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Investigating Life Cycle Cost, Environmental and Social Impacts of a Lithium–Ion Battery Pack

ANTONELLA ACCARDO, GAIA GENTILUCCI, LUCA PONTONE, AND EZIO SPESSA

Department of Energy, Polytechnic of Turin, 10129 Turin, Italy

CORRESPONDING AUTHOR: ANTONELLA ACCARDO (e-mail: antonella.accardo@polito.it).

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ABSTRACT This study evaluates the environmental, economic, and social impacts of the life cycle of a battery pack for automotive applications. The analysis employs Life Cycle Assessment (LCA) for environmental assessment, Life Cycle Costing (LCC) for economic assessment, and Social Life Cycle Assessment (S-LCA) for social impact analysis. Key locations of non-European raw material extraction and refining are considered for the supply chain. Instead, European countries are considered the final destination for battery pack manufacturing and assembly, use, and End-of-Life (EoL). For the use and EoL phases, three scenarios are analyzed. The LCA results indicate that greenhouse gas emissions vary from 77.2 kg CO_2 -eq/kWh to 80.7 kg CO_2-eq/kWh across the evaluated scenarios. Similarly, the economic assessment estimates LCCs between 77.7 EUR/kWh and 79.4 EUR/kWh, depending on the scenario. The S-LCA results highlight significant risks related to fair pay across numerous countries during the raw material extraction phase, particularly for cobalt (Democratic Republic of the Congo), manganese (South Africa), nickel (Australia), lithium (Australia), and graphite (China). In addition, the score for health and safety concerns presents high risks associated with cobalt, manganese, and nickel mining. In contrast, no significant critical social impacts are found for the use and EoL phases.

INDEX TERMS Life cycle assessment (LCA), life cycle costing (LCC), social life cycle assessment (S-LCA), battery, electric vehicles.

I. INTRODUCTION

The goal of sustainable development is to ensure human well-being for present and future generations. Various methodologies support Life Cycle Sustainability Assessment (LCSA), including Social Life Cycle Assessment (S-LCA) for social impacts, Life Cycle Assessment (LCA) for environmental effects, and Life Cycle Costing (LCC) for economic factors [1]. The novelty of this research lies in its comprehensive integration of LCA, LCC, and S-LCA applied to electric vehicle battery systems, which addresses a critical gap in the existing literature.

A systematic review of studies published between 2016 and 2024 was carried out using specific keyword combinations. The first query, "lca AND lcc AND (slca OR s-lca)", returned 100 articles, providing an overview of the prevalence of studies integrating all three methodologies without focusing specifically on the automotive sector. In [2], the authors conducted a systematic literature review on the application of LCSA in the automotive sector, analyzing 18 automotive-related case studies in the 2008–2023 timeframe. These findings underscore the limited adoption of full LCSA frameworks in the automotive sector (18 out of 100). The review highlighted significant discrepancies across studies, primarily due to the absence of a harmonized methodological approach. These discrepancies include differences in functional units, system boundaries, indicator selection across the three sustainability pillars, result interpretation, databases used, selective aggregation of results, and the application of multi-criteria decision analysis methods [2]. In [3], [4], [5], [6], [7], [8], [9],

[10], battery electric vehicles (BEVs), fuel cell electric vehicles, and internal combustion engine vehicles are evaluated at vehicle level, considering the three pillars of sustainability. In [11], [12], [13], [14], [15], LCSA is applied to the production and use of fuels and electricity for road transport. In [16], [17], the analysis is moved to the component level, addressing alternators and vehicle dashboards. These findings underscore that past publications have predominantly focused on vehicles, often comparing different powertrain technologies, while batteries have not been commonly analyzed [2].

Only two studies have been found that focus on the application of LCSA on batteries. In [18], current and prospective battery technologies for electric vehicles are evaluated under environmental and socio-economic assessments. Results are reported per battery pack, assuming 80 kWh battery capacity, and a mileage of 150,000 km within a use period of 10 years. The three assessments are all conducted on the cradle-to-gate boundary. In [19], lithium-ion batteries with nickel-manganese-cobalt (NMC) cathode and iron-phosphate cathode for use in electric vehicles are compared under LCSA methodology. Results are reported per battery pack, assuming 150,000 km in a 10 years battery lifespan. While a cradle-to-grave boundary is assumed for LCA and LCC, S-LCA is evaluated for the manufacturing step only.

The second query, "(lca OR lcc OR slca OR s-lca) AND ev", identified 360 articles that focus on at least one of the three methodologies and apply it to electric vehicles. In [20], LCA is applied to evaluate a magnesium battery for potential use in electric vehicles. In [21], [22], several powertrain types for cars and buses are compared, focusing on the environmental pillar. In [23], [24], [25], different vehicles are evaluated, focusing on the economic pillar, sometimes considering the Total Cost of Ownership (TCO) method as in [23], sometimes considering the LCC method as in [25]. In [25], [26], both environmental and economic pillars are considered to evaluate electric vehicles [25] and batteries [26]. The query "(slca OR s-lca) AND ev", aimed at identifying social life cycle analyses specific to electric vehicles, returned three studies [18], [27], [28]. The most recent is [27], where social life cycle impact assessment from cradle to grave is applied to an electric vehicle lithium-ion battery.

Furthermore, NMC cathodes are analized under LCA in 13 articles, under S-LCA analysis in 3 articles, under a more comprehensive LCSA in 2 articles. Specifically for the NMC811 cathode, the search "nmc811 AND lca" resulted in 12 articles, including [29], which identifies the processing phase as the main source of environmental impacts in the cobalt production chain.

As a summary, only a limited number of articles fully incorporate the three methodologies, especially when batteries are concerned. Lastly, significant methodological inconsistencies were identified, particularly in terms of metric comparability and system boundary alignment. This study builds on these findings by providing a holistic and harmonized assessment framework that considers the entire battery pack life cycle from cradle to grave.

II. METHODOLOGY

Hereafter, all assumptions shared across LCA, LCC, and S-LCA are presented, while pillar-specific assumptions are detailed in the following sub-sections. This study assesses the sustainability of a 518 kg, 75 kWh lithium-ion battery pack by assessing its economic, environmental, and social impacts. The 75 kWh battery pack was selected as a representative value for a mid-size battery electric vehicle, which constitutes a large share of the current EV market in Europe. The battery features a graphite anode and an NMC811 cathode. Sustainability is analyzed using LCC for economic evaluation, LCA for environmental impact, and S-LCA for social implications. The assessment, whether LCA, LCC, or S-LCA, considers alternatively 1 battery pack and 1 kWh of battery pack capacity as the functional unit and follows a cradle-to-grave approach. This includes raw material extraction, processing, utilization, and End-of-Life (EoL). The mass breakdown of the components was based on [30], [31]. The anode and cathode were broken down into their respective processing stages. For the anode, the transformation of raw graphite into concentrated, spherical, and purified forms, including the application of coatings, was considered based on [32]. The material breakdown of the cathode was based on [31]. For logistics, all transport distances have been quantified in tonnes-kilometers. For rail transport, diesel trains have been assumed due to their prevalence in certain regions. Although this assumption holds for African countries and China, it is less accurate for Europe, where approximately 60% of the freight trains operate on electric power [33], but conservative.

