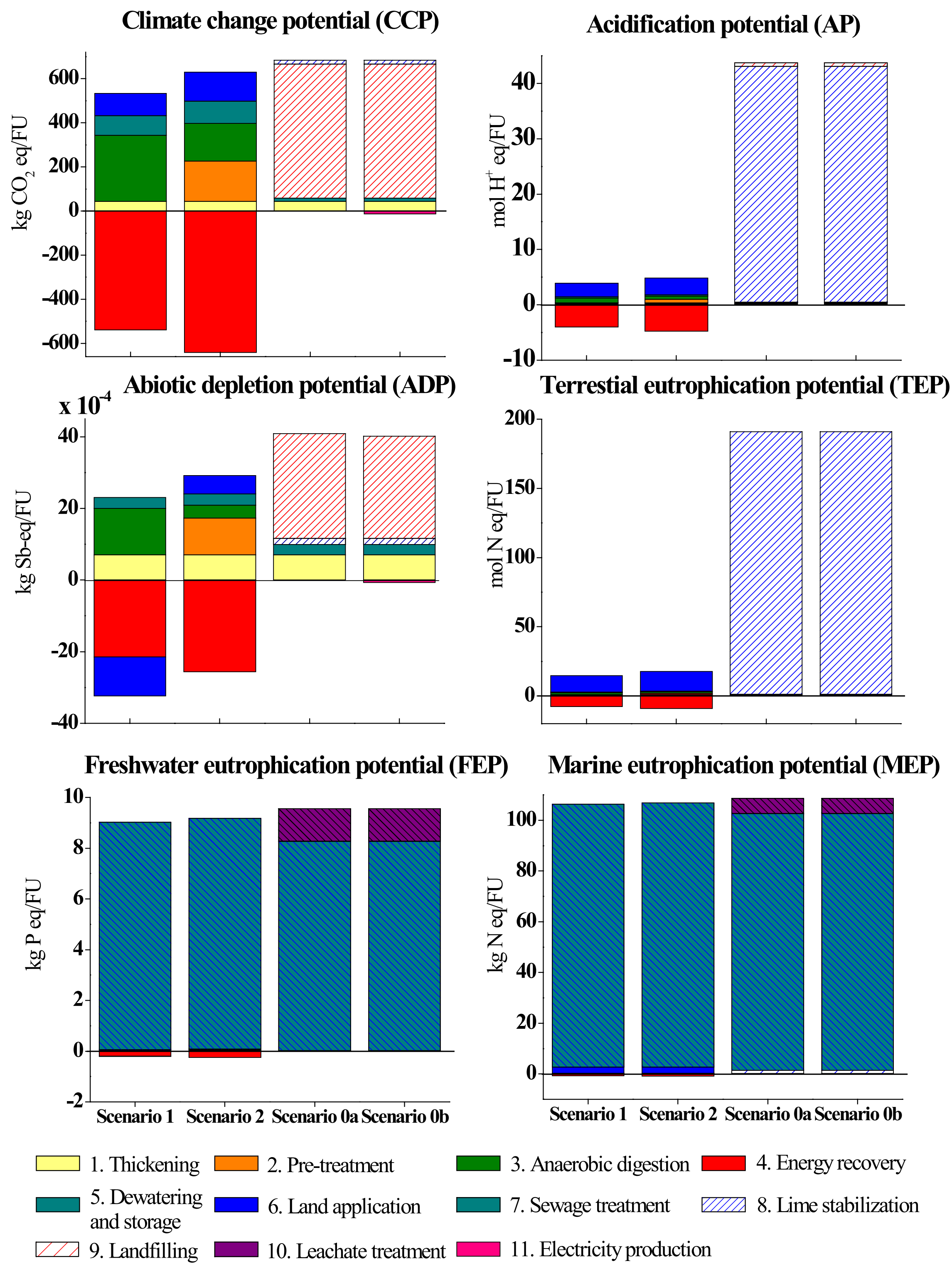

Fig. 2 Comparison of the environmental impact for the four scenarios assessed. Scenario 1: conventional digestion; Scenario 2: advanced digestion; Scenario 0a: lime and landfilling; Scenario 0b: lime and landfilling with electricity recovery

