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Life cycle assessment of lightweight aggregates produced with ashes from municipal solid waste incineration

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Abstract

Air pollution control residues (APCr) from municipal solid waste (MSW) incineration are hazardous waste, and its management requires holistic approaches within technical, economical, legislative, and environmental constraints. This work deals with the recycling of APCr for producing lightweight aggregates (LWA) commonly manufactured by firing natural clay. The main objectives are to evaluate the environmental impacts of LWA with and without incorporating APCr and assess whether APCr recycling in LWA is beneficial, based on the life cycle assessment (LCA) methodology. The system boundary included raw materials extraction, grinding, mixture, firing, cooling, and packing (cradle-to-gate). Results were analysed following the impact assessment methods recommended by the International Reference Life Cycle Data System. Results pointed out a reduction of impacts in all categories when 3% of natural clay is replaced by APCr. The highest gains occurred for toxicity categories (HTc, HTnc) and resulted mainly from avoiding landfill of APCr. For non-toxicity categories, impacts were dominated by the emissions from the kiln firing process, which were similar for both LWA; therefore, impact reductions from APCr use in LWA in these categories were modest. LCA results show that the valorisation of APCr in LWA may be an environmentally-sound solution to avoid landfill disposal practices.

Keywords: Life cycle assessment, Lightweight aggregates, Recycling, APC residues, Incineration

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1. Introduction

Municipal solid waste (MSW) incineration in modern waste-to-energy (WtE) facilities has been increasingly used in developed countries [1-2]. The worldwide capacity of nearly 2000 WtE plants is about 250 Mt/year, with a rising trend [3]. Flue gas cleaning systems contribute to environmental protection but create air pollution control residues (APCr). These residues may represent about 3-6% of the MSW incinerated and have been categorized as hazardous waste. According to Commission Regulation (EU) Nº 1357/2014, the likely properties that render it hazardous are HP 4 – irritant – due to highly alkaline properties, and HP 14 – ecotoxic – mainly due to the presence of heavy metals and soluble salts of acid gases. The hazardous nature of APCr limits its management options. A common technology is the solidification/stabilization (S/S) of APCr before disposal in controlled sanitary landfills [4-5]. However, due to the EU waste acceptance criteria related to strict limits on leaching, this option may not be acceptable in the future [6].

Quina et al. [2] identified six alternative routes to landfill disposal for valorizing APCr: recycling in LWA, glass-ceramics, and cement, and recovery of zinc, rare metals, and salts. The environmental impacts of some of these alternative options for APCr management have been assessed using the life cycle assessment (LCA) methodology [7-9]. Fruergaard et al. [7] evaluated seven scenarios covering the wide range of technologies available worldwide (direct landfilling, backfilling in salt mines, neutralization of waste acid, filler material in asphalt, ferrox stabilization, vitrification, and melting with automobile shredder residues) and pointed out that thermal processes were associated with the highest loads in the non-toxicity categories (due to energy consumption), whereas in the toxicity categories, all treatment options performed better than landfilling. Huang and Chuieh [8] compared four scenarios for reuse and disposal of fly ash and concluded that its reuse in bricks after a washing process had globally the smallest impact. Besides, the authors mentioned that the treatment of wastewater from the washing process must be improved. Huber et al. [9] analysed five different scenarios: underground deposit, cement stabilisation, the FLUREC process, thermal treatment in a dedicated furnace and thermal co-treatment. The main conclusions were that stabilisation with cement and thermal treatment in a dedicated furnace had the highest environmental impact. The thermal co-treatment of fly ash together with combustible waste was a promising scenario.

The recycling of fly ash and/or APCr from MSW incineration in lightweight aggregates (LWA) has been tested by several authors [10-18]. LWA is porous ceramic products often used in the production of light concrete blocks, structural concrete and

pavement. In general, LWA is manufactured from clay, shale or slate. LWA has a density between 0.15 and 1.7 kg/dm³, excellent insulating properties (due to the large intraparticle porosity), thermal conductivity ranging from 0.07 to 0.18 W/m.K, and high compressive strength [19]. Quina et al. [10-12, 20] showed that, while incorporating APCr in LWA did not improve its technical properties, the aggregates had similar technological properties if the APCr content did not exceed 3% (wt) or 5% in the case of pre-washed residues. Nevertheless, the potential environmental benefits of APCr recycling in LWA are yet to be demonstrated. Furthermore, only a few studies report on the environmental profile of LWA production [21].