A. SCENARIO SET-UP

For the use phase, three scenarios have been identified based on country-specific electricity mixes and average annual driving distances. To set-up the scenarios, the following criteria have been considered: (1) charging station availability, (2) BEV-to-charger ratio, and (3) charging costs. According to the first criterion, the five EU countries with the highest number of charging stations were Germany and France (17% each), the Netherlands (16%), Belgium and Italy (6% each). Germany, France, the Netherlands, Belgium, and Italy account for 62% of the EU charging points. In addition, countries were grouped according to the second criterion– ratio of electric vehicles to charging points– with Germany representing the highest BEV-to-charger ratio, France-Italy mid-range, and the Netherlands-Belgium the lowest. Lastly, the third criterion was also taken into account, but no significant differences were found among the considered countries. The scenarios were set up primarily on the basis of the first criterion, but countries were grouped considering a reasonable balance with the other two criteria (Table 1). A sample of 240 EVs was analyzed [34], estimating a 75 kWh battery pack's efficiency at 177.9 Wh/km. Since this value reflects the overall consumption of the vehicle, the delta approach [35] was applied to consider the specific share of consumption of the battery, calculated as 3.6 Wh/km. Then, two additional factors were considered, namely the average annual driving distance and

TABLE 1. Use Phase Scenarios (S)

S	Assumptions			
	Country	*Charging points %*	*BEV-to-charger ratio*	*Average charging cost (€/kWh)*
1	France	17	7.87	0.45
	Italy	6	5.86	0.50
2	Germany	17	14.11	0.53
3	Netherlands	16	3.73	0.53
	Belgium	6	4.26	0.55

TABLE 2. End-of-Life Scenarios (S)

	Country	*Representative company*	*Original process*	*Electricity cost (€/kWh)*
1	Sweden	Northvolt	Hydrometallurgy	0.015
2	Ungary	SungEel Hitech	Combined Pyro-hydro	0.141
	Netherlands	TES-AMM	Combined Pyro-hydro	0.164
	France	Recupyl	Hydrometallurgy	0.168
3	Belgium	Umicore	Combined Pyro-hydro	0.205
	Germany	Primobius	Hydrometallurgy	0.227

the battery life. For the annual driving distance, 2022 data were obtained from [36] and used to calculate a weighted average distance for the five selected countries, based on their respective populations. Since data for Belgium were not available, the EU-wide average was used as a proxy. An eight-year battery life has been considered based on the warranty period usually provided by manufacturers.

Recovery of critical materials is essential not only for environmental sustainability, but also for EU regulatory compliance. Specifically, Article 8, paragraphs 2 and 3 of Regulation 2023/1542 [37] require that by August 18, 2031, the minimum percentage of recovered materials in new batteries must be 16% for cobalt, 6% for nickel, and 6% for lithium. In addition, by August 18, 2036, these targets will increase to 26% for cobalt, 12% for nickel, and 15% for lithium.

Similarly to the use phase, the EoL phase is analyzed under three different scenarios that differ according to recycler countries with consequences derived from the different electricity mixes (Table 2). To set up the scenarios, EU companies operating in the recycling sector were identified. Based on their country of operation, three scenarios were established, considering both the recovery rates of specific materials and the environmental and economic effects of recovery operations. First, the recovery technique used by each company was identified. These techniques fall into two main categories: hydrometallurgical and pyrometallurgical processes. Among these, the hydrometallurgical process was selected for analysis and kept constant across scenarios. Material and energy flows, and recovery rates were taken from [38] considering the scenario based on the hydrometallurgical data of [39]. The same methodological approach was applied across all scenarios, ensuring that the recovery technique, and costs associated with the energy and ancillary materials used in the recycling process are kept constant. The economic benefit of material recovery is evaluated as a cost-saving mechanism that reduces the need for the extraction of new raw materials. For each recovered material, extraction and processing costs were identified and multiplied by the corresponding recovered mass.

For emissions related to logistics along the supply chain, the identification of the main producers and refiners of the materials of interest is based on [40], [41], while [42] and [43] have been used to calculate the land and sea routes. For each component of the battery pack, the main materials have been identified. Subsequently, for each material, the processing steps that the material undergoes before being assembled into the battery pack were identified. For the cathode, the following raw materials have been analyzed: cobalt sulfate, nickel sulfate, manganese sulfate, and lithium hydroxide (from spodumene). The cathode production process was assumed to take place in China because it holds more than 90% of the global cathode production volumes, but also because it holds most of the supply agreements [44]. Cobalt sulfate is assumed to be derived from the Democratic Republic of Congo (DRC) because the share of extraction in the DRC constitutes more than 70% of the total [40]. In addition, about 75% of the refining processes are carried out in China [40]. To estimate the distance of transport, a representative mining site located in Lubumbashi has been assumed based on [45]. The chosen site is the most representative both because it is part of Chinese investments and because it has a railway connection. Among the ports of interest (Durban, Dar es Salaam, Beira), which are connected to the mining sites and the eastern ports, Durban has been taken as a reference, being the only one connected by rail among the three and the main one for cobalt exports according to [46]. The distance between the extraction site and the port is about 2520 km, assuming a rail connection. The port considered for cobalt imports in China is the port of Shanghai, because it is also one of the main ports for the unloading of goods of this type [47]. Once it arrives in Shanghai, it is assumed that cobalt sulfate is transported to Changsha in Hunan province for the production of the cathode, because this is the province where the production is the most concentrated and the city of Changsha has the main plants in the country [48]. Over 70% of global battery cell production occurred in China as of 2019 [49]. In contrast, this study assumes that both cell production and battery pack assembly take place in France. The choice of France for battery pack assembly reflects current industrial practices in Europe, where vehicle manufacturers typically import cells or modules and assemble battery packs locally to meet specific vehicle requirements. Instead, the assumption of European-based cell production—specifically in France—was made to reflect the significant ongoing investments and policy initiatives aimed at establishing a competitive and sustainable

TABLE 3. Raw Material Extraction and Logistic Assumptions

	Departure site	*Arrival site*	*Distance (km)*	*Carrier*
$CoSO_4$	Lubumbashi (DRC)	Changsha (CN)	2520 to Durban port; 12295 to Shanghai port; 1016 to Changsha	train ship train
$NiSO_4$	Palapo Sulawesy (ID)	Changsha (CN)	neglected; 4769 to Shanghai port; 1016 to Changsha	– ship train
$MnSO_4$	Hotazel (ZA)	Changsha (CN)	976 to Durban port; 12295 to Shanghai port 1016 to Changsha	train ship train
LiOH	Mt. Holland mine Bridgetown (AU)	Changsha (CN)	248 to Fremantle port; 7421 to Shanghai port; 1016 to Changsha	train; ship; train
All precursors	Changsha (CN)	Douvrin (FR)	1016 to Shanghai port 19622 to Rotterdam port 192 to Douvrin	train ship train
Natural graphite	Shanxy (CN)	Douvrin (FR)	1256 to Shanghai port 19622 to Rotterdam port 192 to Douvrin	train ship train
EC, DMC	Guangdong (CN)	Douvrin (FR)	1384 to Shanghai port 19622 to Rotterdam port 192 to Douvrin	train ship train
$LiPF_6$	Zhejiang (CN)	Douvrin (FR)	192 to Shanghai 19622 to Rotterdam port 192 to Douvrin	train ship train
Plastics	Weinheim (GE)	Douvrin (FR)	496 to Douvrin	train
Cu sheets	Tatabánya (HU)	Douvrin (FR)	1352 to Douvrin	train
Al sheets	Grevenbroich (HU)	Douvrin (FR)	312 to Douvrin	train
Al plates	Singen (GE)	Douvrin (FR)	616 to Douvrin	train
Steel	Linz (AT)	Douvrin (FR)	992 to Douvrin	train

battery manufacturing ecosystem within the EU. In particular, we refer to the Automotive Cells Company (ACC) gigafactory inaugurated in France in 2023 as a representative facility [50]. Similar reasoning has been performed for the other logistic assumptions based on [40], [46], [47], [50], [51], [52], [53], [54], [55] (Table 3).

