Clean Technologies and Environmental Policy (2026) 28:149
https://doi.org/10.1007/s10098-026-03498-5

ORIGINAL PAPER

Environmental and economic assessment of bio-based anode production

Bening Mayanti[1] · Theodore Azemtsop Manfo[2] · Hafiz Haq[1] · Nebiyu Girgibo[3] · Jonas Markusson[4]

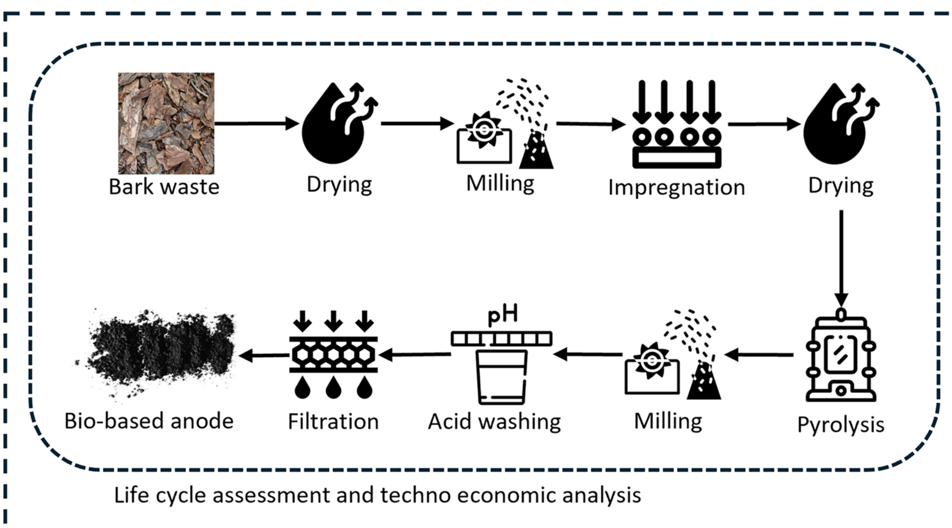
Received: 19 November 2025 / Accepted: 23 April 2026

Abstract

This study evaluates the environmental and economic impacts of bio-based anode material and its implications for policy researchers and businesses. This study is based on upscaling laboratory work to develop and electrochemically test bio-based alternatives for anode materials.

The study uses biochar synthesised from bark waste via pyrolysis. Electrochemical testing is conducted using biochar as the anode in lithium-ion batteries (LIB) and sodium-ion batteries (NIB). Technical economic analysis (TEA) and life cycle assessment (LCA) are employed to assess the economic and environmental viability of producing the materials. The findings suggest a trade-off between economic and environmental aspects compared with a conventional anode material. Both aspects are sensitive to the chemicals used during impregnation and acid washing, $ZnCl_2$ and HCl. It shows the importance of securing the chemical supply chain and experimenting with alternative chemicals to reduce impacts without compromising quality. This work also provides a foundation in green energy storage systems and in evidence-based decision-making for related stakeholders.

Graphical Abstract



Keywords Life cycle assessment · Technical economic analysis · Bio-based battery · Anode · Bark waste

Extended author information available on the last page of the article

Introduction

Climate change is a global threat to humanity, causing food insecurity, unpredictable weather patterns, displacement, and worsening inequality (International Energy Agency 2021; NASA 2024). Decarbonisation becomes a priority, with batteries and energy storage technologies playing a key role in advancing renewable energy and transport electrification to achieve net-zero (Abdelmotteleb 2022; World Economic Forum 2022). The importance of energy storage is shown by the increase in demand for graphite, the primary anode material, which has tripled between 2015 and 2020 and is predicted to grow by 6.2% annually between 2022 and 2026 (ECGA 2022; Surovtseva et al. 2022). Graphite used in batteries is either natural graphite (NG) or synthetic graphite (SG). It remains the leading anode material known for its stable layered structure, capacity of 372 mAh g^{-1}, and excellent cycling stability of over 1000 cycles Adebanjo et al. 2025). NG is mostly mined in China, Mozambique, and Brazil (Alaraudanjoki 2016). Its manufacture consists of three basic steps: (1) mining and flotation, where graphite is extracted using coal-type or shaft methods and separated from impurities through crushing, milling, and flotation (Chemanalyst 2025; Choi et al. 2020; U.S. Geological Survey 2021); (2) material processing, which includes purification to 95%–99% purity (Dolega et al. 2020; Damm 2020); and (3) refinement, where particles are shaped via spheroidization (Dolega et al. 2020; Chehreh 2016) and carbon-coated to improve performance and electrolyte resistance (Lämmerer and Flachberger, 2017). In contrast, SG is synthesised from carbon precursors such as petroleum coke or coal tar pitch. The process begins with calcination at 800–1200 °C to produce soft carbon, followed by crushing and classification (Fig. 1). The process involves graphitization at temperatures above 2500 °C, followed by grinding, classification, and coating (Chehreh et al. 2016; Dühnen et al. 2020). Differences in origin and production methods between NG and SG not only affect quality and cost, but also have significant environmental consequences, as investigated in life cycle assessment (LCA) studies cited in the following section (Dühnen et al. 2020).

Even though graphite is not considered rare, its trade is controlled by China, which dominates 95% of the international supply of battery-grade graphite (Golden 2025). Finding alternatives for virgin graphite becomes important not only to secure the supply chain but also to make batteries greener. Manufacturing graphite from carbon-rich biomass waste has gained popularity as a sustainable alternative. Agricultural bio-waste is an abundant, renewable, and low-cost source of carbon (Gangopadhyay 2021; Haluska et al. 2023; Xia et al. 2022; Sun et al. 2022; Wang et al. 2022a, b, c, d; Tan et al. 2022; Gong et al. 2017). These high-carbon biomasses make excellent feedstocks for graphite synthesis, providing environmental and economic benefits (Hoekstra et al. 2015; McHenry 2009). Moreover, biochar, another carbon-rich material derived from biomass pyrolysis or gasification, has multiple industrial and environmental applications (Jiang et al. 2013). Pyrolysis techniques efficiently convert biomass into high-energy carbon compounds (Tripathi et al. 2016; Wang et al. 2018a, b), and cost–benefit analyses support the economic viability of biowaste recycling (Sun et al. 2018). In addition to addressing waste management challenges, biomass-derived carbons have shown broad applicability in batteries (Feng et al. 2020), supercapacitors

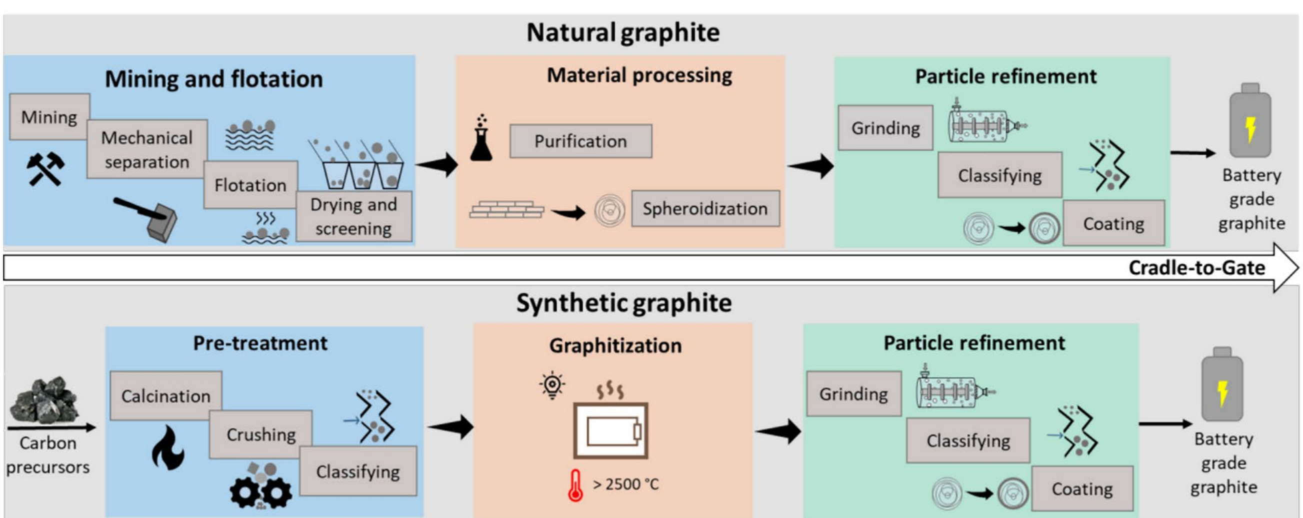


Fig. 1 Mining and production routes for natural and manufactured battery-grade graphite

(Naik et al. 2022), water purification systems (Valdés-Rodríguez et al. 2022; Wang et al. 2022a, b, c, d), and catalytic supports (Zou et al. 2022), all of which contribute to global sustainability and decarbonization objectives.

