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Life cycle assessment of concrete with wind turbine blade waste: A real case study

Javier Manso-Morato [a], Nerea Hurtado-Alonso [b], Marta Skaf [b], Víctor Revilla-Cuesta [a], Vanesa Ortega-López [a,*]

[a] Department of Civil Engineering, Escuela Politécnica Superior, University of Burgos, c/ Villadiego s/n, 09001 Burgos, Spain
[b] Department of Construction, Escuela Politécnica Superior, University of Burgos, c/ Villadiego s/n, 09001 Burgos, Spain

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ABSTRACT

Europe is among the most important wind-energy producers in the world, yet a commonly accepted solution is to be found towards Wind Turbine Blade Waste (WTBW) from wind turbine blades at the end of their lifespan. In this research, Life Cycle Assessment (LCA), regarding Global Warming Potential (GWP) and Abiotic Depletion Potential for fossil fuels (ADPf), was first used to study landfilling, incineration and mechanical recycling of WTBW. Mechanical recycling was highlighted as the best option, as incineration showed higher impacts (x4.5 GWP, x1.2 ADPf) and landfilling is forbidden by the European Union. Afterwards, WTBW management was combined with concrete production by considering both incineration and mechanical recycling, WTBW being used as aggregate replacement (2%, 5%, and 10% vol.) to create fiber-reinforced concrete. Mechanical recycling of WTBW always yielded lower results (-28.3% GWP, -5.9% ADPf), even when including larger transportation impacts in a real case in *Castilla y León* (-28.0% for GWP, -5.4% for ADPf), a region in Spain among the top producers of WTBW in Europe in the next 5-10 years, which is in need for a recycling strategy to follow. Lastly, four mechanical recycling plants would be needed in *Castilla y León* to minimize WTBW transportation impacts, thus the average environmental damage being reduced by 0.2% GWP and 0.3% ADPf per cubic meter of ready-to-cast concrete. These key findings emphasize the benefits of mechanically recycling WTBW and its potential when combined with concrete production through LCA, yielding promising results that can be implemented in different regions around the world.

1. Introduction: Current situation of the wind energy sector

As society is steadily developing, energy demands are arising, and the need for sustainable and renewable energy sources keeps on growing (Ecer, 2021). The wind-energy sector is a powerful ally towards responsible consumption and production of energy (Asociación Empresarial Eólica, 2024; United Nations, 2023), and also helps to reduce the price of the energy for the consumers by up to 19% in some countries (Asociación Empresarial Eólica, 2024). Wind energy is affordable, reliable, locally produced, scalable, emits zero carbon and consumes negligible amounts of water, while paying off its life-cycle emissions before the first year of operation (Nan Cong et al., 2023; WindEurope, 2024).

Nowadays, wind plays one of the main roles for energy production worldwide. Currently, the global installed power for wind-energy production is 1,020.7 GW, with over 10% of this power installed just in 2023 (Asociación Empresarial Eólica, 2024). Furthermore, it is destined to be the backbone of Europe's energy production system, expected to be the first source of power in Europe by 2027 (International Energy Agency, 2024; WindEurope, 2023). Therefore, rapid growth and expansion in following years are projected (Rebolo-Ifrán et al., 2025). According to Asociación Empresarial Eólica (2024), Spain ranks fifth in the world in terms of global installed power and second in Europe, accounting 30.43 GW of installed capacity. In this country there are a total of 22,210 wind turbines in 1,371 wind farms along 1,053 different municipalities in 16 regions. Electricity produced by wind powers around 17 million Spanish homes and provides a salary for over 39,000 people (Asociación Empresarial Eólica, 2024). However, in Spain, only around 2% of the total installed power was added in 2023, making its fleet rather old in comparison to the global and European values (Asociación Empresarial Eólica, 2024).

The expected lifespan of a wind turbine is 20-25 years (Beauson and

* Corresponding author.
E-mail address: vortega@ubu.es (V. Ortega-López).

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Brøndsted, 2016; Gaertner et al., 2020; IEC, 2019; Jiang et al., 2022; Zhang et al., 2023a), although in some particular occasions it can be extended up to 40 years, if the logistics and economic issues related to remanufacturing and reuse can be tackled directly (Ortegon et al., 2013). A high number of the components of wind turbines, can be either repurposed or easily recycled, as up to 80% wt. of the turbines is made out of mechanical components or steel (Baturkin et al., 2021; Ozment and Tremwell, 2007). Regarding the decommissioned blades, they have to be first weighed and inspected to assess their final state (Skrainka, 2012), to then decide if these elements can be reused elsewhere with lower standards (usually third-world countries) or must be recycled as they can no longer be used safely, becoming a proven sustainability blind spot of these wind-energy systems (Beauson et al., 2022; Mishnaevsky, 2021; Revilla-Cuesta et al., 2024b; Sakellariou, 2018), with many scholars addressing this issue (Li et al., 2023). On the one hand, when recycling these blades, their complex composition generally made out of Glass Fiber-Reinforced Polymer (GFRP), polymeric particles, balsa wood and resins (Nagle et al., 2020; Revilla-Cuesta et al., 2023) makes them impossible to melt, remold or degrade (Beauson and Brøndsted, 2016). On the other hand, the lack of official regulations for recycling or disposal of these end-of-life blades in some countries such as Spain (Ziegler et al., 2018) makes these issues very difficult to deal with. Spain will be one of Europe's top producers of Wind Turbine Blade Waste (WTBW) in the upcoming years, with around 14-17% of Europe's onshore waste production, with the country's central regions being the top producers among the whole continent, according to the analysis performed by Lichtenegger et al. (2020). In addition, offshore wind farms are to be also included, being crucial towards global decarbonization targets (Abramic et al., 2022).

2. Brief literature review: Blade recycling and life cycle assessment

The decommissioned blades have been commonly landfilled or incinerated in Europe (Corinaldesi et al., 2015; Ribeiro et al., 2015), as they were the easiest and fastest alternatives, yet it was not sustainable at all (Gopalraj et al., 2021). In the last years, landfilling of these elements has been prohibited and deemed illegal in the European Union (Gharfalkar et al., 2015). New ways to reuse or recycle WTBW have therefore to be sought (Cooperman et al., 2021; Ramirez-Tejeda et al., 2017; Rani et al., 2021), which constitutes one of the most important challenges that is being studied at the moment (Beauson et al., 2022; Cooperman et al., 2021).

WTBW can be recycled by either thermal, chemical, or mechanical treatments. Thermal recycling is based on using high temperature to decompose the resin in the blades to recover fiberglass. Nevertheless, it is high-energy demanding, and emits pollutants during the process, showing also negative impacts on the quality of the retrieved material (N Cong et al., 2023). Chemical recycling relies on solvents to separate the different blade components and recover fiberglass, but it produces hazardous residues. It also worsens the mechanical performance of the recovered material (Sorte et al., 2023). Lastly, mechanical recycling consists of shredding and grinding the WTBW to different sizes and shapes, in order to incorporate it into other materials in the form of powders, fibers or needles (Beauson et al., 2016; Palmer, 2009). In fact, mechanical recycling is one of the most promising options for end-of-life WTBW, with low associated energy consumption and no secondary residues being produced (Liu et al., 2019).

The elements from the mechanical recycling of WTBW can be added into concrete mixes in different ways, helping reduce their high carbon footprint, providing WTBW with a second life without generating any other by-products (N Cong et al., 2023; Revilla-Cuesta et al., 2024a; Yazdanbakhsh et al., 2017), thus avoiding landfilling and incineration (Khalid et al., 2023). Some approaches include their addition as powders, aggregates or fibers (Xu et al., 2024), in turn reducing the need for natural raw materials while maintaining or even enhancing the mechanical performance of the final concrete (Baturkin et al., 2021; Hasheminezhad et al., 2024a; Revilla-Cuesta et al., 2023). Also, the chemical composition of WTBW, rich in SiO_2, Al_2O_3, and CaO, allows producing supplementary cementitious materials based on these wastes, reducing the need for other non-sustainable binders, posing both environmental and economic benefits (Zhang et al., 2023b).

However, concrete is one of the highest pollutants due to the use of Portland cement. Therefore, it is important to assess through Life Cycle Assessment (LCA) the environmental burdens associated to this alternative for WTBW recycling, to assess the sustainability of the whole process (Manso-Morato et al., 2024b). This kind of analysis has been successfully conducted to evaluate the validity of incorporating recycled and upcycled discarded materials from different industries and origins, and their applications towards construction and infrastructure development (Milad, 2025). Some examples include the incorporation of recycled aggregates to lower both energy consumption and CO_2 emissions (Hasheminezhad et al., 2024b; Neupane et al., 2025); using sustainable supplementary cementitious materials such as fly ash and slags to reduce the need for cement (Radwan et al., 2021; Thorne et al., 2024); or the incorporation natural fibers or fibers from other by-products to give them a second-life and avoid landfilling (Balea et al., 2021; García et al., 2024).

The relevance of LCA analyses even goes a step further. The environmental impacts of WTBW disposal or recycling have been traditionally ignored (Alsaleh and Sattler, 2019), because these environmental burdens were low for a single turbine in comparison to the tower or nacelle (Alsaleh and Sattler, 2019; Baturkin et al., 2021; Ozment and Tremwell, 2007). However, the number of wind turbine blades to be dismantled in the next 10 years accounts up to around 77,000 units in Europe, making up for over 200 ktons of WTBW (Beauson and Brøndsted, 2016; Liu and Barlow, 2017). As a conclusion, it is safe to say that the need for LCA in terms of WTBW recycling or disposal grows to ensure that the best option environmentally can be chosen with full knowledge of its advantages and disadvantages.

