

Lithium-ion Batteries: Green Solution or Grey Paradigm of the Modern World

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Article Info	ABSTRACT
Article history: Received , 04/03/2026 Revised , 01/06/2026 Accepted , 05/06/2026	The paper evaluates the impact of global lithium-ion battery (LIB) production on greenhouse gas (GHG) emissions, represented as CO_{2,eq}. In the context of the Paris Agreement and decarbonization, a conservative CtG-GtW (Cradle-to-Gate + Grid-to-Wheel) methodology is used to analyze the production cycle of the two dominant technologies: lithium-nickel-manganese-cobalt (NMC) and lithium-iron-phosphate (LFP). The study tracks the supply chain from raw material extraction for Li (spodumene, lepidolite, and brines), metal concentrate, and intercontinental shipment to China, through to final cell manufacturing, and battery shipment. Results show that the global production of 1.545 TWh generates a total mean of 288 Mt CO_{2,eq}, accounting to 0.8% of global fossil fuel emissions. The mean specific emissions are ~190 kg CO_{2,eq} kWh⁻¹, which is nearly seven times higher than the sustainability target of ~27 kg CO_{2,eq} kWh⁻¹ for the electric vehicles. By comparing diesel and EV-NMC and EV-LFP cars, emissions of EV become lower than those of diesel, after ~6.3 years in the EU, ~8 years in the USA, and globally practically never occur. The findings suggest that current lithium-based energy storage requires urgent decarbonization, lowering CO_{2,eq} emission for charging from the electric grid, or substitution by alternative systems, such as sodium-ion batteries, intensification of biofuel technologies, to align with global climate goals. The paper also, consider socio-ecological impact of Li-ion battery productions.
Keywords: Cradle-to-Gate; Grid-to-Wheel; CO_{2,eq} emission; Energy transition; Green agenda; Decarbonization.	

I. Introduction

Climate change is an extremely challenging factor for the normal functioning of civilization [1]. The use of non-renewable energy has emitted large quantities of various greenhouse gases (GHGs), with carbon dioxide contributing ~80% as one of the main gases [2-4]. According to the World Meteorological Organization (WMO), total carbon dioxide equivalent (CO_{2,eq}) emissions are estimated at 42 Gt in 2024 [5]. This includes fossil CO_{2,eq} emissions of 37.4 Gt, while the remainder comes from land-use changes (deforestation), which is 2.0% above the previous record set in 2023.

The energy sector produces the most greenhouse gas emissions, approximately 68%, of which 26% comes from electricity generation, 11% from industry, 15% from transport, 6% from buildings, and 10% from fuel

production. Emissions from industrial processes account for 9%, agriculture emits 11%, land-use change and forestry 7%, and 4% comes from waste and other sources [6].

According to the Statistical Review by the Energy Institute in London [7], 592 EJ exajoules (EJ) of energy (592×10^{18} J) were produced globally in 2024, which is equivalent to 165 petawatt hours, PWh (165×10^{15} Wh). Due to various losses, only ~ 110 to 125 PWh is usefully utilized. To understand the order of magnitude of energy production, if we convert 165 peta-hours to light-years, the value of this energy is hypothetically equal to $\sim 20,000$ billion (20×10^{12}) of light-years, or corresponding to more than 1500 times larger than the visible Universe!. For comparison, the Andromeda Galaxy (M31) is approximately 2.5 million light-years away from Earth. Unfortunately, of that amount, only $\sim 3\%$ is hydropower, 5.6% is renewable sources, 5.2% is nuclear, and all the rest are fossil fuels, as shown in Figure 1.

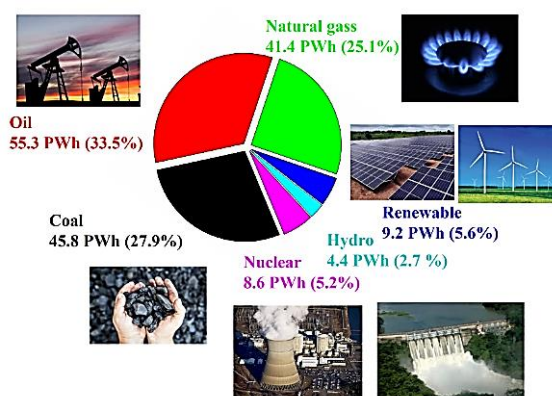


Figure 1. Total primary energy production in the world in 2024 (165 PWh, 1 PWh = 1×10^{12} kWh) [7].

To reduce greenhouse gases (GHG, usually expressed as equivalent carbon dioxide, $\text{CO}_{2,\text{eq}}$) emissions (so-called decarbonization), several agreements have been established, such as the Paris Agreement, a legally binding international treaty on climate change. It was adopted by 196 countries at the UN Climate Change Conference (COP21) in Paris on December 12, 2015, and approved on November 4, 2016. In 2023, the first "global stock take" of the world's efforts under the Paris Agreement concluded at COP28 with a decision on how to accelerate action in all areas – mitigation, adaptation, and finance – by 2030, including a call to governments to accelerate the transition from fossil fuels to renewable energy sources, mainly solar and wind energy, which includes the transition to electrochemical energy storage and electric vehicles. Unfortunately, energy harvesting from wind and solar sources possesses a latent "Mr. Hyde" aspect, characterized by the accumulation of non-recyclable components that persist in the environment for decades [8]. This includes the environmental burden of discarded solar panels, where the recovery of high-purity silicon, silver, and lead remains a significant industrial and ecological challenge.

In addition, the European Commission adopted a set of proposals, the European Green Deal, on December 11, 2019, to make EU climate, energy, transport, and tax policies fit for reducing net greenhouse gas emissions by at least 55% by 2030 compared to 1990 levels [9]. Also, European Union aims to be climate-neutral by 2050., meaning an economy with net-zero greenhouse gas emissions [10]. It should be noted that in 1990, emissions in what is today the European Union amounted to 5 Gt $\text{CO}_{2,\text{eq}}$ [11]. Total net greenhouse gas emissions in the EU were 2.91 Gt $\text{CO}_{2,\text{eq}}$ in 2023. This was a year-on-year reduction of about 9% and a total reduction of 37% compared to 1990 levels [12]. With a 37% decrease over 33 years, linear projection suggests the system will reach 100% depletion in about 90 years.

The green energy transition largely involves lithium-ion batteries as the most advanced electrical energy storage at this time for large, medium, and small-scale storage, as well as an increased number of electric vehicles [13], and many politicians claimed as zero-carbon transition. In overall, concerning GHG emissions for LIB battery manufacturing, many studies summarize reports range from 36 to 312 kg $\text{CO}_{2,\text{eq}}$ kWh^{-1} of the cell capacity [14] or, similarly, from 38 to 356 kg $\text{CO}_{2,\text{eq}}$ kWh^{-1} [15], and from 37 to 313.0 kg $\text{CO}_{2,\text{eq}}$ kWh^{-1} [16] as summarized by [17]. A conservative estimate of emissions from Li-ion battery production is approximately 150–200 kg $\text{CO}_{2,\text{eq}}$ kWh^{-1} [18], although many recent studies show a median of 100–110 kg $\text{CO}_{2,\text{eq}}$ kWh^{-1} , where "Well-to-wheel" emissions are dominantly derived from raw materials and anode and cathode active materials production. The average target value representing the maximum tolerable impact on climate and climate change is estimated to be ~ 27 kg $\text{CO}_{2,\text{eq}}$ kWh^{-1} for electric cars [19]. Such high dissipation of reported data could be connected with different models and input parameters used in Life Cycle Assessments (LCA) or Monte Carlo simulations.

Monte Carlo simulations are increasingly used in battery Life Cycle Assessment (LCA) studies. While they can help explore uncertainty, their outputs depend heavily on the quality of input assumptions. In many cases,

the connection to actual plant-level energy consumption becomes less transparent. This study therefore adopts a fully deterministic, first-principles approach instead of relying on stochastic methods. Unfortunately, many highly cited papers that use "Life Cycle Assessment" (LCA) methodology or Monte Carlo simulations with predictions to 2050 usually focus on the price and second life of EV batteries [20, 21] and do not consider some very important points. In some cases, stochastic models may unintentionally obscure empirical gaps or inconsistencies in inventory data. However, reality is different. This study prioritizes physical accountability by using a transparent, process-based energy and mass balance, rather than relying on statistical distributions. The current geopolitical volatility underscores the delusion of a monolithic energy transition; energy security is a prerequisite for environmental sustainability, and a deterministic audit proves that local resource integrity must precede theoretical carbon targets. We are witnesses that the Ukrainian-Russian war has completely changed the EU energy infrastructure. As an alternative to natural gas, Liquefied Natural Gas (LNG) has become in many counties substitutions for Russian natural gas. The United States is the world's largest producer and exporter of LNG, but the emission of $\text{CO}_{2,\text{eq}}$ before long distance transport and use, is very high, $\sim 0.25 \text{ t}$ of $\text{CO}_{2,\text{eq}}$ per ton of LNG. The recent attack by the USA on Venezuela, the USA and Israel on Iran, and Iran's attack on the surrounding countries, could dramatically change the global oil supply chain. Therefore, the geopolitical situations in today's world can change as fast as the "blink of the eye."

To claim a truly green future, we must stop ignoring the massive environmental "overheads" that current industrial models conveniently omit. The reality is that producing high-purity metal precursors and lithium compounds requires staggering amounts of freshwater, often in regions already facing acute water stress. This "hydrological debt," caused by the endless washing and chemical refining of ores into battery-grade sulfates, is a direct threat to regional stability [22]. Recently, in the Republic of Serbia, a multinational company, with the support of the Government, due to the possible exploitation of lithium and boron ore, Jadarite, $\text{LiNaSiB}_3\text{O}_7(\text{OH})$, on fertile land near the floodplains of the Jadar, Korenita and Drina rivers and the possibility of contamination of large underground reserves of drinking water, mass protests broke out among citizens, which led to the probable postponement of the opening of the mine and processing plant.

Furthermore, we must account for the loss of carbon sequestration whenever forests and, especially, rainforests are cleared for mining. This is not just an energy loss; it is a permanent biological deficit, and probably will lead to the extinction of many species. When we add the material weight of cell casings, copper busbars, and complex internal wiring, it becomes clear that the current narrative is incomplete. Until every liter of water and every acre of lost forest is included in the balance sheet, beyond the reach of simple Monte Carlo simulations, the promise of "clean" energy storage remains a misleading abstraction. True sustainability requires facing these hard numbers, not hiding behind partial data.

The prevailing narrative of a "clean energy" transition appears increasingly detached from the physical and ethical realities of the global supply chain. While Western policy targets focus on localized tailpipe emissions, a deterministic audit reveals a profound geographical displacement of ecological ruin. We are not eliminating environmental debt; we are merely outsourcing it to the most vulnerable biomes on the planet. The hydrological toll in arid lithium-mining regions is a silent catastrophe. For every metric ton of lithium produced, approximately 2 million liters of brine are pumped to the surface and lost to solar evaporation, irreversibly disrupting the hydrostatic equilibrium of ancestral aquifers. This is compounded by the direct consumption of an additional 50,000 liters of freshwater for chemical purification, water that is the primary lifeline for local agricultural communities and hosts a unique biodiversity, including several rare (endemic) plant and animal species, such as three of the world's six flamingo species [23, 24]. As these vital sources vanish, soil salinization and the collapse of regional ecosystems become the permanent price for temporary battery storage. This environmental degradation is inextricably linked to a broader humanitarian crisis. The "cobalt rush" in the Democratic Republic of Congo continues to rely on the unethical exploitation of child labor [25-27], while the aggressive expansion of nickel mining has catalyzed the destruction of Indonesian tropical rainforests [28, 29], leading to the extinction of endangered rare animal and plant species and the total loss of critical hotspots. If externalities are not fully captured in probabilistic LCA models, there is a risk that the energy transition may shift environmental burdens to vulnerable regions. A more explicit, deterministic accounting of ecological and social impacts is needed to avoid unintended consequences. We are rushing toward a solution that may prove as toxic as the problem it seeks to replace. It should also be mentioned that the mass production of especially NMC batteries provokes more and more fires. Due to the thermal runaway [30], we are witnessing that EV battery electric storage systems (BESS) are growing into potentially dangerous emitters of toxic chemicals during fire that is practically impossible to extinguish. The reasons for thermal runaway are different, but the rise of the temperature causes the spinel oxide cathode to release pure oxygen, and in the presence of flammable electrolyte, fire occurs inside the battery. As the pressure increases, a crack in the cell or even an explosion could occur. In addition to toxic solvent contamination, hydrogen fluoride (HF) generation could range from 4.6 to 22 g kWh^{-1} , as a highly toxic gas [31]. Among EV and

BESS, there are numerous fires recorded, in homes or buildings, for example, provoked by a mobile phone or small scooters, fires in the waste recycling facilities, and waste landfills, where a single small battery could provoke a huge fire. In addition, in airplanes, accidents are very common.