Agricultural bio-waste is an excellent source of carbonaceous materials for the production of high-performance carbon products, including activated carbon, mesoporous carbon, CNTs, fullerenes, and graphene (Patnaik et al. 2016). These wastes are largely lignocellulosic, consisting of cellulose, hemicellulose, and lignin, all of which are high in carbon. Lignin, with a carbon content of around 60%, is an ideal precursor (Xu and Li, 2017; Chen et al. 2018; Souto et al. 2018). Materials such as rice husk and coconut coir have shown great promise. Rice husks, which contain 38% cellulose, 18% hemicellulose, and 22% lignin, have been utilised efficiently to extract carbon (Rybarczyk et al. 2016). Coconut coir has permitted the low-cost synthesis of graphitic carbon with well-organised nanostructures (Destyorini et al. 2021). A variety of agricultural leftovers, including nut shells, bagasse, and crop stalks, have also been investigated as carbon sources (Abdeljaoued et al. 2018; Bae and Su 2013; Ameen et al. 2019; Mahmood et al. 2014; Xia et al. 2020; Wu et al. 2019; Janković et al. 2019; Kan et al. 2018). Table 1 summarises the proximate and final analyses of typical lignocellulosic biowaste, highlighting the potential of lignocellulosic biomass as an effective carbon precursor even before carbonisation.

Despite the potential of biomass as an alternative anode material, there is still limited research on its economic viability and sustainability. Most research on bio graphite has primarily concentrated on laboratory investigations of its morphological and electrochemical properties, utilising various biomass feedstocks and catalytic methods for production. Previous works applied pyrolysis to manufacture porous carbon (buckwheat hulls), activated carbon (cotton stalks), spherical mesoporous carbon (waste green tea), and graphitic carbon (coconut coir) as anode alternatives that showed capacity and cycles of 715 mAh/g (150 cycles), 217.7 mAh/g (100 cycles), 498 mAh/g (100 cycles), and 208.2 mAh/g (30 cycles), respectively (Destyorini et al. 2022; Sankar et al. 2019; Wang et al. 2022a, b, c, d; Yu et al. 2021). Other studies used carbonization to produce porous carbon (banana peel), porous hard carbon (mango peel), activated carbon (avocado seeds) with capacity and cycles of 272 mAh/g (200 cycles), 628 mAh/g (200 cycles), and 400 mAh/g (100 cycles), respectively (Luna-Lama et al. 2021; Muruganantham et al. 2021; Yokokura et al., 2020).

To address the gaps, the study implements life cycle assessment (LCA) and a technical–economic analysis (TEA) to study biochar production. LCA is a standardised tool under ISO 14040/14044 that can quantify the environmental impacts of products or activities, including identifying impact hotspots. Combining LCA with TEA enabled us to achieve the goal of this work: to identify the economic and environmental feasibility of biochar as a bio-based alternative to the conventional anode. The assessment is based on laboratory work conducted by our research partners, who developed a bio-based anode using spruce bark (Simões Dos Reis et al. 2022). The developed material was then electrochemically tested as an anode in sodium-ion batteries (NIB) and lithium-ion batteries (LIB). This assessment would also highlight potential trade-offs between economic and environmental aspects. This paper hence addresses the following two research questions: RQ1–What are the environmental

Table 1 Proximate and ultimate analysis of various agricultural bio-waste carbonaceous precursors

Agricultural waste	Proximate analysis (% *w/w*)			Ultimate analysis (% w/w)					References
	Moisture content	Ash content	Volatile content	C	H	N	O	S	
Coconut shell	12.55	0.42	NA	51.60	5.60	0.10	42.70	0.00	Abdeljaoued et al. (2018)
Macadamia nutshell	10.00	0.40	71.0	57.50	5.95	0.33	36.20	0.06	Bae et al., 2013
Coconut coir	9.50	9.30	69.80	42.00	4.85	0.42	40.50	0.13	Ameen et al. (2019)
Kenaf	8.35	16.32	64.20	39.20	5.12	0.35	45.60	0.22	Ameen et al. (2019)
Rice husk	9.40	13.20	62.00	37.80	4.73	0.45	43.50	0.17	Ameen et al. (2019)
Oil palm fronds	6.00	1.00	76.00	42.88	7.06	0.52	49.54	NA	Mahmood et al. (2014)
Palm kernel shell	8.44	8.79	82.58	50.42	9.74	0.52	39.32	NA	Mahmood et al. (2014)
Chinese chestnut shell	NA	0.00	73.23	46.68	5.94	0.62	46.76	NA	Xia et al (2020)
Bamboo	NA	1.02	82.08	54.30	5.57	0.21	38.91	NA	Xia et al (2020)
Jatropha shell	NA	6.69	70.94	47.35	5.88	1.00	39.08	NA	Xia et al (2020)
Cotton stalk	NA	3.14	80.54	47.12	6.25	0.57	42.79	NA	Xia et al (2020)
Saw dust	NA	0.42	85.14	50.64	6.00	0.07	43.29	NA	Wu et al. (2019)
Raw straw	NA	9.98	74.39	50.01	5.54	0.81	43.64	NA	Wu et al. (2019)
Apricot kernel shell	9.71	0.94	73.84	46.88	6.38	0.25	45.45	0.00	Jankovic et al. (2019)
Macadamia nut shell	NA	2.51	77.68	49.15	5.51	0.59	42.12	0.12	Kan et al. (2018)

and economic implications of producing bio-based anode material? RQ2–What implications can be identified for policymakers, researchers, industry, and other stakeholders?

Methods

Life cycle assessment

The impacts were assessed using life cycle assessment (LCA) in accordance with ISO 14040 and ISO 14044 (ISO 2006a, b). This section proceeds through the LCA phases, namely: goal and scope definition, life cycle inventory, impact assessment, and interpretation.

Goal and scope definition

The study assessed the environmental impact of producing bio-based anode material (biochar or bio-carbon) from cradle to gate using attributional LCA. The data and process used were based on laboratory work on bio-based anode development by our project partners (Simões dos Reis et al. 2022). The laboratory results were then scaled up and translated into an industrial implementation to assess its costs and environmental implications, as reported in a previous study conducted by Piccinno et al. (2016). The functional unit (FU) adopted was 1 kWh of energy delivered. The test was done on lithium-ion batteries (LIB) and sodium-ion batteries (NIB) (Simões dos Reis et al. 2022). The electrochemical performance for NIB was 126 Ah/kg, 440 cycles, and an average 0.3 V; meanwhile, the LIB resulted in 319 Ah/kg, 5000 cycles, and 0.5 V. The LCA calculation uses the same lab-scale values, since batteries produced on an industrial scale can have lifetimes longer or shorter than those in laboratory tests, depending on the application (Golden 2024; Kim et al. 2025). The cradle-to-gate process of manufacturing bio-based anode was partitioned into eight sub-processes—drying, milling, impregnation, drying, pyrolysis, milling, acid washing, and filtration—as shown by Fig. 2. The first dryer was assumed to be a belt dryer, and the second, a spray dryer. The raw input biomass was waste bark, so the zero-burden principle was applied, assuming the raw material had no prior environmental impacts that would incur an upstream burden. The study's geographical scope was Sweden, and its temporal scope was 2024.