B. LIFE CYCLE COSTING

The development of the LCC model in this study was structured into the following key phases: 1) identification of material costs sourced in the upstream phase of the life cycle; 2) collection of energy cost data by country; 3) collection of fuel cost data by country; 4) calculation of logistics costs for material transportation; 5) identification of costs for the end-of-life phase, 6) differentiation of cost scenarios based on country-specific energy prices; 7) estimation of secondary material residual value. Additional costs—such as capital equipment, labor, value-added taxes, and manufacturer profit margins—were not included in this analysis. According to [26], these excluded costs account for approximately 30% of the total battery production cost. The market prices of the materials were obtained from various sources [56], [57], [58], [59]. The electricity price data for each country was mainly obtained from [60], with the exception of Sweden, where [61] was used. All values, expressed in EUR/kWh, refer to the industrial electricity market and are based on prices recorded on 30 September 2024. In addition to electricity prices, fuel prices were collected for the relevant countries from [60] based on data recorded on September 30, 2024. Further consideration was given to the primary fuels used in maritime and air transport: heavy fuel oil (HFO) for cargo ships and kerosene for aviation. Fuel prices were obtained from [62] for HFO and [59] for kerosene, both subject to market fluctuations. For maritime transport, HFO prices are typically reported as USD/T for the four major ports, with an average calculated for the four top ports and the twenty top ports. This study adopts the top 20-port average, converted and expressed in EUR/T. Similarly, kerosene prices, initially reported as USD/kg, were converted into EUR/T. Price data for both HFO and kerosene were taken on 4 November 2024. In this study, LCC results are expressed in EUR per kWh of battery capacity, aligning with the functional unit commonly used in battery LCAs. It is important to note that these costs are not annualized, and no amortization or discounting of investments has been applied. The EUR/kWh values reflect the total life cycle costs allocated per unit of battery capacity, without consideration of time-based financial flows. This approach facilitates comparability with environmental indicators calculated using the same functional unit.

C. LIFE CYCLE ASSESSMENT

The assumptions underlying the LCA analysis are presented in this section, where not already detailed in the previous sections, to avoid repetition. To evaluate the environmental impacts of the battery pack, the LCA methodology has been used according to ISO 14040 and ISO 14044 standards [63], [64]. The software used for the LCA analysis is SimaPro [33], in conjunction with the Ecoinvent database [65]. The adopted life cycle impact assessment method is the Environmental Footprint (EF) 3.0, which includes a broad set of environmental impact categories. The most relevant categories resulted to be the following: use of mineral and metal resources, freshwater ecotoxicity, climate change, acidification, and the use of fossil resources.

D. SOCIAL LIFE CYCLE ASSESSMENT

This study performs a S-LCA to evaluate the social impacts of a battery pack. S-LCA follows the UNEP guidelines [1] and the standard ISO 14075 [66]. Stakeholder categories are defined based on life cycle stages: "workers" and "society" are considered for raw material extraction, production, and recycling, while "consumers" and "society" are assessed for the use phase. For the impact assessment, a reference scale assessment approach is used. This method uses a reference scale to categorize specific intensity levels of social attributes [66]. Each indicator is assigned a risk value on a scale from 1 (low risk) to 4 (very high risk). In cases where data are unavailable for certain subcategories within a country, a risk value of 4 is assigned to account for the lack of information and ensure a precautionary approach.

The production phase considers key extraction locations of raw materials as in Table 3. Workers and society are chosen as the main stakeholder categories. For workers, the following impact subcategories are considered based on [67]: freedom of association, child labor, fair pay, hours of work, forced labor, health and safety, and social security. The freedom of association indicator evaluates the extent to which workers can form and join trade unions. It is measured on a scale from 0 to 10, where 0 signifies maximum freedom, and 10 represents the highest level of restriction. Among the countries analyzed, China exhibits the highest restriction (i.e., 9.1), while Austria demonstrates the highest level of freedom (i.e., 0.2). Regarding child labor, the value is expressed as a percentage and DRC has the highest reported prevalence, with 15.1% of children engaged in labor. In contrast, data on child labor are unavailable for China. The fair pay indicator measures the wage disparity between men and women, expressed in purchasing power parity (i.e., PPP $). In many countries, data on this indicator are unavailable, including the DRC, Indonesia, Australia, and China. The lowest value is found in southern Africa. With regarding to weekly working hours, the highest value is recorded in China. Forced labor is measured using the Global Slavery Index. According to [68], estimates of prevalence per thousand people and the total number of affected individuals were calculated based on both individual and country-level risk factors for modern slavery. The highest values are reported in the DRC, Indonesia, and Hungary. Workplace safety is assessed by considering the number of occupational accidents. Data is unavailable for the DRC, Indonesia, and Southern Africa, while China reports the highest incidence, with 1,539.5 non-fatal occupational injuries per 100,000 workers in Hong Kong and Macau. The final indicator is social security, defined as the proportion of the population not covered by at least one social protection benefit. The DRC records the highest value, with 94.8% of the population lacking coverage. For society, the following impact subcategories are considered: technological development, corruption and public commitments to sustainability. In the case of technological development, the global innovation index is examined, that is an annual ranking of countries by their capacity for, and success in, innovation, published by the World Intellectual Property Organization. The corruption is based on Corruption Perceptions Index in [69]. Lastly, the Environmental Performance Index (EPI) is considered to evaluate the public commitments to sustainability. Overall, the EPI determines the degree to which a country works toward mitigation of climate change, ensuring ecosystem vitality, and maintaining environmental health [70]. For the use phase, the subcategories considered are consumers and society. The evaluation of consumer-related aspects focuses exclusively on health and safety. From the consumer perspective, this criterion is assessed based on the presence or absence of key consumer protection measures identified by [71], including the establishment of a designated consumer protection contact point, the existence of comprehensive consumer protection laws, the presence of a dedicated regulatory authority

TABLE 4. Life Cycle Phases and Total Costs for Different Combined Scenarios

Life cycle phases	Scenario 1-1	Scenario 1-2	Scenario 1-3
Extraction and production	5704	5704	5704
Logistics	2.07	2.07	2.07
Use	133.4	133.4	133.4
EoL	-10.4	41.84	63.28
Total EUR	**5829**	**5882**	**5903**
Life cycle phases	**Scenario 2-1**	**Scenario 2-2**	**Scenario 2-3**
Extraction and production	5704	5704	5704
Logistics	2.07	2.07	2.07
Use	185.9	185.9	185.9
EoL	-10.4	41.84	63.28
Total EUR	**5882**	**5934**	**5956**
	Scenario 3-1	**Scenario 3-2**	**Scenario 3-3**
Extraction and production	5704	5704	5704
Logistics	2.07	2.07	2.07
Use	161.9	1161.9	161.9
EoL	-10.4	41.84	63.28
Total EUR	**5858**	**5910**	**5932**

Table structure: Combined scenarios 1-1, 1-2, 1-3 share the use phase assumptions (France-Italy); Combined scenarios 1-1, 2-1, 3-1 share the EoL assumptions (Sweden) (see Tables 1–2).

