

## Article

# The Cobalt Supply Chain and Environmental Life Cycle Impacts of Lithium-Ion Battery Energy Storage Systems

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**Abstract:** Lithium-ion batteries (LIBs) deployed in battery energy storage systems (BESS) can reduce the carbon intensity of the electricity-generating sector and improve environmental sustainability. The aim of this study is to use life cycle assessment (LCA) modeling, using data from peer-reviewed literature and public and private sources, to quantify environmental impacts along the supply chain for cobalt, a crucial component in many types of LIBs. The study seeks to understand where in the life cycle stage the environmental impacts are highest, thus highlighting actions that can be taken to improve sustainability of the LIB supply chain. The system boundary for this LCA is cradle-to-gate. Impact assessment follows ReCiPe Midpoint (H) 2016. We assume a 30-year modeling period, with augmentation occurring at the end of the 3rd, 7th, and 14th years of operations, before a complete replacement in the 21st year. Three refinery locations (China, Canada, and Finland), a range of ore grades, and five battery chemistries (NMC111, NMC532, NMC622, NMC811, and NCA) are used in scenarios to better estimate their effect on the life cycle impacts. Insights from the study are that impacts along nearly all pathways increase according to an inverse power-law relationship with ore grade; refining outside of China can reduce global warming potential (GWP) by over 12%; and GWP impacts for cobalt used in NCA and other NMC battery chemistries are 63% and 45–74% lower than in NMC111, respectively. When analyzed on a single-score basis, marine and freshwater ecotoxicity are prominent. For an ore grade of 0.3%, the GWP values for the Canada route decrease at a rate of 58% to 65%, and those for Finland route decrease by 71% to 76% from the base case. Statistical analysis shows that cobalt content in the battery is the highest predictor ( $R^2 = 0.988$ ), followed by the ore grade ( $R^2 = 0.966$ ) and refining location ( $R^2 = 0.766$ ), when assessed for correlation individually. The results presented here point to areas where environmental burdens of LIBs can be reduced, and thus they are helpful to policy and investment decision makers.

**Keywords:** life cycle assessment; cobalt; supply chain; lithium-ion batteries; environmental sustainability



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## 1. Introduction

Cobalt is a key ingredient in lithium-ion batteries (LIBs). Demand for LIBs is expected to increase by 15 times by 2030 [1,2] due to increased wind and solar generation paired with battery energy storage systems (BESS). By 2025, the International Energy Agency (IEA) [3] predicts that a rise in LIB demand, to meet the goals outlined in the Paris Climate Accords, could increase the need for cobalt by 2500% (Supplementary Materials Figure S1). LIBs used in electric vehicles (EVs) represent the main catalyst for the increased demand of cobalt, rising with the increasing shift to low-carbon fuel technologies. Most of the cobalt supply chain is controlled by a few countries (Table S1). For example, in 2020, the Democratic Republic of the Congo (DRC) accounted for 69% of global mine production, 10 times greater than Russia, the next largest producer [4,5]. According to the United

States Geological Survey [6], the DRC has the world's largest reserves of cobalt ore, so its dominance in the production of mined cobalt will likely continue in the foreseeable future.

Like the market for mined cobalt, refined cobalt products are centralized in a single country, with China accounting for 67% of the world's refined battery-grade cobalt sulfate ( $\text{CoSO}_4$ ) capacity in 2020 [7]. There is little possibility for substitution in the near term because, other than China, only Finland has significant refining capacity for cobalt materials needed for LIBs, accounting for only 10% of total supply in 2020 [7]. The United States recently has taken steps to increase domestic cobalt supplies with the announcement in February 2022 of a USD 3 billion investment toward increasing domestic supplies of refined battery metals, including cobalt [2,8]. Data and insights into the environmental impacts of the cobalt supply chain are necessary for policy makers as they consider choices in planning for at-scale energy systems by the middle of the 21st century.

Previous research has demonstrated human health impacts, water pollution, acid mine drainage, greenhouse gas (GHG) pollution, and agricultural contamination associated with cobalt mining in the DRC [9–14]. Others have found that cobalt-bearing LIB chemistries can impart greater impacts on the environment than cobalt-free chemistries [15]. As cobalt demand increases so too will the need to understand the environmental impacts more fully, so they can be managed effectively and potentially mitigated or avoided altogether.

Life cycle assessment (LCA) is an internationally standardized, analytical method used in evaluating the full scope of environmental impacts across an entire product life [16]. LCA studies of systems or processes can identify prospects of enhancements and/or highlight appropriate substitutions [17–21]. Recently, LCA has grown in popularity for comparing the direct and indirect environmental burdens of energy systems, especially for renewable technologies such as solar panels, wind turbines, battery energy storage systems (BESS), or LIBs [22]. LIBs consist of many significant materials, including metals. LCA studies of metals from mining to end of life have been reported in the literature. Arshad et al. (2022) [23] presented a critical review of noteworthy LCA studies on LIBs, some of which focus on metals and mainly on the extraction process. There are published LCA studies on different variants of LIBs (Table S2) [24–36]. Some relevant studies of metals essential to an LIB are cited in Table 1.

**Table 1.** LCA studies of metals used in LIBs.

Reference	Metal under Study	Life Cycle Phases Included in System Boundary	Scope of the Analysis
Schenker et al., 2022 [37]	Battery-grade $\text{Li}_2\text{CO}_3$	Cradle to gate, recycling	Life cycle environmental assessment
Farjana et al., 2019 (a) [38]	Cobalt	Extraction	Life cycle environmental assessment
Mistry et al., 2015 [39]	Nickel	Cradle to gate	Primary energy demand, Global warming potential
Schmidt et al., 2016 [15]	Nickel and cobalt	Cradle to gate	Life cycle environmental assessment
Engels et al., 2022 [40]	Natural graphite	Cradle to gate	Global warming potential
Farjana et al., 2019 (b) [41]	Aluminum	Cradle to gate	Life cycle environmental assessment

Even given the results of these studies, intercomparison is often difficult because of lack of transparency in data sources in LCAs, or lack of data themselves, including data on cobalt [42]. For example, in a frequently cited LCA on LIBs, Majeau-Bettez et al. (2011) [43] used data for the production of metallic cobalt, which is not used in LIBs, due to a lack of data availability for the refining of  $\text{CoSO}_4$ . Their study did not disaggregate the results across the cobalt supply chain processes, so differentiating impacts between specific processes or identifying the sources contributing to the impacts is not possible. Other studies have since relied on the results from Majeau-Bettez et al. (2011) [43] when conducting additional LCAs [44,45], indicating that the results of these LCAs may not reflect the impacts of the real-world cobalt supply chain. Dai et al. (2019) [35] do include

site-specific data from their earlier study of the cobalt supply chain [46], but the results are not disaggregated at a process level. Kelly et al. (2020) [47] analyzed  $\text{CoSO}_4$  refining in five countries (the United States, China, Japan, South Korea, and Europe) and showed that changing the refinery location will affect the environmental impacts of the cobalt supply chain. However, the results also are not disaggregated. Crenna et al. (2021) [48] noted that allocation method (i.e., how the impacts are apportioned when a process has multiple products) in LCA can vary the results attributed to cobalt. Because cobalt was worth five times more than copper at the time of the study, the impacts attributed to cobalt, when allocated based on economic value, were much higher than when they were allocated based on mass. Crenna et al. (2021) [48] also emphasized the value of disaggregating the supply chain because, for example, varying electricity mixes used for different industrial processes can change the results compared to an aggregated analysis.

LCAs specific to the cobalt supply chain often only include data from a single production route, limiting identification of how the impacts vary as new refineries come online [46,49–51]. Like other types of LCAs, studies pertaining to cobalt often rely on background databases as data sources, especially the Ecoinvent product [52], providing results that may not be representative of products and technologies used today [38,53]. Because data on cobalt refining in Ecoinvent (v 2.2) used for these studies were an approximation based on data gathered from nickel refining decades prior, Arvidsson et al. (2020) [54] have questioned the accuracy of the results. Arvidsson et al. (2022) [55] subsequently presented the health impacts of an LIB containing cobalt using an aggregated LCA approach. A detailed, cradle-to-gate LCA of  $\text{CoSO}_4$  production from a nickel–copper cobalt mine in China has been reported by Zhang et al. (2021) [51].

While the studies listed above provide important information on environmental impacts from LIBs containing cobalt, they tend to aggregate the results across the supply chain; attributing impacts to specific supply chain processes is not possible. To our knowledge, no studies have reported the environmental impacts of each process in the cobalt supply chain that begins in the DRC, the largest producer of cobalt in the world, only a total for the entire supply chain. Moreover, they do not consider how changes in ore grade will manifest in environmental impacts.

This research paper presents a comprehensive disintegrated life cycle assessment of the environmental impacts of cobalt production routes for LIBs, including a range of ore grades. We assume that cobalt is sourced only from copper–cobalt deposits in the DRC, although we assess different production routes. The goal of this research is to address the following questions:

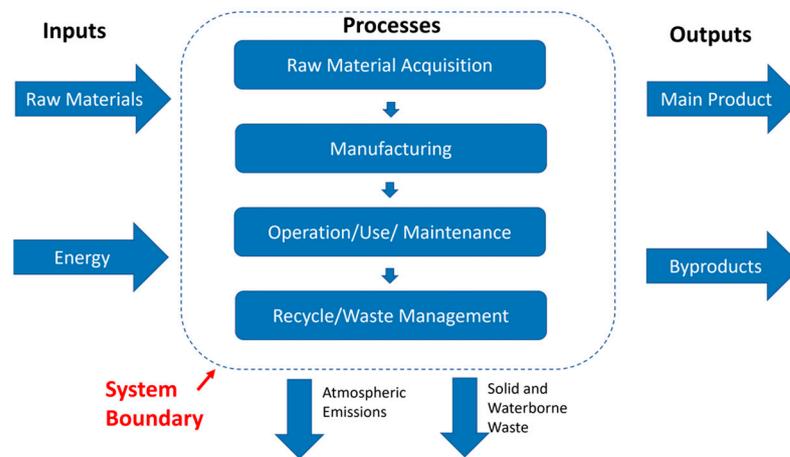
- Q1. What physical pathways and processes does cobalt undergo from extraction until use in an LIB?
- Q2. What are the environmental (greenhouse gases (GHG), air, land, and water) impacts of each process and life-cycle phase?
- Q3. How will changes in ore quality impact the environmental sustainability of extraction?
- Q4. How will changes in refinery location impact the environmental sustainability of the cobalt supply chain for LIBs?
- Q5. How do the impacts of nickel manganese cobalt (NMC) and nickel cobalt aluminum (NCA) compare?

## 2. Materials and Methods

### 2.1. Life Cycle Assessment (LCA)

According to International Organization for Standardization (ISO) 14040—principles and framework [56] and ISO 14044—requirements and guidelines [57], the LCA approach consists of four main phases: (1) goal and scope definition, (2) life cycle inventory, (3) life cycle impact assessment, and (4) interpretation and inferences. These phases are described below. LCA model calculations used in this study were carried out using an OpenLCA software package (version 1.10.3, Green Delta, Berlin, Germany).

A pictorial representation of the material and energy flows of a general process is shown in Figure 1.