Therefore, I pose the question, does “The butterfly effect” remain only a phrase from chaos theory, or does it truly exist [32]?

The primary goal of this paper is to perform a deterministic, first-principles audit of the global Cradle-to-Gate energy debt for the entire 1.545 TWh [33] battery-manufacturing sector, based on reliable 2024 industrial and literature data. By replacing opaque Monte Carlo simulations with a transparent “pencil-and-calculator” mass-and-energy balance, this study establishes a rigorous “min-max” range for the cumulative energy demand required for precursor synthesis and industrial assembly. The resulting analysis provides a foundational dataset for estimating total global GHG emissions ($\text{CO}_{2,\text{eq}}$) based on primary energy consumption, exposing the systemic gap between theoretical climate targets and the physical reality of battery production. By employing a rigorous, deterministic mass-and-energy balance, it is prioritized the first principles of chemical engineering are prioritized over statistical speculation. This ensures that every kilogram of $\text{CO}_{2,\text{eq}}$ calculated is a direct consequence of physical energy expenditures, providing a transparent and auditable “summa summarum” that remains immune to the systemic biases of probabilistic modeling.

Therefore, the main goal of this paper is to estimate the total global $\text{CO}_{2,\text{eq}}$ emissions from ore production, shipment, ore processing, and the production of major battery materials and components, up to the final assembly of the batteries and their transport.

II. Research Methodology

The starting point is the recognition that 1.545 TWh [33, 34, 35] was produced globally in 2024, while China dominated the global lithium-ion battery market, with approximately 76%–79% with production reaching ~1.2 TWh. To provide a conservative and data-transparent estimate, this study assumes a total Chinese production of 1.2 TWh [33–35]. Given that China is significantly shifting to the production of lithium-iron-phosphate (LFP) batteries as cheaper and safer systems (with a market share of over 65%) compared to lithium-nickel-manganese-cobalt (NMC), and considering that the specific energy of NMC batteries are 250–300 Wh/kg, while LFP batteries achieve 120–160 Wh/kg (48% to 53% compared to NMC), the analysis is based on the production of ~70% (0.84 TWh) of LFP systems, and ~30% (0.36 TWh) of NMC systems. For the total balance, to account for non-active materials, see Table 4, and unmodelled logistical steps (*e.g.*, and local ground ore and different compounds transport by rail/truck, production of the cell casings, production of the battery pack with cooling system, copper busbars, binders, separators, transports of labors and material transport inside manufacturer facilities, *etc.*), a 15% uncertainty correction factor (UCF) was applied to the total emission [36, 37, 38].

Finally, to evaluate the impact of global production outside China, the remaining 345 GWh (30% share) was added. This non-Chinese share was assumed to be dominated by NMC chemistry (70% NMC or 241 GWh, and 30% LFP or 104 GWh) due to different regional supply chains accounting the different emissions resulting from the use of different energy mix not based solely on fossil fuels, a 20% share (instead of 30%), and 15% uncertainty correction factor (UCF) was also applied to the total global emission.

In Life Cycle Assessment, several inherent limitations are recognized, including data uncertainty, system boundary truncation are often excluded, subjective allocation choices, and the inability to quantify confidence in deterministic results. Additionally, many LCA studies rely on unreliable or outdated databases such as Ecoinvent or GaBi, in which generic emission factors are used that do not reflect specific supply chains, regional conditions, or technological improvements. Monte Carlo simulation is frequently employed to address these limitations, as input distributions are propagated through thousands of iterations and confidence intervals are generated. However, the usefulness of Monte Carlo simulations is limited by input quality, correlation assumptions, and the transparency of underlying databases. This study complements such approaches by providing a fully disaggregated, process-based model with clearly defined system boundaries in which the uncertainty correction factor (UCF) was explicitly included. Unlike aggregated LCA studies in which these components are hidden in generic factors from sometimes unreliable databases, and unlike Monte Carlo simulations in which statistical complexity is added without process transparency. The methodology used in this study provides deterministic transparency with clearly defined system boundaries. Direct sensitivity analysis on each parameter is enabled, full replicability is ensured, and no hidden correlations or distribution assumptions are required. Consequently, a conservative, transparent, and auditable CtG-GtW (Cradle-to-Gate + Grid-to-Wheel) methodology for $\text{CO}_{2,\text{eq}}$ emission estimate was produced by this study. Finally, the sensitivity analysis was performed, varying the LFP/NMC ratio from 60%, 70%, 85% to 100%, and the UCF from 5%, 10% and 15%.

The algorithm of $\text{CO}_{2,\text{eq}}$ estimations was given in Figure 2.

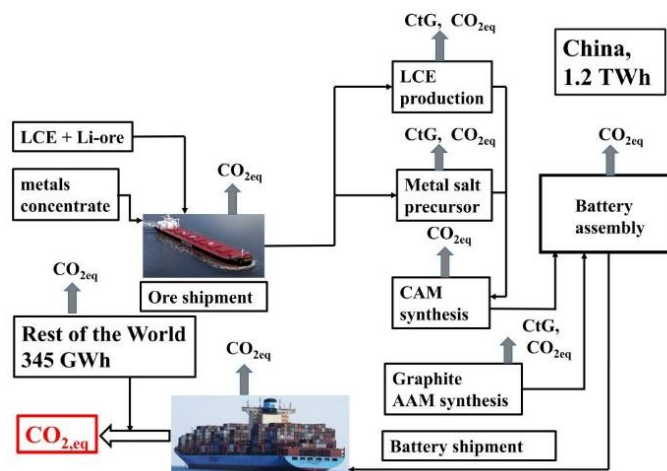


Figure 2. The algorithm of the $\text{CO}_{2,\text{eq}}$ emission estimation for battery production in China and World.

III. Results

III.1. Lithium-ion Batteries

The first commercial Sony lithium-ion battery from 1991 had a specific energy of only 80 Wh/kg [39], which today represents barely one-fourth of the capacity of the most modern NMC systems. Subsequently, billions of dollars were invested in development to reach a specific energy of ~ 300 Wh/kg. The construction of lithium batteries is relatively simple, as can be seen in Figure 3. It consists of a housing containing a positive electrode (cathode) based on $\text{LiCo}_x\text{Mn}_y\text{Ni}_z\text{O}_2$, $x+y+z=1$ (NMC) or LiFePO_4 (LFP). The negative electrode (anode) is based on processed or synthetic graphite. The electrolyte is based on aprotic organic solvents with the addition of mainly LiPF_6 as a conductive salt. A separator based on porous polypropylene or polyethylene prevents a short circuit between the positive and negative electrodes. The operating principle is relatively simple: during charging, lithium deintercalates from the positive electrode and intercalates between the graphite layers, and during discharging, the reverse reactions take place. It is important to note that LIB uses anodic current collectors made from Cu (whose price rises constantly, at 27.03.2026 was over 12,000 \$ per ton) [40], which can corrode and provoke short circuit, thermal runaway, and self-ignition, because aluminum with lithium forms an irreversible alloy.

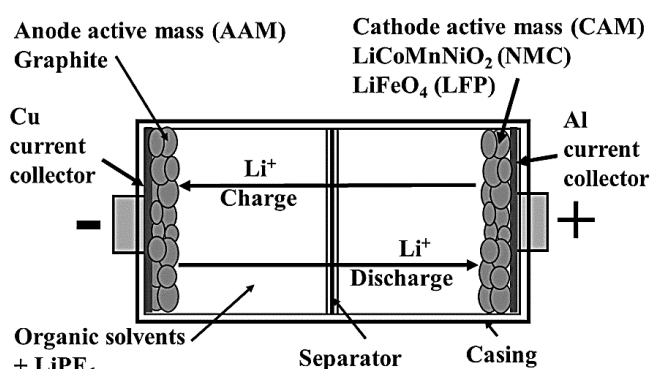


Figure 3. Simplified construction of a lithium-ion battery

For the lithium-ion batteries with a capacity of 1 kWh, the amount of lithium carbonate equivalent (LCE) depends on the cathode chemistry. It is interesting that in NMC systems (LiNiMnCoO_2), with a typical battery mass of about 4-5 kg, the lithium carbonate (LCE) content is approximately 180-220 g, corresponding to a share of about 4.5 wt.%. In LFP systems (LiFePO_4), the battery mass is slightly higher, about 6-7 kg per kWh, while

the lithium content is lower, approximately 140–180 g of lithium carbonate, or only about 2–3 wt.% of the total mass. This indicates that although mainly lithium is discussed in a controversy of the supply chain, practically >95% of the battery mass depends on other materials and strategic raw materials. In addition to these basic parts, a whole range of materials is used in battery manufacturing, see Table 3, for which, due to unreliable data, it was not possible to accurately estimate $\text{CO}_{2,\text{eq}}$ emissions.

III.2. Lithium Sources, Other Materials, and $\text{CO}_{2,\text{eq}}$ Emissions

Various lithium compounds are used for the production of lithium batteries: lithium carbonate (Li_2CO_3), lithium oxide (Li_2O), and lithium hydroxide (LiOH). In the literature, the Li-equivalent is sometimes misinterpreted, so the conversion factors are shown in Table 1.

Table 1. Conversion factors for different Li compounds based on 1 t of metallic Li.

	Li_2CO_3	Li_2O	LiOH	$\text{LiOH} \times \text{H}_2\text{O}$
Li / 1 t	5.32 t	2.15 t	3.45 t	6.05 t

The main sources of lithium are hard lithium ores (spodumene, lepidolite, zinnwaldite, petalite, etc.) [41–43] and brine lakes, especially in the "lithium triangle," an Andean region rich in lithium reserves that spans the borders of Argentina, Bolivia, and Chile [44]. In 2024 total lithium carbonate equivalent (LCE) production reached 1,323,000 t, as shown in Table 2, with 418,000 t LCE from brines, 688,000 t LCE from spodumene ($\text{LiAl}(\text{SiO}_3)_2$), and 217,000 t from lepidolite ($\text{K}(\text{Li},\text{Al},\text{Rb})_2(\text{Al},\text{Si})_4\text{O}_{10}(\text{F},\text{OH})_2$) [45], as well as smaller amounts from zinnwaldite ($\text{KLiFeAl}(\text{AlSi}_3)\text{O}_{10}(\text{OH},\text{F})_2$), petalite ($\text{LiAlSi}_4\text{O}_{10}$).

According to InfoLink's global lithium-ion battery supply chain database [46], global demand for lithium carbonate equivalent reached 1,189,000 t in 2024, which includes 759,000 t for automotive lithium-ion batteries, 119,000 t for energy storage batteries, and 311,000 t for lithium-ion batteries for consumer electronics, ceramics, and glass-ceramics. Consequently, production surplus is approximately 10% higher than demand, which caused a drastic drop in the price of lithium carbonate on the world market.

Table 2. Total production of lithium carbonate equivalent (LCE) by country in 2024 [45, 46].

Country	LCA / t	%
Australia	468,160	35.4
Chile	260,680	19.7
China	218,120	16.5
Zimbabwe	117,040	8.8
Argentina	95,760	7.2
Brazil	53,200	4.0
Canada	22,876	1.7
Namibia	14,364	1.1
Portugal	2,022	0.2
The rest of the world	47,900	3.6
Total	1,323,000	100

Lithium carbonate production processes from ores are very complex and include multiple technological phases, such as mining, concentration, thermal treatment, leaching with concentrated acids, etc. [41, 47, 48]. Production of LCE from brines is much simpler than from hard rock. However, for the brine concentration via evaporation, an average of 500,000 L of water must evaporate per ton of lithium carbonate produced, along with a non-negligible volume of fresh water of about 50,000 L per ton of LCE in various production or purification stages [48]. The mineral composition of several brine lakes can be found in reference [49], where lithium content varies between 0.23 and 1.57 g L^{-1} . In brine lakes, an intense exchange of $\text{CO}_{2,\text{eq}}$ between air and water occurs, and they are important reservoirs of dissolved inorganic carbon, with average concentrations of 7.7 mmol L^{-1} (~3 g L^{-1}), which is about 10 to 15 times higher than typical concentrations in freshwater lakes and rivers [49–51]. $\text{CO}_{2,\text{eq}}$ emissions per 1 t of hard rock and/or brine vary during processing between 5 t and 25 t, as seen in Table 3 [41, 48, 52–54]. Due to China dominating the global lithium-ion battery market, with approximately 76%–79%, for the calculations is accounted that China consume 77% or ~1 million tons of LCA.

