




Full length article

Reducing the environmental footprint of the battery sector: assessment of a new recycling process from a techno-economic and life cycle perspective

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ARTICLE INFO

Keywords:

Direct recycling
Techno-economic assessment
Life cycle assessment
Circular economy
Batteries

ABSTRACT

The rising demand for lithium-ion batteries increases the need for sustainable end-of-life pathways that recover critical materials with low environmental burden. This study assesses a novel direct recycling process for recovering battery-grade graphite from spent NCA lithium-ion cells using primary data from a continuous pilot-scale plant. A combined techno-economic and life cycle assessment is performed, supported by sensitivity analysis, scenario analysis, and Monte Carlo simulation. The process delivers low environmental impacts, with a global warming value of 4.06 kg CO₂-eq per kg of graphite and achieves economic viability with a positive net present value at year ten. The minimum selling price is 9.87 €/kg when co-product revenues are included. The findings indicate that direct recycling provides an efficient and environmentally preferable route for high-quality graphite recovery. The technology can enhance circularity in battery value chains and reduce reliance on virgin resources in Europe and the United States.

1. Introduction

The global demand for lithium-ion batteries (LiBs) has risen due to the shift to renewable energy, prompting the EU to adopt sustainable and circular battery production strategies (Albertsen et al., 2021). Various programmes, such as Horizon 2020, the Battery Strategic Action Plan, and the Battery 2030+ initiative, support the entire battery life-cycle, from raw material extraction to recycling (Miao et al., 2022). However, a major sustainability challenge is the lack of strategies for valuing spent batteries. Despite a projected global demand for LiBs exceeding 2 million metric tonnes by 2030, <1% of spent batteries are currently recycled or reused (Ma et al., 2021), raising environmental and resource-efficiency concerns, especially due to limited availability of critical raw materials. Battery recycling plants face challenges like low yields, high costs, process complexity, difficulties in collecting and storing spent LiBs, and excessive water usage (Baum et al., 2022). Despite these challenges, recycling remains the most promising way to reduce the environmental impact of the batteries sector, recover critical materials, and support a circular battery value chain (Srinivasan et al.,

2025; Rezaei et al., 2025).

Battery recycling technologies are mainly classified into three main categories: pyrometallurgical, hydrometallurgical, and direct recycling processes. Pyrometallurgical routes involve high-temperature treatment to recover metals, resulting in energy-intensive operations, high CO₂ emissions, and reduced Li quantities, impacting environmental performance (Zhao et al., 2024; Pan and Shen, 2023). In contrast, hydrometallurgical techniques use strong chemical solutions to dissolve battery components, followed by separation and purification steps. While they achieve higher recovery yields than pyrometallurgical methods, their high chemical and process water use, along with slow processing rates, raise environmental concerns due to polluted waste streams (Vieceli et al., 2021; Davis and Demopoulos, 2023). Direct recycling is considered the most environmentally preferable option, with lower energy consumption, shorter process times, and high recovery rates. However, it faces technical challenges such as precise separation techniques, sensitivity to battery heterogeneity, high operational costs (Dobó et al., 2023; Wei et al., 2023), and complex, costly battery deactivation processes involving CO₂ or liquid nitrogen (Rezaei et al., 2025;

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Windisch-Kern et al., 2022).

Current battery recycling methods primarily focus on recovering cathode active materials, neglecting graphite, the dominant anode material in lithium-ion batteries. Graphite, crucial for energy, electronics, and metal manufacturing, accounts for 15–25% of battery mass (Dai et al., 2018) and plays a vital role in electrochemical performance. Its rapid growth in non-Ni/Co LFP chemistries is a significant concern. The global demand for graphite is projected to rise with the expansion of electric vehicles (IEA, 2023). Natural graphite reserves are geographically concentrated, posing supply risks and price volatility, while synthetic production is energy-intensive and contributes to greenhouse gas emissions (Ngoy et al., 2025; Olivetti et al., 2017).

Recovering graphite from spent lithium-ion batteries (LIBs) reduces resource usage, strengthens domestic supply, and lowers environmental impacts. However, technical challenges like contamination and structural degradation affect its reuse in high-performance applications (Gao et al., 2024). Innovative recycling technologies are needed to restore electrochemical properties while minimising energy and reagent consumption. Graphite recovery complements high-value metal extraction, enhancing integrated recycling processes (Das et al., 2025). Recent studies show that graphite inclusion in recycling strategies improves material recovery and reduces waste (Maheswari et al., 2025; Sahivirta et al., 2024).

Battery recycling technologies (hydrometallurgical, pyrometallurgical, direct) have been assessed in LCA and TEA studies for recovering cathode active materials and critical minerals. LCA studies showed significant variability in global warming potential (GWP) and environmental performance.

Hydrometallurgical recycling reduces GWP compared to conventional LIBs production. Du et al. (2022) reported 20.8 kg CO₂ eq./kg of ternary cathode material recovered, with electricity requirements causing most of the environmental impact. Wang et al. (2024) found values between 1.5–4.2 kg CO₂ eq./ton of spent batteries treated, implying a lower GWP compared to virgin manufacturing. Machala et al. (2025) found a 57.5% reduction in environmental impact compared to using virgin materials. Process design influences results, with Premathilake et al. (2024) finding that using deep eutectic solvents for leaching can increase GWP to 280.27 kg CO₂ eq./kg of recycled cathode, while electrolysis reduces it to 6.39 kg CO₂ eq./kg.

Pyrometallurgical processes, though energy-intensive, can yield environmental credits when integrated with closed-loop systems. Rajaeifar et al. (2021) reported, per tonne of recycled batteries, credits of –1200 to –770 kg CO₂ eq. for closed-loop recycling and –290 to 1410 kg CO₂ eq. for open-loop systems. Cusenza et al. (2019) confirmed that pyrometallurgical routes combined with hydrometallurgical steps enhance recovery efficiency and reduce impacts, noting recycling credits of –360 kg CO₂ eq. for an EV battery pack. Yu et al. (2021) found up to 35% GWP reduction compared to LIBs production when using mixed technologies. Also, Perocillo et al. (2025) found that coke consumption drives the impacts of pyrometallurgical recycling, while chemical use dominates the impacts of hydrometallurgical processes.

Zhou et al. (2025) estimated that pyro-hydro combined recycling could reduce GWP by 132 kg CO₂ eq./kWh recovered compared to other recycling technologies. Pražanová et al. (2025) demonstrated a 33% GWP reduction by incorporating secondary material recovery during pretreatment, compared to virgin material production. Also, Rinne et al. (2025) found that pretreatment of spent batteries enhances environmental benefits in pyrometallurgical recycling. Direct recycling generally offers lower energy consumption and emissions. Chen et al. (2023) reported GWP values ranging from 12.7 to 41.7 kg CO₂ eq./kWh of nominal capacity for LFP and NMC batteries, observing that electricity and nitrogen consumption are the main drivers for environmental impacts, while Kurz et al. (2021) found only 0.92 kg CO₂ eq./kg of NMC active material, with the purification stage contributing with the highest environmental load. On the other hand, Yang et al. (2024) demonstrated significant benefits of direct recycling with –12.13 kg CO₂ eq./kg of

recycled LIBs.

Several authors have investigated the economic performance of battery recycling technologies. Woeste et al. (2024) compared pyrometallurgical and hydrometallurgical processes, finding that energy and material costs drive operating expenses, with profitability depending on market prices for recovered materials and electricity price for pyrometallurgy. Choux et al. (2024) investigated direct recycling technologies, finding higher revenues compared to module-level processing, with potential revenues ranging from \$4.39 to \$10.15 per kg of recycled cells, depending on high-value material recovery rates. Vu et al. (2024) assessed industrial-scale hydrometallurgical recycling, highlighting the extraction section as the largest investment share and the price of recovered lithium carbonate (Li₂CO₃) as a key factor in process feasibility. Li et al. (2025) compared multiple recycling technologies, concluding that direct recycling offers the most advantageous economic performance (\$10.07, \$4.18, and \$2.64 revenue per kg treated for direct, hydrometallurgical, and pyrometallurgical recycling respectively). Earlier work by Gutsch and Leker (2024) combined environmental and economic assessments, showing that hydrometallurgical recycling reduces overall cell costs by 44% and global warming potential by 37%, while delivering substantial environmental benefits. Xiong et al. (2020) reinforced these findings, reporting that LIB remanufacturing remains viable until the purchase price of spent batteries exceeds \$2.87/kg, with cost savings of approximately \$1.87/kg of hydrometallurgical recycled cells.