**Figure 1.** Definition of a unit process.

The cradle-to-gate burden vector for life cycle inventory  $\{B\}$  for unit production can be expressed as in Equation (1):

$$\{B\}ctg = \{B\}mp + \{B\}mnf + \{B\}tr + \{B\}oper \quad (1)$$

where  $ctg$  stands for cradle-to-gate,  $mp$  is the material production stage,  $mnf$  is the manufacturing stage,  $tr$  is transportation, and  $oper$  is operation. The material production energy of any process or product is expressed as  $PE_j$  which is the sum of the input energy components (materials, fuel, and electricity) of the  $j^{\text{th}}$  component. This can be mathematically expressed as in Equation (2):

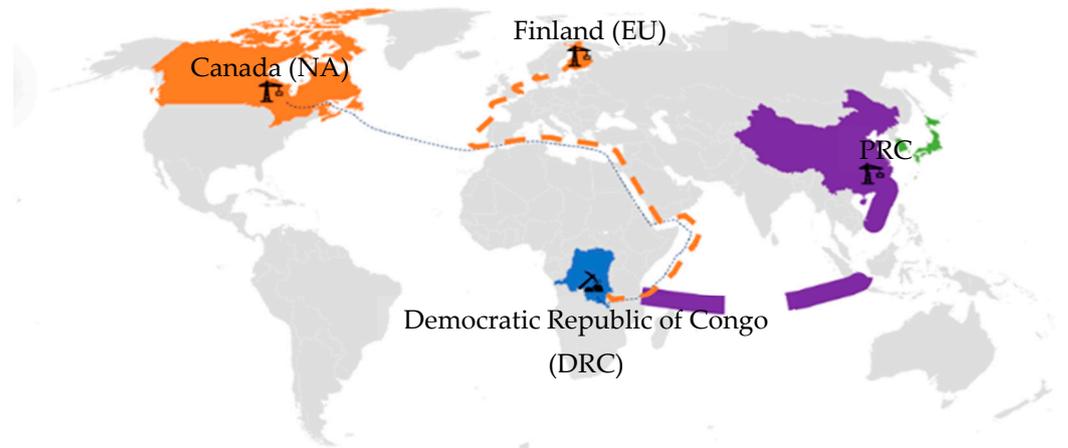
$$E_{mp} = \sum PE_j \times m_j / C_j \quad (2)$$

where  $E_{mp}$  is the material production energy;  $j$  is the number of components in the process or system;  $PE_j$  is the production energy of the  $j^{\text{th}}$  component;  $m_j$  is the number of components; and  $C_j$  is the production efficiency of putting material  $j$  into the unit in the amount  $m_j$ .

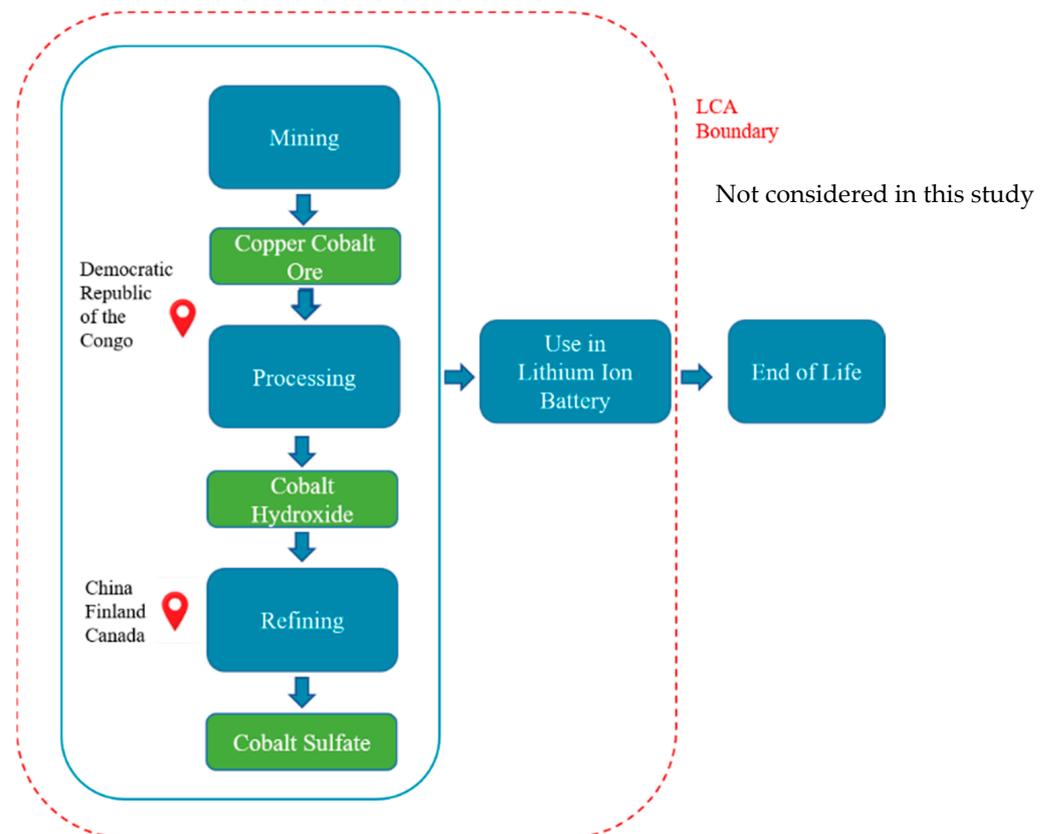
### 2.1.1. Goal and Scope Definition

The goal of this study is to analyze the cradle-to-gate environmental impact of cobalt when used as raw material for LIB production. The functional unit is 1 MWh of LIB storage capacity.

The multiple geographic regions under study (Figure 2) represent prevailing conditions or those expected in the future. In all scenarios, mining and processing are assumed to be in the DRC. Three different geographic locations—China, Canada (North America), and Finland (Europe)—are used for refining cobalt hydroxide ( $\text{Co}(\text{OH})_2$ ) into battery-grade cobalt sulfate ( $\text{CoSO}_4$ ). These locations were chosen given China and Finland's dominance in global refining capacity (67% and 10%, respectively) and the reported construction of a new refinery in Canada. The different locations on three continents provided enough spatial separation to show the significance of transportation distances. Cobalt supply chain and inventory data are largely sourced from Dai et al. (2018) [46], other literature sources, and the Ecoinvent database (version 3.8, [52]). We considered five battery technologies for this study: NMC111, NMC532, NMC622, NMC811, and NCA. The cycle life, cobalt content, and energy density of these different chemistries were gathered from literature sources (Table 2). System boundaries for this analysis are shown in Figure 3.



**Figure 2.** Map of geographic locations incorporated in this study; blue = cobalt mining and processing; orange = cobalt refining; green = cathode manufacturing; purple = refining and cathode manufacturing. Dashed lines indicate the flow of  $\text{Co(OH)}_2$  for the refinery location scenario analysis, and line thickness indicates the relative flow by weight.



**Figure 3.** System boundary for LCA of cobalt in LIBs.

To ensure that environmental impacts are properly attributed to cobalt when a process produces multiple coproducts, the material inputs and emissions are allocated based on the mass of the coproducts. For cobalt chemicals, the impacts are allocated based on the ratio of the molecular mass of cobalt to the molecular mass of the entire chemical, so that the impacts are attributed only to cobalt rather than to other components [58].

**Table 2.** Battery Specifications <sup>†</sup>.

Battery	Cathode	Formula	Anode	Cycle Life (@80% DoD)	Life (Years) ‡	Specific Energy (Wh/kg) [25]	Round Trip Efficiency (%)
NMC111	Nickel manganese cobalt (in the ratio 1:1:1)	$\text{LiNi}_{1/3}\text{Mn}_{1/3}\text{Co}_{1/3}\text{O}_2$				143	
NMC532	Nickel manganese cobalt (in the ratio 0.5:0.3:0.2)	$\text{LiNi}_{0.5}\text{Mn}_{0.3}\text{Co}_{0.2}\text{O}_2$				259.26	
NMC622	Nickel manganese cobalt (in the ratio 0.6:0.2:0.2)	$\text{LiNi}_{0.6}\text{Mn}_{0.2}\text{Co}_{0.2}\text{O}_2$	Graphite	7000–7300	20	269.17	90
NMC811	Nickel manganese cobalt (in the ratio 0.8:0.1:0.1)	$\text{LiNi}_{0.8}\text{Mn}_{0.1}\text{Co}_{0.1}\text{O}_2$				278.75	
NCA	Nickel cobalt aluminum	$\text{LiNi}_{0.8}\text{Co}_{0.15}\text{Al}_{0.05}\text{O}_2$				279.12	

†—Data from [59]; ‡—assuming 1 cycle/day.

The cradle-to-gate environmental impact of the system considered for analysis is given by Equation (3):

$$EI_{ctg} = EI_{(mining)} + EI_{(processing)} + EI_{(transportation)} + EI_{(refining)} \quad (3)$$

where *EI* stands for environmental impact and *ctg* stands for cradle-to-gate. In our study, the total environmental impact of the system within the boundary is the sum of the impacts during mining of the ore, ore processing, transportation, and ore refining to be used in the battery.

Referring to the general methodology in Section 2.1, material production in our study is the sum of all the materials that go into the mining, processing, and refining stages. Manufacturing refers to all the processes that contribute to the above stages.

### 2.1.2. Life Cycle Inventory

Individual processes and inventories used in the cobalt supply chain for LIBs are identified from different geographic locations to assess how the impacts vary among China, Canada, and Finland. Cobalt is incorporated into five types of LIBs: nickel cobalt aluminum (NCA) and four types of nickel manganese cobalt, NMC111, NMC532, NMC622, and NMC811, to show how the life cycle impacts vary across battery types.

The life cycle inventory (LCI) includes numerous processes and inputs that are fully described in Section II of the Supplementary Materials section. We note that the analyses assume an initial battery rating of 500 MW capacity and battery replenishment, due to degradation, according to a specific schedule (Table 3).

**Table 3.** Battery augmentation rate used for analysis.

Year End	Battery Capacity (% of Rating)	Augmentation (% of Rating)	Battery Installed Capacity (% of Rating)	Useable Capacity (% of Rating)
0	100	-	120.5	100
3	88.7	10	118.0	100
7	82.0	10	119.3	100
14	74.5	15	123.3	100
21	64.9	112 (Battery replacement)	122	100

### 2.1.3. Life Cycle Impact Assessment

The life cycle impact assessment (LCIA) method converts the material inputs, outputs, and emissions of the LCI into quantifiable impacts on land, air, and water resources [60]. The ReCiPe 2016 midpoint H (hierarchist) LCIA method [61], used in this study is based on a scientific consensus of the time frame and likelihood of impact mechanisms. Because impacts can persist after the 30-year useful life of the electricity generating option, the time horizon for the impacts included in this LCIA method is 100 years, which is the same standard used by the EIA (2021) [62]. Impact categories included in the analysis are given in Table 4.