Table 3. $\text{CO}_{2,\text{eq}}$ emissions per 1 t of lithium source during processing, and the total emission.

Lithium source	Chemical formula	$m(\text{CO}_{2,\text{eq}})$, t, per 1 t of LCE	Annual production of LCE, t	Estimated emission per year, $m(\text{CO}_{2,\text{eq}})$, Mt
Spodumene	$\text{LiAl}(\text{SiO}_3)_2$	~17-22	688,000	11.7-15.1
Lepidolite	$[\text{K}(\text{Li},\text{Al},\text{Rb})_2(\text{Al},\text{Si})_4\text{O}_{10}(\text{F},\text{OH})_2]$	~12	217,000	2.6
Salt lakes (brine)	Li , 0.23-1.57 g L ⁻¹	~5-25	418,000	2.1-10.4
Total LCA			1,323,000	16.4-28.1
China's contribution			~1,000,000	12.7-21.7

Because battery-grade Li_2CO_3 has different purity requirements ranging from 99.5% to 99.8%, there is a significant range of impurities within that scope; for example, 150–650 ppm of sodium (Na^+), 30–300 ppm of chloride (Cl^-), and 150–500 ppm of sulfate (SO_4^{2-}). Nevertheless, for some elements, smaller variations are allowed. For example, iron (Fe) is extremely harmful in battery production, with a tolerance of only 2–10 ppm. Similarly, metals such as aluminum (Al), lead (Pb), and zinc (Zn) have very strictly limited concentrations [55]. Australia, as the largest producer of lithium, harvests spodumene concentrate containing up to 6% Li_2O [56]. Calculated, 6% Li_2O corresponds to 14.8% LCE, so the mass of spodumene concentrate is estimated at 3.2 Mt. Australia transports most of the concentrate to China for further processing into lithium carbonate. Some interesting transport routes from Australia to China can be seen in Figure 2 in citation [56]. Chile and Argentina produce practically battery-grade LCE in the amount of 360 kt from brines, but most of the product is also transported to China [57]. Bolivia, as one of the countries with the largest lithium reserves in the world, estimated at 23 Mt (or 243 Mt LCE), produces only about 600 t due to political issues [58]. In Zimbabwe, which could have the largest lithium production in Africa at 117,040 t LCE annually (mainly from spodumene), a ban on the export of raw lithium ore and a 5% export tax on concentrate were introduced, requiring companies to open plants in the country and process ore into concentrate before export to encourage local employment and revenue [59]. Subsequently, China, despite its own large production, imports most of the raw materials to maintain a monopoly. It is necessary to estimate the mass of ores used for Li battery production to calculate $\text{CO}_{2,\text{eq}}$ emissions from transport ships. In 2023, China imported cobalt ores and concentrates amounting to 220,000 t of pure cobalt equivalent per year, mainly from the Democratic Republic of Congo [60], where questions regarding human rights violations, and particularly child labor, are still being raised [61]. Some Cu-Co ore enrichment is carried out in the DR Congo [62], but most cobalt concentrate is sent to China. About 28,000 t of cobalt was imported from Indonesia in 2023 [63], and ~1 Mt of refined nickel. The largest exporters of nickel ores and concentrates to China were the Philippines (39,687,100 t), the Ivory Coast (984,716 t), Guatemala (539,290 t), Brazil (19,813 t), and the European Union (13,803 t), while China produces 110,000 t [64]. China also imports 31,351,400 t of the manganese ores and concentrates from South Africa (14,624,900 t), Australia (5,245,460 t), Gabon (4,860,340 t), Ghana (3,153,920 t), and Brazil (1,236,360 t) [65]. Furthermore, in 2024, China increased iron ore imports by 4.9% compared to 2023 on 1.24 billion tons [66]. Currently, China mainly imports iron ore from Australia, Brazil, India, North Africa, and other countries, but Australia supplies 736 million tons, which is more than 60% of China's iron ore imports. For the fourth consecutive year, China recorded growth in the consumption of copper ores and concentrates, which rose by 0.6% to 28 million tons (Mt), while imports of unwrought copper amounted to 5.68 million tons in 2024 [67]. China imported copper ores and concentrates mainly from Chile (8,433,410 t) and Peru (7,248,330 t). In the whole of 2024, Chinese imports of aluminum ore and its concentrate amounted to 158.8 million tons, a 12.4% increase compared to the previous year [68]. Chinese imports of unwrought aluminum and aluminum products fell by 24.3% between January 2024 and January 2025 to 290,000 tons [69]. In 2024, the main source of China's aluminum ore imports was Guinea. Globally, lithium-ion batteries were mainly produced by companies from China, Japan, and South Korea. Driven by China's policy of strong development of electric vehicles and renewable energy storage [70], the scale of the Chinese lithium-ion battery industry began to grow rapidly from 2015, overtaking South Korea and Japan to become the global leader. In 2024, global lithium-ion battery production reached about 1.545 TWh [33]. In China, with a global market share exceeding 76% of world production in 2024, it reached ~1.2 TWh, [33, 34] for the $\text{CO}_{2,\text{eq}}$ is estimated on the production 30% (0.36 TWh) of lithium-nickel-manganese-cobalt (NMC), and 70% (0.84 TWh) of lithium-iron-phosphate (LFP) [33-34]. For batteries based on NMC chemistry, there is a visible transition to cathode materials with higher nickel content (which is also a strategic raw material) and lower content of expensive cobalt in the form of $\text{LiNi}_{0.5}\text{Mn}_{0.3}\text{Co}_{0.2}\text{O}_2$ (NMC532), $\text{LiNi}_{0.6}\text{Mn}_{0.2}\text{Co}_{0.2}\text{O}_2$ (NMC622), and $\text{LiNi}_{0.8}\text{Mn}_{0.1}\text{Co}_{0.1}\text{O}_2$ (NMC811), from the basic NMC structure,

$\text{LiNi}_{1/3}\text{Mn}_{1/3}\text{Co}_{1/3}\text{O}_2$ (NMC333) [71]. Table 4 shows the material composition for different NMC LIB systems per 1 kWh.

Table 4. Material composition for different NMC LIBs per 1 kWh, calculated from ref. [71].

LIB component, kg	NMC111	NMC532	NMC622	NMC811
Active material	1.78	1.72	1.50	1.27
Graphite/carbon	0.94	0.92	0.92	0.99
Binder	0.08	0.05	0.05	0.09
Copper	0.34	0.32	0.29	0.29
Wrought aluminum	0.80	0.77	0.74	0.74
Electrolyte: LiPF_6	0.06	0.06	0.06	0.06
Ethylene carbonate	0.18	0.16	0.16	0.16
Dimethyl carbonate	0.18	0.16	0.16	0.16
Polypropylene	0.03	0.04	0.03	0.03
Polyethylene	0.01	0.01	0.01	0.01
Polyethylene terephthalate	0.01	0.01	0.01	0.01
Steel	0.03	0.03	0.02	0.03
Thermal insulation	0.02	0.02	0.01	0.01
Glycol	0.12	0.12	0.12	0.12
Electronic parts (BMS)	0.08	0.08	0.08	0.08
Total (kg)	4.66	4.46	4.16	4.04

Based on Table 4, it can be calculated how much different metals are contained in NMC batteries per kWh (Table 5).

Table 5. Materials content in different NMC batteries per kWh.

NMC component, kg	NMC111	NMC532	NMC622	NMC811
CAM	1.78	1.72	1.50	1.27
Ni	0.59	0.86	0.9	1.02
Co	0.59	0.34	0.3	0.13
Mn	0.59	0.52	0.3	0.13
AAM Graphite/carbon	0.94	0.92	0.92	0.99
Electrolute	0,38	0,38	0,38	0,38
Cu-current collectors	0.34	0.32	0.29	0.29
Al-current collectors	0.80	0.77	0.74	0.74

The material balance of the LFP batteries is shown in Table 6 [72].

Table 6. Materials content in different LFP batteries per kWh [72].

LFP Component	Material	$m, \text{kg kWh}^{-1}$
Active Material	LiFePO_4	2.5
In-cathode Iron Content	Iron (Fe)	0.93
Anode Material	Graphite	1-1.4
Electrolyte	LiPF_6 in Carbonates	~ 1.0
Current Collector (Cathode)	Aluminum (Al)	0.74-0.8
Current Collector (Anode)	Copper (Cu)	0.29-0.34

III.3. Transport of Ore Concentrates and Salts

In order to estimate the amount of concentrate that has to be shipped to China, the following estimations are performed. The mass of lithium concentrate in spodumene concentrate from Australia with 14.8% LCE is estimated at 3.2 Mt (corresponding to about 474 Mt LCE), and the mass of battery-grade Li_2CO_3 from Chile and Argentina is about 418,000 t. In Table 7, the masses of metals in Mt are estimated for 0.36 TWh NMC and 0.84 TWh LFP, based on the data from Tables 5 and 6, with the note that China mainly imports ore concentrates. The mass of the concentrate can be estimated by considering that an ore deposit is a rock mass in which one or more metals are concentrated to the point where it is economically viable to extract them. Some basic concentrations of the mentioned metals in average concentrates are estimated and shown in Table 7 [73, 74]. To calculate $\text{CO}_{2,\text{eq}}$ emissions for ore transport by transport ships, the gross tonnage was assumed as a function of the molded volume of all enclosed spaces of the ship. Net tonnage is obtained by a formula that is a function of the molded volume of all cargo spaces of the ship. Net tonnage must not be less than 30% of gross tonnage. Deadweight tonnage (DWT) is a measure of how much weight a ship can carry. It is the sum of the weights of cargo, fuel, fresh water, ballast water, provisions, passengers, and crew. The cargo itself should not be less than 30% of DWT. The optimal speed of a cargo ship ranges between 17 and 24 knots. Speed depends primarily on the size of the ship, with larger ships traveling more slowly. For example, a container ship can go up to 24 knots, while a large bulk carrier travels at a speed of about 15 knots. To avoid overestimation, a Very Large Ore Carrier (VLOC) of 400,000 DWT with 70 tons of heavy fuel oil consumption per day and 0.140 Mt of net cargo was selected for $\text{CO}_{2,\text{eq}}$ estimation [75]. Each kilogram of heavy fuel oil burned produces 3.23 kg of greenhouse gases according to GREET [76]. By simple calculation, it can be determined that one such cargo ship produces more than 230 tons of greenhouse gases every day during the cruise, which is what ~100 cars produce in a single year. To calculate distances, the "Sea-distances" internet portal [77] was used, which calculates the distance and number of sailing days based on ship speed and suggests several routes.

Table 7. Total mass of metals required for 1.2 TWh of batteries, metal concentration in ore for profitable exploitation, calculated mass of concentrate, and exporting countries.

Metal	m , kg kWh ⁻¹	Mass of metals Mt	Minimum ore grade for profitable extraction, wt. %	Mass of concentrate Mt	Import
Ni	0.59-1.02	0.21-0.37	0.5	42-74	Philippines
Co	0.13-0.59	0.050-0.21	0.4	12.5-52.5	DR Congo Indonesia
Mn	0.13-0.59	0.050-0.21	35	0.14-0.60	South Africa Australia
Fe	0.93	0.78	20	3.9	Australia
Cu	0.29-0.34	0.35 0.24	0.5	70-82	Chile
NMC		0.10-0.12			
LFP		0.25-0.29			
Al	0.74-0.8	0.89-0.92	30	2.96-3.05	Guinea
NMC		0.27-0.29			
LFP		0.62-0.67			

Table 8 summarizes the results of the $\text{CO}_{2,\text{eq}}$ emission estimate for the transport of concentrates for NMC and LFP batteries from different countries to China.

Table 8. Estimates of CO_{2,eq} emissions for the transport of concentrates for 1.2 TWh of NMC and LFP batteries (ship cargo, 0.140 Mt).

