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ENVIRONMENTAL AND SOIL IMPACTS OF END-OF-LIFE LITHIUM-ION BATTERIES: QUANTITATIVE POLLUTION SCENARIOS, EMERGING CONTAMINANTS AND RECYCLING-ROUTE ASSESSMENT

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Abstract

The rapid deployment of lithium-ion batteries in electric vehicles and stationary energy-storage systems is generating an increasing end-of-life waste stream with potential impacts on soil, water and air. This study combines an integrative literature review with mass-balance calculations based on a representative NMC811 battery composition. Electric-vehicle battery deployment reached approximately 1.2 TWh in 2025 and may approach 3 TWh by 2030, corresponding to estimated masses of 7.0 and 17.5 million tonnes, respectively. Global end-of-life battery waste is projected to increase from 0.9 million tonnes in 2025 to 20.5 million tonnes in 2040. Under the 2040 landfill/leaching stress scenario, potential releases were estimated at 7,754 t Li, 4,949 t Ni, 3,075 t Cu, 1,655 t Mn and 623 t Co. Relevant pollutants also include fluorides, electrolyte degradation products, fine cathode particles and persistent fluorinated compounds. High collection rates, chemistry-based sorting, fire-safe storage, controlled recycling and integrated soil and water monitoring are essential for reducing environmental risks and improving material circularity.

Key words*: battery recycling, end-of-life batteries, lithium-ion batteries, soil contamination, waste management.*

INTRODUCTION

Lithium-ion batteries (LIBs) have become essential for electric mobility, portable electronics and stationary energy storage because of their high energy density, efficiency and long operating life. According to the latest international outlook, global electric-vehicle battery deployment reached approximately 1.2 TWh in 2025 and is expected to approach 3 TWh by 2030. This rapid increase in deployment is creating a delayed but substantial flow of batteries that will require collection, transport, reuse and recycling after reaching the end of their service life (Harper et al., 2019; International Energy Agency, 2026).

End-of-life lithium-ion batteries, hereafter referred to as EoL-LIBs, contain a complex mixture of potentially recoverable and hazardous materials. Depending on cathode chemistry, these materials may include Li, Ni, Co, Mn, Fe and P, together with Cu and Al current collectors, graphite, polymeric binders, separators, organic solvents and fluorinated lithium salts. NMC and NCA batteries contain economically valuable Ni and Co, whereas LFP batteries contain no Ni or Co in the active cathode material but retain Li, Cu, Al, graphite, phosphate compounds and fluorinated electrolyte components (Harper et al., 2019; Mrozik et al., 2021).

Environmental releases may occur through corrosion, mechanical damage, uncontrolled shredding, landfill disposal, battery fires and inadequately controlled recycling operations (Mosnegutu et al., 2025a). Soil may receive contaminants through direct leakage, contaminated dust, atmospheric deposition, leachate, firefighting water and the disposal of slags, sludge or other process residues. The subsequent mobility and persistence of these contaminants are influenced by pH, redox

conditions, soil texture, clay content, organic matter, rainfall and groundwater depth (Bolan et al., 2021; Mrozik et al., 2021).

The environmental assessment of EoL-LIBs must therefore consider more than lithium and the economically valuable cathode metals. Fluorides, electrolyte decomposition products, fine cathode and graphite particles, and emerging per- and polyfluoroalkyl substances may contribute to soil acidification, metal mobilisation, groundwater contamination and ecological toxicity (Guelfo et al., 2024; Rensmo et al., 2023; Mosnegutu et al., 2025b; Shojaei et al., 2025). Accordingly, the objectives of this study were to estimate the mass of batteries entering use and the projected EoL-LIB waste stream, quantify the potential release of selected contaminants under three management scenarios, examine a site-scale soil-contamination scenario involving an NMC811 battery pack, and compare the environmental release pathways associated with uncontrolled disposal, battery fires, mechanical pretreatment, pyrometallurgy, hydrometallurgy and direct recycling.