TABLE 5. Life Cycle Phases and Total CO_2-Eq Emissions for Different Combined Scenarios

Life cycle phases	Scenario 1-1	Scenario 1-2	Scenario 1-3
Extraction and production	6128	6128	6128
Logistics	41.6	41.6	41.6
Use	66.9	66.9	66.9
EoL	-444	-329	-313
Total kg CO_2-eq	**5793**	**5908**	**5924**
Life cycle phases	**Scenario 2-1**	**Scenario 2-2**	**Scenario 2-3**
Extraction and production	6128	6128	6128
Logistics	41.6	41.6	41.6
Use	199	199	199
EoL	-444	-329	-313
Total kg CO_2-eq	**5925**	**6040**	**6056**
	Scenario 3-1	**Scenario 3-2**	**Scenario 3-3**
Extraction and production	6128	6128	6128
Logistics	41.6	41.6	41.6
Use	131	131	131
EoL	-444	-329	-313
Total kg CO_2-eq	**5857**	**5972**	**5988**

Table structure: Combined scenarios 1-1, 1-2, 1-3 share the use phase assumptions (France-Italy); Combined scenarios 1-1, 2-1, 3-1 share the EoL assumptions (Sweden) (see Tables 1–2).

TABLE 6. Results of the LCA in the Combined Scenario 1-1

Impact category	**Results**			
	Total	***Extraction, Production, & Logistics***	***Use***	***EoL***
Climate change (kg CO_{2eq})	5.79E3	6.17E3	6.69E1	-4.44E2
Use of mineral and metal resources (kg Sb_{eq})	1.16E0	1.24E0	2.53E-4	-7.67E-2
Freshwater ecotoxicity (CTU_e)	4.94E5	5.42E5	1.37E3	-5.01E4
Acidification (mol $H+_{eq}$)	6.60E1	1.16E2	3.22E-1	-5.02E1
Use of fossil resources (MJ)	8.61E4	8.80E4	2.66E3	-4.47E3

Results obtained considering scenarios 1 for use and recovery.

overseeing consumer protection, and the implementation of research and analytical initiatives focused on consumer protection. For each country, the presence of these consumer protection measures was indicated with a value of 0, while their absence was marked with a value of 1. Notably, only Italy and Germany received a value of 1, indicating a lack of research and analytical initiatives related to consumer protection. For the society as stakeholder, corruption and public commitments to sustainability are evaluated. For the end-of-life phase, workers and society are identified as the primary stakeholder categories. For workers, the assessment includes the same impact subcategories considered in the production phase (i.e., freedom of association, child labor, fair pay, working hours, forced labor, health and safety, and social security). Instead, for society, the study considers technological development, corruption, and public commitments to sustainability. Among this phase, the most concerning issues include inadequate fair pay in Hungary, significant health and safety risks in France due to a high rate of work accidents (i.e., there were 3,043 non-fatal incidents per 100,000 workers), and a poor score on the Corruption Perceptions Index in Hungary.

III. RESULTS

Figs. 1–2 show the greenhouse gas (GHG) emissions and LCC results of the combined scenarios. Fig. 1 shows the results per battery pack, while Fig. 2 shows the results per 1 kWh battery capacity.

The results show that life cycle costs range between 77.7 EUR/kWh and 79.4 EUR/kWh. The extraction and production phases account for more than 90% of the total costs, while logistic costs were negligible. The costs of the use phase vary by country, with France-Italy being the cheapest (133.4 EUR/batterypack) and Germany the most expensive (185.9 EUR/batterypack) (Table 4).

The LCA results indicate that the GHG emissions range between 77.2 kg CO_2 -eq/kWh and 80.7 kg CO_2 -eq/kWh, depending on the scenario (Fig. 1). The use and EoL phases are electricity mix-dependent, with Italy-France having the

TABLE 7. Social Assessment in the Main Producing Countries of the Raw Materials Used in Battery Pack

Stakeholder category	Subcategory	Co DRC	Mn ZA	Ni ID	Li AU	Gr CN	Al GE	Cu HU	Stl AT	Assembly FR
Workers	Freedom to associate	2	1	1	1	4	1	1	1	1
	Child labor	4	1	2	1	4	1	1	1	1
	Fair pay	4	4	4	4	4	1	3	1	1
	Work hours	2	3	3	2	3	2	2	2	2
	Forced labor	2	1	2	1	2	1	2	1	1
	Health and safety	4	4	4	2	2	2	1	2	3
	Social security	4	2	2	1	1	1	1	1	1
Society	Technological development	4	3	3	2	1	1	2	1	1
	Corruption	4	3	3	1	3	1	3	2	2
	Public commitments to sustainability	3	3	3	2	3	2	2	2	2

Co: Cobalt; DRC: Democratic Republic of Congo; Mn: Manganese; ZA: South Africa; NI: Nickel; ID: Indonesia; LI: lithium; AU: Australia; Gr: Graphite; CN: China; Al: Aluminum; GE: Germany; Cu: Copper; HU: Hungary; Stl: Steel; AT: Austria; FR: France.

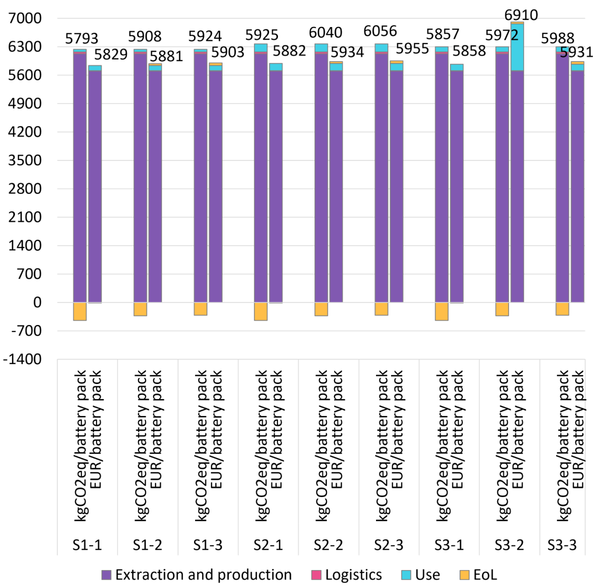

FIGURE 1. Results of the combined scenarios per battery pack.

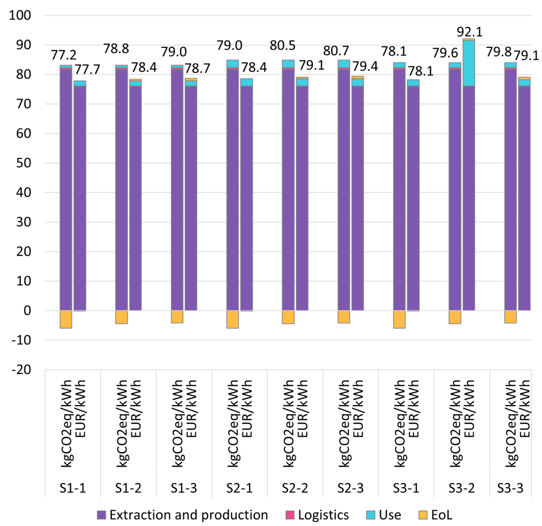

FIGURE 2. Results of the combined scenarios per 1 kWh capacity.

TABLE 8. Social Assessment in the Countries Considered for Use Phase

Stakeholder category	Subcategories	FR	IT	GE	BE	NL
Consumers	Safe and security	1	1	1	1	1
	Public commitment to sustainability	2	2	2	2	2
Society	Corruption	2	2	1	2	1

FR: France; IT: Italy; GE: Germany; BE: Belgium; NL: Netherlands

TABLE 9. Social Assessment in the Countries Considered for End-of-Life Phase

Stakeholder category	Subcategories	SE	NL	FR	HU	BE	GE
Workers	Freedom to associate	1	1	1	1	1	1
	Child labor	1	1	1	1	1	1
	Fair pay	1	1	1	3	1	1
	Work hours	2	1	2	2	2	2
	Forced labor	1	1	1	2	1	1
	Health and safety	1	2	3	1	2	2
	Social security	1	1	1	1	1	1
	Technological development	1	1	1	2	1	1
Society	Corruption	1	1	2	3	2	1
	Public commitment to sustainability	2	2	2	2	2	2

SE: Sweden; NL: Netherlands; FR: France; HU: Hungary; BE: Belgium; GE: Germany

lowest use phase impact (66.9 kg CO_2 -eq/batterypack), Germany the highest (199 kg CO_2 -eq/batterypack), and Sweden being the most sustainable in the EoL contribution (−444 kg CO_2-eq/batterypack) (Table 5).