The above paragraphs demonstrate the need to further explore direct recycling technologies since they have been indicated to lead to considerable economic and environmental benefits.

Existing LCA and TEA studies on battery recycling have largely focused on cathode material recovery, leaving a clear research gap on technologies targeting graphite. This study fills that gap by providing an integrated TEA–LCA and Monte Carlo analysis of a novel direct-recycling process for recovering battery-grade graphite from spent NCA cells. Using primary data from a semi-automated, continuous pilot-scale facility in Chicago, we build a detailed techno-economic and life-cycle inventory, offering new high-quality data to the literature. The process combines advanced disassembly operations (Kyselyova et al., 2025) with coal tar pitch treatment (McKinney et al., 2023), achieving >99% purity. The assessment highlights trade-offs between economic and environmental performance and evaluates the potential contribution of graphite recovery to strengthening material autonomy in Europe and the United States.

2. Methods

2.1. Battery recycling: process description, technologies and system configurations

A direct recycling process is developed to recover high-quality synthetic and natural graphite from spent cylindrical lithium-ion NCA battery cells (5000 mAh, 9.8 V). End-of-life cells are first discharged and mechanically disassembled. The graphite is then liberated from the Cu current collector with a recovery rate > 99%. The process yields approximately 1 kg of battery-grade graphite from 6.8 kg of waste cells. In addition to graphite, the process recovers 1.8 kg of recycled cathode concentrate and 0.06 kg of separator, although these byproducts require further treatment, which falls outside the scope of this study.

The recycling process begins with cryogenic deactivation to safely discharge the batteries (Fig. 1A). A custom rack is used to discharge an incoming lot of 100 cells simultaneously over the course of one hour. Following this, the cells are placed in a freezer at –20 °C for 30 min to complete the deactivation. This stage consumes approximately 0.50 kWh per battery. Once deactivated, the plastic casing is separated for recycling, while the stainless-steel components are downcycled.

Dismantling is carried out on a machine shop lathe equipment by cutting the top and bottom of each cell to access the internal electrode

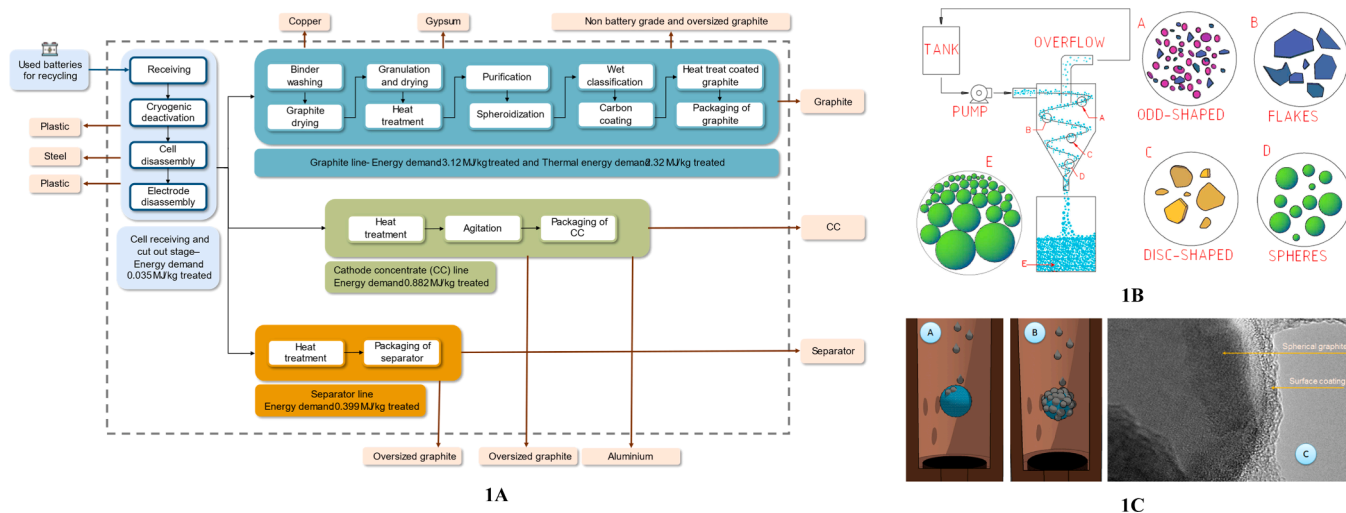


Fig. 1. 1A) Complete process diagram including the most important process stages as well as inputs and outputs of the process (CC: cathode concentrate). 1B) Segregation of graphite by shape and size in a wet classification process. 1C) Surface coating of spherical graphite precursor: (a-b) conceptual depiction of a coating process; (c) Transmission Electron Microscopy image of AETC's typical spherical nano-carbon coated natural crystalline flake graphite after the coating has been cured on graphite surface.

roll. The cathode, anode, and separator are then separated using machine tools, including lathes and carbide cutters. To scale up the process, automation is planned using picker-placer robots and sorting trays to classify components for subsequent stages.

Binder removal in graphite anodes is achieved by soaking the retrieved negative electrode rolls *en-masse* in warm water, which instantly dissolves the water-soluble polymer binder. After 3 to 5 s of exposure, graphite detaches from the copper foil, settles at the bottom of the vessel, and is collected and dried for further processing.

The cathodes are heat treated at 430 °C to prevent aluminum melting and lithium ignition, followed by ultrasonic treatment at room temperature to further enhance particle separation. Cathode materials are granulated and dried in a continuous fluidized bed dryer, then calcinated under nitrogen atmosphere to remove residual electrolytes. This process yields 99.25% cathode concentrate, while the remaining 0.75% is separated into recyclable plastics (5%) and aluminum (95%). This can then be converted into newly made cathode active material through electrowinning (Barsukov et al., 2023) and subsequent co-precipitation (Lantto et al., 2022; Tynjälä et al., 2023) in another facility.

Graphite purification is carried out in an electrothermal fluidized bed reactor operating at 2700 °C under nitrogen flow to prevent oxidation (Fedorov et al., 2015). This stage produces pure graphite, achieving battery-grade purity of greater than 99.95 wt.%, along with side streams of mineral impurities and gypsum, the latter resulting from neutralization reactions. Impurities are captured in a scrubber.

Purified graphite undergoes rapid re-spheroidization using mechanochemical equipment consisting of a rotating vessel and stationary stator. Operating at 4500 rpm, the system applies compressive forces to shape the graphite particles. A water-based slurry with surfactant is used to separate ultra-fine and spherical particles via hydrocyclone technology. The cut products are collected, allowing precise classification by size, shape and density (Fig. 1B). Each fraction is analyzed for surface area, particle size, purity, and packing density, enabling identification of grades suitable for anode applications and those requiring surface coating.

A key step in preparing graphite for lithium-ion battery applications is the surface coating of the classified spherical precursor powder. Typically composed of nanosized carbon, this coating reduces the BET surface area of the graphite, which helps minimize irreversible capacity loss and enhances battery safety. High BET values have been linked to catastrophic failures on the cathode side during post-thermal runaway diagnostics.

The coating is produced using nano-carbon technology with a graphitizable soft carbon precursor, forming a protective shell around each graphite particle (Fig. 1C). This shell allows lithium-ion transport while blocking unwanted species and suppresses exposed edges of graphene layers. The process involves carefully controlled heat treatment schedules, reaching temperatures above 1300 °C. Transmission electron microscopy shows a 20–70 nm thick, partially graphitic coating surrounding the spherical graphite core, with accessible pathways for lithium ions, which is an essential feature for anode functionality. Electrochemical tests confirmed the successful regeneration of the recovered graphite, which delivered a specific capacity of 350 mAh/g and maintained a cycling stability of > 99% over 300 cycles (Surace et al., 2025).