To manufacture 1 tonne of bio-based anode with 94% carbon content, 6 tonnes of bark waste (50% moisture content) were needed. In addition to energy, chemical inputs were needed—zinc chloride ($ZnCl_2$), nitrogen, hydrochloric acid (HCl), and distilled water. The bark waste was impregnated with $ZnCl_2$ by combination at a 1:1 ratio, achieving 25% ZnCl2 absorption, with the remaining 75% entering the wastewater stream. For acid washing, 6 M HCl was required, and wastewater accounted for 90% of the influent (10% loss). The study followed an attributional approach and employed system expansion to handle co-products. The electricity generated from syngas was assumed to replace the average regional electricity production in Sweden.

Life cycle inventory

The inventories used in the foreground system were based on the lab work done by the project partners (Simões Dos Reis et al. 2022). Previous studies concerning bio-carbon manufacturing were also used to build the industrial-level model (Liu et al. 2024; Piccinno et al. 2016; Wang et al. 2018a, b). Background data were drawn from the Ecoinvent 3.11 Cut-off database (Ecoinvent 2024), and the model was executed using OpenLCA 2.6. Raw material transportation was assumed to be 30 km. The value for construction and equipment was a generic value for manufacturing 1 tonne of chemicals; it was split equally across the eight sub-processes for simplicity. Meanwhile, the quantity of direct emissions from drying was adopted from previous studies (Imalpal Group 2023; Lagleder & Obermeier-Hein 2015; Spets & Ahtila 2004). $ZnCl_2$ was not available in the Ecoinvent database; therefore, it was modelled stoichiometrically as the product of the reaction between hydrochloric acid and zinc oxide. Electricity and heat needed were estimated using generic chemical manufacturing processes from a previous study and Ecoinvent 3.11 (Ecoinvent 2024; Kim & Overcash 2003). Meanwhile, the syngas inventory for syngas burning was taken from previous research (Gu & Bergman 2017). Table 2 shows the overall inventories for manufacturing one tonne of bio-based anode from six tonnes of biomass.

Life cycle impact assessment

The impact assessment translates and simplifies the inventories into more meaningful impact categories. Eleven impacts were assessed, namely abiotic depletion (AD), abiotic depletion of fossil fuels (AD-FF), acidification (AC), eutrophication (EU), freshwater aquatic ecotoxicity (Ecotox-FW), climate change (CC), human toxicity (HT), marine aquatic ecotoxicity (Ecotox-MA), ozone layer depletion (ODP), photochemical oxidation (PO), and terrestrial ecotoxicity (Ecotox-tr). The CML-baseline method was used because the focus was on midpoint impacts rather than an aggregate damage score; it is widely used by academia and industry to ensure comparability and was developed for the European context. It also covers a range of impacts that suffice for baseline assessment without generating so many impact categories as to be unwieldy.

Normalisation and weighting were not applied because the aim of this study was not to compare unitless results across various impact categories, nor to create a single score.

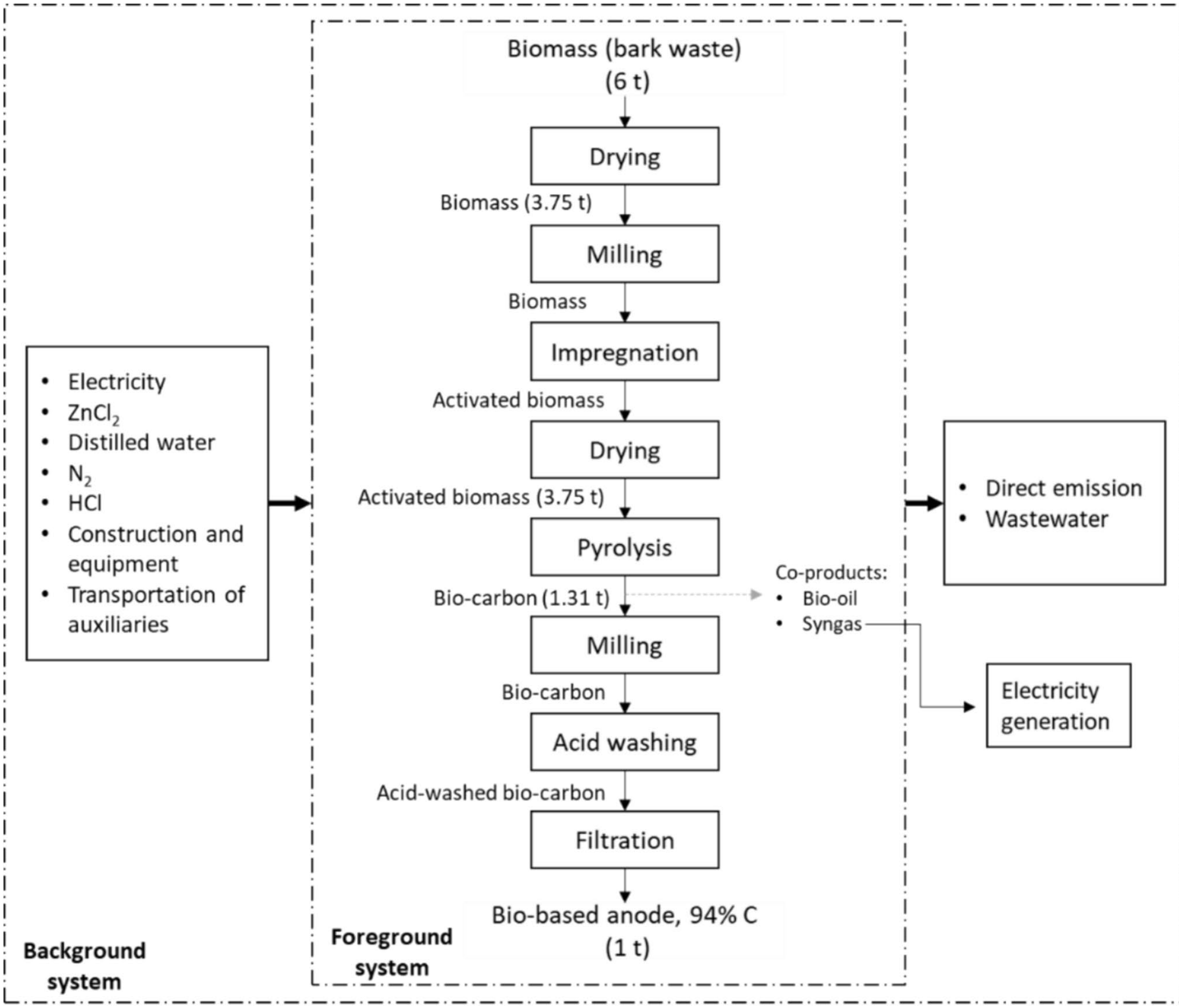


Fig. 2 System boundaries of bio-based anode manufacturing

Some analyses focused on climate change due to its effects on the global population and the urgent need for action. In addition to the overall results, a contribution analysis was performed. This analysis focused on the manufacturing process for a bio-based anode to identify which process caused the environmental hotspots. Contribution analysis was also applied using input and output inventories to highlight the most critical factors affecting GWP. Conducting contribution analysis allows stakeholders to identify environmental hotspots and make informed decisions.