The best combined scenario for LCC turned out to be the one in which the vehicle is used in France or Italy and the battery pack is recycled in Sweden (Scenario 1–1). This optimal combination resulted in a total battery cost of 5829 EUR. Specifically, in the Swedish recycling scenario, the financial return compensates 5.73% of the associated costs,

TABLE 10. Social Assessment Across Life Cycle Phases Under Different Scenarios

Life cycle phase	Stakeholder	Impact subcategory	Scenario					
			1-1 3-1	1-2 3-2	1-3 3-3	2-1	2-2	2-3
Raw material extraction and manufacturing	Workers	Freedom to associate	1	1	1	1	1	1
		Child labor	2	2	2	2	2	2
		Fair pay	3	3	3	3	3	3
		Work hours	2	2	2	2	2	2
		Forced labor	1	1	1	1	1	1
		Health and safety	3	3	3	3	3	3
		Social security	2	2	2	2	2	2
	Society	Technological development	2	2	2	2	2	2
		Corruption	2	2	2	2	2	2
		Public commitments to sustainability	2	2	2	2	2	2
Use phase	Consumers	Safe and security	1	1	1	1	1	1
	Society	Public commitments to sustainability	2	2	2	2	2	2
		Corruption	2	2	2	1	1	1
EoL	Workers	Freedom to associate	1	1	1	1	1	1
		Child labor	1	1	1	1	1	1
		Fair pay	1	2	1	1	2	1
		Work hours	2	2	2	2	2	2
		Forced labor	1	1	1	1	1	1
		Health and safety	1	2	2	1	2	2
		Social security	1	1	1	1	1	1
	Society	Technological development	1	1	1	1	1	1
		Corruption	1	2	2	1	2	2
		Public commitments to sustainability	2	2	2	2	2	2

demonstrating a potential economic advantage in the hydrometallurgical recycling process under certain conditions.

The best combined scenario for GHG emissions turned out to be the so-called Scenario 1–1, namely the one in which the vehicle is used in Germany and the battery pack is recycled in Sweden. This optimal combination resulted in total battery GHG emissions of 5793 kg CO_2-eq.

Table 6 shows the LCA results of the best scenario combination selected based on Figs. 1–2. Results are given per battery pack. The production phase (extraction, production & logistics) has the highest GHG emissions (6170 kg CO_2-eq/batterypack) due to the raw material extraction.

Table 7 shows the social risk in the main producing countries of the raw materials used in the battery pack. Each country is assigned a score from 1 to 4, indicating the level of risk. It is evident that the highest risks (4) are recorded mainly in the extraction countries, particularly for cobalt (DRC), manganese (South Africa), nickel (Australia), and graphite (China). These countries face significant social challenges, especially regarding issues such as child labor and fair pay. In addition, the score for health and safety concerns presents high risks associated with cobalt, manganese, and nickel mining.

In contrast, no significant critical social impacts are found for the use and end-of-life phases in EU countries. Table 8 highlights the social risk related to the use phase. The values range from 1 to 2, indicating a low to medium risk. A medium risk level is consistently recorded in terms of public commitments to sustainability in all countries. Table 9 highlights the social risk related to the end-of-life phase. The risk values differ, generally ranging from 1 to 2. The only exceptions are France, where the risk level for occupational safety is 3, and Hungary, where the risk level for fair pay and corruption is 3.

Table 10 presents the social risk levels across different life cycle phases under various scenarios, as defined in Tables 2 and 3. Certain combined scenarios (such as Scenarios 1-1 and 3-1) are presented in the same column, as the averaging of different scores led to identical outcomes, despite differing underlying assumptions. As in the LCA and LCC assessments, the raw material extraction and manufacturing phases are not affected by scenario variation. Overall, the risks range from low to medium for most subcategories. However, the "fair pay" and "health and safety" subcategories exhibit a higher risk level (i.e., 3) in all scenarios during the raw material extraction and manufacturing phases. These elevated risks are primarily associated with countries where mining activities tend to have more severe social consequences. In contrast, the remaining subcategories consistently show low to medium risk levels across all life cycle phases, suggesting relatively lower social impact. This variation in risk reflects differences in national social responsibility standards and regulatory frameworks, which influence the degree of social risk at each stage of the product life cycle.

Fig. 3 presents the S-LCA results for the battery pack under the best combined scenario (i.e., scenario 2-1), namely the one in which the vehicle is used in Germany and the battery pack is recycled in Sweden. Risk levels are categorized on a scale from 1 to 4, where 1 indicates low risk (green), 2 medium

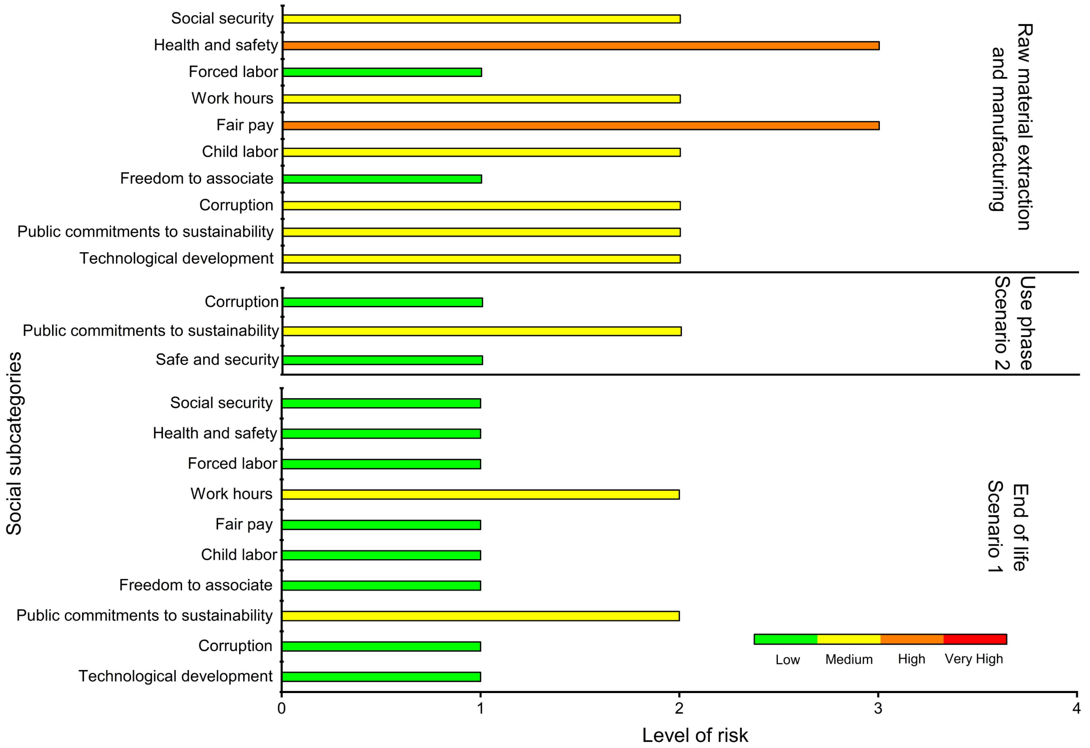

FIGURE 3. **S-LCA results of battery pack in the best combined scenario (2-1).**

risk (yellow), 3 high risk (orange), and 4 very high risk (red), as shown in the color bar. Scenario variations were applied only to the use and EoL phases, in line with the approach used in the LCA and LCC assessments. Among these phases, only "public commitments to sustainability", in both the use and EoL phases, and "work hours", in the EoL phase, were identified as medium risk. While the overall social risks in the use and EoL phases appear generally low under optimal conditions, these medium-risk subcategories highlight areas where additional efforts in transparency and labor practices are still needed.