MATERIALS AND METHODS

The study was conducted as an integrative review combined with mass-balance calculations, quantitative pollution-scenario modelling and a technology-specific environmental assessment. Scientific articles published mainly between 2015 and 2026 were identified through Scopus, Web of Science, ScienceDirect, SpringerLink, PubMed and Google Scholar, while older regulatory and foundational sources were retained where relevant. Institutional data were obtained from the International Energy Agency, the United Nations Development Programme, Argonne National Laboratory and the European Commission. The literature review focused on battery deployment, EoL-LIB waste generation, contaminant composition, environmental release pathways, soil impacts and recycling technologies.

Battery mass was estimated from energy deployment using a representative pack-level mass of 5.85 kg kWh^{-1}. This central value represents a simplified mixed NMC–LFP battery flow and falls within the range derived from BatPaC automotive pack simulations. In practical units, the calculation was:

Battery mass (Mt) = energy deployment (TWh) × 5.85 Mt TWh^{-1}.

Consequently, 1.2 TWh corresponds to approximately 7.02 Mt of batteries, while 3.0 TWh corresponds to approximately 17.55 Mt. These values represent batteries newly introduced into use and not waste generated during the same year (Ahmed et al., 2023; International Energy Agency, 2026).

Annual EoL-LIB quantities were obtained from the UNDP projection, which reports 0.90 Mt in 2025, 1.85 Mt in 2030, 7.85 Mt in 2035 and 20.50 Mt in 2040. The published values were used directly, without interpolation between the reported years (United Nations Development Programme, 2024).

An NMC811 battery pack was selected as a representative metal-containing chemistry for the pollution-screening model. The active cathode material, $LiNi_{0.8}Mn_{0.1}Co_{0.1}O_2$, contains approximately 7.13% Li, 48.27% Ni, 5.65% Mn and 6.06% Co by mass. Assuming that active cathode material represents 25% of total pack mass, the corresponding pack-level fractions were estimated at 1.78% Li, 12.07% Ni, 1.41% Mn and 1.52% Co. Cu was assumed to represent 7.5% of pack mass. These fractions are modelling assumptions and do not describe every commercial battery pack (Ahmed et al., 2023; Rinne et al., 2024).

The potential annual release of contaminant *i* was calculated using the following mass-balance relationship:

Released mass = annual battery waste × mismanaged fraction × contaminant fraction × environmental release fraction.

Three quantitative screening scenarios were evaluated. Scenario A assumed that 1% of EoL-LIBs were mismanaged, with environmental release of 10% of Li and 1% of Ni, Co, Mn and Cu. Scenario B assumed 5% mismanagement, with release of 42.5% of Li, 11.45% of Mn and 4% of Ni, Co and Cu. Scenario C represented severe fragmentation, fire or uncontrolled disposal, assuming 15% mismanagement and release of 75% of Li, 20% of Mn and 10% of Ni, Co and Cu. Scenario B release factors reflect the order of magnitude reported in landfill-related leaching studies, whereas Scenarios A and C were used as lower and upper sensitivity

boundaries (Bolan et al., 2021; Mrozik et al., 2021). These scenarios represent comparative screening conditions rather than predictions of the actual proportion of battery waste that will be released into the environment.

In addition to the quantitative NMC811 release scenarios, a technology-specific assessment was performed for uncontrolled disposal, battery fires, mechanical pretreatment, pyrometallurgy, hydrometallurgy and direct recycling. Each management or treatment option was compared qualitatively according to its main potential releases, the principal affected environmental medium and the expected environmental implications. The assessment considered metal-bearing leachates and residues, fluorinated emissions, cathode and graphite particles, electrolyte vapours, acidic or saline wastewater, sludge and solid process residues. The comparison was based on evidence reported for EoL-LIB pollution and recycling systems (Harper et al., 2019; Mrozik et al., 2021; Rensmo et al., 2023; Claassen et al., 2024; Guelfo et al., 2024; International Energy Agency, 2024; Rinne et al., 2024; Aiello et al., 2025; Shojaei et al., 2025).

