

Environmental benefits of a new recyclable thermoplastic resin for wind turbine blades[☆]

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HIGHLIGHTS

- End-of-life management of WTBs remains a major environmental challenge.
- A new thermoplastic resin enables the manufacture of highly recyclable WTBs.
- LCA compares conventional and recyclable blades up to the year 2100.
- Recyclable WTB reduce impacts across several environmental impact categories.
- Sensitivity analysis of system and process parameters reveals key parameters and future research needs.

ARTICLE INFO

Keywords:

Wind energy
Eco-design
Recycling
Life cycle assessment

ABSTRACT

Wind energy is a key renewable energy source in the current energy landscape, with further growth expected. Nevertheless, its expansion raises an important environmental challenge due to the End-of-Life management of Wind Turbine Blades (WTB), as landfill disposal and incineration are the most common options. To address this issue, more sustainable alternatives have emerged to improve blades recyclability and circularity, notably the use of recyclable thermoplastic resins. This study applies a Life Cycle Assessment (LCA) to compare the environmental impacts of a WTB manufactured with a recyclable thermoplastic resin and a conventional blade. The main advantage of recyclable resin is that it facilitates the separation of its main components, enabling them to be reused in new recyclable blades. The LCA considers three full life cycles for both blade configurations, separated by 25-year intervals. The results indicate a substantial reduction in environmental impacts in several categories evaluated after the second and third life cycles compared to the use of conventional resin. In particular, the recyclable blade achieves a 30% reduction in climate change impact category and a 50% improvement in the material resources: metals/minerals impact category. These results highlight the potential of ecodesign and recyclable materials to significantly enhance the long-term sustainability of the wind energy sector.

[☆] This article is part of a special issue entitled: 'Sustainability Assessment' published in Composites Part A.

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<https://doi.org/10.1016/j.compositesa.2026.110282>

Received 1 July 2026; Received in revised form 3 September 2026; Accepted 12 September 2026

Available online 13 September 2026

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1. Introduction

Wind energy is the most widely used renewable energy source in the European Union (EU). In 2025, the installed wind power capacity reached approximately 291 GW, of which 254 GW were onshore and 37 GW offshore. It is projected that the EU will increase its installed capacity to 344 GW between 2025 and 2030; however, its target for 2030 is 425 GW [1]. Wind energy is produced locally, is reliable, does not generate direct carbon emissions, and requires only very small amounts of water [2]. Energy production is based on the conversion of the kinetic energy of the wind into electrical energy by means of wind turbines [3]. These devices consist of more than 8,000 components, organised into different main sections. The rotor, which includes the hub and Wind Turbine Blades (WTB); the nacelle, located at the top of the wind turbine, which houses the main mechanical and electrical systems; the tower, whose function is to support both the nacelle and the rotor, designed to withstand high dynamic loads; and, finally, the internal and external cabling system, which allows the connection and transmission of the energy generated by the wind turbine [4]. With an average operational lifetime of 20 years, large amounts of wind turbines are expected to reach End-of-Life (EoL) over the next decades, with the recycling of these complex components representing one of the most significant environmental challenges for the sector [5]. The EoL waste management for WTB is particularly difficult because they are made of hybrid materials that are hard to separate, also the WTB are very bulky structures and there is no dominant recycling route [6].

The EoL of WTB can take several paths [7]. As WTB are composed of various materials and elements, such as glass fibre (or carbon fibre) composites, thermoplastic coatings, balsa wood and adhesives, it is extremely difficult to separate and recycle the parts [8]. Across Europe, the most common EoL management routes for WTBs continue to be landfilling and incineration [9]. Nevertheless, increasing policy pressure is encouraging the adoption of higher-value recovery alternatives. Among the available recycling technologies, mechanical recycling, often associated with co-processing in the cement industry, is currently the most widely implemented and economically viable option at an industrial scale [10]. In contrast, pyrolysis and solvolysis show considerable potential but remain at an early stage of development; consequently, they are not yet widely established in European practice. However, their relevance is expected to grow as these technologies mature and markets for high-quality recycled materials continue to expand. Through landfilling, the materials that compose the blades cannot be recovered and it often involves high taxes and very specific ground conditions [11]. Incineration is a common thermal method for generating heat and electricity by incinerating WTB in combined heat and power plants (CHP), however, the inorganic components of composite materials used to manufacture WTB can be a source of hazardous emissions of flue gases containing small glass and dioxins [12]. Pyrolysis consists of the thermal decomposition of carbon-based components in the absence of oxygen (temperatures between 300 and 1200 °C), producing new raw materials, but requires specific pressure and temperature conditions and a very high energy input [13].

The waste – management hierarchy – prevention (at the top), reuse, recycling, recovery and disposal (at the bottom) – underpins EU waste policy, directing material flows towards a circular economy [1,3]. Scientific literature emphasises that improving recycling efficiency depends on integrating eco-design principles that reduce material complexity and enhance EoL recovery [4]. Furthermore, research shows that new recyclable materials benefit from design strategies that facilitate dismantling and clean material separation, thereby strengthening the outcomes of the circular economy [5]. Together, the hierarchy and ecodesign principles facilitate the creation of more sustainable, high-quality recycling loops. In this context, thermoplastic resins [14] have emerged as a more sustainable alternative to the thermoset epoxy-type resins commonly used in the manufacture of WTB. This trend is mainly attributed to the ability of the resin to be recyclable and

separates from the reinforcing fibres. Commercial thermoplastic resins such as Elium® [15], allow curing at room temperature without the need for moulds or ovens, significantly shortening manufacturing time and reducing tooling and labour costs. They also facilitate thermal welding, eliminating the need for conventional adhesives, which can result in lighter, stronger, and more reliable structures. Unlike conventional epoxy resins, thermoplastics can be melted, renovated or dissolved to recover both the polymer and the reinforcing fibres, enabling more efficient recycling [16]. Although the initial prototypes using thermoplastic resins such as Elium® were experimental, it has been possible to manufacture large-scale blades with mechanical properties comparable to those of epoxy-based blades [17]. Taken together, these capabilities make thermoplastic resins a viable and competitive alternative, offering improvements in cost, structural performance, sustainability, and recyclability, while allowing for the use of formats and manufacturing processes similar to those employed with conventional thermoset resins.

In recent years, there has been growing academic and industrial interest in the potential environmental benefits of various process for the disposal of WTB, typically based on the Life Cycle Assessment (LCA) method. The LCA is a systematic method for evaluating the environmental impacts associated with a product or service. The LCA allows the identification of environmental hotspots (e.g., the most critical life cycle stages or processes) as well as potential burden shifting (e.g., from one impact category to another, from one region to another, or from one life cycle stage to another). This approach provides a holistic view of environmental aspects associated with the product or service, which is essential for making decisions in pursuit of sustainability [18]. A number of LCA studies have addressed the environmental performance of different EoL options for WTB. For example, Alavi et al [19] conducted an LCA of four EoL destinations for WTB generated in Australia: landfilling, mechanical recycling, pyrolysis, and solvolysis. There are many EoL alternatives for WTB; however, it is essential to emphasize strategies that prioritize human health, optimize energy consumption, and take into account the environmental impact of recycling processes for the future of renewable energy. Alsaleh and Sattler [20] focused on onshore WTB in the USA in accordance with US regulatory standards, they can be concluded that increasing the service life of wind turbines significantly reduces the impact per kWh of electricity produced. Nagle et al. [21] performed a comparative study of three scenarios for Ireland, looking at new alternatives for a second life. N. Cong et al. [22], evaluated the CO₂ emission reduction potential of four different WTB management scenarios in China, with emphasis on scenarios in which blade waste is utilised as a raw material in concrete production. The reduction of waste associated with wind energy, an ever-expanding clean energy source, offers a promising prospect globally. In this context, the levels of the waste hierarchy highlight the importance of reuse, recycling and energy recovery as fundamental tools to reduce energy consumption, a crucial challenge in today's world [23]. Otherwise, several LCAs have been conducted on other recyclable WTB manufactured using different types of resins. For instance, Farazmandnia and Ilinca [24] evaluated a blade produced with Elium resin by analysing the environmental impacts associated (embodied energy and carbon footprint) with the various stages of the complete life cycle of blades featuring different design configurations. This study considers a complete life cycle of the WTB, following a cradle-to-grave approach. The EoL stage includes both incineration and landfill disposal. The results indicate that the blade materials, particularly the resin and glass fibres, are the main contributors to the overall environmental impacts.

A study closely related to the present research is the work conducted by K. Pedersen et al. [25], in which the authors present an LCA that incorporates the thermolysis of waste from thermoplastic WTBs based on PMMA resin. The study evaluates the impact categories associated with global warming potential and the production of virgin materials. The scope of this study includes a single complete blade cycle. The results of this study can serve as a useful benchmark for comparison with

the findings of the present research.

Regarding the limits of the system, these LCAs studies lack a forward-looking perspective, as they do not evaluate the future development of the wind energy sector from a long-term point of view. Including this type of prospective approach is essential, since wind energy is expected to play a key role in the energy transition. In this context, it is important to assess not only the sustainability of the new materials that will be used in the future manufacturing of WTB, but also the environmental performance of the recycling processes currently under development. In addition, the ongoing transition of current electricity mixes towards configurations with a higher share of renewable energy sources should be considered, as this change may significantly influence the life-cycle environmental impacts associated with these systems and the recycling processes that use electricity as the main energy input.

In this study, we address this knowledge gap by conducting a comparative LCA between WTB manufactured with a new recyclable thermoplastic resin, called Akelite [26], and that of a conventional blade made with Epoxy resin. We have considered a 75-year time horizon, divided into three complete cycles of 25 years each (the lifetime of a wind turbine [9,10,27]), which includes two blade replacement events. This novel LCA framework represents one of the main contributions of the present study compared with previous works [24,25], which consider only a single life cycle, use a different type of thermoplastic resin, and assume a single electricity mix scenario. In contrast, the work incorporates multiple life cycles and considers different electricity mix scenarios over time, providing a more comprehensive assessment of the long-term environmental performance of recyclable WTBs. Both blades are designed for installation in onshore wind turbines; therefore, the operation and maintenance (O&M) processes considered in this study correspond to those of onshore wind energy systems.

Since the manufacturing and recycling processes of the recyclable WTB evaluated in this study is still at the laboratory scale, a scaling approach has been applied to estimate their potential environmental performance at an industrial scale. Section 2 describes the methodological framework used for the industrial scale-up, as well as the LCA methodology adopted in the study. Section 3 presents and discusses the results of the environmental analysis comparing the two WTB. Moreover, to test the robustness of our results, we performed a sensitivity analysis considering the following elements: a future scenario for the national electricity mix in 2050, and the potential improvement of the acetone recovery process. Finally, section 4 presents the main conclusions derived from the research conducted.