Ore resource	Cargo Mt	Import from	No of ships	Distance nautical miles*	Days	Fuel consumption per ship, t	Total fuel consumption kt	m(CO _{2,eq}) Mt
Spodumene	3.2	Australia	23	3521	9.8	686	15.8	0.05
LCA	0,261	Chile	2	10354	28.7	2009	4.02	0.013
LCA	0.096	Argentina	1	10578	29	2030	2.0	0.002
LCA	0.05	Brazil	1	10442	29	2030	2.0	0.002
Ni	42-74	Philippines	300-530	1091	3	210	63-334	0.2-1.1
Co	12.5-52.5	DR Congo	89-396	8917	24.7	1730	154-685	0.5-2.2
Mn	0.14-0.60	South Africa	1-5	7118	19.8	1386	1.39-6.9	0.005-0.022
Fe	3.9	Australia	28	3521	9.8	686	19.2	0.06
Cu	70-82	Chile	500-586	10354	28.7	2009	1.004-1.180	3.2-3.8
Al	2.96-3.05	Guinea	21-22	10303	28.6	2002	42-44	0.14-0.14
Total								11.4

* nmi - nautical mile 1.852 km

Summary data from Table 7 show that total transport emissions of the raw materials are ~11.4 Mt CO_{2,eq}.

III.4. CO_{2,eq} emissions during the production of lithium batteries

III.4.1. The cathode and anode active mass production

Considering that metals concentrate are used for cathode production, from which salts (usually sulfates) or oxides are later produced, as well as current collectors. The comprehensive diagrams of ore processing to sulfates are given in ref [78]. The cradle-to-gate carbon footprint of nickel sulfate hexahydrate (NiSO₄×6H₂O) production, particularly from high-pressure acid leaching of the Indonesian laterite, is significant. This impact typically ranges from 33.8 to 36.8 t CO_{2,eq} per ton of Ni, which corresponds to recalculated emissions of 7.54 to 8.21 t CO_{2,eq} per ton of sulfate. This footprint encompasses mining, processing, and refining, with primary contributions from coal-based energy and grid electricity [79]. For cobaltite-dominated ores, the emission for cobalt sulfate heptahydrate (CoSO₄×7H₂O) is estimated at 20.9 kg CO_{2,eq} per kg of sulfate, of which 12.7 kg CO_{2,eq} is attributed to the hydrometallurgical process; for linnaeite-dominated ores, the equivalent value is 20.4 kg CO_{2,eq} (11.0 kg CO_{2,eq}, hydrometallurgical process) [80]. Manganese sulfate (MnSO₄×H₂O) produced from higher-grade ores (20–40 wt% Mn) is generally more energy-efficient and produces lower emissions. Assuming a standard electrical grid mix, the total impact is expected to be 4.8 t CO_{2,eq} per ton of high-purity manganese sulfate monohydrate [81]. The total carbon footprint of iron sulfate (FeSO₄×7H₂O) is determined by integrating cradle-to-gate emissions from iron ore extraction with specific chemical processing requirements. Typical greenhouse gas emissions for iron ore extraction, concentration, and pelletizing range from 12 to 25 kg CO_{2,eq} per ton of ore, depending on the study. Following leaching and crystallization phases for 30–60 wt% Fe ores results in an additional 0.03 to 0.07 t CO_{2,eq} per ton of purified sulfate. These values account for all processing steps from ore concentrate to purified sulfate but exclude emissions from mining, transport, and downstream battery material synthesis [82, 83].

To ensure a rigorous deterministic audit, emission data for precursor sulfates are recalculated to a per-ton-of-metal basis using stoichiometric conversion factors. The carbon intensity for nickel is 33.8 to 36.8 t CO_{2,eq} / t(Ni), and manganese is determined at 14.78 t CO_{2,eq} / t(Mn). Cobalt exhibits a significantly higher footprint of 95.40 t CO_{2,eq} / t(Co) due to the energy-intensive purification of CoSO₄·7H₂O. In contrast, the cumulative footprint for iron—integrating both ore extraction and crystallization, remains an order of magnitude lower at approximately 0.40 t CO_{2,eq} / t(Fe), reflecting the high efficiency and lower energy threshold of the iron sulfate supply chain. For the current collectors, CO_{2,eq} is for copper 4.1 t, while for aluminum 16 t [84, 85]. The emission for the production of current collectors' foils, Al and Cu, is neglected. However, to make the data more realistic, the mean value of

CO_{2,eq} t per t of metal is used for the calculation, as is presented in Table 9.

Table 9. Cradle to gate CO_{2,eq} emissions for production of the metal salt precursors, and metals for current collectors on the metal basis (mean value) for NMC and LFP batteries.

LIB- NMC	NMC kg kWh⁻¹	m(Mt) metals per 0.36 TWh	m(CO_{2,eq}) t per t metals	m(CO_{2,eq}) Mt 0.36 TWh
CAM	1.27-1.78	0.47-1.0		
Ni	0.59-1.02	0.21-0.37	35.3	7.6-13.1
Co	0.13-0.59	0.050-0.21	95.5	4.8-20.0
Mn	0.13-0.59	0.05-0.21	14.8	0.74-3.1
Cu	0.29-0.34	0.10-0.12	4.1	0.41-0.49
Al	0.74-0.80	0.27-0.29	16	4.3-4.6
m(CO_{2,eq}) Mt				17.9-41.3
LIB-LFP	LFP kg kWh⁻¹	m(Mt) per 0.84 TWh		m(CO_{2,eq}) Mt 0.84 TWh
CAM FePO ₄	2.5	2.1		
Fe	0.93	0.78	0.4	0.31
Cu	0.29-0.34	0.24-0.29	4.1	1-1.2
Al	0.74-0.80	0.62-0.67	16	10-11
m(CO_{2,eq}) Mt				11.3-14.5
Total m(CO_{2,eq})Mt				29.2-55.8

By rough estimation using GtG, the total CO_{2,eq} for the production of metals (in the form of salt) used in batteries is estimated at 29.2-55.8 Mt. The conversion of precursor sulfates into final lithiated metal oxides (NMC CAM) represents a distinct manufacturing burden beyond upstream mining. In the solution of metal salt, NaOH is usually added to coprecipitation, metal hydroxide, followed by filtration and drying. This solid-state synthesis, primarily driven by high-temperature calcination, requires a specific energy input of 7.3 to 12.0 MWh per ton of CAM. Depending on the local electrical grid mix, this processing stage alone contributes an additional 2.9 to 4.8 t CO_{2,eq} per t of CAM. These values account for the thermal energy required for atmospheric control and crystal lattice stabilization, independent of the carbon footprint associated with raw material extraction and purification [86-88]. The production of LiFePO₄ (LFP) cathode materials is significantly less energy-demanding than that of high-nickel NMC variants. While NMC requires 7.3 to 12.0 MWh/t for complex multi-stage calcination, the thermal treatment of LFP precursors (typically between 600–800 °C) demands a specific energy input of approximately 3.5 to 5.5 MWh per ton of CAM. Consequently, the direct manufacturing emissions for LFP synthesis are estimated between 1.4 and 2.2 t CO_{2,eq} per ton of CAM, which is roughly 50% lower than the synthesis burden of NMC. This reduction is attributed to the lower processing temperatures and the shorter residence time required for crystal lattice stabilization in the olivine structure [72].

For the production of anode active materials (AAM), the environmental footprint widely accepted until recently, is heavily influenced by the choice of AAM [89]. While natural graphite (NG) is inherently less carbon-intensive than its synthetic counterpart, the multi-stage purification and shaping required for battery-grade material significantly elevate its cradle-to-gate (CtG) burden. The transition from raw flake graphite to spherical, coated AAM involves intensive mechanical shaping and chemical purification. This process typically requires 15 to 22 MWh per ton of AAM, resulting in a carbon footprint ranging from 5.0 to 11.0 t CO_{2,eq} per t of AAM. The higher end of this range reflects the low yield during spheronization (often <50%) and the energy-intensive thermal coating phase. Synthetic graphite (SC), derived from petroleum coke or coal tar pitch, remains the most energy-demanding component due to long-lasting graphitization at 3000 °C. The energy input ranges from 25 to 35 MWh per ton, with emissions scaling between 17 and 28 t CO_{2,eq} per t of, depending on the regional electricity mix [89]. From an environmental standpoint, natural graphite is the superior choice due to its significantly lower cumulative energy demand and carbon intensity. However, synthetic graphite is often preferred in high-performance applications due to its superior structural stability and longer cycle life. Consequently, most commercial anodes utilize a strategic blend of both materials to balance electrochemical longevity with sustainability targets.

But at the end of 2025 and beginning of 2026, significant changes have been reported about CtG CO_{2,eq} emission for AAM. By a deeper analysis, it is reported that synthetic graphite emission is 49.0 kg CO_{2,eq}kWh⁻¹

[72]. Also, it is reported that the SG value is 42.2 t CO_{2,eq}/t, which is 2 to 10 times higher than presently existing values [90]. Some “intriguing report” comes from SG production in the USA, with a favorable electric energy mix. The results show that GHG emissions of ~30 kg CO_{2,eq} per kg of SG, with overall energy use of 580 MJ kg⁻¹, and water consumption of ~120 L kg⁻¹, for washing [91]. This number is one assume that for LIB ~1 kg of SG per kWh is used, so for 1.545 TWh it is counted to ~1.85 or more millions of liters of purified water. A lot of energy has to be used to purify water (demineralization), but how much energy has to be used to purify water after washing SG that contains a significant percentage of toxic and environmentally dangerous compounds is uncertain. Consequently, the Cradle-to-Gate data for active cathodic and anodic materials are given in Table 10. Because in the USA, the SG emission is ~30 kg CO_{2,eq} per kg, for China will be conservatively assumed to be ~35 t CO_{2,eq}/t of AAM, and 1 kg of AAM per kWh. So, for 0.84 TWh of LFP, 0.84 Mt, and for NMC, 0.36 Mt of SG is estimated.

Table 10. Estimation of the CO_{2,eq} for cathodic (CAM) and anodic (AAM) active mass synthesis, excluding Al and Cu.

Process	0.36 TWh NMC <i>m</i> (CO _{2,eq}) Mt	0.84 TWh LFP, <i>m</i> (CO _{2,eq}) Mt	1.2 TWh <i>m</i> (CO _{2,eq}) Mt
CAM synthesis	0.47-1.0 Mt NMC 2.9-4.8 t CO _{2,eq} / t	2.1 Mt LFP 1.4-2.2 t CO _{2,eq} / t	
CAM, <i>m</i> (Mt) CO _{2,eq}	1.36-4.8	2.94-4.62	~4.30-9.42
AAM	0.36 Mt SG	0.84 Mt SG	
GtG AAM, <i>m</i> (Mt) CO _{2,eq}	12.6	29.4	42
Total <i>m</i> (Mt) CO _{2,eq}	14-17.4	32.9-34.0	46.9—51.4

III.4.2. Battery Manufacturing

For the positive and negative electrode manufacturing, the most common is the wet process, which uses N-Methyl-2-pyrrolidone (NMP) to make a slurry with CAM and AAM powder, casting on the current collectors, and hot solvent evaporation. This conventional method requires energy-intensive drying and solvent recovery (NMP) [92], contributing 11.2 kg CO_{2,eq} kWh⁻¹ to the total cell footprint. A lot of efforts have been made to replace the toxic NMP or to switch to a dry process, but they are still far from industrial scale [93].

Beyond material extraction and synthesis, cell assembly and electrochemical formation represent a critical energy overhead. According to recent high-energy density cell models [72], cell assembly in controlled dry room environments consumes 15–20 kWh per kWh of battery capacity. The subsequent formation, numerous charge/discharge cycles, and aging process add 5–8 kWh per kWh. Depending on the electrical grid carbon intensity, these final manufacturing stages contribute an additional 8.0 to 11.2 kg CO_{2,eq} per kWh to the total cradle-to-gate balance. Also, it is taken that kg CO_{2,eq} is 4.1 kg for Cu, and 16 kg for Al. Table 11 presents the estimated CO_{2,eq} emission for the battery assembly.

Table 11. Estimated CO_{2,eq} emission for the battery assembly.