Interpretation, scenario, and sensitivity analysis

Interpretation, the final phase of LCA, involves evaluating and analysing the results to draw meaningful conclusions. Scenario and sensitivity analysis were included to enhance the understanding of the system, with a view to guiding decision-making. Various assumptions regarding alternative options were applied to understand how the model behaved by observing the change in the outcomes under different circumstances: i) electricity in system expansion was assumed to be generated from oil, ii) changing the production method of HCl to chlor-alkali electrolysis (for HCl used in the acid washing and as a chemical input for producing $ZnCl_2$), iii) full system expansion covering syngas and bio-oil, assuming the bio-oil replacing heavy fuel oil production, and iv) chemicals recovery for $ZnCl_2$ and HCl. Chlor-alkali electrolysis was applied in the scenario analysis because Sweden has invested in green hydrogen production, which links to HCl production (Thyssenkrupp Nucera 2023). Meanwhile, chemical recovery through evaporation and distillation could reduce the consumption of virgin $ZnCl_2$ and HCl. The

Table 2 Inventories for manufacturing 1 tonne bio-based anode from 6 tonne of biomass

Item	Drying	Milling	Impregnation	Drying	Pyrolysis	Milling	Acid washing	Filtration	Unit
Inputs									
Electricity	2019	750	34	1862	1200	750	7	34	kWh
C&E	5E-08	5E-08	5E-08	5E-08	5E-08	5E-08	5E-08	5E-08	item(s)
$ZnCl_2$			3						t
Distilled water			2250					15,844.5	l
Nitrogen					72				l
HCl (6 M)							15,655.5		l
Water, cooling (resource)					65.5				m^3
Co-products									
Bio-oil					1.13				
Syngas					1.31				
Waste									
Wastewater			1012				14,090	14,260	l
Direct emissions									
VOCs	810			90					
CO_2 (non-fossil)					3530.5				kg
CO (non-fossil)					248.9				kg
Ethane					9.2				kg
Ethylene					3.1				kg
CH_4 (non-fossil)					52.8				kg
PM < 2.5 µm	200.8				0.590				kg
PM > 10 µm					0.072				kg
PM 2.5–10 µm					0.066				kg
Water, emission to air	2.25			1.01	25.4				m^3
Water, emission to water					40.1				m^3

C&E: construction and equipment, VOCs: volatile organic compounds, CO_2: carbon dioxide, CO: carbon monoxide, CH_4: methane, PM: particulate matter

chemical recovery rate was assumed to be 90% for both of the chemicals. The inputs required per m^3 of wastewater treated were 1000 MJ of heat, 10 kWh of electricity, and 30 m^3 cooling water. The inventory was estimated using a proxy of chemical process in Ecoinvent database, other studies, and latent heat of vaporisation (Ecoinvent 2024; IMPEL 2012; Jiangsu Co Ltd 2024; Ramanathan et al. 2025).

A sensitivity analysis was also conducted to examine how outcomes differ with changes in parameter inputs. Each input was increased by 10% while holding the other parameters at baseline. Sensitivity ratios (SRs) were calculated for the parameter according to Eq. (1).

$$SR = \frac{\Delta\text{result}/\text{result}_{\text{baseline}}}{\Delta\text{parameter}/\text{parameter}_{\text{baseline}}} \tag{1}$$

where Δresult is the difference generated from the baseline scenario after modifying the parameter by Δparameter from the baseline scenario. The parameters varied for sensitivity analysis were electricity, distilled water, nitrogen, HCl, $ZnCl_2$, construction, and equipment.

Methodology for techno-economic analysis

The industrial upscaling of bio-carbon synthesis was investigated through a laboratory experiment (Simões dos Reis et al. 2022), using Norway spruce bark as biomass for pyrolysis. The process included biomass drying, milling, impregnation, drying again, pyrolysis, and filtration. Industrial upscaling required the addition of a gas-cleaning module before converting the syngas into electricity. The bio-carbon synthesis method was set up to produce 1 tonne of bio-carbon initially, anticipating an input biomass of 6 tonnes with 50% moisture content. The dryer reduces the moisture to 20%. The milling reduces the size of biomass to dimensions where it can be impregnated with a catalyst at a ratio of 1:1, along with water at a water-to-biomass ratio of 0.75:1. After drying, the mixture is pyrolysed by heating to 800 °C at a heating rate of 10 °C/min in an anoxic atmosphere with nitrogen fed at 200 mL/min. The pyrolysis is expected to produce 35% bio-carbon, 35% syngas, and 30% bio-oil. The bio-oil is ignored; meanwhile, the syngas is cleaned and burned to generate electricity at 20% efficiency. It reflects a realistic scenario in which Sweden has a strong grid connection, so selling electricity generated from syngas can be done seamlessly with the existing infrastructure, whereas selling raw bio-oil would require a new supply chain setup. The bio-carbon is milled and washed with hydrochloric acid (HCl) to produce bio-graphite. The equipment sizing was conducted based on work by Liu et al. (2024), Piccinno et al. (2016), and Wang et al. (2018a, b). The power consumption of each piece of equipment was calculated.

Data collection

The material and energy costs come from various sources. The prices for nitrogen and hydrochloric acid are from a market seller (Chemanalyst 2025), as is the price for zinc chloride (Synthetika 2025). The prices of biomass and water are assumed based on prior research (Alaraudanjoki 2016). The price of electricity is set at 0.1 €/kWh (Eurostat 2024). The energy consumption for gas cleaning is assumed to be accounted for in the electricity generated. The quantities needed and their unit costs are summarised in Table 3.

The initial investment cost in equipment and facilities is assumed to be €15 million, based on the assumptions for individual equipment by Haeldermans et al. (2020). The estimated initial capital investment comprises both direct and indirect costs associated with plant construction and commissioning. Direct costs include purchased equipment, installation, piping, instrumentation, electrical systems, and buildings, whereas indirect costs include engineering and design, site preparation, permitting, contractor fees, and contingency allowances to account for project development uncertainties. In techno-economic assessments of emerging chemical and energy technologies, these additional capital components are commonly incorporated to provide realistic estimates of total plant investment and economic feasibility. According to recent techno-economic research, installation costs, engineering design, and contingency considerations can account for a significant fraction of total capital expenditure depending on project size and technical maturity (Wahid 2024; Badgett et al. 2024; Gomes et al. 2025). The cost of labour and the interest rate are assumed to be 8% (Thewys & Kuppens 2008). The maintenance cost is assumed to be 6% of the capital investment (Haeldermans et al. 2020). The property tax and contingency costs are assumed to be 2.5% and 5% of the investment's capital cost, respectively (Haeldermans et al. 2020). General expenses are assumed at 2% of the capital investment, in line with standard practice. These costs are summarised in Table 4, where general expenses have been increased to account for unexpected overheads.

Table 3 Materials and energy are used to produce 1 ton of bio-carbon

Item	Quantity		Unit cost (€)
Materials			
Biomass	6	t	360
Distilled water	2250	L	4.18
$ZnCl_2$	3	t	3000
N_2	72	L	16.70
HCl	15,656	L	2630
Distilled water for washing	15,845	L	29.47
Energy			
Drying	2019	kWh	201.9
Milling	750	kWh	75
Impregnation	34	kWh	0.34
Drying	1862	kWh	186.2
Pyrolysis	1200	kWh	120
Milling	750	kWh	75
Acid washing	7	kWh	0.7
Filtration	34	kWh	3.4
Electricity generated	1048	MWh	0.05

Economic assessment

The economic assessment of bio-carbon production is based on the net present value of the production facility. The value of the production facility can be estimated as (Short et al. 1995; Thewys & Kuppens 2008):

$$\text{net present value} = \sum_{a=1}^{b} \text{cash flow}_i (1+r)^{-y} - \text{capital investment} \tag{2}$$

where.

- net present value is the value of the production facility over several years.
- cash flow_i is the income in year I after expenses.
- r is the real discount rate on the capital investment.
- capital investment is the initial capital invested in the project.
- y is the number of years.

Table 4 Operating costs of the bio-carbon production

Item	€ million
Labor cost	1.20
Interest rate	1.2
Maintenance	0.9
Property tax & insurance	0.375
Contingency	0.75
Revenue from electricity generated	(0.0524)
General expenses	0.3

The cash flow is the income minus expenses, including taxes and depreciation, formulated as (Short et al. 1995; Thewys & Kuppens 2008):

$$\text{cash flow} = (1 - \text{tax}) \times (\text{revenue} - \text{cost}) + \text{tax} \times \text{depreciation} \quad (3)$$

where.