2. Methodology

The LCA methodology is standardised by the international standards ISO 14040 [28] and ISO 14044 [29]. The LCA consist of four phases in an iterative process that allows for comprehensive analysis:

1. Goal and scope definition: This phase involves the definition of the system boundaries and the functional unit (FU) to which the inputs and outputs refer.
2. Life Cycle Inventory (LCI) analysis: In this phase, data on inputs and outputs of unit processes included within the system boundaries are collected (e.g., raw materials, energy, emissions, waste, etc.).
3. Life Cycle Environmental Impact Assessment (LCIA): The inventory results are translated into potential environmental impacts by using a set of characterization factors that determine the potential impact of environmental flows into impact categories such as climate change and resource depletion, among others.
4. Interpretation: In this last phase, the results are analyzed to draw conclusions, identify critical points and propose improvements in the processes [30].

In this study, the LCA was performed using the open-source software Activity Browser combined with background LCI data from the

ecoinvent database v3.10.1 [31].

2.1. Goal and scope

The goal of the present study is to assess the environmental impact associated with the life cycle of a recyclable blade (RB) manufactured with the recyclable resin (RR) and to compare it with that of a conventional blade (CB) made with Epoxy resin (conventional resin, CR). Both blades are designed for use on onshore wind turbines.

The LCA of WTBs was conducted using a cradle-to-grave approach. This means that the analysis considers all stages, from resin manufacturing (RR and CR) to EoL, including both the waste treatment of the CB and the recycling process of the RB. The functional unit is the amount of blade mass required to generate 1 kWh of electricity supplied to the grid over a full life cycle (25 years). The reference flow for the amount of blade required to generate 1 kWh is 0.000239 kg of WTB [27].

The system boundaries cover: (i) the extraction of raw materials, mainly glass fibres and the two resins (including the production of resins), (ii) the use phase, the blade contributes to electricity generation, (iii) the EoL stage, treatment of CB waste and recycling of RB. The reference point is the manufacturing of a CB compared with the emerging technology of the RB made with Akelite resin [26,32].

A reference inventory based on a previous study was used [27], and background data were obtained from the ecoinvent 3.10.1 database. Finally, environmental impacts were evaluated using the Environmental Footprint 3.1 method [33], recommended by the European Commission, focusing on 16 impact categories.

The geographical scope of this study is Spain. Three temporal cycles have been established for each blade (CB and RB), using the standard useful life of a reference WTB [26]. The Cycle 1 extends from 2025 to 2050 (a critical moment when half of Spain's WT will reach the end of their useful life), assuming the complete substitution of the CB. By contrast, the RB undergoes an initial recycling process, enabling the recovery of a specific percentage of glass fibres and resin, as detailed in Section 2.2.4. Afterwards, Cycle 2 with a timeline extending to the year 2075, mandates the manufacture of a second CB and a subsequent recycling process for the RB, maintaining the established material recovery rates. Finally, Cycle 3 a third CB and an additional recycling cycle are required for the RB (Fig. 1). Additional cycles were not considered, as results over such extended timeframes would involve a high degree of uncertainty due to the lack of reliable technical data for such long-term periods.

Once the three complete life cycles of the recyclable WTB have concluded (year 2100), a set of materials is generated as a result of the third recycling cycle for RB. These materials mainly consist of recycled glass fibres and RR. Although these materials could be reused to manufacture new blades, the materials recovered after the Cycle 3 were excluded from the system boundaries in the LCA model. This decision was made due to the high level of uncertainty associated with defining plausible EoL scenarios for these materials in the year 2100, which could significantly affect the robustness of the results.

Fig. 2 represent the stages and sub-stages corresponding to resin production, blade manufacturing, and EoL processes for RB and CB.

As shown in the figure, the complete life cycle of the RB includes a recycling stage in which the main materials composing the blade, primarily glass fibre and RR, are recovered and reused in the production of a subsequent blade generation. In addition, the acetone involved in the acetone bath during the recycling process is also recovered. Throughout the three complete RB cycles, the impacts associated with WTB recycling are incorporated within the cycles themselves by applying a system expansion approach. Fig. 3 presents a detailed scheme of the processes included within the LCA system boundaries, as well as those excluded from the analysis. Transport associated with the production of the raw materials has been included within the system boundaries, as "market for" activities were selected for their modelling of the background

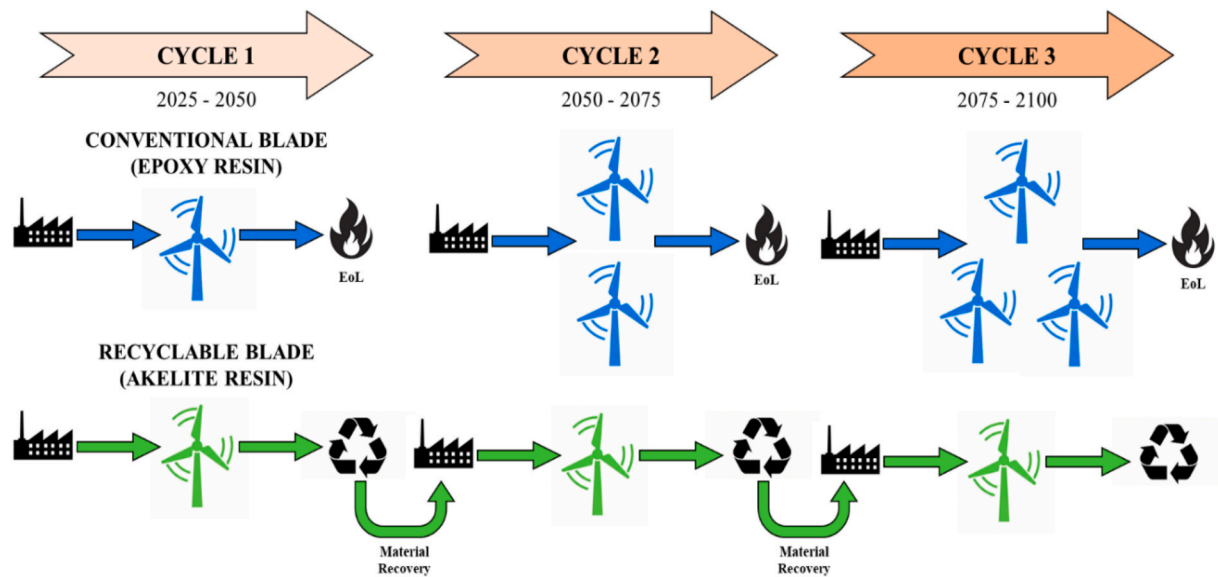


Fig. 1. Lifetime framework for the service life of the two types of WTBs during the three cycles of use.

processes. In contrast, transport processes associated with the installation, dismantling, and decommissioning of the WTB are excluded from the system boundaries (box called “transport” outside the system boundaries); material and energy consumed for O&M activities during the wind turbine use phase; and blade substructures, such as balsa wood and electronic components, are also excluded (boxed called “operation and maintenance” and “substructures” respectively, outside the system boundaries). These exclusions apply equally to the CB system. The reason why these processes have not been included within the system boundaries is explained in the following section. The impacts associated with the products involved in the manufacturing and recycling of the RB are allocated according to the mass distribution of the main materials composing the WTB.

2.2. Life cycle inventory (LCI)

This section describes in detail the stages, processes, and data used to compile the inventory for CB and RB used in onshore wind turbines. The LCI has been subdivided into several stages: resin production, blade manufacturing, use, and EoL.

The Bill of Materials (BoM) data required for the construction of the CB were collected from existing literature [27]. While this same baseline reference was applied to the RB, the data were modified and refined incorporating specific insights provided by the CIEMAT Wind Energy Group [32,34]. Primary data from the processes, material and energy balances of the manufacture of both blades on a laboratory scale have been used. Some energy and yield data have been scale up to industrial conditions using mathematical equations related to energy requirements and chemical process to make a more realistic comparison [35].

The specific processes related to the transport of raw materials to the manufacturing plant have been included within the background processes selected from the ecoinvent database, since all market processes already encompass the full set of activities associated with each material. However, transport processes from the manufacturing plants to the wind farm have not been included, as the RB is still under development and the most appropriate location for installing these new blades made from RR has not yet been defined. Furthermore, in similar case studies [24], CO₂ emissions from this transport do not exceed 1% of the total emissions; therefore, they have not been included within the system’s boundaries.

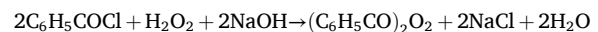
Electricity consumption in the stages developed is based on a model electricity mix scenario in Spain in 2025 (data from Red Eléctrica

Española) [36]. The proxy ecoinvent datasets used to represent these activities are detailed in the [supplementary material](#).

2.2.1. Resin production

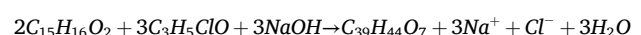
The RR (Akelite resin) is a reactive acrylic thermoplastic resin that polymerizes by adding 3% benzoyl peroxide as an initiator at a temperature of 70 °C for 2 h. After use, it can be recovered by simply bath it in acetone for 24 h at 23 °C. This polymer has been recently developed by ICTP- CSIC (Instituto de Ciencia y Tecnología de Polímeros – Consejo Superior de Investigaciones Científicas) [26]. Primary data for the manufacture of Akelite resin have been obtained from [23]. The main raw materials required for the manufacturing of the Akelite resin are acrylic monomers (0.97 kg/kg Akelite resin) and benzoyl peroxide (0.03 kg/kg Akelite resin). The main monomer used is methyl methacrylate (MMA), which is mixed with its polymer, poly (methyl methacrylate) (PMMA) to obtain the appropriate viscosity (MMA:PMMA ratio 89:11). Overall, 0.782 kg MMA and 0.118 kg PMMA are required to manufacture 1 kg of Akelite resin. Moreover, about 0.070 kg of methacrylic acid (MAA) is added per kg of resin to improve the resin formulation.

The LCIs to represent the production of these materials were selected from the ecoinvent database. Yet, ecoinvent does not contain an LCI for the production of benzoyl peroxide. Therefore, this process has been modelled considering the materials and energy requirements according to the literature [37]. Benzoyl peroxide is prepared by treating hydrogen peroxide with benzoyl chloride under alkaline conditions according to Reaction 1 using stoichiometric amounts.



Reaction 1. Benzoyl peroxide production.

With regard to CR (epoxy resin), the LCI is based on an existing European production process obtained from the ecoinvent database, which was modified to incorporate the Spanish electricity mix for 2025. In addition, an epoxy hardener was included in the inventory. CR is produced by reacting bisphenol-A and epichlorohydrin in presence of a base catalyst (here represented by sodium hydroxide, NaOH) according to Reaction 2. The process selected to represent the epoxy resin hardener was [market for hexamethylenediamine, GLO], as hexamethylenediamine is commonly used as an epoxy resin hardener [38]. The amount of epoxy hardener used to model this activity was set at 30 wt% of the epoxy resin mass, based on data reported in previous studies [34].



