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Review

Decarbonizing lithium-ion battery primary raw materials supply chain

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SUMMARY

Decarbonizing the supply chain of raw materials for electric vehicle (EV) batteries is the ultimate frontier of deep decarbonization in transportation. While circularity is key, decarbonizing primary production is equally imperative. Here, we provide a blueprint for available strategies to mitigate greenhouse gas (GHG) emissions from the primary production of battery-grade lithium hydroxide, cobalt sulfate, nickel sulfate, natural graphite, and synthetic graphite. Shifting to renewable electricity and electrifying heat for mining and refining operations and reagents production emerges as a promising avenue. Combined, these measures can reduce the GHG emissions intensity by 53%–86% for the analyzed production routes. However, these reductions may not achieve absolute decoupling of GHG emissions from the growing demand driven by the rollout of EVs. Bridging this gap may require additional strategies, including low-carbon haul trucks, electrification of processing equipment, reagents regeneration and/or substitution, alternative reducing agents, improvements in material recovery rates, or new and emerging production technologies. Ultimately, an optimized portfolio of strategies is crucial for decarbonizing the production of raw materials that will power a net-zero future.

CONTEXT & SCALE

The demand for raw materials for lithium-ion battery (LIB) manufacturing is projected to increase substantially, driven by the large-scale adoption of electric vehicles (EVs). To fully realize the climate benefits of EVs, the production of these materials must scale up while simultaneously reducing greenhouse gas (GHG) emissions across their supply chain. This paper identifies available strategies to decarbonize the supply chain of battery-grade lithium hydroxide, cobalt sulfate, nickel sulfate, natural graphite, and synthetic graphite, assessing their mitigation potential and highlighting techno-economic challenges.

Low-carbon electricity, heat, and reagents are fundamental for decarbonizing battery-grade raw materials. However, even with a supply chain fully powered by renewable electricity and electrified heat, reducing future total emissions under an ambitious EV adoption scenario remains unlikely. Absolute decoupling of GHG emissions from demand growth would require additional decarbonization efforts, including low-carbon haul trucks, electrification of processing equipment, reagents regeneration and/or substitution,

INTRODUCTION

The Paris Agreement goal of limiting global warming to well below 2°C requires achieving global net-zero greenhouse gas (GHG) emissions around the second half of the 21st century.[1] Numerous scenarios can meet this target, all hinging on a massive deployment of clean energy technologies[2] and triggering an unprecedented surge in demand for raw materials pivotal to the manufacturing of electric vehicles (EVs), wind turbines, solar photovoltaic (PV) panels, fuel cells, and more.[3] Estimates for future raw materials demand vary depending on factors like the deployment of clean energy technologies, the specific technologies considered, and their material intensity.[4] For example, the International Energy Agency's (IEA) Net-Zero Emissions (NZE) by 2050 scenario forecasts a remarkable 3.5-fold increase in mineral demand for clean energy technologies from 2022 to 2030.[5] Most of this demand will come from the deployment of EVs, projected to grow from 30 million in 2022 to 380 million by 2030 under a net-zero scenario.[6] As a result, substantial spikes in demand for raw materials used in lithium-ion batteries (LIBs) are expected, including lithium (with a projected 8.6-fold increase by 2030), graphite (7.6-fold increase), nickel (7.6-fold increase), and cobalt (a three-fold increase) (Figure 1A).

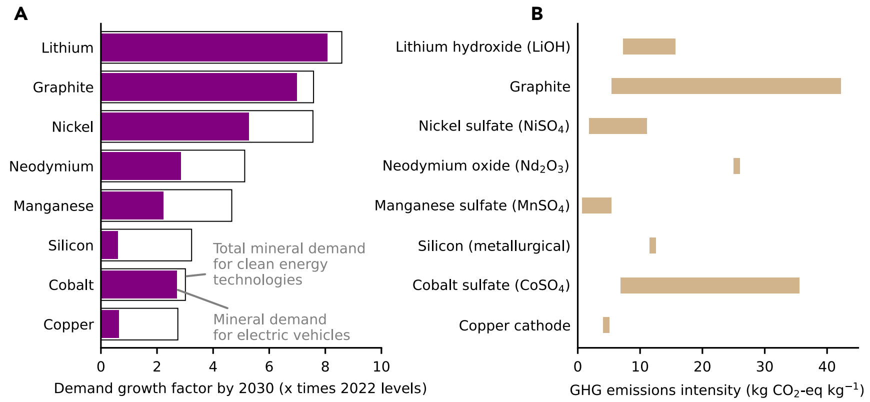

Figure 1. Minerals demand by 2030 in the IEA's Net-Zero Emissions scenario and GHG emissions intensities of associated commodities

(A) Demand growth factor for minerals used in clean energy technologies (including EVs) by 2030 compared with 2022 in the IEA's Net-Zero Emissions scenario.

(B) Cradle-to-gate life cycle GHG emissions of the primary production of associated commodities as reported in the literature (data sources can be found in Table S1). The range reflects the large variation in GHG emissions attributable to differences in resource characteristics, technologies, and production routes.

Paradoxically, mineral resources serve as the fundamental building blocks of a future net-zero world, while their supply stands as a large source of GHG emissions. Direct emissions stem primarily from high process emissions and energy use, making the mining and metal sector responsible for 40% of all industrial GHG emissions and for over 10% of global GHG emissions.[7,8] Lithium, cobalt, nickel, and graphite currently make a modest contribution to global emissions due to relatively low production volumes.[9] However, battery-grade lithium hydroxide, cobalt sulfate, nickel sulfate, and graphite are highly GHG emission-intensive commodities (Figure 1B). This raises concern about their future emission levels in light of skyrocketing demand. The production of battery-grade raw materials also contributes substantially to the carbon footprint of LIBs (e.g., 5%–15% for lithium and about 10% for graphite).[10,11] While it is highly unlikely for EVs to exhibit higher life cycle GHG emissions than fossil fuel vehicles, substantial emissions from the raw materials supply chain can potentially reduce their climate benefits. Additionally, potential barriers to mining and refining expansion may emerge if permits and licenses become contingent on progressively stringent GHG emissions requirements.[12]

Transitioning to a net-zero world evokes a systematic transformation across all industrial sectors, compelling the mining and metal sector to significantly scale up production to meet the growing demand while simultaneously reducing its GHG emissions.[12–14] The transition to a circular economy holds significant promise in this endeavor, as evidenced by the consideration of reduction, reuse, and recycling practices in numerous sector-specific decarbonization roadmaps (see Table S2). However, decarbonizing primary production remains imperative. Current efforts in this regard have predominantly focused on major raw materials such as steel, aluminum, and copper.[15–18] The academic literature has also primarily centered on the decarbonization of materials that currently contribute the most to GHG emissions, e.g., iron and steel,[19–21] aluminum,[21,22] copper,[23] and structural alloys.[24] While this body of knowledge provides possible decarbonization pathways, it is worth noting that decarbonization strategies are often process-specific, which limits their direct application to battery-grade raw materials.[21]

alternative reducing agents, improvements in material recovery rates, and new and emerging production technologies. All these strategies face techno-economic challenges that hinder widespread adoption in the short- and medium-term. These findings underscore the considerable challenge of fully decarbonizing the transportation sector—an effort that cannot be achieved solely through EVs. Now is the time to take decisive action on the raw materials supply chain.

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Recent reports from the IEA have highlighted decarbonization opportunities for lithium, cobalt, nickel, and graphite linked to low-carbon electricity, energy efficiency, and fuel switching.[12,25] However, these reports offer only a brief overview without fully exploring the spectrum of available options, their mitigation potential, and the associated challenges. More focused studies have delved into the decarbonization of specific hydrometallurgical[26] and pyrometallurgical processes,[27] thereby addressing only a segment of the extensive supply chain of LIB raw materials. Finally, a few prospective life cycle assessment (LCA) studies have assessed future GHG emissions for the supply of lithium,[28] cobalt,[29] and nickel,[30,31] yet also focusing solely on the evolution of specific parameters like declining ore grades, decarbonization of the power sector, secondary production share, and/or energy efficiency.[32] Therefore, a comprehensive assessment of the complete range of available decarbonization strategies throughout the entire supply chain of LIB raw materials, along with a discussion on their mitigation potential and challenges to be overcome, is still lacking.

Here, we analyze available strategies for decarbonizing the supply chain of battery-grade lithium hydroxide, cobalt sulfate, nickel sulfate, natural graphite, and synthetic graphite. While we recognize the importance of recycling and secondary production, our focus in this work is solely on primary production due to its anticipated dominance in the near future.[33] We start by identifying GHG emissions sources across the principal production routes based on LCA. Next, we identify the array of available decarbonization options for every source, discussing their mitigation potential and the techno-economic challenges hindering widespread adoption in the short- and medium term (i.e., 5–15 years). Moreover, we quantify the mitigation potential of a proposed decarbonization scenario and determine whether it could enable absolute decoupling of GHG emissions from demand across various EV adoption scenarios. This work can serve as a blueprint for supporting manufacturers and policymakers in identifying impactful decarbonization strategies and prioritizing future research and development.