Process Stage	<i>m</i> (kg) kWh ⁻¹	kg CO _{2,eq} kWh ⁻¹	0.36 TWh NMC, <i>m</i> (CO _{2,eq}) Mt	0.84 TWh LFP <i>m</i> (CO _{2,eq}) Mt
Negative current collector (Cu)	0.29-0.34	1.2-1.4	0.41-0.49	1-1.2
Positive current collector (Al)	0.74-0.80	12-13	4.3-4.6	10-11
Positive electrode	-	11.2	4	9.4
Negative electrode	-	11.2	4	9.4
Electrolyte (LiPF ₆ / Carbonates)		5.3		
NMC	0.38		0.72	
LFP	1			4.5
Cell Assembly (Dry Room)	-	6.0 – 8.0	2.2-2.9	5-6.7
Formation & Aging	-	2.0 – 3.2	0.72-1.15	1.7-2.7
Total Manufacturing	-	-	16.3-17.9	41.1-45

The total CO_{2,eq} emission for the battery assembly is approximately 57.3-62.9 Mt CO_{2,eq}

The summary calculation of emissions for the production of the battery in China by type, with the estimated CO_{2,eq}

emission, is given in Table 12.

Table 12. Total CO_{2,eq} emissions by battery type

Components	NMC <i>m</i> (CO _{2,eq}) Mt 0.36 TWh	LFP <i>m</i> (CO _{2,eq}) Mt 0.84 TWh	Total <i>m</i> (CO _{2,eq}) Mt 1.2 TWh
Raw materials shipment			
Li-ore	0.02	0.047	0.067
Metals ore NMC	4.03		4.03
Metals ore LFP		0.06	0.06
Al-ore	0.08	0.2	0.28
Cu-ore	2.52	5.9	7.0
Raw materials shipment	3.42	8.0	11.42
CtG LCE	3.81-6.51	8.9-15.2	12.7-21.7
GtG metal salt precursor			29.2-55.8
NMC	17.9-41.3		
LFP		11.3-14.5	
CAM synthesis	1.36-4.8	2.94-4.62	~4.30-9.42
CtG AAM synthesis	12.6	29.4	42
Battery assembly			57.3-62.9
NMC	16.3-17.9		
LFP		41.1-45	
Total Emission, <i>m</i>(CO_{2,eq}) Mt	55.3-86.5	100.6-116.7	157.0-203.2
Uncertainty correction factor (UCF) 15%	8.3-13.0	15.1-17.5	23.5-30.5
Total Emission, <i>m</i>(CO_{2,eq}) Mt	63.3-99.5	115.7-134.2	180.5-233.7
Emission, <i>m</i>(CO_{2,eq}) kg kWh⁻¹	175.8-276.4	137.7-159.8	150.4-194.7
Mean emission, <i>m</i>(CO_{2,eq}) kg kWh⁻¹	225.7	148.7	172.6

As can be seen from Table 12, the total emission of CO_{2,eq} in China for 70% LFP and 30% NMC is estimated between 180.5 and 233.7 Mt, with the specific emission of 150.4-194.7 kg CO_{2,eq} kWh⁻¹. Concerning battery by type, for NMC chemistry, the total emission is 63.3-99.5 Mt (175.8-276.4 kg CO_{2,eq} kWh⁻¹), while for LFP, it is 115.7-134.2 Mt (137.7-159.8 kg CO_{2,eq} kWh⁻¹). Using the transparent, deterministic methodology, for the estimation of the CO_{2,eq} kWh⁻¹ during the production is similar as reported for ternary NMC of 334 kg CO_{2,eq} phase, and slightly higher than to lithium iron phosphate batteries of 93.1 kg CO_{2,eq} [94]. It should also be concluded that Chinese shifts to LFP chemistry significantly decrease CO_{2,eq} emission.

III.5. Transport of batteries

Global lithium-ion battery production increased by 29% compared to the previous year, 2023, and reached 1,545 GWh in 2024, including 1,215 GWh in China, representing a 37% growth and a 79% share of total battery production. Continuous demand growth in China and increased investment in production abroad further boosted global shipments [95]. Production of active cathode materials (ACM) shows that China's control is almost complete for LFP (98.45%), while it is about 70% for NMC [96, 97]. As the main center of battery production, China produces both for the domestic market and for export. Total exports reached a record 197.1 GWh in 2024, representing a growth of 29.2% and 19% of total sales. Electric vehicle (EV) batteries accounted for 133.7 GWh (67.8%) with 5% growth, while the battery energy storage systems (BESS) reached 63.4 GWh with an exceptional growth of 151.6% and now make up 32.2% of exports [98].

A container ship remains key to global supply chains, allowing for more efficient transport of goods than other modes of transport. The average mass of an empty 20 FT TEU (Twenty-foot Equivalent Unit). A 20 FT TEU is a standardized metal container for transporting goods about 6.1 m long, 2.44 m wide, and 2.59 m high, which forms the basic unit in maritime transport known as 1 TEU. The mass of the container is about 2,500 kg, while the average net mass of the container is 24,000 kg, and the maximum gross mass is 30,480 kg. Assuming a specific energy of NMC batteries of 200–250 Wh kg⁻¹, the average mass is about 4.5 kg kWh⁻¹, while for LFP (160 Wh kg⁻¹), it is about 6.25 kg kWh⁻¹. For a total of 197.1 GWh, assuming 70% is NMC (138 GWh) and 30% is LFP

(59 GWh), the total mass is 621 Mkg for NMC and 369 Mkg for LFP batteries. Taking a Panamax container ship with a capacity of 5,000 TEU and a net payload of about 120 Mkg as an example, approximately five ships are needed for NMC battery transport and three ships for LFP batteries [75]. If only export from China to Europe (Germany) and the USA is assumed, neglecting other countries, at a speed of 10 knots, the distance to Germany is 13,469 nautical miles (56 days), and to the USA 5,802 nautical miles (24 days). Fuel consumption is about 8,400 t (0.027 Mt CO_{2,eq}) for Germany and 6,000 t (0.02 Mt CO_{2,eq}) for the USA, giving a total of about ~0.05 Mt CO_{2,eq}.

It should also be mentioned that a long-term strategy has allowed China to form a broad battery supply chain. In 2024, the production, for domestic use and export of cathode materials reached 3.35 million tons (+35%), of which LFP dominates with 2.46 million tons (75%). Production of anode materials was 2.08 million tons (+26%), including 1.81 million tons of artificial and 260,000 tons (~9%) of natural graphite. Separator production reached 22.3 billion m² (+30%), while the electrolyte reached 1.47 million tons (+32%) [98].

III.6. Sensitivity analysis

To estimate the variation of the emission of CO_{2,eq} for the different Scenarios of the LFP/NMC ratio and Uncertainty Correction Factor (UCF), the sensitivity analysis is performed, with the following data:

Variables Definition:

x = LFP share (60-100%, e.g., 0.60 to 1.0)

y = Uncertainty Correction Factor (UCF) (5%, 10% and 15%, e.g., 0.05, 0.1 and 0.15)

e_{LFP} = emission factor for LFP battery production (kg CO_{2,eq} kWh⁻¹)

e_{NMC} = emission factor for NMC battery production (kg CO_{2,eq} kWh⁻¹)

Total production = 1.2 TWh

Baseline Values

Baseline LFP share = 0.70 (70%)

Baseline NMC share = 0.30 (30%)

Baseline mean (including UCF 15%), $e_{mean} = 172.6 \text{ kg CO}_{2,eq} \text{ kWh}^{-1}$

Baseline LFP emission factor (mean) = **148.7 kg CO_{2,eq} kWh⁻¹**

Baseline NMC emission factor (mean) = **225.7 kg CO_{2,eq} kWh⁻¹**

Weighted average without UCF, using the mean emission factors: $e_{weighted} = (x \times 148.7) + [(1 - x) \times 225.7]$

and with UCF (y): $e_{total} = e_{weighted} \times (1 + y)$

Percentage change from Baseline, $e(\%)$: $e(\%) = 100 \times (e_{total} - e_{mean}) / e_{mean}$

where Baseline mean is, $e_{mean} = 172.6 \text{ kg CO}_{2,eq} \text{ kWh}^{-1}$

The summarized data of the sensitivity analysis are given in Table 13, while the graphical presentation is shown in Figure 4.

Table 13. Results of the sensitivity analysis for 11 scenarios, changing the LFP/NMC ratio from 100% to 60%, and the UCF from 5% to 15%.

Scenario	LFP Share	UCF	Mean kg kWh ⁻¹	Min kg kWh ⁻¹	Max kg kWh ⁻¹	ΔCO _{2,eq} kg kWh ⁻¹	ΔCO _{2,eq} %
Scenario 1	60%	15%	188.8	164.5	213.1	+16.2	+9.4%
Scenario 2	60%	10%	180.6	157.4	203.8	+8.0	+4.6%
Scenario 3	60%	5%	172.4	150.3	194.5	-0.2	-0.1%
Baseline	70%	15%	172.6	150.4	194.8	0.0	0.0%
Scenario 4	70%	10%	165.1	143.9	186.3	-7.5	-4.3%
Scenario 5	70%	5%	157.6	137.4	177.8	-15.0	-8.7%
Scenario 6	85%	15%	156.0	136.1	175.9	-16.6	-9.6%
Scenario 7	85%	10%	149.2	130.2	168.2	-23.4	-13.6%
Scenario 8	85%	5%	142.4	124.3	160.5	-30.2	-17.5%
Scenario 9	100%	15%	148.8	129.9	167.7	-23.8	-13.8%
Scenario 10	100%	10%	142.3	124.2	160.4	-30.3	-17.6%
Scenario 11	100%	5%	135.9	118.6	153.2	-36.7	-21.3%

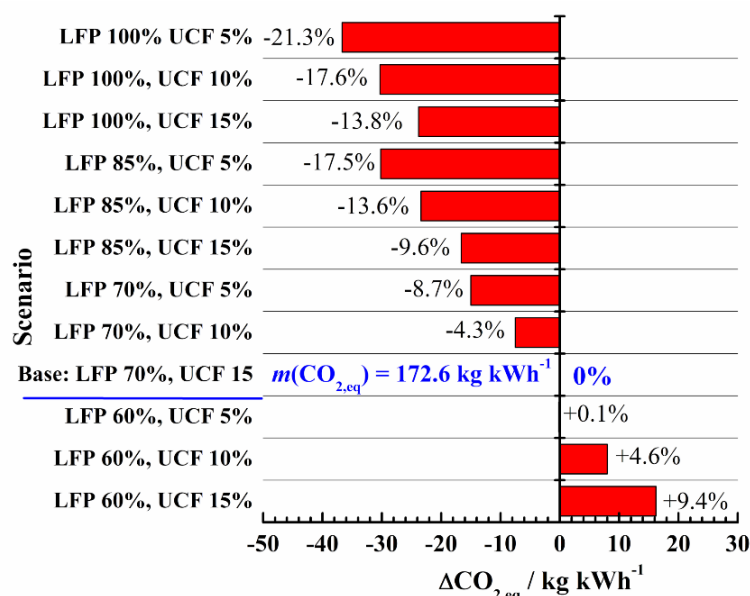


Figure 4. Results of the sensitivity analysis based on the baseline mean of $172.6 \text{ kg CO}_{2,\text{eq}} \text{ kWh}^{-1}$, and using the mean value for all treated Scenarios (Table 13).

The lowest emissions ($135.9.0 \text{ kg CO}_{2,\text{eq}} \text{ kWh}^{-1}$, -21.3% , and decrease of $-36.7 \text{ kg CO}_{2,\text{eq}} \text{ kWh}^{-1}$) are achieved with 100% LFP and UCF 5%, while the highest emissions ($188.8 \text{ kg CO}_{2,\text{eq}} \text{ kWh}^{-1}$, $+9.4\%$ and an increase of $+19.7 \text{ kg CO}_{2,\text{eq}} \text{ kWh}^{-1}$) occur with 60% LFP and UCF 15%, demonstrating that both technology mix and uncertainty correction factor significantly affect the carbon footprint. So, the intensive production of LFP and the decrease of NMC chemistry, which dominate in the rest of the world, are good examples of how to decrease GHG emissions. The decrease in GHG emission, accounted for in the Uncertainty Correction Factor, significantly lowers the emission, but interestingly, increases to 100% LFP with a UCF of only 5%, emission decrease of only -21.3% .