For example, Li release under Scenario B in 2040 was calculated as:

$20{,}500{,}000\ t \times 0.05 \times 0.0178 \times 0.425 = 7{,}754\ t\ Li\ yr^{-1}$.

The calculated values represent contaminant masses potentially released from the waste stream and not final concentrations in a particular soil or water body. Site-level concentrations require additional information on the affected area, mixing depth, hydrological conditions, soil properties, background concentrations and contaminant transport. Therefore, the results should be interpreted as comparative pollution-potential indicators rather than site-specific environmental predictions (Mrozik et al., 2021).

RESULTS AND DISCUSSIONS

The conversion of energy deployment into mass indicates that approximately 7.0 Mt of batteries may have entered electric-vehicle applications in 2025. Deployment approaching 3 TWh in 2030 would correspond to approximately 17.5 Mt of batteries per year. The range could be wider because NMC packs generally have higher gravimetric energy density than LFP packs, while changes in cell-to-pack design or the introduction of solid-state and sodium-ion batteries may reduce or increase the future mass per unit of energy (Ahmed et al., 2023; International Energy Agency, 2026).

The annual quantity of EoL-LIBs is much lower than annual deployment because batteries normally remain in service for several years. However, the waste stream is projected to expand from 0.90 Mt in 2025 to 20.50 Mt in 2040, representing an increase of more than twenty times. Production scrap currently represents an important recycling feedstock, but approximately 94% of the projected 2040 flow is expected to originate from batteries removed from vehicles and other applications. The estimated mass of newly deployed batteries and the projected evolution of the EoL-LIB waste stream are summarised in Table 1.

Table 1. Estimated battery deployment and projected end-of-life waste

Year	EV battery deployment	Estimated mass newly deployed	Projected EoL-LIB waste
2025	≈1.2 TWh	≈7.02 Mt	0.90 Mt
2030	≈3.0 TWh	≈17.55 Mt	1.85 Mt
2035	n.a.	n.a.	7.85 Mt
2040	n.a.	n.a.	20.50 Mt

Source: Authors' calculations based on Ahmed et al. (2023), International Energy Agency (2026), and United Nations Development Programme (2024).

As shown in Table 1, the projected EoL-LIB waste stream increases rapidly after 2030 and gradually shifts from production scrap towards used batteries. This transition has practical consequences because production waste is generally homogeneous, documented and relatively uncontaminated, whereas used battery packs differ in chemistry, state of charge, degradation level, accident history and structural design. Recycling systems will therefore require improved identification, automated disassembly, chemistry-based sorting and safer transport as the proportion of used packs increases (Harper et al., 2019; International Energy Agency, 2024).

Li is potentially important because it is relatively mobile in many soils and may be weakly retained compared with transition metals. A landfill-related study synthesised by Mrozik et al. (2021) reported that approximately

42.5% of the Li inventory entered the leachate, compared with less than 4% for Ni, Co, Cu, Al and Fe. Mn release was approximately 11.45%. These values are not universal leaching factors, but they illustrate that contaminant mobility does not necessarily correspond to the total metal content of the battery (Bolan et al., 2021; Mrozik et al., 2021).

Ni, Co and Cu generally have stronger interactions with clay minerals, iron and manganese oxides and soil organic matter. Nevertheless, their mobility and bioavailability may increase when soil pH decreases. At elevated concentrations, Ni and Co can affect microbial enzymatic activity and plant growth, while Cu may impair microbial processes and modify the physical behaviour of clayey soils. Mn presents redox-dependent behaviour and may become more soluble under reducing or waterlogged conditions (Mrozik et al., 2021; Mishra et al., 2026).

Electrolyte degradation may amplify metal contamination. $LiPF_6$ is moisture-sensitive and may form HF and other fluorinated products. Local acidification can increase metal solubility, while organic carbonate solvents and their degradation products may affect soil microorganisms. Simulated landfill experiments have reported PFAS concentrations of up to approximately 100 mg L^{-1} and inorganic fluoride concentrations of up to 1.4 g L^{-1} for specific batteries and test conditions. These experimental maxima should not be treated as typical landfill concentrations but demonstrate the possibility of concentrated fluorochemical releases (Guelfo et al., 2024; Shojaei et al., 2025).