GHG EMISSIONS ACROSS LIB RAW MATERIALS SUPPLY CHAIN

Pursuing decarbonization efforts requires a thorough assessment of direct and indirect GHG emissions across the supply chain. LCA is the most commonly used method for quantitatively assessing the environmental impacts of product systems throughout their life cycle.[34] High-quality and disaggregated life cycle inventory (LCI) data are crucial for conducting an LCA, yet such data remain scarce for battery-grade raw materials. Therefore, here we analyze the principal production routes for which LCI datasets are publicly available (Table 1). For each production route, we quantify the life cycle GHG emissions linked to mining, concentration, and refining (i.e., cradle-to-gate), distinguishing between emissions arising from on-site electricity consumption (i.e., electricity required for the operation of mining, concentration, and refining activities), process heating, diesel consumption, reagents consumption, process emissions (i.e., due to chemical reactions), and other minor sources (Figure 2). Details of the LCA methodology can be found in experimental procedures, and a description of the assessed production routes is provided in Note S1.

Electricity consumption

Electricity consumption represents the largest emission source for the natural graphite, synthetic graphite, and nickel sulfate production routes analyzed here, accounting for 79%, 79%, and 42% of their respective life cycle GHG emissions. Its role

Table 1. Battery-grade raw materials production routes analyzed in this study

Commodity	Production route	Representativeness	LCI data
Lithium hydroxide, from brine	lithium hydroxide monohydrate ($\geq$97% $LiOH \cdot H_2O$; $\geq$16% lithium) production in Chile from lithium brine extracted at Salar de Atacama	Salar de Atacama contributes to over 70% of the global supply of brine-based lithium chemicals[35]	Schenker et al.[36]; Kelly et al.[37]
Lithium hydroxide, from spodumene	lithium hydroxide monohydrate ($\geq$97% $LiOH \cdot H_2O$; $\geq$16% lithium) production in China from spodumene concentrate produced in Australia	Australia contributes to over 90% of the global supply of spodumene concentrate.[35] China dominates the refining stage.[5]	Kelly et al.[37]; Wernet et al.[38]
Cobalt sulfate	cobalt sulfate heptahydrate (21% cobalt) production in China from cobalt hydroxide produced in the Democratic Republic of Congo (DRC) from copper-cobalt ores	the DRC contributes to over 70% of the global supply of cobalt.[39] China accounts for over 75% of refined cobalt production.[5]	Arvidsson et al.[40]; Dai et al.[41]
Nickel sulfate	nickel sulfate hexahydrate (22% nickel) production reflecting the 2020 technology mix: pyrometallurgical and hydrometallurgical processing of both sulfide ores (44%) and laterite ores (56%). About 24% of total production comes from the processing of laterite ores through high-pressure acid leaching (HPAL)	accounting for 15% of the global production of nickel sulfate hexahydrate	The Nickel Institute [42,43]
Graphite natural	coated spherical graphite (99.95% C) production from flake graphite in Heilongjiang province in China	China supplies over 80% of natural graphite and 99% of the spherical graphite.[33]	Engels et al.[11]
Graphite synthetic	coated synthetic graphite production using the Acheson powder route in China	China supplies over 70% of synthetic graphite,[5] while Acheson powder is the mainstream production route.[44]	Carrère et al.[44]

is comparatively more modest for other commodities, representing 20% for spodumene-based lithium hydroxide, 14% for brine-based lithium hydroxide (7% if considering lithium carbonate; see Figure S1), and 8% for cobalt sulfate. Electricity-related GHG emissions are more significant in the refining and concentration stages. Its importance in mining is minor, as only a relatively small amount of electricity is used for ancillary applications such as pumping and conveyors. For synthetic graphite, the major source of electricity-related GHG emissions is the graphitization of calcined coke.

The varying contribution of electricity to life cycle GHG emissions across commodities stems from several factors, including the level of electrification and the location. Synthetic and natural graphite have the highest on-site electricity consumption, at 36 and 8.7 kWh kg^{-1}, respectively, primarily due to electrified high-temperature processes like graphitization (3,000°C) and coating (1,300°C).[11,44] Moreover, both types of graphite are produced in China (Table 1), where the electricity mix has an emissions intensity of 0.95 kg CO_2-equiv kWh^{-1}, varying substantially across graphite hubs—from 0.68 kg CO_2-equiv kWh^{-1} in Gansu province to 1.14 kg CO_2-equiv kWh^{-1} in Anhui.[45] Electricity GHG emissions intensity in the main graphite-producing regions of Heilongjiang and Inner Mongolia[46] is at the higher end, at 1.08 and 1.06 kg CO_2-equiv kWh^{-1}, respectively. Cobalt sulfate has the next highest electricity consumption at 5.7 kWh kg^{-1}, with about 80% of this occurring during the concentration stage in the Democratic Republic of Congo (DRC), where 96% of the electricity mix is hydropower. Spodumene-based lithium hydroxide requires 3.7 kWh kg^{-1}, nickel sulfate 3.1 kWh kg^{-1}, and brine-based lithium hydroxide 0.5 kWh kg^{-1}. It is noteworthy that electricity consumption is influenced by resource characteristics (e.g., ore grade and mineralogy)[29,30] and technology. Therefore, the relative contribution of electricity consumption (as well as the other contributors) to life cycle GHG emissions is likely to vary for different production routes.

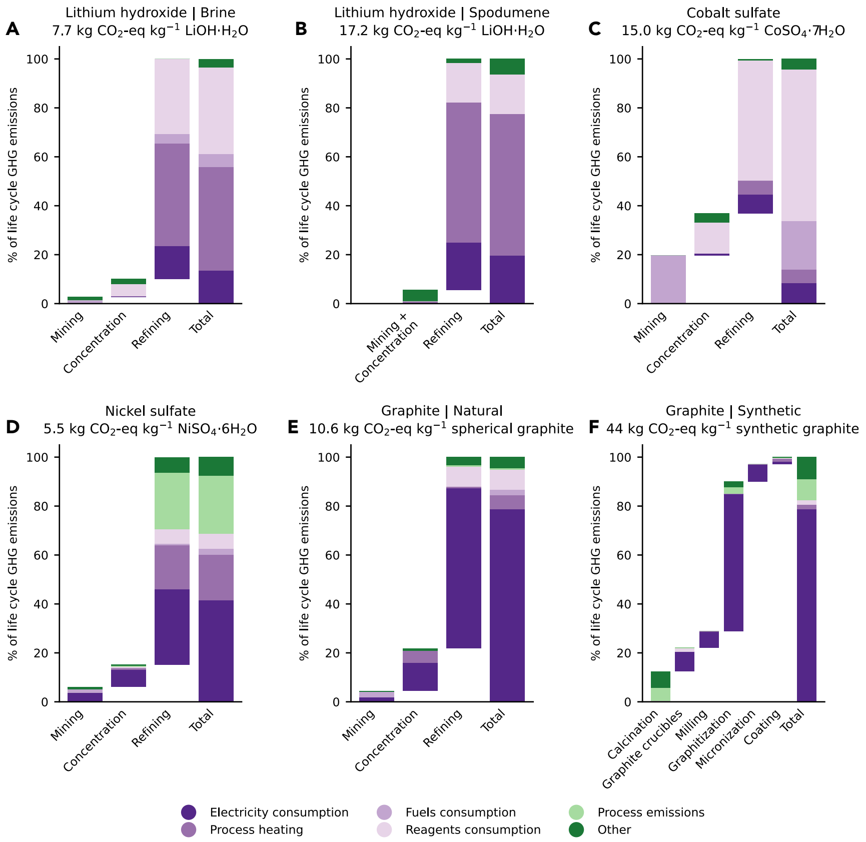

Figure 2. Breakdown of life cycle GHG emissions from the principal production routes for battery-grade raw materials

(A) Lithium hydroxide monohydrate production in Chile from brine extracted at Salar de Atacama.
(B) Lithium hydroxide monohydrate production in China from spodumene concentrate produced in Australia.
(C) Cobalt sulfate heptahydrate production in China from cobalt mined in the DRC.
(D) Nickel sulfate hexahydrate production through pyrometallurgical and hydrometallurgical processing of sulfide and laterite ores reflecting the 2020 technology mix.
(E) Coated spherical graphite production in China from flake graphite mined in China.
(F) Coated synthetic graphite production in China based on the Acheson powder route.
Emissions from on-site electricity generation from fossil fuels (e.g., diesel generators) and emissions from electrified heating are included within the category "electricity consumption." Results for lithium carbonate are provided in Figure S1.

Process heating

The combustion of fossil fuels for generating process heat constitutes the major source of emissions for the spodumene- and brine-based lithium hydroxide production routes, contributing to 58% and 42% of their respective life cycle GHG emissions (78% and 48% if considering lithium carbonate; see Figure S1). Moreover, process heat accounts for about 19% of emissions from nickel sulfate, 5.7% for cobalt sulfate, 5.5% for natural graphite, and 1.7% for synthetic graphite. The less significant role of process heating in these cases should be understood in the context of electric furnaces used for certain high-temperature processes such as the pyrometallurgical processing of nickel laterite ores and the graphitization and/or coating in synthetic and natural graphite production, resulting in heat-related emissions attributed to electricity consumption.