2.2. Techno-economic assessment

A techno-economic assessment was conducted to estimate the cost of recovering graphite from spent NCA batteries and to guide the development of direct recycling plants by identifying the main cost-drivers, in parallel with a Life Cycle Assessment that identifies the activities dominating the environmental impacts of the plant. The analysis therefore adopts a costing perspective and does not aim to provide a full-scale financial evaluation of an investment in such plants.

The lifetime of the plant is assumed to be 10 years with a capacity factor of 7500 h/year, and a discount rate of 8.7% is used (PWC, 2025). To convert USD to EUR, the 2024 exchange rate (0.92) was used. Operational costs arising from material, water, and energy consumption, as well as waste generation and materials used for maintenance purposes (or equipment replacement), are estimated primarily based on actual operational data and prices provided by the plant operators. When primary data was not available, literature data was used instead (Supplementary Excel file, SM). In contrast, personnel costs and capital expenditure were calculated using the Everbatt model for a generic direct recycling plant (Dai et al., 2019), adapted to the actual plant based on the amount of feedstock processed. In addition to the purchased equipment cost (PEC), the costs for installation, piping, instrumentation, electrical works, buildings and auxiliary facilities, service facilities, land, engineering and supervision, construction, and contingency were calculated as percentages of the PEC. The value of the equipment at the end of its lifetime was assumed to be zero. The relevant assumptions and exact values used in the economic and environmental analyses, together with the associated references, are listed in the

Supplementary Excel file (SM).

The costing analysis excludes several elements typically included in investment-oriented TEA studies, such as taxes, depreciation, and financing structure (debt/equity), as the objective is to identify techno-economic process-related hotspots and assess process feasibility in parallel with its environmental performance. Although this approach facilitates comparability and consistency with the LCA results, the omission of the investment financial structure may influence the TEA indicators. For example, including taxes and debt/equity financing would likely result in poorer economic performance. Therefore, the results of the techno-economic analysis should be interpreted as process-level metrics rather than estimates of investment profitability. The simplified equations used to calculate the techno-economic metrics considered: Net Present Value (NPV), cost of graphite production, and Minimum Selling Price (MSP) are presented below:

$$NPV = -CAPEX + \sum_t^T (R_t - Cop_t) * e^{-rst} \quad (1)$$

$$Cost\ of\ production = \frac{CAPEX + \sum_{t=1}^T Cop_t * e^{-rst}}{\sum_{t=1}^T Am_t * e^{-rst}} \quad (2)$$

$$MSP = \frac{CAPEX + \sum_{t=1}^T Cop_t - R_t * e^{-rst}}{\sum_{t=1}^T Am_t * e^{-rst}} \quad (3)$$

Where t denotes the operational years, T the lifetime of the plant (10 years), r the discount rate, CAPEX the capital expenditure, Cop_t the operational costs occurring in year t (material, water, energy consumption, waste generation, personnel, materials used for maintenance, an equipment replacement), R_t the revenues arising at year t from selling the products and co-products of the plant (Table 1), Am_t the amount of graphite produced at year t, and R*_t the revenues arising at year t from all co-products except of graphite. The difference between the cost of graphite production and MSP is that the latter considers revenues from selling the by-products.

The plant is considered to process annually 34,628 tonnes of spent battery cells resulting into 7500 tonnes of graphite going through the thermal purification furnace (see Fig. 1A) and eventually into 1255 tonnes of non-spherical graphite (sold into conductivity enhancement market) and 5074 tonnes of battery grade graphite. An analysis of the costs and revenues was performed to identify the processes causing the largest share of cost and generating the highest share of revenues. The lcpy package (Gkousis and Katsou, 2025) is used to perform the techno-economic modelling.

Local sensitivity and Monte Carlo analysis are performed to explore the uncertainty of the techno-economic analysis results. The results of the baseline analysis guide the selection of the parameters to be explored in the sensitivity and Monte Carlo analysis. For parameters having the same effect to the economic feasibility of the process, only one was

Table 1
By-products obtained from the production of 1 kg of graphite.

Process	By-product	Amount (kg)
Cell disassembly	Scrap steel	1.08
	Scrap nickel	0.028
	Scrap polypropylene	0.01
	Scrap nylon	0.015
	Scrap polyethylene	0.01
Binder washing	Scrap copper	0.715
	Purification	Gypsum
Carbon Coating	Agglomerated graphite	0.011
Packaging of anode	Non-spherical graphite	0.247
	Oversized graphite	0.01
Mechanical Agitation	Scrap Aluminium	0.77
Cathode packaging	Screened cathode concentrate	1.80
Cathode packaging	Oversized particles of aluminium	0.095
Separator packaging	Synthetic graphite component	0.056

selected. For example, the anode amount per cell, total graphite produced, and graphite price influence the revenues from graphite selling. The sensitivity analysis considers only the former which also affects other process parameters such as weight treated and electricity consumption. A total of eleven parameters were selected for sensitivity analysis: anode amount in cell, cell weight, number of batteries processed, capital expenditure, discount rate, tungsten consumption, personnel costs, deionized water consumption, electricity consumption, and revenues from by-products. These parameters are varied by 10% to identify how they influence the plant's techno-economic performance.

The parameters whose variation leads to the largest changes in the NPV and MSP of the plant are identified as critical and included in a Monte Carlo analysis. Table 2SM in the SI summarizes the selected parameters and their corresponding distributions. Triangular distributions are assigned to all variables with the lower and upper bounds defined based on scientific-based values (see Table 2SM). A total of 50,000 simulations were run to ensure convergence, using a random seed of 0. When varying the number of simulations between 40,000 and 50,000, the relative error in the calculated NPV was <2.5%. The parameters were allowed to vary independently, without considering correlations between their distributions. Furthermore, a scenario analysis is performed to investigate the process performance under different conditions. Five distinct scenarios are examined; S1: increasing the plant's operational lifetime to 15 years, S2: neglecting the marketable value of by-products, S3: assuming a purchase price for the feedstock (spent battery cells), S4: considering that the produced NCA black mass is ready for sale, S5: changing the selling price of the produced graphite.

2.3. Environmental analysis: life cycle assessment

The Life Cycle Assessment (LCA) methodology standardized in the ISO 14,040 series is considered for the environmental assessment. An LCA study consists of 4 phases: goal and scope definition, life cycle inventory, life cycle impact assessment, and interpretation of the results. Each of these stages is defined and described in the following subsections.

2.3.1. Definition of the goal and scope of the assessment

An attributional LCA approach is adopted to evaluate the environmental impacts of producing 1 kg of battery-grade graphite from spent batteries, being this the functional unit considered for assessment. A cradle-to-gate approach is considered for the system boundaries, which include all the value chain stages from the extraction of the raw materials to the production of the graphite (Fig. 1A). The transport of raw materials and the handling of waste generated have been also considered within the system boundaries.

Notably, producing 1 kg of battery-grade graphite through the recycling process also generates co-products, including 1.8 kg of recycled cathode concentrate and 0.06 kg of separator material, which could be utilized in subsequent processes to create valuable products.

To allocate the environmental impacts of the direct recycling process between the three co-products, the recommendations of ISO 14,044:2006 are followed (ISO, 2020). The process is subdivided to three production lines (cathode concentrate, graphite, and separator), assigning the impacts of each line to the product it relates to. Next, the impacts of the remaining four sub-processes (Fig. 1A, from receiving to disassembly) that cannot be further subdivided are allocated to the three co-products based on the mass extracted (a physical relationship between the products as mentioned in ISO 14,044:2006). Mass allocation is preferred over economic allocation following the recommendations of the ISO and because the produced cathode concentrate and separator need to be further processed before they have commercial value (e.g., the cathode concentrate is not market-grade NCA black mass yet). This assumption is in line with the recommendations of the Nickel Institute for allocation, to use economic allocation in case the price difference between the co-products is more than a factor of 5 which is not the case

for battery-grade graphite and NCA black mass (Nickel Institute, 2024), following a study on harmonizing LCA methods for the metal sector (Santero and Hendry, 2016). Table 1 lists additional scrap materials produced during the process and their amounts. The treatment of these scrap materials and their upgrading to valuable materials is not included in this analysis, following the end-of-life recycling approach (Nordelöf et al., 2019).