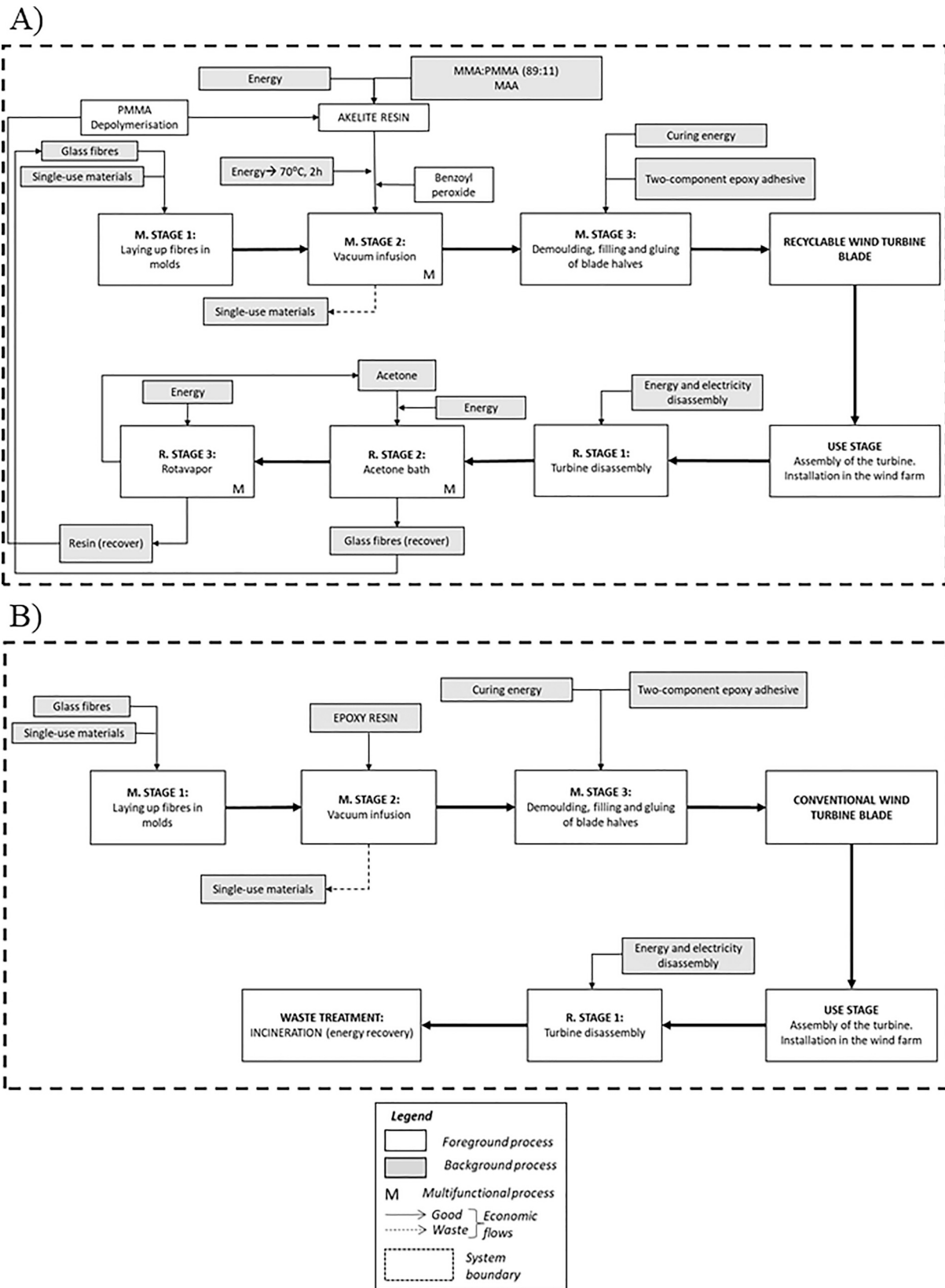


Fig. 2. Life cycle scheme of the product system of the RB (A) and CB (B).

Reaction 2. Epoxy resin production.

2.2.2. Blade manufacturing

The first three stages of the blade manufacturing process are similar

for the CB and the RB. The main difference appears in the second stage, which corresponds to the moment when the resin is incorporated into the glass fibres to form the reinforced composite material. CB uses CR, while RB uses RR.

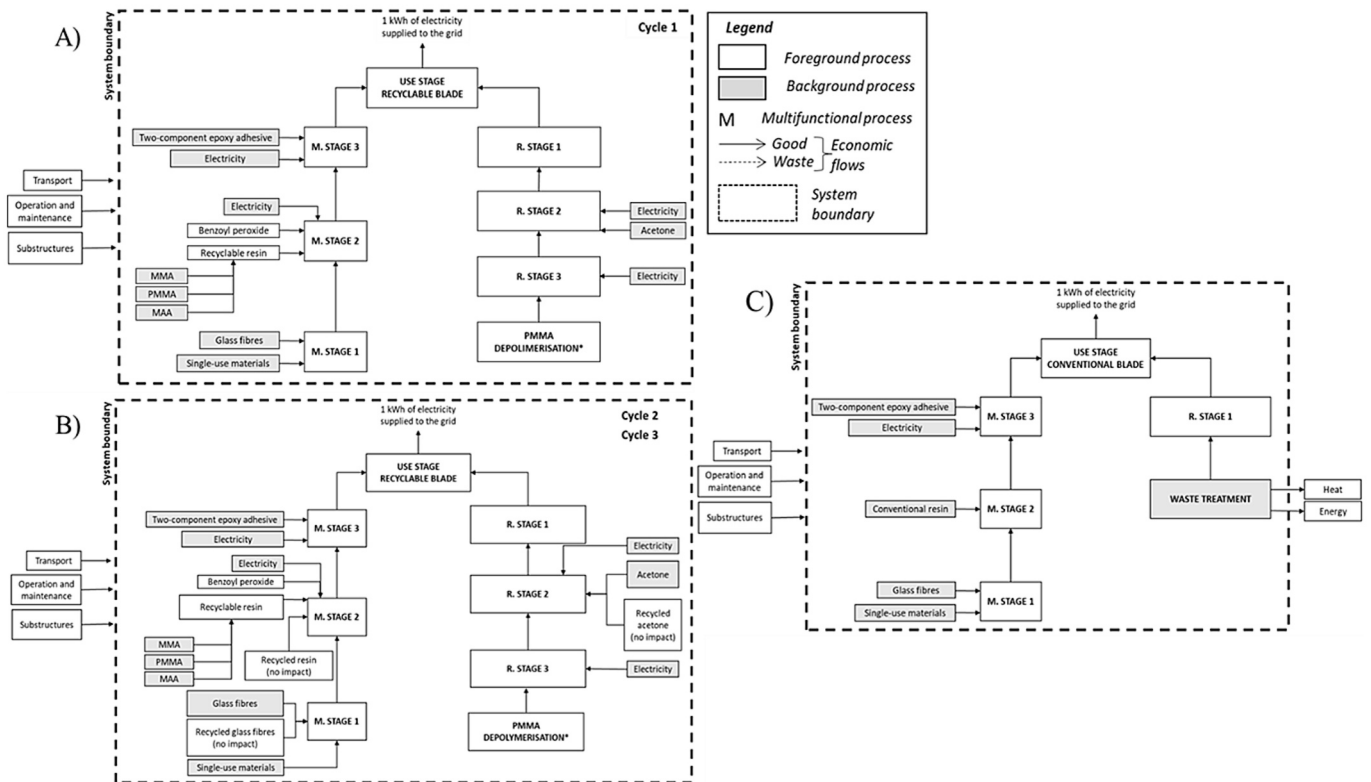


Fig. 3. Flow chart with system boundaries in the LCA showing the primary and secondary processes involved in RB manufacturing and recycling during cycle 1 (A) and during cycles 2 and 3 (B), and CB manufacturing and waste treatment (C).

- **Manufacturing Stage 1 (M Stage 1): Laying up fibres in molds.** This first stage consists of placing commercial glass fibres in the molds. This process is the same for both blades. The materials involved in this stage are commercial glass fibres and single-use materials (release agent film, coat UV protection, distribution mesh, peel ply, filling tube, vacuum tube, vacuum bag and taky tape). The LCIs for the manufacturing activities for these materials have been obtained from the ecoinvent database, and the amount of materials used was provided by our colleagues in Wind Energy Unit, in the Energy Department from CIEMAT.

The output of this production stage is the glass fibres strategically placed in the molds. It should be noted that the blades developed in this project are manufactured in two halves, which are assembled at the end of the manufacturing process.

- **Manufacturing Stage 2 (M Stage 2): Vacuum infusion.** The resins are incorporated into the glass fibres previously placed in the moulds using the technique VARTM (*Vacuum-Assisted Resin Transfer Molding*) [39] to create a reinforced hybrid material that achieves the specific properties of the WTB.

The main inputs for this stage are the fibres from M Stage 1 and the resins. For the RB, benzoyl peroxide is required. Moreover, a supply of electrical energy is required to maintain a constant temperature of 70 °C for 2 h (experimental conditions defined to ensure correct matrix polymerisation). In the case of a CB made from epoxy resin, the epoxy resin hardener has been included in the activity selected from the database. The activity selected for modeling the CB includes two activities: the epoxy resin activity [market for epoxy resin, liquid, RER] and the epoxy resin hardener activity [market for hexamethylenediamine, GLO], with a mass ratio of 1:0.3, respectively [34].

The main output in this stage is one-half of the WTB.

Regarding the single-use materials used in this phase, they are treated as municipal solid waste, since the minor non-hazardous laboratory waste experimentally generated can be assimilated to those. However, industrial deployment of the RB manufacturing could include different industrial waste pathways, such as segregation increasing recycling of a greater portion of wastes.

- **Manufacturing Stage 3 (M Stage 3): Unmolding, filling and glueing.** At the end of the previous stage, one-half of the WTB is obtained. Once both halves are manufactured, they are glued together to form the complete blade. This assembly is carried out using a two-component epoxy adhesive that requires heat for curing. In industrial blade manufacturing, an epoxy foam is incorporated as a filling material; however, in our study, this component was intentionally omitted in order to simplify the composition of the blades, limiting it to the glass fibres and the resin that, together, form the hybrid material.

2.2.3. Use stage

Once the blade manufacturing process using different resins has been completed, the WTB are incorporated into the wind turbine, with the aim of installing multiple units that will form an operational wind farm. The size of the blade used for the primary data collection was based on the literature [27], which described a blade 2 m long; however, when modeling the LCA, a full-scale blade was used. A real turbine has been considered based on the bibliographic data from [40], assuming both blades are installed in Nordex N60/1300 kW wind turbine, equipped with three WTB, with capacity factor of 42%, a rotor diameter of 60 m and a hug height of 60 m.