**Table 4.** Impact categories considered in this study.

Impact Category	Abbreviation	Unit	Damage Pathways
Fine particulate matter formation	PMFP	kg PM <sub>2.5</sub> <sub>eq</sub>	Increased respiratory illness
Fossil resource scarcity	FFP	kg oil <sub>eq</sub>	Scarcity in fossil resources on earth
Freshwater ecotoxicity	FETP	kg 1,4-DCB	Loss of plant and aquatic life; increased risk of cancer
Freshwater eutrophication potential	FEP	kg P <sub>eq</sub>	Loss of aquatic species
Global warming potential	GWP	kg CO <sub>2</sub> <sub>eq</sub>	Increased flood risk, human disease, species decline
Human carcinogenic toxicity	HTP <sub>c</sub>	kg 1,4-DCB	Increased toxicity and human disease
Human non-carcinogenic toxicity	HTP <sub>nc</sub>	kg 1,4-DCB	Increased toxicity and human disease which are non-carcinogenic
Ionizing radiation	IRP	kBq Co-60 <sub>eq</sub>	Increased DNA damage
Land use	LOP	m <sup>2</sup> a crop <sub>eq</sub>	Increased land footprint
Marine ecotoxicity	METP	kg 1,4-DCB	Loss of plant and aquatic life; increased risk of cancer
Marine eutrophication potential	MEP	kg N <sub>eq</sub>	Loss of aquatic species
Mineral resource scarcity	SOP	kg Cu <sub>eq</sub>	Scarcity of minerals on the earth
Ozone formation, human health	OFHH	kg NO <sub>x</sub> <sub>eq</sub>	Increased threat to human health
Ozone formation, terrestrial ecosystems	OFTE	kg NO <sub>x</sub> <sub>eq</sub>	Increased threat to terrestrial ecosystems
Stratospheric ozone depletion	ODP	kg CFC-11 <sub>eq</sub>	Increased risk of disease
Terrestrial acidification potential	TAP	kg SO <sub>2</sub> <sub>eq</sub>	Loss of plant life
Terrestrial ecotoxicity	TETP	kg 1,4-DCB	Loss of plant species
Water depletion potential	WDP	m <sup>3</sup>	Loss of aquatic species; malnutrition

### 2.1.4. Experimental Design and Data Analysis

To study the influence of input parameters on the LCA impact pathways, 150 scenarios were generated that include 10 variations in ore grade, 3 refinery locations, and 5 battery types (Table 5). The base case considers an ore grade of 0.3% [63] and NMC111 battery

chemistry. The mining and processing location is in the DRC, and the refining is performed in China for the base case. The details are taken from Golder Associates, 2007 [64]. The transportation details were from [65,66].

**Table 5.** Description of scenarios used in the analysis.

Scenarios	Ore Grade	Mining + Processing Location	Battery Chemistry	Refining Location
Base Case	0.30%	DRC	NMC111	China
1–30			NMC111	China
				Canada (NA)
				Finland (EU)
31–60			NMC532	China
				Canada (NA)
				Finland (EU)
61–90	0.1–1%	DRC	NMC622	China
				Canada (NA)
				Finland (EU)
91–120			NMC811	China
				Canada (NA)
				Finland (EU)
121–150			NCA	China
				Canada (NA)
				Finland (EU)

The output from each scenario includes results for each impact category (listed in Table 4). Stepwise regression analysis is used to identify statistically relevant variables [67,68] when determining environmental burden. The method analyzes the sensitivity of the dependent variable to different independent variables. This method is performed to rule out potentially insignificant independent parameters.

$$y = y_0 + a_1x_1^{\varepsilon_1} + a_2x_2^{\varepsilon_2} + a_3x_3^{\varepsilon_3} + \dots a_nx_n^{\varepsilon_n} \quad (4)$$

where  $x_1, x_2, \dots, x_n$  are the  $n$  independent variables,  $a_1, a_2, \dots, a_n$  are their coefficients, and their powers are given by  $\varepsilon_1, \varepsilon_2, \dots, \varepsilon_n$ .  $y$  is the dependent variable, and  $y_0$  is the constant.

In this analysis, ore grade, refining location, and battery type are the independent variables, and  $GWP$  is the dependent variable (Table 6).

**Table 6.** Variables used in sensitivity analysis.

Ore Grade	Independent Variables		Cobalt Content in the Battery Type (kg/MWh)		Dependent Variable
	Fossil Fuel Generation at Refining Location (%)				
0.1–1%	China Canada Finland	75 18 29	NMC111	394	Global Warming Potential (GWP)
			NMC532	230	
			NMC622	190	
			NMC811	94	
			NCA	143	

The regression equation for the present study is given as:

$$GWP = y_0 + a_1(g)^{\epsilon_1} + a_2(loc)^{\epsilon_2} + a_3(cob)^{\epsilon_3} \quad (5)$$

where  $GWP$  is the global warming potential in kg CO<sub>2eq</sub>;  $g$  is the ore grade (percentage);  $loc$  is the percentage of fossil fuel generation at the refining location; and  $cob$  is the cobalt content in the battery (kg/MWh). Categorical variables were converted to nominal values, where refining location was expressed as the percentage of the grid mix sourced from fossil fuel generation at the location, and battery type was quantified based on the mass of cobalt required for 1 MWh of energy storage (Table 6).

### 3. Results

#### 3.1. Base Case Scenario-Analysis of Impact Factors

The base-case scenario is an NMC111 battery with refining in China and an ore grade of 0.3%. The major environmental impacts are shown in Figure 4. Of the total GWP of 8568 kg CO<sub>2eq</sub>, refining contributed the most, relative to other life cycle phases (47.3% of total), within which the contributions of sodium hydroxide (15.5%) and the emissions associated with the electricity used during the refining process (14.1%) are substantial. The burning of diesel in machinery used to mine, haul, and grind the copper–cobalt ore was the largest single source of GWP (22.64%) (Figure 4a). The use of sulfur during processing in its solid state and as sulfur dioxide gas contributed 16% and 15.6%, respectively. Sulfuric acid use during refining contributed 10.8% of the total release of 70.1 kg of SO<sub>2eq</sub> of the terrestrial acidification potential (TAP). The sodium hydroxide (8.7%) and electricity (7.3%) used during refining also contribute to acidification (Figure 4b). Impacts from the treatment and disposal of tailings during the processing phase dominated the freshwater eutrophication potential (FEP) (6.19 kg P<sub>eq</sub>) along the supply chain (73.4%). For the total marine eutrophication potential (MEP) along the supply chain (2.56 kg N<sub>eq</sub>), refining is the largest contributor (38.7%), stemming mostly from the use of sodium hydroxide (15.2%). Mining was the second largest contributor to marine eutrophication (36.6%) from the burning of diesel fuel in mining machinery. Processing accounted for 18.1% of the total impact, stemming mostly from the embodied impacts of the magnesia (11.89%) (Figure 4c). The mineral resource scarcity (SOP) of 29.18 kg Cu<sub>eq</sub> was dominated by refining processes that contributed 87.8% (Figure 4d). Mining was the largest contributor to fine particulate matter formation (PMFP), primarily from machinery exhaust and emissions from the crushing, mining, and drilling of ore, road entrainment, and wind erosion (73.6% of 90.2 kg of PM<sub>2.5eq</sub>, Figure 4e).

The treatment and disposal of tailings from the processing of Co(OH)<sub>2</sub> accounted for the vast majority of both freshwater (FETP) (Figure 4f) and marine ecotoxicity (METP) (Figure 4g) (87.8% of 1989 kg 1,4-DCB and 87.1% of 1727 kg 1,4-DCB, respectively). The terrestrial ecotoxicity (TETP) impacts are spread across the supply chain processes (Figure 4h), where the total release was calculated as 1.1 kg of 1,4-DCB. Refining was the largest contributor to TETP (61.4%) from hydrochloric acid (25.7%), sulfuric acid (17.2%), and sodium hydroxide (9.7%). TETP was influenced by processing of Co(OH)<sub>2</sub> (12.1%), stemming from MgO (6.5%) and sulfur (3.7%). Mining accounted for 7.1% of the total TETP impacts. Transportation contributed a higher percentage to terrestrial ecotoxicity than any other impact category (19.0%). These impacts originated from non-exhaust emissions, such as tire, road, and brake wear from the trucks used to transport cobalt products. Refining contributed the largest impact to ozone depletion (ODP) (68.5% of 0.0016 kg CFC-11<sub>eq</sub>), mostly from the methane tetrachloride emitted during the manufacturing of sodium hydroxide (Figure 4i). Ozone depletion from mining (20.2%) was due to the emissions from the combustion of diesel fuel used in the mining equipment. Processing was the most water-intensive step in the supply chain (63.3% of the total of 192 m<sup>3</sup>), which was mainly attributed to the loss of water from evaporation at the hydropower facilities used to generate electricity for processing plant operations (60.3%). Although

not as water-intensive as processing, refining still accounted for 35.5% of the total water depletion potential (WDP) (Figure 4j). Other major environmental impacts with detailed classification are tabulated in Table S3.

Knowing the relative proportion of environmental impacts per MWh of NMC111 storage across all life cycle stages considered in the base case (Figure S2) helps in visualizing the environmental intensity of each life cycle stage, which provides the opportunity to assess relative risk to local communities and natural bodies, and to identify processes in the supply chain that can be modified. For example, electrifying the mine equipment would significantly reduce the PMFP rate (over 66% of total inventory), thus improving conditions for workers and potentially for local communities. Likewise, improving industrial practices during refining could reduce kg 1,4-DCB release rates to freshwater and marine water bodies, which in this base case represent almost 90% of the total inventory.