This work aims to complement the two previous ones. In Quina et al. [11], the feasibility of producing LWA by replacing a certain amount of natural clay by APCr was investigated. Later on, the effect of the incorporation of APCr (with and without washing pre-treatment) on the main technological properties of LWA as well as on the environmental impacts due to leaching processes were investigated [12]. The authors found that the substitution of 3% of natural clay by APCr did not compromise the technical characteristics of the LWA neither exceeded the leaching legal limits for disposed of waste in hazardous landfills. The present study performs a comparative LCA of LWA produced with and without APCr to evaluate whether the incorporation of APCr into LWA is an environmentally beneficial solution.

2. Materials and methods

2.1. Goal and scope

This study aims to compare the life cycle environmental impacts of LWA produced in an industrial plant located in Portugal considering two scenarios: (i) LWA produced using the traditional production process in which only natural clay is used (T-LWA) and (ii) LWA incorporating APCr from MSW incineration (APCr-LWA), in which 3% of clay input is replaced by APCr without compromising the technical properties of the final product [11-12], i.e. both products are equivalent. The LCA methodology was used to assess the environmental impacts of both scenarios [22-23]. The simulation and modelling were conducted using the LCA software SimaPro 8.0.

The functional unit is 1 Mg of LWA as a final product (density: 0.168 kg/dm^3), and the system boundary is presented in Fig. 1 for the T-LWA and in Fig. 2 for the APCr-LWA. A cradle-to-gate approach is followed, i.e., both the use and end-of-life phases of the LWA are excluded from the analysis. The following phases, common to both systems, are included:

clay extraction, LWA industrial process, and production and transport of auxiliary materials (oil, plastic) and energy carriers. For the APCr-LWA, the transportation of the APCr from the incineration plant to the LWA industrial plant was considered. Since the incorporation of APCr in LWA avoids its solidification/stabilization treatment and landfill disposal, the environmental burdens of these processes are considered a credit to the APCr-LWA system (avoided burdens or substitution approach).

2.2. Inventory analysis

The inventory analysis was based on data collected from an industrial plant that produces LWA using natural clay, data from the literature, and data from the ecoinvent v.3 database.

2.2.1. Clay extraction and transport

Clay is extracted from a clay-pit using diesel-powered backhoes. The extraction of 1 Mg of clay requires 0.19 L of diesel. Clay is then transported to the industrial plant in a 25-ton lorry, over 40 km. Data for the operation of the backhoes and the lorry was obtained from the ecoinvent v3 database.

2.2.2. Industrial process

The industrial process includes: (i) grinding of clay in electric-powered mills; (ii) mixing the clay with water, a small portion of waste lubricant oil (as expansion agent) and APCr (only for the production of APCr-LWA), requiring electricity; (iii) firing in a rotary kiln at near 1170ºC using petroleum coke and electricity; in this phase, emissions to air arise due to both burning of fuel to obtain heat and volatilization/reactions of raw materials; (iv) cooling through contact with air and water showers, requiring electricity and water; and (v) packaging with plastic film, requiring electricity.

Data for the industrial process were collected on-site and include input material flows (clay, auxiliary materials, and packaging materials), energy consumption (electricity and heavy fuel oil), and water consumption. The main inputs for both systems are summarized in Table 1.

According to industrial information, Table 2 shows the main outputs of the T-LWA production process (direct emissions). The main pollutants emitted by the LWA industry (kiln firing) are particulate matter (PM), sulphur oxides (SO_x), nitrogen oxides (NO_x), carbon monoxide (CO), carbon dioxide (CO_2), volatile organic carbon (VOC), and hydrogen chloride (HCl). In general, low content of heavy metals is detected in clays, and thus no emissions of

these pollutants are expected [11, 24]. In the case of the production of APCr-LWA, the main outputs were considered equal to those of T-LWA, with the exception of HCl, due to the content of chloride in APCr (as indicated in Table 3). The industrial plant used fabric filters to control particulate emissions from the rotary kiln.

2.2.3. Solidification and landfill disposal of APCr

The recycling of APCr from MSWI avoids the landfill disposal of this residue and hence the burdens associated with this process. Therefore, the substitution or avoided burdens approach is used and credit is given for the impact of the APCr disposal process that is avoided by its use in the LWA matrix.

Before its disposal in a residual landfill, APCr are usually solidified using a mixture of residue (42%), cement (25%), and water (33%) [25]. This mixture is transported in a 25-ton truck lorry to the residual landfill. Emissions from residual landfill disposal of APCr were modelled using the Excel calculation tool from Doka [26] and considering the chemical composition of APCr residue indicated in Table 3 [20, 27].