In other studies such as [24], the use phase of the wind turbine includes O&M activities, which are determined by two main factors: routine maintenance and operations derived from accidental damage. Results from those studies indicate that O&M would account for less than 1% of the carbon footprint. In our study, O&M activities were not

included, since no real repair tests were carried out to assess the maintenance requirements of the RBs throughout their service life. Anyway, it was assumed that evaluated blades would require the same type of maintenance, and it would likely have negligible influence on results and none over the comparative results.

2.2.4. EoL stage

When WTB reach the end of their service life, they follow a process of dismantling and decommissioning (R_Stage 1). The first aspect to consider is whether some of the equipment can be recovered, in our case, the blades. How this process is carried out depends on several factors, such as the location of the wind farm, the proximity of treatment facilities, and the number of turbines installed.

From this point onward, the two blades follow different paths. The CB is sent to the waste treatment process. Energy recovery incineration was selected as the EoL scenario, as it is one of the most common disposal routes, alongside landfilling. In addition, this option for CB leads to a higher impact in the climate change category, due to the significant emissions of pollutant gases. Therefore, this scenario is appropriate for assessing the avoided impacts associated with incineration [5,41]. The LCI for the incineration activity was obtained from the ecoinvent database.

The RB uses a simple recycling process in an acetone bath. The main goal of this treatment is to recover both the glass fibres and the RR that compose the blade, thereby allowing them to be reused in the manufacturing of new blades. The recycling process is as follows:

- Recycling Stage 2 (R_Stage 2): Acetone bath – Fibres recovery.

In the recycling process, the WTB is immersed in the acetone bath. The solvent interacts with the RR that forms the blade, dissolving it and thereby separating it from the glass fibres, which facilitates their recovery. The treatment lasts for 24 h at room temperature (23 °C).

Three main inputs are required for this process: the blade previously dismantled from the wind turbine, commercial acetone and the electricity needed to keep the bath at a constant temperature. The latter two inputs were modelled using LCI data from the ecoinvent database.

According to the literature [32] the resin-acetone mass ratio must be maintained at 1:20 (20 kg of acetone for each 1 kg of resin). This proportion reduces the amount of resin remaining on the recovered fibres, resulting in fibres that are more flexible and of higher quality, which improves their suitability for reuse in the manufacturing of new blades.

- Recycling Stage 3 (R_Stage 3): Rotavapor – Resin and acetone recovery.

The final stage of the recycling process focuses on recovering both the RR used in blade manufacturing and the acetone applied in the previous bath. This is achieved by recovering acetone via evaporation using a rotary evaporator (rotavapor). The equipment enables the separation of the two components by taking advantage of their different boiling points. The main input for this stage is the resin-acetone mixture obtained from R_Stage 2. In addition, electricity is required to ensure the proper operation of the rotavapor throughout the process.

After the recycling process, the two main materials composing the WTB are prepared for the manufacture of a new generation of WTB. On the one hand, the glass fibres recovered from the acetone bath are air-dried, and the different fibre fabrics belonging to the blade are separated. In other similar studies [42], glass fibres recovered through alternative recycling processes require an additional treatment to obtain a suitable format for reuse. However, in our study, the fibres recovered after the acetone bath maintain both their structural stability and continuity, making them suitable for the manufacture of a new blade without the need for further treatment. To support this observation, several photographs were taken during the laboratory-scale blade manufacturing process, showing the recovered glass fibres and

confirming that they retain an appropriate format for reuse (Fig. 4). The [supplementary material](#) includes additional photographs of the glass fibres recovery process. In addition to the attached images, previous published studies of the authors responsible for the manufacturing and recycling of the recyclable blade support the conclusion that the recovered glass fibres do not require any post-treatment. In the 2023 study [43], the mechanical properties of a 100% recycled WTB laminate obtained after the acetone bath were shown to be improved compared with those of a laminate manufactured using virgin glass fibres and virgin RR, as well as a laminate produced with a commercial epoxy resin. In the [supplementary material](#), a table has also been included showing the mechanical properties of the epoxy resin laminate and the Akelite resin laminates after the recycling process. A similar approach was followed in the 2025 study [44], in which carbon fibres were used instead.

On the other hand, the RR recovered from the rotary evaporator remains in a solid state, confirming that its main component is the polymerized monomer, namely PMMA. In order to reuse the RR, the polymer must undergo a depolymerization process to recover MMA [45]. The depolymerisation of PMMA is typically carried out by pyrolysis or controlled thermal degradation; this process allows macromolecular chains to break down, regenerating the monomer. This procedure generally involves three main stages:

1. PMMA preparation. Mechanical grinding of the polymer to improve heat transfer and remove impurities or additives present.
2. Thermal depolymerization. Heating of the material within a temperature range of 375–500 °C, depending on the specific methodology applied. The reaction takes place in an inert atmosphere to prevent unwanted secondary processes. Under standard conditions, it does not require catalysts.
3. Monomer recovery. Volatilized MMA is separated from the solid residue, then condenses through a cooling system and is purified by fractional distillation. This stage allows high recovery yields, in the range of 92–98% of the original monomer [46].

To model the resin depolymerisation process within the system boundaries, the study by Pender K. et al. [25] was used, in which an inventory for the depolymerization of a PMMA-based resin is developed. In a similar manner to the approach described in this literature, the process was modelled following the scheme shown in Fig. 5.

The recovery rates of the inputs after the recycling process are as follows: glass fibre has a recovery rate of 81.5%, the RR 88.6% and acetone 54.7% [32]. The relatively low acetone recovery rate is attributed to the fact that the recycling process is carried out at laboratory scale, where process efficiencies are typically lower than those achievable under industrial conditions. In this case, acetone recovery was performed under open-air conditions, which leads to significant losses



Fig. 4. Process for using glass fibres recovered from recycling to manufacture a second-generation WTB.

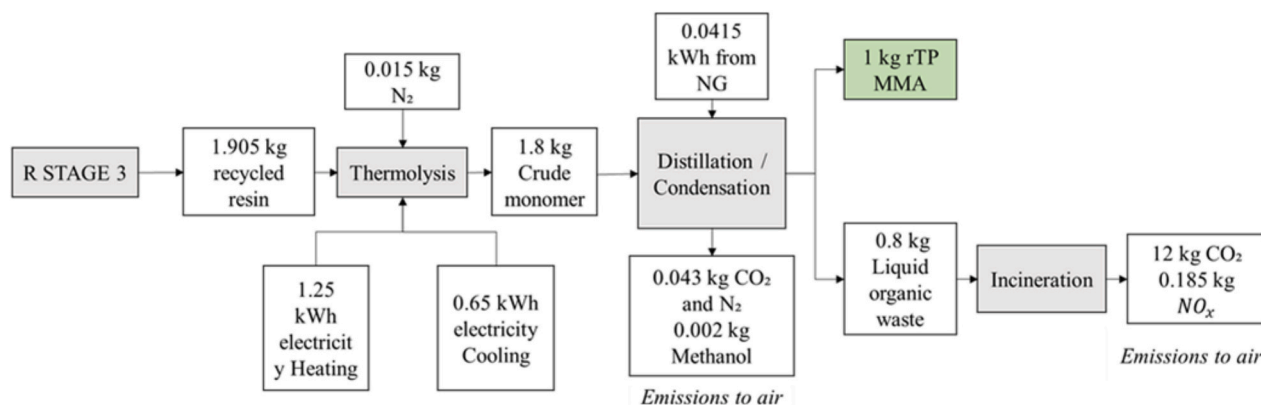


Fig. 5. Depolymerisation process of RR scaled up to 1 kg of recycled MMA (green box) as described in the literature [25].

due to the high volatility of this compound. However, since the present analysis is assumed to represent an industrial-scale scenario, an acetone recovery rate of 95% is considered [47], as this value is more representative of an optimized and controlled industrial process [46].

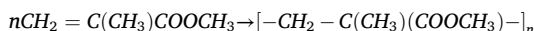
2.2.5. Cut-off criteria

The following section describes the processes that were excluded from the system boundaries in the environmental assessment. Regarding transport activities, transport associated with the production of raw materials (i.e., background processes) has been included within the system boundaries. However, transport related to the installation of the wind turbines at the wind farm, as well as their subsequent dismantling and decommissioning, has been excluded from the assessment. In addition, transport from the wind farm to waste treatment facilities has also been excluded from the system boundaries. The O&M activities of the WTBs throughout their service life were not included in the assessment. Similarly, secondary structural components of the WTBs (wood, foams, adhesives, metals), and core materials [41], were excluded, as the evaluated WTBs were assumed to be composed exclusively of glass fibres and resin. Furthermore, under the recycling approach assessed in this research, the recovered glass fibres do not require any additional pre-treatment before being reused in the manufacturing of new blades. Consequently, this process was not incorporated within the system boundaries.

2.2.6. Scaling

The LCI data for some processes have been scaled up from the laboratory level to the industrial level based on the bibliography [35].

- Manufacture of the thermoplastic resin, based on the polymerization reaction of MMA to obtain PMMA, using benzoyl peroxide at a concentration of 3% as the initiator. For this process, the benzoyl peroxide production reaction (Reaction 1) and the MMA polymerization reaction (Reaction 3) have been scaled up as follows [48].



Reaction 3. MMA polymerization reaction.

The quantities of reactants for the reactions were determined according to stoichiometric requirements. The amount of solvent was reduced by 20%, since at the industrial scale, it can be recovered through distillation processes.

- Scaling of the energy requirements for the RR manufacturing processes. To scale the benzoyl peroxide reaction, the reaction enthalpies of the reactants and products were used [49], allowing the calculation of the reaction heat per mole of benzoyl peroxide. Using the same approach, the reaction energy of the MMA polymerization process was calculated.

2.2.7. Modelling future electricity mixes

Data for the Spanish electricity mix in 2025 have been obtained from Red Eléctrica Española [36]. Future electricity mixes used for sensitivity analysis have been modelled using an energy system model of the Spanish energy system.

Energy system models are analytical tools used to explore plausible future configurations of the energy sector by simulating alternative technology, policy, and demand pathways [50]. In this work, the TIMES-Sinergia model is employed to estimate the potential electricity generation portfolio in Spain by 2050. TIMES-Sinergia is the most recent update of the TIMES framework tailored to the Spanish energy system, incorporating an explicit representation of technologies, fuels, and resource constraints across the full energy chain. The model identifies the lower cost system configuration that satisfies energy service demands while complying with imposed technical and policy constraints.

The resulting projections describe the 2050 electricity mix under a business-as-usual scenario aligned with the Spanish National Energy and Climate Plan (PNIEC) [51]. This scenario was developed within the ManOEUVRE project [52] and was designed to be robust to key uncertainty pillars [53], thereby providing a consistent reference case for subsequent analyses. The energy contribution to the electricity grid of each type of energy in each scenario is included in the [supplementary material](#).