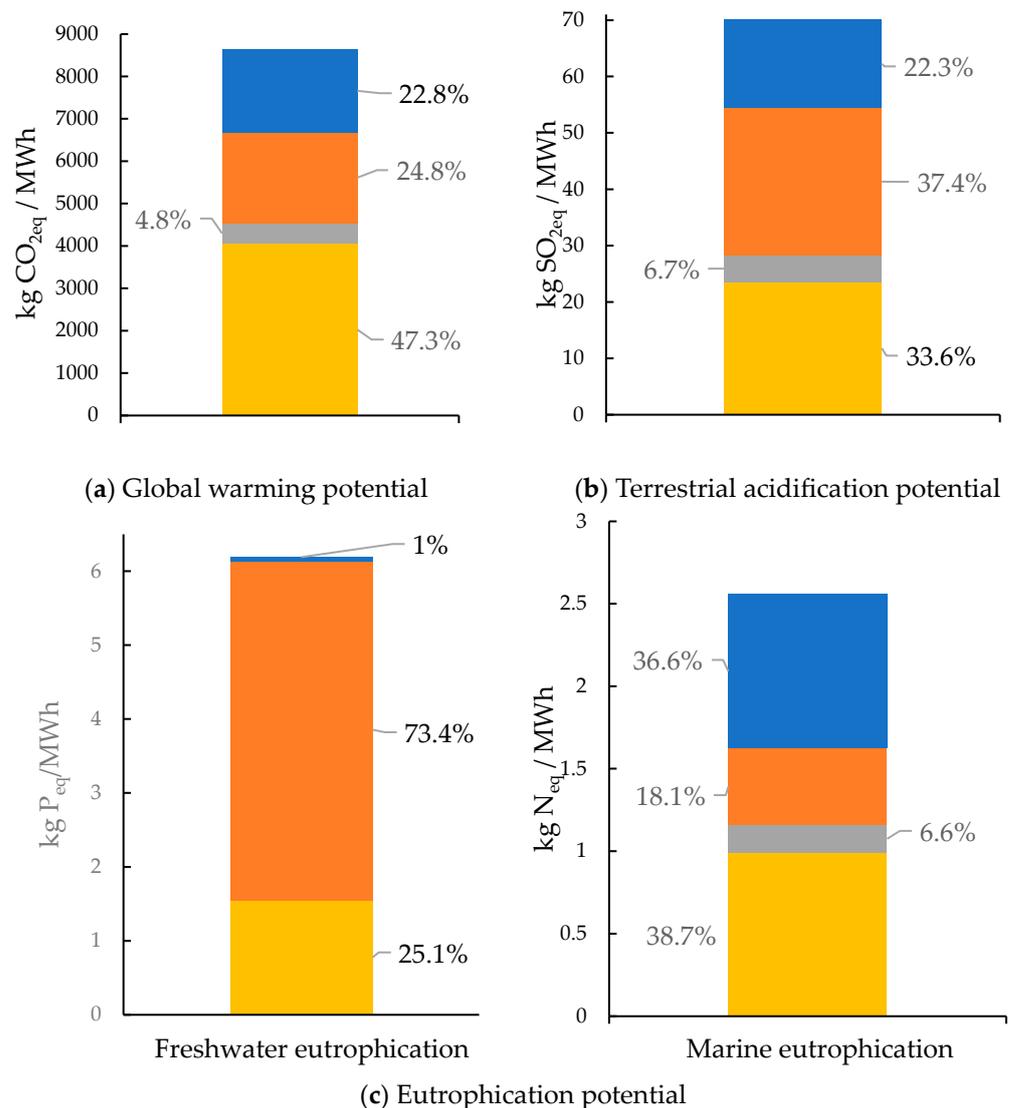
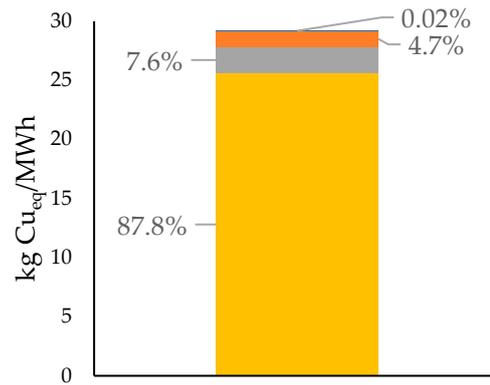
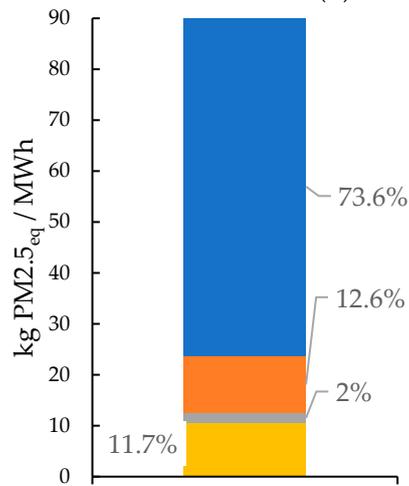


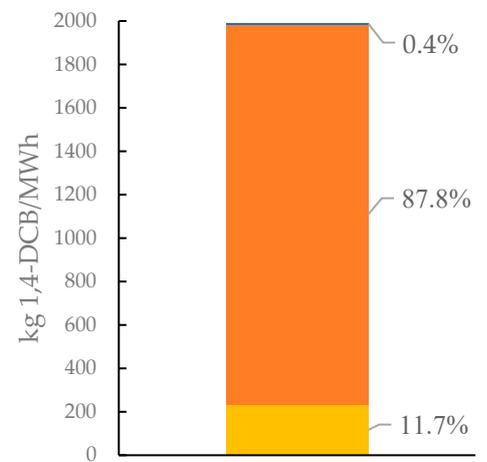
Figure 4. Cont.



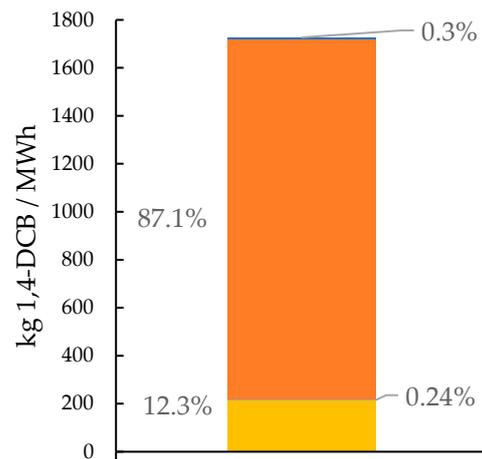
(d) Mineral resource scarcity



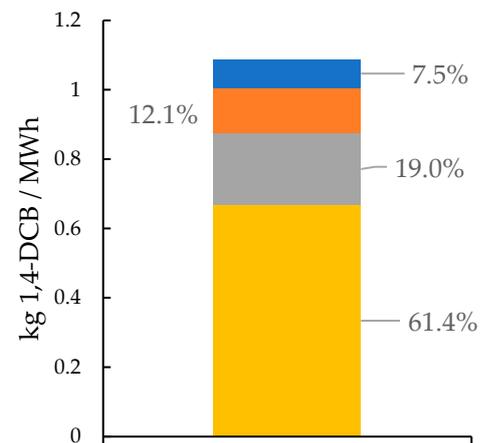
(e) Particulate matter formation



(f) Freshwater ecotoxicity

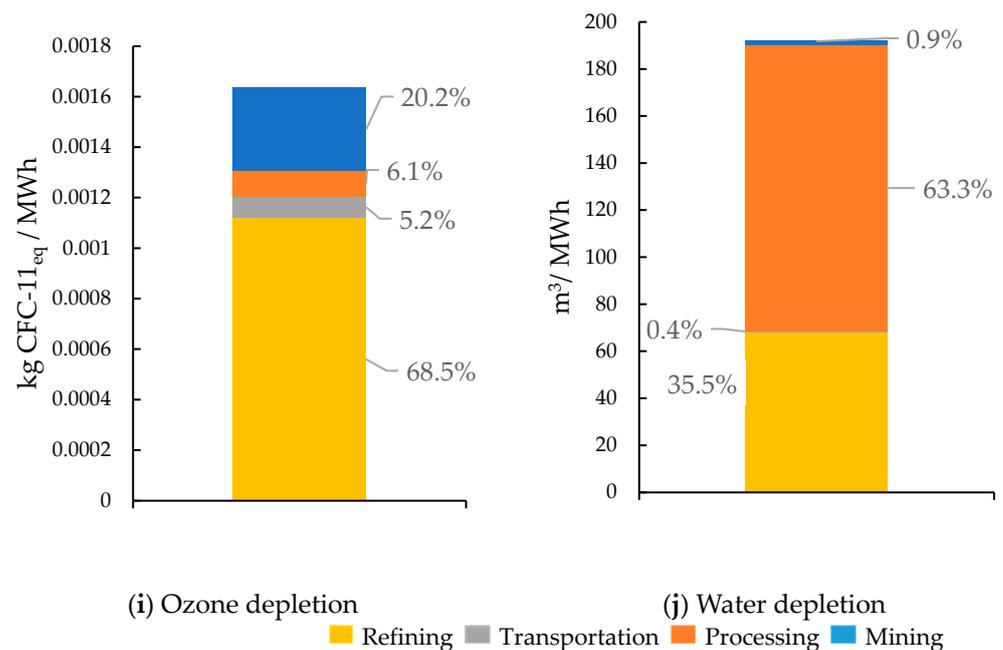


(g) Marine ecotoxicity



(h) Terrestrial ecotoxicity

Figure 4. Cont.

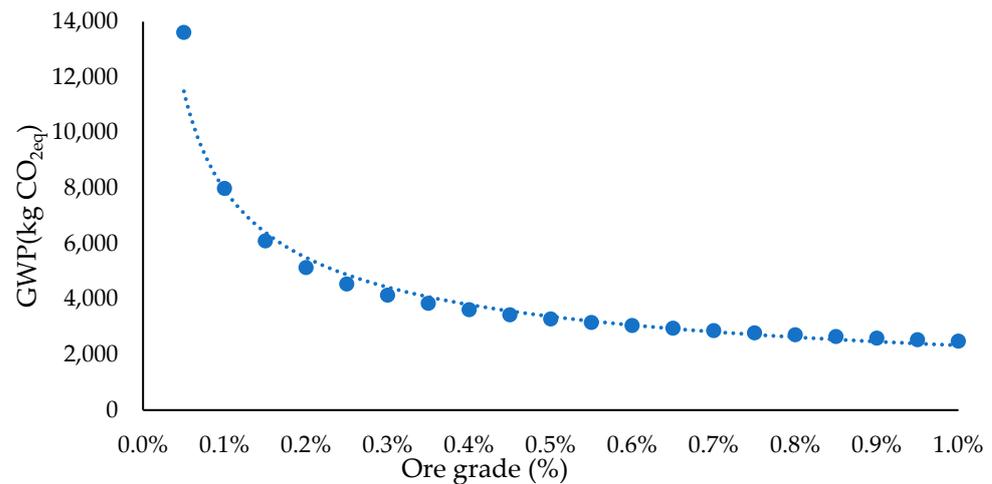


**Figure 4.** Base-case major life cycle impacts of NMC111.

### 3.2. Scenario Analysis

#### 3.2.1. Impact of Ore Grade

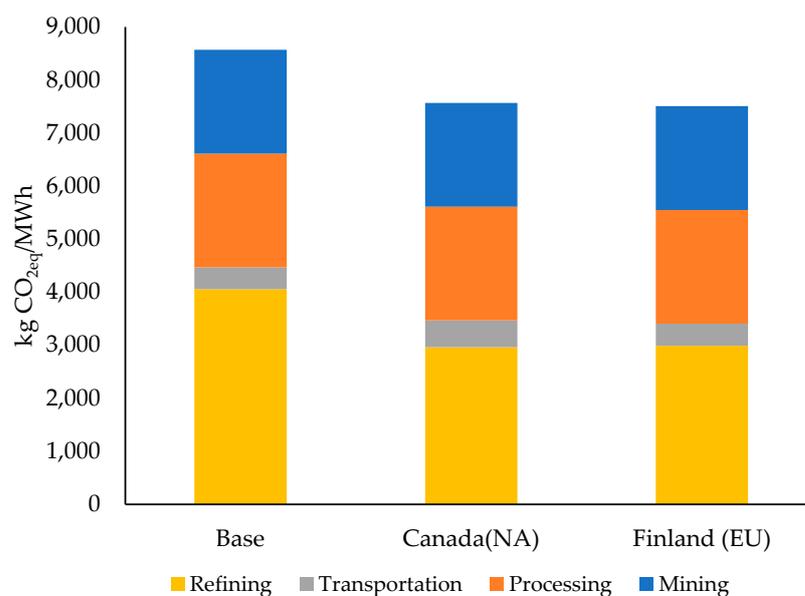
The life cycle impacts of mining and processing copper–cobalt ore to produce 1 ton of  $\text{Co}(\text{OH})_2$  were analyzed across a range of ore grades from 0.05% to 1% for NMC111. While processing lower-grade ore may become economically infeasible, the low-grade scenario was considered for analytical purposes. The results show that as ore grade decreased, the GWP of mining and processing increased according to a power-law relationship, with the GWP increasing substantially for cobalt ore grades of less than 0.3% (Figure 5). With a cobalt grade of 0.05%, the GWP of mining and processing the same amount of  $\text{Co}(\text{OH})_2$ , using base-case assumptions except for ore grade, resulted in a 230% increase in GWP compared to the base-case value (0.3%) (13,606 kg  $\text{CO}_{2\text{eq}}$  versus 4139 kg  $\text{CO}_{2\text{eq}}$ ). In the high-ore-grade scenario of 1%, or more than 3 times the base case, the GWP was reduced by 40% compared to the base case (2482 kg  $\text{CO}_{2\text{eq}}$ ), illustrating how increasing the ore grade led to reduced mining and processing needs (per kg of cobalt) and, hence, a lower total GWP. Detailed results for varying the ore grade are tabulated in Table S4. The results show that the power-law relationship with decreasing ore grade holds true for all impact categories, though it is less pronounced for water depletion (Figure S3), which is most likely because water consumption was mostly through hydropower and less through hydrometallurgy or other processing routes. This power-law relationship is consistent with observations reported by Eckelman (2010) [69], who noted that energy consumption for mining and beneficiation for nickel, scaled non-linearly with ore grade. Others have noted this as well for copper (e.g., [70,71]), especially for pyrometallurgy routes that use significant fossil-fueled thermal processes. In our case, we fitted the power-law curve to the modeled data. The impact becomes less pronounced as the ore grade increases above 0.5%.