Nevertheless, moving to LFP chemistry, some questions arise. Because of the lower specific energy, it is reasonable to conclude that the range and weight of EV will be higher than for EV using an NMC battery. Therefore, let us consider the following. For the same 75 kWh battery pack using LFP cells, the specific energy is assumed at 140 Wh kg^{-1} , while NMC cells are assumed at 270 Wh kg^{-1} . Thus, the LFP cells for a 75 kWh pack weigh approximately 536 kg, compared to 278 kg for NMC. The mass penalty for LFP is therefore 258 kg, or 93% heavier than NMC at the cell level. In terms of volume, LFP cells deliver 330 Wh L^{-1} , compared to 720 Wh L^{-1} for NMC. The LFP cells occupy 227 liters, while the NMC cells occupy 104 liters. The volume penalty for LFP is 123 liters, or 118% larger than NMC at the cell level.

However, at the complete battery pack level, the gap narrows because LFP allows simpler thermal management and more efficient cell-to-pack integration. The total LFP pack weight for 75 kWh reaches approximately 600 kg, while an NMC pack weighs 400 kg. The LFP pack is therefore 200 kg or 33% heavier than the NMC pack. The volume of the LFP pack is approximately 270 liters, compared to 180 liters for NMC, making the LFP pack 90 liters or 50% larger than the NMC pack. The heavier LFP pack increases vehicle mass by approximately 200 kg compared to an NMC vehicle. This additional mass raises energy consumption by only 6%. For a 75 kWh battery with 80% depth of discharge, DoD, (60 kWh usable), the LFP vehicle consumes 19.1 kWh per 100 km, while the NMC vehicle consumes 18.0 kWh per 100 km, according to Battery to Will (BtW) methodology. The range per charge, 60 kWh, for LFP is therefore, 19.1 kWh per 100 km, or in total $\sim 314 \text{ km}$. For the NMC, the range of 60 kWh usable energy is 333 km. The LFP vehicle thus has only 19 km less range per charge, or only 6% lower than the NMC-based vehicle. Despite these disadvantages, LFP remains less expensive to produce, reducing raw material costs by approximately 40% or more compared to NMC. However, the gap in mass, volume, range, and energy consumption is insignificant for passenger EVs. However, packaging space and range are critical for expensive EVs, and could explain why automakers still prefer NMC for premium or long-range vehicles, where 50% more “energy” could be packed in the same space as for LFP.

IV. Discussion

IV.1. Green House Gas (GHG) emissions

Table 14 summarizes the $\text{CO}_{2,\text{eq}}$ emissions for all technological operations considered in this work, given for production in China. It should be noted that 20% of emissions from other global producers were conservatively added to China's emissions. As can be seen, the most intense $\text{CO}_{2,\text{eq}}$ are metal salt precursor production (17.58-25.0%), AAM synthesis (25.0-19.33%), and battery assembly (30.0-26.3%).

Table 14. Summary of $\text{CO}_{2,\text{eq}}$ emissions estimation of the different technological phases in lithium-ion battery production in China (1.2 TWh) and the whole world (1.545 TWh).

Table 14. Total world $\text{CO}_{2,\text{eq}}$ emissions by battery type

Emission in China	Total	NMC	LFP
Total Emission, $m(\text{CO}_{2,\text{eq}})$ Mt	157.0-203.2	55.3-86.5	100.6-116.7
	(1.2 TWh)	(0.36 TWh)	(0.84 TWh)
Uncertainty correction factor (UCF) 15%	23.5-30.5	8.3-13.0	15.1-17.5
Total Emission, $m(\text{CO}_{2,\text{eq}})$ Mt	180.5-233.7	63.3-99.5	115.7-134.2
$m(\text{CO}_{2,\text{eq}})$ kg kWh^{-1}	150.4-194.7	175.8-276.4	137.7-159.8
Mean $m(\text{CO}_{2,\text{eq}})$ Mt	207.1	81.4	125
Mean emission, $m(\text{CO}_{2,\text{eq}})$ kg kWh^{-1}	172.6	225.7	148.7
Emission of the rest of the World	0.34 TWh	0.241 TWh	0.104 TWh
Mean $m(\text{CO}_{2,\text{eq}})$ Mt	70.4	19.6	13
Uncertainty correction factor (UCF) 15%	10.6	2.94	1.35
Mean $m(\text{CO}_{2,\text{eq}})$ Mt	81	22.5	14.5
Mean $m(\text{CO}_{2,\text{eq}})$ Mt 1.545 TWh	288.1	106.9	139.5
Mean emission, $m(\text{CO}_{2,\text{eq}})$ kg kWh^{-1}	186.5 (1.545 TWh)	177.4 (0.6 TWh)	147.8 (0.944 TWh)

Comparing the total mean emission outcome from world battery production of ~ 290 Mt $\text{CO}_{2,\text{eq}}$ with emissions from fossil fuels of 37.4 Gt in 2024, the percentage share of lithium-ion battery production in total emissions is estimated to be $\sim 0.8\%$. However, because the energy sector produces the most greenhouse gas emissions, 68% (25.4 Gt), of which 15% (3.8 Gt) originated from transport, ~ 290 Mt $\text{CO}_{2,\text{eq}}$, corresponds to $\sim 8\%$ contributions, without EV being produced.

In 2024, global demand for electric vehicle (EV) batteries installed on vehicles reached a historic milestone, with total demand for EV batteries growing to over 950 GWh (nearly 1 terawatt-hour) as EV sales expanded worldwide. Electric cars accounted for more than 85 % of this installed capacity, while electric trucks, buses, and other heavy vehicles comprised the remainder of demand [100].

To estimate annual electricity throughput for the 2024 fleet, the installed battery energy is adjusted for depth of discharge (DoD = 80 %), reflecting realistic usable energy per cycle. Under this assumption, the usable energy stock becomes approximately 0.64 to 0.72 TWh for passenger EVs and 0.04 to 0.08 TWh for buses and heavy-duty vehicles. Assuming passenger EVs are charged once every ~ 8 -9 days (~ 40 -45 cycles per year) and heavy-duty vehicles (buses and trucks) charge daily (~ 300 -350 cycles per year), the annual net electricity demand is estimated at approximately 60-90 TWh per year. Applying cumulative system losses of 10% to 15% for grid transmission, AC/DC conversion, inverters, and charging inefficiencies increases this to a gross electricity demand of ~ 67 to 105 TWh per year. Using a global average electric energy mix emission factor of 475 g $\text{CO}_{2,\text{eq}}$ per kWh for electricity generation in 2024 [101], this gross demand corresponds to an operational carbon footprint of approximately 30-47 Mt $\text{CO}_{2,\text{eq}}$ annually for the global EV fleet under current utilization and charging patterns. These values reflect operational emissions only; they do not include lifecycle emissions from vehicle or battery manufacturing, end-of-life processing, or infrastructure production. While the typical world energy mix continues to evolve, the absolute carbon output remains a direct function of the total energy throughput required to maintain global fleet mobility and overcome these inherent thermodynamic and electrical losses.

Accordingly, if one adds GHG from annual charging of 30-47 Mt $\text{CO}_{2,\text{eq}}$ to mean batteries production emission of ~ 290 Mt $\text{CO}_{2,\text{eq}}$, this gives ~ 320 -337 Mt $\text{CO}_{2,\text{eq}}$. Again, comparing with emissions from fossil fuels of 37.4 Gt in 2024, the total GHP is around $\sim 0.85\%$ to 0.9% .

IV.2. Cumulative CO_{2,eq} Emissions Comparison: Diesel vs. Electric Vehicle (EV)

To better understand emissions over the lifetime, the following analysis is performed. This comparison is based on the following input data. For both cars, the vehicle's production (body, chassis, interior, electronics, and assembly, excluding any traction battery) emits ~6,000 kg of CO_{2,eq}. The vehicle is driven 15,000 kilometers per year (~40 km/day) over a hypothetical 15-year lifetime.

The diesel vehicle consumes approximately 6.0 liters of fuel per 100 kilometers. For diesel fuel, the Well-to-Wheel (WtW) emission factor of approximately ~3.1 kg CO_{2,eq} per liter is composed of 0.6 kg CO_{2,eq} per liter for the Well-to-Tank (WtT) stage, which covers crude oil extraction, refining, and transport, and 2.4–2.6 kg CO_{2,eq} per liter for the Tank-to-Wheel (TtW) stage, which covers the exhaust emissions from combustion in the vehicle engine [102]. It should be noted that in many developed countries, the blend of fossil diesel and biofuels is used, with much smaller emissions, for example, B20 ~2.67 kg CO_{2,eq} per liter [103]. This gives the diesel vehicle an emission of 18.6 kilograms of CO_{2,eq} per 100 kilometers, or for B20, ~16 kilograms of CO_{2,eq} per 100 kilometers. The vehicle is driven 15,000 kilometers per year (40 km/day) over a 15-year lifetime, totaling 225,000 km. Annual fuel consumption is 900 liters, and the annual diesel fuel emission of CO_{2,eq} is 2,790 kg for pure diesel, or 2,400 kg for B20. The 15-year fuel CO_{2,eq} totals 41,850 kg for pure diesel or 36,000 kg for B20. Adding the 6,000 kg of vehicle production gives the diesel a 15-year total emission of 47,850. or 42,000 kg CO_{2,eq} for B20. At the start of life, before any driving occurs, the diesel vehicle has upfront emissions of 6,000 kilograms from vehicle production.

The same assumption, 15-year, 225,000 km lifetime (15,000 km/year) is used for EV calculations [104]. The vehicle production (without battery) emission is assumed to be 6,000 kg CO_{2,eq}. Battery installed capacity is 75 kWh, which has to be charged at a depth of discharge (DoD) of 80 %, giving usable battery energy of 60 kWh. The battery production emission factor (global average) for NMC is 177 kg CO_{2,eq} kWh⁻¹, and for LFP, 148 kg CO_{2,eq} kWh⁻¹, is taken from Table 11. For the NMC battery production, CO_{2,eq} (mean) is 75 × 177 = 13,275 kg CO_{2,eq}, and adding 6,000 kg for EV productions gives 19,275 kg CO_{2,eq}, and for LFP 11,100 kg CO_{2,eq} or for EV 17,100 kg CO_{2,eq}. For the charging, it is conservatively assumed a charging efficiency of 85 % (15% energy loss). Accordingly, for the calculated energy from the grid per 100 km: 19.0 kWh (at wheels) / 0.85 (charging efficiency) = 22.3 kWh/100 km. Accounting for the average range per year is 15,000 km, the required energy is 3,345 kWh/year. To estimate CO_{2,eq} emission, the grid mix has a crucial role. According to Ember [105], in 2024, the CO_{2,eq} emission was in the European Union (EU) 213 g kWh⁻¹, the United States (USA) 384 g kWh⁻¹, China 578 g kWh⁻¹ and the global average 475 g kWh⁻¹. Hence, charging emissions per 100 km by regions with today's grid emissions are:

- EU: 21.18 × 0.213 = 4.51 kg CO_{2,eq}/100 km
- USA: 21.18 × 0.384 = 8.13 kg CO_{2,eq}/100 km
- China: 21.18 × 0.578 = 12.24 kg CO_{2,eq}/100 km
- Global: 21.18 × 0.475 = 10.10 kg CO_{2,eq}/100 km

or, annual charging emissions (15,000 km/year):

- EU: 15,000 × (4.51/100) = 677 kg CO_{2,eq}/year
- USA: 15,000 × (8.13/100) = 1,220 kg CO_{2,eq}/year
- China: 15,000 × (12.24/100) = 1,836 kg CO_{2,eq}/year
- Global: 15,000 × (10.10/100) = 1,515 kg CO_{2,eq}/year

From this data, it is plausible to draw a hypothetical diagram of cumulative emission over 15 years (if energy grid mix has not changed significantly), starting with 0 years diesel emission of 6,000 kg CO_{2,eq}, and annual diesel fuel emission of CO_{2,eq} 2,790 kg and 2,400 kg for B20. The EV gives the annual charging emissions multiplied by the number of years. The starting emission of NMC EV before traveling, 0 km is 19,275 kg CO_{2,eq}, and for LFP 17,100 kg CO_{2,eq}. The comparisons of two different battery chemistries and different diesel cumulative emissions results are shown in Figures 5a and 5b.