In general, the bulk of heat demand occurs during the refining stage. In the case of brine-based lithium hydroxide, lithium carbonate precipitation (requiring ca. 80°C) and the subsequent conversion of lithium carbonate to lithium hydroxide are the most energy-intensive stages. Additional heat is required for washing and drying the final product. All the operations rely on natural gas for heating, as defined in the original LCI dataset.[36] For spodumene-based lithium hydroxide, calcination (where the spodumene concentrate is heated to 1,100°C) and roasting (where the concentrate is roasted with sulfuric acid at 250°C) are the most heat-demanding processes. Additionally, steam (at 85°C) is used in the carbonation process, and a minor heat demand is required for drying the final product. This heat is generated from coal.[37] For nickel sulfate, a mix of natural gas, coal, and heavy fuel oil is used to supply process heat for the pyrometallurgical processing of sulfide ores and high-pressure acid leaching (HPAL) of laterite ores (involving the reaction of a slurry with sulfuric acid in an autoclave at ca. 255°C).[42] Overall, heat sources are typically specific to each production route, implying that they might differ for other routes not analyzed in this study.

Fuel consumption

Diesel and gasoline are used in various mobile and stationary equipment at every stage, including bulldozers, excavators, front end loaders, front shovels, surface miners, and/or drillers for ore mining, haul trucks for ore transportation, and grinding and milling facilities. The supply and combustion of diesel contribute to 20% of life cycle GHG emissions for cobalt sulfate, but less than 5% for other commodities. The cobalt sulfate LCI dataset indicates a consumption of about 15 kg of diesel per ton of mined ore based on primary data from a copper-cobalt mine in the DRC.[40,41] About 57% of this consumption is associated with mining equipment such as front end loaders and drillers, and the other 43% is due to haul trucks. For comparison, the LCI datasets for spodumene and nickel mining report 0.6 and 1.6 kg of diesel per ton of mined ore, respectively.[38,42] The reasons behind this variation are not clarified, but it might be related to factors such as fleet size, equipment employed, load factors, and the distance covered.

Reagent consumption

A wide variety of reagents are employed in processes like leaching, precipitation, and solvent extraction. Reagent consumption emerges as the primary source of GHG emissions for cobalt sulfate, accounting for 62% of its life cycle GHG emissions, and the second-largest source for brine-based lithium hydroxide, representing 36% of emissions. Moreover, reagents contribute to 16% of emissions from spodumene-based lithium hydroxide (9% if considering lithium carbonate; see Figure S1), 8% for natural graphite, 6% for nickel sulfate, and less than 2% for synthetic graphite. The majority of GHG emissions linked to reagents are associated with the refining stage. In cobalt sulfate production, the consumption of sodium hydroxide and hydrochloric acid alone contributes to over 45% of life cycle emissions (see Figure S2). Quicklime and soda ash stand out as the primary contributors to reagents-related emissions in brine-based lithium hydroxide production, representing 18% and 14% of emissions, respectively. In spodumene-based lithium hydroxide production, sodium hydroxide is the most noteworthy reagent, accounting for 10% of emissions. Reagents-related emissions in natural graphite production are related to nitric acid, hydrogen fluoride, and hydrochloric acid. Hydrated lime, sodium hydroxide, and ammonia are the major contributing reagents for nickel sulfate.

Process emissions

Process emissions account for over 24% of life cycle GHG emissions in nickel sulfate production. Direct CO_2 emissions primarily stem from the use of coke, coal, and petroleum coke as reducing agents in pyrometallurgical processing of nickel ores, as well as from the use of limestone as a neutralization agent in hydrometallurgical processing of laterite ores. Due to the aggregation of the nickel sulfate LCI dataset, it was not possible to determine the specific contribution of process emissions to each production route.[42] Moreover, process emissions account for 8.5% of the life cycle GHG emissions in synthetic graphite production. These emissions are primarily generated during the calcination of petroleum coke and the subsequent graphitization stage, due to the partial oxidation of the carbon-rich packing media used in the electric furnace.[44] The production of the other commodities does not involve direct emissions of CO_2 from chemical reactions.

Other emission sources

Other minor sources of GHG emissions may include, depending on the production route: raw materials (such as petroleum coke used for synthetic graphite production), waste treatment, tailings management, long-distance transportation, or the embodied emissions in infrastructure. Overall, these other sources contribute to 1%–9% of life cycle GHG emissions. These findings are aligned with findings for the mining and metal sector as a whole, e.g., concerning the lower relevance of GHG emissions from explosives,[47] tailings disposal,[48] or long-distance transportation.[49]

DECARBONIZATION STRATEGIES

Here, we describe available strategies to reduce GHG emissions from the supply chain of LIB raw materials and discuss their relevance across commodities and the key techno-economic challenges that could hinder their adoption and effectiveness.

Blueprint for LIB raw materials decarbonization

Building on the GHG hotspot analysis and an extensive review of industry roadmaps and scientific literature (see Table S2), we developed a tailored blueprint for decarbonizing LIB raw materials (Figure 3). Available strategies include the following, ordered by emission source: (1) decarbonization of on-site electricity consumption, (2) improvements in energy efficiency, (3) decarbonization of process heat (4) low-carbon haul trucks, (5) electrification of processing equipment, (6) reagents regeneration and reuse, (7) reagents substitution, (8) decarbonization of the chemical industry (i.e., reagents production), (9) hydrogen and biomass-based reducing agents, (10) carbon capture, utilization and/or storage (CCUS), (11) improvements in material recovery rate, (12) new and emerging production technologies, (13) decarbonization of electricity consumption elsewhere in the global economy, and (14) carbon dioxide removal (CDR, considered as an offsetting mechanism).

It is worth noting that a single emission source may have several mutually exclusive strategies for reducing emissions. For example, GHG emissions from process heat can be reduced by shifting away from fossil fuels (i.e., decarbonized heat) or by implementing CCUS. Similarly, multiple alternatives may exist within decarbonization strategies, such as electrified heat, low-carbon fuels, and renewable heat sources for process heat, as well as batteries, hydrogen fuel cells, or biofuel for haul trucks. Moreover, the relevance of decarbonization strategies varies across commodities. To identify the most relevant strategies for each commodity, we assigned a low, medium, high, or indeterminate mitigation potential based on a qualitative assessment of the emission source significance and the technology potential (for further details, see experimental procedures).