2.2.4. Background processes

The production of auxiliary materials, water, and energy carriers was considered and data for these processes were taken from the ecoinvent v.3 database, except for electricity generation, which was based on life-cycle inventories implemented for Portugal by [28 -30] considering the average electricity mix in 2012-2016. The transport of materials and fuels was also taken into account considering the average distances and transport modes depicted in Table 4. Data for the operation of the lorries was obtained from the ecoinvent v3 database.

2.3. Impact assessment

The selection of impact categories and life cycle impact assessment (LCIA) methods were based on those recommended by the International Reference Life Cycle Data System [31] (levels I or II), for which enough inventory data could be obtained. Table 5 summarizes the impact categories considered in this study, which were analysed at midpoint level: climate change (CC), ozone depletion (OD), human toxicity – cancer (HTc) and non-cancer (HTnc) effects, particulate matter (PM), photochemical ozone formation (PO), acidification (AC), terrestrial (TE) and marine eutrophication (ME), freshwater ecotoxicity (FE), and mineral, fossil and renewable resource depletion (RD). For toxicity-related categories (HTc, HTnc, FE), impacts were calculated using recommended characterization factors (CFs) as default,

but a sensitivity analysis considering also interim CFs (i.e. with higher uncertainty) was performed to test the sensitivity of the conclusions.

2.4. Sensitivity analysis

A sensitivity analysis of the electricity generation mix used in the production of LWA was performed to evaluate the effect of a change in this process in the impact assessment results. The Portuguese electricity mix (PT mix) has a high contribution from renewable resources, attaining 40-55% in 2012-2016 [41]. Nonetheless, the contribution of renewables is expected to increase with the phase-out of coal power plants and the need to meet the EU renewable energy goals. Therefore, the influence of changing the electricity mix used in LWA production to a renewable-based source in the life cycle environmental impact assessment was assessed. Two renewable energy systems were considered: wind and hydro reservoir, currently representing 20% and 12% of total electricity generated in the country, respectively [41]. The life-cycle inventories used in this assessment were based on [29 -30].

3. Results and discussion

3.1. Environmental impacts of T-LWA production

The first simulations for estimating the environmental impacts were performed for traditional LWA manufacturing. Figure 3 summarises the contribution of the different industrial processes to the environmental impacts of T-LWA. The LWA industrial process (including grinding, mixing, kiln firing, cooling, and packaging) is responsible for more than 3/4 of the impacts in CC, OD, PM, PO, AC, TE, ME, and FE. The kiln firing process alone contributes to more than 84% of the impacts in AC, CC, TE, PM, PO, and ME, mostly due to combustion emissions (CO_2 for CC, NO_x for PO, AC, TE, and ME, SO_x for PM, PO and AC, particulates for PM, and VOCs for PO) resulting from the use of petroleum coke. The production of petroleum coke is responsible for 62% of the impacts in OD, due to the use of fire extinguishers in the crude oil production chain. The generation of electricity used in the various production processes is the second highest contributor to the impacts in HTnc (32%), due to emissions of 2,3,7,8-tetrachlorodibenzo-p-dioxin and formaldehyde from the combustion of volatile organic compounds, such as hydrocarbons in fossil fuels, and RD (18%), mainly from the use of metals in renewable energy infrastructure (e.g., zinc, lead, ferronickel). The packaging of LWA is the highest contributor to the impacts in FE (55%), mainly due to the burdens of petroleum extraction for the production of the plastic film. The

transport of clay to the industrial plant accounts for more than half of the impacts in HTc and RD, mostly resulting from the disposal of slag from steel production and lead production for the lorry, respectively. Clay extraction has a negligible contribution to the impacts.

3.1.1 Sensitivity analysis: electricity generation mix

Energy consumption is one of the main causes of environmental impacts of the LWA industrial process in several categories. In particular, the electricity generation mix in Portugal is highly variable leading to variations in the environmental impacts associated with the consumption of electricity [28]. Since it is expected that the contribution of renewable resources in the energy matrix increase, a sensitivity analysis regarding the electricity source was performed, considering wind and hydro generation (Fig. 4). Results in Fig. 4 show that renewable-sourced electricity only modestly reduces the environmental impacts of LWA in most categories (<10%). The highest reductions are achieved in HTnc (29%, for wind, and 32%, for hydro) and RD (19%, for hydro). However, an increase in impacts in RD (by 12%) is observed when wind-sourced electricity is used due to the metals used in the infrastructure.