Battery fires create an acute pathway from the waste stream to air, soil and water. Experimental studies reported gaseous HF emission factors of approximately 10-81 mg Wh^{-1}, depending on chemistry and test conditions. Metal-containing particles and fluorinated combustion products may be deposited on soil or transported in firefighting water. Following the January 2025 Moss Landing battery-storage fire, increased Ni, Mn and Co concentrations were measured in nearby surface wetland soils, with a Ni ratio consistent with NMC532 cathode material. Enrichment was concentrated in the upper few millimetres and subsequently decreased after rainfall and tidal flushing, demonstrating both deposition and remobilisation (Claassen et al., 2024; Aiello et al., 2025).

The main EoL-LIB pollutants, their sources, dominant environmental pathways and potential effects are summarised in Table 2.

Table 2. Main EoL-LIB pollutants, pathways and environmental effects

Pollutant or material	Main source	Dominant pathway	Potential environmental effect
Li	Cathode and electrolyte	Leachate and groundwater transport	Salinity and ionic imbalance; plant uptake
Ni and Co	NMC, NCA and LCO cathodes	Leachate, dust and fire deposition	Microbial toxicity, phytotoxicity and oxidative stress
Mn	NMC and LMO cathodes	Leachate under acidic or reducing conditions	Nutritional imbalance and toxicity at elevated concentration
Cu	Anode current collector	Dust, leachate and recycling residues	Microbial inhibition and alteration of clayey soil properties
HF and inorganic fluorides	$LiPF_6$ degradation and fires	Gas emissions, leachate and firefighting water	Acidification, metal mobilisation and acute toxicity
PFAS/bis-FASI	Some electrolytes and fluorinated materials	Manufacturing, recycling and landfill leachate	Persistent and mobile contamination
Cathode particles and graphite	Shredding and fires	Atmospheric dispersion and soil deposition	Particle toxicity and gradual metal dissolution

Source: Authors' synthesis based on Mrozik et al. (2021), Rensmo et al. (2023), Claassen et al. (2024), Guelfo et al. (2024), Aiello et al. (2025), and Shojaei et al. (2025).

As shown in Table 2, EoL-LIB pollution involves both dissolved and particulate contaminants transported through leachate, atmospheric deposition, firefighting water and recycling residues. In addition to their chemical and biological effects, battery-derived contaminants may also influence the physical and geotechnical properties of soil. In controlled experiments using clayey soil contaminated with 3-9% Cu, Co or Ni powder, maximum permeability increases of 84%, 46% and 68%, respectively, were recorded, while shear strength decreased by 34%, 11% and 27%. These concentrations are much higher than typical diffuse contamination and should not be directly extrapolated to ordinary sites. Nevertheless, they demonstrate that severe battery-related contamination may influence soil plasticity, permeability, consolidation and geotechnical stability, in addition to chemical and biological functions (Mishra et al., 2026).

A representative 60 kWh NMC811 pack with an estimated mass of 312 kg contains approximately 5.6 kg Li, 37.7 kg Ni, 4.4 kg Mn, 4.7 kg Co and 23.4 kg Cu under the adopted composition assumptions. If 4% of the Ni inventory were released and mixed into 26 t of soil, corresponding to 100 m² to a depth of 0.20 m and an apparent density of 1.3 t m^{-3}, the Ni concentration would increase by approximately 58 mg kg^{-1}. When added to a background value of 20 mg kg^{-1}, the resulting concentration of approximately 78 mg kg^{-1} would slightly exceed the Romanian alert threshold of 75 mg kg^{-1} for sensitive land use (Ahmed et al., 2023; Ministry of Waters, Forests and Environmental Protection, 1997).

This calculation represents a conservative hotspot rather than an expected average concentration. It assumes that released material is retained within a small soil volume and does not account for runoff, wind dispersion, deep leaching or removal during emergency response. It nevertheless shows that even one damaged traction battery may generate locally relevant contamination when its contents are fragmented or deposited over a limited area (Mrozik et al., 2021).

