Economic Viability of Lithium-Ion Battery Recycling

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Abstract

The rise of electric vehicles (EVs) has created a greater focus on lithium ion battery (LIBs) recycling. This paper compares the economic viability of LIB recycling in the United States, China, and Europe. It examines the three major recycling processes: hydrometallurgy, pyrometallurgy, and direct recycling. The analysis looks at how viable these processes are for both lithium iron phosphate (LFP) and nickel manganese cobalt (NMC) chemistries. It breaks down both the capex and opex as well as what the value of the recovered materials are based on current market values. This study and its underlying model may be used as a guide for investment and to direct policy for long term viable LIB recycling processes.

Introduction

There has been an increase in the use of electric vehicles and stationary energy storage systems (SESS) due to global decarbonization goals that started in 2015. In the United States 12.3 GW of SESS was installed in 2024. This was a 33% year on year increase [1]. The need for lithium ion batteries (LIBs) is increasing every year. This rise in LIB use creates new problems in end-of-life (EOL) management and the recovery of critical materials. Recycling EOL batteries would allow for the critical materials within the batteries like lithium, cobalt, nickel, and graphite to not go to waste and be reused [2]. This also reduces the dependency battery manufactures have on buying new, virgin materials to create batteries [3]. Despite there being many technical advances, the financial viability of large scale LIB recycling remains uncertain and varies significantly across regions, battery chemistries, and the recycling process used [4].

There are three major processes for LIB battery recycling. These processes are Pyrometallurgy, Hydrometallurgy, and Direct Recycling. Pyrometallurgy (Pyro) is a process that uses high temperature smelting in order to recover metallic alloys like nickel and cobalt. While this is a mature process it still has flaws. This process is the most energy intensive of the three and loses most, if not all, of the lithium and graphite within the batteries. Hydrometallurgy (Hydro) is a more selective recovery process that uses chemical leaching and precipitation to recover materials. This process has its own downsides in that it requires many different types of chemicals to recover materials and has high wastewater treatment costs. Direct recycling is the newest process still under development.. This process differs from the other two in that instead of breaking down the chemical structures in the battery it rejuvenates the cathode materials instead. This leads to lowering energy use and lower processing costs [5]. Direct recycling is being looked at due to the rising use of lithium iron phosphate (LFP) batteries, which contain fewer high value metals compared to nickel manganese cobalt (NMC) batteries [6].

In this paper we consider the financial viability of each process in the United States, China, and the EU, the three major geographic regions for LIB recycling. The opex consists of factors like the cost of buying the EOL batteries, transportation, energy, labor, etc. Financial viability is also heavily dependent on the amount of materials recovered and the current market price these materials sell for. China currently has the biggest market for LFP batteries. They use them the most and also are their biggest recycler. Recycling in China is supplemented by various government mandates and has lower costs for things like energy and labor. Europe’s policy framework focuses on making a more circular economy. The United States currently remains at a more developmental stage. It has many emerging facilities for both hydro and direct recycling with more regulatory support being developed [7].

Despite a growing technical literature on LIB recycling, comprehensive economic comparisons across different recycling processes, chemistries and regional markets remain limited and inconsistent due to varying assumptions and data scopes. This paper will address the economic viability of LIB recycling across the United States, China, and Europe. It will compare hydro, pyro, and direct recycling processes for both LFP and NMC chemistries, analyzing both capex and opex, the value of recovered materials, and various regional factors. It will introduce a simple model to analyze the financial viability. While there have been different models published before like Everbatt [8]. Everbatt’s model is spread out across more than 20 sheets where formulas cross reference between sheets, making them hard to track and see what affects each other. Adapting a model to rapidly evolving recycling processes is difficult and the authors of EverBatt are not keeping it current. This new model that we developed can have values easily updated and will show a corresponding update to capex, opex, or revenue that can be analyzed. This flexibility allows users to easily test how variations in different parameters affect the gross profits. The model can also be updated with new process data or regional economic conditions, making it a useful tool for scenario based forecasting.

Regional Overview of LIB Recycling

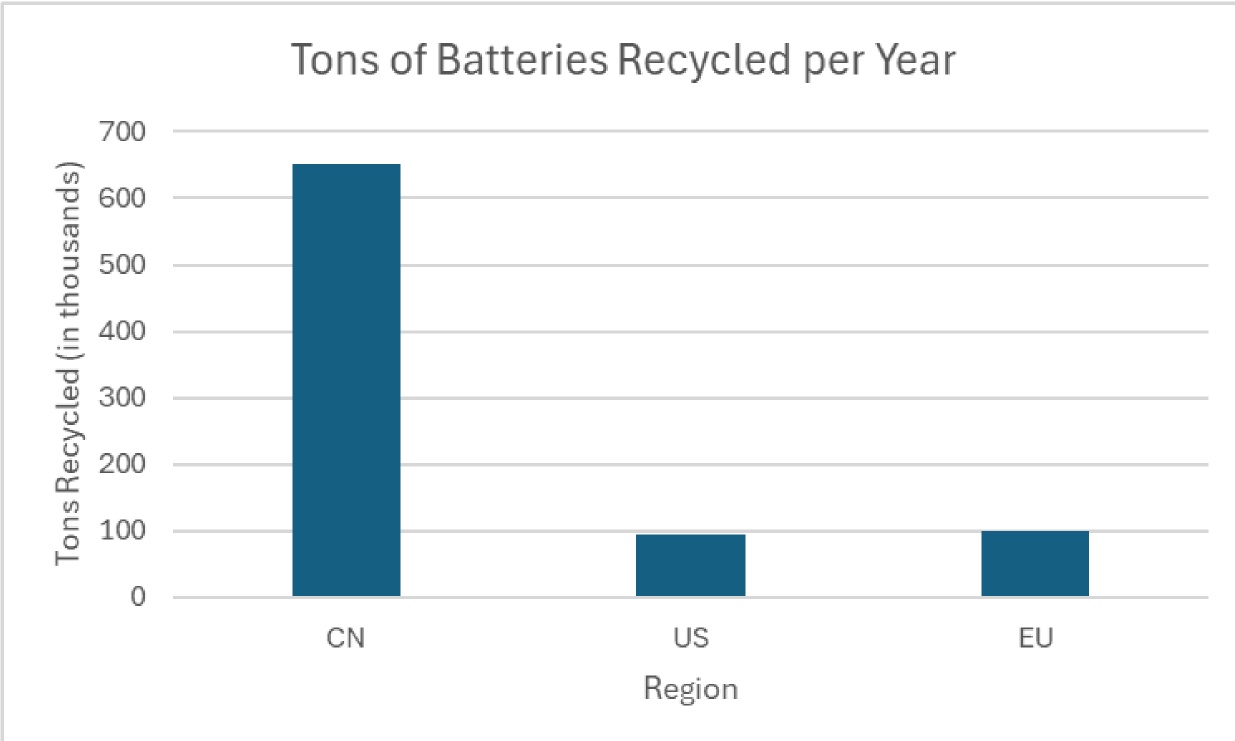

Fig. 1 Total annual volume of lithium-ion batteries recycled in China, the United States, and Europe.
Bars represent the estimated tons of EOL batteries processed per year in each region based on available industry and regulatory data.