**Figure 5.** Impact of ore grade on GWP of mining and processing of  $\text{Co}(\text{OH})_2$ . (Dashed line represents fitted power-law curve).

### 3.2.2. Refining Location

Three locations were chosen for refining cobalt hydroxide to cobalt sulfate (China, Canada, and Finland), each assuming a 0.3% ore grade. Figure 6 shows that the GWP for the base case (8568 kg  $\text{CO}_{2\text{eq}}$ ) can be reduced by 11.7% using Canadian routes and 12.4% using the Finland route (to 7568 and 7507 kg  $\text{CO}_{2\text{eq}}$ , respectively) for the NMC111 battery (base case). The largest factor contributing to reduced GWP stems from the electricity mix for the Canada and Finland scenarios. The electricity generated in Ontario, Canada, is dominated by low-carbon generation sources, with hydropower and nuclear being the two largest sources (75%) of regional electricity generation and only 18% coming from carbon-intensive sources [72]. In Finland, the electricity grid mix is composed mostly of low-carbon generation from nuclear, wind, and hydropower sources, with only 29% of electricity production from the high-carbon-footprint sources of coal, natural gas, and biofuels [72].

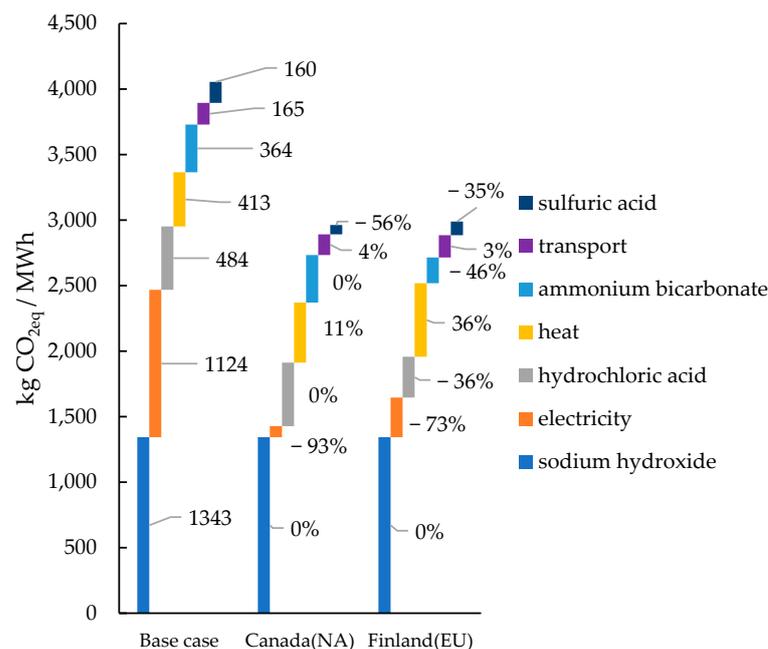


**Figure 6.** Impact of refinery location on GWP (ore grade of 0.3%).

Freshwater and marine eutrophication and ecotoxicity are almost same for the base case (China) and the Canada scenario, while the Finland scenario yields lesser impacts.

Other impacts also show similar trends. Water consumption is highest for the Canada scenario, followed by the Finland and China scenarios.

These less carbon-intensive sources in Canada and Finland contrast with the grid mix in the State Grid Corporation of China (i.e., base case), which is dominated by coal-fired electricity production (>75%) (Ecoinvent v 3.8, [40]). The GWP attributed to transportation to Canada and Finland is 4% and 3% greater, respectively, compared to the base case, while transportation only accounts for a relatively small proportion of the total GWP for the base case (4.8%), Canada (6.7%), and Finland (5.6%). The material and energy components that contribute to the GWP, with percentage changes for Canada (NA) and Finland (EU) (Figure 7), show that the electricity generation mix is the main component impacting the GWP, with a 93% decrease in footprint for Canada and a 73% decrease in Finland.

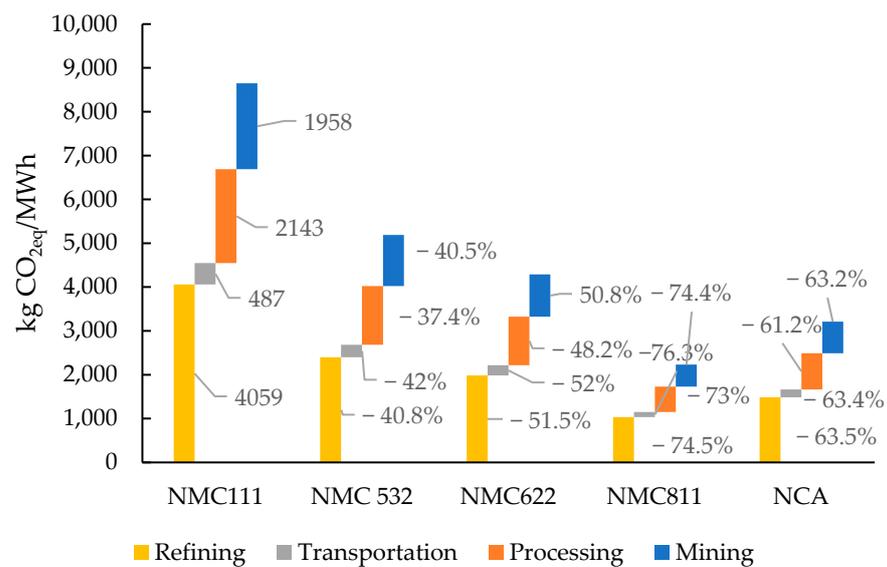


**Figure 7.** Material-wise impact of refinery location on GWP. (Numbers for base case reflect kg CO<sub>2eq</sub>, while numbers for Canada and Finland routes reflect percent change from base case).

Water depletion is the only impact category where the impacts for the base case (191.7 m<sup>3</sup>) are lower than those for Canada (223.83 m<sup>3</sup>) and Finland (201.2 m<sup>3</sup>). This difference can be attributed to the higher use of nuclear electricity generation, which is water-intensive, in the Canadian and Finland grid mixes.

### 3.2.3. Battery Chemistry

Impacts for each category scale linearly and are directly related to the cobalt intensity for NMC111, NMC532, NMC622, NMC811, and NCA LIBs (battery characteristics are given in Table 2 in Section 2.1.1). The process-wise GWP impacts of cobalt for 1 MWh of the five battery chemistries (Figure 8) show that, relative to the base case (location based in China and an ore grade of 0.3%), refining contributes the highest percentage of GWP, followed by processing and mining. Though NCA battery chemistry requires less cobalt per MWh compared to NMC111, NMC532, and NMC622 due to its higher energy density, the larger mass of cobalt required in the NCA battery itself offsets the higher energy density, still leading to a higher environmental load than for a less-cobalt-intensive NMC811 battery. Other impacts of the different processes, calculated from the ReCiPe 2016 Midpoint H analysis, also show similar trends, owing to the environmentally intensive refining procedures in China.



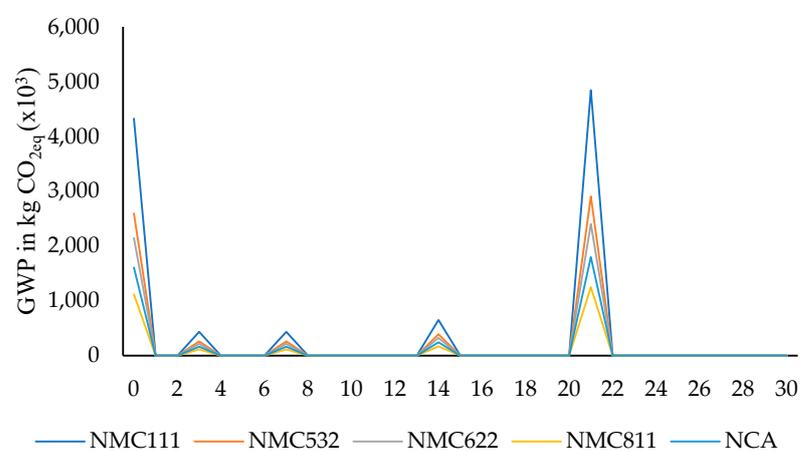
**Figure 8.** Process-wise comparison of GWP for the three battery chemistries. Numbers shown for NMC111 show actual numbers and the percentage decrease for the other battery chemistries (location—China and ore grade—0.3%).

Detailed environmental impacts, calculated using the ReCiPe 2016 Midpoint (H) method [49], are shown in Table S5.

The life cycle impacts were also normalized using World 2010 (H/H) normalization factors (Figure S4) [73]. It can be seen that marine ecotoxicity and freshwater ecotoxicity have higher impact when compared on a single-score basis.

### 3.2.4. Use

As mentioned in Section 2.1.2, the GWP footprint due to battery augmentation is included as part of this research. Using original cobalt contents for the five different batteries, the results show that the GWP is highest for NMC111 (8647 kg CO<sub>2eq</sub>/MWh), followed by NMC622 (4287 kg CO<sub>2eq</sub>/MWh), NMC532 (3630 kg CO<sub>2eq</sub>/MWh), NCA (3211 kg CO<sub>2eq</sub>/MWh), and NMC811 (2230 kg CO<sub>2eq</sub>/MWh), assuming other base-case conditions. In all cases, for a battery capacity of 500 MWh and a 30-year use phase, the original battery would need to be augmented with additional storage capacity at the end of the 3<sup>rd</sup>, 7<sup>th</sup>, and 14<sup>th</sup> years, before a complete replacement in the 21<sup>st</sup> year. The yearly GWP emissions, due to the original battery manufacturing and subsequent augmentation of the five battery chemistries with 500 MWh capacity, are compared in Figure 9.



**Figure 9.** Yearly variation of GWP during use phase for the five battery chemistries under study.

### 3.2.5. Combined Scenario Analysis

We combined the results of all 150 scenarios, in which the ore grade, refinery location, and battery chemistry are varied (see Table 5). The comparative analyses for each environmental impact listed in the ReCiPe 2016 Midpoint H [61] analysis for these scenarios are shown in Figure 10. The importance of battery chemistry (Figure 10), especially for the most-cobalt-intensive NMC111 chemistry, exceeds those of the ore grade and location, even given the noted importance of the former. We note, however, that, as battery chemistries vary away from NMC111, there is still a substantial difference in GWP emission rates by chemistry. For example, the GWP for the NCA battery chemistry and Finland route (2111 kg CO<sub>2eq</sub>) is nearly half the GWP modeled for the NMC622 and China route (3908 kg CO<sub>2eq</sub>), assuming a 0.3% ore grade.