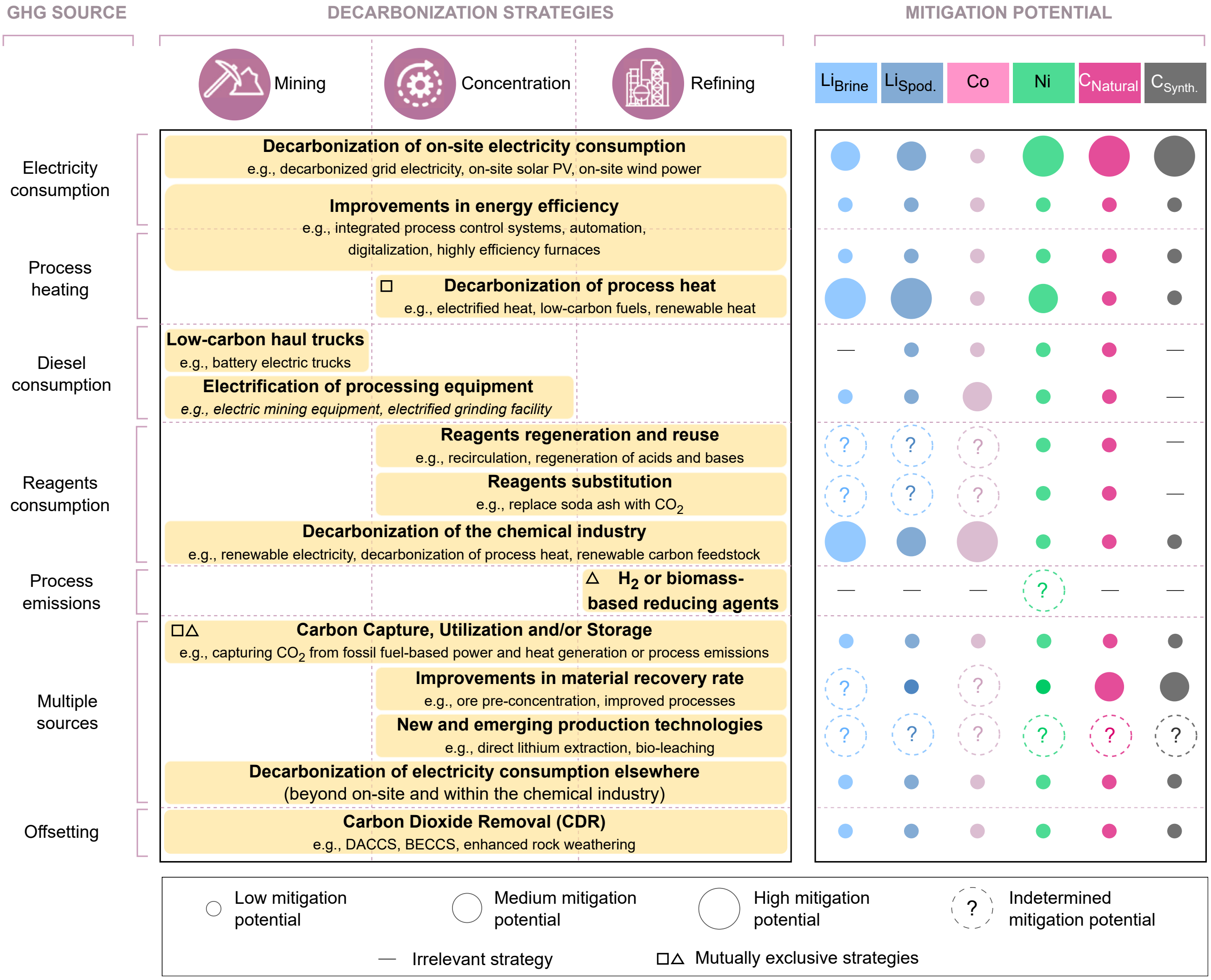

Figure 3. Strategies for reducing GHG emissions from battery-grade raw materials supply chain

Strategies are organized by emission source and relevance for each production stage (mining, concentration, and refining). The mitigation potential was derived from the qualitative assessment of the significance of the emission source and the technology potential (for more details, see experimental procedures). Li$_{Brine}$ and Li$_{Spod.}$ refer to brine- and spodumene-based lithium hydroxide monohydrate, respectively; Co refers to cobalt sulfate heptahydrate; Ni refers to nickel sulfate hexahydrate; C$_{Natural}$ and C$_{Synth.}$ refer to coated spherical natural graphite and coated synthetic graphite, respectively.

Decarbonization of on-site electricity consumption

Low-carbon electricity is key for all the production routes analyzed, except for cobalt sulfate, which predominantly relies on hydropower. The mitigation potential is noteworthy for natural graphite, synthetic graphite, and nickel sulfate due to their reliance on electrified high-temperature processes. Transitioning to low-carbon electricity can be achieved either indirectly through a decarbonized power grid (as for cobalt sulfate) or directly by installing renewable power generation technologies on-site. Mining and metal companies are increasingly adopting wind turbines and solar PV panels, with renewable energy capacity at mining sites growing from 0.6 GW in 2015 to 5 GW installed or announced by 2019, and further expansion expected.[50,51] Examples of mines operating on a substantial share of wind or PV power include the Chuquicamata copper mine in Chile, the DeGrussa copper-gold mine in Australia, or the Veladero gold mine in Argentina.[50,52] Other low-carbon options include geothermal power, adopted in lithium extraction from geothermal brines,[53] or on-site natural gas-based power generation coupled with CCUS.[21] These options have the advantage of co-generating power and heat, though they may face limitations such as availability (geothermal) and operational challenges (CCUS, discussed below). Finally, mining and metal companies may also use power purchase agreements to access low-carbon power generation such as wind, solar, hydro, or nuclear.[51]

Improvements in energy efficiency

Energy consumption in the mining and metal sector has been continuously optimized over time, suggesting relatively modest additional energy efficiency gains and thus mitigation opportunities in the short- and medium-term.[54,55] For example, an analysis of the European Union (EU) non-ferrous metal industry indicates an economic potential to reduce overall energy consumption by about 0.5% per year (for a cumulative reduction of 12% by 2050) through measures such as highly efficient furnaces, integrated control systems, waste heat recovery, and combustion optimization.[18] Other studies have reported a 5% reduction in GHG emissions from the Australian mining sector due to optimization and efficiency,[56] a 6% energy savings in class 1 nickel production through waste heat recovery,[16] and a 10%–30% energy consumption reduction from adopting modern boilers equipped with integrated control systems.[57]

Decarbonization of process heat

Shifting from fossil fuel combustion to alternative methods for generating process heat has large mitigation potential for lithium hydroxide and nickel sulfate production. Electrification of heating, when supplied with low-carbon electricity, has gained prominence in climate change mitigation scenarios, with almost 80% of industrial heat demand electrifiable with current technologies.[58] Heat pumps and electric boilers are commonly used for various industrial low-temperature applications (<100°C) such as product drying.[59] Established technologies for electrifying high-temperature operations, like calcination in spodumene-based lithium hydroxide production (1,200°C), include resistance furnaces and electric arc furnaces.[58] Examples of electric furnace adoption can be seen in the natural and synthetic graphite coating process, graphitization for synthetic graphite, and the pyrometallurgical processing of nickel laterite ores. Electric smelting is also applicable to the pyrometallurgical processing of nickel sulfide ores.[43] Alternatively, green hydrogen, produced via water electrolysis powered by renewables, is also gaining momentum for high-temperature applications.[60] Other low-carbon fuel options include biomass and ammonia.[60] Renewable heat sources, such as solar and geothermal energy, face significant challenges, including solar intermittency, limited availability (e.g., geothermal energy), and difficulty in achieving the required high temperatures. Alternatively, CO_2 emissions from fossil fuel-based process heat could be reduced through CCUS, as discussed below.

Low-carbon haul trucks

Low-carbon haul trucks present a promising avenue for reducing mining-related emissions, yet their mitigation potential remains low due to the minor contribution of mining to life cycle GHG emissions and the technology's current stage of development. Among available alternatives—such as natural gas-powered trucks, battery-electric trucks, hydrogen fuel cell trucks, and biofuel-powered trucks—battery-electric trucks powered by renewable electricity emerge as the frontrunner.[61] According to the International Copper Association, the electrification of haul trucks is anticipated to have a limited impact until 2030, while the large-scale production of battery-electric and fuel cell trucks is foreseen to emerge after 2040.[17] For example, BHP's Escondida copper mine in Chile aims to replace all trucks with battery-electric trucks and other low- or zero-emission trucks by 2033,[62] while Teck Resources Limited, one of the largest mining companies in Canada, plans to deploy 30 low-carbon haul trucks by 2027.

Electrification of processing equipment

Machinery and auxiliary equipment employed for ore mining (e.g., front end loaders and drillers) and concentration (e.g., grinding and crushing equipment) could also be electrified,[52] yet with generally limited mitigation potential due to minor contribution of fuel consumption in mining and concentration to life cycle GHG emissions. For cobalt sulfate, a medium mitigation potential is reported due to the high amount of fuel used by processing equipment in mining operations in the analyzed production route.[40,41]

Reagents regeneration and reuse

Regenerating and reusing reagents can minimize their consumption and reduce the GHG emissions associated with their production. This includes acids and bases used for leaching and pH control, solvents used in solvent extraction processes, and reducing and oxidizing agents.[26] Recovery and recirculation of solutions produced from process reactions is already common for some commodities. For example, 98% of organic solvents used in lithium brine purification are recovered.[36,63] Pyrohydrolysis, a highly energy-intensive process, has been traditionally used in industry for regenerating hydrochloric acid (key for cobalt sulfate and natural graphite).[64] Nonetheless, opportunities for further improvements exist for every commodity. The precipitation of lithium hydroxide with calcium hydroxide generates large amounts of solid calcium carbonate, which is often discarded.[65] Ongoing research explores less energy-intensive hydrochloric acid regeneration methods, including metal sulfate salt crystallization, electrowinning, solvent extraction, and hydrolytic distillation.[66] The mitigation potential of emerging circular hydrometallurgical concepts depends on their recovery efficiency and energy requirements, necessitating comprehensive process simulations.

Reagents substitution

Manufacturers could also explore alternative reagents options to reduce GHG emissions. Many reagents used in LIB raw materials production have potential substitutes. For example, sulfuric acid could be replaced with organic acids such as citric, oxalic, and formic acid, or sulfuric acid recovered from acid mine drainage.[64,67] CO_2, which can be captured from the flue gas generated by burning fossil fuels for process heat generation, has been proposed as a substitute for soda ash in the precipitation of lithium carbonate.[68] Steel slags have been used as neutralization agent substituting limestone.[69] The mitigation potential of reagent substitution depends not only on the GHG emissions intensity of producing alternative reagents but also on their performance relative to the benchmark. Therefore, the potential for reducing GHG emissions needs to be confirmed by comprehensive LCAs.[70]

Decarbonization of the chemical industry

Beyond minimizing consumption or replacing certain reagents, the decarbonization of LIB raw materials could largely benefit from the decarbonization of the chemical industry. The chemical industry is actively pursuing renewable electricity sources, electrification of heating, and renewable carbon feedstocks.[71–73] Efforts such as gradually replacing coal-fired boilers with cleaner alternatives like wood waste in soda ash production (critical for lithium hydroxide) are already underway at Solvay facilities.[74] Similarly, emissions from sodium hydroxide production, used in spodumene-based lithium hydroxide and cobalt sulfate, largely stem from electricity consumption in electrolysis, suggesting potential reductions through a shift to decarbonized power grids or on-site renewable power generation.