2.3. Life cycle environmental impact assessment

The method used to assess the environmental impact of the WTB was Environmental Footprint 3.1 method included in the ecoinvent v3.10.1 database. This method includes the impact categories listed in [Table 1](#). The definition of each impact category can be found in the [supplementary material](#).

3. Results and discussion

In this section, the results of the environmental assessment of the two WTBs analyzed in this study are presented: the RB manufactured using RR and the CB manufactured using CR. First the results of the comparison of the resins production is presented. Second, the results of the complete life cycle per cycle and cumulative are showed. Third, the two-sensitivity analysis performed are exposed. Finally, limitations and future lines of research are discussed.

3.1. Environmental comparison of resin production

The LCA results for the production of 1 kg of the two resins (FU) used to manufacture WTB are shown in [Fig. 6](#).

The results indicate that neither resin performs better across all impact categories. For instance, in the CC impact category, the

Table 1
Impact Categories in Environmental Footprint 3.1 method.

Impact Categories	NAME	Units
Acidification	AC	mol H ⁺ – Eq
Climate change	CC	kg CO ₂ – Eq
Ecotoxicity: freshwater	ECOTOX-F	CTUe
Energy resources: non-renewable	ER	MJ net calorific value
Eutrophication: freshwater	EUT-fresh	kg P – Eq
Eutrophication: marine	EUT-marine	kg N – Eq
Eutrophication: terrestrial	EUT-ter	mol N – Eq
Human toxicity: carcinogenic	HTOX_c	CTUh
Human toxicity: non-carcinogenic	HTOX_nc	CTUh
Ionising radiation: human health	IR	kBq U235 – Eq
Land use	LU	dimensionless
Material resources: metals/minerals	RU-materials	kg Sb – Eq
Ozone depletion	ODP	kg CFC – 11 – Eq
Particulate matter formation	PMF	disease incidence
Photochemical oxidant formation: human health	POF	kg NMVOC – Eq
Water use	WU	m ³ world Eq deprived

production of RR generates almost the same impact compared to CR. Specifically, the CC impacts associated with RR production amount to 7.5 kg CO₂-eq, whereas CR production results in 7.8 kg CO₂-eq. The primary reason for the CC impact associated with RR lies in the consumption of MMA. The production of MMA is characterized by high energy requirements, which lead to significantly elevated GHG emissions.

In the AC impact category, the difference between the two resins is more pronounced, the production of 1 kg of RR has a 40% higher impact than that of CR. Once again, this result is mainly attributed to the MMA production process. In the EUT-marine and PMF categories, the trend is similar to AC. In ER, EUT-ter, POF, and WU, the differences are similar to those in CC, mainly due to the energy demand associated with the MMA production process.

In addition, the RR generates substantially lower impacts compared with the CR across a range of impact categories, including ECOTOX-F (90% lower), EUT-fresh (70% lower), HTOX_c (90% lower), HTOX_nc

(60% lower), and RU-materials (60% lower). This reduction in impacts is mainly due to the synthesis of bisphenol A process involved in the production of CR. This production demands high consumption of fossil resources and releases potentially harmful substances, which explains the improved performance of RR in these impact categories.

Moreover, implementing a recycling process for the RR reduces the amount of resin required for WTB manufacturing during the second and third life cycles of wind turbines. Thanks to the intrinsic properties of RR and the efficiency of its recycling process, this resin represents a highly promising sustainable alternative for application in WTB production within the wind energy sector.

3.2. Environmental comparison of CB and RB across cycles

The results of the environmental comparison of CB vs. RB in each time cycle of the 16 impact categories of the EF 3.1 method are shown in Fig. 7. The impacts of each cycle are disaggregated according to different processes: depolymerisation (only in RB), EoL, fibres, resin, and others (single-use materials, curing, etc.). These results demonstrate the impact of the amount of material required to produce 1 kWh fed into the grid in each of the cycles (FU).

To discuss the results, the 16 impact categories have been grouped into four groups: global and regional impacts (CC, ODP, AC and POF), ecosystem damage impacts (ECOTOX-F, EUT-fresh, EUT-marine and EUT-ter), health impacts (HTOX_c, HTOX_nc, IR and PMF) and resources depletion impacts (LU, RU-materials, ER and WU).

Within the “global and regional impacts” group corresponding to the first life cycle, the RB exhibits a higher environmental impact than the CB in the AC and POF categories, with values approximately 25% and 30% higher, respectively. This difference is mainly attributed to the production of MMA in the RR stage and to the resin depolymerization process carried out after recycling. Both processes require a considerable amount of energy, which significantly increases the impact in these categories. By contrast, in the CC and ODP categories, both blades generate a similar environmental impact. In this case, the impacts are primarily associated with the energy consumption of the manufacturing and recycling processes, including the resin depolymerization stage in the RB. After the second and third life cycles, the impact associated with

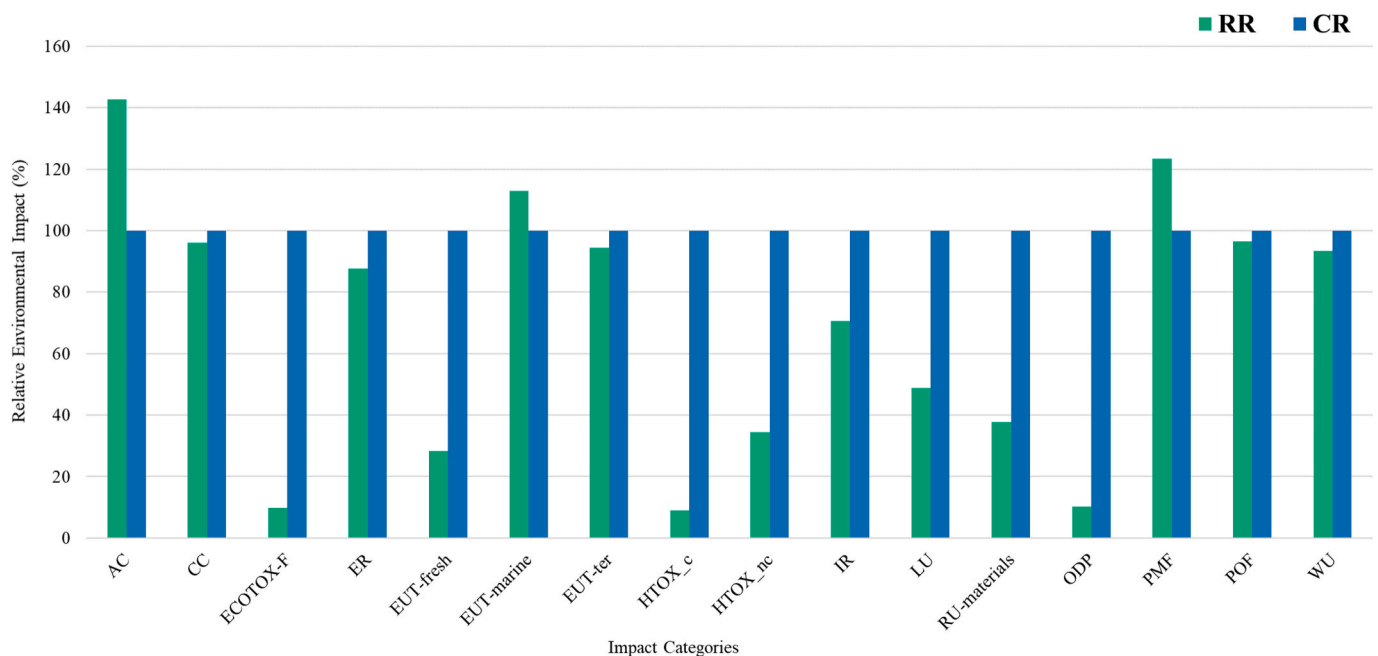


Fig. 6. Results of the environmental assessment of the manufacture of 1 kg of RR (green bars) vs. 1 kg CR (blue bars). Impacts are normalized to the CR manufacture, represented as 100%.

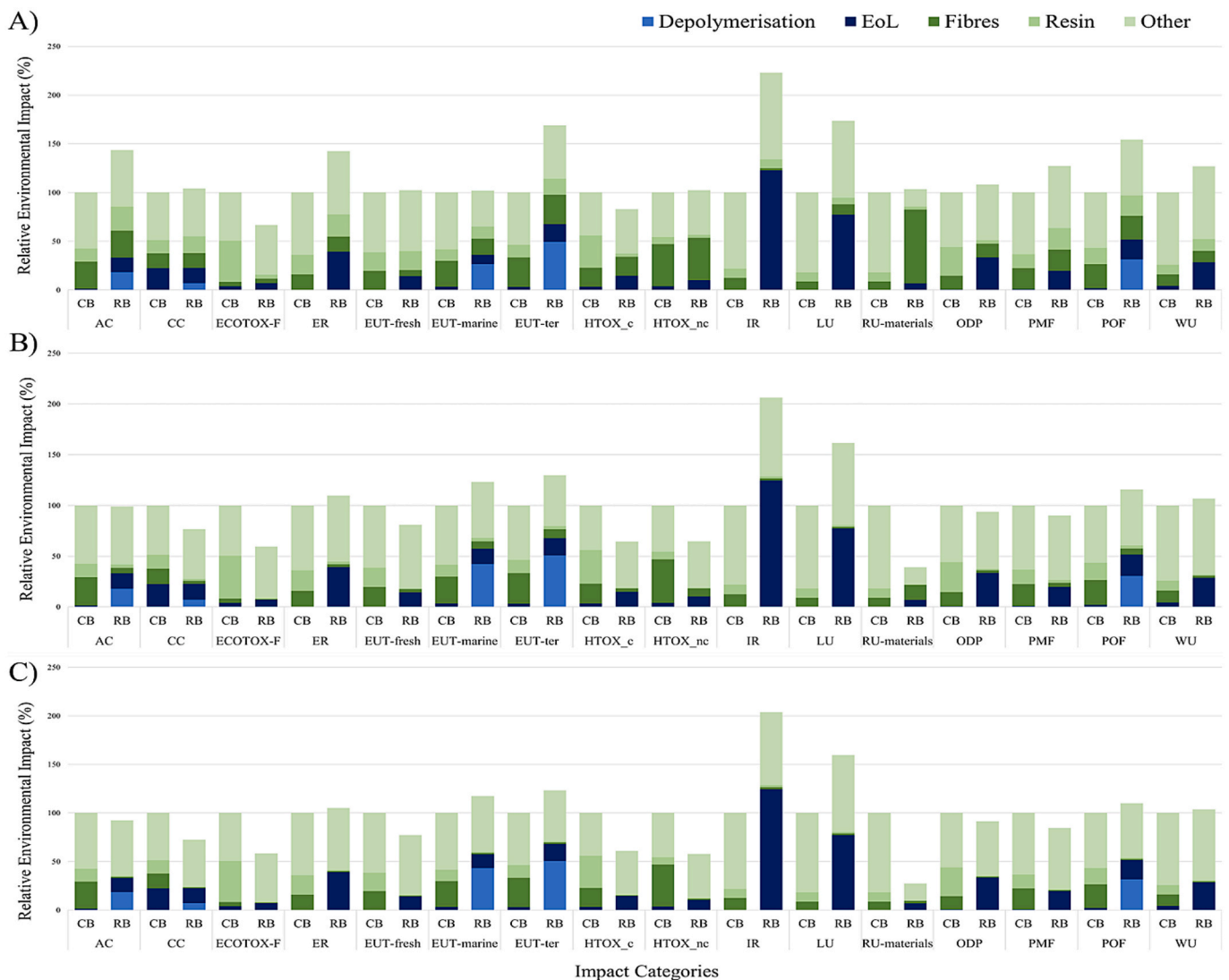


Fig. 7. Results of the environmental assessment of the 16 impact categories of the EF 3.1 method, for the amount of RB and CB material required to produce 1 kWh supplied to the grid in each of the cycles. Cycle 1 (A), Cycle 2 (B) and Cycle 3 (C). Impacts are normalized over the entire CB life cycle, represent as 100%. Light blue bars: PMMA depolymerisation (RB only), dark blue bars: EoL, dark green bars: glass fibre production, medium green bars: resin production, light green bars: other materials.