2.3.2. Life cycle inventory

The foreground inventory used consists mostly of primary data gathered from the owners and operators of the recycling plant. The detailed foreground inventory for each process line is summarized in the supplementary information along with additional information used for techno-economic calculations. For each process taking place during the recycling of the batteries, the materials, energy, and water used are considered along direct emissions of substances, waste generated, and by- or co-products produced.

The foreground inventory was first collected considering the processing of 5000 spent battery cells and was projected assuming an annual throughput of 500 million cells based on the projections of the plant owners and operators. According to plant owners and operators, the scale-up from the experimental 5000-cell inventory to an annual throughput of 500 million cells was performed using process-specific methods. Mechanical disassembly steps, such as lathe and grinder operations, were scaled linearly by using the measured processing time per cell and determining the number of mechanical units required to achieve the projected throughput. For powder-handling steps, the rate-limiting unit operation was used as the basis for scaling. In this stage, recovered graphite powder is treated in continuous fluidized-bed furnaces with a nameplate capacity of 1000 kg/hour. Although large-tonnage recycled graphite has not yet been processed, the same furnaces have been operated with non-recycled graphite of comparable purity. Because the impurity-removal mechanism depends on mineral content rather than on whether the graphite is primary or recycled, operating parameters (energy, chemicals, water, and gas flows) are considered equivalent. These prior experimental furnace data support the inventory values applied in the scaled-up model.

For the background inventory, the ecoinvent 3.9.1 database was used (Wernet et al., 2016).

The electricity mix of Illinois, United States is considered for the supply of the plant. Table 1SM presents the electricity mix based on data from the International Energy Agency. Background processes for the different energy carriers are sourced from the ecoinvent 3.9.1 database for the region. It is noteworthy that the calorific energy required for the drying step is recycled from the subsequent purification stage (Refer to Supplementary Material for further details). To model for coal tar pitch consumption, coal tar is considered assuming a production process efficiency of 0.83 (Crenna et al., 2021; Dunn et al., 2015). The solvents used during granulation and coating were modelled as NMP. Adhesives used during packaging for placing the materials in pallets are also considered, and their composition is provided in the Supplementary Excel file (SM). Finally, the volatiles produced during the purification and heat treatment steps are considered to be treated in a scrubber. An efficiency of 0.9 is considered for the scrubber (U.S. EPA, 2025) resulting in 10% of the volatiles emitted in the atmosphere as CO₂ and the remaining 90% being landfilled as inert material.

2.3.3. Environmental assessment

The Icp software (Gkousis and Katsou, 2025) is used for system modelling, LCI, and impact assessment. For the latter, the TRACI 2.1 method was used. Nine impact categories are evaluated: acidification potential (AP, kg SO₂ eq.), global warming potential (GWP100, kg CO₂ eq.), ecotoxicity freshwater (ECFW, CTUe), eutrophication potential (EP, kg N eq.), human toxicity carcinogenic (HTC, CTUh), human toxicity non-carcinogenic (HTNC, CTUh), ozone depletion potential (ODP, kg CFC11 eq), particulate matter formation potential (PMFP, kg

PM_{2.5} eq.) and maximum incremental reactivity (MIR, kg O₃ eq.).

2.3.4. Interpretation of results and sensitivity

A hotspot analysis was conducted to identify the processes contributing most to the overall impact and identify potential areas for improvement. The results of the hotspot analysis guided the selection parameters found to drive the environmental performance of the system. To identify the influence of these parameters on the environmental performance of the plant their values were varied by 10% in a local sensitivity analysis. The parameters found to influence the GWP the most were considered in a Monte Carlo analysis (following the approach outlined in Sartori et al.) to investigate the variability of the potential environmental impacts. A total of fourteen parameters were considered in the local sensitivity analysis: final graphite produced, cell weight, number of batteries processed, anode content per cell, electricity consumption, natural gas consumption, coal tar pitch consumption, scrubber efficiency, transportation distances for all materials, cathode content per cell, building housing area, nitrogen consumption, adhesive usage, and tungsten consumption. Of these, six parameters were varied in the Monte Carlo simulation of the LCA results. Each variable was modeled using a triangular distribution with the baseline as the mean and the maximum values set within realistically expected ranges. The variability ranges for the parameters are described in Table 2SM. In addition, two scenarios considering that the plant is powered by 100% solar- or wind-generated electricity are explored.

3. Results

3.1. Techno-economic assessment

The purchased equipment cost and total CAPEX are calculated using the Everbatt software (Dai et al., 2019), leading to a total capital expenditure of 316.7 million EUR. Table 4SM summarizes the different capital cost categories and their disaggregated contribution to the CAPEX (e.g., purchased equipment, installation, etc.). Moreover, the personnel costs were estimated using the Everbatt model and resulting into 3771,729 EUR annually, assigned equally between the different processes of the plant. Of the total incurred costs, 36% is due to operational activities and 64% due to the plant construction.

Fig. 2A shows the distribution of the operational cost of the process. Personnel expenses caused during all process stages contribute almost 16% of the total operational costs. The tungsten carbide tools used for cutting the cell during the cell disassembly stage drive the operational costs (31%), while a large portion of these is also caused due to deionized and grid water, mostly consumed during the wet classification stage. Electricity consumption contributes 7.8% of the operational costs, with the purification and carbon coating stages being the most electricity-intensive stages of the process. Notably, solvent use (during the granulation, and carbon coating stages) and petroleum pitch consumed (during the carbon coating stage) also contribute considerably to the operational expenses (4 and 5.6% respectively).

Accordingly, Fig. 2B shows the distribution of the revenues generated by the various products produced demonstrating that most of the revenues (69%) originate from the sale of the battery-grade graphite (main product), while a considerable portion (24%) is due to the sale of the non-spherical battery-grade graphite produced as by-product during the wet-classification step of the process which can be used as a component of electrically conductive matrix in NCA or NMC or LFP cathodes (Surace et al., 2025). Regarding the economic viability, the plant has a Net Present Value (NPV) of 4.2 million EUR, which becomes positive at the tenth operational year. The corresponding cost of graphite production is 15.18 €/kg, while when considering the revenue streams from the various co-products the minimum selling price (MSP) is at 9.87 €/kg.