China: Policy Driven Expansion and Cost Competitiveness

China currently dominates the global LIB recycling industry, accounting for more than 70% of the current global recycling capacity (Fig. 1) [9, 10]. There are many large recyclers in China like GEM, Brunp (CATL), and Huayou Cobalt. China's growth has mainly been driven by various state policies, including the Ministry of Industry and Information Technology (MIIT) recycling management regulations and the Extended Producer Responsibility (EPR) framework. These mandate traceability and promote closed-loop material flows [11]. Hydro remains the dominant process in the region while direct recycling research is beginning to scale through pilot programs (Fig. 2). Overcapacity and uneven collection systems are still ongoing challenges in China. The majority of recycling facilities in China are located within a few of its provinces and so collecting recycling batteries outside of those provinces is much more difficult and less developed (Fig. 3). Recent estimates indicate that by the end of 2024, China's installed recycling capacity reached about 4.2 million tons per year, while only 650,000 tons of spent batteries were actually processed. This is only a utilization rate of around 15% [12, 13]. Recycling profitability is heavily dependent on government incentives available and having a stable flow of feedstock from partnerships with the original equipment manufacturers (OEM) [14]. Regulations in China are tightening. An example of this is a draft specification issued in August 2024 by the MIIT mandates improved traceability and industrial-park location for recycling firms [15]. Domestic feedstock supply is constrained, emphasising the value of OEM partnerships.

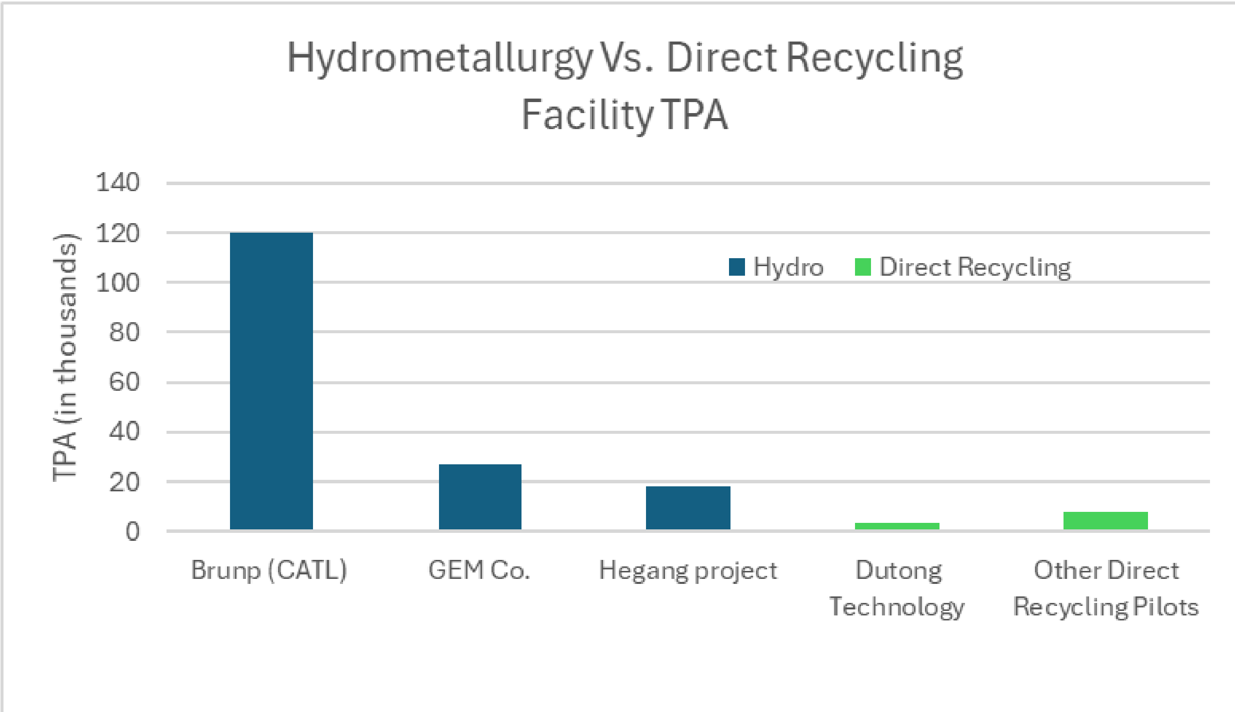

Fig. 2 Facility TPA for hydrometallurgy and direct recycling plants in China.
Large hydrometallurgical plants operate at far higher capacities than existing direct-recycling pilots. [16, 17, 18, 19]

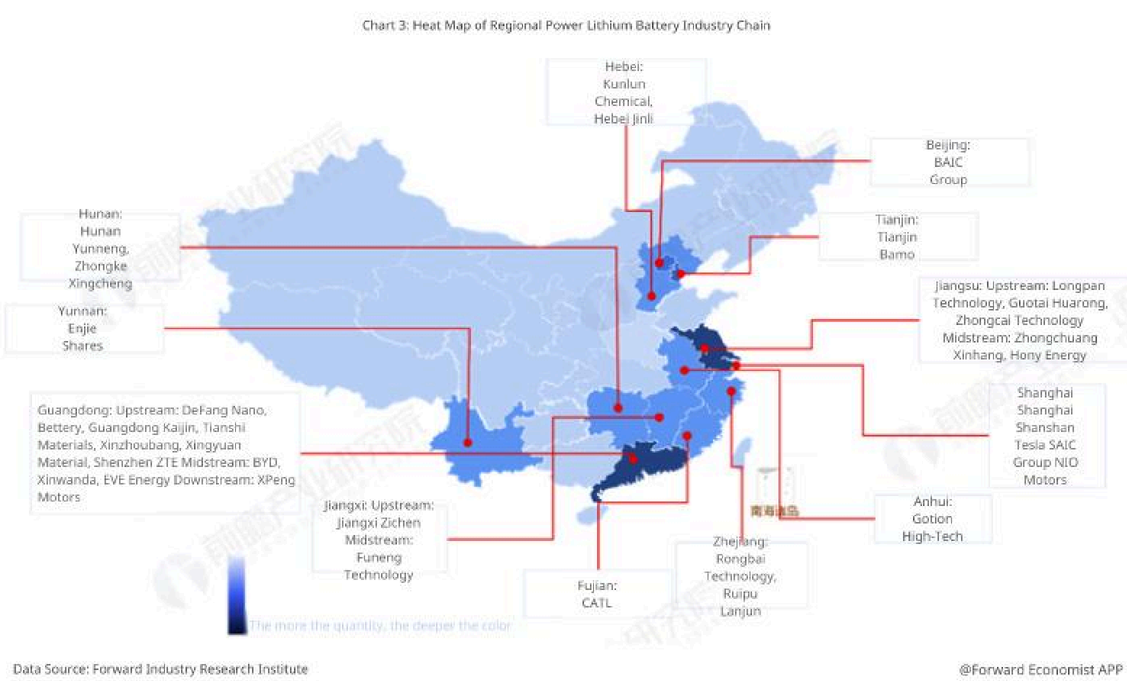

Fig. 3 Geographic distribution of major lithium-ion battery recycling in China.
The map highlights key provinces with significant hydrometallurgical recycling capacity and battery-materials manufacturing activity, illustrating the regional concentration of China's recycling industry. [20]

Europe: Regulation and Circular Economy

In Europe, recycling growth has been primarily policy-driven under the EU Battery Regulation that was introduced in 2023. This regulation enforces strict collection, recycling efficiency, and material recovery targets that recyclers have to follow [21]. European recyclers such as Umicore, and Northvolt have become leaders in sustainable and traceable supply chains. The region has higher energy costs and more stringent environmental standards than the U.S. and China. Even with these increased costs the region has strong public funding and a coordinated circular-economy framework that helps to offset it [22]. Most facilities rely on a combined hydro and pyro process. There is growing development in direct recycling to meet future recovery mandates. Europe's more fragmented collection of LIBs and higher capex limit the profitability compared to China (Fig. 4) [23].