In this research, the environmental impacts through LCA regarding climate change and use of fossil fuels of the end-of-life management of WTBW in the Spanish region with the greatest need to recycle WTBW in the next few years will be studied, compared and optimized. This innovative research line is highlighted by environmentally assessing a sustainable Fiber-Reinforced Concrete (FRC) that has already been proven to show proper mechanical and durability behavior for a wide variety of applications in building and civil engineering (Manso-Morato et al., 2025; Revilla-Cuesta et al., 2024a). Meanwhile, the literature gap regarding GFRP recycling methodology and, therefore, WTBW, is filled (Gopalraj et al., 2021). In addition, this study provides a full environmental description of the most common processes of GFRP and WTBW management (landfilling, thermal, and mechanical recycling), and establishes a novel methodology for transportation-impact reduction, that can be used in any problematic area with a high WTBW-production rate. Through these environmental analyses, conclusions are drawn regarding which recycling/disposal method yields best environmental results and ensures sustainability, combined with the main regulations for WTBW management in Europe.

3. Geographical framework

This research work takes place in Spain, focused on one of its 17 autonomous regions called *Castilla y León*. It is in the North-West of the Iberian Peninsula, with an area of around one fifth of the whole country, *i.e.*, 95,000 km^2 (del Río et al., 2005). This region is the largest one in Spain and the third one in Europe (Garrote et al., 2020; Junta de Castilla y León, 2024), and it is made of 9 provinces, detailed in Table 1, along with relevant data about their existing windfarms.

This region is mainly an elevated plateau framed by important mountain ranges, with an average altitude above sea level of 830 meters (del Río et al., 2005; Junta de Castilla y León, 2024). This region of Spain

Table 1
Wind sector data in *Castilla y León* (EREN, 2024).

PROVINCE	INSTALLED POWER (MW)	NUMBER OF WIND TURBINES	AVERAGE AGE (YEARS)	PERCENTAGE OF WIND FARMS OVER 15 YEARS (%)	NUMBER OF BLADES TO DISMANTLE
ÁVILA (AV)	260.68	281	18.40	60	561
BURGOS (BU)	2129.58	1,428	14.50	56	2,883
LEÓN (LE)	443.75	300	14.00	41	573
PALENCIA (PA)	918.00	560	14.28	50	1,023
SALAMANCA (SA)	192.64	118	11.50	25	132
SEGOVIA (SG)	75.72	78	20.33	100	234
SORIA (SO)	1249.47	972	17.05	74	2,391
VALLADOLID (VA)	755.70	310	6.32	9	141
ZAMORA (ZA)	612.11	515	17.18	79	1,428
TOTAL	6637.64	4,562	14.84	55	9,366

is extensive, and its population density is quite low, which allows for large open spaces that can be used for different purposes (Junta de Castilla y León, 2024). Therefore, *Castilla y León* has favorable wind conditions combined with vast areas, which makes it a perfect place to allocate wind farms (Iberdrola España, 2024). This region is at the top of Spain's total wind power installed, with over 260 different wind farms and 6.6 GW alone, and it generates 23% of the national production (Asociación Empresarial Eólica, 2024; EREN, 2024; Iberdrola España, 2024). Around 65% of wind farms are located in the North-East of the region (Burgos, Soria, and Palencia), shown in Fig. 1.

Between 1998 and 2013, there was an important growth in wind power capacity in this region, which led *Castilla y León* to its current position (Matti et al., 2017). Therefore, most of its installed wind farms are currently approaching the average lifespan of around 20-25 years (Beauson and Brøndsted, 2016; Gaertner et al., 2020; IEC, 2019), the average age of these wind farms in *Castilla y León* being 14.84 years (Table 1). This means that 145 wind farms will need to be decommissioned in the next 5-10 years. In Table 1, data about the wind farms to be decommissioned per province is also collected and displayed, with over 9,300 blades to be managed shortly in their end-of-life scenario (68% of the total), mainly located in Burgos, Soria, Zamora and Palencia (Fig. 2).

According to all the above, the need for recycling strategies and their environmental analysis has been proven to be crucial in this region, being important not only locally but also in a much higher scale. By finding a solution in this problematic region, a proper management option could be extrapolated to other areas, being able to find the best environmental strategy to deal with this worldwide issue.

4. Methodology

The methodology used in this research is summarized in Fig. 3. In order to conduct a proper LCA, the applicable regulations were addressed, such as ISO 14040/44 (ISO, 2017), EN 15804 and EN 15978 (EN-Euronorm, 2020), to adequately choose the correct scopes and system boundaries for each scenario. SimaPro v9 (Database and Support Teams at PRé Sustainability, 2023) was used to calculate the environmental impacts, supported by the Ecoinvent v3 database (Ecoinvent Centre, 2023) and following the CML-IA Baseline methodology (CML - Department of Industrial Ecology, 2016). Specific data regarding the Life Cycle Inventory (LCI) used in the present study can be found in the Supplementary Material. Among the different LCA environmental indicators, Global Warming Potential (GWP) and Abiotic Depletion Potential of fossil fuels (ADPf) were chosen. They are the most usual in the existing literature regarding GFRP (Gopalraj et al., 2021; Pillain et al., 2017), and are commonly applied in the environmental assessment of concrete (Manso-Morato et al., 2024b). These indicators refer to two of the main issues of today's world: emissions of greenhouse gases to the Earth's atmosphere (GWP) and the overutilization of a rather scarce and non-renewable material such as fossil fuels (ADPf) (Frischknecht et al.,

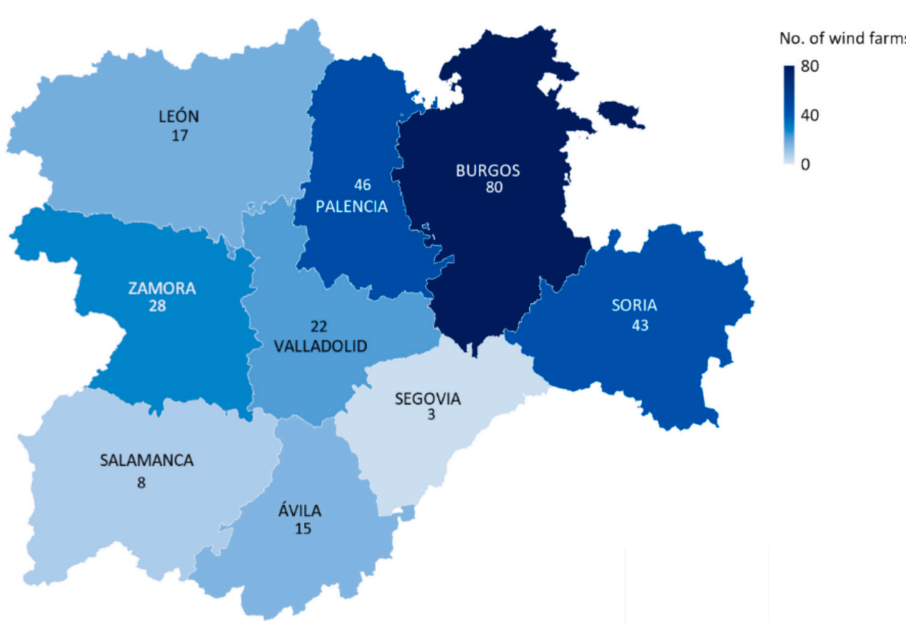

Fig. 1. Heatmap of existing wind farms per province in *Castilla y León* (EREN, 2024).

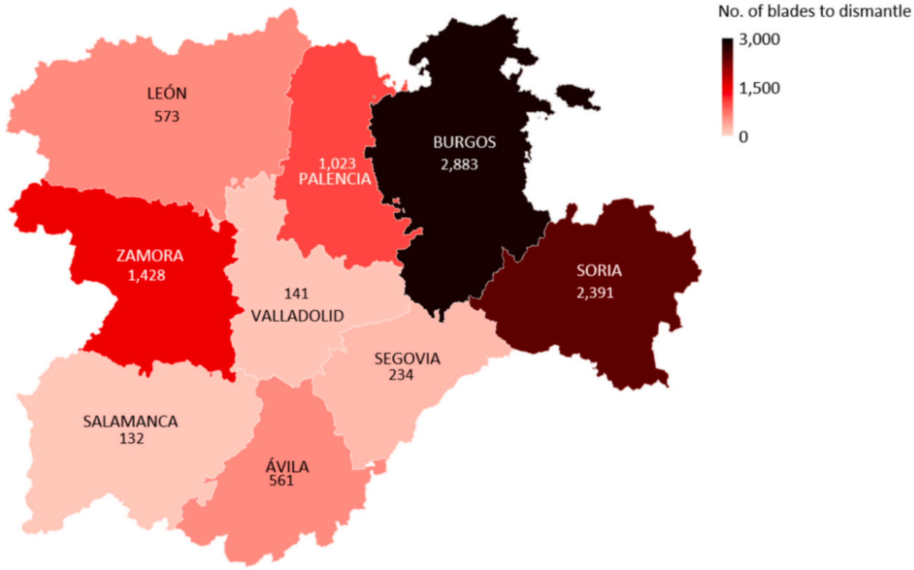

Fig. 2. Heatmap of blades to be dismantled in the next 5-10 years per province in *Castilla y León* (EREN, 2024).