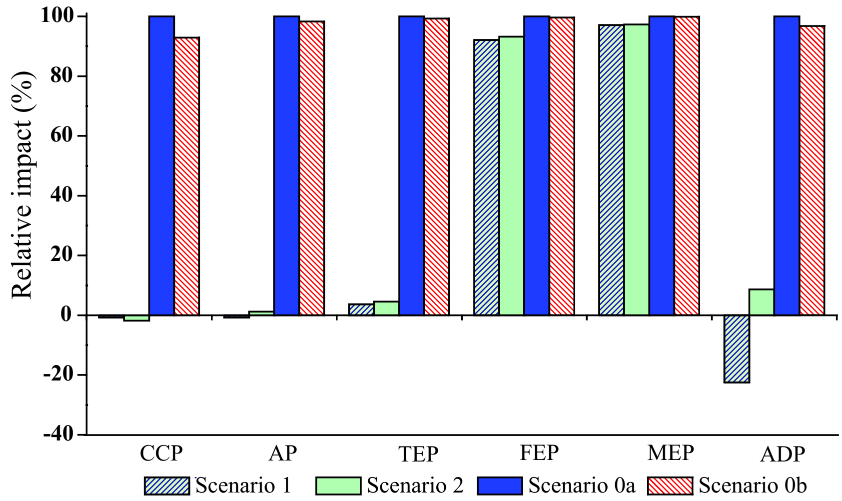

Fig. 3 Relative impact of the four scenarios under study, taking the highest value for every category as baseline. Scenario 1: conventional digestion; Scenario 2: advanced digestion; Scenario 0a: lime and landfilling; Scenario 0b: lime and landfilling with electricity recovery

Acidification and terrestrial eutrophication potential

S_1 resulted in the lowest AP and TEP, as observed in Fig. 3 ($S_{0a} > S_{0b} > S_2 > S_1$). The incorporation of the PT slightly increased both categories mainly due to emissions during land application, which represented the main contributor in the corresponding scenarios (Fig. 2). This was associated with increased NH_3 volatilization due to a higher N-NH_4^+ load in sludge applied to soil (Table SI2). The use of management practices oriented toward the efficient use of sludge N in soil could be used to mitigate AP and TEP impacts in those scenarios.

The AP and TEP values for S_{0a} and S_{0b} were more than 700 times higher than those of S_1. This was mostly related to the emission of NH_3 during lime stabilization, representing approximately 98% of the total contribution to both impact categories. The use of odor and NH_3 filters during this step could lead to decreased AP and TEP, in addition to the increased acceptance of WWTPs by the surrounding population.

Freshwater eutrophication potential

Regarding FEP, S_{0a} and S_{0b} presented higher values than S_1 and S_2, as observed in Fig. 3 ($S_{0a} > S_{0b} > S_2 > S_1$). However, as this category was mostly associated with P emissions during sewage treatment (80% of contribution; Fig. 2), the differences between scenarios were less than 9%.

The overall effect of PT on FEP was only a 1.3% variation with respect to S_1, mainly due to the increased P concentration and load in the dewatering process supernatant (Table SI2). The overall contribution of land application to FEP was negligible for both S_1 and S_2 (< 0.03%), which was related to the low mobility of P in soil assumed in this study [34]. However, it is important to note that the application of fertilizers such as sludge over a long period of time or with high rates could result in soil oversaturation of P and its release to water bodies [45], which can be expected if disposal is the primary goal of sludge land application.

Marine eutrophication potentials

Figure 3 shows that MEP presents the same trend that FEP in terms of scenario performance ($S_{0a} > S_{0b} > S_2 > S_1$) and main contributor (i.e., sewage treatment; Fig. 2), with differences of less than ~ 3% between scenarios. However, NO_3^- emissions from land application in S_1–S_2 and leachate treatment in S_{0a}–S_{0b} were also relevant. The principal effect of PT was related to an increased N concentration and load in the dewatered supernatant re-circulated to sewage treatment (Table SI2), which resulted in a negligible ~ 0.3% variation in MEP in S_2 compared to S_1.

Sensitivity analysis

Table 2 shows the sensitivity analysis performed in the presence of fossil carbon in sludge CO_2 emissions.

A higher rate of change was observed for S_2 due to the higher CO_2 emissions associated with biogas leakage and burning. When the fossil carbon presence was higher than 6.3%, CCP in S_2 was higher than in S_1.

Table 3 shows the sensitivity analysis performed on the heat replaced by biogas burning.

The heat recovery strategy associated with AD was of importance for the CCP, ADP and AP impact categories, and S_2 had the highest slope. S_2 exhibited the highest values for ADP and AP under all conditions, while CCP was higher for this scenario when less than 82.2% of the produced heat was used to replace the heat from natural gas.

Table 2 Sensitivity analysis for fossil carbon presence in sludge-related CO_2 emissions

	% fossil carbon	$CO_{2biogenic}$ (kg/FU)	$CO_{2fossil}$ (kg/FU)	CCP ($kgCO_{2eq}$/FU)	ΔCCP ($kgCO_{2eq}$/FU)	CCP slope[a]
S_1 (conventional digestion)	0	597.4	0.0	− 5.4	–	6.0
	10	537.7	59.7	54.4	59.8	
	20	477.9	119.5	114.1	119.5	
	30	418.2	179.2	173.9	179.3	
S_2 (advanced digestion)	0	712.6	0.0	− 12.6	–	7.1
	10	641.3	71.3	58.6	71.2	
	20	570.1	142.5	129.9	142.5	
	30	498.8	213.8	201.1	213.7	
S_{0a} (lime and landfilling)	0	307.7	0.0	713.9	–	3.1
	10	276.9	30.8	744.7	30.8	
	20	246.2	61.5	775.5	61.6	
	30	215.4	92.3	806.0	92.1	
S_{0b} (lime and landfilling + ER)	0	307.7	0.0	677.5	–	3.1
	10	276.9	30.8	708.3	30.8	
	20	246.2	61.5	739.1	61.6	
	30	215.4	92.3	769.8	92.3	