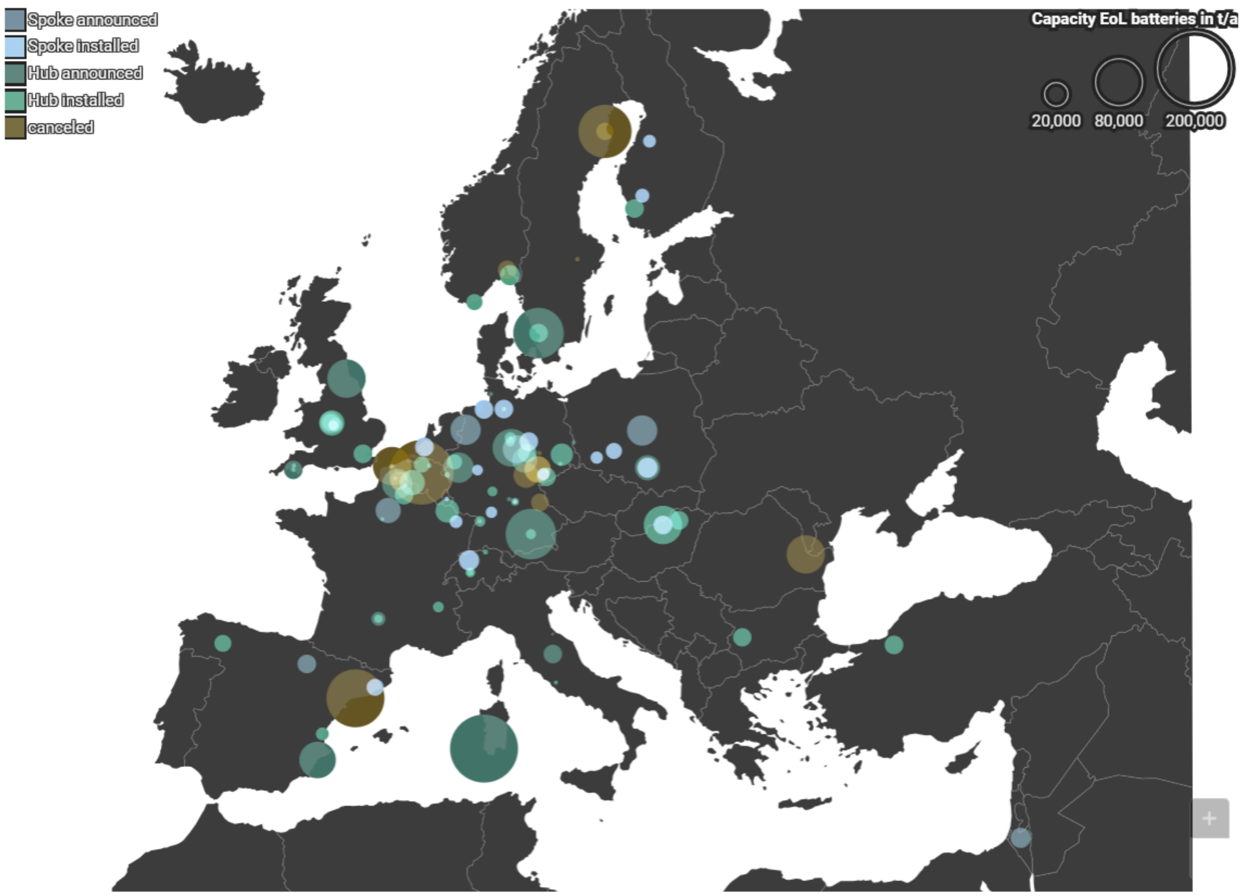

Fig. 4 Geographic distribution of existing and planned lithium-ion battery recycling facilities across Europe [24]

United States: Emerging Market

The U.S. LIB recycling market remains in a growing, development stage. It is mainly supported by large investments under the Inflation Reduction Act (IRA) and Bipartisan Infrastructure Law. Although portions of the IRA have faced delays, it remains an important factor that influences the financial market of LIB recycling in the United States. The IRA's clean manufacturing tax credits and domestic content incentives have reduced effective capital costs for companies such as Redwood Materials and Ascend Elements. This encourages large scale facility development. With ongoing policy uncertainty and the discontinuance of IRA payments, there will be growing uncertainty in major LIB investments for both new and existing facilities. Together these two allocate billions of dollars toward domestic battery materials recovery [25]. Companies such as Redwood Materials, Li-Cycle, and Ascend Elements are scaling hydro and direct recycling operations aimed at establishing domestic supply chains for critical materials (Fig. 5) [26]. The U.S. currently faces challenges in battery collection. It also lacks an Extended Producer Responsibility (EPR) system that would make producers responsible for the entire lifecycle of their products.

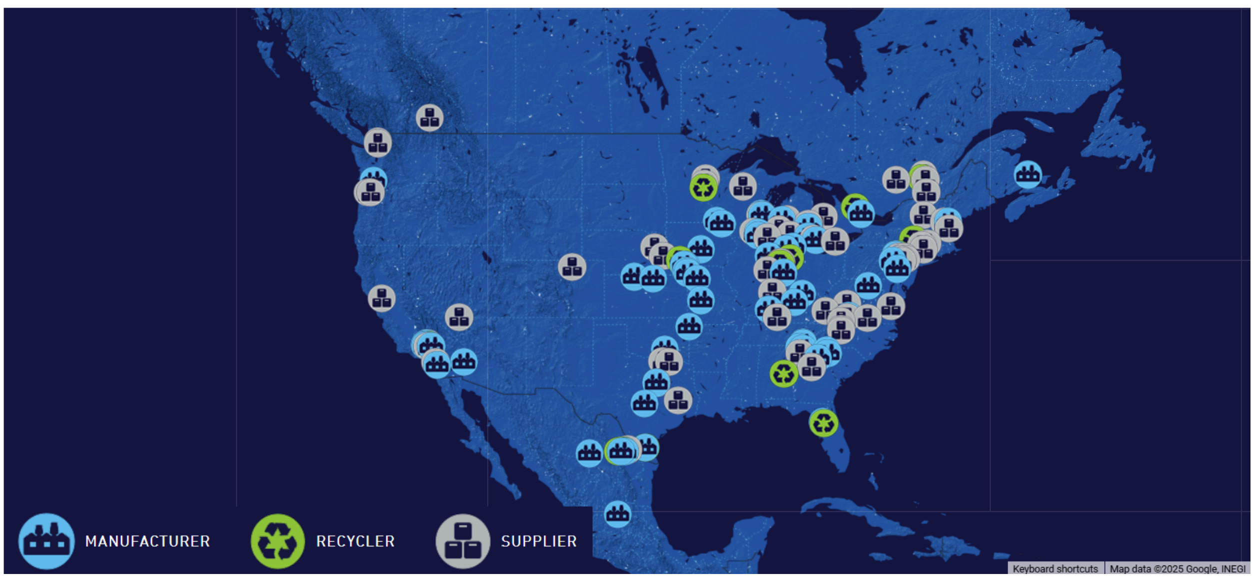

Fig. 5 Geographic distribution of end-of-life recycling facilities in the United States
The map highlights the rapid growth of diverse recycling-facility types within just one year. [27]