2015).

A cradle-to-gate approach was taken in order to assess the environmental impacts of WTBW management according to the most common practices in the field, including energy and resource consumption during the processing of this waste at treatment plant level. Then, the different management available options according to the regulations were applied in combination with concrete production, with two different approaches: first, concrete-manufacturing plant and waste-treatment plant (A1-A3); and second, adding the on-site installation, therefore including the environmental burdens of transportation (A1-A5) (EN-Euronorm, 2020). Lastly, as these transportation impacts can represent a high environmental impact due to the high volume of concrete and WTBW to be produced and managed, they were minimized in Castilla y León. Further details of all the cases analyzed can be found in the next sections of the article. As a final stage towards LCA validity and robustness, a sensitivity analysis and threshold and range calculations were also performed.

5. Environmental impact of blade waste

A cradle-to-gate system boundary (A1-A3) (Xia et al., 2020) was chosen, as it is thought to calculate the impact of recycling/disposal of the WTBW right at the management center, which is a common approach for waste materials (Manso-Morato et al., 2024b). Furthermore, the chosen Functional Unit (FU) in this analysis was one kilogram of already managed WTBW, which will have undergone different treatments at its end-of-life stage. By choosing this FU, all variations of sizes and origin of the different blades (Mishnaevsky et al., 2017; Revilla-Cuesta et al., 2023) can be avoided.

5.1. Scenario formulations

The first scenario under analysis is the LandFilling (LF) of 1 kg WTBW. Due to current regulations (European Parliament and Council, 2008; Gharfalkar et al., 2015), the recycling of the blades is desired, and landfilling is either prohibited or suffers heavy taxation, depending on the country. Besides, it requires large areas that cannot be destined for any other use for long periods of time (Gopalraj et al., 2021). Consequently, landfilling WTBW is not a viable long-term solution, yet it is included in the current section for comparison's sake, and to gather information about the environmental impacts that 1 kg of WTBW has been producing in the last years.

The second scenario of analysis is the INcineration (IN) of 1 kg WTBW. This methodology is often used for WTBW management in Europe (Sakellariou, 2018), and can be used with or without energy recovery (European Composites Industry Association, 2011; Gopalraj et al., 2021; Nagle et al., 2020). In the present study, incineration is considered without energy recovery, as glass fiber, balsa wood and the polymeric particles present in the blades do not have a high calorific value (Beauson et al., 2022). Also, no further treatment of the remaining ashes from the incineration process has been considered, which can lead to even higher environmental impacts (Beauson et al., 2022; Gopalraj et al., 2021).

The third scenario addressed is the production of 1 kg of WTBW through Mechanical Recycling (MR) of the end-of-life blade as a whole. The WTBW undergoes a process of non-selective crushing of the blade as a whole, yielding Raw-Crushed Wind-Turbine Blade (RCWTB) through knife milling and sieving. RCWTB consists in a mixture of GFRP fibers and microfibers, small semi-spherical particles of balsa wood and polymers, and non-separable particles in the shape of a fluff. RCWTB has been studied in further detail in a previous work by the authors (Revilla-Cuesta et al., 2023).

5.2. Life cycle inventory

LCI requires data collection of the inputs/outputs of the system under study, such as energy, raw materials, products, produced waste, and emissions, among others (ISO, 2017). As mentioned above, information on GFRP treatment and recycling is not common at all, thus being hard to find the necessary data in the available databases or literature (Gopalraj et al., 2021). The composition of the WTBW under study per FU can be seen in Table 2, following field studies by the authors (Revilla-Cuesta et al., 2023). The GFRP was considered to be made out of 55% wt. virgin glass fibers and 45% wt. thermoset resins (Gopalraj et al., 2021; Vita et al., 2019).

5.2.1. Scenario: Waste disposal by landfilling

No specific environmental data has been found for blade landfilling, just of the GFRP part (Gopalraj et al., 2021). Therefore, this scenario has been addressed by the sum of impacts of landfilling each component of WTBW, detailed in Table 2, considering their location in Europe (Ecoinvent Centre, 2023; Wernet et al., 2016).

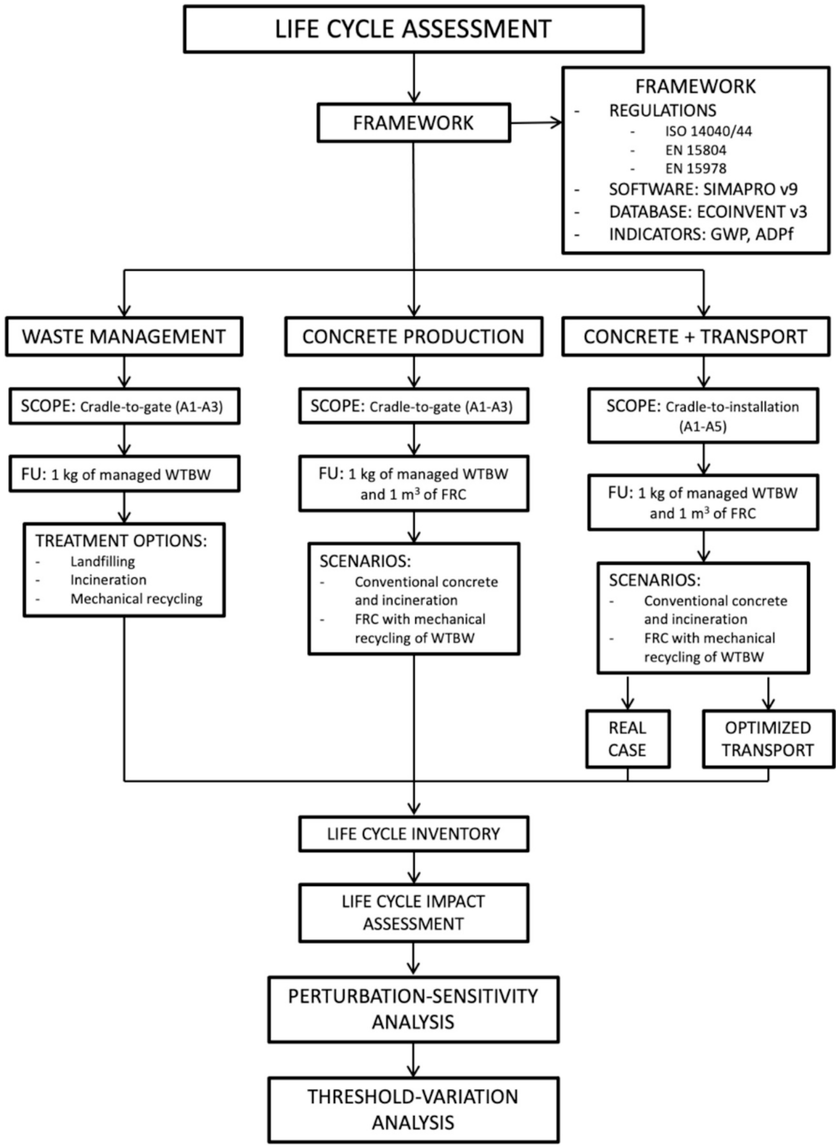

Fig. 3. Flowchart of the methodology of the study.

Table 2
Composition by mass of a FU (1 kg of WTBW).

Material	Glass fiber	Resin	Polymeric particles	Balsa wood
Mass (kg)	0.4646	0.3801	0.0870	0.0683

5.2.2. Scenario: Waste disposal by inceration

No specific data regarding the incineration of WTBW was found either. Some LCA information was found regarding incineration of GFRP, but it was modelled as "plastic in a waste incineration plant" with energy recovery (Gopalraj et al., 2021), which was not as accurate as desired for this research. Thus, incineration without energy recovery in a municipal treatment plant was modelled in the LCA as the addition of the environmental impacts of the components of the WTBW according to the composition of the FU (Table 2). The resins were modelled as hazardous waste for incineration, due to the chemical emissions that take place in this process (Ecoinvent Centre, 2023; Wernet et al., 2016).

5.2.3. Scenario: Waste mechanical recycling

Mechanical recycling creates an all-in-one composite in an effective way, as no by-products are produced (Revilla-Cuesta et al., 2023). Data regarding the mechanical recycling of the whole blades are unavailable, and data on GFRP mechanical recycling is scarce as well. Thus, the process addressed was the mechanical recycling of WTBW into RCWTB (Revilla-Cuesta et al., 2024b; Revilla-Cuesta et al., 2023). The process from blade decommissioning to RCWTB production has been already researched by the authors (Revilla-Cuesta et al., 2023): the decommissioned blade is divided into one-meter segments and then cut into regular pieces (20-30 cm), which are then introduced into a knife mill for crushing and sieving to produce fragments smaller than 10 mm (Revilla-Cuesta et al., 2023). All energy inputs for these processes were taken from the local electricity network and modelled as the market for medium voltage electricity in Spain (Ecoinvent Centre, 2023; Wernet et al., 2016). The process (performance ratio of machinery, method and energy consumption) is detailed in Fig. 4.

5.3. Life cycle impact assessment and interpretation

The results of the LCA in terms of GWP (kgCO_2-e) and ADPf (MJ) can be seen in Table 3 and Table 4, respectively.