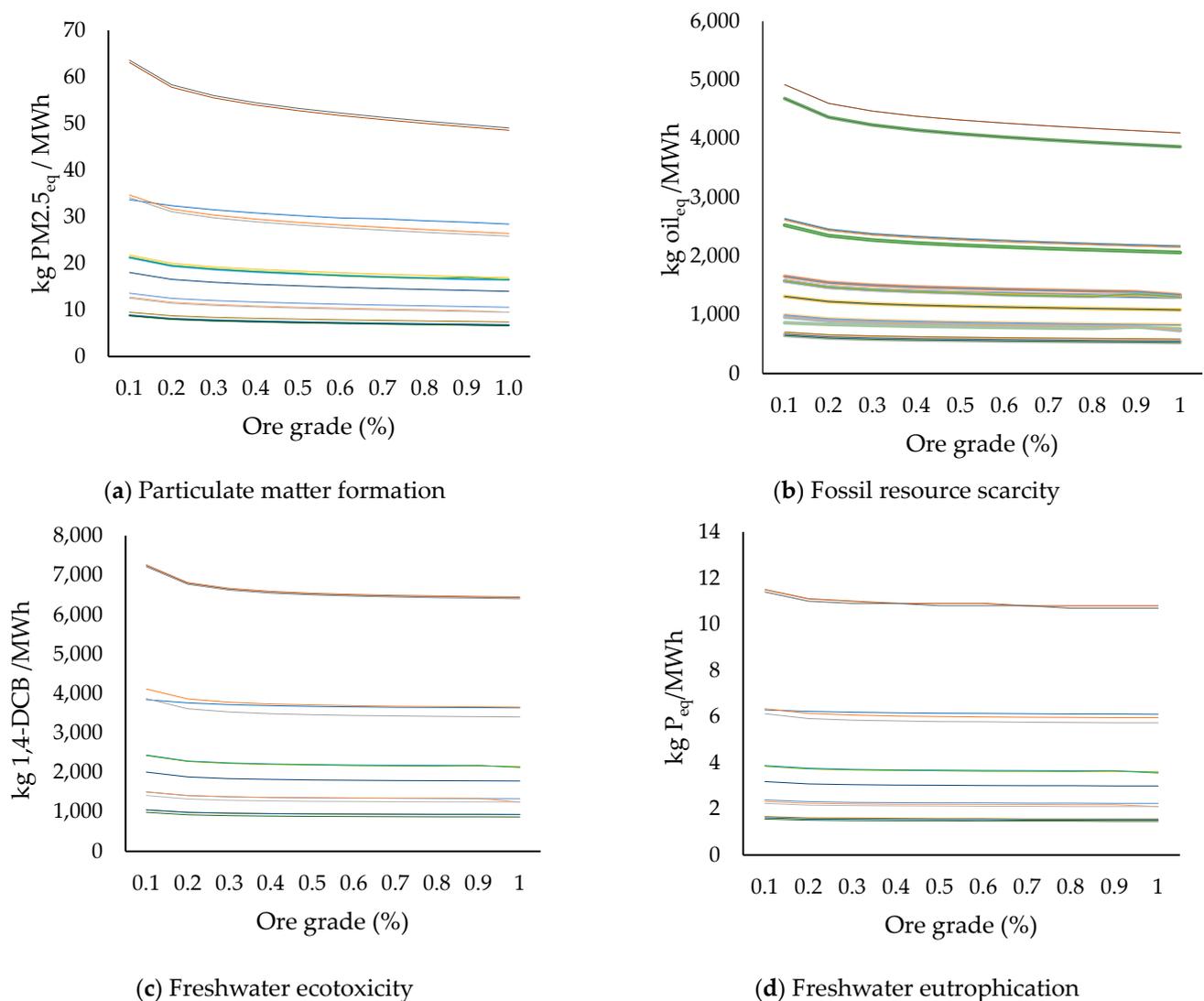
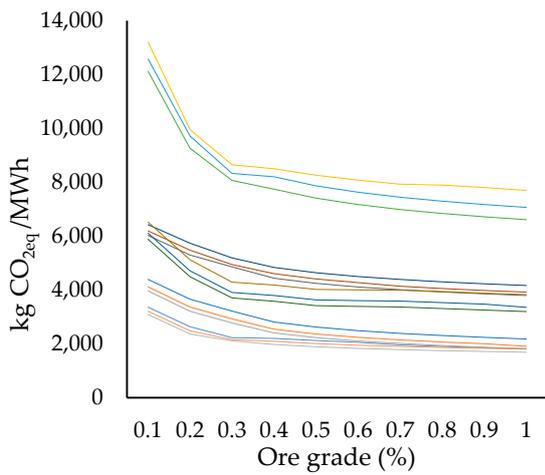
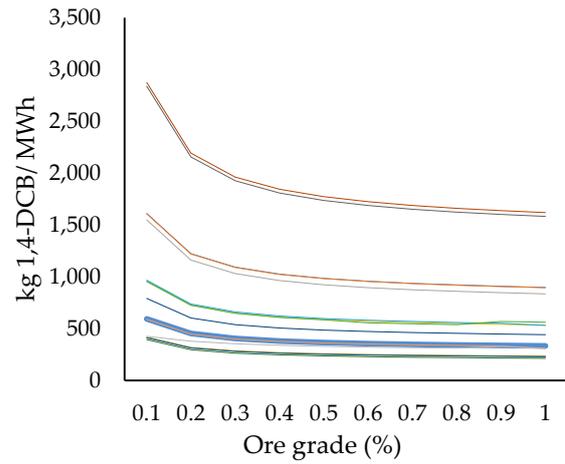


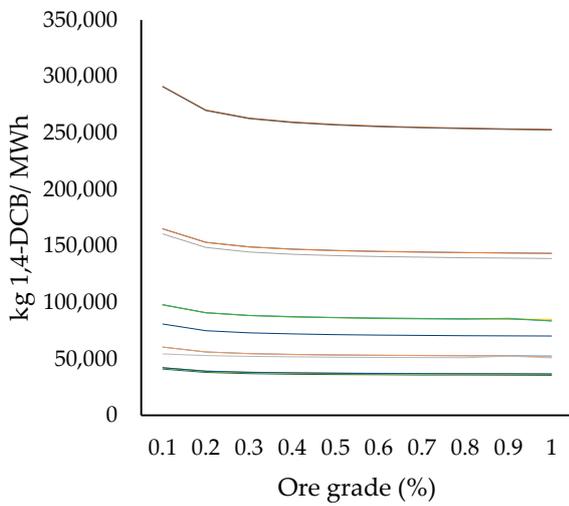
Figure 10. Cont.



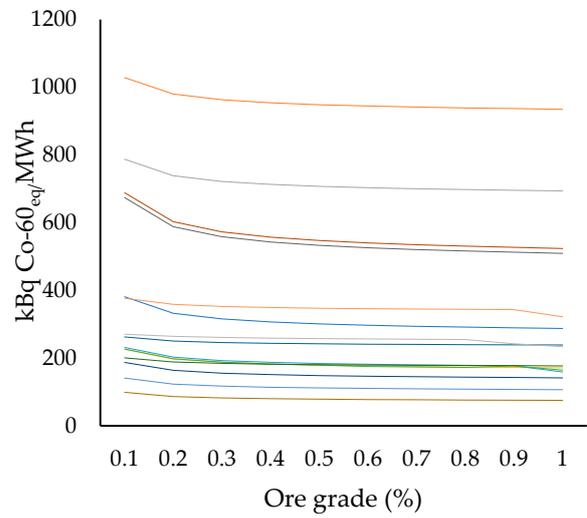
(e) Global warming potential



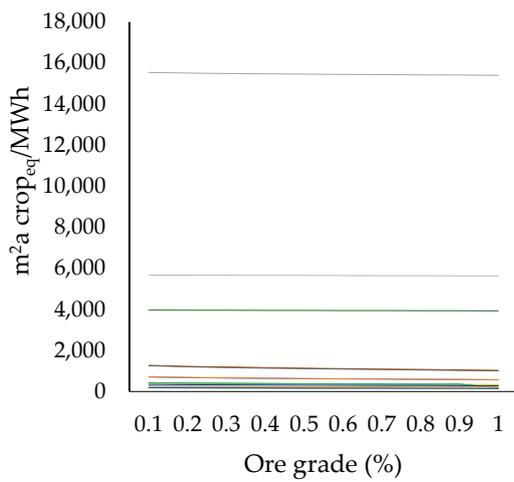
(f) Human carcinogenic toxicity



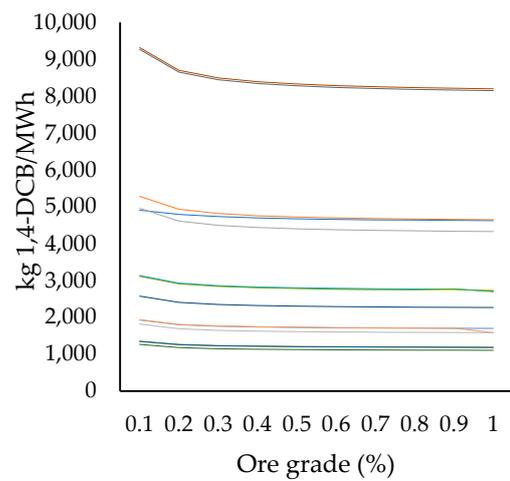
(g) Human non-carcinogenic toxicity



(h) Ionization radiation potential



(i) Land use potential



(j) Marine ecotoxicity

Figure 10. Cont.