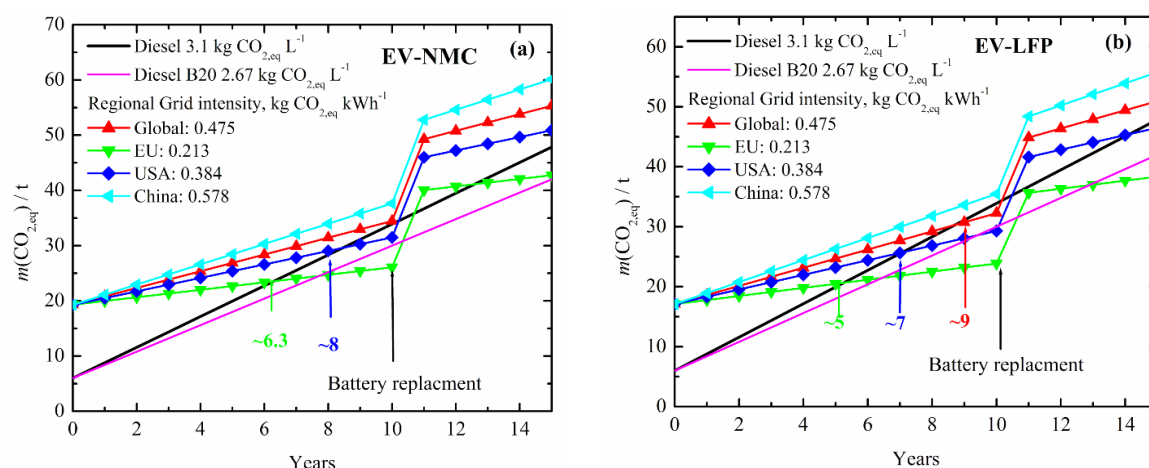


Figure 5. Cumulative $\text{CO}_{2,\text{eq}}$ emission of diesel car (consumption: pure diesel and blend B20, 6 L/100 km), and EV over years under different electricity grid emission mix (a) EV NMC, (b) EV LFP.

As can be seen from Figure 5a, the lower emission of EV-NMC than diesel will occur after 6.3 years in the EU, 8 years in the USA, globally, and in China, practically never occurs. The situation is slightly better with EV-LFP, as can be seen from Figure 5b. This is mainly connected with the starting disadvantage of emission of EV and $\text{CO}_{2,\text{eq}}$ from battery production. Unfortunately, under different driving scenarios, the life span of LIB is ~ 10 years, (due to 2-3% of annual aging, or for 10 years 20% to 30% loss of initial capacity, temperature, SoC, DoD, fast charging, and many different internal and external factors that shorten life span [106, 107, 108] and has to be replaced, mainly due to increase of internal resistance and capacity loss. Consequently, during the present energy mix for charging, most of the LIB will produce more $\text{CO}_{2,\text{eq}}$ than corresponding diesel cars, especially with diesel biofuel blend. Under a hypothetical scenario where the global energy mix remains static at current levels, the transition to electric mobility fails to achieve net decarbonization. The cumulative system losses, combined with the high carbon intensity of battery manufacturing, create a “carbon trap” where the environmental debt of the EV is never fully amortized within its operational lifespan.”

A comparison with diesel car emissions can be made. The average diesel car travels $\sim 15,000$ km/year, consumes ~ 6 L of diesel per 100 km, and emits an annual diesel fuel emission of $\text{CO}_{2,\text{eq}}$ is 2,790 kg. Comparing this data to annual charging of 30–47 Mt $\text{CO}_{2,\text{eq}}$ and adding to mean batteries production emission of ~ 290 Mt $\text{CO}_{2,\text{eq}}$, gives ~ 320 –337 Mt $\text{CO}_{2,\text{eq}}$. Comparing this number with diesel emissions yields an incredible number of 115 million to 120 million diesel cars per year that would emit the same amount of gases as lithium-ion battery production and EV charging. The question remains regarding the production of electric cars, battery storage, and especially the source of electricity for battery charging, which can drastically change the share of greenhouse gas emissions and the impact on climate change.

Several European companies and industry associations actively promote the narrative of a low-carbon battery with a cradle-to-gate emission factor of 50–75 kg $\text{CO}_{2,\text{eq}}$ kWh $^{-1}$. BASF claims that its cathode active material production in Schwarzheide, Germany, reduces the carbon footprint by up to 40% compared to the industry benchmark, achieved through energy-efficient processes and renewable energy. Company Northvolt had set a target of producing cells with a carbon footprint as low as 10–20 kg $\text{CO}_{2,\text{eq}}$ kWh $^{-1}$ by 2030, powered by locally sourced renewable energy. On March 12, 2025 Northvolt AB, the Swedish electric vehicle battery manufacturer with 5000 employees, officially filed for bankruptcy in Sweden, following unsuccessful attempts to restructure debt, marking the largest bankruptcy in modern Swedish industrial history. The company, whose slogan was “make oil history”, received more than 10 billion US \$ in equity, debt and public financing since it was founded in 2016. Facing severe cash shortages, high operational costs, and production failures, the former European battery champion entered bankruptcy to allow a trustee to sell off assets. Also, Norwegian lithium-ion battery cell producer Morrow Batteries ASA filed for bankruptcy on May 6, 2026, after failing to secure sufficient new investment and liquidity. About 210 employees were affected. Total funding exposure exceeded ~ 470 million €, including private investment, public loans, and grants. The European Battery Alliance and various policy documents routinely cite figures in the range of 50–70 kg $\text{CO}_{2,\text{eq}}$ kWh $^{-1}$ for European cell production. These claims are not supported by the physical reality of the supply chain. What these statements omit is that Europe produces almost none of the upstream battery materials. BASF admits that it must “begin at the mines and metal refineries” and that “isolating and refining metals requires a lot of energy”. However, BASF does not own these mines or

refineries in Europe. The cathode active materials produced in Schwarzheide rely on precursors and refined metals imported from China, where they are produced on coal-intensive grids. In the “business plan| Northvolt similarly planned sources its active materials from Asian suppliers because Europe lacks domestic refining capacity for lithium, nickel, cobalt, and graphite. A battery assembled in Germany or Sweden still carries the carbon debt of Chinese upstream production. The clean European grid only affects the final assembly, which accounts for less than 15% of total cradle-to-gate emissions. The remaining 85% is already locked in before the materials arrive at the European factory.

Therefore, the claim of a $>50 \text{ kg CO}_{2,\text{eq}} \text{ kWh}^{-1}$ battery is not a scientific finding. It is a marketing narrative – a convenient illusion that allows companies and policymakers to claim progress while continuing to rely on the same carbon-intensive Chinese supply chain. Until Europe builds its own refining capacity for lithium, nickel, cobalt, and graphite – and until cathode and anode active materials are produced locally using renewable energy – the honest emission factor for any battery assembled in Europe remains $150\text{--}180 \text{ kg CO}_{2,\text{eq}} \text{ kWh}^{-1}$. The $50 \text{ kg CO}_{2,\text{eq}}$ myth is not a target. It is a deception. The green transition cannot be built on marketing claims. It must be built on real numbers.

To compare the $\text{CO}_{2,\text{eq}}$ emission of EV-LFP with pure diesel and B20, the existing literature data is used. For the comparison from the reported data using the LCA method for LFP $\text{CO}_{2,\text{eq}}$ emission ranges $90\text{--}127 \text{ kg kWh}^{-1}$ for China [109] that has a world market share of LFP over 90%, the conservative value of 100 kg kWh^{-1} for the cell, and an additional 20 kg kWh^{-1} for the battery pack. For the LFP (75 kWh, 80% DoD $\sim 60 \text{ kWh}$) is also assumed 2.5% calendar aging, and the cleanest EU $\text{CO}_{2,\text{eq}}$ emission for electric energy production of 0.23 kg kWh^{-1} . With all the above data considered, the comparisons are shown in Figure 6. As can be seen from Figure 6, the EV, while a pure diesel car, will emit more $\text{CO}_{2,\text{eq}}$ after 4.5 years, B20 after 5.5 years. But due to battery replacement after ~ 10 years (that cost over 10,000 US \$), the emission of EV is $31 \text{ t}/10.5$ years, the emission of B20, $\sim 30 \text{ t}/10.5$ years, and only $\sim 8\%$ lower than for a pure diesel car, $33.8 \text{ t}/10.5$ years. Even the some LCA study predict battery life of 15+ years, reality is different. During calendar aging, the growth of the Surface Interphase Layer (SIL) is a purely chemical, diffusion-limited passivation process following non-linear parabolic kinetics, $\sim t^{0.5}$, with degradation accelerating at elevated temperatures and higher States of Charge (SoC) due to the increased thermodynamic driving force for electrolyte decomposition. However, the introduction of aggressive fast charging-driven by high localized overpotentials, and increased resistivity due formation of thick non protective SIL, and intense Joule heating (I^2R), triggering severe lithium plating on graphite, active material micro-cracking, and accelerated chemical electrolyte decomposition that violently compresses the battery's expected lifespan from 8 to 10 years. In addition, with the development of advanced internal combustion engines, the fuel consumption will be significantly lower.

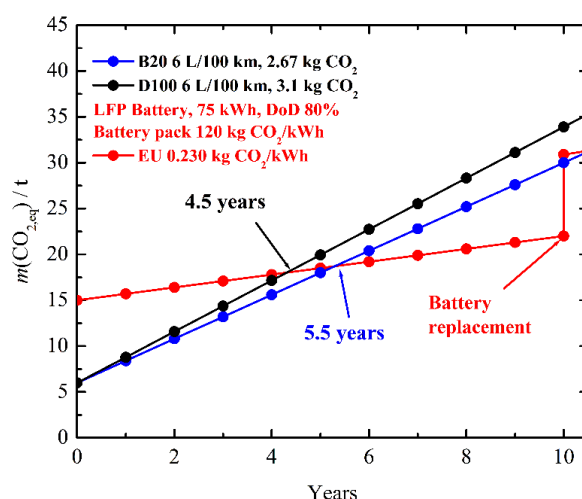


Figure 6. Comparison of EV-LFP based on the literature data [109], with pure diesel and B20 $\text{CO}_{2,\text{eq}}$ emission over 10.5 years.

Assessment (LCA) models, which frequently underestimate the cumulative “efficiency leak” occurring during the Grid-to-Battery (WtB) phase. While traditional studies often rely on static, manufacturer-provided figures of $16\text{--}18 \text{ kWh}/100 \text{ km}$, a stochastic evaluation of real-world variables reveals a much harsher reality. By accounting for ambient temperature extremes ranging from -10°C to $+40^\circ\text{C}$, where parasitic energy demands for thermal

stabilization and cabin climatization can increase consumption by 50%, the engineering focus must shift to the massive overhead of fast-charging cycles. For a standard 75 kWh (60 kWh for 80% DoD) battery pack, a 30-minute rapid charge necessitates currents of approximately 300 A. Even with an optimistic internal resistance at the milliohm scale, the resulting Joule heating (I^2R) generates kilowatts of localized thermal energy. During summer peaks (+40°C), the vehicle's Thermal Management System (TMS) must operate at maximum capacity to dissipate this internal heat while simultaneously fighting ambient thermal loads, consuming substantial additional power just to maintain battery stability. When these factors are combined with grid transmission losses (7–10%) and AC/DC conversion inefficiencies (~5%), the total energy withdrawal from the primary grid (Grid to Will, GtW) escalates to 32 kWh per 100 km. When scaled to a global fleet of, for example, 100 million electric vehicles, this results in an annual primary energy demand of 480 TWh. **Crucially, approximately 160 TWh of this total represents pure parasitic loss, energy dissipated as waste heat through resistance and active cooling—an amount equivalent to nearly five times the total annual electricity consumption of Serbia.** To offset the inefficiencies inherent in this 32 kWh reality, over 400,000 hectares of dedicated photovoltaic installations would be required, representing a massive land-use externality that further solidifies the "Grey Paradigm" and exposes the underlying "energy accounting fraud" in current green transition roadmaps.

IV.3. The Additional Problems

IV.3.1. Battery Recycling

One important conclusion derived from this deterministic audit is that current enthusiasm for battery recycling remains an industrial aspiration rather than a physical reality. While theoretical models projecting a closed-loop system are very often published [110-113], the global industrial landscape of 2024 reveals a stark divergence: the vast majority of battery chemistries undergo negligible elemental recovery due to unfavorable thermodynamic and economic margins, as well as significant environmental hazards associated with industrial-scale processing [114, 115].