Both in GWP and ADPf terms, landfilling reached the lowest impacts, followed by mechanical recycling and incineration. The results for GFRP

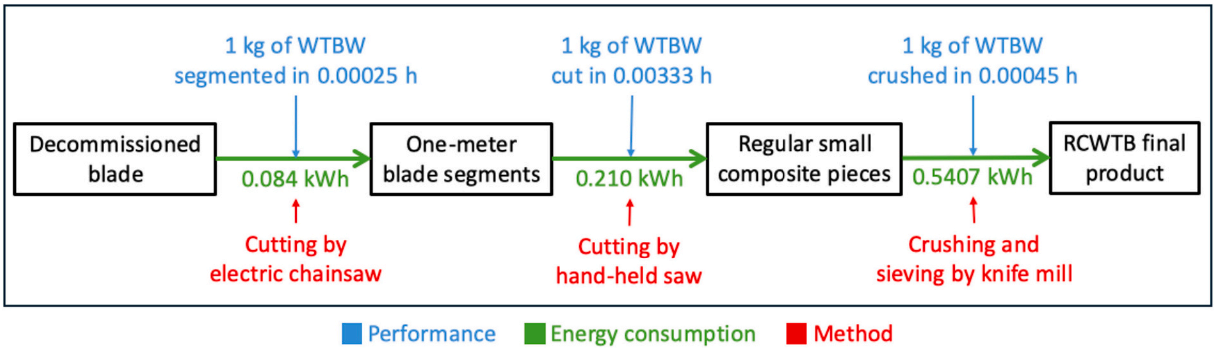

Fig. 4. Process of WTBW mechanical recycling, including performance ratio of the machinery, method, and energy consumption.

Table 3
GWP (kgCO$_2$-e) for landfilling, incineration and mechanical recycling of 1 kg of WTBW.

Material	Landfilling	Incineration	Mechanical recycling
GFRP	0.0676	0.9661	-
Balsa wood	0.0052	0.0009	-
Polymeric particles	0.0083	0.2130	-
TOTAL	0.081	1.1800	0.2680

Table 4
ADPf (MJ) for landfilling, incineration and mechanical recycling of 1 kg of WTBW.

Material	Landfilling	Incineration	Mechanical recycling
GFRP	0.9630	3.6015	-
Balsa wood	0.0167	0.0085	-
Polymeric particles	0.0196	0.0600	-
TOTAL	0.9993	3.6700	3.0700

recycling were in all cases in accordance with the existing bibliography (Gopalraj et al., 2021). Impacts in terms of GWP were about 15 times higher for incineration than landfilling, and about 3 times higher for mechanical recycling (Table 3). ADPf of incineration represented about 4 times the value of landfilling, and mechanical recycling about 3 times (Table 4).

Landfilling had the lowest environmental impact as the treatment of WTBW is not carried out fully, which can result in methane and volatile-organic-compound emissions in the long term that LCA does not consider in the present stage. Besides, this procedure does not follow European directives towards recycling (Cooperman et al., 2021; Ramirez-Tejeda et al., 2017; Rani et al., 2021), and therefore it is out of the table for long-term WTBW management. Consequently, for further analysis in the following sections, landfilling will not be considered as an option, leaving mechanical recycling and incineration as the viable solutions to be explored.

Regarding the two other methods, mechanical recycling accounted for 22.7% of the GWP of incineration (Table 3), and for 83.6% of its ADPf (Table 4). GFRP and polymers accounted for 81.9% and 18.1% of the GWP impact, respectively, in WTBW incineration (Table 3), while GFRP represented a staggering 98.1% of the ADPf during incineration (Table 4). Besides, the impacts from the treatment of the ashes related to WTBW incineration (Beauson et al., 2022) were not considered in this LCA, which made it even more undesirable as a management route for WTBW (Gopalraj et al., 2021). Therefore, mechanical recycling can be deemed as the best environmental alternative for long-term management of end-of-life blades, showing the least environmental impacts and resulting in no further treatment as no by-products are created.

6. Joint environmental impact of concrete and blade waste

Concrete industry is one of the most polluting sectors worldwide, as it is responsible for a great deal of emissions (ANEFHOP, 2022; GCCA, 2021; Kirthika et al., 2020; Xing et al., 2023). The addition of wastes from other industries to concrete helps reduce the quantity of raw materials needed and in turn the emission of greenhouse gases (Rahimpour et al., 2024; Soltanzadeh et al., 2022), apart from giving a second life to waste materials (Revilla-Cuesta et al., 2024b).

6.1. Scope and goals

In this study, incineration and mechanical recycling are explored in conjunction with concrete production. The first scenario considered the joint impacts of the incineration of WTBW and the production of conventional concrete, in order to understand the environmental output of not giving a second life to WTBW and the inevitable production of concrete due to society's needs. The second scenario involves mechanically recycled WTBW in the form of RCWTB used to produce FRC, adding this waste as a percentage in volume of the aggregates (% vol. agg.) (Revilla-Cuesta et al., 2023), to show its potential benefits by comparing total environmental burdens.

Three amounts of already managed WTBW, replacing aggregates, were studied: 2%, 5%, and 10% vol. agg. of conventional concrete, in accordance with the usual percentages of fibers used in FRC production (Manso-Morato et al., 2024b; Manso-Morato et al., 2024a). When WTBW incineration was simulated, the environmental damage of conventional concrete production and that of the incineration of such amounts of WTBW being added. The environmental impact of concrete produced with those amounts of WTBW was calculated when employing mechanically recycled WTBW as a raw material in concrete.

6.2. Impact of conventional concrete production

For the LCA on conventional concrete, these environmental impacts were calculated in previous research (Manso-Morato et al., 2025). Concrete production was conducted in a vertical-axis mixer, which used electricity from the local network supply, and lasted 15 minutes. All material and energy inputs for the LCA were based on these indications (Ecoinvent Centre, 2023; Wernet et al., 2016). In accordance with existing literature, the FU for this LCA was one cubic meter of concrete (1 m^3) (Acosta-Calderon et al., 2022; Frazão et al., 2022).

In terms of GWP, the production of conventional concrete accounted for 286.18 kgCO$_2$-e, from which cement had the highest contribution (around 85%), followed by aggregate production (around 10% of the total GWP impact). These findings were in accordance with previous work (Manso-Morato et al., 2025; Revilla-Cuesta et al., 2024b; Revilla-Cuesta et al., 2023), where concrete with similar dosage incorporating CEM I represented around 320 kgCO$_2$-e/m^3 (Hafez et al., 2019). On the other hand, ADPf of conventional concrete production yielded 1,819.37

MJ/m^3, from which cement production represented around 66%, and aggregate extraction 18%.

6.3. Impacts without transport

The FU that better represented the study was the combination of one cubic meter of conventional concrete and one kilogram of already managed WTBW. In the first scenario, concrete production and WTBW incineration (C+IN), WTBW was not introduced into the concrete mix. Therefore, the impact of concrete production was considered to be constant and the impacts of incinerating WTBW were added. In the second scenario, mechanically recycled WTBW was incorporated (C+MR), replacing the aggregates of the mix by 2%, 5% and 10% vol. agg. Such concrete mixes were evaluated in previous research of the authors (Manso-Morato et al., 2025). The final dosages adjusted to one cubic meter for all mixes can be seen in Table 5.

6.3.1. Life cycle impact assessment and interpretation

The results obtained from the LCA can be seen in Table 6. All the impacts were lower when incorporating mechanically recycled WTBW into the concrete mix, as this process is less environmentally damaging than incineration (Section 5.3). Also, the quantity of aggregates added as raw materials into the concrete mixes was reduced, as well as those of other raw materials due to the high volume and low density of the mechanically recycled WTBW. However, the impacts were not linearly reduced with higher amounts of mechanically recycled WTBW, because slightly higher amounts of water and admixtures had to be incorporated to achieve proper fresh characteristics (EN-Euronorm, 2020). Finally, the higher the amount of WTBW that had to be managed, the greater the difference between both scenarios.

On the one hand, the GWP of both scenarios showed increasing differences with the quantity of WTBW to be managed. The reduction from C+IN to C+MR was 17.3% when considering 2% vol. agg. WTBW, while this reduction was 28.3% for 10% vol. agg. WTBW. Besides, the GWP of mechanically recycling WTBW was around one fifth of that of WTBW incineration. Finally, the reductions in GWP due to the lower consumption of raw materials in the C+MR scenario were 0.2% and 3.0% for 2% and 10% vol. agg. WTBW, respectively. Aggregates and cement manufacturing represented around 96% of the GWP of concrete production, so changes in these quantities had relevant effects in the overall impact.

On the other hand, the difference was less noticeable in terms of ADPf. The C+MR scenario represented a reduction of 5.9% compared to the C+IN scenario for 10% vol. agg. WTBW. The reductions from the C+IN to the C+MR scenario were 4.0% and 1.2% for 5% and for 2% vol. agg. WTBW, respectively. These trends could be caused because cement and aggregates only represented 85% of the total ADPf of concrete production, and their ADPf decrease for C+MR when considering 10% vol. agg. WTBW was only 16.4% compared to C+IN. All these percentage contributions were less than for GWP.

Table 6
Environmental impacts per scenario (C+IN *vs.* C+MR) and WTBW amount to be managed (cradle-to-gate A1-A3).