the CC and AC categories decreases substantially. The main reason for this reduction is the recycling process itself, since the recovery of the materials that compose the blade leads to a significant reduction in environmental burdens. However, the changes observed in the ODP and POF categories are less pronounced. As previously mentioned, this is due to the high energy demand of the resin depolymerization process required for resin reuse, which results in a considerable impact in these categories.

In the “ecosystem damage impacts” group, during the first life cycle, the CB generates a higher impact than the RB in the ECOTOX-F category. This result is mainly attributed to the production process of bisphenol A, a compound involved in the manufacture of the CR. The difference in impact between both blades in the first cycle is approximately 35%, whereas in the second and third cycles this difference increases to about 40% in both cases.

The three eutrophication impact categories show different trends. In the EUT-fresh category, both blades generate a very similar impact during the first life cycle. However, during the second and third cycles, the RB exhibits an environmental impact approximately 20% lower than that of the CB. This reduction is mainly due to the decreased requirement for virgin resin in the manufacturing of second and third

generation blades, enabled by the recycling process applied to the RB. In contrast, within the EUT-marine category, the CB generates a lower environmental impact than the RB throughout all three life cycles. This is primarily attributed to the significant impact associated with the resin depolymerisation process in the RR stage. A similar trend can be observed in the EUT-ter category, where the resin depolymerisation process in the RR stage also contributes considerably to the overall environmental impact. Nevertheless, when comparing the first and the final life cycles, it can be seen that the impact of the RB decreases due to the recycling process of the glass fibres.

Within the “health impacts” group, the IR impact category exhibits the most distinct behavior when comparing both blades. Throughout the three life cycles, the RB generates nearly twice the environmental impact of the CB in this category. The main reason for this result lies in the substantial contribution of nuclear energy within the electricity mix used to model the energy inputs involved in the RB recycling process. The use of nuclear energy is associated with an increase in ionizing radiation emissions, which negatively affects this impact category related to human health. Therefore, replacing this type of energy source with more sustainable alternatives, in terms of their effects on human health, would be advisable. In the sensitivity analysis section (section 3.4), an

additional environmental assessment is conducted using electricity mixes with a lower contribution from nuclear energy and a higher share of renewable energy sources, with the aim of evaluating the sensitivity of this parameter to the mitigation of this issue within the RB production process.

The remaining categories within the “health impacts” group (HTOX_c, HTOX_nc, and PMF) follow a similar trend. Due to the reduction in the use of virgin materials associated with the recycling process (glass fibers), the environmental impacts in these three categories decrease significantly.

The impact categories in the “resource depletion impacts” group do not follow a consistent overall trend; therefore, each category must be analyzed individually.

During the Cycle 1, in the WU category, the RB generates a higher impact than the CB, this is mainly associated with the recycling process, which involves relatively high electricity consumption. However, after Cycle 3, the impacts of the RB and CB become comparable in the WU category. This convergence is primarily due to the benefits of the reduction in impacts associated with the extraction and production of raw materials.

Across the three life cycles, in ER, the CB consistently generates a lower impact than the RB. Although the ER impact associated with the RB decreases by approximately 35% from Cycle 1 to Cycle 3, this WTB still exhibits a higher impact than the CB. The main reason for this result is the phenol production process via cumene oxidation, a co-product required for the manufacture of acetone used in the RB recycling process. This reaction demands a significant energy input, which is currently supplied by non-renewable energy sources. Consequently, more sustainable alternatives will need to be developed in the future in order to reduce this impact.

In LU, RB has a much more significant impact than CB throughout all life cycles. The RB generates approximately 70% more impact than the CB in the first cycle, and this difference decreases to about 55% in the second and third cycles. This result is mainly due to the use of non-renewable energy sources in processes that require high energy inputs, such as acetone production (via phenol production), as previously discussed in the ER category, as well as in recycling processes where the energy demand is linked to the electricity mix employed. This further highlights the need to perform a sensitivity analysis of the electricity mix used and to explore alternative energy sources.

Finally, the results for the RU-materials category are the most promising. During the first cycle, both the CB and the RB generate similar impacts; however, after the second and third cycles, the impact associated with the RB decreases by approximately 60% and 70%, respectively. The main reason for this reduction is that the primary materials composing the WTB, mainly glass fibres and resin, are recovered and reused in subsequent blade life cycles through the recycling process. These results demonstrate that blades manufactured using the new recyclable resin represent a promising alternative for the wind energy sector, owing to their eco-design and recyclability.

A comparison with other studies addressing the environmental assessment of WTB manufacturing reveals trends that are consistent with the findings of the present work. The study by Farazmandnia et al. [24] identified raw material production and blade manufacturing as the stages with the highest environmental impact, as they account for approximately 97% of total CO₂ emissions associated with the entire life cycle. These findings are consistent with the results obtained in the present study, which also highlights material production and manufacturing processes as the main factors contributing to the overall environmental impacts. Similarly, the study by Carallo, G.A., Casa, M., et al. [53], which evaluated the environmental performance of a recyclable wind turbine blade manufactured using an alternative recyclable resin through the Environmental Footprint 3.1 methodology, supports the conclusions of the present research. Their results indicate that the incorporation of RR systems represents a promising strategy for reducing the environmental footprint of wind turbine blades while

facilitating material recovery and reuse at the end of their service life. This analysis presents limitations that should be taken into account when interpreting the results. These limitations are explained in [section 3.5](#).

3.3. Cumulative environmental impacts of the three cycles

In the previous section, a comparative LCA results of the CB and RB was carried out for each of the three life cycles considered. In this section, the focus shifts to the assessment of the environmental impacts accumulated over the three life cycles.

The analysis addresses the following impact categories: CC and RU-materials ([Fig. 8](#)), providing a comprehensive evaluation of the overall environmental performance of the both alternatives from a long-term perspective. These two impact categories were chosen because are the most frequently analyzed when assessing the sustainability and eco-design of the new WTBs [2,22]. (The results of the cumulative impacts of all impact categories are presented in the [supplementary material](#)).

In CC, the results in the Cycle 1 show that the RB and CB generate approximately the same amount of CO₂ equivalent. After the completion of the second and third life cycles, the cumulative impact of the RB becomes lower than that of the CB. This fact is mainly attributed to the recovery of the materials that constitute the WTB. Through the recycling process, the RR and the glass fibres used in cycles 2 and 3 are sourced from recycled materials, thereby reducing the need for virgin raw materials and the environmental impact associated with them. As a result, the amount of CO₂ equivalent emissions is significantly reduced by the time the year 2075 is reached. The overall improvement is 30%.

In the RU-materials impact category, the RB shows a higher impact than the CB during Cycle 1. This fact is mainly attributed to two factors: the use of the recyclable resin, which entails additional material consumption associated with the polymerization of MMA; and the use of virgin glass fibre. However, when considering the cumulative RU-materials impacts of Cycles 1, 2 and 3, the CB exhibits a higher impact than the RB. The significant reduction observed for the RB after the first two cycles is explained by the incorporation of a large share of recycled materials in its manufacturing process, including glass fibres, resin, and the acetone used for recycling. The overall improvement is 50%.

When analyzing the cumulative impact over the three complete life cycles, it is evident that the impact associated with the CB is more than twice that of the RB. This results highlights the benefits of material recovery through the recycling process and their subsequent reuse in the production of successive WTB, reinforcing the advantages of the RB from a material resource efficiency perspective.

3.4. Sensitivity analysis

In this section, a sensitivity analysis is conducted on two key parameters that play a significant role in the environmental assessment of the RB based on RR. This sensitivity analysis is conducted exclusively for the manufacturing and recycling processes of the RB, with the aim of identifying key hotspots and potential improvement opportunities that may contribute to enhancing the sustainability and eco-design of the WTB's full life cycle.

The first parameter analyzed is the electricity mix used in the LCA. The baseline scenario assumes the Spanish electricity grid mix for the last available year at the start of the research (2025) as representative of the current WTB industry's performance. However, additional sensitivity analyses are performed to evaluate how the environmental impacts would change if a projected electricity mix for the year 2030, based on data from the National Integrated Energy and Climate Plan (PNIEC) [51], were applied, as well as if a prospective electricity mix for the year 2050, derived from a long-term energy system model [52], were used (The contribution of the three scenarios is shown in the [supplementary material](#)).

The second sensitivity analysis is carried out by varying the resin

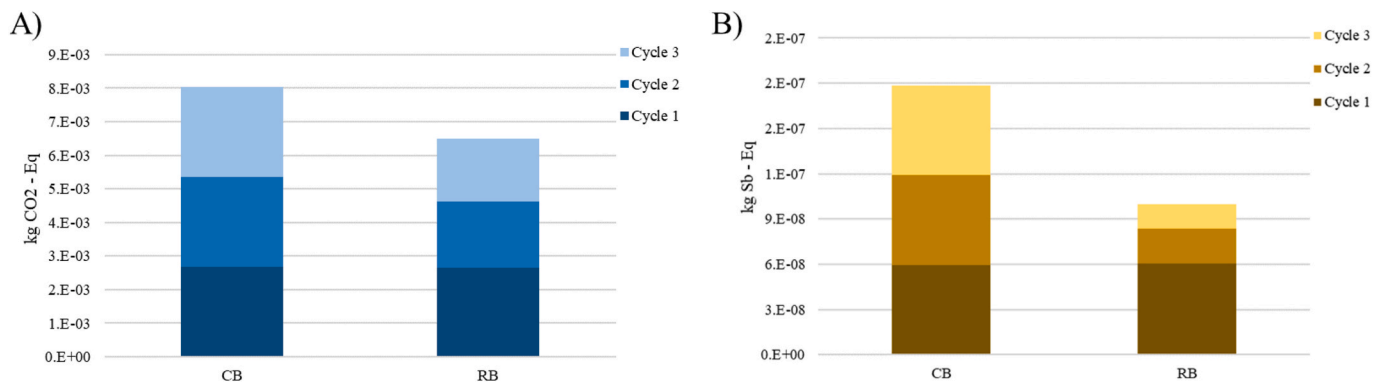


Fig. 8. Cumulative environmental impacts during the three life cycles of the conventional blade (CB) and the recyclable blade (RB) for the impact categories of CC (A) and RU-materials (B).

recovery rate in the recycling process. In the baseline scenario, a 95% recovery rate is assumed as a result of process scaling. Alternative scenarios are evaluated by considering a 54% recovery rate, corresponding to values obtained at the laboratory scale, and a 75% recovery rate, representing an intermediate scenario between the scaling scenario and the laboratory scenario.