Taken together, the experimental, field and modelling evidence indicates measurable forms of environmental degradation rather than only a theoretical possibility of contamination. Reported effects include HF emissions during battery fires, the deposition and remobilisation of cathode metals in surface soils, increased soil permeability, reduced shear strength and the potential exceedance of soil-quality thresholds (Claassen et al., 2024; Aiello et al., 2025; Mishra et al., 2026). Although these effects may be concentrated at individual accident, disposal or recycling sites, they demonstrate that EoL-LIB releases can affect air quality, soil chemistry, contaminant mobility, hydraulic behaviour and geotechnical stability.

At the global scale, the 2030 stress scenario produced potential releases of approximately 700 t Li, 447 t Ni, 278 t Cu, 149 t Mn and 56 t Co. These estimates were calculated by the authors using the waste projections reported by the United Nations Development Programme (2024) and the release assumptions derived from Mrozik et al. (2021). Although these values remain small compared with total battery waste, they represent significant dispersed contaminant loads, particularly when concentrated around informal processing sites, landfills or storage facilities.

By 2040, the strong increase in the EoL-LIB stream becomes the dominant driver of potential pollution. Scenario A, representing relatively effective control, produced releases ranging from approximately 29 t Mn to 365 t Li. Under Scenario B, the calculated quantities reached thousands of tonnes for Li, Ni and Cu. Scenario C generated substantially larger upper-bound values and illustrates the consequences of combining high waste quantities with fragmentation, fires or uncontrolled disposal. The increase in contaminant releases under Scenario B between 2030 and 2040 is illustrated in Figure 1, while the results obtained for all three scenarios in 2040 are presented in Table 3. As shown in Figure 1, lithium had the highest estimated release under the landfill/stress scenario, increasing from approximately 700 t $year^{-1}$ in 2030 to 7,754 t $year^{-1}$ in 2040. Nickel increased from 447 to 4,949 t $year^{-1}$, while the estimated Cu release rose from 278 to 3,075 t $year^{-1}$. Lower quantities were calculated for Mn and Co; however, their environmental relevance remains considerable because of their potential ecotoxicity and increased mobility under acidic or reducing soil conditions. The marked increase between 2030 and 2040 is mainly explained by

the expansion of the global EoL-LIB waste stream, since the assumed mismanagement and contaminant-release fractions remained unchanged.

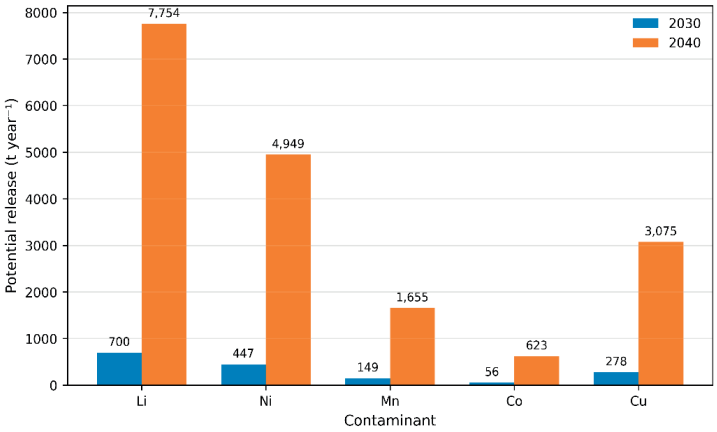


Figure 1. Estimated potential releases of lithium, nickel, manganese, cobalt and copper under the landfill/leaching stress scenario in 2030 and 2040 (Source: authors' calculations based on Mrozik et al., 2021, United Nations Development Programme, 2024, and the modelling assumptions described in Materials and Methods)

Table 3. Estimated potential contaminant releases from NMC811-type EoL-LIB waste in 2040

Scenario	Li	Ni	Mn	Co	Cu
A	365	247	29	31	154
B	7,754	4,949	1,655	623	3,075
C	41,051	37,115	8,672	4,674	23,062

Source: Authors' calculations based on Mrozik et al. (2021), United Nations Development Programme (2024), and the assumptions described in Materials and Methods.