ENVIRONMENTAL IMPACT		GWP (kgCO2-e)			ADPf (MJ)		
WTBW AMOUNT (% vol. agg)		2	5	10	2	5	10
C+IN	CONCRETE	286	286	286	1819	1819	1819
	INCINERATION OF WTBW	29	73	147	91	228	456
	TOTAL	316	360	433	1911	2047	2276
C+MR	CONCRETE	286	281	277	1811	1775	1759
	MECHANICAL RECYCLING OF WTBW	7	17	33	76	191	382
	TOTAL	292	297	310	1887	1966	2140

6.4. Impacts with transport: A real case

6.4.1. Functional unit and scenario formulations

The next step in the LCA evaluation was to adopt a cradle-to-installation approach (EN-Euronorm, 2020), which added transportation impacts, as road transportation is one of the main contributors to GWP and ADPf (Khanna et al., 2025). The chosen FU was the same as in Section 6.3: the combination of one cubic meter of conventional concrete and one kilogram of already managed WTBW, but ready to be casted at a construction site.

In order to analyze a real case, the wind farm selected was the last one dismantled near the city of Burgos in 2024, where the researchers are located. This Wind Farm (WF) was in the municipality of *Unzúe*, in the autonomous region of *Navarra*. Then, the two closest locations to this WF available for incineration and mechanical recycling were chosen. For incineration (scenario C+IN+T), the closest incineration plant to this WF was located in the municipality of *Miranda de Ebro*, 107.40 km away from the WF. Mechanical recycling (scenario C+MR+T) was to be done in Burgos, 174.23 km away from the WF. Concrete was assumed to be produced in the same municipality as the treatment plant in both scenarios, therefore not adding further significant transportation impacts.

6.4.2. Life cycle inventory

All raw materials and energy inputs from Section 6.3 were used. The environmental impact of logistics to carry the WTBW to each treatment plant was modeled by a diesel truck with a payload capacity under 32 metric tons (EURO 4), according to the Ecoinvent v3 database (Ecoinvent Centre, 2023; Wernet et al., 2016). This impact was calculated per kilogram of WTBW and per kilometer of transport, to be in accordance with the FU adopted in the present study and to later be multiplied by the transport distance in each scenario.

6.4.3. Life cycle impact assessment and interpretation

The results of this real case LCA can be seen in Table 7. Logically, all impacts grew as higher amounts of WTBW were managed, due to the higher transportation burdens, being them calculated as kg·km. However, a noticeable difference between C+IN+T and C+MR+T scenarios was noted: mechanical recycling always yielded lower impacts for all considered percentages of WTBW, even though the mechanical recycling plant was 66.83 km further away than the incineration plant.

First, in terms of GWP, the C+MR+T scenario led to a reduction of up to 28.0% compared to C+IN+T when considering 10% vol. WTBW. As can be seen in Table 7, the GWP contribution of the incineration of 10% vol. WTBW accounted for around 34% of the total impacts, while mechanical recycling only represented around 11%.

Conversely, transportation impacts were more important for ADPf than for GWP, as fossil fuels are the main source of energy for logistics. These transportation burdens were 1.62 times higher for C+MR+T than

Table 5
Dosage (kg/m^3) of concrete mixes incorporating different amounts of WTBW.

WTBW (% vol. agg.)	Cement	Water	Plasticizers	Gravel 12/22	Gravel 4/12	Sand 0/4	Sand 0/2	WTBW
2	320	128	3.2	764	544	377	271	25
5	315	140	3.2	730	520	361	259	61
10	312	149	3.9	684	487	338	242	121

Table 7
Environmental impacts per scenario (C+IN+T vs. C+MR+T) and WTBW amount to be managed (cradle-to-installation A1-A5).

ENVIRONMENTAL IMPACT		GWP ($kgCO_2$-e)			ADPf (MJ)		
WTBW AMOUNT (% vol. agg)		2	5	10	2	5	10
C+IN+T	CONCRETE	286.18	286.18	286.18	1,819.37	1,819.37	1,819.37
	WTBW INCINERATION	29.33	73.34	146.67	91.24	228.09	456.18
	TRANSPORT	0.24	0.60	1.19	3.70	9.25	18.50
	TOTAL	315.75	360.11	434.05	1,914.31	2,056.71	2,294.06
C+MR+T	CONCRETE	285.51	280.73	277.17	1,810.54	1,774.90	1,758.81
	WTBW MECHANICAL RECYCLING	6.66	16.66	33.31	76.32	190.80	381.60
	TRANSPORT	0.39	0.97	1.93	6.00	15.01	30.01
	TOTAL	292.56	298.35	312.42	1,892.86	1,980.71	2,170.42

for C+IN+T when considering 10% vol. agg., but it was compensated for by the environmental difference between both treatments. C+MR+T represented a 1.1% reduction for 2% vol. agg. WTBW, 3.7% reduction for 5% vol. agg., and 5.4% reduction for 10% vol. agg. compared to the C+IN+T scenario.

Therefore, the difference between both scenarios was large, even for the small-scale production analyzed (one cubic meter of concrete and management of only one kilogram of WTBW). If the huge amounts of concrete to be produced and WTBW to be managed in the following years were accounted (GCCA, 2021; Lichtenegger et al., 2020), mechanical recycling would result in a substantial reduction of the environmental impacts, yielding it as the better alternative for the management of decommissioned wind turbine blades.

6.5. Optimization of concrete with wtbw: A castilla y león case

Mechanical recycling is the best environmental alternative for WTBW management, even if further transport is needed, as highlighted in Section 6.4. However, the vast area that *Castilla y León* represents and the high volumes of WTBW to be treated shortly lead to the need of minimizing transportation to environmentally optimize FRC production incorporating mechanically recycled WTBW.

6.5.1. Existing wind farms

To gather accurate transportation information, all currently existing wind farms in *Castilla y León* were first precisely located, as well as the number of turbines to be dismantled in the next 5-10 years per wind farm (EREN, 2024), shown in Fig. 5. The UTM coordinates of all these wind farms were also collected. There are two main hotspots for WTBW production in *Castilla y León*. On the one hand, Burgos (2,883 blades to dismantle) and Soria (2,391 blades to dismantle), located in the northeast of the autonomous region. On the other hand, Zamora (1,428 blades to dismantle), which is the third higher WTBW-producer province, and is located in the west of the region.

6.5.2. Optimal location of WTBW treatment plants

The aim of this analysis was to define specific locations for the plants for the mechanical recycling of WTBW for which the WTBW transport was minimized for all WF in *Castilla y León*. For this purpose, the WF were identified as point masses concentrated in their UTM coordinates, whose value of "mass" was the number of turbines to be dismantled. Then, the weighted centroid of the WF was calculated with a city as a reference using Eq. (1).

$$|distance_{CITY-CENTROID}| = \frac{\sum (turbines\ to\ dismantle_{WF} \cdot distance_{CITY-WF})}{\sum turbines\ to\ dismantle} \quad (1)$$

When using a single city as a reference, the result was a circumference on whose entire perimeter the centroid could be located. Therefore, in order to find the exact location of the centroid, it was calculated through three different reference cities (Burgos-BU, Valladolid-VA, and Salamanca-SA), as the resulting three circumferences only intersected in a specific location: the centroid where the treatment plant had to be located for minimized WTBW transportation impact. This procedure, applied for the obtention of a unique centroid for the whole autonomous

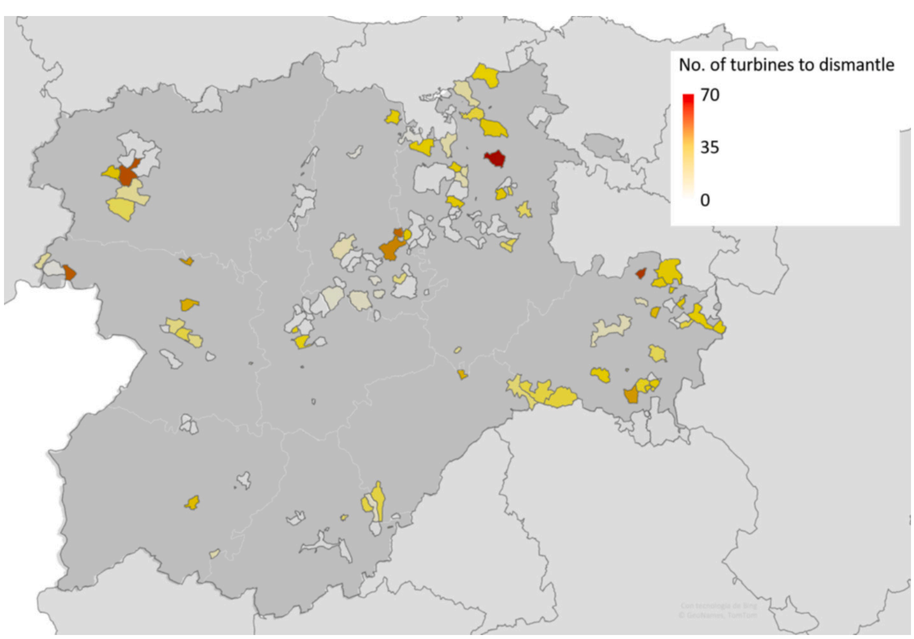

Fig. 5. Heatmap of the locations of wind farms related to the number of turbines to be dismantled in the next 5-10 years (EREN, 2024).

region that represents all WF (option A, blue), can be seen in Fig. 6. This centroid was located in the far-east end of *Castilla y León*, where the Burgos-Soria hotspot was identified, as 56% of the WTBW is to be produced in these two provinces.