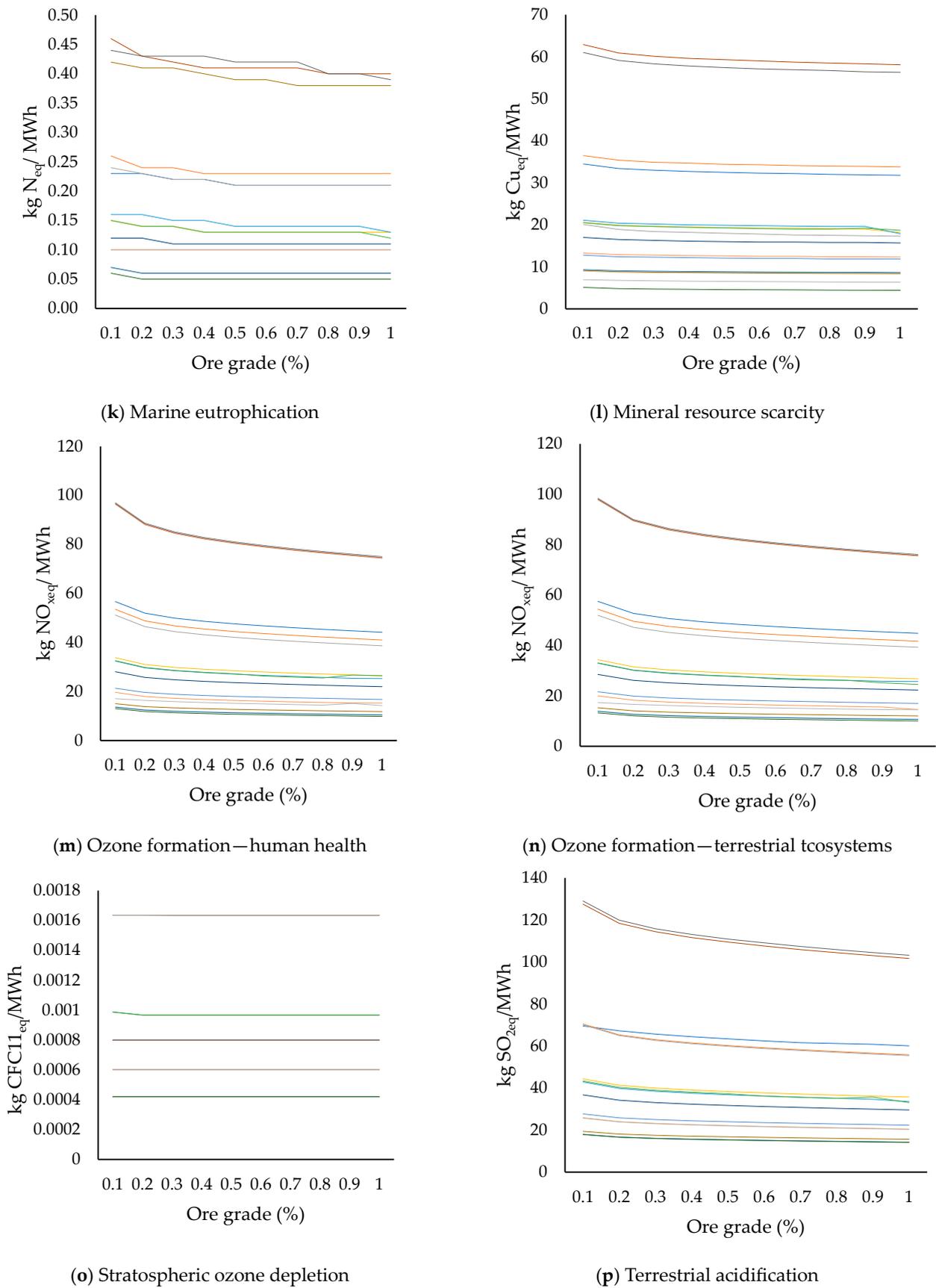
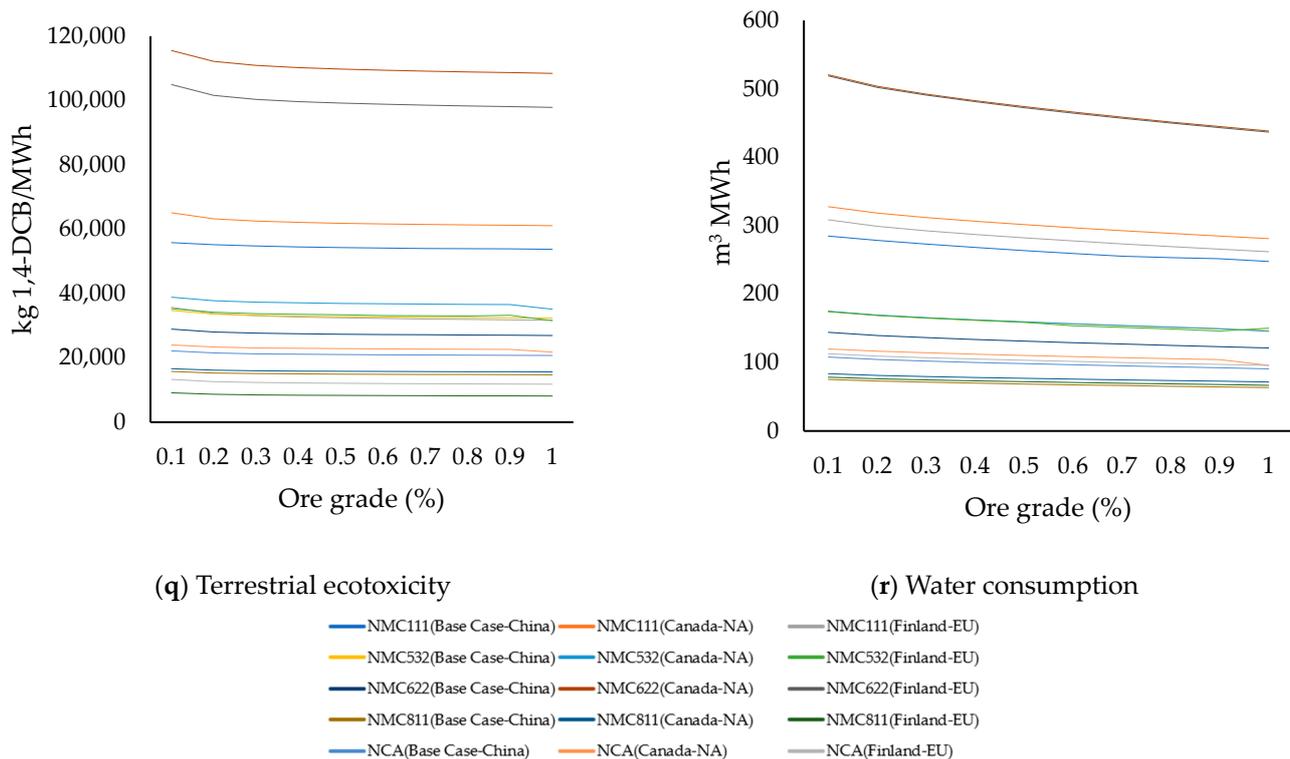


Figure 10. Cont.



**Figure 10.** Variation in environmental impacts with ore grade, refining location, and battery chemistry.

For GWP and carcinogenic toxicity, in particular, the dependency of environmental pathways on the ore grade is apparent, and this manifests in a lower (more negative) exponent on the power-law relationship, as shown in Table 7. For the other impact pathways, the dependency on the ore grade is subdued (i.e., a nearly straight-line relationship indicates an exponent close to zero). The results also highlight the highest dependency of the NMC111 chemistry on ore grade amongst all the battery chemistries studied. Production routes with the highest impacts follow the order of China, Canada (NA), and Finland (EU). For all five batteries, the results suggest an 8 to 12% decrease in the GWP in the Canada route and a 12 to 18% decrease in the GWP in the Finland route. The enhanced thermal combustion or chemical processing needed for low-quality ore will lead to impacts on some impact pathways that are higher than others. Variations of the different environmental parameters with ore grade, refining location, and battery chemistry are included in Table S6.

The LCA results were normalized to a common reference using factors introduced by Sleswijk et al. (2008) [73] for different environmental impacts. Normalized values of all 150 scenarios were analyzed. The comparison of major normalized impacts for an ore grade of 0.3% (Figure 11) shows that freshwater and marine ecotoxicity dominate the environmental impacts. When compared using the actual absolute values, although the magnitudes are less, the environmental impact of FETP and METP are more prominent in comparison to other numbers, owing to the emissions to the marine ecosystem from the life cycle routes. The detailed values are given in SI (Table S7).

The cumulative single score (Pt) for different battery chemistries is highest for NMC111 and lowest for NMC811 for the base case (where the refining location is in China and the ore grade is 0.3%). For the Canada refining location, the highest impact was noted for the NMC622 chemistry and the lowest impact for the NMC811 chemistry. The same trend also is seen in the Finland scenario. Other environmental impacts, when represented using a single-score scale, follow the same pattern as GWP. The results show the range of environmental impacts for the Canada route are between 58% and 65% of the base case, and impacts for the Finland route are between 71% and 76% of base case. This analysis gives an insight into the sensitivity to battery chemistry and refining location for the single-

score environmental impacts. Single-score calculations provide a first-order comparison of overall sustainability of different LIB chemistries and cobalt supply chains.

Table 7. Results of multivariate regression analysis.

	Constant	Ore Grade	Refining Location-Grid Mix	Cobalt Content in Battery	R <sup>2</sup>
Global Warming Potential (GWP)		$y_0 + a_1(g)^{\epsilon_1}$ $a_1 = 4866.2$ $\epsilon_1 = -0.271$			0.97
	4952.7		$y_0 + a_2(loc)^{\epsilon_2}$ $a_2 = 1604.78$ $\epsilon_2 = 1$		0.77
	1490			$y_0 + a_3(cob)^{\epsilon_3}$ $a_3 = 51.1$ $\epsilon_3 = 1$	0.99
		$y_0 + a_1(g)^{\epsilon_1} + a_2(loc)^{\epsilon_2}$ $a_1 = 4684.5$ $a_2 = 1632.7;$ $\epsilon_1 = -0.271$ $\epsilon_2 = 1$			0.94
	-1122	$a_1 = 1682.9$ $\epsilon_1 = -0.8323$		$y_0 + a_1(g)^{\epsilon_1} + a_3(cob)^{\epsilon_3}$ $a_3 = 62.7$ $\epsilon_3 = 1$	0.95
			$y_0 + a_2(loc)^{\epsilon_2} + a_3(cob)^{\epsilon_3}$ $a_2 = 4522.29$ $a_3 = 72.4185$ $\epsilon_2 = 1$ $\epsilon_3 = 1$		0.97

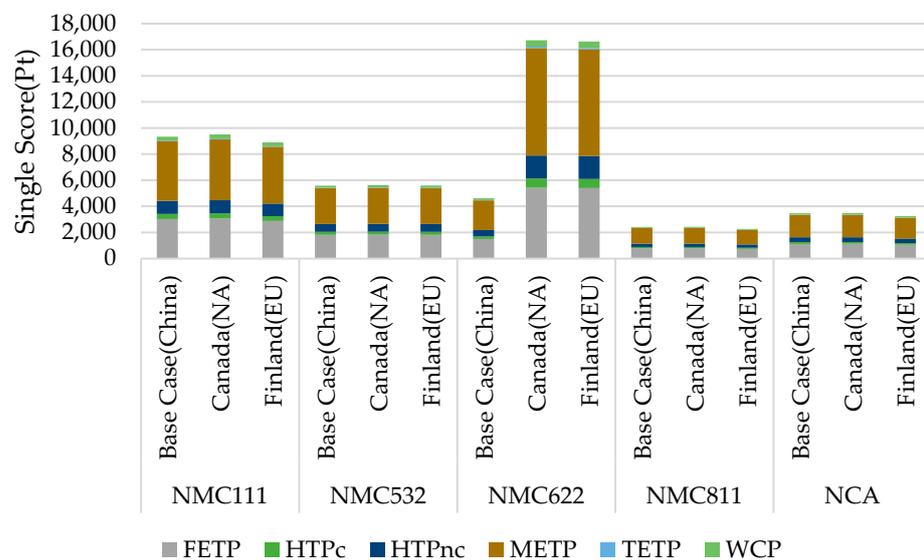


Figure 11. Single-score results for major environmental impacts comparison of the different combined scenarios (ore grade—0.3%).

### 3.3. Statistical Analysis

From the scenario analysis results (Table S6) focusing on GWP, the results ranged from a high of 13,200 kg CO<sub>2eq</sub> (0.05% ore grade, Chinese refining route, and NMC111 battery) to a low of 1700 kg CO<sub>2eq</sub> (1% ore grade, Finland refining route, and NMC811 battery). The median was found to be 3040 kg CO<sub>2eq</sub>.

We have already discussed that GWP varies with ore grade according to an inverse power law (Figure 5); thus, to assess the relative importance of ore grade, relative to other variables, the negative power coefficient was used as the independent variable term (Table 7). The percentage of fossil fuel used in the electricity grid mix (used as a surrogate for location) and cobalt content in the battery were other variables. Correlation coefficients for each relationship are also shown on the right-hand side of the table. The results show that cobalt content in the battery is the highest predictor ( $R^2 = 0.988$ ) of GWP, followed by ore grade ( $R^2 = 0.966$ ) and refining location ( $R^2 = 0.766$ ), when parameters are assessed for correlation individually.