IV. DISCUSSION

While the study explores scenario-based variability related to use and EoL phase countries, additional sources of variability could further influence the outcomes. For example, electric vehicle efficiency can vary significantly between vehicle models and could affect life cycle energy use and emissions per km driven. Including this parameter in the LCA could add nuance to the results, particularly when comparing different use cases or battery chemistries. Similarly, although we argue that cathode production is likely to occur in China, based on current market conditions, domestic or European production could become more relevant in future supply chains. This China-to-Europe market shift would affect transportation emissions and energy sources, potentially lowering the overall carbon footprint depending on the regional energy mix. Furthermore, battery life cycle costs may vary on the basis of regional-dependent prices such as electricity and human labor prices. Similarly, social life cycle risks may vary on the basis of different labor conditions, health and safety measures, and the level of societal awareness about sustainability. In the economic analysis, fixed material prices were used for all components. However, historical data shows that prices of key materials such as lithium, cobalt, and nickel can fluctuate significantly over time. Incorporating high- and low-price ranges (e.g., over the past decade) could better represent the economic uncertainty in battery production and usage. Lastly, the use of recycled materials would affect LCA, LCC, and S-LCA.

V. CONCLUSION

This study assesses the environmental, economic, and social impacts of a 75–kWh NMC811 battery pack throughout its life cycle using LCA, LCC, and S-LCA. The LCA results indicate that the GHG emissions range between 77.2 kg CO_2 -eq/kWh and 80.7 kg CO_2 -eq/kWh, depending on the scenario. The best combined scenario for GHG emissions turned out to be the so-called Scenario 1-1, namely the one in which the vehicle is used in France or Italy and the battery pack is recycled in Sweden. This optimal combination resulted in total battery GHG emissions of 5793 kg CO_2-eq/batterypack. The upstream phase has the highest GHG emissions due to raw material extraction, while the use and EoL phases depend on the electricity mix. Furthermore, extraction and production account for the main cost drivers, with France-Italy as the cheapest scenario for the use phase and Sweden as the most cost-effective for recycling, leading to a total battery cost of 5829 EUR/batterypack. The results show that the LCC ranges between 77.7 EUR/kWh and 79.4 EUR/kWh. Lastly, in terms of S-LCA, results revealed that raw material extraction occurs in high-risk regions, while use and EoL phases take place in regulated environments with strong sustainability and labor protections. Unlike the LCA and LCC results, the social assessment identified scenario 2-1 as the most favorable. Notably, all three methodologies share the EoL phase in Sweden, suggesting that battery recycling in this context contributes positively across environmental, economic, and social dimensions. Additionally, Germany emerges as the most advantageous option for the use phase in the S-LCA. Future work could benefit from a more comprehensive uncertainty analysis, including probabilistic modeling or Monte Carlo simulations. Nevertheless, the current scenario-based approach provides useful insights into how system-level decisions and regional configurations influence the environmental and economic performance of the battery system.

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REFERENCES

[1] C. Norris et al., "Guidelines for social life cycle assessement of products and organizations," 2020. [Online]. Available: https://www.lifecycleinitiative.org/library/guidelines-for-social-life-cycle-assessment-of-products-and-organisations-2020/
[2] S. Ostojic and M. Traverso, "Application of Life Cycle Sustainability Assessment in the automotive sector–A systematic literature review," *Sustain. Prod. Consumption*, vol. 47, pp. 105–127, Jun. 2024. [Online]. Available: https://linkinghub.elsevier.com/retrieve/pii/S2352550924000940
[3] D. Alramadhan, S. Alabduljaleel, D. Almulla, and E. Aleisa, "Multi-criteria optimization for electric vehicles in an arid oil state based on a triple bottom line approach," *Int. J. Life Cycle Assessment*, vol. 27, no. 1, pp. 138–156, Jan. 2022, doi: 10.1007/s11367-021-02009-6.
[4] N. Elagouz et al., "Rethinking mobility strategies for mega-sporting events: A global multiregional input-output-based hybrid life cycle sustainability assessment of alternative fuel bus technologies," *Sustain. Prod. Consumption*, vol. 33, pp. 767–787, Sep. 2022. [Online]. Available: https://linkinghub.elsevier.com/retrieve/pii/S2352550922002068
[5] M. Haase et al., "Multi-criteria decision analysis for prospective sustainability assessment of alternative technologies and fuels for individual motorized transport," *Clean Technol. Environ. Policy*, vol. 24, no. 10, pp. 3171–3197, Dec. 2022, doi: 10.1007/s10098-022-02407-w.
[6] N. C. Onat, M. Kucukvar, O. Tatari, and G. Egilmez, "Integration of system dynamics approach toward deepening and broadening the life cycle sustainability assessment framework: A case for electric vehicles," *Int. J. Life Cycle Assessment*, vol. 21, no. 7, pp. 1009–1034, Jul. 2016, doi: 10.1007/s11367-016-1070-4.
[7] N. C. Onat, "How to compare sustainability impacts of alternative fuel vehicles?," *Transp. Res. Part D: Transport Environ.*, vol. 102, Jan. 2022, Art. no. 103129. [Online]. Available: https://www.sciencedirect.com/science/article/pii/S1361920921004247
[8] N. C. Onat, M. Kucukvar, N. N. M. Aboushaqrah, and R. Jabbar, "How sustainable is electric mobility? A comprehensive sustainability assessment approach for the case of Qatar," *Appl. Energy*, vol. 250, pp. 461–477, Sep. 2019. [Online]. Available: https://www.sciencedirect.com/science/article/pii/S0306261919309304
[9] B. Sen, M. Kucukvar, N. C. Onat, and O. Tatari, "Life cycle sustainability assessment of autonomous heavy-duty trucks," *J. Ind. Ecol.*, vol. 24, no. 1, pp. 149–164, 2020. [Online]. Available: https://onlinelibrary.wiley.com/doi/abs/10.1111/jiec.12964
[10] Y. Wang, G. Zhou, T. Li, and X. Wei, "Comprehensive evaluation of the sustainable development of battery electric vehicles in China," *Sustainability*, vol. 11, no. 20, Oct. 2019, Art. no. 5635. [Online]. Available: https://www.mdpi.com/2071-1050/11/20/5635
[11] E. Ekener, "Developing Life Cycle Sustainability Assessment methodology by applying values-based sustainability weighting - Tested on biomass based and fossil transportation fuels," *J. Cleaner Prod.*, vol. 181, pp. 337–351, 2018.
[12] N. Hoque, W. Biswas, I. Mazhar, and I. Howard, "Life cycle sustainability assessment of alternative energy sources for the western australian transport sector," *Sustainability*, vol. 12, no. 14, Jan. 2020, Art. no. 5565. [Online]. Available: https://www.mdpi.com/2071-1050/12/14/5565
[13] N. Mahbub, A. O. Oyedun, H. Zhang, A. Kumar, and W.-R. Poganietz, "A life cycle sustainability assessment (LCSA) of oxymethylene ether as a diesel additive produced from forest biomass," *Int. J. Life Cycle Assessment*, vol. 24, no. 5, pp. 881–899, May 2019, doi: 10.1007/s11367-018-1529-6.
[14] P. Masilela and A. Pradhan, "A life cycle sustainability assessment of biomethane versus biohydrogen–For application in electricity or vehicle fuel? Case studies for African context," *J. Cleaner Prod.*, vol. 328, Dec. 2021, Art. no. 129567. [Online]. Available: https://www.sciencedirect.com/science/article/pii/S0959652621037458
[15] J. Ren, A. Manzardo, A. Mazzi, F. Zuliani, and A. Scipioni, "Prioritization of bioethanol production pathways in China based on life cycle sustainability assessment and multicriteria decision-making," *Int. J. Life Cycle Assessment*, vol. 20, no. 6, pp. 842–853, Jun. 2015, doi: 10.1007/s11367-015-0877-8.
[16] E. M. Schau, M. Traverso, and M. Finkbeiner, "Life cycle approach to sustainability assessment: A case study of remanufactured alternators," *J. Remanufacturing*, vol. 2, no. 1, pp. 1–14, Dec. 2012, doi: 10.1186/2210-4690-2-5.
[17] L. Zanchi, M. Delogu, C. A. Dattilo, A. Zamagni, and F. Del Pero, "Integrating life cycle sustainability assessment results using Fuzzy-TOPSIS in automotive lightweighting," *SAE Int. J. Mater. Manuf.*, vol. 14, no. 3, pp. 317–341, 2021. [Online]. Available: https://www.jstor.org/stable/27095864
[18] J.-L. Popien, C. Thies, A. Barke, and T. S. Spengler, "Comparative sustainability assessment of lithiumion, lithiumsulfur, and allsolidstate traction batteries," 2023, doi: 10.1007/s11367-023-02134-4.
[19] B. Reuter, "Assessment of sustainability issues for the selection of materials and technologies during product design: A case study of lithium-ion batteries for electric vehicles," *Int. J. Interactive Des. Manuf.*, vol. 10, no. 3, pp. 217–227, Aug. 2016, doi: 10.1007/s12008-016-0329-0.
[20] S. Pinto-Bautista, M. Baumann, and M. Weil, "Prospective life cycle assessment of an electric vehicle equipped with a model magnesium battery," *Energy, Sustainability Soc.*, vol. 14, no. 1, Jul. 2024, Art. no. 44. [Online]. Available: https://energsustainsoc.biomedcentral.com/articles/10.1186/s13705-024-00475-y
[21] A. Bouter, E. Hache, C. Ternel, and S. Beauchet, "Comparative environmental life cycle assessment of several powertrain types for cars and buses in France for two driving cycles: "Worldwide harmonized light vehicle test procedure" cycle and urban cycle," *Int. J. Life Cycle Assessment*, vol. 25, no. 8, pp. 1545–1565, Aug. 2020, doi: 10.1007/s11367-020-01756-2.
[22] J. Van Mierlo, M. Messagie, and S. Rangaraju, "Comparative environmental assessment of alternative fueled vehicles using a life cycle assessment," *Transp. Res. Procedia*, vol. 25, pp. 3435–3445, Jan. 2017. [Online]. Available: https://www.sciencedirect.com/science/article/pii/S2352146517305513
[23] A. Desreveaux, E. Hittinger, A. Bouscayrol, E. Castex, and G. M. Sirbu, "Techno-economic comparison of total cost of ownership of electric and diesel vehicles," *IEEE Access*, vol. 8, pp. 195752–195762, 2020. [Online]. Available: https://ieeexplore.ieee.org/document/9239269