Castilla y León was therefore divided into different sections, in order to minimize WTBW transport from all places amongst the area of study. Eight different divisions were considered in order to evenly separate the number of blades to be managed in them, in order to position the treatment plants close to the centroids with the highest point-masses. First, two sections and therefore two centroids as a solution were considered (options B and C); then three parts and three centroids (options D and E); and lastly four parts and four centroids (options F, G and H). Table 8 lists the considered sections, along with the UTM coordinates of the calculated centroids and their distances to each reference city. All centroids tended to gather around the same areas, independently of the option chosen. These areas were close to where most of the WTBW is produced, as the mass of these places was higher.

Consequently, a centroid was placed in each resulting area, and all the WF were assigned to the closest centroid. These centroids were labelled as *"M"* (*"mechanical"*) followed by the acronyms of the provinces they give service to, and are graphed in Fig. 7 and listed in Table 9:

- The one in the north was located in the province of Burgos (M-BU/PA/VA, in blue). It would receive a total of 4,047 blades to manage, accounting for around 43% of the total.
- The one in the south was located in Segovia (M-AV/SA/SG, in red). It would receive 927 blades (around 10%).
- The one in the east was located in Soria (M-SO, in green), receiving 2,391 blades to manage (around 26%).
- The last one was placed in Zamora (M-LE/ZA, in purple), for a total of 2,001 blades to manage (around 21%).

To validate the solution, an index was calculated for each province of *Castilla y León*. This index was defined as the total distance from dismantling the blade to incorporate it into concrete divided by the number of blades to be dismantled in that province in the next 5-10 years. The distance for the index of one province was therefore obtained by precisely calculating and averaging all distances from each WF of that province to their assigned centroid and adding to it the distance between the WTBW treatment and concrete plants. The concrete plant was assumed to be located in the capital city of each province and then sourced within itself. The obtained indexes are recorded in Table 10, and plotted in Fig. 8. It was found that the average distance from the wind farms to the centroid was lower for the provinces with higher amounts of WTBW to be managed (Burgos, Soria, and Zamora). Furthermore, the provinces with lower expected quantities of WTBW (Valladolid and Salamanca) were located further from these treatment plants, being possible to afford these high transportation distances in compensation for lower values where it is most needed. Therefore, the transportation in kilometer per kilogram of WTBW to be managed was adequately minimized in *Castilla y León*.

6.5.3. *Life cycle impact assessment and interpretation*

The results of the cradle-to-installation LCA by considering optimized transport distances in *Castilla y León* is depicted in Table 11, both in terms of GWP (kgCO_2-e) and ADPf (MJ). These values represent the environmental impact of the WTBW transportation from the WF to the construction site ("transport" rows). Then, these results are added to the impacts of concrete production and WTBW management shown in Table 6 ("total" rows). The results from the LCA are also represented in heatmaps for GWP in Fig. 9 and for ADPf in Fig. 10.

The provinces with the best values were the ones that had the lowest distance to number of blades to be decommissioned, as less transport distance was needed to obtain one kilogram of mechanically recycled

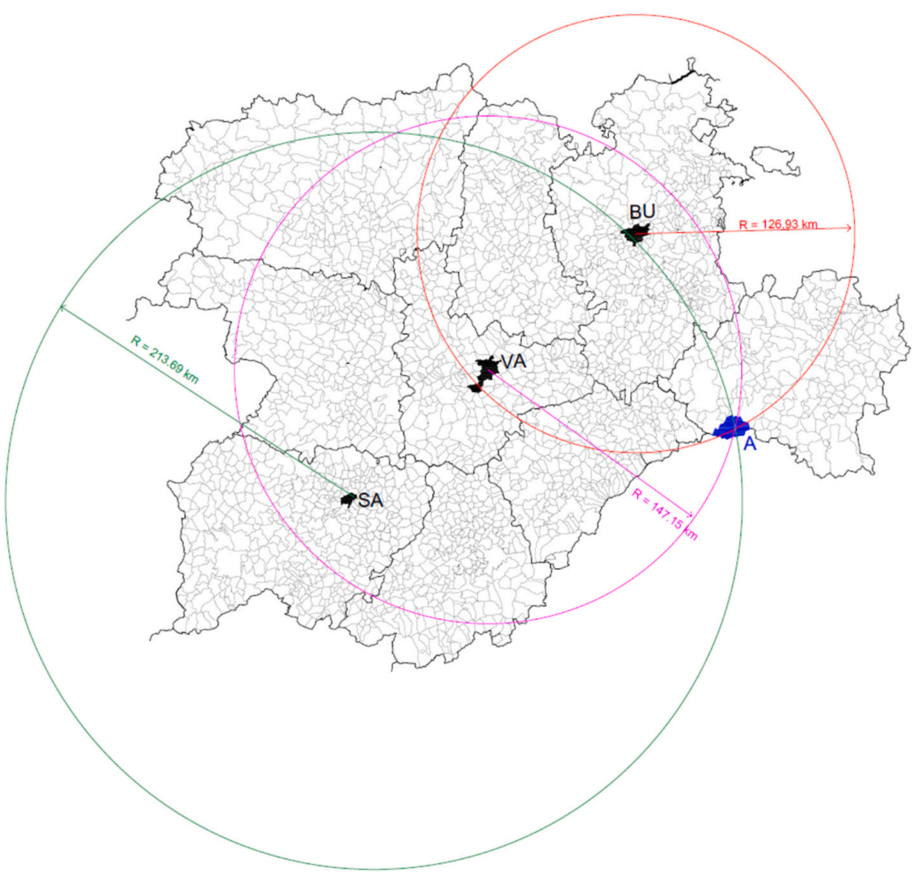

Fig. 6. Calculation procedure of a unique centroid for *Castilla y León*.

Table 8
Location of centroids for the different sections studied.

OPTION	DESCRIPTION	UTM X (m)	UTM Y (m)	DIST. TO BU (km)	DIST. TO VA (km)	DIST. TO SA (km)
A	UNIQUE	498,007	4,573,778	126.93	147.15	213.69
B	BU-PA-SO	506,647	4,636,070	83.94	151.38	251.88
	AV-LE-SA-SG-VA-ZA	228,238	4,665,260	215.13	138.47	135.33
C	BU-LE-PA-SO	508,314	4,618,891	95.16	152.47	245.97
	AV-SA-SG-VA-ZA	230,490	4,654,748	214.37	132.51	124.85
D	BU-SO	520,386	4,647,991	87.02	168.28	267.83
	LE-PA-VA-ZA	265,663	4,692,879	175.59	121.86	153.76
	SG-SA-AV	384,233	4,506,226	187.84	113.30	110.31
E	BU	435,463	4,732,199	45.21	143.48	251.29
	PA-SO	515,429	4,602,082	116.64	158.05	252.38
	LE-VA-ZA-SA-AV-SG	221,033	4,662,593	222.88	143.92	136.00
F	BU-PA	409,590	4,724,090	51.19	122.74	229.91
	SO	554,480	4,609,140	137.43	198.17	287.77
	LE-ZA-VA	217,630	4,670,540	226.95	149.36	146.16
	SA-AV-SG	384,233	4,506,226	187.84	113.30	110.31
G	BU-PA-VA	402,910	4,724,970	54.08	119.32	225.13
	SO	554,480	4,609,140	137.43	198.17	287.77
	AV-SA-SG	209,520	4,672,510	187.84	113.30	110.31
	LE-ZA	382,387	4,509,853	233.50	158.15	149.92
H	BU-PA-VA-SG	398,400	4,722,520	57.89	117.93	221.38
	SO	554,480	4,609,140	137.43	198.17	287.77
	LE-ZA	382,387	4,509,853	233.50	158.15	149.92
	SA-AV	209,520	4,672,510	209.47	119.85	94.74

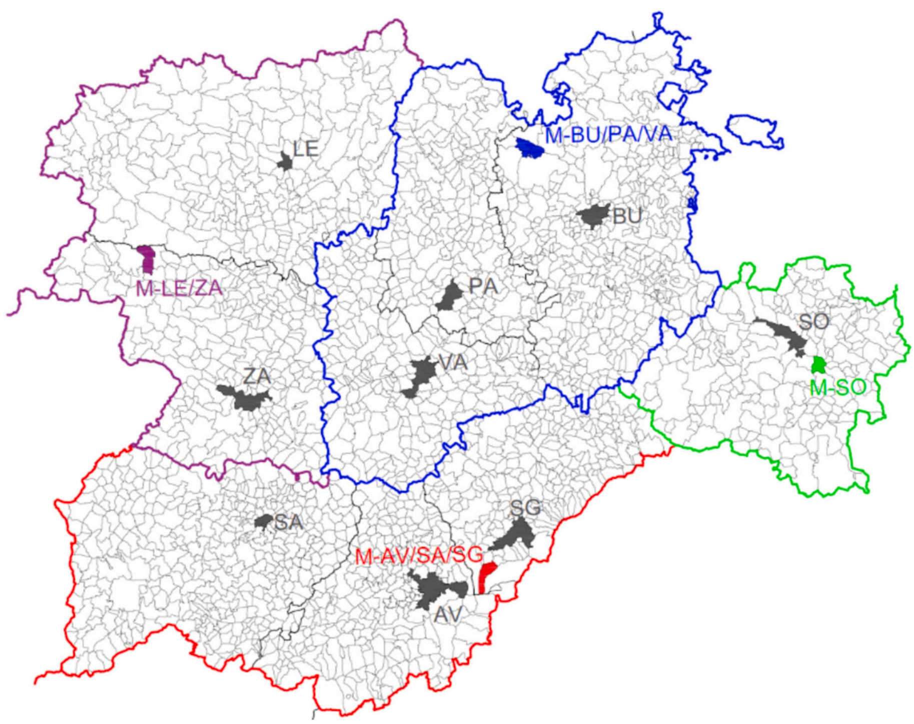

Fig. 7. Final location of the centroids and the territory covered by each.