- tax is the corporate tax rate.
- revenue is the monetary value achieved after selling the product.
- cost is the cost of operations, including energy, materials, and water.
- depreciation is the cost of depreciated assets.

Sensitivity analysis was performed by changing the prices of the inputs one at a time, followed by the calculation of the sensitivity ratio.

Results

Environmental assessment

This section assesses the environmental impacts of manufacturing bio-based anode material in Sweden, including scenario and sensitivity analysis. The results covered 11 impact categories following the CML baseline model, with further examination of global warming potential (GWP) given its importance as a driver of battery development.

Environmental impacts

The cradle-to-gate environmental impacts of 1 kWh of energy delivered are shown in Table 5. For completeness, the table also includes the results per 1-ton bio-based anode. The analysis indicated that the GWP per kWh energy delivered was 4.21E-01 kg CO_2 eq (NIB) and 8.77E-03 kg CO_2 eq (LIB). Both battery types showed the same patterns in hotspot analysis, with impregnation as the biggest contributor to GWP (46%), followed by acid washing (27%) and pyrolysis (22%).

Identical patterns were also observed for the remaining impact categories in both NIB and LIB. For both batteries, more than 70% of total impacts were generated from impregnation and acid washing, except in PO. It was found that 93% of the PO was due to pyrolysis. In all categories except PO, impregnation contributed 13% to 69%. Hotspot analysis thus identified impregnation and acid washing as the most significant contributors to almost all impact categories in both NIB and LIB (Fig. 3).

Given the importance of climate change impacts, another hotspot analysis was conducted to identify the primary contributors to GWP (Fig. 4). Zinc chloride production, hydrochloric acid production, and direct emissions (mainly occurring during pyrolysis) contributed to 94% of GWP. Electricity consumption, in contrast, accounted for only 3% of GWP due to Sweden's high share of non-fossil electricity. The significance of zinc chloride and hydrochloric acid production mirrors the earlier, inventory-based result, which was driven largely by high zinc chloride consumption. The GWPs due to zinc chloride production and hydrochloric acid were 46% and 27%, respectively. Meanwhile, overall impacts from wastewater treatment, nitrogen production, construction, and raw material transportation contributed about 1.7% of GWP.

The burden avoidance from system expansion was insignificant. For every unit of energy delivered from the batteries, the avoided GWP is 0.35 kg CO_2 eq (NIB) and 0.0073 kg CO_2 eq (LIB). The benefits obtained from the system expansion application depend on the system or process being replaced. The expanded system in this study was

Table 5 Environmental impacts of bio-based anode

Impact category	Bio-based anode (per tonne)	NIB (per kWh energy delivered)	LIB	Unit
Abiotic depletion (AD)	9.87E-02	5.93E-06	1.24E-07	kg Sb eq
Abiotic depletion, fossil fuels (AD-FF)	7.15E+04	4.30E+00	8.97E-02	MJ
Acidification (AC)	2.31E+01	1.39E-03	2.90E-05	kg SO_2 eq
Eutrophication (EU)	2.45E+01	1.47E-03	3.07E-05	kg PO_4 eq
Freshwater aquatic ecotoxicity (Ecotox-FW)	5.93E+03	3.57E-01	7.44E-03	kg 1,4-DB eq
Global warming potential (GWP)	6.99E+03	4.21E-01	8.77E-03	kg CO_2 eq
Human toxicity (HT)	1.07E+04	6.42E-01	1.34E-02	kg 1,4-DB eq
Marine aquatic ecotoxicity (Ecotox-MA)	9.20E+06	5.53E+02	1.15E+01	kg 1,4-DB eq
Ozone layer depletion (ODP)	6.73E-04	4.04E-08	8.43E-10	kg CFC-11 eq
Photochemical oxidation (PO)	1.70E+01	1.02E-03	2.14E-05	kg C_2H_4 eq
Terrestrial ecotoxicity (Ecotox-Tr)	2.83E+01	1.70E-03	3.54E-05	kg 1,4-DB eq

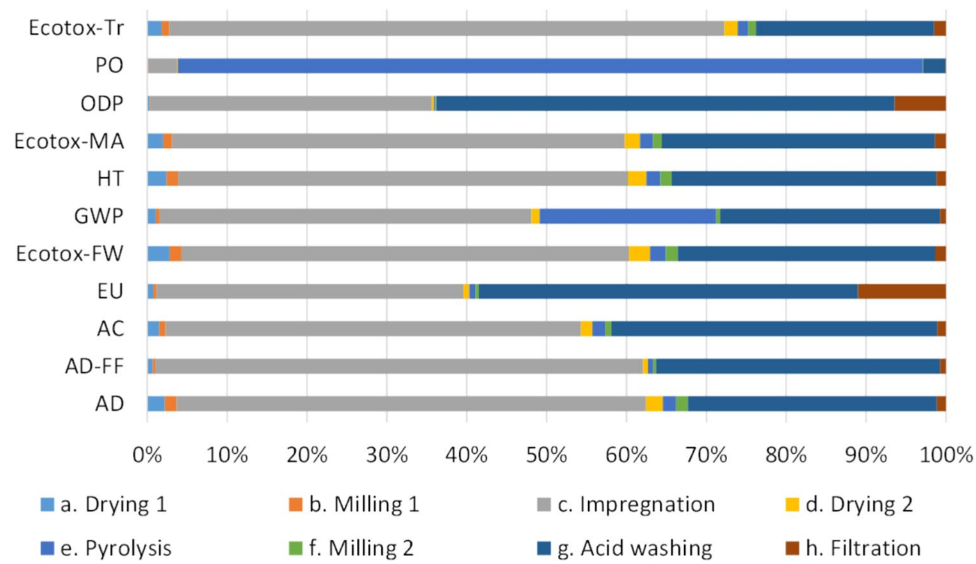


Fig. 3 Process contribution across impact categories for NIB and LIB

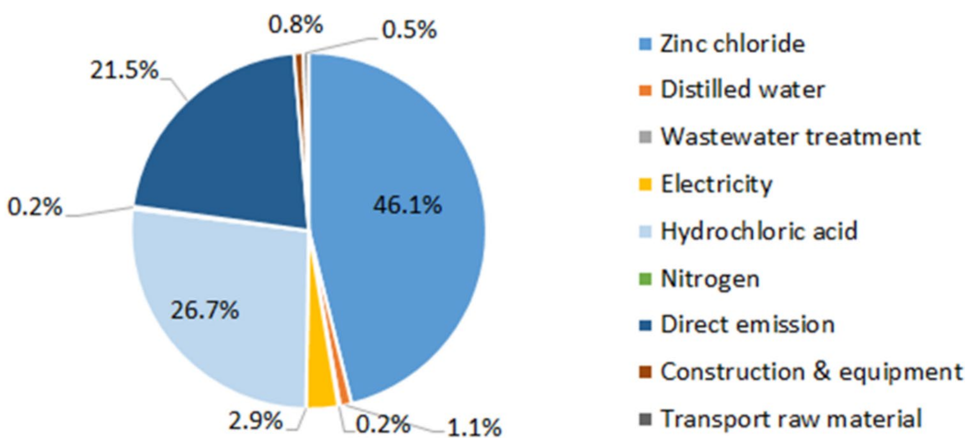


Fig. 4 Hotspot analysis of GWP in NIB and LIB

the Swedish electricity mix, which was dominated by green electricity sources; thus, the benefits from avoiding electricity production were low.