[24] M. Woody, S. A. Adderly, R. Bohra, and G. A. Keoleian, “Electric and gasoline vehicle total cost of ownership across US cities,” *J. Ind. Ecol.*, vol. 28, no. 2, pp. 194–215, 2024. [Online]. Available: https://onlinelibrary.wiley.com/doi/abs/10.1111/jiec.13463

[25] A. E. P. Abas, J. Yong, T. M. I. Mahlia, and M. A. Hannan, “Techno-economic analysis and environmental impact of electric vehicle,” *IEEE Access*, vol. 7, pp. 98565–98578, 2019. [Online]. Available: https://ieeexplore.ieee.org/document/8765562

[26] N. Di Vittorio, A. Accardo, E. Spessa, L. Viscido, and E. Tam, “LCA and LCC of a li-ion battery pack for automotive application,” *SAE Int. J. Adv. Curr. Practices Mobility*, vol. 6, pp. 1995–2004, Aug. 2023. [Online]. Available: https://www.sae.org/content/2023-24-0170

[27] A. Sanchez, G. Benveniste, V. J. Ferreira, I. Bulfaro, L. Igualada, and C. Corchero, “Methodology for social life cycle impact assessment enhanced with gender aspects applied to electric vehicle Liion batteries,” *Int. J. Life Cycle Assess*, 2024, doi: 10.1007/s11367-024-02329-3.

[28] M. Koese, C. F. Blanco, V. B. Vert, and M. G. Vijver, “A social life cycle assessment of vanadium redox flow and lithium-ion batteries for energy storage,” *J. Ind. Ecol.*, vol. 27, no. 1, pp. 223–237, Feb. 2023, doi: 10.1111/jiec.13347.

[29] J. Das, A. Kleiman, A. U. Rehman, R. Verma, and M. H. Young, “The cobalt supply chain and environmental life cycle impacts of lithium-ion battery energy storage systems,” *Sustainability*, vol. 16, no. 5, Feb. 2024, Art. no. 1910. [Online]. Available: https://www.mdpi.com/2071-1050/16/5/1910

[30] S. Bawankar et al., “Environmental impact assessment of lithium ion battery employing cradle to grave,” *Sustain. Energy Technol. Assessments*, vol. 60, Dec. 2023, Art. no. 103530. [Online]. Available: https://linkinghub.elsevier.com/retrieve/pii/S2213138823005234

[31] Q. Chen et al., “Investigating carbon footprint and carbon reduction potential using a cradle-to-cradle LCA approach on lithium-ion batteries for electric vehicles in China,” *J. Cleaner Prod.*, vol. 369, Oct. 2022, Art. no. 133342. [Online]. Available: https://linkinghub.elsevier.com/retrieve/pii/S0959652622029286

[32] P. Engels et al., “Life cycle assessment of natural graphite production for lithium-ion battery anodes based on industrial primary data,” *J. Cleaner Prod.*, vol. 336, Feb. 2022, Art. no. 130474. [Online]. Available: https://linkinghub.elsevier.com/retrieve/pii/S0959652622001172

[33] Pré Sustainability B. V., “SimaPro,” 2024. [Online]. Available: https://simapro.com/

[34] EVDB, “All electric vehicles in Europe - EV database,” 2025. [Online]. Available: https://ev-database.org/

[35] L. Zampori and R. Pant, “Suggestions for updating the product environmental footprint (PEF) method,” 2019. [Online]. Available: https://publications.jrc.ec.europa.eu/repository/handle/JRC115959

[36] Enerdata, “Change in distance travelled by car,” 2022. [Online]. Available: https://www.odyssee-mure.eu/publications/efficiency-by-sector/transport/distance-travelled-by-car.html

[37] The European Parliament and the Council of the European Union, “Regulation (EU) 2023/1542 of the European Parliament and of the Council of 12 Jul. 2023 concerning batteries and waste batteries, amending Directive 2008/98/EC and Regulation (EU) 2019/1020 and repealing Directive 2006/66/EC (Text with EEA relevance),” Jul. 2023. [Online]. Available: https://eur-lex.europa.eu/eli/reg/2023/1542/oj/eng

[38] M. Gutsch and J. Leker, “Costs, carbon footprint, and environmental impacts of lithium-ion batteries–From cathode active material synthesis to cell manufacturing and recycling,” *Appl. Energy*, vol. 353, Jan. 2024, Art. no. 122132. [Online]. Available: https://www.sciencedirect.com/science/article/pii/S0306261923014964

[39] M. Mohr, J. F. Peters, M. Baumann, and M. Weil, “Toward a cell-chemistry specific life cycle assessment of lithium-ion battery recycling processes,” *J. Ind. Ecol.*, vol. 24, no. 6, pp. 1310–1322, Dec. 2020, doi: 10.1111/jiec.13021.