Comparative Summary

China currently leads in scale and cost efficiency due to its centralized policy and mature infrastructure. Europe leads in its regulations and sustainability frameworks. The United States currently shows strong growth potential driven by new incentives and private investment. These regional differences shape the economics of hydro, pyro, and direct recycling processes, as explored in the following section.

Recycling Process Economics and Comparison

Financial Viability Model

To evaluate the viability of LIB recycling we created a process based economic model. The model estimates the capex and opex for each process and region by using data from various research papers and industrial reports. It incorporates region specific variables such as labor costs, electricity costs, EOL battery prices, etc. to create accurate economic differences among the U.S., China, and Europe.

Opex include reagents, energy, utilities, maintenance, transportation, feed stock, and labor. The capex consists of research, equipment, and regulation fees which will scale based on the size of the facility's operating capacity. All the costs are standardized to USD to easily compare them. Material recovery rates for lithium, nickel, cobalt, manganese, graphite, and iron are estimated based on multiple research papers for LFP and NMC chemistries [28, 29, 30]. The amount of material that is recovered is then priced based on the current market prices to determine the total value of all the materials.

The model's outputs include estimated capex and opex for each recycling process, broken down by cost category and region, as well as the overall cost per ton of processed battery material. The initial reference values for the capex were for various industrial and research papers [31, 32, 33]. The method used to estimate the capex at higher TPAs was the cost-capacity scaling relationship, also known as the power-law or exponential scaling model [34, 35]. The values acquired from this relationship were then set up into a lookup table that would change the capex depending on how large the facility TPA input was. The model also calculates the gross profit of the facility. This allows us to see if the facility is able to make money to eventually pay back the capex for the facility. By comparing gross profits across regions and process routes, the model highlights which technologies and locations are more financially sustainable under current market conditions. These results provide the foundation for the analysis presented in the following subsections.

		LFP Direct			NMC Direct			NMC Pyro			LFP Hydro		
		CN	US	EU	CN	US	EU	CN	US	EU	CN	US	EU
Capex	$K	2,005	1,525	1,525	2,005	1,525	1,525	406,000	152,000	78,500	8,355,000	2,582,000	2,425,000
Opex	$/ton	1,110	301	383	2,910	2,301	383	3,190	2,604	965	1,652	881	1,166
Revenue	$/ton	1,891	1,891	1,891	4,045	4,045	4,045	4,108	4,108	4,108	1,955	1,955	1,955
Gross Profit	$/ton	782	1,590	1,509	1,135	1,744	3,662	917	1,503	3,143	303	1,074	788
Break-Even	TPA	600	300	100	400	200	100	88,700	20,000	5,000	1,020,000	292,000	5,400

Table 1 Comparison of Capex, Opex, revenue, profit, and break-even throughput for three major LIB recycling pathways

The results highlight substantial regional variation in cost structure and economic performance. China generally displays lower Capex and Opex for both direct and hydrometallurgical routes, while the EU shows comparatively higher profitability in NMC direct recycling. Pyrometallurgical facilities exhibit the highest break-even TPA requirements.

Hydrometallurgy

Hydro recycling generally involves higher capex due to the need for corrosion resistant reactors, solvent extraction units, and effluent treatment systems. The process offers strong scalability once operational. The most common LIB battery chemistry used in this process are LFP batteries. At their respective break-even TPAs, hydro recycling requires capex of roughly $8.36 million in China, $2.58 million in the United States, and $2.43 million in Europe (Table 1). Chinese facilities benefit from lower equipment and construction costs and established supply chains for reagents such as sulfuric acid. However, U.S. and EU recyclers often benefit from not having to pay as much to acquire batteries [36].

Hydro uses on average around 4166 kWh/ton [37, 38, 39]. This total consists of around 18% electricity and 82% natural gas. Based on these calculations the energy costs in Europe are the highest at around $482/ton [40, 41]. This is almost double compared to China and the U.S. where it costs $207-229/ton [42, 43]. This is a part of the opex where Europe is the weakest (Fig. 6).

Labor costs also vary greatly from region to region. An assumption of 25 full time employees (FTE) was made for both the U.S. and Europe facilities. Combining all the working hours of the workers we scaled it down to hours per ton and calculated the labor cost per ton from the region's wages. This leads to the Chinese labor cost being $160/ton [44]. This is advantageous for China since labor in the U.S. costs about $254/ton while in Europe it is around $283/ton (Fig. 6).

Transportation of LIBs is also an important part of the opex. The model assumes the distance LIBs have to travel to get to a recycling facility to be around 140 miles or 225 km for all three regions [45]. The cost per ton-km for each region was used to determine the total transportation cost per ton [46, 47, 48]. The calculated costs were all very close to each other ranging from $13-18/ton with China being to lowest and Europe being the highest (Fig. 6).

Europe and the U.S. don't have to pay to acquire LFP batteries while China has to pay around $900/ton [49, 50]. This is the most significant advantage these regions have over China. This causes China to have a higher opex costs compared to the U.S. and Europe. The consumables cost the last thing to consider but are very close in the regions ranging from $310-350/ton [44]. This all leads to the overall opex cost in China to be $1,652/ton, $881/ton in the U.S., and $1,166/ton in Europe (Table 1).