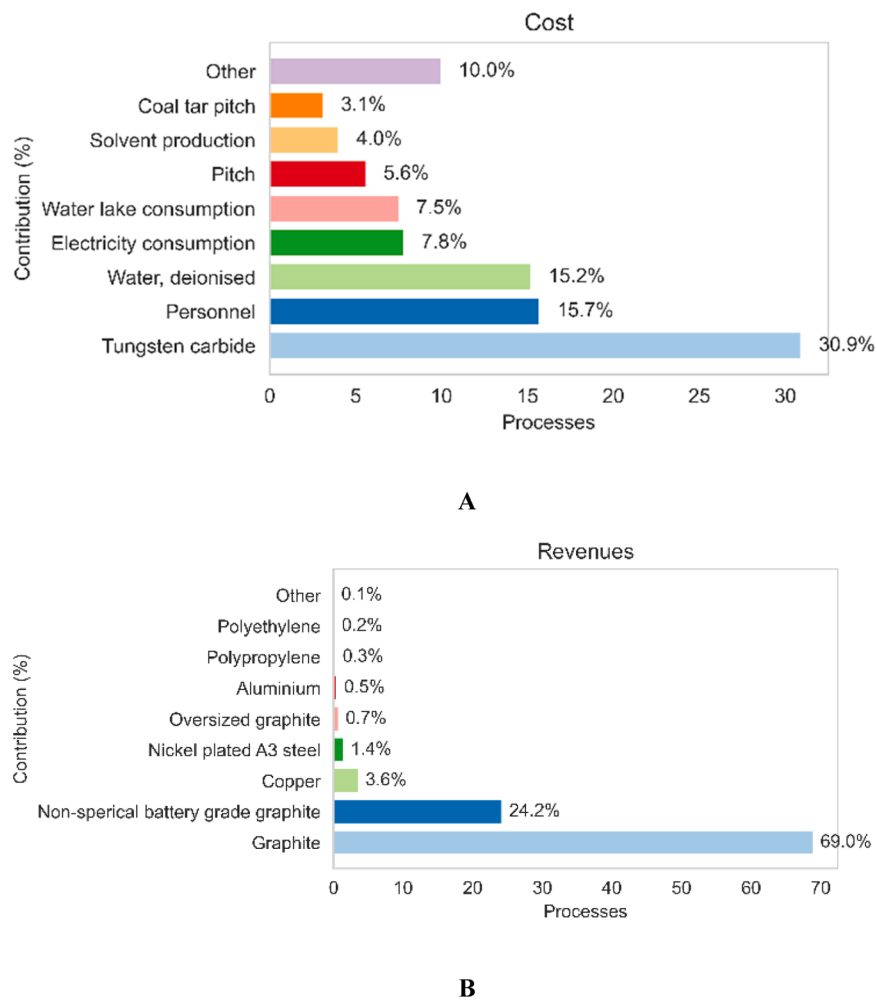


Fig. 2. Cost contribution (A) and revenues (B) distribution.

3.1.1. Sensitivity and scenario analysis

The sensitivity analysis shows that the parameters with the strongest effect on production costs are the amount of anode in the cell, cell weight, number of batteries, and final graphite produced (Figure 1SM). Reducing these parameters has a greater impact than equivalent increases, because operational costs are scaled with the number of batteries processed, but not entirely linearly, while CAPEX remains fixed. A 10% reduction in these parameters lead to over 10% increase for the production cost, while a 10% increase reduces the production cost by circa 9% (Figure 2SM). The CAPEX and discount rate significantly influence production costs as well, while costs such as tungsten consumption, personnel, and electricity have a limited direct impact on production costs, as they represent only a portion of the operational expenses.

To provide a more thorough picture of the feasibility of direct recycling technologies, five scenarios are investigated (Table 2). S1 shows that increasing the operational lifetime of the plant to fifteen years leads to a much higher NPV, and lower production costs and MSP. This is mainly because in this case more graphite is produced over the lifetime of the plant for the same initial investment. Notably, direct recycling plants can operate for periods longer than 15 years, a fact indicating the potential of the investigated technology. In contrast, when the revenues from the by-products are omitted (S2), the plant's NPV becomes negative, and the MSP becomes equal to the production cost, since no other revenues are considered. This scenario demonstrates the importance of extracting and utilizing the by-products for feasibility of the plant. S3 demonstrates the large value potential of the process,

Table 2

Alternative scenarios for assessing economic performance. Acronyms: NPV (Net Present Value), PC (Production Costs), MSP (Minimum Selling Price).

Scenario	NPV (M€)	PC (€/kg)	MSP (€/kg)
S0: Baseline	4.2	15.18	9.87
S1-Lifetime:15 years	88.3	13.15	7.84
S2: No revenues from by-products	-16.90	15.18	15.18
S3: Cathode concentrate price: 5700 EUR/tonne	674	15.18	N.A.
S4-1: Graphite Price 5000 EUR/tonne	-168	15.18	9.87
S4-2: Graphite Price: 15,000 EUR/tonne	176	15.18	9.87
S5-1: Spent battery price: 4000 EUR/tonne	-885	42.48	37.16
S5-2: Spent battery Price: 1500 EUR/tonne	-329	25.42	20.10

showing that if refinement processes for turning the cathode concentrate produced, to market-ready material with a value similar to black mass, the profitability of the plant can increase radically, making the extracted cathode concentrate the main source of revenue. A price of 5700 EUR/tonne was considered for S3, though a potential sale of the produced cathode concentrate can have a large impact on the plant's NPV even for lower price since the amount of produced cathode concentrate is almost double that of graphite. Moreover, S4 shows the sensitivity of the plant's performance to the graphite price, despite this does not influence the production costs or MSP. It is shown that in case the selling price of battery-grade graphite reduces, the NPV of the plant becomes negative, a fact highlighting the importance of ensuring stable prices and other

sources of revenue (e.g., from the recycled cathode concentrate) for widening the adoption of such technologies. This observation is also strengthened by the results of S5 which shows that in case the spent battery cells need to be bought, the plant's NPV becomes negative and the production cost and MSP increase considerably. Therefore, to derisk investment in such technologies the other extracted materials need to be also utilized for revenue.

3.1.2. Monte Carlo simulation

To assess the variability of the techno-economic results, a MC analysis was performed considering the variability of the parameters described in **Table 2SM**, based on the results of the sensitivity analysis presented in the previous paragraph. **Figure 2SM-A** shows the MC results for the cost of production and NPV. The results indicate that in a significant portion (42.4%) of the simulations the NPV is negative. These negative outcomes correspond to scenarios with lower cell weight and reduced anode content, resulting in a lower amount of graphite produced while CAPEX and fixed operational costs remain unchanged, leading to lower NPV and higher production costs. However, when the plant lifetime is extended to 15 years, the economic performance of the plant improves considerably, as revenue is generated for an additional five years compared to the baseline case (**Figure 2SM-B**).

3.2. Environmental analysis

Table 3 shows the environmental impacts for producing 1 kg of battery-grade graphite with the direct recycling process of spent NCA battery cells. The hotspot analysis for the graphite production (**Fig. 3**) demonstrates the stages and processes that drive the impacts. The GWP impact is driven by the binder washing, purification, and wet classification stages, with drying, coating, and heat treatment processes also contributing non-negligibly (detailed values can be found in **Table 4SM**). **Fig. 3b** shows that the consumption of electricity and natural gas is what causes the majority (over 75%) of the GWP. The cells are first received, discharged, and disassembled to separator, and anode and cathode electrode materials, with relatively low energy intensity for these phases (0.035 MJ of electricity per kg of treated cell, **Fig. 1A**). The energy needs of the anode material processing to graphite are much higher (3.1 MJ of electricity and 2.3 MJ of thermal energy per kg of treated cell), driving the GWP of graphite production. The thermal energy is almost entirely consumed at a boiler during the binder washing stage (with copper being also produced as a by-product, **Fig. 1A**), explaining the increased contribution of the binder washing stage to the GWP impact (**Fig. 3a**). In contrast, the majority of the electricity is consumed at the purification and wet classification stages (over 65%) at the kilns used (**Fig. 1A**). The relatively high electricity consumption in these stages is the reason of their increased contribution to the GWP impact. Moreover, the heat treatment and carbon coating stages (**Fig. 1A**) also consume a large amount of electricity (over 25%) at the compressors and kilns used, explaining their non-negligible contribution to the GWP impact.

Table 3

LCA results for the three products produced.