CCP climate change potential; *ΔCCP* variation in CCP compared with the base scenario. *ER* electricity recovery

[a]Expressed as $kgCO_{2eq}/FU - \%_{fossil\ carbon}$

Table 3 Sensitivity analysis for the heat replaced by biogas burning in S_1 and S_2

	% of produced heat used as replacement of natural gas	CCP ($kgCO_{2eq}$/FU)	ADP ($\times 10^{-4}$) ($kgSb_{eq}$/FU)	AP ($\times 10^{-2}$) (mol_cN_{eq}/FU)
S_1 (conventional digestion)	100	− 5.4	− 9.3	− 6.2
	80	37.4	− 6.5	2.7
	60	80.2	− 3.6	11.5
	40	123.0	− 0.8	20.4
	20	165.8	2.0	29.2
	0	208.6	4.8	38.1
	Slope[a]	− 2.1	− 0.14	− 0.44
S_2 (advanced digestion)	100	− 12.7	3.6	9.9
	80	38.3	6.9	20.4
	60	89.3	10.3	31.0
	40	140.3	14.0	41.5
	20	191.2	17.0	52.0
	0	242.2	20.4	65.3
	Slope[a]	− 2.5	− 0.17	− 0.55

CCP climate change potential; *ADP* abiotic depletion potential; *AP* acidification potential

[a]Expressed as the reference unit for corresponding impact category per FU and % replaced heat

Overall comparison between scenarios

Figure 3 shows the comparison between the scenarios in terms of their relative impact. From an overall perspective, the utilization of AD followed by land application was the most appropriate alternative, with lower impacts in all selected categories. It is important to note that our results are valid in similar conditions to those assessed, as previous reports show that management strategies and selected impact categories could greatly affect the outcome results of sludge management LCA and, therefore, other alternatives could result in even better environmental performance than AD [14]. However, this work illustrates that sludge valorization through energy recovery and land application represents a central aspect of sustainable sludge management, as opposed to practices that consider sludge only as waste (i.e., landfilling). In the alkalinization scenarios, stabilization and landfilling represented the main contributors to environmental impacts, which is in agreement with reports identifying emissions, lime production and the lack of valorization of sludge as important environmental hotspots [14].

The comparison between advanced and conventional digestion indicates that the burdens associated with PT need to be considered. Sequential PT had positive effects on energy recovery and CCP, but due to the greater needs associated with transport/spreading and the lower replacement of commercial fertilizers, ADP was negatively affected. This was associated with operational effects such as the lower water recovery from the digestate and changes in NH_4^+, N and P concentrations and loads in the sludge and supernatant. These factors also affected the AP, TEP, FEP and MEP impact categories, but the overall values were similar to the conventional digestion scenario. The sensitivity of the results to the assessed parameters should also be accounted when comparing the performance of the different scenarios, as those show different behaviors when modified (Tables 2, 3).

It is important to note that one main effect of PT on digestion is associated with increased biodegradation kinetics [8, 12]. Intensification of the digestion process could lead to smaller digester volumes and decreased capital costs, increasing the attractiveness of the technology for smaller WWTPs. Currently, there are only 6 WWTPs in Chile that use AD for sludge stabilization; these plants treat sewage that exceeds 50,000 person equivalent [2]. As the studied PT resulted in a similar performance to conventional digestion, this may become an interesting tool for the energetic valorization of sludge in smaller plants, which could lead to a decrease in the environmental burdens of sludge management in Chile compared to the current scenario. To further optimize the environmental performance of advanced AD, operational aspects such as organic loading rate and retention time should also be considered prior to its widespread application, as previous reports state their relevance over the life cycle impacts of AD [30].

Overall, the environmental performance of the advanced digestion scenario was relatively more sensitive to the presence of fossil carbon in sewage and to the effective valorization of the produced heat. This was related to process intensification and the corresponding higher emissions and energy consumption and generation. The chemical stabilization scenarios showed relatively lower sensitivity to the presence of fossil carbon in CO_2 emissions, which was mostly related to its lower contribution compared with leaked methane from landfill sites. However, even the worst case scenarios of digestion in terms of fossil carbon presence exhibited lower CCP than the best chemical stabilization scenarios.

Conclusions

The combination of AD and the agricultural valorization of digestate exhibited lower potential impacts than the chemical stabilization scenarios in all selected categories. When sequential ultrasound-thermal PT was included, the main effects were a decrease in CCP and an increase in ADP. The influence of PT on digestion performance was related to its effects on energy recovery, transport requirements and nutrient loadings, highlighting the need to assess its performance from a life-cycle perspective. Advanced digestion including the assessed PT showed a similar performance to conventional digestion, but the results were more sensitive to the possible emission of fossil CO_2 and heat valorization strategies. Considering the results, PT can represent an interesting alternative for the implementation of AD in WWTPs lacking sludge valorization strategies, which in the case of Chile could led to decreased environmental burdens compared to the current scenario.

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