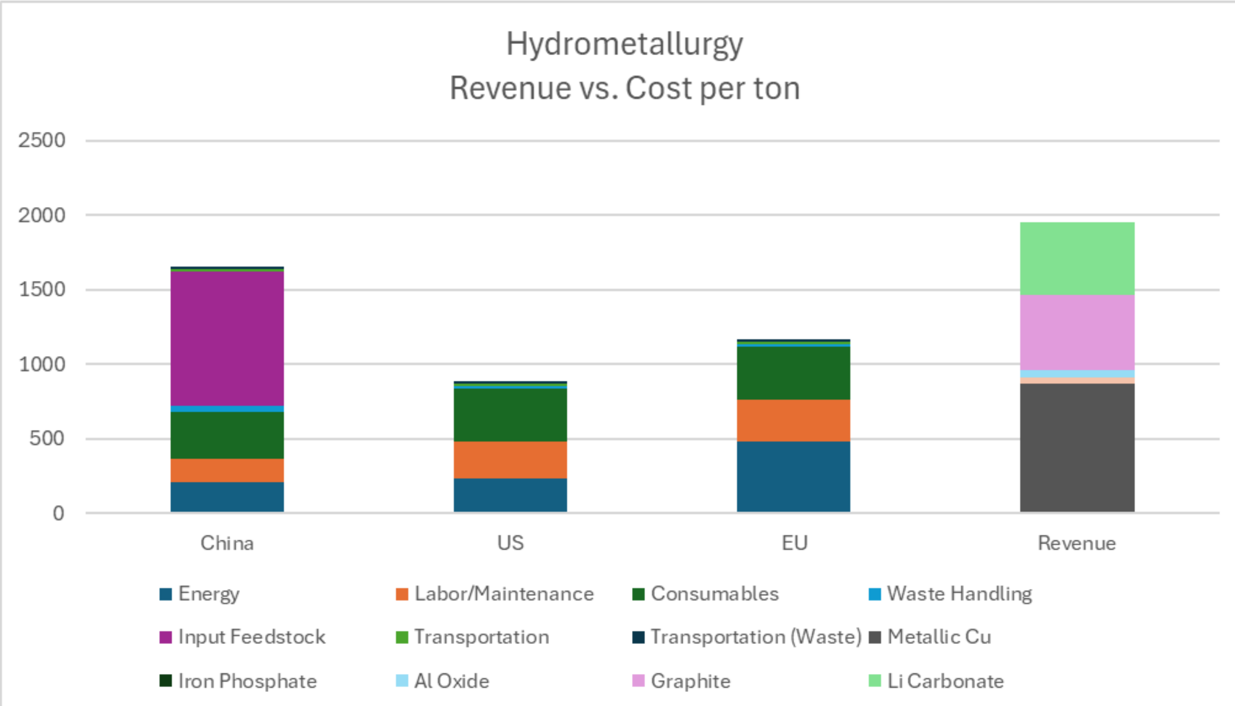

Fig. 6 Breakdown of hydrometallurgical recycling costs and revenues per ton for China, the United States, and Europe. Stacked bars show contributions from major cost categories while the revenue bar reflects the estimated market value of recovered products such as metallic copper and lithium carbonate. The comparison shows the regional differences in cost structure and the relative margin between total opex and revenue.

Pyrometallurgy

Pyro systems demand significant upfront capital for smelters and emission-control units. The capex at the break-even TPA ranges from $78.5 million to $152 million in Europe and the U.S., about 75% lower in China where it is $406 million (Table 1).

However, the opex for pyro is dominated by energy consumption, contributing up to 60% of total opex depending on the region. Pyro can consume on average 5500kWh/ton compared to hydro which consumes around 4100kWh/ton [37, 38]. Using a similar energy split like in hydro China's energy cost is $207/ton while in the U.S. it is $229/ton. However, Europe is much more expensive again at $482/ton (Fig. 7).

While the energy is more, labor for pyro ends up being lower than hydro. It generally requires less workers with only around 20 FTEs needed. Using the same calculations, the model estimated that labor costs $95/ton in China, $203/ton in the U.S., and $227/ton in Europe. Since pyro is commonly used on NMC batteries over other chemistries the model's feedstock price is for NMC batteries instead of LFP batteries. This causes a large change since in China NMC batteries cost up to $2,700/ton (Fig. 7) [51]. The U.S. also experiences high costs to acquire NMC batteries at around $2,000/ton (Fig. 7) [52]. Europe has an advantage in this case since it does not need to pay for these batteries [50]. This causes large differences in the total opex where it is $3,190/ton in China, $2,604/ton in the U.S. and only $965/ton in Europe (Table 1).

Pyro remains feasible primarily for NMC feedstocks rich in cobalt and nickel, while LFP yields negligible value after accounting for slag losses and lithium losses [53]. In the U.S., limited access to grid cheap energy constrains standalone pyro economics, leading to preference for hydro or direct processes.

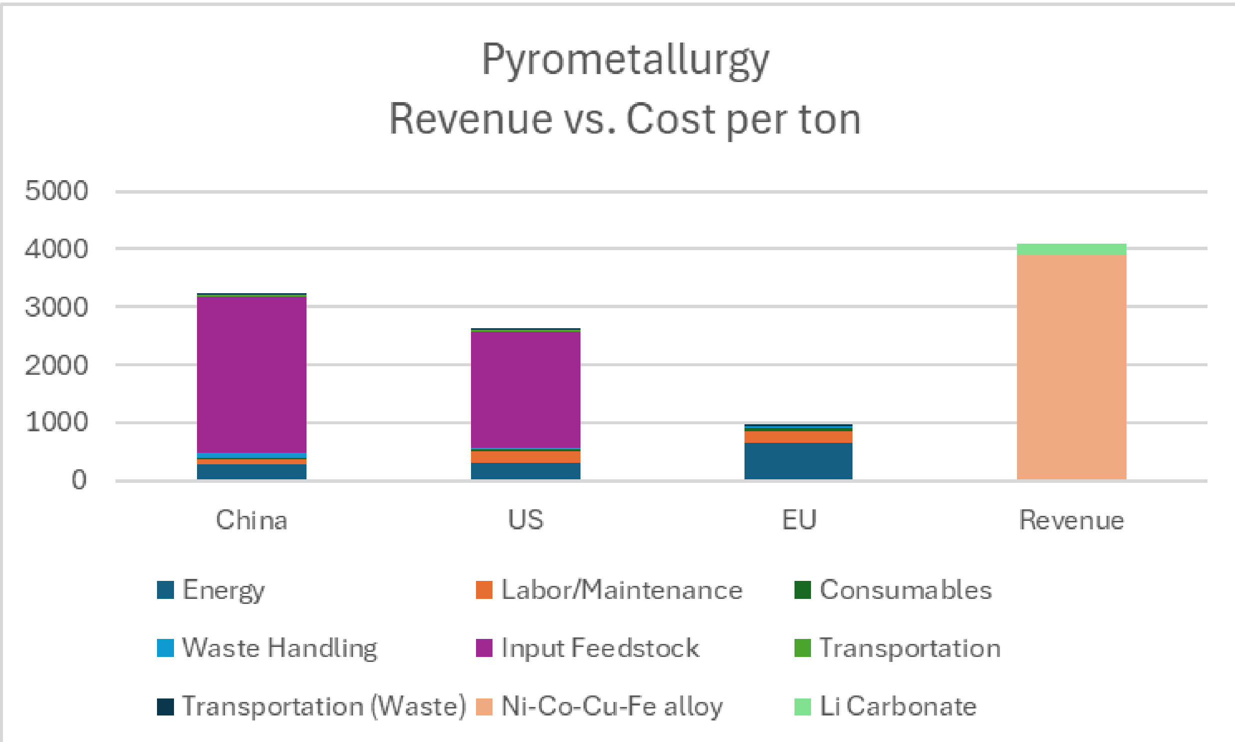

Fig. 7 Breakdown of pyrometallurgical recycling costs and revenues per ton for China, the United States, and Europe.
Stacked bars show contributions from major cost categories while the revenue bar reflects the estimated market value of recovered products like lithium carbonate and the metal alloy. The comparison shows the regional differences in cost structure and the relative margin between total opex and revenue.