Table 9
Optimal location of the centroids.

OPTION	DESCRIPTION	UTM X (m)	UTM Y (m)
OPTIMAL	BU-PA-VA	409,590	4,724,090
	SO	554,480	4,609,140
	LE-ZA	217,630	4,670,540
	SA-AV-SG	384,233	4,506,226

WTBW. Valladolid and Salamanca can be highlighted for their high results due to the low quantity of blades to be treated in the next 5-10 years. However, Ávila was the province with the lowest impacts, as both the WF and the management center (M-AV/SA/SG) was close to the capital city of the province, highly reducing the transportation impacts compared to other provinces. Both hotspots for WTBW production mentioned in previous sections were also properly addressed, achieving low values regarding both GWP and ADPf. Compared with the real case explained in Section 6.4, a slight decrease in environmental impacts was also found: a reduction of up to 0.2% for GWP and 0.3% for ADPf on average when producing FRC with 10% vol. agg. WTBW.

Regarding this data and accounting the around 286,078 metric tons of WTBW to be managed in *Castilla y León* in the next 5-10 years, the optimization of the transport distance of the WTBW would allow saving up to 1,111,212 kgCO_2-e and 17,046,459 MJ on average compared to

Table 10
Data per province in *Castilla y León* regarding optimal mechanical treatment locations.

PROVINCE	WIND FARMS OVER 15 YEARS	AVERAGE DIST. FROM WF (km)	DIST. TO CITY (km)	DISTANCE/ No. BLADES
ÁV	9	13.17	27.93	0.0733
BU	45	42.97	48.84	0.0318
LE	7	40.46	71.65	0.1957
PA	23	68.94	80.19	0.1458
SA	2	118.57	112.81	1.7529
SE	3	74.47	34.89	0.4674
SO	32	39.74	18.01	0.0242
VA	2	136.18	123.77	1.8436
ZA	22	49.22	89.17	0.0969
AVERAGE	-	64.86	67.47	-

the real case (C+MR+T) explained in Section 6.4 when producing FRC with 10% vol. agg. of mechanically recycled WTBW. When compared with the incineration treatment shown in the real case (C+IN+T), the savings make for up to 288,578,588 kgCO_2-e (28.1% reduction) and 309,366,038 MJ (5.7% reduction) on average.

7. Sensitivity analysis

Sensitivity analysis is a key tool to analyze robustness and uncertainty factors in LCA (Wei et al., 2015). This procedure is used to quantify the importance of each parameter towards the final LCA result as a consequence of a variation of its input values (Bisinella et al., 2016). In this research, a sensitivity-perturbation analysis was performed in order to assess the robustness of the results. Variations of ±12.5%, ±25.0%, ±30.0%, ±37.5% and ±50.0% for each parameter involved in every proposed scenario were conducted with a one-at-a-time approach (Islam et al., 2025), thus yielding a value range for each parameter and

Fig. 8. Heatmap of the distance (km) to number of blades to be decommissioned ratio per province of *Castilla y León*.

Table 11
Environmental impacts of concrete per WTBW percentage and province in *Castilla y León*.

ENVIRONMENTAL IMPACT		GWP (kgCO2-e)			ADPf (MJ)		
WTBW AMOUNT (% vol. agg.)		2	5	10	2	5	10
AV	TRANSPORT	0.09	0.23	0.46	1.42	3.54	7.08
	TOTAL	292.26	297.61	310.94	1,888.27	1,969.24	2,147.49
BU	TRANSPORT	0.20	0.51	1.02	3.16	7.91	15.82
	TOTAL	292.37	297.89	311.50	1,890.02	1,973.61	2,156.22
LE	TRANSPORT	0.25	0.62	1.24	3.86	9.66	19.31
	TOTAL	292.42	298.00	311.73	1,890.72	1,975.36	2,159.72
PA	TRANSPORT	0.33	0.83	1.66	5.14	12.85	25.69
	TOTAL	292.50	298.21	312.14	1,892.00	1,978.54	2,166.10
SA	TRANSPORT	0.51	1.28	2.57	7.97	19.93	39.86
	TOTAL	292.68	298.67	313.05	1,894.83	1,985.63	2,180.27
SG	TRANSPORT	0.24	0.61	1.21	3.77	9.42	18.84
	TOTAL	292.41	297.99	311.70	1,890.63	1,975.12	2,159.25
SO	TRANSPORT	0.13	0.32	0.64	1.99	4.97	9.95
	TOTAL	292.30	297.70	311.12	1,888.85	1,970.67	2,150.36
VA	TRANSPORT	0.58	1.44	2.89	8.96	22.39	44.78
	TOTAL	292.75	298.83	313.37	1,895.81	1,988.09	2,185.19
ZA	TRANSPORT	0.31	0.77	1.54	4.77	11.92	23.84
	TOTAL	292.48	298.15	312.02	1,891.63	1,977.62	2,164.25
AVERAGE		292.46	298.12	311.95	1,891.51	1,977.53	2,163.21

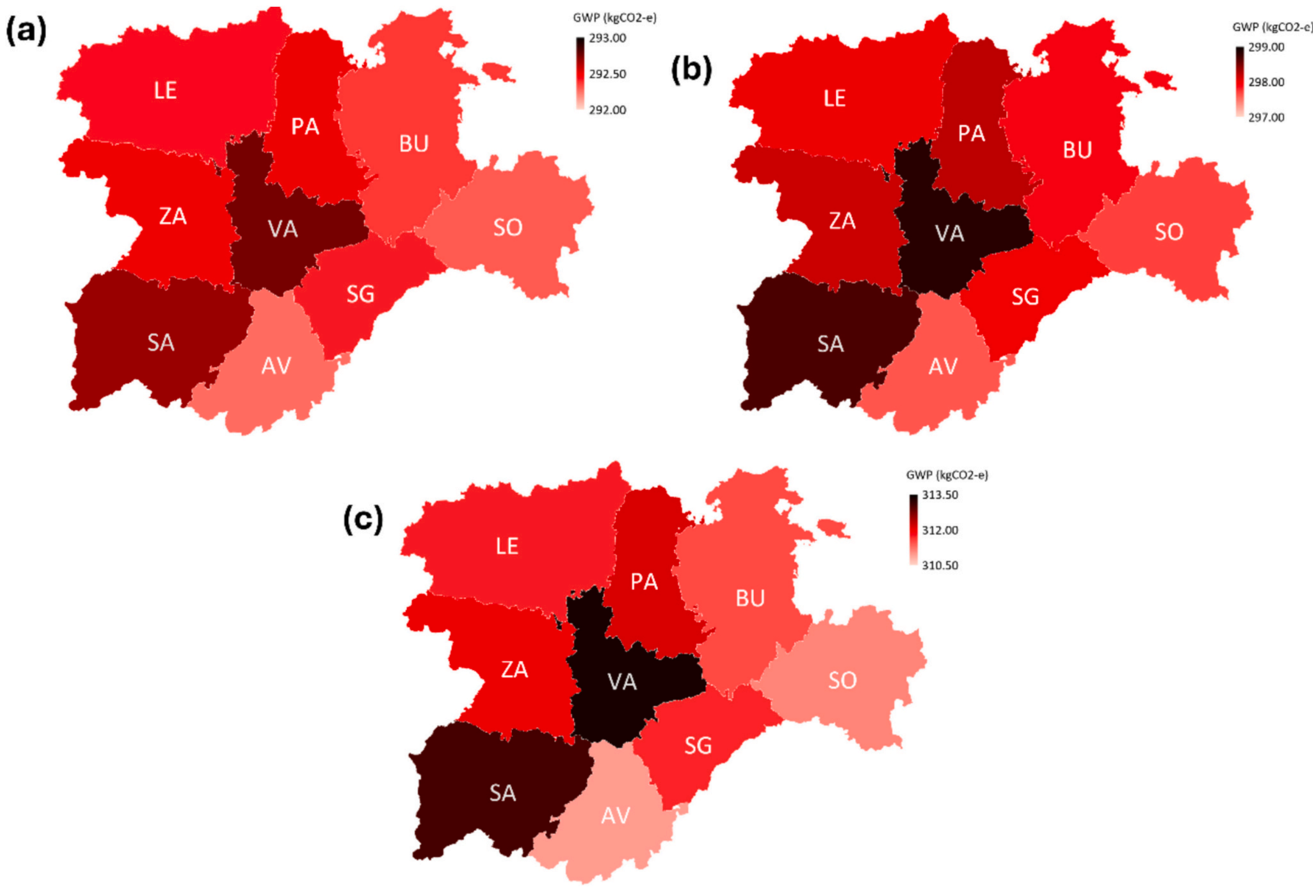

Fig. 9. Heatmap per province of GWP: (a) 2% WTBW incorporation; (b) 5% WTBW incorporation; (c) 10% WTBW incorporation.

variation. The Sensitivity Ratio (SR) was also calculated according to Eq. (2) (Bisinella et al., 2016; Islam et al., 2025). Finally, a threshold analysis was also performed, thus defining the necessary percentage variations of the parameters in one scenario to yield the same LCA results than in the other scenarios. All the results can be found in the Supplementary Material.

$$SR_a^b = \frac{\left(\frac{\Delta result}{initial\ result}\right)_a}{\left(\frac{\Delta parameter}{initial\ parameter}\right)_b} \quad (2)$$

According to the literature (Sobek et al., 2024), values of SR between 0.20 and 0.80 can be considered significant, and when a SR is higher than 0.80, it can be affirmed that the LCA result is highly affected by a variation of this parameter. For the waste management scenario, GFRP was the most influential parameter for the processes of landfilling and incinerating the WTBW, always yielding a SR higher than 0.80 for both GWP and ADPf. However, both blade segment cutting and crushing were found relevant for WTBW mechanical recycling, with a SR equal to 0.25 and 0.65, respectively, for both GWP and ADPf. For scenarios that involved conventional concrete production and WTBW incineration as the recycling alternative, the SR of the WTBW management yielded values of up to 0.34 (GWP) and 0.20 (ADPf) when considering a WTBW amount in concrete of 10%. However, the significance of WTBW mechanical recycling on the overall environmental impact was lower, SR results of up to 0.11 (GWP) and 0.18 (ADPf) being obtained. This demonstrates the high dependence of the whole environmental impact on the WTBW management when incineration is conducted, which renders non-significant when mechanical recycling is performed.