Hydrogen and biomass-based reducing agents

Process emissions from the pyrometallurgical processing of nickel ores could be reduced by substituting coke and coal reducing agents with hydrogen or biomass-based materials. Green hydrogen has generated considerable attention as a more sustainable reducing agent in metallurgy.[27] Biomass-based materials such as wood waste or biochar have also been proposed as promising alternatives because they are compatible with current infrastructure relying on solid reducing agents.[75] For example, a previous LCA demonstrated that replacing coal with biochar could reduce GHG emissions by 45% in ferronickel smelting.[76] However, similar studies are not available for nickel sulfate production.

CCUS

CCUS offers a potential solution for reducing CO_2 emissions from various sources, including electricity consumption, process heat, and process emissions. CCUS involves engineering solutions for capturing CO_2 from flue gas and either storing it underground permanently or utilizing it in production processes requiring carbon (e.g., chemicals production). CCUS can be applied to fossil fuel-based power and heat cogeneration, addressing two major emission sources while providing baseload power.[21] However, as far as we are aware, there are currently no known examples of CCUS applied to LIB raw materials production,[77] likely due to economic and operational challenges. The economic feasibility of CCUS for non-ferrous raw materials remains uncertain due to high capital costs and the relatively lower direct CO_2 emissions compared with iron and steel production.[18] Additional operational challenges include the need for access to CO_2 geological storage sites (likely unavailable at most mining and refining locations)[78] and the complex supply chain of transporting and injecting CO_2.[79] For CO_2 utilization, on-site demand is preferable, but this is limited for LIB raw materials production processes, except for substituting soda ash in lithium hydroxide production. These factors limit the mitigation potential of CCUS for LIB raw materials.

Improvements in material recovery rate

Improving material recovery rates can mitigate GHG emissions by reducing the amount of material processed, thus decreasing energy and resource consumption. The mitigation potential can be important for natural and synthetic graphite due to the relatively low production yields achieved in the spheronization (45%) and micronizing (60%) processes.[44,80] Improved processes have been proposed in the scientific literature, capable of increasing yields up to 80% while simultaneously reducing energy consumption.[81] A prior LCA demonstrated that achieving an 80% micronizing yield could reduce the life cycle GHG emissions of synthetic graphite by about 23%.[44] For other commodities, refining processes often achieve recovery rates above 90%,[82] while the concentration stage remains relatively inefficient, with recovery rates of around 40%–60% for lithium extraction from brines, 66% for lithium extraction from spodumene, 55% for cobalt, and 79% for nickel.[82,83] Ore pre-concentration techniques like dense media separation, gravity separation, magnetic separation, electrostatic separation, and flotation could increase the recovery rate, yet it remains uncommon largely due to complex mineralogies.[84] Moreover, the concentration stage often has a minor influence on life cycle GHG emissions (e.g., as in spodumene-based lithium hydroxide or nickel sulfate production), limiting the mitigation potential of this strategy. It is worth mentioning that reprocessing of mining waste and tailings represents another relevant strategy for improving material recovery, especially for cobalt and nickel.[85] However, this strategy is not further explored in this study as it is more closely associated with recycling and secondary production, which are beyond the scope of our assessment.

New and emerging production technologies

New and emerging technologies for concentration and refining could drastically affect the GHG emissions of LIB raw materials production. Often, these innovations aim to maximize material recovery. For example, direct lithium extraction (DLE) technologies like adsorption, ion-exchange, solvent extraction, membrane separation, and electrochemical separation can increase lithium recovery from brines up to 80%–99%.[83,86] Various DLE concepts are currently being tested at scale, with commercial implementation expected between 2025 and 2030.[83] Cobalt recovery can benefit from the adoption of new leaching methods that maximize its recovery (e.g., cobalt recovery rate in the DRC varies from 36% to 77%, compared with 97% for copper).[25,87] Bio-leaching is gaining attention as an alternative for processing complex low-grade nickel laterite ores. This process operates at atmospheric pressure, potentially reducing electricity consumption, and does not require acids.[88,89] There is also growing interest in using biomass as a precursor for producing battery-grade graphite, which offers an alternative to the energy-intensive natural and synthetic graphite production routes.[90] Catalytic graphitization of biochar, obtained from the pyrolysis of biomass waste or energy crops, has shown GHG emissions intensities between 1.2 and 3.5 kg CO_2-equiv kg^{-1} battery-grade graphite.[91,92] However, despite the promising mitigation potential of emerging technologies, most are still in the early development stages and far from industrial deployment. Therefore, we do not assign mitigation potentials to this decarbonization strategy due to the large uncertainties associated with their future scale up. For example, relative uncertainty in GHG emissions of DLE technology has been found as high as 79% elsewhere.[53]

Decarbonization of electricity consumption elsewhere

In addition to electricity consumed for on-site operations and reagents production, electricity consumption elsewhere in other sectors of the global economy also affects the life cycle GHG emissions associated with LIB raw materials. This includes, e.g., electricity used in fuel production and the manufacturing of renewable power generation technologies. However, the overall mitigation potential might be limited, as upstream production processes other than reagents production contribute relatively little to the life cycle GHG emissions.

CDR

In addition to available strategies for reducing GHG emissions, a portfolio of CDR technologies is available for compensating emissions. These include bioenergy with carbon capture and storage (BECCS), direct air capture and storage (DACCS), forestation, soil carbon sequestration, biochar, ocean fertilization, and enhanced rock weathering.[93] Enhanced rock weathering, which involves the reaction of atmospheric CO_2 with minerals to form bicarbonate solutions, is particularly noteworthy due to the large volume of rock waste generated during mining and concentration activities.[94] The estimated CDR potential of tailings associated with LIB raw materials reaches 200–250 kg CO_2 t^{-1} for sediment-hosted copper-cobalt deposits, 10–30 kg CO_2 t^{-1} for nickel laterite deposits, and 140–990 kg CO_2 t^{-1} for nickel sulfide deposits.[95] Globally, the CDR potential based on current tailings production volumes is estimated at 290–642, 0.002–0.02, and 14–175 Mt CO_2 $year^{-1}$ for copper-cobalt, nickel laterite, and nickel sulfide deposits, respectively. This potential could increase substantially in the future driven by increased demand and declining ore grades. Companies like BHP Nickel are already exploring methods for exploiting the CDR potential of tailings.[96] However, enhanced rock weathering is still in the early stages of development (with a technology readiness level [TRL] of 5),[93] which coupled with the absence of a policy framework for accreditation and assurance of carbon offsets makes its future deployment largely uncertain.[93] Even with technological advancements, mineral carbonation will require long intergenerational timeframes, rendering this strategy less relevant in the short- and medium-term but potentially impactful over decades or centuries.

Techno-economic challenges in LIB raw materials decarbonization

The decarbonization strategies outlined above face distinct techno-economic challenges that ultimately influence their adoption and shape their mitigation potential. Table 2 summarizes these challenges and identifies key opportunities to address them.

Securing low-carbon electricity is fundamental to decarbonizing LIB raw materials, as the effectiveness of various decarbonization strategies—such as electrification, hydrogen use, and emerging, potentially more energy-intensive production technologies—depends heavily on the GHG emissions intensity of electricity. Leveraging a decarbonized power grid provides access to low-carbon electricity without major investments from mining and metal companies, though this approach is constrained by the uneven global progress in power sector decarbonization and may be impractical for isolated production sites. Unlike mining, refining offers greater flexibility in relocation (acknowledging potential logistical challenges), allowing for the strategic planning of future facilities in regions with decarbonized power grids or strong commitments to achieve it. This ''green relocation'' can offer competing benefits such as cost savings, infrastructure availability, and political and economic stability,[97] and aligns with regulatory frameworks like the European Critical Raw Materials Act's goal for the EU to domestically process 40% of its demand for strategic raw materials.[98]

Shifting to on-site renewable electricity generation comes with inherent risks due to the variability and intermittency of renewable energy sources. The mining and metal sector typically operates under a continuous operation model, which necessitates a constant electric load.[61] Therefore, transitioning to renewable electricity under this business model requires either grid connection, backup generators, or battery energy storage systems (BESS). Currently, remote mining sites without direct access to the power grid resort to diesel generators.[51,99] In the future, expanding grid infrastructure and progressively adopting BESS will be crucial. Moreover, producers could explore shifting toward flexible operation modes, where mining, concentration, and refining processes are synchronized with renewable energy availability. Many processes can be adapted for flexible operation, including crushing, grinding, flotation, electric haul trucks, and electrified heat (heat pumps, electric boilers, or electric furnaces).[100,101]

High investment costs and increased operational expenditures pose significant barriers to decarbonizing the supply chain of LIB raw materials. Integrating renewable electricity can already improve operating margins for mining and metal companies,[50,51] especially at isolated sites where fossil fuel supplies are expensive. However, at the current prices, the costs of fully operating on renewable electricity with BESS are currently prohibitive.[61] Addressing the intermittency of renewable energy