Direct Recycling

Direct recycling has the lowest theoretical capex and opex because it avoids high temperature smelting and chemical leaching. The break-even TPAs are far lower than breakeven TPAs for hydro and pyro. The break even TPA capex for China is $2 million, $1.5 million in the U.S., and Europe also being around $1.5 million (Table 1). Pilot scale estimates for opex are between $1,000-3,000 per ton, depending on the region and the chemistry of the battery being recycled [51, 52].

Direct recycling requires much less energy only at around 1000 kWh/ton [54, 55, 56]. The model calculates that the cost in China is $50/ton. This is lower than hydro and pyro. This is the same for both the U.S. and Europe where the costs are $55/ton and $116/ton respectively (Fig. 8).

Direct recycling also requires much less labor to recycle the batteries. It only requires around 15 FTEs. This clearly leads to lower labor for all the regions. China is calculated to have a cost of $75/ton. The U.S. has a cost of $152/ton while Europe ends up with a cost of $170/ton (Fig. 8).

The U.S. currently leads pilot investments through DOE's ReCell Center and private firms such as Ascend Elements, targeting both LFP and NMC materials. These facilities use localized feedstock and renewable energy, offering long term cost advantages once large scale production is achieved.

In China, direct recycling research and development is expanding, but deployment remains limited to demonstration plants. Cost advantages are offset by feedstock sorting and quality control costs. Europe funds similar pilots but faces higher opex due to labor and compliance costs. For LFP, direct recycling offers the most viable economic pathway for all regions, since it preserves cathode value without breaking it down chemically. For NMC, while hydro still dominates in recovery rate, direct recycling could become cost competitive if material rejuvenation maintains more than 90% of original electrochemical performance.

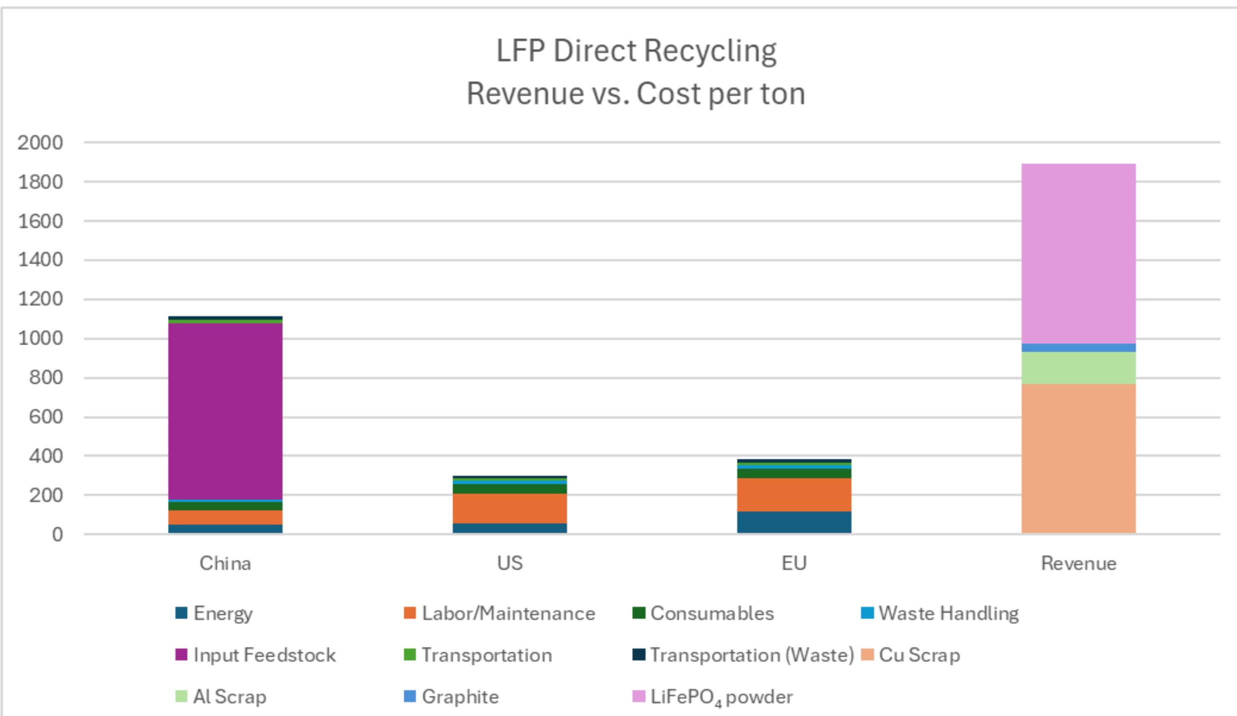

Fig. 8 Breakdown of direct recycling costs and revenues per ton for China, the United States, and Europe.
Stacked bars show contributions from major cost categories while the revenue bar reflects the estimated market value of recovered products. The comparison shows the regional differences in cost structure and the relative margin between total opex and revenue.

Process Comparison

China maintains the lowest capex and opex across methods due to its lower labor and energy costs. Europe's high compliance costs are partially offset by regulatory incentives and the U.S. has strong capital funding but higher logistics expenses. Process wise, hydro remains the most mature and profitable for NMC chemistries. Pyro serves as an industrial baseline. Direct recycling shows the highest long term potential, especially for LFP chemistries.

		Direct Recycling		Pyro		Hydro	
		Our Model	Schlott Data	Our Model	Schlott Data	Our Model	Schlott Data
Opex	$/ton	1,110	5,260	3,190	4,300	1,652	4,140
Revenue	$/ton	1,891	9,610	4,108	4,690	1,955	8,180
Profit	$/ton	782	4,350	917	390	303	4,040

Table 2 Comparison of estimated U.S. LIB recycling costs and performance from values reported by Schlott et al. (2025).
The table summarizes differences in Opex, revenue, and profits for hydrometallurgical, pyrometallurgical, and direct recycling processes. Showing similarities and differences between modeled results and literature values. [57]

To evaluate the accuracy and representativeness of the economic model, the estimated U.S. cost and revenue values were compared with those reported by Schlott et al. (2025) [57] for hydrometallurgical, pyrometallurgical, and direct-recycling processes (Table 2). While the overall trends are consistent like hydro showing moderate costs and revenues, pyrometallurgy showing high energy driven opex, and direct recycling showing lower cost but limited revenue there are several notable differences in magnitude between the two datasets.