3.2. Comparative environmental assessment of T-LWA and APCr-LWA production

The results obtained for both T-LWA and APCr-LWA regarding the different impact categories are shown in Table 6. Incorporating 3% of APCr in the LWA matrix (APCr-LWA) resulted in an overall reduction of impacts in all categories. The highest savings (27-42%) occurred for human toxicity categories (HTc, HTnc). Reductions between 10 and 18% were found in RD, OD, and FE; for the remaining impact categories, savings were modest (below 5%).

Impact reductions were obtained mostly by avoiding solidification/stabilization with cement and landfill disposal of the APCr residue, particularly for HTc and HTnc (Fig. 5). Avoiding the transport of APCr after solidification with cement was also important to reduce RD. The reduction of clay input had a minor contribution to decrease environmental impacts of LWA production by reduced clay transport and electricity use (0.1-2.1% reduction). The increase in transport distance from replacing clay by APCr offset that impact reduction (the transport distance of APCr is five times higher than clay). Impact reductions from the use of APCr in LWA production could be further increased by optimizing the distance between the LWA industrial plant, the MSW incineration plant, and the clay pit. In the remaining impact

categories (CC, PO, AC, TE and ME) the main impacts arise from the kiln firing process and the reductions can be achieved by changing the fuel used, influencing positively both LWA.

3.2.1 Sensitivity analysis: toxicity assessment

The highest reduction in environmental impacts achieved with the incorporation of APCr in LWA occurred for toxicity categories, which were assessed using the USETox method considering recommended characterization factors (CFs) only. The recommended CFs do not include all substances causing toxicity effects; therefore, a sensitivity analysis was performed including interim CFs, which have higher uncertainty (Table 7). Impacts in all toxicity categories increased for T-LWA by 3 (HTc and FE) to 4 (HTnc) orders of magnitude. For HTc, most impacts resulted from emissions of chromium emissions to water from the treatment of spoil from coal mining for electricity generation and incineration residues in the packaging production chain. For HTnc, impacts were mostly due to zinc and arsenic emissions to water from treatment of spoil from coal mining for electricity generation, and zinc emissions to air from tire wear in transportation processes. For FE, the main contributors were copper and zinc emissions to water from the treatment of scrap copper from the electricity transmission network and spoil from coal mining for electricity generation, respectively. On the other hand, impacts for APCr-LWA decreased and became negative, as a result of avoided APCr landfill emissions of chromium VI for HTc, arsenic for HTnc, and antimony for FE. All those substances were characterized by zero impact on the previous assessment (considering recommended CFs only). Although the magnitude of the impact reduction is highly uncertain, the trend remains: incorporating APCr in LWA can potentially reduce toxicity impacts compared to landfill disposal of these residues.

3.3. Considerations about the use and end-of-life stages

This study followed a cradle-to-gate approach and assessed the impacts of both T-LWA and APCr-LWA production processes up to the factory gate. Several applications for this type of ceramic materials can be found, including in structural lightweight concrete, lightweight concrete masonry, asphalt, geotechnical, and landscape applications [16]. To the best of our knowledge, the main aspect to be considered in the use and end-of-life stages of APCr-LWA is the possible leaching of toxic metals and salts. The literature has shown that leaching processes from LWA do not appear to be significant [11-14, 16, 42]. Indeed, Table 8 provides a quantitative overview of the leaching from T-LWA and APCr-LWA, tested according to the standard EN 12457-2:2002 after grinding. Besides, for comparison in

common baselines, the waste acceptance criteria (WAC) at inert and non-hazardous landfills were also included (defined in Council Decision 2003/33/EC, which establishes criteria and procedures for the acceptance of waste at landfills according to Directive 1999/31/EC). The data in Table 8 shows that except for Ca, leaching of all the other elements is similar between T-LWA and APCr-LWA. All WAC_NH limits were fulfilled, while WAC_inert criteria were likely complied (but further analyses are required using an analytical technique with lower quantification limits). This means that no additional environmental impact is expected during the use and end-of-life stages (phases not included in the LCA) of APCr-LWA over T-LWA. Additional technical difficulties for using APCr in practical applications such as in LWA are due to the high concentration of chlorine (more than 10 wt%), namely halite (NaCl), sylvite (KCl) and calcium hydroxychloride (CaOHCl), as well as calcium chloride hydrate ($CaCl_2 \cdot 2H_2O$) [12, 43-48]. A possible approach for reducing chlorine in APCr (up to 90-93 w% of removal) is through a washing treatment with water [44,45, 47]. However, from the data in Table 8, if 3 wt% of APCr is used in LWA no additional Cl^- leaching is detected from APCr-LWA when compared to the control (T-LWA). Even so, the study of chloride stability in these ceramic materials should be further investigated.