Table 2. Key techno-economic challenges associated with decarbonization strategies for LIB raw materials and opportunities to address them

Decarbonization strategy	Key challenges	Key opportunities
Decarbonization of on-site electricity consumption	uneven global progress in power sector decarbonization; intermittent renewable energy requires grid connection, backup generators, or energy storage; integrating energy storage would currently make costs prohibitive	on-site renewable power generation is already cost-competitive; relocating refining capacity; expanding power grid infrastructure; declining energy storage technology costs; shifting to flexible operation modes
Improvements in energy efficiency	many processes are already optimized; high costs for marginal gains in energy efficiency	upgrading old equipment to modern, more efficient alternatives (e.g., boilers equipped with integrated control systems and waste heat recovery)
Decarbonization of process heat	effective mitigation of electrified heat or hydrogen requires access to low-carbon electricity; costs of electrified heat or hydrogen are heavily influenced by electricity costs	electrification of process heat possible with current technologies; declining costs of renewable power generation and energy storage technologies; development of policy frameworks supporting an hydrogen economy
Low-carbon haul trucks	electric haul trucks are not yet a mature technology; effective mitigation requires access to low-carbon electricity	advancements in battery systems for large haul trucks
Electrification of processing equipment	effective mitigation requires access to low-carbon electricity	–
Reagents regeneration and reuse	conventional regeneration methods are energy-intensive	development of innovative regeneration methods that are less energy-intensive
Reagents substitution	limitation of alternative reagents to achieve the desired performance	organic acids; recovery of sulfuric acid from acid mine drainage; substitution of soda ash with captured CO_2
Decarbonization of the chemical industry	hard-to-abate chemicals; increased reagents production costs impact battery-grade raw material prices	chemical industry explores CCS for hard-to-abate chemicals
Hydrogen and biomass-based reducing agent	technologies not yet demonstrated at scale; effective mitigation requires access to low-carbon electricity	development of policy frameworks supporting an hydrogen economy; declining costs of water electrolysis
CCUS	high costs; requires access to CO_2 geological storage sites or on-site CO_2 demand	CCUS applied to co-generation can simultaneously reduce emissions from electricity consumption and process heat; CO_2 utilization can enhance economic viability
Improvements in material recovery rate	limited opportunities for improving the recovery efficiency for conventional concentration processes; increase in energy and resource consumption	development of advanced concentration technologies
New and emerging production technologies	early stages of development, entailing high costs; effective mitigation for energy-intensive emerging technologies requires access to low-carbon electricity	growing interest in innovations driven by the rising demand for LIB raw materials
Decarbonization of electricity consumption elsewhere	uneven global progress in power sector decarbonization	–
CDR	unclear role of CDR and large uncertainty on their deployment due to still-low TRL and lack of policy framework	enhanced rock weathering using waste from mining and concentration activities

sources often requires overbuilding generation and storage capacity, further escalating costs.[100] The economic viability of electrifying process heat, haul trucks, processing equipment, and hydrogen use also heavily depends on electricity prices.[21] For example, even with cheap on-site renewable power generation and BESS, electrified heat remains more expensive than natural gas-based heat.[102] Substantial upfront investment is also expected for adopting technologies such as heat pumps, electric furnaces, electric haul trucks, electrolytic hydrogen, or emerging production technologies. Moreover, the decarbonization of the chemical industry would also raise reagent prices, potentially impacting battery-grade raw material prices.

Despite these challenges, future trends present promising opportunities to enhance the economic feasibility of decarbonization. Notably, the costs of BESS are projected to decline drastically, with the IEA forecasting a 40% reduction in the upfront costs of utility-scale BESS by 2030.[103] Shifting to flexible operation modes offers further cost-saving opportunities by reducing the need for expensive energy storage capacity.[100] Regulatory measures will also play a role. For example, the EU Battery Regulation sets that a carbon footprint declaration shall be drawn up for a number of battery types (e.g., LIBs), presumably by early 2025.[104] By early 2028, the EU will additionally prevent entry into the market for those batteries whose carbon footprint is larger than a (still to be defined) "maximum threshold." One may expect that these regulatory constraints may incentivize the adoption of decarbonization strategies across the LIB raw materials supply chain and beyond. Moreover, the role of a carbon tax in facilitating the decarbonization of LIB raw materials warrants further research. While carbon pricing has proven effective in reducing industry GHG emissions,[105] previous studies suggest that energy transition metals may be less affected by a global carbon tax compared with sectors like energy or agriculture, owing to the high economic value of metal commodities per unit of carbon emission.[106] Nonetheless, a global carbon tax could potentially accelerate the shift toward EVs and increase demand for battery-grade raw materials, underscoring the need for adopting decarbonization strategies.

MITIGATION POTENTIAL, DECOUPLING LEVEL, AND TRADE OFFS OF A DECARBONIZATION SCENARIO

Here, we quantitatively assess a decarbonization scenario for the analyzed production routes that involves transitioning to renewable electricity and electrifying heat across the LIB raw materials supply chain. This scenario considers the adoption of the following strategies: (1) switch to renewable electricity for on-site operations, (2) improvements in energy efficiency, (3) electrification of process heat, (4) switch to renewable electricity and electrified heat in the chemical industry, and (5) switch to renewable electricity elsewhere in the global economy. Wind power paired with a BESS serves as the representative for dispatchable supply of renewable electricity in this scenario, and it was further assumed that electrified heat is also powered by wind energy (see experimental procedures for further modeling details).

Combined, these measures could reduce GHG emissions intensity by 53% to 86%, depending on the commodity and production route (Figure 4). Natural graphite and synthetic graphite see the most substantial reduction due to electricity's large contribution to their life cycle GHG emissions. By contrast, the lowest reductions are observed for cobalt sulfate and nickel sulfate, attributed to a higher relevance of other emission sources not addressed in our decarbonization scenario, such as diesel use in mining (cobalt sulfate) and process emissions (nickel sulfate). It is noteworthy that the decarbonization of the chemical industry reduces GHG emissions from lithium hydroxide production by only 9%–10%, despite a high/medium mitigation potential being identified in Figure 3. This discrepancy arises because the production of the key reagent for lithium hydroxide, quicklime, generates a large amount of CO_2 during limestone decomposition that is not alleviated by a switch to renewable electricity and electrified heat. CCS is often considered the main option for decarbonizing such hard-to-abate chemicals, which has not been considered in our scenario.[107]

The estimated mitigation potentials are affected by uncertainty in the LCI data, which should not be overlooked. To illustrate this, we conducted a Monte Carlo analysis to quantify the propagation of uncertainty related to LCI data (see experimental procedures for methodological details). Cobalt sulfate and nickel sulfate exhibit the highest uncertainty, with the uncertainty range for mitigation potential (based on the 5th–95th percentiles) spanning 20%–68% and 34%–75%, respectively (Figure S3).

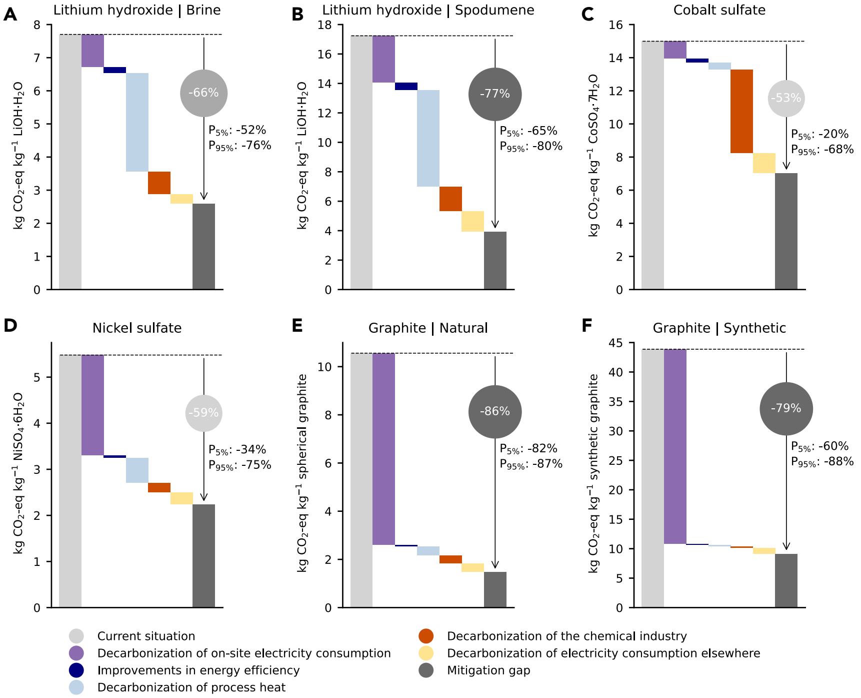

Figure 4. Mitigation potential of a decarbonization scenario for battery-grade raw materials involving the transition to renewable electricity and heat electrification across their supply chain

"Decarbonization of on-site electricity consumption" is achieved by utilizing wind power paired with a BESS. "Improvements in energy efficiency" assumes a 5% reduction in overall energy consumption. "Decarbonization of process heat" is achieved through electrified heat powered by wind energy. Decarbonization of the chemical industry encompasses the use of wind power and electrified heat (also based on wind energy). Decarbonization of electricity consumption elsewhere entails transitioning to wind power for all remaining activities within the global economy. $P_{5\%}$ and $P_{95\%}$ indicate the 5th–95th percentiles from the Monte Carlo uncertainty analysis (full results are presented in Figure S3).