Values are expressed in t yr^{-1}. A - controlled management; B - landfill/leaching stress; C - severe mismanagement.

The values presented in Table 3 are conservative screening estimates rather than forecasts. The future waste stream will include an increasing proportion of LFP batteries, which would reduce Ni and Co loads, while released contaminants may be distributed among soil, water, air, sludge and engineered residues. Consequently, the results are primarily useful for comparing management scenarios and demonstrating the importance of collection efficiency rather than predicting pollution at a specific location (Harper et al., 2019; International Energy Agency, 2024).

Mechanical pre-treatment generally involves discharge, dismantling, crushing, sieving and physical separation. It concentrates cathode and anode materials into black mass but may generate metal-bearing dust, volatile electrolyte compounds and fire risks. The quality and chemical homogeneity of black mass strongly affect the environmental and economic performance of subsequent recycling stages (Harper et al., 2019; Rinne et al., 2024).

Pyrometallurgical recycling can process mixed or contaminated feedstocks and efficiently transfer Ni, Co and Cu into an alloy. However, the process requires high temperatures, destroys graphite and organic components and may transfer Li, Al and Mn to slag. Fluorinated emissions require gas treatment, while the long-term leaching behaviour of slag and dust must be assessed before reuse or disposal. The low Ni and Co content of LFP batteries further reduces the economic attractiveness of conventional pyrometallurgy for this chemistry (Harper et al., 2019; Mrozik et al., 2021).

Hydrometallurgical recycling uses acid or alkaline leaching, precipitation, solvent extraction and crystallisation to recover Li and transition metals. It generally operates at lower temperatures and may achieve high recovery efficiencies, but the process consumes water, acids, bases and reducing agents and generates saline wastewater and sludge. A process-based life-cycle assessment found that reducing the initial acid concentration from approximately 2 M to below 0.5 M lowered several environ-

mental impacts by approximately 30-40%, illustrating the importance of reagent optimisation and internal recycling (Rinne et al., 2024).
Direct recycling seeks to preserve the functional structure of cathode materials through separation, purification, relithiation and thermal treatment. It may reduce energy and chemical requirements because the cathode is not completely decomposed into metals or salts. Its main limitations are the need for accurate chemistry identification, clean feedstocks and control of Cu, Al and electrolyte contamination. Direct recycling may be particularly important for LFP, where recovery based solely on the market value of individual metals is less favourable (Harper et al., 2019; International Energy Agency, 2024).
Recent hydrometallurgical work on waste LFP batteries reported Li leaching efficiencies of 54-97% using organic acids and reductants, with Li recovery of approximately 90% after precipitation. Such results indicate that LFP recycling can be technically feasible, although industrial environmental performance will also depend on reagent recovery, wastewater treatment, energy use and product quality (Avcı & Özdemir, 2026).
To assess the influence of management and treatment technologies on environmental pollution, the principal release pathways associated with uncontrolled disposal, battery fires, mechanical pretreatment, pyrometallurgy, hydrometallurgy and direct recycling are compared in Table 4.

Table 4. Technology-specific pollution pathways and environmental implications of EoL-LIB management and recycling

Technology or scenario	Main potential releases	Main affected medium	Environmental implication
Uncontrolled disposal/landfill	Li, Ni, Co, Mn, Cu, fluorides, PFAS	Soil and groundwater	Chronic leaching and contaminant accumulation
Battery fire	HF, metal-rich particles, fluorinated compounds	Air, surface soil and water	Acute deposition and remobilisation
Mechanical pretreatment	Fine cathode dust, graphite, electrolyte vapours	Air and workplace surfaces	Particulate and VOC exposure
Pyrometallurgy	Combustion gases, slag containing Li, Al and Mn	Air and solid residues	High energy demand and residue-management risks
Hydrometallurgy	Acidic or saline wastewater and metal-bearing sludge	Surface water, groundwater and soil	High recovery but significant wastewater burden
Direct recycling	Dust, rejected mixed materials and damaged cathode fractions	Air and solid waste	Lower processing burden but requires clean feedstock

Source: authors' synthesis based on Harper et al. (2019), Mrozik et al. (2021), Rensmo et al. (2023), Claassen et al. (2024), Guelfo et al. (2024), International Energy Agency (2024), Rinne et al. (2024), Aiello et al. (2025), and Shojaei et al. (2025).