4. Conclusions

A life cycle assessment of lightweight aggregates produced with and without incorporating APCr from MSWI was performed based on eleven midpoint indicators: climate change (CC), ozone depletion (OD), human toxicity carcinogenic (HTc) and human toxicity non-carcinogenic (HTnc), particulate matter (PM), photochemical ozone formation (PO), acidification (AC), terrestrial eutrophication (TE), marine eutrophication (ME), freshwater ecotoxicity (FE) and resource depletion (RD). The LWA industrial processed had the highest contribution to most of the impact categories, namely CC, OD, PM, PO, AC, TE, ME, and FE. In particular, the kiln firing process contributed the most to the impacts in non-toxicity categories, due to both production and combustion of petroleum coke. The generation of electricity used in the industrial process was found to be an important source of impacts on human toxicity and resource depletion. Whilst the former is likely to decrease as the renewable share in the electricity generation mix increases, RD impacts may increase if more wind-based generation is used. Furthermore, results showed that 3% of the APCr may be incorporated in LWA with positive gains in all categories, and in particular for human toxicity. Although our results show that the recycling of APCr in LWA may be a sound solution from an environmental perspective, further studies should be conducted to deepen the

understanding of the process under analysis, namely regarding the washing process for the removal of salts from APCr.

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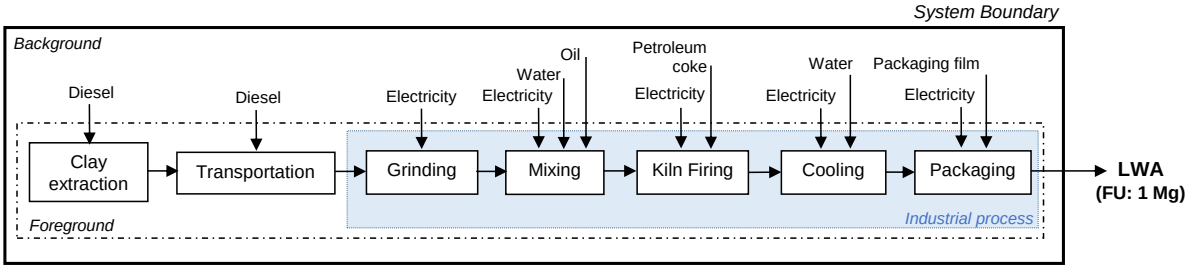

Fig. 1- System boundary of the LCA model of conventional LWA production (T-LWA).

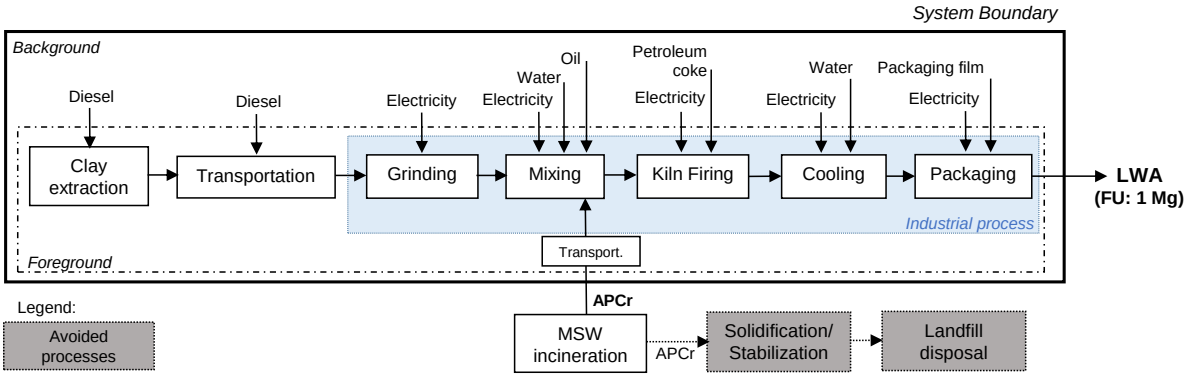

Fig. 2 - System boundary of the LCA model of LWA production incorporating APCr (APCr-LWA).