3.4.1. Long term electricity mixes

A sensitivity analysis was conducted considering two different electricity mix scenarios, in order to assess the impact of changes in the electricity system on foreground processes. This approach was selected

because no additional assumptions were introduced for the background processes throughout the study, whereas several assumptions were made for the foreground processes. In particular, all foreground processes were modeled under a geographical limitation, assuming that they take place in Spain. This methodological decision has several important implications for the interpretation of the results. By applying the electricity mix variations only to the production processes of the both WTBs and EoL treatment, the influence of the electricity mix can be isolated specifically for these stages. As a result, the impacts directly associated with manufacturing and EoL can be more clearly identified, facilitating the evaluation of potential strategies to optimize both blade

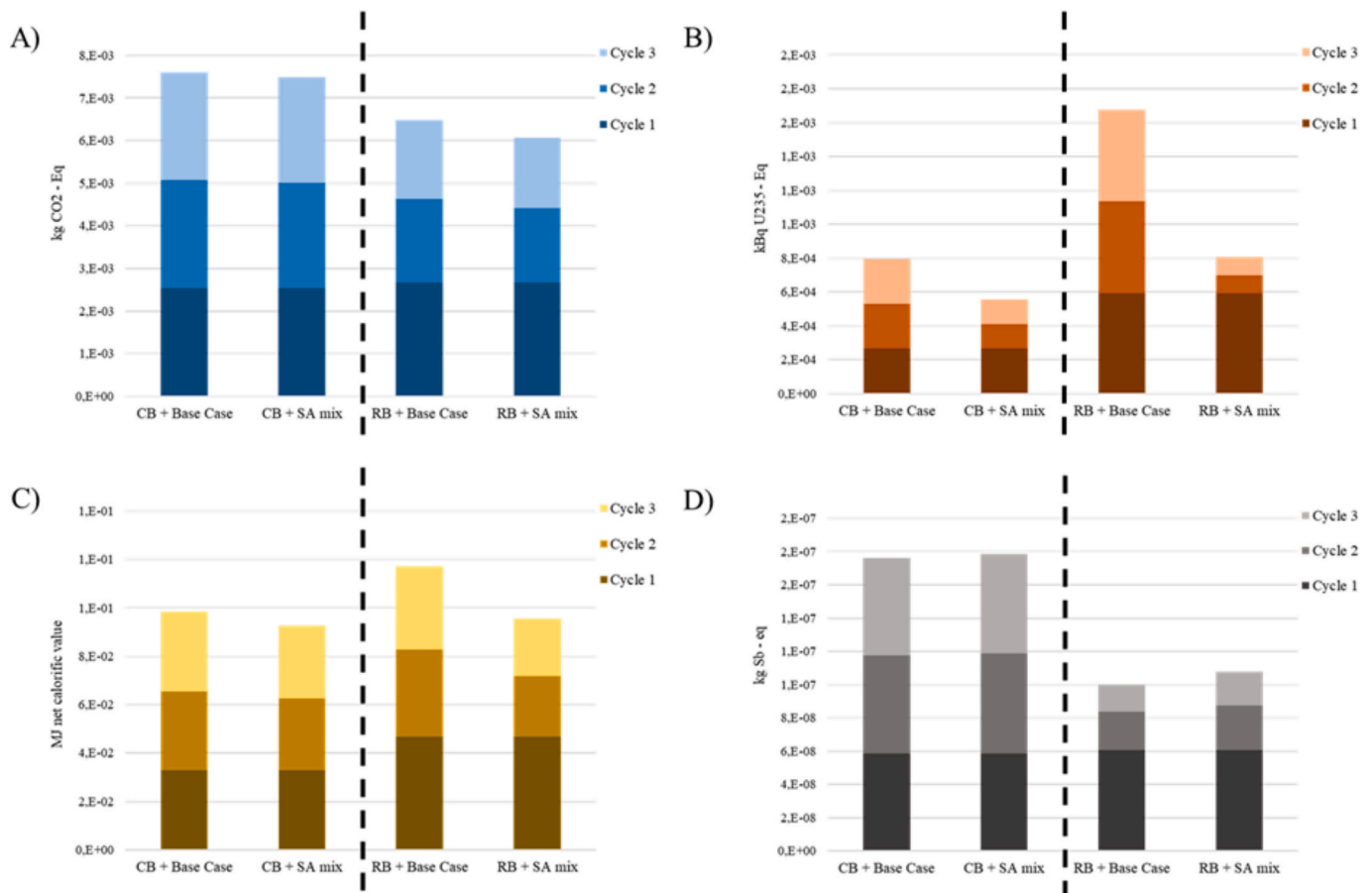


Fig. 9. Results of the cumulative LCA sensitivity analysis of long term- electricity mixes of RB and CB. Base Case: all cycles with 2025 electricity mix. SA mix: sensitivity analysis considering Cycle 1 with 2025 electricity mix and Cycles 2 and 3 with 2050 and 2100 electricity mixes. The impact categories are CC (A), IR (B), ER (C) and RU-materials (D).

production and waste management.

In this context, studies such as that conducted by Šimaitis et al. [53] have implemented prospective Life Cycle Assessment (pLCA) methodologies to incorporate the temporal dimension into the analysis of lithium-ion batteries. This approach integrates future scenarios derived from Integrated Assessment Models (IAMs) in order to project the evolution of global energy mixes under different climate trajectories.

The purpose of conducting a sensitivity analysis of the two energy mix scenarios in the foreground processes was to perform a cumulative LCA that included the energy mixes for the year 2050. Two scenarios were created to conduct the environmental assessment: Scenario 1, the 2025 electricity mix is applied to all stages of the both WTBs. Scenario 2, a temporal electricity mix is applied: the 2025 mix for Cycle 1, and the 2050 mix for Cycle 2 and Cycle 3. This approach allows for a comparison between using a single electricity mix across all cycles versus applying a changing mix adapted to each life cycle. The CC, IR, ER and RU-materials impact categories are chosen for the environmental assessment analysis, and the results are shown in Fig. 9. These impact categories were selected because similar studies have considered them [2,22]. In addition, the IR category was specifically included because, in the environmental comparison between the CB and the RB (Section 3.2), this category showed a particularly high impact for the RB. The results for the other impact categories are included in the [supplementary material](#).

The results obtained for these four impact categories exhibit a variable trend. When electricity mixes with a higher proportion of renewable energy sources are considered in the sensitivity analysis (SA mix), the environmental impact associated with the manufacturing and EoL processes decreases, except in the RU-materials. In this impact category, a higher contribution of renewable energy sources leads to an increase in environmental impact. This is mainly because renewable energy technologies require significant amounts of specific materials for their manufacturing and infrastructure. For example, the production of WTBs and photovoltaic panels involves technologies that require intensive use of materials. In addition, the deployment of these systems requires the use of various minerals and metals, such as aluminum, silicon, and lithium, whose extraction and processing contribute to increased impacts in the resource use category.

In the CC category, the results show that scenarios with a higher contribution of renewable energy sources in the electricity mix lead to lower CO₂ emissions across the three complete life cycles of both blade types. This improvement is mainly due to the reduced contribution of non-renewable energy sources, such as combined-cycle power plants and cogeneration systems. The reduction in GHG emissions is particularly significant during the third time cycle. In the case of IR category, a decrease in impact is already evident in the second cycle and becomes even more pronounced in the third. The main reason for this improvement is the elimination in the contribution of nuclear energy in the energy mix for the years 2050. For the ER impact category, the reduction in environmental impact becomes more evident during Cycle 3, although an improvement can already be observed in Cycle 2, trend similar to CC. This trend is mainly attributed to the decreased use of non-renewable energy sources and the increased share of cleaner energy sources, such as solar and wind power.

When the results of both WTB are compared simultaneously, it can be observed that, in the CC, IR, and ER categories, the SA mix scenario presents lower impacts than the Base Case scenario. However, for the RU-materials category, the SA mix scenario shows a higher impact than the Base Case scenario, mainly due to the demand for critical materials associated with renewable energy technologies. Despite this effect, the difference between the two scenarios does not exceed 5%.

This sensitivity analysis shows that the electricity mix used for the manufacture and EoL is a hot spot in our study. The results indicate that, as electricity generation in Spain incorporates a higher proportion of renewable energy sources, the manufacturing, operation, maintenance and EoL processes follow the same trend of improvement, resulting in a

progressive reduction in environmental impacts.

3.4.2. Improvement in acetone recovery

As mentioned above, the data used for the LCI of the RB were scaled up to industrial scale in order to allow for a more realistic and representative comparison with the CB. One of the key parameters in this scaling up to industrial scale was the acetone recovery rate in the recycling process. At the laboratory scale, the recovery rate was 54%, while in the LCA a recovery rate of 95% was assumed based on industrial-scale conditions [39]. The final sensitivity analysis of this study aims to evaluate how variations in this parameter influence the environmental performance results of the RB.

To this purpose, a comparative LCA has been carried out following the same methodological framework applied throughout the study. The scenarios considered are as follows: Conventional scenario (CB), Low scenario (RB with a recovery rate of 54%, corresponding to laboratory scale), High scenario (RB with a recovery rate of 95%, corresponding to industrial scale); and Medium scenario (recyclable blade with a recovery rate of 75%, representing an intermediate value between laboratory and industrial scales). All scenarios evaluated correspond to Cycle 3, in which EoL is assumed to occur in 2100. The results of this assessment are presented in Fig. 10.

The LCA results show that the High scenario generates the lowest environmental impacts across almost all the impact categories assessed. In contrast, Low scenario exhibits higher environmental impacts, in many cases even exceeding those of the Conventional scenario. This finding highlights that the acetone recovery rate is a critical hotspot within the scope of this research.