Scenario analysis

Several scenario variations were applied to evaluate how LCA outcomes may change with factors such as methodological choices, technologies, economic conditions, and social factors. The scenario employed in this study includes system expansion, replacing oil-based electricity generation, HCl production through chlor-alkali electrolysis, full system expansion covering all byproducts (syngas and bio-oil), and recovery of $ZnCl_2$ and HCl. Figure 5 presents the relative changes in NIB and LIB for each scenario compared to the baseline. For example, the graph of AD under the 1 M HCl scenario shows a value of 0.75; this means the impact is 75% of the baseline.

The greatest overall improvement was observed in the chemical recovery scenario (52% of baseline impacts), followed by the chlor-alkali electrolysis scenario (78% of baseline impacts). AD showed a significant reduction in the chemical recovery scenario (37% of the baseline), while ODP showed the greatest improvement in the HCl-chlor scenario (28% of the baseline). Meanwhile, the overall improvement observed in the other two scenarios ranged from 89 to 99% of the baseline impact. The least improvement was found in avoided burden in generating electricity from oil (99%), followed by full expansion (89%). A more detailed analysis of GWP was conducted, as shown in Fig. 6. The reduction patterns were similar between NIB and LIB, although the absolute values were dissimilar. Scenario analysis showed that the most significant change occurred in the chemical recovery scenario. When chemical recovery was implemented, the climate change hotspot shifted from impregnation to acid washing. The difference was caused by the high volume of

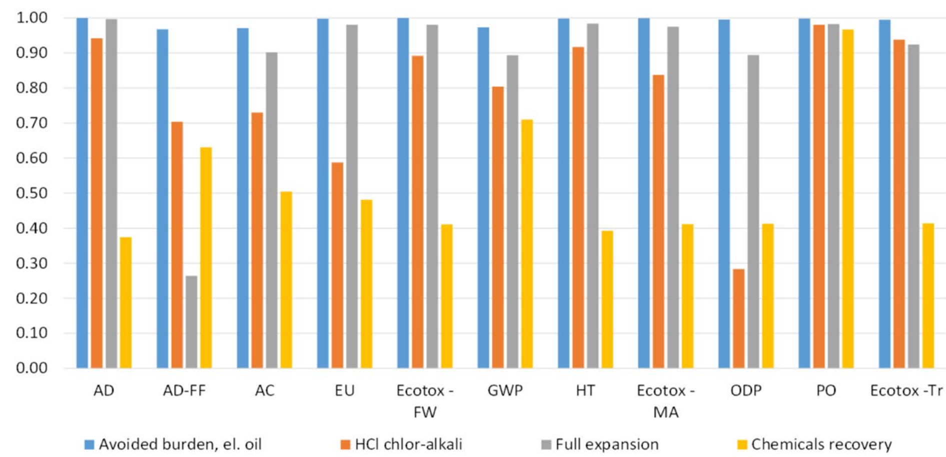


Fig. 5 Relative changes compared to baseline for NIB and LIB

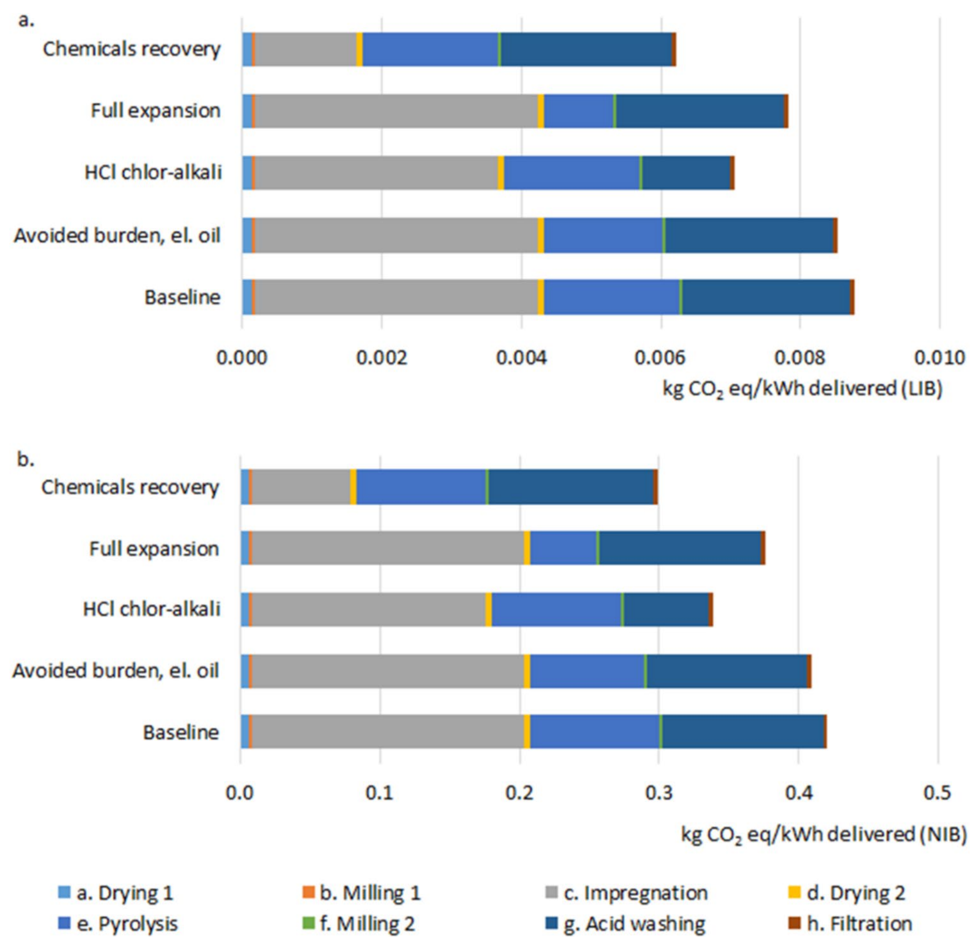


Fig. 6 Scenario analysis of the GWP: **a** LIB, **b** NIB

wastewater from the acid-washing process, which required significant energy for evaporation during chemical recovery. The chemical production pathway was also an essential aspect in impact reduction, as shown by reduced impacts when HCl was produced using different methods.

Sensitivity analysis

The input parameters were set to be 10% higher than the baseline one-at-a-time. The parameters were electricity, zinc chloride ($ZnCl_2$), distilled water, nitrogen, hydrochloric acid (HCl), construction, and energy delivered. The sensitivity ratios per impact category ranged from – 0.9 to 0.7, as shown in Fig. 7.

Energy delivered by NIB and LIB was the most sensitive parameter, with the same level of sensitivity (SR – 0.9). When NIB and LIB could increase their energy delivered by 10%, the overall impact would decrease by 9%. As for the $ZnCl_2$ and HCl inputs, they were also identified as influential candidates for optimisation across nearly all the impact categories. These outcomes were also supported by a scenario analysis showing the importance of chemical use in bio-based anode production. Meanwhile, the LCA outcomes were markedly less sensitive to increases in electricity, distilled water, and construction and equipment.

Techno-economic assessment

The production cost is the most important parameter for the profitability assessment. Production cost is directly related to the amount of product made in a year. The study estimated the production cost for producing bio-carbon at 2000 to 8000 tonnes per annum, ranging from €9080 to €7508 per tonne. The results were higher than the selling price of conventional anode materials. The price of battery-grade graphite ranged from 3000 to 7000 USD per tonne, with an average of 5200 USD by 2022 in North America (East Carbon 2024; Pro Graphite Shop 2026; Westwater Resources 2020). The net present value was estimated at €12.72 million and €72.08 million for annual quantities of 2000 and 8000 tons, respectively. A 15% profit margin was assumed. To represent the economic implications of the bio-based anode's performance, production costs were expressed in EUR/kWh of delivered energy (Fig. 8).