Impact		1 kg graphite	1.8 kg cathode concentrate	0.056 kg separator
AP	kg SO ₂ eq	1.09·10 ⁻²	3.63·10 ⁻³	7.87·10 ⁻⁴
GWP100	kg CO ₂ eq	4.06	0.82	0.28
ECFW	CTUe	46.72	35.34	3.37
EP	kg N eq	1.37·10 ⁻²	3.79·10 ⁻³	8.86·10 ⁻⁴
HTC	CTUh	2.72·10 ⁻⁷	3.04·10 ⁻⁷	1.99·10 ⁻⁸
HTNC	CTUh	1.12·10 ⁻⁶	9.23·10 ⁻⁷	7.82·10 ⁻⁸
ODP	kg CFC11eq	1.97·10 ⁻⁷	8.11·10 ⁻⁹	1.41·10 ⁻⁹
PMFP	kg PM _{2.5} eq	2.19·10 ⁻³	6.61·10 ⁻⁴	7.08·10 ⁻⁵
MIR	kg O ₃ eq	1.52·10 ⁻¹	4.09·10 ⁻²	8.64·10 ⁻³

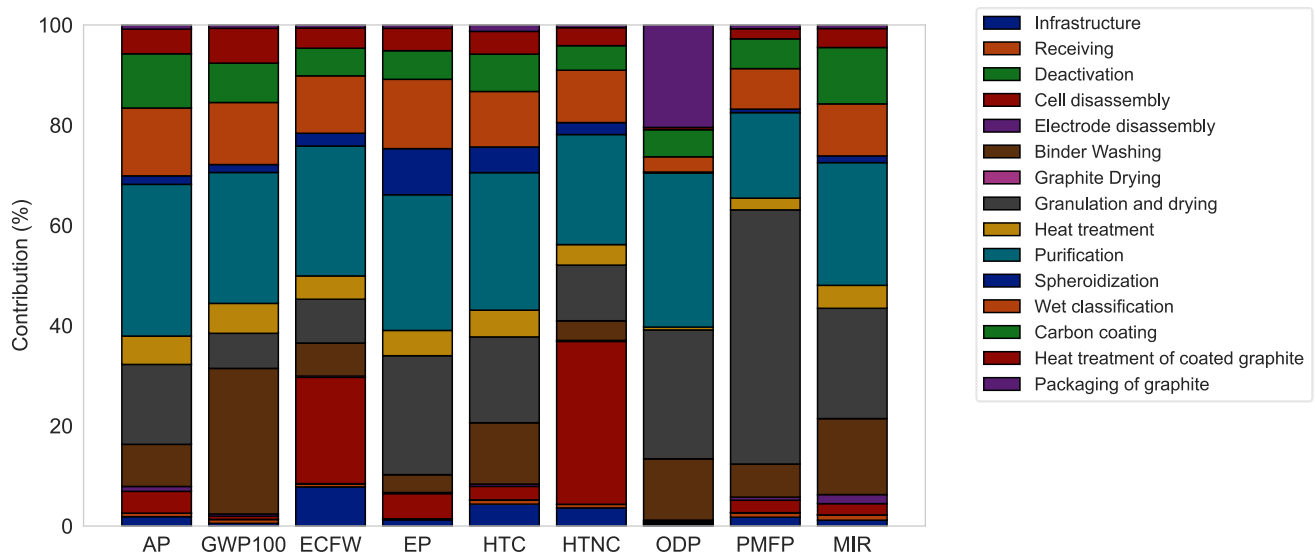
Fig. 3b demonstrates that the electricity and natural gas consumption are major contributors to all impact categories investigated. Coal tar pitch and tungsten consumption are also shown causing a significant portion of several impact categories with the former causing 10–25% of the AP, EP, HTC, HTNC, ODP, and MIR impacts and 45% of the PMFP impact, and the latter causing 21% and 32% of the ECFW and HTNC impacts respectively. Coal tar pitch is added to dried graphite during the granulation stage (**Fig. 1A**), explaining the high contribution of this stage at the PMFP and other impact categories (**Fig. 3a**). By contrast, the tungsten carbide refers to cutting tools used for the cell disassembly (**Fig. 1A**), leading to its high contribution to the ECFW and HTNC impacts. Finally, it is shown that the NaOH consumed (mostly for purification purposes) drive, along with the coal tar pitch consumed, the ODP impact.

The energy (electricity and natural gas) consumption drives the environmental profile of graphite production, with materials consumed (such as tungsten, coal tar pitch, and chemicals) also contributing considerably in some impact categories. **Figures 3–6SM** and **Tables 5SM–8SM** show the contribution analysis values for the stages driving the GWP impact (purification, classification, binder washing, and granulation, **Fig. 1A**). The investigated technology focuses on the production of battery-grade graphite from the spent battery cells, and thus most of the processes concern the processing of anode electrode to graphite. The cathode concentrate and separator material are processed slightly to a point that can be sent to other plants for further processing and recovery. The GWP impact of the cathode concentrate line is dominated by the electricity consumed during its heat treatment and agitation of the electrode, while for the separator line it is dominated by the electricity consumed for its heat treatment (**Fig. 1A**). As shown in **Fig. 1A**, the cathode line is responsible for the majority of the process's direct energy consumption (100% of the thermal energy and 70% of the electricity), with the cathode and separator lines consuming 20% and 9% of the total electricity respectively.

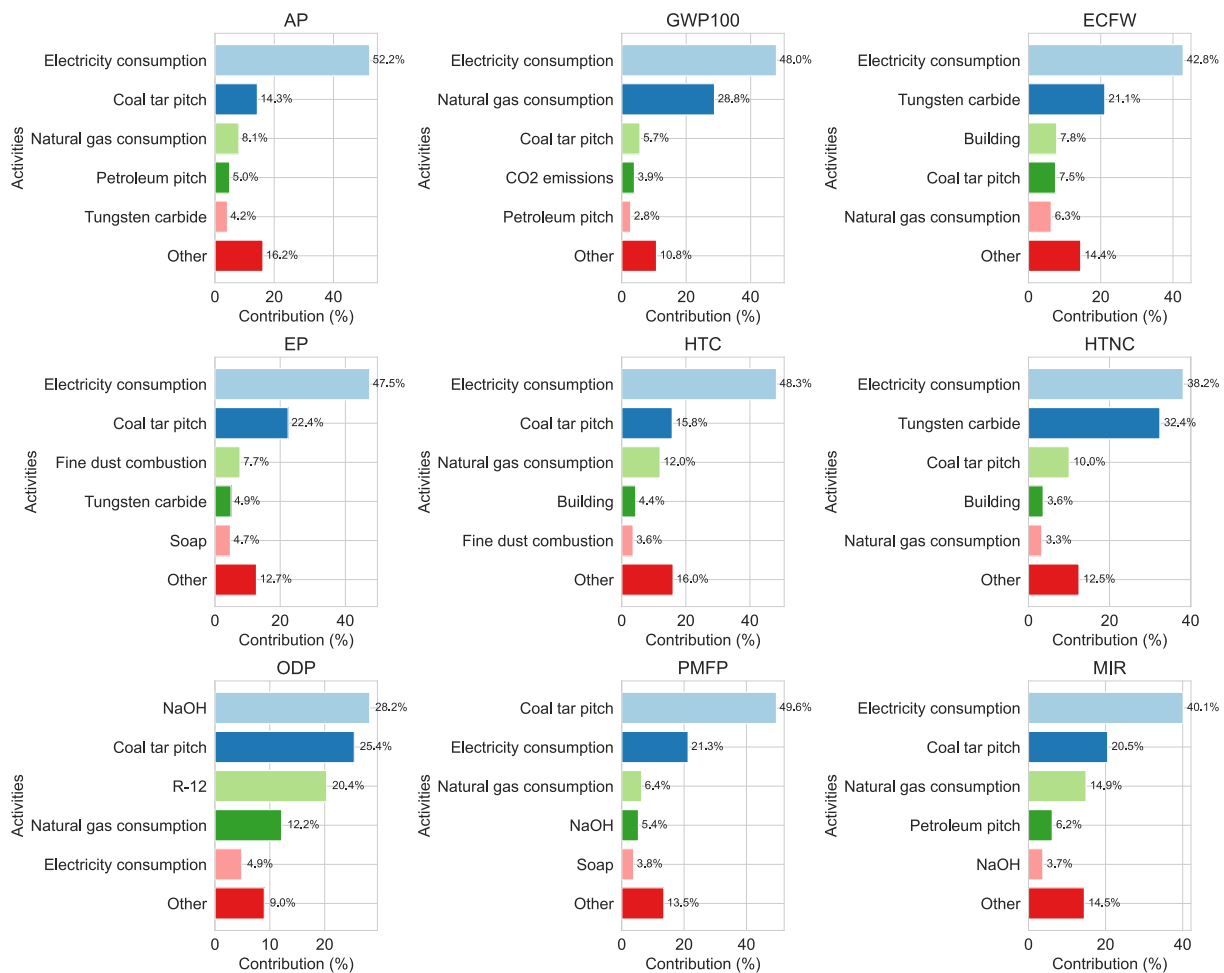
Two scenarios were explored considering alternative supply sources for the electricity of the plant: powered entirely by solar or wind power. In both cases the environmental impacts of the plant reduce, with higher reductions noted for the case considering wind energy supply. For the GWP a reduction of 38% and 46% is observed for solar and wind energy respectively. It is observed that the choice of wind energy is the best alternative, as its impacts are significantly lower compared to solar energy. Notably, in some impact categories, the use of solar renewable energy leads to a higher environmental contribution than the base case, such as the ECFW, ODP and PMFP, mainly due to the high consumption of mineral resources required for the manufacturing of the solar panels. The detailed numerical results for these two additional scenarios are presented in **Table 9SM**. An additional scenario was considered in which the impacts of coal tar pitch production are set to zero, as it was identified as a major impact driver and is a by-product of coke production. The impacts are reduced in this scenario with the greatest reduction, compared to the base case, in environmental impact being observed in PMFP, HTC, and EP impacts that are mostly affected by the coal tar pitch consumption respectively.