General factorial regression was also used to analyze the impact of the independent variables. The results showed that ore grade and cobalt content in the battery contributed equally to the GWP footprint. Refining location had a comparatively lesser contribution when compared with the other two parameters.

#### 4. Discussion

The results of this research provide several insights regarding the cobalt supply chain. Addressing Q1 and Q2, no single process accounts for most of the contributions in all impact categories. Impacts from mining varied substantially, depending on which impact pathways were analyzed. Mining contributes significant levels of particulate matter formation potential but has lesser impacts on freshwater eutrophication, freshwater ecotoxicity, marine ecotoxicity, and water depletion. The refining processes led to the highest GWP, ozone depletion, and terrestrial ecotoxicity emissions, while refining contributed less to freshwater eutrophication, freshwater ecotoxicity, marine ecotoxicity, and particulate matter formation. Transportation contributed less than 10% of impacts to most categories compared to other life cycle phases, except for terrestrial ecotoxicity, to which its contribution was nearly 20%.

Addressing Q3, as ore grades decline over time below 0.3% cobalt by weight, the environmental impacts of mining and processing could begin to increase substantially. This finding is noteworthy because with cobalt demand expected to grow, higher ore grades could eventually become depleted below the 0.3% level used in the DRC base case. Alternative sources of cobalt (e.g., reprocessing of tailings or improved value chain) or recycling of batteries may extend the time that higher ore grades are available.

Addressing Q4, additional refinery locations outside of China could reduce the environmental impacts of the cobalt supply chain, but only to a certain degree. We found that altering location to Canada in North America and to Finland in Europe reduced the overall GWP, even given the offset from the longer travel distances. In these cases, the lower carbon intensiveness of the grid mix was the decisive factor. Addressing Q5, environmental impacts from the cobalt supply chain are directly proportional to the cobalt intensity of the battery. Reducing the cobalt intensity lowered the GWP by nearly half, as well as reducing other environmental pathway impacts.

The impacts in the presented study have been compared with values already available in the published literature (Table 8). The numbers shown for this study are for the base case with a refining location in China and an ore grade of 0.3%. However, the studies differ in several crucial aspects: battery chemistry, data sourcing and modeling—especially regarding the energy and material requirements along the supply chain—refining location, and the adopted LCIA method. As a result, we observe large differences among the reported impacts. For example, GWP is the lowest in this study, while TETP is the highest by an order of magnitude. Interestingly, the same study (Jiang et al. (2023) [74]) that reports the lowest TETP reports an METP that is 10,000 times larger than that found in this study (for the same battery chemistry). Peters et al. (2017) [42], in their review of the literature at the time, reported a range of 40 to 250 kg CO<sub>2eq</sub> per kWh. The HTP in this study is roughly in the middle of the range in the literature presented in Table 8. Overall, these differences highlight the importance of system boundaries and input assumptions as well as the influence of updated data sets when comparing results from different studies.

Moreover, the presented studies deal with complete-battery LCA, with some including the manufacturing of all the components, and do not allocate environmental impacts only to the cobalt portion. We could not find any publications which investigate the environmental impacts of only the cobalt supply chain on batteries; therefore, direct comparison can lead to incorrect conclusions. At the same time, a careful analysis of these systems and assumptions can help identify opportunities to reduce impacts if, for example, lower impacts in a study can be explained by switching energy sources or certain processes that use less water or energy.

Our results show that electricity used during the refining process is the major contributor to global warming potential, freshwater ecotoxicity, and carcinogenic and non-carcinogenic emissions. This stems from the diesel fuel used in refining processes that also contributes to ozone depletion. Transportation of refined products to battery manufacturing sites also depends on liquid fuels and can contribute to terrestrial ecotoxicity. Other mining and material separation processes such as blasting of the ore can have direct impact as well. It has already been reported that when the ore grade decreases, there is an increase in or no effect on environmental impacts.

Lastly, because battery storage systems will need to be augmented to maintain capacity and dispatch electricity more readily from intermittent generation sources, we can expect ongoing demand for cobalt. The extent of the battery augmentation needed for future deployments depends, in part, on how batteries are used in BESSs (e.g., for dispatching electricity when wind and solar are inactive, for maintaining frequency from variability of wind and solar resources, etc.). As battery technology and grid management improve with time, the upper limit of lifetime charge/discharge cycles might improve as well, also reducing the need for augmentation.

**Table 8.** Comparative analysis with select existing literature on LIBs (The studies referred to are lithium-ion battery LCAs including all components and battery manufacturing).

Ref	LCA Boundary/ Method	Fun. Unit	Battery	Impact Category									
				GWP (kg CO <sub>2eq</sub> )	HTP <sub>eq</sub> (1,4 DCB <sub>eq</sub> )	FETP (kg 1,4-DCB)	METP (kg 1,4-DCB)	TETP (kg 1,4-DCB)	TAP (kg SO <sub>2eq</sub> )	FEP (kg P <sub>eq</sub> )	MEP (kg N <sub>eq</sub> )	WCP m <sup>3</sup>	
Gutsch et al., 2024 [75]	Cradle to grave/ ReCiPe 2016 Midpoint	1 kWh	811	64.5									
Tabrizi et al., 2024 [76]	Cradle to gate/ ReCiPe 2016 Midpoint	1 kWh	111	85									
			622	76									
Popien et al., 2023 [77]	Cradle to gate/The ReCiPe Midpoint (H) V1.13	1 kWh	NCA	110.54	254.46								
			622	110.76	254.13								
Jiang et al., 2023 [74]	Cradle to grave/CML-IA	1 kWh	622	132	13.5	70.3	1.77 × 10 <sup>5</sup>	0.199	2.76				
			811	121	14.4	68.8	1.76 × 10 <sup>5</sup>	0.189	2.65				
Sun et al., 2020 [30]	Cradle to grave/ReCiPe Midpoint (H) V1.11/	1 kWh	622	93.57	12.53	1.5		0.01	0.49	0.01	0.02		
			111	59.5									0.44
Winjobi et al. 2022 [78]	Cradle to gate/ GREET	1 kWh	532	59.3									0.42
			622	56.4									0.39
Dai et al., 2019 [35]	Cradle to gate/ GREET	1 kWh	811	55.1									0.39
			111	72.87									0.752
Orozco et al., 2023 [79]	Cradle to gate/ GREET	1 kWh	111	78.10									
			532	81.13									
Peters et al., 2017 [42]	Review	1 kWh	622	77.43									
			811	77.37									
This study	Cradle to gate/The ReCiPe Midpoint (H) V1.13	1 kWh	NCA	82.33									
			NMC	40–250									
			<b>111</b>	8.5	150.15	3.7	4.7	54.8	0.07	6 × 10 <sup>-3</sup>	2 × 10 <sup>-4</sup>	0.27	
			<b>532</b>	5.2	88.89	2.2	2.8	33.2	0.04	4 × 10 <sup>-3</sup>	1 × 10 <sup>-4</sup>	0.17	
			<b>622</b>	4.3	73.46	1.8	2.4	27.6	0.03	3 × 10 <sup>-3</sup>	1 × 10 <sup>-4</sup>	0.14	
		(Base Case)	<b>811</b>	2.2	38.37	0.96	1.2	15.1	0.02	16 × 10 <sup>-2</sup>	6 × 10 <sup>-4</sup>	0.07	
			<b>NCA</b>	3.2	54.99	1.4	1.8	21.3	0.03	3 × 10 <sup>-3</sup>	8 × 10 <sup>-4</sup>	0.11	

## 5. Conclusions

The results from this study offer many opportunities for reducing the environmental impacts of the cobalt supply chain for LIBs used in BESSs and improving environmental sustainability. The results indicate that the largest influence on the environmental sustainability of the cobalt supply chain was that of battery type selection; therefore, reducing cobalt levels in future battery chemistries will lead to lower impacts from the cobalt, though it is unknown whether metals replacing cobalt are more or less environmentally sustainable. There may be additional impacts from other materials and processes, such as the refining of nickel or graphite, that could offset any environmental gains from reduced cobalt requirements. Each of the critical materials found in these batteries should be assessed in an integrated LCA, which was not possible in this study.

Such an integrated LCA could also include alternatives to diesel-powered mining equipment, which contributes most of the environmental impacts. The potential to substitute equipment powered by diesel fuel with equipment (even partially) powered by batteries could be one strategy for reducing these impacts. A recent case study highlights the potential to reduce the GWP of mining operations through the electrification of mining equipment [80]. Earl et al. (2022) [81] suggest that current mining practices are unsustainable and that responsible sourcing of ores, especially in the largest producer in the world, the DRC, is needed. In addition, a circular cobalt economy is needed via recycling and reuse. However, they observe that advances in identifying primary and secondary sources of cobalt, mining methods, and recycling are yet to be fully optimized. The lack of standardization of LIB chemistries and designs is offered as a particular handicap to improving cobalt sustainability.

As also observed by Earl et al. (2022) [81], LIB recycling is a nascent industry, and no recycling technologies are currently used at scale; the potential for environmental impact reduction from using recycled versus virgin cobalt could be substantial. However, as demand for cobalt continues to scale, efforts should be made to quantify the impacts of recycling, as its share of cobalt production will likely grow in the future.

Bamana et al. (2021) [82] offer important content on the responsible mining idea by focusing on human health and quality of life associated with cobalt mining in Lualaba, DRC. The authors focus on artisanal mining and gather data on social impacts such as health, safety, migration, child labor, and others to inform a social LCA approach. Social LCA can also be conducted to understand the impact on human well-being from emerging technologies. Environmental sustainability thus should be measured not only in terms of technical benefits, but also in terms of quality of life. They suggest that the joint use of social and environmental LCAs will help better inform the choice of mining technologies and practices to minimize impacts [82].

Finally, although this study focuses on cobalt, it expands on the environmental impact pathways that are typically reported. We include results on all ReCiPe 2016 (Midpoint H) categories that are included in the database. This choice was made to highlight the global ( $\text{CO}_{2\text{eq}}$ ) versus local impacts (all other categories) that are manifested through acquisition of cobalt. Our study also includes the element of time through the augmentation of the BESS, which will be necessary to maintain usable capacity when paired with intermittent energy generation. This more-comprehensive treatment of the supply chain and environmental impacts could provide other opportunities for understanding where and when impacts might occur, and how to mitigate them.

**Supplementary Materials:** The following supporting information can be downloaded at: <https://www.mdpi.com/article/10.3390/su16051910/s1>, Figure S1: Projected cobalt demand, 2020–2040; Figure S2: Relative proportion of environmental impacts per MWh of NMC111 storage, across all life cycle stages considered in the base case; Figure S3: Impact of ore grade on water depletion of mining and processing of  $\text{Co}(\text{OH})_2$ ; Figure S4: Single-score results for environmental impacts comparison of different battery chemistries (ore grade—0.3%); Table S1: Top five global producers of cobalt; Table S2: LCA studies of LIBs published in the literature; Table S3: Base case contribution analysis results;

Table S4: LCA results for mining and processing with varying ore grades; Table S5: LCA results for three different battery chemistries for an ore grade of 0.3% and a refining location in China; Table S6: LCA results for the combined scenarios; Table S7: Single-score results for battery chemistries and locations for an ore grade of 0.3%.

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