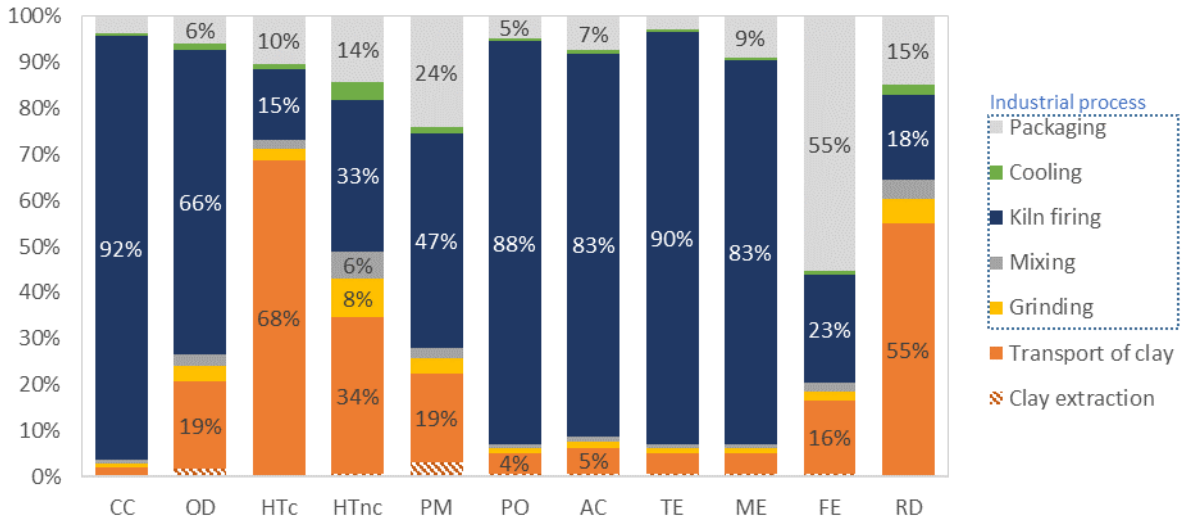

Fig. 3- Contribution of the different processes on the environmental impacts of T-LWA production.

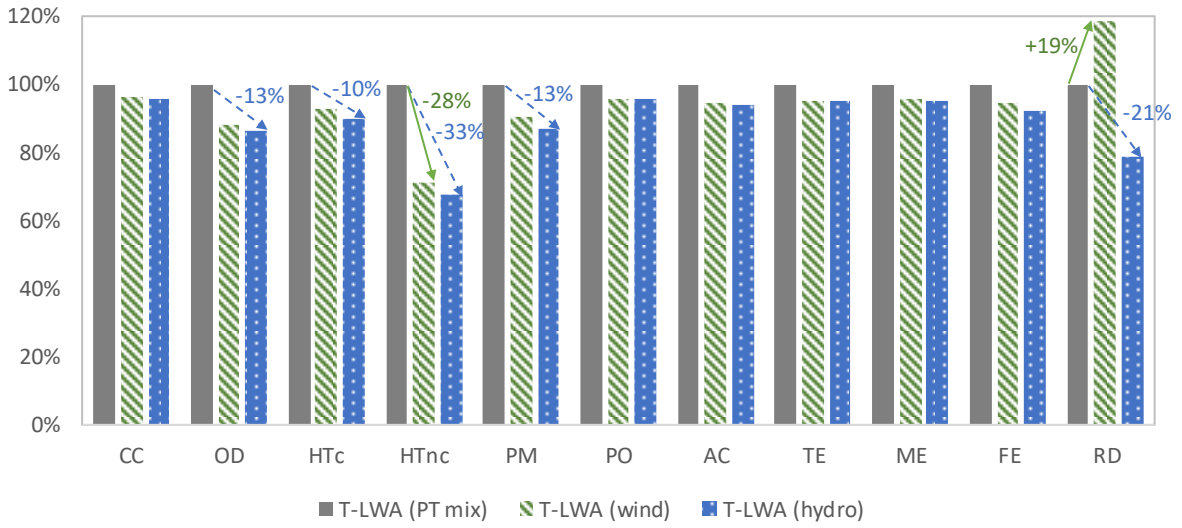

Fig. 4 - Sensitivity analysis to the electricity source used in T-LWA industrial process.