3.2.1. Sensitivity assessment

A sensitivity analysis was conducted to identify the parameters with the greatest influence on environmental impacts. Fourteen parameters were varied locally by 10% as described in **Section 2.3.4**. **Figures 7SM–15SM** in the SM present the sensitivity analysis results. The parameters show a relatively slight effect on environmental impacts. This is because electricity and material consumption, as well as the final graphite output, are largely proportional to the input batteries. Reducing the graphite produced by 10% results in less than a 10% increase in GWP, due to the mass allocation of impacts from the first four process stages among the cathode concentrate, separator, and graphite. Accordingly, a 10% increase in graphite production leads to less than a 10% decrease in environmental impact.



A



B

Fig. 3. (A) Environmental profile of the graphite process line and (B) Contribution of LCI items over the total impact.

Parameters related to the amount of material treated also influence environmental impacts, though to a lower extent. Increasing the number of cells, their weight, or anode content by 10% results in a 2–4% variation in GWP, as higher inputs drive greater energy and material consumption. Energy use for cell treatment is among the most influential factors, being the largest contributor to GWP, whereas the coal tar pitch consumption and other parameters have a smaller effect. A similar picture is observed for the other environmental impacts investigated (Figures 7SM–15SM), though the effect of coal tar pitch consumption is relatively stronger for the PMFP, ODP, EP, and MIR impacts. Also, the amount of cathode has a considerable effect on the toxicity impacts because of the high steel consumption, relative to the produced cathode concentrate, required for the screening screens.

3.2.2. Monte Carlo simulation

To assess the variability of the calculated environmental impacts, a Monte Carlo simulation is performed for the parameters shown in Table 2SM based on the results of the sensitivity analysis performed in the previous paragraph. In the absence of more detailed data, a triangular distribution is assigned to the variables. The graphite produced, which has the greatest influence on the results and is directly linked to cell weight and anode content, is not assigned a separate variability range. In total 30,000 simulations were conducted to reach convergence. The mean impact calculated is slightly lower than that reported in the baseline analysis for all impact categories. The reason is mainly that the triangular distributions assigned for the cell weight is left-skewed. As a result, more simulation runs pick relatively lower cell weight that leads to less amount of graphite produced compared to the baseline consideration for the same energy consumption. The GWP impact caused due to graphite production remains low ranging between 3.35–5.07 kg CO₂-eq/kg graphite produced. Figure 16SM shows the Monte Carlo results for the GWP caused by graphite production, while similar figures for the other impact categories can be found in Figures 17SM–24SM in the SI. Table 10SM summarizes the MC results for all impact categories.

3.2.3. Effect of allocation method

To allocate the environmental impacts of the process between the three co-products (battery-grade marketable graphite, and not-ready precursors of cathode and separator) the sub-processes were subdivided as far as possible, except of the first four sub-processes (receiving to cell disassembly, Fig. 1A), the impacts of which were allocated via mass allocation. Notably, these processes cause <10% of the total impacts for all impact categories, except of the ECFW and HTNC categories. This indicates that the allocation of the impact has relatively low effect on the final result. Following the recommendations of the ISO documents, an analysis considering economic allocation was performed as well. For this set the price of battery grade graphite was set to 10,000 EUR/tonne (as in the TEA) and for the cathode concentrate precursor a value of 5700 EUR/tonne was assigned considering prices of black mass, while the price of the separator is set to 0. It is shown that the results do not alter considerably with the graphite impacts increasing slightly and the cathode impacts reducing accordingly due to the higher graphite price than the black mass. The largest changes concern the ECFW and HTNC impacts that are those affected the most by the impacts of the first four sub-processes the impacts of which are allocated. The detailed results are shown in Table SM11.

4. Discussion

4.1. Graphite availability and importance of battery recycling

Natural graphite is considered a critical mineral as it has wide applications in several sectors such as energy, electronics and metal manufacturing. The global demand for graphite is expected to rise to 10,419 ktons by 2030, with China producing 82% of all graphite grades and 93% of battery-grade graphite (IEA, 2024). In this context,

technologies aiming on graphite recovery are highly relevant since they can reduce the dependency of Europe and the US on external graphite suppliers, contributing to the material independency and autonomy. While most current battery recycling technologies focus on recovering nickel and cobalt, which constitute only a small fraction of the battery materials (Shang et al., 2024), the direct recycling technology investigated in this study targets the recovery of battery-grade graphite from spent cells, along with the concurrent retrieval of cathode concentrate and other scrap metals. This method is designed for NCA batteries, with modifications under investigation to accommodate other Li-ion chemistries.

Currently, NCA batteries account for roughly 8 percent of Li-ion battery capacity, a share expected to rise to 13% of batteries available for recycling in North America by 2030 (Gonzales-Calienes et al., 2023). NCA chemistry is used in electronic devices and in prismatic and cylindrical cells, with major producers including Panasonic, Sony, and Samsung. In the electric vehicle market, companies like Tesla also employ NCA-based batteries (Sanders, 2017). The growing adoption of batteries increases the volume of end-of-life waste that must be managed. In Europe, for instance, 420 ktons of batteries are projected to be available for recycling by 2030, rising to 2100 ktons by 2040 (Schmaltz, 2023). In the United States, batteries are considered restrictive waste, making their disposal in municipal streams illegal.

In this context, developing recycling technologies that recover critical materials and reintegrate them into domestic supply chains is essential. The NCA recycling process investigated demonstrated efficient recovery and proven economically and environmentally viable. Further research is needed to expand the process to other battery chemistries and to explore post-processing techniques for recovering lithium, manganese, and cobalt from the produced cathode concentrate. These improvements could enhance economic performance, reduce environmental impact, and strengthen domestic supply of critical battery materials.

4.2. Cost and impact mitigation strategies and policy recommendations

The TEA and LCA performed highlight promising benefits but also potential areas of improvement for the techno-economic and environmental efficiency of the plant. The electricity consumption drove the environmental impacts of the plant, which improved significantly when RES-dominated mixes were considered. Therefore, such plant in the EU can be developed in regions with high RES-share in the electricity mix (e.g., Austria, Denmark (IEA, 2025)). Also, the EU and US can endorse the use of utility PPAs for such plants, building on existing practices (U.S. EPA, 2025), to supply them with RES-generated electricity regardless of the region, even at a higher price since the electricity cost is a relatively low fraction of the total cost. In terms of process design, focus can be put to reduce the electricity consumption for the purification of recycled graphite particles, by for example exploring different kiln configurations and also explore different furnace types for reducing the energy intensity (heat treatment stages) of the cathode and separator lines.