First, Schlott, et al, reports substantially higher opex values, particularly for direct recycling and hydrometallurgy, compared to the model results. This is primarily due to differences in some underlying assumptions. Schlott et al. also assumes higher reagent and waste-management factors, reflecting precautionary industrial bounds. In their supplementary tables S6 to S23 [57], many of the cost studies operate at laboratory or pilot scale. Schlott et al. classify these scales as non-industrial where operational expenditures dominate due to high energy, reagent, and labor intensity. Schlott's review shows that hydrometallurgical models often omit fixed-cost dilution and instead emphasize direct operational inputs like acids, bases, leaching agents, water, and thermal energy which increases opex relative to full-scale industrial assumptions. Their framework also incorporates all 11 direct opex cost types and several indirect categories that many primary studies leave out, which further raises the $/kg estimates in their dataset. This method is not fully appropriate for comparing against a single region model because the added cost categories do not reflect actual expenditures in many of the underlying studies. Schlott et al. classify R&D, marketing, administrative, and insurance costs as "general expenses," yet these are not tied to process level operating activity and should not be included when assessing process opex. These costs belong in facility overhead or accounting, not in process opex, because they do not scale with kilograms of black mass processed. They are normally handled as corporate overhead in industrial cost accounting rather than as a part of the opex. Supervision and construction expenditures are also categorized as indirect capital costs in their framework and function as capex rather than recurring opex. Under standard techno-economic analysis practice, construction and facility development are treated as capex and amortized over the plant lifetime rather than charged as per-kg costs. When these non-operational categories are added to the cost structure, they artificially elevate Schlott et al.'s reported opex relative to our model. In the supplementary tables, these categories represent a large share of the total reported opex because they are allocated on a per-kilogram basis even in laboratory and pilot-scale studies. For various entries in their tables S10-S12, non-process expenses account for 6-28% of the reported opex. When things like supervision, construction, administrative overhead, and insurance are distributed across annual throughputs that are smaller than commercial facilities, their per-kg contribution of the opex becomes disproportionately large. At pilot-scale throughputs, these non-operational categories can contribute between 35-50% of the total reported opex, which is far higher than would be observed in commercial facilities. This scaling effect increases reported opex by two to three times relative to models that restrict opex to process dependent costs such as reagents, labor, and utilities. So Schlott's higher opex comes from including several categories that do not scale with the amount of blackmass processed and therefore dominate the cost.

Second, Schlott, et al, reported revenues are significantly higher across all processes. This difference comes from assumed battery prices during a period of higher cobalt and nickel value. They also use NMC-rich feedstock in their base-case and assume 100% collection and refined-material sales assumptions, which are more optimistic than many current U.S. operational conditions. In our model, we assume between 70% to 90% collection efficiency depending on the material which reflects a more realistic performance level for current U.S. operations. Nickel and cobalt prices have also fallen by 10 to 17% over the past year, the value of NMC-rich black mass today is substantially lower than during the period used by Schlott et al. Their model was developed using commodity prices from a time (2018-2021) when Ni and Co markets were significantly higher, which naturally leads to higher projected revenue per ton. These lower percentages and material values lead to lower revenue estimates under today's market conditions. Schlott et al.'s revenue values reflect normalization where most papers report revenues for cobalt and nickel rich chemistries, which dominate the dataset and yield higher product values. Their summarized revenue tables also rely on studies that assume near-complete recovery of high value metal salts under idealized laboratory conditions rather than commercial-scale yields, reinforcing higher revenue figures per kilogram. An example of this is the hydrometallurgical process data from Yang et al. (2021) [58] in their supplementary table S12, which reports 99% Co and 98% Ni recovery under controlled laboratory leaching and solvent-extraction conditions. Recovery yields alone cannot explain the large revenue gap. Most of the difference comes from system boundaries. Schlott et al. value fully refined sulfate products at optimistic market prices and assume all recovered metals are sold at those grades. Our model instead reflects current commercial practices, where recyclers primarily sell black mass or partially refined intermediates at significantly lower realized prices. An important distinction is that Schlott et al. assign revenues based on fully refined sulfate products, which carry much higher market values because they include the added value of downstream refining. Crediting recyclers with these refined product prices inflates revenue estimates. In contrast, most U.S. recyclers sell black mass or partially refined intermediates at much lower realized prices. These choices, also used by EverBatt, explain why Schlott's revenues are higher.

Overall, these differences between the two datasets do not indicate disagreement but rather reflect different modeling goals. Schlott et al. emphasize upper bound economic potential, whereas our model provides a more grounded and conservative estimate of U.S. economic performance under today's economic environment. Their assumptions also align more closely with conditions in China and Europe than in the U.S., since most studies in their analysis originate from these regions and reflect higher collection efficiencies, more centralized logistics, and feedstock typical of their recycling systems. In China, many referenced studies [59, 60, 61] assume lower labor costs and mature hydrometallurgical infrastructure, making Schlott's opex and revenue structures more representative of Chinese-scale operations. For Europe, the underlying literature often models coordinated collection networks and pilot-scale hydrometallurgical pathways making their cost more applicable to European contexts than to current U.S. conditions. The Schlott et al. analysis is an upper-bound dataset that is meant to be comparable internationally. It reflects the recycling conditions in China and Europe more accurately than it does the U.S. conditions. This is supported by their supplementary information, where the majority of hydrometallurgical and pyrometallurgical cost entries come from Chinese studies (Tables S6-S15) and European studies (Tables S16-S23) [57]. These tables also show that the referenced studies assume higher collection efficiencies and NMC-rich feedstock compositions characteristic of EU and CN systems. This explains why Schlott's values align more closely with these regions. Several of the key references included in Schlott et al., such as Xu et al. (2020), Harper et al. (2019), and Zeng et al. (2014), provide cost and recovery data specifically for China and Europe [59, 60, 61]. Our results for China and Europe still differ from Schlott et al. because their cost estimates represent an upper-bound abstraction that smooths across dozens of studies, many of which report pilot-scale or laboratory-scale performance rather than industrial data. Our regional models use cost and yield inputs drawn from recent industrial and pilot-scale data rather than laboratory-scale values [62, 63, 64, 65, 66, 67]. For example, commercial hydrometallurgical facilities in China and the U.S. report cobalt recoveries in the 80-92% range and nickel recoveries of 85-90% [68], which are consistent with pilot-scale observations in Tables S6-S8 of Schlott et al. rather than the higher yields of 98+% reported in several laboratory studies included in their dataset. We also incorporate updated material prices from 2024-2025 market data, region-specific labor rates, and realistic transportation distances. This includes current material prices, realistic labor rates, transportation distances, and collection efficiencies. This divergence arises because of several studies in Schlott et al.'s dataset which report performance that is not representative of commercial plants. Because Schlott et al.

combines many studies with near-idealized recoveries and minimal reported reagent or energy use, the resulting cost and revenue estimates exceed real commercial performance by a large margin. While most studies in the Schlott et al. dataset report substantially higher revenue values than our model, a subset of the pilot-scale hydrometallurgical studies [69, 70, 71, 72, 73], in Tables S9-S10, produce revenue values in the same range as our hydro estimate of approximately $1.3-2.8/kg compared to our $1.9/kg. These cases reflect more moderate yields and reagent intensities that are closer to industrial practice, and therefore show smaller deviations from our results.

Break-Even Analysis

The break-even point that our model uses for direct recycling is the closest hundreds TPA that is needed for the facility's gross profit to become close to 0. For the various LIB recycling processes it shows significant differences. For LFP direct recycling, the break-even threshold is relatively low, ranging from 100 TPA in the EU to 600 TPA in China and 300 TPA in the United States. This low threshold indicates that direct recycling of LFP cathodes can achieve economic viability at relatively small scales, largely due to lower opex. China's higher break-even capacity is primarily attributed to its higher feedstock price.

For NMC direct recycling, the break-even capacities are slightly higher. They range from 100 to 400 TPA. This shows stronger sensitivity to both feedstock pricing and labor costs. China and the U.S. exhibit lower economic thresholds compared to Europe, because established industrial infrastructure and lower labor rates in these regions help to offset the high feedstock input costs.