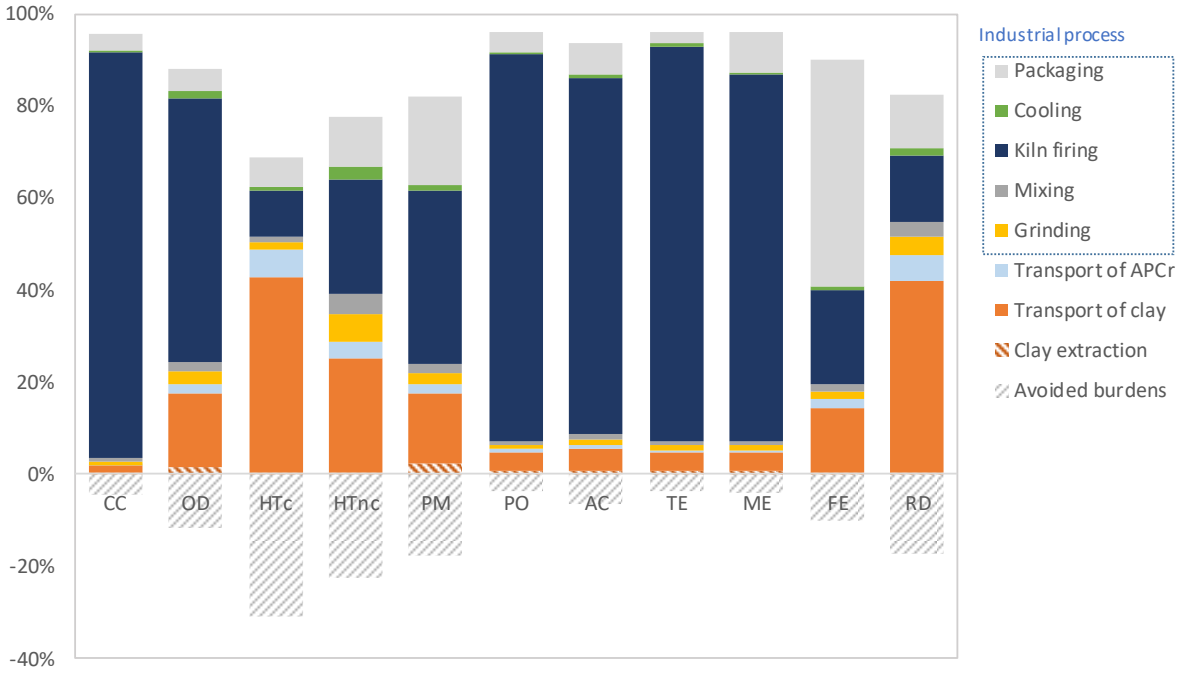

Fig. 5 - Contribution of the different processes to the environmental impacts of APCr-LWA production.

Table 1- Main inputs to produce 1 Mg of LWA of both T-LWA and APCr-LWA.

Input	T-LWA[(a)]	APCr-LWA[(b)]
Clay (Mg)	1.10	1.07
APCr (Mg)	0	0.03
Water (Mg)	0.201	0.201
Oil (Mg)	0.01	0.01
Electricity (kWh)	49	48
Petroleum coke (kg)	42	42
LDPE packaging film (kg)	5.5	5.5

(a) Data provided by an industrial plant (Leca Portugal – Avelar); [(b)] Data considering that 3% of clay is replaced by APCr.

Table 2- Main outputs (kg/Mg LWA) from LWA industrial production for both T-LWA and APCr-LWA (direct emissions, at the industrial plant).

Emissions	T-LWA[(a)]	APCr-LWA[(b)]
CO_2	390	390
CO	1.65	1.65
PM	0.34	0.34
SO_x	1.7	1.7
NO_x	1.0	1.0
VOC	0.39	0.39
HCl	10	13

(a) Data provided by an industrial plant (Leca Portugal – Avelar); [(b)] Data considering that 3% of clay is replaced by APCr.

Table 3- Composition of APCr.

Substance	Fraction[(a)]	Substance	Fraction[(b)]
H_2O	0.0242	P	0.0095
O	0.1924	Ag	0.00001
H	0.0189	As	0.000025
C	0.0655	Ba	0.000365
S	0.0185	Co	0.000017
Cl	0.1383	Mn	0.00052
Cd	0.000087	Mo	0.0000098
Cr	0.000259	Sb	0.00038
Cu	0.000647	Se	0.0000029
Hg	0.0000163	Sn	0.00039
Ni	0.000132	V	0.000019
Pb	0.002408	Be	0.00000032
Zn	0.006367	Sc	0.0000025
Si	0.0831	Sr	0.00035
Fe	0.0161	Ti	0.0015
Ca	0.3105	Tl	0.00000029
Al	0.04	W	0.000017
K	0.0302	Mg	0.0062
Na	0.0331		

[a] Quina et al. (2008b); [b] Bogush et al. (2015)

Table 4- Average distance and transport modes for materials used in both product systems.

Material	Transport mode	Distance
Clay	25-ton lorry (EURO 3)	40 km
Waste lubricant oil	16-ton lorry (EURO 4)	36 km
Plastic	16-ton lorry (EURO 4)	50 km
APCr to industrial plant[a]	25-ton lorry (EURO 3)	200 km
Cement for solidification of APCr[a]	25-ton lorry (EURO 3)	60 km
Solidified APCr to landfill[a]	25-ton lorry (EURO 3)	108 km

[a] Only for APCr-LWA.