NMC packs contain valuable cobalt and nickel – approximately 9 to 12 kilograms of cobalt and 30 to 40 kilograms of nickel in a 75 kWh pack. These metals drive recycling, with recovery rates reaching 90 percent for cobalt and 70 percent for nickel. However, the recycling process itself uses toxic organic solvents such as N-methyl-pyrrolidone (NMP), which is reprotoxic and hazardous [116, 117]. In addition, strong inorganic acids are used, generating hazardous waste streams. Recycling presents a devastating picture for LFP [118]. LFP recycling is an environmental disaster in waiting. A typical 75 kWh LFP pack contains no cobalt, no nickel, and only about 3 percent lithium by weight (approximately 2 to 3 kilograms of lithium per pack). The remaining 97 percent of the pack – iron, phosphate, graphite, aluminum, copper, binders, separators, electrolytes – has virtually zero market value. Traditional pyrometallurgical and hydrometallurgical methods are completely unprofitable for LFP. Industry experts conclude that LFP recycling has "almost no commercial value" because only about 5% of the battery's materials (lithium) have economic worth, while the remaining 95% must be paid to dispose of. The accumulation risk is catastrophic. Globally, only about 5% of lithium-ion batteries are recycled (note: *even the 5% is highly cited, the exact number is unclear; most often, the collection rate is reported*). In Europe, less than 5% were recycled in 2022, with only 1% of lithium recovered. For LFP, the situation is far worse. Because there is no economic incentive, most spent LFP batteries are simply not collected. They end up in landfills, stockpiled in warehouses, or illegally dumped. For example, the LiPF_6 salt in LIB organic solvents (highly flammable and irritable primarily the cyclic carbonate ethylene carbonate (EC) mixed with linear carbonates such as dimethyl carbonate (DMC), ethyl methyl carbonate (EMC), and diethyl carbonate (DEC) is known for causing systemic toxicity, cardiac arrest, respiratory failure, and death even with only limited physical contact with the compound [119]. The LiPF_6 salt also reacts readily with moisture or water to liberate dangerous hydrofluoric acid or gas, HF, which is a highly corrosive and toxic [119]. Over time, heavy metals and toxic compounds leach into soil and groundwater, causing long-term, irreversible environmental damage.

While NMC recycling has its own toxicity issues (solvents, acids), at least NMC batteries are collected and processed because they have value. LFP batteries have no such luck. Even when LFP batteries are collected, their low value means they are often stockpiled indefinitely, shipped to developing countries for crude and unsafe processing, or simply abandoned. The term "waste to wealth" for LFP is misleading – there is no wealth, only waste. This crisis is particularly compounded by the exponential market growth of Lithium Iron Phosphate (LFP) battery within the global growth to TWh capacity. Due to the absence of high-value cathode metals, the energy debt required for LFP elemental recovery far exceeds the market value of the reclaimed precursors, rendering the process an economic absurdity. Until the energy-to-mass debt of recovery is reconciled with the actual value of recovered materials, an exponential volume of toxic electronic waste will continue to be generated. It could be concluded that without a fundamental shift from speculative "green" marketing to rigid first-principles engineering, the transition to a global battery fleet risks substituting a carbon crisis with a permanent, multi-

generational ecological and toxicological debt. In addition, the question of safety is very relevant with growing numbers of Li-ion batteries, because there has been a significant rise in fires related to lithium-ion batteries. [120, 121]. This is not a future problem – it is already happening.

In recent studies [122], some authors proposed that a new market is rising that reuses retired full batteries for less rigorous applications before sending them for recycling. According to them these battery packs can be stacked in parallel to create high-capacity energy storage systems, extending life span second-life batteries (SLB) that may last an additional 5 to 30 years before reaching end-of-life (EOL) defined as 40% state of health (SOH) years. The claim of a 25+ year battery service horizon through second-life grid storage represents a structural mismatch with fundamental electrochemical kinetics. Under continuous calendar aging, a lithium-ion cell inevitably encounters a non-linear degradation knee-point, or "aging cliff," where internal resistance escalates and triggers rapid capacity decay after approximately 8 to 10 years. Attempting to harvest and repurpose these unevenly degraded, volatile cells into stationary infrastructure presents severe capital expenditure and safety challenges, as it requires operating a compromised chemical system at the very threshold of its kinetic collapse.

V.3.2. Ecological, Social, and CO_{2,eq} Impacts of Ore Mining and Deforestation

Moreover, subsequent calculations must account for the devastation of vast forested regions. These areas provide not only irreversible CO_{2,eq} assimilation, but also have profound ecosystem implications, as evidenced by the situation in Indonesia and other countries.

Mining for battery metals—lithium, cobalt, nickel, and copper—inevitably requires deforestation to access ore bodies and construct mine infrastructure. Clearing tropical rainforest releases approximately 500–800 tons of CO_{2,eq} per hectare from biomass burning and decomposition, while simultaneously destroying biodiversity hotspots and fragmenting habitats for endemic species [123]. Beyond forest loss, mining operations generate acid mine drainage and heavy metal leaching (copper, cobalt, nickel) that contaminates rivers and groundwater for decades, killing aquatic life and rendering drinking water unsafe for nearby communities. Soil degradation follows topsoil removal, leading to erosion and desertification that permanently removes agricultural land. Socially, these impacts disproportionately affect indigenous and rural populations: communities are forcibly displaced without adequate compensation, lose ancestral territories, and suffer increased rates of respiratory diseases and heavy metal poisoning from dust and contaminated water. In artisanal cobalt mines in the Democratic Republic of Congo, children as young as seven work in hazardous conditions without protective equipment, exposing them to cobalt dust and chronic health risks [25, 26]. Mining revenues in resource-rich regions often fuel armed conflicts, corruption, and weak governance, perpetuating poverty despite resource wealth. From a CO_{2,eq} perspective, direct emissions from diesel-powered haul trucks, excavators, and blasting add 30–50 kg CO_{2,eq} per ton of ore mined, while processing and refining ores (laterite nickel, copper smelting) consume fossil-fuel-based electricity, contributing another 50–100 kg CO_{2,eq} per ton of final metal. Converting forests to mine sites also eliminates future carbon sequestration potential (approximately 10–15 tons CO_{2,eq} per hectare per year), creating a long-term carbon debt that extends far beyond the battery's operational life. A single large lithium or nickel mine can destroy 1,000–5,000 hectares of forest, emit 0.5–2 million tons of CO_{2,eq} from deforestation alone, generate 100,000–500,000 tons of CO_{2,eq} annually from mining and processing, and affect 10,000–50,000 people through water contamination and displacement [124]. For example, nickel mines in Indonesia could destroy 500,000 hectares of forest [125]. These ecological, social, and climate burdens constitute a hidden debt that life cycle assessments rarely capture fully, yet they are essential for a complete cradle-to-gate accounting of battery production. Mining for battery metals drives deforestation that releases 500–800 tons of CO_{2,eq} per hectare, displaces indigenous communities, contaminates water resources, and in regions like the DRC involves child labor—creating ecological and social debts that life cycle assessments must account for alongside direct process emissions.

IV.4. What next?

The necessity of prioritizing biological restoration over industrial toxicity is reflected in the timeless wisdom of the prominent Serbian poet and physician, Jovan Jovanović Zmaj (1833–1904), who wrote: *"Wherever you find a suitable place, plant a tree! And the tree is grateful...., so it will reward either you or your brothers."* Global forests act as a massive natural sink, absorbing roughly 15.6 billion tons of carbon dioxide every year, over the years 2001–2019. Even though deforestation and wildfires release about 8.1 billion tons back into the air, the net result is a positive one for the planet. Ultimately, forests remain a critical "carbon sink," successfully removing 7.6 billion tons of CO₂ from the atmosphere annually and helping to slow down global warming [120]. The research from 2021 shows that the world's forests emitted an average of 8.1 billion metric tons of carbon dioxide into the

atmosphere each year due to deforestation and other disturbances, and absorbed 16 billion metric tons of CO₂ per year [126]. So, if we triple forests worldwide, a huge “decarbonization” will occur, much higher than with Li-ion batteries.

For example, in the near future, coastal Africa country and other waterless coastal nations should have mandate manifests in the strategic expansion of, for example, mangrove and acacia ecosystems [127]. The planting of these species serves as a fundamental biological catalyst; through nitrogen fixation, these “noble” trees convert atmospheric nitrogen into nutrient-rich organic matter, stabilizing the soil and initiating a cycle of fertility. Beyond their role as superior carbon sinks, mangroves function as biological soil engineers. Similar to the role of black locust (*Robinia pseudoacacia*) in stabilizing degraded soils and fixing nitrogen in temperate regions like the Balkans, mangroves and acacias transform barren, saline coastlines into fertile substrates. They trap sediments and produce organic detritus, literally building “new” land where once there was only salt and sand. In the context of the global 1.545 TWh energy transition, such nature-based solutions represent the only truly noble investment, one that returns fertility to the environment, rather than leaving behind a legacy of toxicological and economic debt.

One transition option will be hybrid cars using small sodium ion batteries (SIB) with the energy of 20-25 kWh (with a range of 110-140 km), for urban driving, and an internal combustion engine with a higher amount of biofuels for outside town. This will highly eliminate urban pollution, and biofuel will reduce fossil fuel emissions. Sodium ion batteries, due to being battery-based on bio hard carbon anodes, which emit during synthesis, are ten times smaller than for synthetic graphite, on an industrial scale, 3.2-3.6 t CO_{2,eq} per ton, compared to 30-40 t CO_{2,eq} per ton of anode materials, that for LIB account for 20-25% of total emission [128]. In addition, SIB uses aluminum as a negative current collector, less flammable electrolyte, *etc.* [129], and CATL (Contemporary Amperex Technology Co., Limited, China) launched SIB with 200 Wh/kg [130], that excide LFP specific energy. Due to the necessity of using BESS for sun and wind energy, SIB could play a significant role in the further installations of PV and wind grid.

Ultimately, the deterministic energy and carbon audit presented in this study reveals that the current “green” transition is dangerously decoupled from the thermodynamic and biogeochemical realities of our planet. While global policy remains hyper-focused on the industrial scaling of battery-grade metals, it fails to account for the systemic tipping points that are already being breached. The continued displacement of ecological debt through aggressive mining and carbon-intensive manufacturing accelerates the very warming that threatens to destabilize the Arctic permafrost. The resulting anaerobic decomposition and the catastrophic release of millions of tons of methane (CH₄) represent a positive feedback loop that no amount of lithium-based storage can offset. Furthermore, the rising ocean temperatures and acidification threaten the survival of fundamental calcifying organisms, such as foraminifera, whose extinction would signal a total collapse of marine carbon sequestration. As long as the scientific and political communities continue to prioritize “market-ready” solutions over first-principles ecological integrity, we risk sleepwalking into a climate feedback crisis. Therefore, the urgent need to improve biofuels technologies can be more than beneficial [131].

V. Conclusion

Using transparent, deterministic CtG-GtW, Cradle-to-Gate + Grid-to-Wheel methodology, based on battery energy produced in China (1.2 TWh) in 2024, assuming that 30% were NMC and 70% of LFP, the total emission of GHG in China is estimated to be ~180.5-233.7 CO_{2,eq}, with the most intense CO_{2,eq} are from metal salt precursor production, AAM synthesis, and battery assembly. From the sensitivity analysis, it is concluded that shifting from NMC to LFP has a significant effect. The total world emission for 1.545 TWh is estimated to be ~290 Mt CO_{2,eq} in average. Comparing the total emission outcome from world battery production with emissions from fossil fuels of 37.4 Gt in 2024, the percentage share of lithium-ion battery production in total emissions was ~0.8%. However, because the energy sector produces the most greenhouse gas emissions, 68% (25.4 Gt), of which 15% (3.8 Gt) originated from transport, 239.5-350 Mt CO_{2,eq}, correspond to 8% contributions, without EV being produced. Analysis of the cumulative CO_{2,eq} emissions comparison: diesel vs. electric vehicle, the lower emission of EV than diesel will occur after 6.3 years in the EU, 8 years in the USA, globally, and in China, practically never occurs, because after 8 to 10 years, the battery has to be replaced. The situation is slightly better with EV-LFP.

Considering that for 2030 ~5 TWh of battery production is expected, the total emission could reach ~3-4% of total CO_{2,eq} emission.

Accordingly, the rhetorical question stated in the manuscript is becoming a reality. The “butterfly effect” that started with a small Sony Li-ion battery (80 Wh/kg) in 1991 is no longer a theoretical abstraction; it is an imminent physical reality, and the time for statistical convenience has run out.

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