In contrast, pyro and hydro show substantially higher break-even points. These use the closest thousands TPA since their break-even points are much higher. Pyro routes exhibit extreme capex numbers, with break-even capacities exceeding 20,000 TPA in the U.S. and 88,700 TPA in China. These results highlight the strong dependency of pyroprocessing economics on scale and feedstock price, where profitability can only be realized through very large scale facilities. Similarly, the LFP hydro pathway demonstrates even greater economic challenges, with break-even points exceeding 1,000,000 TPA in China and 292,000 TPA in the U.S.,primarily driven by high capital costs and lower-value cathode material recovery. In the EU the break-even threshold remains high at approximately 226,000 TPA. However the average recycling facility in any of the regions are not close to these sizes (Fig. 9). Even if they were, recycling facilities rarely operate at their full capacity and so they would still be difficult to be profitable. That shows that these two processes can't be viable easily and will take decades to recover capex without government subsidies and will remain negative for a long time.

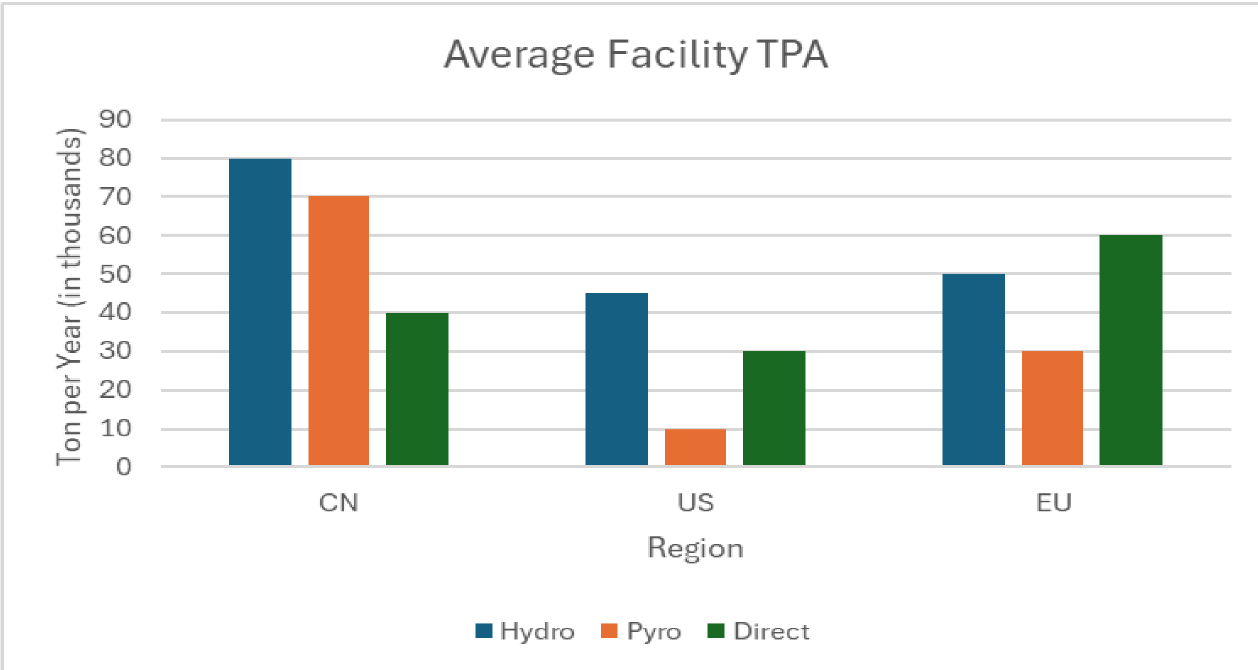

Fig. 9 Average TPA of lithium-ion battery recycling facilities.
The chart compares typical facility scale in China, the United States, and Europe across hydrometallurgical, pyrometallurgical, and direct recycling processes. China shows the largest average hydrometallurgical and pyrometallurgical plant capacities, while the EU exhibits the highest average capacity for direct recycling. [10, 74, 75, 76, 77]

Conclusion

This study shows that the financial viability of LIB recycling depends heavily on both regional economic conditions and the process being used. In China, strong government backing, centralized industry networks, and well developed hydro facilities. These helped keep recycling costs low and allowed the market to expand quickly. Europe benefits from strict environmental regulations and an established circular economy framework. Its higher labor and energy costs make recycling less profitable. The United States is still developing its recycling industry. Building more large scale facilities and introducing supportive policies will be important to becoming more competitive to other regions. Across all regions, direct recycling consistently achieves the lowest breakeven throughput compared to hydrometallurgy and pyrometallurgy. This would indicate that direct recycling offers the most accessible entry point for new commercial facilities, especially for LFP feedstocks.

The breakeven analysis also shows clear regional differences. China reaches breakeven at the smallest plant sizes across all processes due to lower labor, more developed collection networks, and lower energy costs. Europe has the highest breakeven thresholds, driven primarily by labor and compliance expenses, while the U.S. is in between the two, with the main constraint being feedstock cost and collection efficiency rather than process performance.

Hydro is the most commonly used process because it achieves high metal recovery rates and can handle different battery chemistries effectively. Pyro tends to be more energy intensive and recovers less lithium, which makes it less suitable for chemistries like LFP. Direct recycling offers strong potential for cutting costs and improving environmental outcomes, but it still faces challenges with scaling up and maintaining consistent product quality. Some ways direct recycling can overcome these challenges would be investing in automation and the standardization of battery designs. Each battery pack has different formats, chemistries, and cell structures, which makes feedstock preparation inconsistent and more labor intensive. Making these packs more consistent would allow for these early operations to be automated and become faster.

These findings suggest that the future economic viability of LIB recycling will rely not only on improving technologies but also on stable material markets, clear and consistent policies, and standardized feedstock. Consistent policy frameworks would be helpful to get more investments and give more market confidence. Uncertainty in regulatory design or short lived subsidy programs can deter large scale deployment, particularly in the U.S. market where policy coordination is more limited. Standardizing battery formats, labeling, and chemistry identification would significantly lower preprocessing and sorting costs in recycling plants.

Coordinated global efforts among governments, industry, and research institutions will be essential to transition LIB recycling to a profitable industry. More uniform recycling standards, sharing process innovations, and aligning trade and policy frameworks across regions would reduce market fragmentation and improve overall efficiency. This kind of cooperation would speed up the development of the technology and allow for better practices to be adopted sooner.

Further Work

This study provides a comparative analysis of what the current economic viability of LIB recycling looks like. There is still a lot of uncertainty in how these technologies and processes will change in the future. Future research shall focus on developing models that show the expected changes in costs, recovery efficiencies, and material values over the next 5 to 15 years. That would allow for a better understanding of how regional policy incentives, technological advancements, and market conditions change the profitability of recycling.

An area of further study involves using projected trends in electric vehicle adoption, raw material prices, and cathode composition shifts into cost models. What that would do is allow for more accurate predictions of feedstock composition and the recovery value for it. Incorporating sensitivity analyses for energy prices, labor costs, and recycling efficiencies would also be important. It would give more insight into how future scenarios may favor different recycling processes or regions.

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Conflicts of Interest

The author declares no conflict of interest.

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