Fig. 7 Sensitivity ratios (SRs) across impact categories

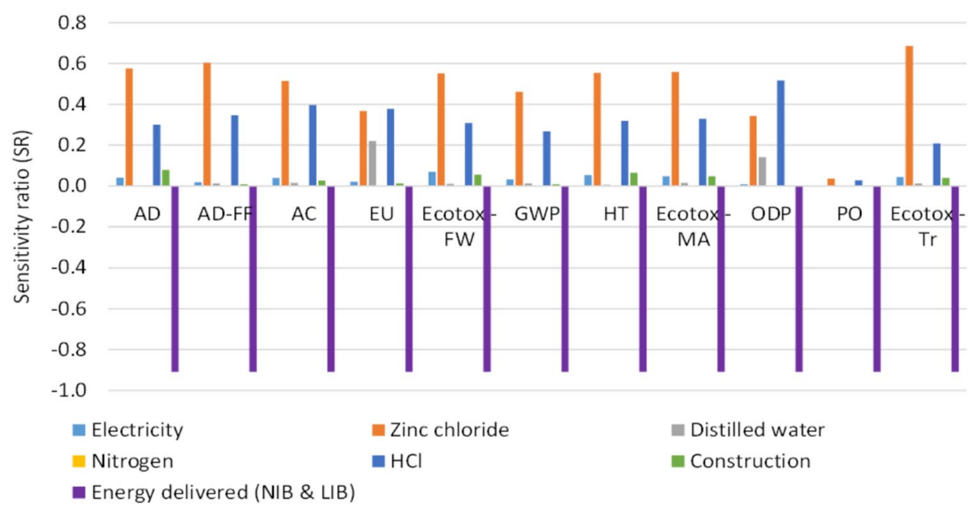


Fig. 8 The cost of production of bio-carbon

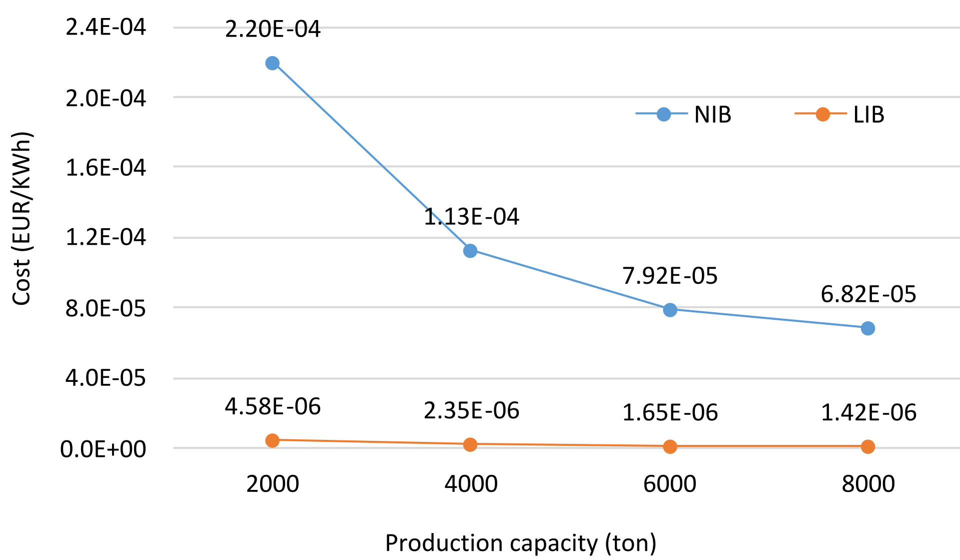


Figure 8 shows that the production of bio-based anodes per unit of energy delivered varies by application. The cost increased significantly when applied in NIB due to its lower specific capacity and fewer charging cycles compared to LIB. Assuming a production capacity of 8000 tonnes, the production cost per kWh would be 98% higher for NIB than for LIB. A sensitivity analysis was conducted to investigate how changes in prices affected production costs by increasing the price of five parameters by 10% each, one at a time. The most price-sensitive parameter was $ZnCl_2$ (SR 0.41), followed by HCl (SR 0.26) and electricity (SR 0.09). Meanwhile, N_2 (SR 0.002) and distilled water (SR 0.005) did not affect the production cost.

Discussion

Comparison with previous studies

Various GWP outcomes of manufacturing a bio-based anode have been reported elsewhere, employing different functional units such as per kg anode, per kWh stored capacity, or per mAh (Kulkarni et al. 2022; Liu et al. 2024; Peters et al. 2022). Our finding of 6.9 kg CO2 eq per kg bio-based anode is higher than those reported in other studies. The lowest GWP was obtained through electrochemical production, transforming amorphous carbon into highly crystalline graphite at a relatively low temperature (Kulkarni et al. 2022). The differences in reported GWP are due to differences in manufacturing method, geographical scope, and choices and LCA assumptions such as impact assessment method, data source, and so on. Liu et al. (2024) considered a production method somewhat similar to ours, but their GWP was only 51% of that in this study, despite higher electricity emissions in China and Germany. The reduced GWP stems largely from the chemicals used during impregnation and the absence of acid washing in the process. They used sulfuric and phosphoric acids for impregnation, whereas we chose zinc chloride and hydrochloric acid. In our study, using zinc chloride and hydrochloric acid during impregnation and acid wash accounted for 74% of the GWP, compared with their much lower acid contribution of about 10.5%. The Ecoinvent 3.11 database indicates that producing 1 kg of sulfuric acid in Europe causes a GWP of 0.09 kg CO_2 eq. These differences illustrate how technological choice, quantity, and the type of acid significantly affect environmental outcomes. Meanwhile, this study showed a lower impact compared with natural and synthetic graphite on a mass-based functional unit. The GWP comparison is summarised in Table 6.

Ideally, the comparison was conducted on a per-energy-delivered basis. Nevertheless, LCA studies at the material level, such as graphite or biochar, usually quantify results on a mass basis. Compared with Liu et al. (2024), our study yielded a much higher impact per unit of specific anode material capacity. However, there was no information on the number of charging cycles in their study, so a performance comparison throughout its lifetime could not be conducted.

Limitations and the implications of choices in the assessment

This study focuses on the costs and environmental impacts of manufacturing bio-based anode material from bark waste. The performance-based FU was selected as the primary FU, while a mass-based FU was also calculated, as it is the common FU in LCA studies of anode materials. Different raw materials, chemical inputs, and battery types, and application result in distinct anode characteristics. For example, the bio-based anode we produced exhibits 94% graphitisation but performs differently when used as an anode in NIB and LIB. Should the bio-based material perform worse than its conventional counterpart, consequences such as more frequent battery replacements may offset the overall impact. Nevertheless, calculating the impacts of material production is important to support the development of alternatives and manufacturing optimisations that can help reduce the net environmental impacts and costs.

The study is also based on laboratory work that was upscaled to an industrial level, which posed challenges and uncertainties. The operating conditions and dynamics between laboratory and industrial scales are obviously different: some reactions or processes may not scale linearly or may vary owing to the industrial environment's lack of

Table 6 Summary of GWP comparison

Type*	Scale	Multi-product handling	GWP (kg CO_2 eq)		References
			per tonne product	per mAh	
BB	Industrial	Internal energy recovery	1.53	n.a	Kulkarni et al. (2022)
BB	Industrial	Internal energy recovery	2.21–3.6	6.2–10.2	Liu et al. (2024)
BB	Industrial	System expansion	6.9	21.9–55.5	Own study
NG	Industrial	Not applied	9.6	n.a	Engels et al. (2022)
SG	Industrial	Physical allocation	13.8	n.a	Surovtseva et al. (2022)

BB = bio-based, NG = Natural graphite, SG = Synthetic graphite laboratory-level control precision. A recent study shows that the battery in an EV performed better in real-world applications than in laboratory tests (Golden 2024). We address this limitation by applying good-practice upscaling and conducting a sensitivity analysis to understand how the model behaves with respect to its underlying data and assumptions. The analysis could thus identify the most crucial input parameters and assumptions, highlighting areas where data quality needs improvement. Practitioners will naturally be concerned about the transferability of choices made throughout the study—for example, the selection of impact assessment methods, the allocation method, water loss during processing, the estimation of energy consumption for zinc chloride, and so on. Transparency and justification are therefore essential to enable reproducibility, improvement, and informed interpretation.