This large uncertainty is primarily due to the quality of the available LCI data, including relatively outdated data for cobalt sulfate and aggregated data for various production routes for nickel sulfate.[41,43] Synthetic graphite also shows substantial uncertainty (60%–88%), despite using a current dataset primarily based on industry data. This uncertainty primarily arises from the reliance on expert assumptions and estimates in part of the LCI dataset,[44] coupled with variability in GHG emissions associated with the electricity mix. For lithium hydroxide, the uncertainty range is 52%–76% for brines and 65%–80% for spodumene, whereas natural graphite displays significantly lower uncertainty (82%–87%). Overall, these findings highlight that mitigation potentials are highly variable depending on the LCI data, underscoring the importance of high-quality data in developing decarbonization strategies.

We next assess the decoupling ratio, defined as the ratio between the demand growth factor by 2030 and the achievable GHG emissions intensity reduction rate after implementing the decarbonization scenario (Figure 5).[108] The decoupling ratio provides a measure of how the change in demand compares to the change in GHG emissions intensity. A ratio exceeding one indicates that demand is growing faster than the GHG emissions intensity is being reduced, implying an increase in total emissions by 2030. In the Stated Policies Scenario (STEPS), which assumes existing policies only, all commodities but brine-based lithium hydroxide exhibit a decoupling ratio below one, suggesting that achieving absolute decoupling of

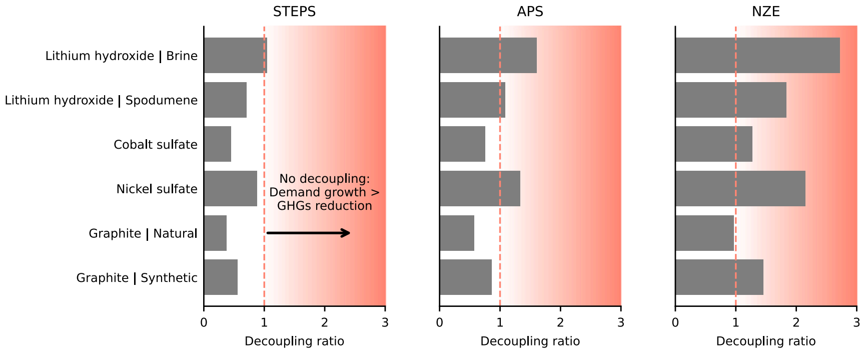

Figure 5. Decoupling ratio between EV's mineral demand by 2030 in the IEA's scenarios and the GHG emissions intensity of associated LIB raw materials considering the hypothetical decarbonization scenario

GHG emissions intensity of LIB raw materials after the adoption of the decarbonization strategies outlined in Figure 4. A ratio exceeding one indicates that demand is growing faster than the reduction in GHG emissions intensity, indicating that achieving absolute decoupling of GHG emissions from demand may not be feasible. STEPS, Stated Policies Scenario; APS, Announced Pledges Scenario; NZE, Net-Zero Emissions by 2050 Scenario.

GHG emissions from demand may be possible. However, in the more ambitious NZE scenario, the increase in demand largely surpasses the mitigation potential for all commodities except natural graphite. Focusing on lithium, its demand for EVs is projected to increase 8.6-fold by 2030 in the NZE scenario. To counterbalance this increase and achieve absolute decoupling between GHG emissions and demand, the GHG emissions intensity of lithium hydroxide production must be reduced by 88% by 2030 (Table S4). Our decarbonization scenario achieved only a 66% and 77% reduction for spodumene- and brine-based routes, respectively (Figure 4). Consequently, additional efforts are required to further reduce GHG emissions by 11%–22%. This gap is about 22% for nickel sulfate, 10% for cobalt sulfate, and 7% for synthetic graphite.

These results suggest that even with a complete transition to wind power and electrified heat throughout the supply chain, meeting future demand in a net-zero scenario is likely to lead to increased emissions from LIB raw materials production. This finding is significant, as residual GHG emissions from industry are frequently identified as a major obstacle to achieving the target of limiting global warming to well below 2°C.[109,110] It is important to note that our analysis assumes the projected demand by 2030 is met by the decarbonized production routes we have analyzed here. In reality, additional production routes with varying GHG emissions intensities are likely to contribute to satisfying this demand. Moreover, additional decarbonization strategies, such as adopting low-carbon haul trucks or addressing process emissions, could further reduce GHG emissions. However, the mitigation potential presented in Figure 4 is already highly ambitious, based on a supply chain entirely powered by wind energy. The mitigation gap that persists even under these optimistic conditions underscores the challenges in decarbonizing LIB raw materials production.

It is equally important to acknowledge that decarbonization can have environmental implications beyond climate change. Transitioning from fossil fuels to wind power and electrified heat substantially reduces other impacts such as acidification, eutrophication, photochemical oxidant production, particulate matter formation, and non-renewable energy use (see Figure S4). However, this shift may also introduce trade-offs, such as increased human toxicity and mineral and metal resource use. These trade-offs are associated with the deployment of wind turbines and batteries across the LIB raw materials supply chain. For example, minerals and metals resource use for synthetic graphite production could increase by a factor of 5.7 due to the high electricity demand. The large-scale adoption of other decarbonization strategies—such as electric haul trucks, electrification of processing equipment, and the use of hydrogen as a fuel or reducing agent—will further increase the requirement for renewable electricity and battery storage, thereby amplifying these trade-offs. Moreover, producing hydrogen via water electrolysis requires substantial water,[111] which would further exacerbate freshwater consumption of mining activities, already a concern at local scales,[112] particularly for lithium extraction from brines.[113] Finally, decarbonizing the supply chain of LIB raw materials will also increase the global demand for various raw materials, including LIB raw materials themselves, neodymium and dysprosium for wind turbines, tellurium, silver, copper, or aluminum for PV panels, or platinum and iridium used in electrolyzers.[114]

OUTLOOK AND FUTURE RESEARCH

In this study, we analyzed strategies for reducing GHG emissions from the production of battery-grade lithium hydroxide, cobalt sulfate, nickel sulfate, natural graphite, and synthetic graphite. We derived these strategies, their mitigation potential, and challenges from an analysis of the principal production routes. While these routes dominate the current market, it is crucial to acknowledge that alternative routes employing different resources and technologies may present distinct emission profiles, leading to different decarbonization priorities and challenges. For example, there is a continuous shift toward nickel laterite ores, projected to account for 90% of global supply by 2040.[33] This trend goes along an increasing role of HPAL of laterite ores, from 24% of global nickel sulfate production in 2020 to 42% by 2030.[115] Prior research on the Indonesian HPAL route has shown that GHG emissions mainly stem from on-site electricity consumption (36%), reagents consumption (25%), process heat (22%), and process emissions (13%).[116] These findings align with our analysis of the 2020 technology mix, although the importance of decarbonizing reagents production will become more pronounced with increased HPAL use. A potential rise in laterite ore processing via rotary kiln electric furnace may also shift decarbonization priorities toward low-carbon electricity and alternative reducing agents.[116] Future research could leverage the methodology used in this study to develop tailored decarbonization blueprints for alternative production routes and other raw materials. Similarly, future work can focus on developing blueprints for mitigating other impacts, such as water consumption or biodiversity loss, thus advancing sustainability efforts in the LIB raw materials industry.

Shifting to low-carbon electricity is fundamental to decarbonizing LIB raw materials supply chain. However, renewable electricity presents significant challenges concerning the variability and intermittency of renewable energy sources. Therefore, the decarbonization scenario quantified in this study is highly illustrative as it envisions the LIB raw materials supply chain and chemical industry being fully powered by wind energy with battery storage for dispatchability. In addition to techno-economic challenges, successful implementation of this scenario requires careful consideration of the interdependence among the mining and metal and chemical sectors. Addressing fundamental practical questions, such as determining implementation responsibilities, is crucial in this context, with regulators potentially playing a pivotal role.