Regarding the value ranges and threshold analysis, it can be seen that the results yielded in this research are rather robust. For the waste management scenarios, the impacts related to the crushing process for mechanical recycling of WTBW would have to be increased up to +525.3% (GWP) and +30.2% (ADPf) to reach the top values obtained in the incineration scenario. Therefore, it can be stated that the process of the mechanical recycling of WTBW would have to be highly altered to lead to different results than those yielded in the present research. In the same way, variations on the mechanical recycling process higher than +348.2% (GWP) and +28.1% (ADPf) would have to take place in order to alter the results obtained in the concrete production scenarios.

8. Conclusions

In this research, the environmental impacts of managing Wind Turbine Blade Waste (WTBW) were studied through Life Cycle Assessment (LCA). The following conclusions can be drawn:

- The best methodology for WTBW management was mechanical recycling, as it caused an assumable environmental impact, and it generated no further waste. Although landfilling was the least polluting option overall, it was forbidden in the European Union, did not provide a real management solution, and could be dangerous in the long run. Incineration was high-energy demanding, produced high environmental impacts compared to other solutions (around 4.5 times the GWP of mechanical recycling and 1.2 times the ADPf), and generated by-products that required further treatment.
- When combining these results with concrete production, mechanical recycling was highlighted as the best option for WTBW management. Mechanical treatment of WTBW yielded a fiber-like composite valid for replacing the aggregates in concrete production, thus reducing the environmental impact from raw material extraction. The incineration of the WTBW resulted in higher environmental impacts due

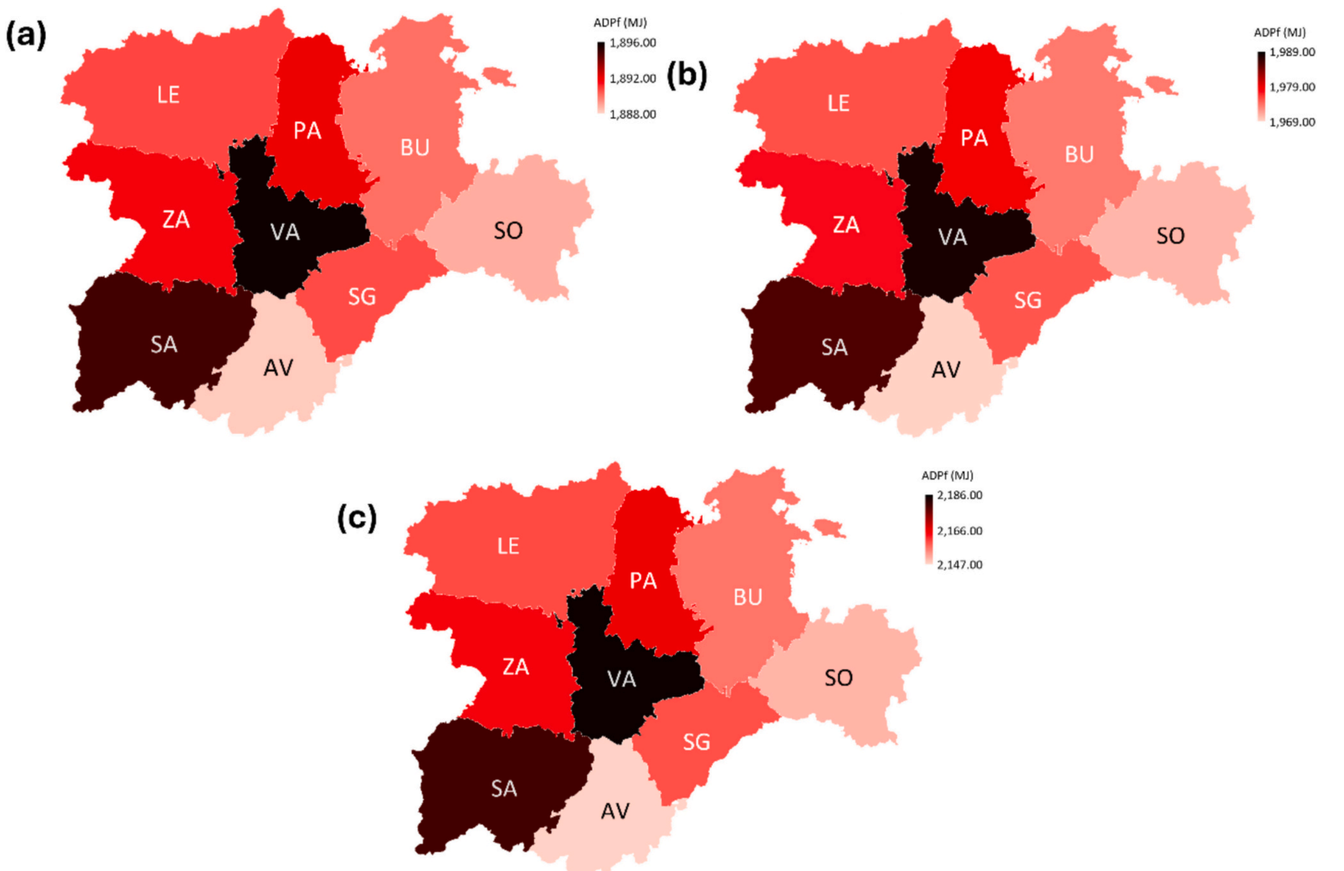

Fig. 10. Heatmap per province of ADPf: (a) 2% WTBW incorporation; (b) 5% WTBW incorporation; (c) 10% WTBW incorporation.

to WTBW treatment and no replacement of concrete raw materials, which led up to +39.4% GWP and +6.3% ADPf.

- In the specific case of *Castilla* y León, only four WTBW treatment plants would be needed to successfully implement the approach of mechanically recycling WTBW and adding it to concrete as a raw material. Furthermore, they would reduce the environmental impacts of this process compared to a current real case, with an average reduction of around 0.2% in terms of GWP and 0.3% in ADPf per cubic meter of concrete. 1,111,212 $kgCO_2$-e and 17,046,459 MJ would be saved if all the blades to be dismantled are accounted. In addition, this methodology could be implemented in any region where the need for WTBW management is required.

As an overall conclusion, this study has shown that the production of concrete with WTBW is an environmentally better option compared to the other solutions currently available. In addition, an adequate planning of the location of the WTBW treatment points would enable to minimize the environmental impact of WTBW transportation in any area with a high demand for wind-turbine blade recycling. Therefore, it is demonstrated that this solution provides proper management of a waste with such a complicated recycling as the out-of-use wind turbine blades in terms of environmental impact.

Nevertheless, several limitations can be found in this research. First, the variability in the sizes, designs, and material compositions of wind turbine blades worldwide poses a substantial challenge to yield homogeneous results. In this study, the authors focused on a specific blade type readily available for analysis, but a broader range of decommissioned blades should be examined to verify and generalize these findings. Second, conducting detailed chemical and morphological analyses on a variety of WTBW samples is essential for building a robust database of material properties. Such data would facilitate more precise calculations of environmental burdens and transportation modeling and would enhance the accuracy of the resulting LCA. Finally, concrete manufacturing at an industrial scale could also be explored, as larger batching processes, improved handling methods, and advanced casting techniques may reveal additional pathways for reducing environmental impacts. By addressing these considerations, future research can refine the proposed methodology, enhance its replicability, and thereby contribute more effectively to circular economy in both the wind-energy and construction sectors.

CRediT authorship contribution statement

Javier Manso-Morato: Conceptualization, Formal analysis, Investigation, Methodology, Validation, Writing – original draft. **Nerea Hurtado-Alonso:** Formal analysis, Investigation, Methodology, Software, Validation, Visualization. **Marta Skaf:** Conceptualization, Funding acquisition, Methodology, Project administration, Supervision. **Víctor Revilla-Cuesta:** Formal analysis, Investigation, Methodology, Software, Validation, Visualization, Writing – review & editing. **Vanesa Ortega-López:** Conceptualization, Writing – review & editing, Funding acquisition, Project administration, Resources, Supervision.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.eiar.2025.107992.

Data availability

All data used in this research can be found in the text of the article and in its supplementary material.

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