The trend observed in the CC category can be considered representative of other impact categories, such as AC, EUT-fresh, EUT-marine, EUT-ter, and HTOX_c. In all these categories, the Low scenario results in higher impacts than the Conventional scenario, whereas the High scenario consistently presents the lowest impacts among the four scenarios evaluated. The primary reason for these results lies in the substantial amount of virgin acetone required in the recycling process when the recovery rate is low. The production of virgin acetone involves the manufacture of phenol through the oxidation of cumene, a necessary co-product in the process. This reaction involves high energy requirements and additional chemical compounds, such as benzene and propylene, whose production is also associated with significant global and regional environmental impacts. Furthermore, cumene is classified as hazardous to both human health and ecosystems [54], which means that increased demand for this substance has a greater impact on categories related to health and ecosystems. Indeed, if a high recovery rate of this compound is not achieved, the conventional CB model is more environmentally sustainable than the alternative RB model.

In the RU-materials category, the Conventional scenario generates higher impacts, as no raw material recovery takes place in this case due to the absence of a recycling process. A similar trend is observed in the HTOX_{nc} category. The comparatively high impact of the Conventional scenario is primarily attributable to the use of glass fibres in the blade manufacturing process. In contrast, the trends observed in the IR and LU categories are less favorable to the recycling process. This is mainly due to the energy requirements associated with the RB recycling process. As noted in the previous sensitivity analysis, increasing the share of renewable energy sources used to supply the recycling process would reduce impacts in these categories.

3.5. Limitations and future key lines of research

The main limitations of this study reside in various factors. First, some processes such as transport process associated with the manufacturing and EoL stages of both CB and RB have been excluded from the analysis. Future research could incorporate these stages, as they may influence specific impact categories [2]. Secondly, the environmental comparison was made between a CB already commercially

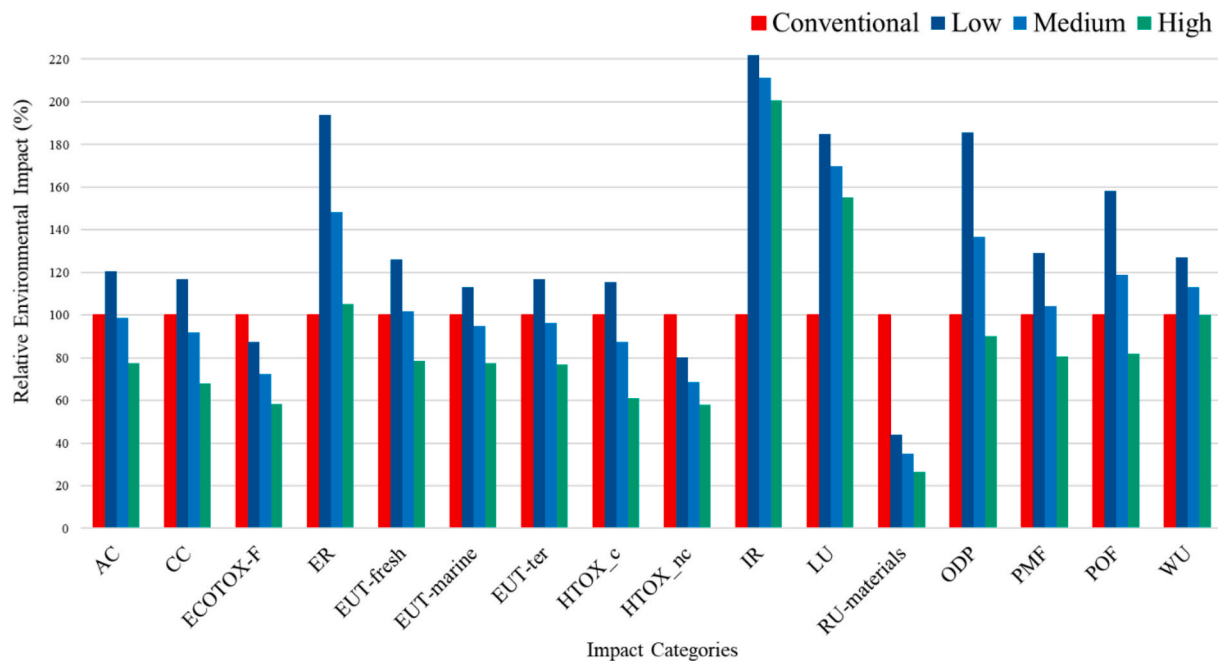


Fig. 10. Results of the environmental assessment of the 16 impact categories of the EF 3.1 method for the sensitivity analysis of acetone recovery. Conventional scenario, RB scenario with 54% recovery (Low scenario), RB scenario with 75% recovery (Medium scenario) and RB scenario with 95% recovery (High scenario). Impacts are normalized according to the conventional scenario and represented as 100%.

available and a RB still in the development phase. This may affect the comparability of the results, as the commercialization of the RB would likely involve the optimization of its parameters and the improvement of its environmental performance. However, there are case studies with other recyclable resins [55] that can serve as a reference for future research.

Furthermore, the RB model was based on a 2-metre WTB composed mainly of resin and glass fibres [29]. This represents a simplification that does not fully reflect the actual conditions of a commercial blade. Related to this, several studies, such as the work by Pender et al. in 2026 [42], have shown that recycled glass fibres require additional processing before being reused in order to recover their original shape and properties. However, as demonstrated in Fig. 4, this additional treatment was not necessary in the present study. Nevertheless, further testing on full-scale WTB will be required to confirm this finding.

The main limitation of the long-term electricity mix sensitivity analysis is that the alternative scenarios were applied exclusively to the foreground processes, while changes in the background processes were not considered. This sensitivity analysis was intentionally restricted to national processes in order to isolate the assessment to the manufacturing and EoL stages of the CB and RB. Future research could address this limitation by incorporating software such as PREMISE [56] to model the LCI with electricity mix variations applied to both foreground and background processes. This would enable a more comprehensive evaluation of the long-term environmental impacts associated with different electricity transition scenarios.

The LCI modeled in this study includes only glass fibres and resin as the main components of the WTBs. However, other studies have considered additional materials commonly present in real WTBs, such as PET, foams, coatings, and core materials [40]. Some components typically used in WTBs have been shown to separate easily during the acetone immersion step. Therefore, future assessments could expand the system boundaries to include both the manufacturing and EoL treatment of these additional components. Even, from an experimental perspective, future work aims to demonstrate that the new WTBs manufactured with thermoplastic resin can incorporate all these materials while still maintaining recyclability.

Finally, recycled materials generated at the end of cycle 3 (year 2100) as a result of recycling were excluded from the system boundaries due to the high uncertainty associated with their final destination over such a long time horizon, which could compromise the reliability of the results. Future studies could design an alternative final destination, such as their use for rural package [21] or glass fibre remelting [41].

4. Conclusions

This study conducted an environmental assessment using the LCA methodology to compare the complete life cycles of two WTBs: a CB manufactured from glass fibres and a traditional epoxy resin, and a RB manufactured from glass fibres and a novel recyclable thermoplastic resin currently under development Akelite. The analysis was based on a long-term time horizon spanning three consecutive 25 year cycles, from 2025 to 2100.

The results showed that, in the short term, the CB exhibited lower environmental impacts than the RB. This result is mainly due to the recycling and depolymerisation processes required for the RB, which involve substantial energy consumption. However, from a long-term perspective, the environmental impacts associated with these processes are more than offset by the benefits derived from material recovery and reuse. The recycling process allows a reduction of approximately 30% in CO₂ emissions across the three life cycles. Furthermore, it decreases impacts associated with raw material extraction by up to 50%. Although the manufacturing of RB is still in the development stage, the capacity to recover the main structural materials places this technology as a highly promising sustainable alternative for future deployment in wind farms, particularly once the process is optimized at an industrial scale.

The assessment also identified two key hotspots that significantly influence the overall sustainability of the system. The first is the electricity mix used during both manufacturing and recycling operations. The sensitivity analysis revealed that increasing the contribution of renewable energy sources substantially reduces the environmental impacts of both WTB. Future research should focus on minimizing the dependence on non-renewable energy sources throughout the WTB

production and recycling value chain. The second hotspot is the acetone recovery rate during the RB recycling process. The results demonstrated that higher acetone recovery rates lead to reductions in environmental impacts across all impact categories.

In our study, alternative EoL pathways for the CB were excluded for different reasons. First, the pyrolysis route [11] was not considered because the recovered fibres may require post-treatment, which could involve a substantial additional environmental impact. Moreover, this pathway theoretically results in the loss of the resin used in blade manufacturing. Another commonly reported alternative is the use of WTB waste as a filler in cement production [10]. However, if this pathway were considered as the final destination for the CB, none of the materials used in blade manufacturing would be recovered. In addition, the system boundaries would need to be expanded beyond the wind energy sector. This would introduce additional assumptions and increase the complexity of the comparative assessment. Therefore, incineration was considered the most appropriate EoL alternative for the CB within the scope and system boundaries defined in this study.

Another possible path for future research is the assessment of EoL management routes for conventional WTBs, which are predominantly manufactured using commercial epoxy resins. In this context, the R3POWER project [57], in which the authors are involved, aims to conduct an environmental and techno-economic assessment of some of the most advanced recycling technologies currently being developed for epoxy resin-based WTBs (mechanical recycling, chemical recycling, thermo-chemical recycling, and biological recycling).

Overall, this study demonstrates that the use of the RR as a substitute for CR represents a highly promising strategy for addressing the EoL challenges associated with WTB. Nevertheless, further research is required to optimize all processes and to facilitate large-scale industrial implementation within the wind energy sector.

CRedit authorship contribution statement

Pablo Atance-Gómez: Writing – original draft, Methodology, Investigation. **Ana Rosa Gamarra:** Supervision, Methodology, Investigation. **Yolanda Lechón:** Supervision, Methodology, Investigation. **Mario Martín-Gamboa:** Supervision. **Robert Istrate:** Supervision, Methodology. **Carlos Mantilla:** Investigation. **Rafael Carnicero:** Investigation. **Miguel Angel López-Manchado:** Investigation. **Raquel Verdejo:** Investigation.

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.

Acknowledgements

The authors acknowledge the financial support of the Agencia Española de Investigación (AEI) of the Ministry of Science, Innovation and Universities of the Spanish Government (MICIU) through the co-ordinated project SUCCESS BLADE, under Grand No. TED2021-130201B-C31 and TED2021-130201B-C33, co-funded by NextGenerationEU European Funds. Dr. Martín-Gamboa would also like to thank the Regional Government of Madrid for financial support (2023-T1/ECO-29202).

Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.compositesa.2026.110282>. The supplementary material includes several data and supporting information. First, a glossary with terms and abbreviations is provided. Second, graphical material of the recycling process of the WTB. Third, details of LCI processes. Fourth,

complete results of LCA, including the 16 impact categories, disaggregated by cycles and cumulative, under the different scenarios. Finally, additional information about the data supporting electricity mixes for scenarios are provided.

Data availability

Data will be made available on request.

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