One critical finding from our decarbonization scenario is that the prospects of reducing total GHG emissions from LIB raw materials production in the future encounters a major challenge—the skyrocketing demand. Achieving absolute decoupling of GHG emissions from demand growth will likely require additional measures beyond the transition to renewable electricity and electrified process heat. Further research is essential, including, but not limited to, quantifying remaining energy savings potential, identifying optimal relocation of refining capacities, and advancing innovations such as battery systems for large haul trucks, low-energy reagent regeneration methods, establishing hydrogen or biomass as reducing agent, and new and emerging production technologies. However, even with this portfolio of strategies, achieving net-zero emissions in LIB raw materials production might not be feasible. Residual emissions that cannot be mitigated would need to be compensated via CDR from the atmosphere. For example, a net-zero roadmap for copper and nickel mining determined that 10% of 2020 industry's GHG emissions need to be compensated via CDR to reach carbon-neutrality.[16]

In this study, we focused exclusively on the decarbonization of current primary production processes. Recently, unconventional production routes have emerged as appealing, such as geothermal brines or deep sea mining. For example, a prior LCA reported that producing cobalt sulfate and nickel sulfate from deep sea polymetallic nodules could result in 80% and 30% lower life cycle GHG emissions, respectively, compared with conventional routes.[117] However, these routes are in an early stage of development. Acting on the demand for raw materials through the type of clean energy technologies deployed and their material intensity could be more relevant. For example, the emergence of post-LIB chemistries, such as sodium-ion batteries, lithium-sulfur batteries, or solid-state batteries, may mitigate the demand for lithium and cobalt.[118] Strategies like using smaller vehicles or extending the lifetime of batteries can further contribute to reducing demand for LIB raw materials.[119] Recycling LIBs emerges as a promising pathway for reducing mining while concurrently delivering battery-grade raw materials with lower GHG emissions. For example, a previous study determined that recycling has the potential to reduce 20%–23% of the demand for lithium by 2050, 26%–44% for cobalt, and 22%–38% for nickel.[120] Similarly, secondary production of cobalt sulfate and nickel sulfate has been shown to yield GHG emissions intensity 74% and 57% lower compared with primary production.[121] Therefore, future research could enrich the decarbonization blueprint outlined in this study by incorporating demand-side actions and quantifying their mitigation potential. Building upon this extended portfolio, efforts could be directed toward proposing optimal decarbonization pathways for LIB raw materials that are both technically and economically feasible.

EXPERIMENTAL PROCEDURES

Life cycle assessment

We performed an LCA of battery-grade raw materials production using publicly available LCI datasets (see Table 1). The goal of the LCA was 2-fold: to identify GHG emissions hotspots across the analyzed production routes and to assess the mitigation potential and environmental trade-offs of a decarbonization scenario. We applied an attributional LCA approach, modeling existing processes as they are based on average data (e.g., electricity consumption in China is provided by the Chinese electricity mix).[123] The functional unit was defined as producing 1 kg of battery-grade commodity (e.g., 1 kg of lithium hydroxide monohydrate [$\geq$97% $LiOH\cdot H2O$; $\geq$16% lithium]). The LCA adopts cradle-to-gate system boundaries, encompassing ore mining, concentration, and refining (system boundaries diagrams can be found in Figures S4–S9). For synthetic graphite, the system boundaries include calcined coke production, calcined coke milling, graphitization (including production of synthetic graphite blocks), micronization, and coating.[44] At each stage, all relevant inputs (e.g., energy, reagents, waste treatment services, etc.) and outputs (i.e., emissions to air, water, and soil) were quantified. LCIs for LIB raw materials production processes (i.e., foreground processes) were obtained from the literature (see Table 1), while LCIs for background processes (e.g., electricity mixes or reagents production) were obtained from the ecoinvent database v3.10, cut-off system model.[38] LCI modeling and LCA calculations were performed using Brightway, a Python-based open-source LCA software.[124] All LCI datasets are made available at Istrate et al.[122] in a format that can be directly imported into Brightway. For impact assessment, we used the 100-year global warming potentials (GWP100, including short-lived climate forcers) from the Intergovernmental Panel on Climate Change (IPCC) Sixth Assessment Report (AR6)[125] and the Environmental Footprint (EF) v3.1 method recommended by the European Commission's Joint Research Centre.[126] Note that the LCA results reported in Figure 2 may differ from the values reported in the original studies due to different versions of the ecoinvent background database or the combination of LCI data from different sources.

Qualitative assessment of the mitigation potential

In Figure 3, we categorize the mitigation potential of each decarbonization strategy for each production route as low, medium, or high. This categorization is based on a qualitative assessment of two key indicators: the significance of the emission source and the technology potential of the decarbonization strategy. The significance of the emission source (e.g., electricity consumption) is determined by its contribution to life cycle GHG emissions, categorized as low if it accounts for <10%, medium if it accounts for 10%–25%, and high if it accounts for >25%. These thresholds are informed by the results for the GHG emissions hotspots presented in Figure 2. Technology potential evaluates the feasibility and effectiveness of the decarbonization strategy in reducing emissions. It is considered low if the strategy relies on unproven or in the early stage of development technologies, which may take decades to deploy at scale with the associated risk of deployment failure.[20] A low technology potential is also assigned if the strategy requires improving technologies that are already optimized, offering limited room for further advancements. Medium technology potential is assigned to strategies employing demonstrated technologies expected to be deployed at scale in the near term (5 to 10 years). Finally, a high technology potential is assigned if the strategy involves proven, commercially deployed at-scale technologies that are capable of significantly reducing emissions. The evaluation of technology potentials is based on evidence from the literature, as detailed in Table S3. It is important to note that this assessment is largely based on expert judgment and that the actual mitigation potential may depend on techno-economic, social, and regulatory factors. Moreover, while this assessment provides a framework, the criteria and thresholds could be refined and adapted for other production routes, depending on specific circumstances.

Decarbonization scenario modeling

In our decarbonization scenario, we assumed that renewable electricity is provided by 3 MW onshore wind turbines paired with lithium-iron phosphate (LFP) batteries. The BESS ensures dispatchability by storing electricity generated during periods of excess wind power and discharging as needed. The LCI dataset for the integrated wind-BESS system was normalized to 1 kWh generated over the system's lifetime, with wind turbines contributing 0.76 kWh and the BESS contributing 0.24 kWh. These proportions are based on the study of Das et al.,[127] which modeled a wind farm generating 90 TWh over 30 years, using 3 MW wind turbines with a 31% capacity factor (close to the global weighted average for onshore wind of 37%),[128] combined with LFP batteries with 500 MWh capacity that deliver 22 TWh over the same period. The LCI dataset for electricity generation from onshore wind turbine was obtained from the ecoinvent v3.10 database. For the BESS, the LCI data for supplying 1 kWh of electricity were modeled according to Krebs et al.,[129] assuming an overall efficiency of 90%, energy density of LFP of 0.11 kWh kg^{-1}, and 80% depth of discharge. The integrated wind-BESS system was assumed to provide electricity for all mining, concentration, refining, and reagent production processes, as well as for powering electrified process heat. It was also considered an electricity source for the decarbonization of electricity consumption elsewhere in the global economy.

For energy efficiency improvements, we assumed a 5% reduction in overall energy consumption based on the 0.5% annual reduction reported in Wyns and Khandekar.[18] and a 10-year timeframe. The 10-year timeframe was selected aligned with the short- to medium-term scope of this review. Process heat is decarbonized through electrification. We assumed that temperatures below 100°C (as required, e.g., in brine-based lithium hydroxide production) are supplied by industrial heat pumps, while high-temperature processes are electrified employing different types of electric furnaces with an electrical-to-thermal efficiency of 90%.[58] The LCI dataset for high-temperature electrified heat supply exclusively comprises the input of electricity. For reagent production, lacking specific data on distribution between low and high temperature for each reagent, we have assumed that all heat demand is high temperature and thus supplied by electric furnaces. This conservative approach to estimating mitigation potential does not consider the possible use of more efficient heat pumps for part of the heat demand.

The Monte Carlo sampling method (based on 1,000 iterations) was used to assess the robustness of the estimated mitigation potentials resulting from the propagation of uncertainty related to LCI data. In each iteration, LCI data values were randomly sampled according to their probability distributions, and life cycle GHG emissions were quantified. Uncertainty in the LCIs of background processes was included per the default in the ecoinvent database v3.10, which employs a semi-quantitative approach based on the pedigree matrix.[38] For the uncertainty in the LCI data of foreground processes, a triangular distribution was used when uncertainty ranges were provided in the original LCI data source (e.g., as for natural graphite).[11] In cases where uncertainty ranges were not available, a log-normal distribution was specified based on the pedigree matrix, similar to the approach applied in the ecoinvent database. The evaluation of the pedigree matrix is included with the LCI datasets in the supplemental information.

RESOURCE AVAILABILITY

Lead contact

Further information and requests for resources should be directed to and will be fulfilled by the lead contact, Robert Istrate (i.r.istrate@cml.leidenuniv.nl).

Materials availability

This study did not generate new, unique materials.

Data and code availability

All the data used in this study can be found from the references mentioned in the manuscript or experimental procedures. The datasets and Python code needed to reproduce the results in the manuscript are available at Istrate et al.[122]

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AUTHOR CONTRIBUTIONS

Conceptualization, R.I. and B.S.; methodology, R.I. and B.S.; investigation, R.I., A.B., and A.M.-F.; writing – original draft, all authors; visualization, R.I.; writing – review & editing, all authors.

DECLARATION OF INTERESTS

K.V. is employed by Northvolt, a battery developer and manufacturer. The information and views set out in this article are those of the authors and do not necessarily reflect the official opinion of the company.

SUPPLEMENTAL INFORMATION

Supplemental information can be found online at https://doi.org/10.1016/j.joule.2024.10.003.

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