[40] IEA, “Global critical minerals outlook 2024,” May 2024. [Online]. Available: https://www.iea.org/reports/global-critical-minerals-outlook-2024

[41] Statista, “Global manganese production by country,” 2023. [Online]. Available: https://www.statista.com/

[42] TopoNavi, “Distanza tra le città,” 2024. [Online]. Available: https://va.toponavi.com/

[43] VesselFinder, “Ship & container tracking,” 2024. [Online]. Available: https://www.vesselfinder.com/

[44] IEA, “Global supply chains of EV batteries,” Jul. 2022. [Online]. Available: https://www.iea.org/reports/global-supply-chains-of-ev-batteries

[45] The Cobalt Global South Project, “Cobalt,” 2023. [Online]. Available: https://chinaglobalsouth.com/category/resources/cobalt/

[46] The Cobalt Global South Project, “Interactive map of the copper and cobalt sector in the DRC,” 2024. [Online]. Available: https://chinaglobalsouth.com/cobaltmap/en/

[47] S. R. Rahman, “Top 5 ports in China 2024,” Dec. 2024. [Online]. Available: https://www.porttechnology.org/news/top-5-ports-in-china-2024/

[48] Tycorun, “Capacity distribution of Chinese lithium battery cathode material companies in 2022,” Jun. 2022. [Online]. Available: https://www.tycorun.com/blogs/news/capacity-distribution-of-chinese-lithium-battery-cathode-material-companies-in-2022

[49] S. Moores, “The global battery arms race: Lithium-ion battery gigafactories and their supply chain,” in *Oxford Energy Forum*, 2021, pp. 26–30.

[50] Stellantis, “Stellantis electrification transition in full swing with first ACC battery gigafactory inaugurated in France,” 2023. [Online]. Available: https://www.stellantis.com/en/news/press-releases/2023/may/stellantis-electrification-transition-in-full-swing-with-first-acc-battery-gigafactory-inaugurated-in-france

[51] Fucheng, “Estrazione mineraria in Indonesia: Guida completa all’estrazione mineraria,” Dec. 2024. [Online]. Available: https://fuchenglhd.com/it/blog/mining-in-indonesia/

[52] South32, “South africa manganese,” 2024. [Online]. Available: https://www.south32.net/what-we-do/our-locations/southern-africa/south-africa-manganese

[53] ILO, “Safety and health in mines convention,” 1995. [Online]. Available: https://www.ilo.org/resource/other/c176-safety-and-health-mines-convention-1995

[54] MDO, “Mt. Holland Mine,” 2024. [Online]. Available: https://miningdataonline.com/property/3417/Mt.-Holland-Mine.aspx

[55] Statista, “Production volume of graphite in China from 2010 to 2023,” 2024. [Online]. Available: https://www.statista.com/statistics/1267633/graphite-production-in-china/

[56] SMM, “Shanghai metals market,” 2024. [Online]. Available: https://www.metal.com/

[57] Trading Economics, “Historical prices,” 2024. [Online]. Available: https://tradingeconomics.com/

[58] World Bank Group, “Commodity markets,” 2024. [Online]. Available: https://www.worldbank.org/en/research/commodity-markets

[59] Business Analytiq, “Natural graphite price index,” 2024. [Online]. Available: https://businessanalytiq.com/

[60] GlobalPetrolPrices, “Energy prices around the world,” 2024. [Online]. Available: https://www.globalpetrolprices.com/

[61] Statista, “The statistics portal,” 2024. [Online]. Available: https://www.statista.com/

[62] S. Bunker, “World bunker prices,” 2024. [Online]. Available: https://shipandbunker.com/prices

[63] *Environmental Management Life Cycle Assessment. Principles and Framework*, UNI EN ISO Standard 14040:2021, 2021. [Online]. Available: https://store.uni.com/uni-en-iso-14040-2021

[64] *Environmental Management. Life Cycle Assessment. Requirements and Guidelines*, UNI EN ISO Standard 14044:2021, 2021. [Online]. Available: https://store.uni.com/uni-en-iso-14044-2021

[65] Ecoinvent, “Ecoinvent database v3.8,” 2021. [Online]. Available: www.ecoinvent.org

[66] *Environmental Management — Principles and Framework for Social Life Cycle Assessment*, ISO Standard 14075:2024, 2024. [Online]. Available: https://www.iso.org/standard/61118.html

[67] ILOSTAT, “Statistics on safety and health at work,” 2024. [Online]. Available: https://ilostat.ilo.org/topics/safety-and-health-at-work/

[68] W. Free, “Global slavery index,” 2021. [Online]. Available: https://www.walkfree.org/global-slavery-index/

[69] Transparency International, “Corruption perceptions index,” 2023. [Online]. Available: https://www.transparency.org/en/cpi/2023

[70] World Population Review, “Environmental performance index by country,” 2024. [Online]. Available: https://worldpopulationreview.com/country-rankings/environmental-performance-index-by-country

[71] UNCTAD, “UN trade and development,” 1964. [Online]. Available: https://unctad.org/

ANTONELLA ACCARDO received the M.S. degree in mechanical engineering and the Ph.D. degree from the Polytechnic of Turin, Italy, in 2019 and 2025, respectively. She was a Research Fellow with the Department of Energy for two years. She is currently with the Department of Energy. She is also a Member of the Interdepartmental Center CARS, Polytechnic of Turin. Her research interests include the sustainability of the transport sector, and application of life cycle assessment and life cycle costing to light- and heavy-duty commercial vehicles and their main components, such as batteries, electric motors, hydrogen and internal combustion engines, and fuel cells.

GAIA GENTILUCCI received the M.S. degree in environmental engineering in 2020 from the Polytechnic of Turin, Italy, where she is currently working toward the Ph.D. degree with the Department of Energy. She was with a consultancy company on sustainability topics for three years. She is also a Member of the Interdepartmental Center CARS, Polytechnic of Turin. Her research interests include the sustainability of the transport sector, and application of life cycle assessment, life cycle costing and social life cycle assessment to light- and heavy-duty commercial vehicles.

LUCA PONTONE was born in Italy, in 2000. He received the M.S. degree in environmental, cultural, and territorial economics from the University of Turin, Turin, Italy, in 2025. He has also studied economics and statistics applied to the environment, sustainability, and energy, with a focus on sustainability reports, as well as insurance and statistical aspects. He has a background in sustainability, energy, and business analysis.

EZIO SPESSA received the M.S. degree in mechanical engineering and the Ph.D. degree in energetics from the Polytechnic of Turin, Turin, Italy, in 1991 and 1995, respectively. He is currently a Full Professor of fluid machinery with the Dipartimento Energia “Galileo Ferraris,” Polytechnic of Turin, and also a Member with Interdepartmental Center CARS, Polytechnic of Turin. His research interests include innovative vehicle technologies and powertrain solutions for the zero-tailpipe emission mobility of people and goods, including LCA approaches and circular economy aspects for sustainable and innovative road mobility solutions.