The TEA and LCA calculations were implemented in parallel, with consistent choices and assumptions, to produce reliable results. The assumptions for both analyses are the same: FU, boundaries, inputs, co-product handling, and production process. The consistency is important so that the trade-offs between environmental and economic factors can be compared fairly. Applying consistent assumptions alongside scenario analysis also provides insight into the insignificant environmental benefits from avoided electricity production. It also explains the importance of conducting LCAs on a case-by-case basis because the background system, methodological choices, and assumptions will yield different outcomes.

Implications and future directions

This research has implications for many stakeholders, including legislators, battery manufacturers, consumers, and LCA practitioners. The growing potential of agricultural biowaste as a carbon precursor, combined with advances in efficient carbonisation and graphitisation technologies, provides a solid foundation for scalable, low-carbon graphite production. As previously discussed, pre-treatment of agricultural residues (drying, crushing, grinding, and screening) ensures homogeneous feedstocks, whereas optimized carbonization (typically 220–900 °C depending on biomass type) and catalytic graphitization at low temperatures convert bio-waste into high-purity graphitic carbon (Fu et al. 2020; Ge et al. 2023; Ceylan and Sungur 2020; Aboudi et al. 2021; Tarelho et al. 2020; Selvarajoo et al. 2022; Wang et al. 2022a, b, c, d; Sieradzka et al. 2022; Weldekidan et al. 2022; Gai et al. 2021; Destyorini et al. 2020). These improvements make bio-based graphite a viable and sustainable alternative to mined graphite, reducing both environmental impact and manufacturing costs when properly integrated into industrial value chains. Nevertheless, the heavy reliance on chemicals ($ZnCl_2$ and HCl) during the production process could pose a vulnerability. When the process relies on specific chemicals, it operates efficiently with fewer contamination issues but is risky, as the entire operation is more susceptible to disruption (AGC Biologics 2025; Aramex 2025). Regional shortages, transportation issues, price volatility, or a government ban on the chemicals could significantly affect production or even stop it. From an R&D standpoint, more research on chemical alternatives is needed to reduce dependency and improve recovery strategies from the waste stream. Companies can also apply strategic stockpiling by purchasing chemicals in bulk to protect against disruption and diversify suppliers.

On the other hand, government funding for bio-based battery research and its facilities is, therefore, imperative: policies such as incentives, green procurement, and labelling can foster bio-based battery development (Flash Battery 2025; Leal Filho et al. 2025), for example, financial incentives, subsidies, or tax breaks for companies developing bio-based batteries, or providing funding accessible by small–medium enterprises (SMEs). The government can pioneer bio-based batteries through public procurement, giving preference to the products, and requiring labelling to build consumer awareness of the composition of the batteries they use. Battery producers may incorporate bio-based components into their products or directly engage in bio-based battery development by tapping government incentives.

The worldwide graphite supply chain is geographically concentrated, with China accounting for most of the natural graphite processing and synthetic graphite production for lithium-ion battery anodes (Golden 2025). China's recent export controls on some graphite components reveal possible vulnerabilities in global battery supply chains. Furthermore, traditional graphite supply chains often involve long-distance transportation from key producing regions such as China, Mozambique, and Brazil to battery production hubs in Europe and North America, thereby increasing logistics costs and associated emissions. In contrast, bio-based carbon compounds derived from locally available biomass residues have the potential to shorten supply chains and improve material security. These aspects were not explicitly included in the techno-economic model, but they may influence the economic and strategic attractiveness of localised bio-based anode production (IEA 2023; USGS, 2024).

They should also develop awareness of the environmental impacts of different materials and production processes, and incorporate this into their decision-making beyond mere financials. They may want to re-evaluate their suppliers as well, recognising that the same materials or products may have different impacts depending on the region and production process. Consumers will thus have alternatives between fossil-based and bio-based batteries, with some considerations such as environmental friendliness, cost, performance, and safety.

Consumers are willing to pay a premium for batteries that are more environmentally friendly (Choi et al. 2020). Transparency through labelling is therefore important for informing consumer decisions about bio-based battery selection. For CA practitioners, including database providers, continuously assessing the impacts and updating the database are necessary as the rate of electrification in various countries, and the differences in bio-based anode production methods can affect the environmental impacts markedly, as can the production methods of chemicals used in manufacturing (e.g., zinc chloride or hydrochloric acid, as this study illustrates). Working together with battery producers and reporting the results would be ideal to maintain the current impacts and database. However, companies are often secretive about proprietary knowledge, so researchers developing bio-based batteries must report their input–output inventories transparently if LCA practitioners are to use such work to assess impacts.

Research on bio-based batteries should be extended by exploring additional bio-materials, optimising production processes, testing components, and developing sustainable recycling methods. The focus can be on production optimisation and component testing: the former is related to production costs that are much higher than for synthetic or natural graphite, making optimisation crucial to reducing the price. The latter concerns battery performance. Bio-based battery performance should be competitive compared to the fossil-based status quo. Lower environmental impacts should not come at the cost of performance losses great enough to require more frequent battery replacement, thereby merely shifting the environmental impacts downstream.

Conclusion

This study contributes to the knowledge regarding the environmental and economic aspects of manufacturing bio-based anodes from waste bark. To date, there remains a paucity of studies examining the manufacturing impacts of bio-based anodes compared to those of graphite production. Our results show that the environmental impacts of bio-based anodes were lower than those of natural or synthetic graphite, although there was a trade-off in production cost. However, the outcomes were promising, as electrification is a key agenda for reducing climate change, so future graphite demand is expected to continue increasing. The hotspot analysis provides insights into how to reduce impacts by optimising the manufacturing process, focusing on the chemicals used for impregnation and acid washing: chemical inputs can be manufactured more sustainably, used in lower quantities, or used at different concentrations that incur lower impacts or costs while still delivering the same function. The energy source also played an important role: the pyrolysis process had only a small impact due to Sweden's clean energy sources, underscoring the importance of choosing a suitable manufacturing location. Our study helps better understand the environmental impacts and cost estimates of bio-based anode production. Future research and its industrial application can improve the current process to advance the decarbonization agenda through cleaner energy storage.

Acknowledgements We would like to thank the project funding for this work is part of the Green Battery project funded by the EU—Interreg Aurora.

Author contributions Bening Mayanti: data curation, LCA formal analysis, visualization, validation, writing draft, revision Theodore Azemtsop Manfo: writing draft, editing, revision Hafiz Haq: data curation, TEA formal analysis, visualization, validation, writing draft Nebiyu Girgibo: writing draft, editing, revision Jonas Markusson: supervision.

Funding Open Access funding provided by University of Vaasa. This work is part of the Green Battery project funded by the EU–Interreg Aurora.

Data availability All data supporting the findings of this study are available within the paper.

Declarations

Conflict of interest There are no conflict of interests between the authors or with the funding organizations.

Ethical approval Not applicable.

Consent to participate Not applicable.

Consent for publication All the authors agreed and gave their consent for publication.

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Authors and Affiliations

Bening Mayanti[1] · Theodore Azemtsop Manfo[2] · Hafiz Haq[1] · Nebiyu Girgibo[3] · Jonas Markusson[4]

✉ Bening Mayanti
bening.mayanti@uwasa.fi

✉ Theodore Azemtsop Manfo
theodore.azemtsop.manfo@uwasa.fi

1 Industrial Management, School of Technology and Innovations, University of Vaasa, Vaasa, Finland

2 Department of Electrical Engineering and Energy Technology, School of Technology and Innovations, University of Vaasa, Wolffintie 32, 65200 Vaasa, Finland

3 Vaasa Energy Business Innovations Centre (VEBIC), University of Vaasa, Vaasa, Finland

4 Research Institute of Sweden (RISE), Örnsköldsvik, Sweden