As shown in Table 4, EoL-LIB management and recycling options generate different release profiles and may transfer environmental burdens between air, soil, water and solid residues. The dominant environmental compartment therefore changes according to the management route: uncontrolled disposal primarily threatens soil and groundwater, battery fires affect air and surface environments, pyrometallurgy concentrates burdens in gaseous emissions and solid residues, while hydrometallurgy transfers a substantial part of the burden to wastewater and sludge.
Recycling performance should therefore not be evaluated only through the total percentage of battery mass processed. A system may achieve a high mass-based efficiency by recovering steel, Cu and Al while losing Li, graphite, fluorinated compounds or fine active material. Environmental assessment should therefore combine overall recycling efficiency with element-specific recovery, energy and reagent consumption, wastewater quality and the leaching stability of final residues (Rensmo et al., 2023; Rinne et al., 2024).
European legislation requires a minimum recycling efficiency of 65% for lithium-based batteries by the end of 2025 and 70% by the end of 2030. Material recovery targets are 90% for Co, Cu, Ni and Pb and 50% for Li by the end of 2027, increasing to 95% and 80%, respectively, by the end of 2031. These targets create

incentives for improved Li recovery and more complete material balances, but effective environmental protection also depends on collection, traceability and control of fluorinated emissions and residues (European Commission, 2025; European Parliament and Council of the European Union, 2023).

The most favourable pollution-prevention scenario combines collection rates approaching 100%, digital battery identification, separation by chemistry, fire-resistant storage, rapid isolation of damaged batteries and recycling routes selected for the feedstock. Monitoring programmes around storage and recycling facilities should include Li, Ni, Co, Mn, Cu, total and soluble fluorides, pH, electrical conductivity and groundwater quality. PFAS screening should be considered where fluorinated electrolytes or polymers are produced or processed (Guelfo et al., 2024; Rensmo et al., 2023).

CONCLUSIONS

Global electric-vehicle battery deployment reached approximately 1.2 TWh in 2025, corresponding to an estimated 7.0 Mt of batteries, and may approach 17.5 Mt annually by 2030.

The annual EoL-LIB waste stream is projected to increase from 0.90 Mt in 2025 to 20.50 Mt in 2040, requiring a rapid expansion of collection, storage and recycling infrastructure.

Environmental degradation may result from the combined action of Li, Ni, Co, Mn, Cu, fluorides, electrolyte degradation products, fine cathode particles and persistent organofluorinated compounds. Soil acidification may amplify pollution by increasing metal mobility and bioavailability.

The stress scenario for 2040 produced potential releases of approximately 7,754 t Li, 4,949 t Ni, 3,075 t Cu, 1,655 t Mn and 623 t Co. These values are sensitivity indicators rather than predictions, but they demonstrate that small percentages of mismanaged waste can generate substantial contaminant loads.

NMC and NCA waste presents a higher Ni and Co contamination potential, whereas LFP reduces these metal-related risks but maintains the need to recover Li, Cu, graphite, phosphate compounds and fluorinated electrolyte materials.

Pyrometallurgy is robust for mixed feedstocks but may transfer environmental pressure to energy use, emissions and slag. Hydrometallurgy provides broader metal recovery but requires strict control of reagents and wastewater. Direct recycling offers high circularity potential for clean, chemistry-specific streams.

Collection efficiency is the principal control variable. Chemistry-based sorting, traceability, fire-safe storage, capture of firefighting water and site-specific soil and groundwater monitoring are required to prevent the transfer of battery pollution between environmental compartments.

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