Table 5 – Impact categories and life cycle impact assessment (LCIA) methods used.

Impact categories	Units	LCIA Methods
Climate change (CC)	kg CO_2 eq	IPCC 2013 GWP 100a [32]
Ozone depletion (OD)	kg CFC-11 eq	[33]
Human toxicity – carcinogenic (HTc)	CTUh	USETox v1.04 [34][a]
Human toxicity – non-carcinogenic (HTnc)	CTUh	USETox v1.04 [34][a]
Particulate matter (PM)	kg PM2.5 eq	RiskPoll [35, 36]
Photochemical ozone formation (PO)	kg NMVOC eq	LOTUS-EUROS [37]
Acidification (AC)	mol_c H^+ eq	Accumulated Exceedance [38-39]
Terrestrial eutrophication (TE)	mol_c N eq	Accumulated Exceedance [38-39]
Marine eutrophication (ME)	kg N eq	EUTREND model [40]
Freshwater ecotoxicity (FE)	CTUe	USETox v1.04 [34][a]
Resource depletion (RD)	kg Sb eq	CML-IA [22]

CTUh : comparative toxic unit for human toxicity impacts; CTUe : comparative toxic unit for aquatic ecotoxicity impacts; NMVOC: non-methane volatile organic compounds.

[a] Recommended characterization factors only as default; Recommended and interim characterization factors were used in the sensitivity analysis presented in section 3.2.1.

Table 6 – Comparison of LCA results (per ton of LWA) for LWA produced with (APCr-LWA) and without (T-LWA) APCr.

Impact category	Unit	T-LWA (X)	APCr-LWA (Y)	Savings (Y-X)/X×100
Climate change (CC)	kg CO_2 eq	434	415	-4,5%
Ozone depletion (OD)	kg CFC-11 eq	7,24E-06	6,37E-06	-11,9%
Human toxicity, cancer effects (HTc)	CTUh	3,08E-09	1,80E-09	-41,6%
Human toxicity, non-cancer effects (HTnc)	CTUh	1,94E-10	1,42E-10	-26,8%
Particulate matter (PM)	kg PM2.5 eq	2,09E-01	2,04E-01	-2,8%
Photochemical ozone formation (PO)	kg NMVOC eq	1,43	1,38	-3,2%
Acidification (AC)	mol_c H^+ eq	3,19	3,13	-1,9%
Terrestrial eutrophication (TE)	mol_c N eq	4,89	4,71	-3,6%
Marine eutrophication (ME)	kg N eq	0,48	0,46	-3,8%
Freshwater ecotoxicity (FE)	CTUe	0,22	0,20	-9,6%
Mineral, fossil & ren resource depletion (RD)	kg Sb eq	8,62E-04	7,14E-04	-17,1%

Table 7 – Comparison of LCA results (per ton of LWA) for LWA produced with (APCr-LWA) and without (T-LWA) APCr for toxicity categories considering both recommended and interim CFs.

Impact category	Unit	T-LWA (X)	APCr-LWA (Y)	Savings (Y-X)/X×100
Human toxicity, cancer effects (HTc)	CTUh	1,78E-06	-1,94E-05	-1191%
Human toxicity, non-cancer effects (HTnc)	CTUh	6,77E-06	-2,32E-05	-442%
Freshwater ecotoxicity (FE)	CTUe	321	-2327	-825%

Table 8- Comparison of leaching (mg/kg) from T-LWA and APCr-LWA through equilibrium test data.

	Potentially toxic metals						Elements associated with soluble salts			
	Pb	Cd	Zn	Cr	Ni	Cu	K	Na	Ca	Cl^-
T-LWA[(a)]	<0.9	<0.2	<0.1	<1.0	<1.4	<0.9	41	30	201	43
APCr-LWA[(a)]	<0.9	<0.2	<0.1	<1.0	<1.4	<0.9	54	11	779	44
WAC inert landfill(b)	0.5	0.04	4	0.5	0.4	2	nd	nd	nd	800
WAC NH landfill(c)	10	1.0	50	10	10	50	nd	nd	nd	15000

[(a)] - Data from [12]; [(b)] - waste acceptance criteria in the inert landfill, defined in Council Decision 2003/33/EC; [(c)] - waste acceptance criteria in the non-hazardous landfill, defined in Council Decision 2003/33/EC; nd- not defined.