This study showed that the costs are dominated by the construction costs, which can lead to a reluctance from private investors due to high upfront costs and associated financial risks. Subsidization, and low or zero-interest loans for the construction of such plants can help derisk investment and make investment more appealing, as already done in the US (U.S. DOE, 2025) and the EU (Skyrman, 2024) for energy investments. Including investment in direct recycling technologies and their R&D in such schemes can increase adoption and accelerate investment. Another way to support the development of such plants is to ensure their profits. Demand for recycled battery-grade graphite is expected to increase with the EU battery regulation (E.C., 2025) endorsing the use of battery-grade graphite originating from secondary sources, as well as the demand for recycled Li, Co, Ni that can be subsequently derived from the product of the cathode line. The amount of spent

batteries and the graphite demand is increasing both in the US and the EU. Ensuring the selling price of the produced graphite as well as the low-cost supply of spent batteries can help ensuring the profits of such plants and thus widen investment and development. Moreover, the direct recycling technology produces graphite with a low GWP compared to natural graphite production. Considering a carbon tax, (e.g., 0.1–134 EUR/tonne CO₂-eq, gathered from Tax Foundation Europe) based on the lifecycle GHG emissions of the plant lead to an additional cost increasing the MSP in the baseline scenario between 0–0.5%. However, a potential expansion of the EU ETS to include recycling plants and allocate benefits compared to its alternative could generate revenues for the plant reducing more the MSP, as compared to natural graphite production (battery-grade) the current technology introduces carbon savings.

Further cost reductions can be achieved through adopting longer-lasting cutting tools (e.g., diamond-coated blades) and closed-loop water systems to minimize deionized water consumption in wet classification. Additionally, solvent reuse in the granulation and drying step already shows cost-reduction potential over time and should be emphasized. Extending operations from 10 to 15 years (a highly likely case) improved NPV to 88.3 million € and lowered the minimum selling price (MSP) by 20%.

4.3. Comparison with other studies on graphite production

In this section, the GWP achieved for producing 1 kg of graphite with the proposed direct recycling technology (4.06 kg CO₂-eq/kg) is compared to the reported GWP for graphite production in the literature to evaluate the environmental desirability of the proposed technology. Abdelbaky et al., reported a more than three times higher GWP for natural graphite production (13 kg CO₂-eq/kg) (Abdelbaky et al., 2023). Other authors have reported lower GWP values for the production of natural graphite for battery-grade applications. For example, Engels et al. (2022) found a GWP of 9.6 kg CO₂-eq/kg, while Gao et al. (2018) reported a value of 5.3 kg CO₂-eq/kg and Zhang et al. (2018) of 7.8 kg CO₂-eq/kg. Nevertheless, all these reported values remain above the one found for the direct recycling process analysed in this article, thus showing the potential of enhancing recycling over natural graphite use. Additionally, the GWP results found in this study fall in the lower range reported by Rey et al. (2021) for graphite recovery from spent LiBs using combinations of hydrometallurgy and pyrometallurgy processes (0.53–9.76 kg CO₂-eq/kg of recycled graphite). The authors reported that energy consumption drove the environmental impacts. Notably, the latter study used inventories derived from the literature for lab-scale analysis and scaled the inventory for pilot scale operation.

4.3.1. Setting the functional unit to 1 kg of batteries treated

The investigated technology targets the graphite contained in the spent battery cells. Therefore, the functional unit considered is 1 kg of battery grade graphite retrieved. The results were recalculated considering 1 kg of treated cell as functional unit. The functional unit and credits considerations in LCA studies of battery recycling vary widely in the literature hindering comparisons (Domingues and Souza, 2024). For example, in the literature review performed in the introduction studies were found reporting GWP values of 1.5–4.2 kg CO₂ eq./ton of spent batteries for hydrometallurgical recycling (Wang et al., 2024), –12.13 kg CO₂ eq./kg of recycled batteries for direct recycling (Yang et al., 2024), or –1.2 to 1.4 kg CO₂ eq./kg of recycled batteries (Rajaeifar et al., 2021), while other studies also use different functional units such as 1 kWh of battery produced (Paul et al., 2024). To facilitate future usage of the results of this study, the GWP impact was recalculated using 1 kg of treated battery cell as a functional unit, resulting to a value of 0.7565 kg CO₂-eq/kg treated. However, this value does not consider any benefit from the products produced (even from the produced graphite). Considering such credits would result to a highly negative impact per kg of battery cell treated, depending also on the products that are assumed

to be substituted. Notably, and as shown in Fig. 1A, a total of 4.43 MJ of electricity and 2.32 of MJ of natural gas are consumed (directly) to treat 1 kg of battery cell.

5. Conclusions

This study presents a comprehensive environmental and economic evaluation of a novel direct recycling process for recovering battery-grade graphite from spent NCA lithium-ion batteries, using detailed primary data from a continuous pilot-scale plant. The integrated assessment framework, combining life cycle and techno-economic analysis with sensitivity testing, scenarios, and Monte Carlo simulations, provides strong evidence of the process's feasibility. The technology delivers clear environmental benefits, achieving a global warming impact of 4.06 kg CO₂-eq per kg of graphite, significantly lower than values reported for natural and synthetic graphite production. Electricity use is the dominant impact driver, followed by coal tar pitch consumption. Purification has the highest climate impact, while granulation and drying contribute notably to particulate matter formation. Replacing grid electricity with renewables, particularly wind, substantially reduces impacts, and further improvements can be achieved by sourcing recycled pitch and steel. Economic results show viability, with a production cost of 15.18 €/kg and a minimum selling price of 9.87 €/kg when co-product revenues are included. Capital expenditure represents the largest cost share, while throughput and plant lifetime strongly influence profitability. Extending plant operation from 10 to 15 years increases the Net Present Value from 4.2 million € to 88.3 million € and lowers the minimum selling price by about 20%. Overall, direct recycling emerges as a technically feasible, environmentally favorable, and economically competitive route for graphite recovery. The process can strengthen circular battery value chains and reduce dependence on virgin critical materials. Policy support enabling renewable electricity use, stable markets for recycled graphite, and design-for-disassembly standards will be essential to accelerate deployment and unlock the full potential of this technology.

CRediT authorship contribution statement

Spiros Gkousis: Writing – original draft, Investigation, Formal analysis, Data curation, Conceptualization. **Ana Arias:** Writing – review & editing, Writing – original draft, Formal analysis, Conceptualization. **Anna Doninger:** Writing – review & editing, Resources, Data curation. **Zoe Umlauf:** Writing – review & editing, Resources, Data curation. **Emily Schmidt:** Writing – review & editing, Resources, Data curation. **Igor V. Barsukov:** Writing – review & editing, Resources, Data curation. **Maria Teresa Moreira:** Writing – review & editing, Supervision. **Evina Katsou:** Writing – review & editing, Supervision.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Anna Doninger, Zoe Umlauf, Emily Schmidt, Igor V. Barsukov reports a relationship with American Energy Technologies Co., 265 Alice St., Wheeling, IL 60,090, USA that includes: employment. If there are other authors, they 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 work by Imperial College of London has been funded by STREAMS (Sustainable Technologies for Reducing Europe's battery raw Materials dependence) project, funded by the European Union's Horizon Europe research and innovation programme under the Grant Agreement No 101137771. The work by American Energy Technologies Co. was

funded by the aforementioned company as part of its Consortium membership within STREAMS. Views and opinions expressed are however those of the author(s) only and do not necessarily reflect those of the European Union or the European Climate, Infrastructure and Environment Executive Agency (CINEA). Neither the European Union nor the granting authority can be held responsible for them. A. Arias also thanks the Galician Government for financial support (Grant reference ED481B-2023-072). A. Arias and M.T. Moreira, authors belong to the Galician Competitive Research Group (GRC ED431C 2025/19) and to the Cross-disciplinary Research in Environmental Technologies (CRETUS Research Center, ED431G 2023/12). I.V.B. submits this work in eternal memoriam of Dr. Nina Georgievna Barsukova and Mr. Leonid M. Vepriński, who have done much to advance the science and technology of industrial carbon and graphite recycling via energy-efficient continuous thermal methods. Dr. Barsukova rose against the odds to become a true mentor, inspirer, and role model for dozens of scientists and engineers advancing the carbon and graphite industry. Authors acknowledge notable contributions of the following employees of American Energy Technologies Company, into the development of process data for LCA inventories, used to develop this paper; Mansoor Qureshi, Anzori Kuchaidze, Matthew Meyers, Michael Durbin, Jayden Chow and Savita Thompson.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at [doi:10.1016/j.resconrec.2026.108993](https://doi.org/10.1016/j.resconrec.2026.108993).

Data availability

Data will be made available on request.

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