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# Environmental Impact of Material Supply Chain Disruptions

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## Executive Summary

Recent geopolitical events, such as the Ukraine war and its subsequent repercussions, have opened a new phase of instability and uncertainty in the international relations landscape. This has sparked concerns about the European Union’s ability to ensure a stable and affordable supply of raw materials from global markets. Of particular concern are critical raw materials (CRMs), crucial for facilitating the transition to green energy, as highlighted in the 2023 Critical Raw Materials Act (CRMA).

In response to potential disruptions in the supply chain, the EU is considering various de-risking strategies and instruments outlined in the CRMA and other relevant policy measures. However, these strategies have predominantly focused on economic and geopolitical aspects, with less attention given to the environmental dimension.

This report aims to explore the environmental consequences arising from disruptions and interruptions in the international supply of raw materials. In doing so, it seeks to enrich the ongoing policy debate by providing valuable insights and knowledge pertaining to this particular aspect of the strategic shifts anticipated within the EU.

The report introduces an analytical framework designed to address the environmental implications of supply chain disruptions, showcasing the complex nature of potential impacts and corresponding responses. This elucidates how environmental implications are the result of market dynamics and diverse reactions from economic and policy stakeholders within the EU. The framework also underscores the critical role played by the timing of responses, ranging from short- to long-term, where the latter offers greater flexibility, such as the development of domestic supply.

The framework is subsequently applied to investigate two case studies on critical raw materials: nickel and rare earth elements, specifically neodymium and dysprosium. In these case studies, the report delves into three key potential responses from the EU perspective across varying timeframes, including:

- Short term scenario (2023-2025): Shift to an alternative supplier;
- Medium term scenario (2025 – 2030): Growth in domestic capacity for recycling;
- Long term scenario (2030 – 2040): Improved domestic capacity for primary extraction, processing and refining.

The study finds that, in a short-term scenario, a complete or partial shift of nickel and neodymium supply from China to Canada and the USA, respectively, can result in reduced environmental impacts—particularly in terms of global warming potential, energy consumption, and ecotoxicity to water—as viewed from the perspective of EU consumption.

In the medium-term scenarios, recycling emerges as a preferable option for reducing various environmental pressures. However, to ensure that the nickel recycling from batteries has lower energy consumption, there is a need to implement renewable solutions.

Finally, in the long term, the growth of domestic mining and refining in Europe could also lead to reduced environmental impacts. Currently, Finland is one of the primary suppliers of refined nickel class I to the EU and has plans to increase its refining capacity by 50 %. If the refining operations in Finland were to achieve carbon neutrality in the future, as claimed by the respective company, this could lead to a reduction in carbon emissions within the scope of EU production and consumption. The same environmental benefits could be achieved by domestic production of REE. If the newly discovered REE deposits in Sweden could satisfy the entire EU demand, it is estimated that many environmental impacts of the current REE supply chains could be reduced by 70-80 %.

While the study primarily focuses on the environmental impacts caused by the EU responses, it recognizes that any benefits accruing to the EU do not necessarily translate into reduced global impacts. This outcome depends significantly on various other factors. For instance, in light of the increasing demand for nickel in battery production, a shift from China to Canada for Europe, does not guarantee a reduction in China's supply due to decreased European demand. Thus, especially for raw materials with anticipated future demand growth, the environmental effects of an EU supply shift may not align on a global scale. Furthermore, if a reduction in global supply of these materials were to hinder the progress of the clean energy transition, it could delay the phase-out of fossil fuel systems, resulting in far-reaching consequences for global climate change.

The current study, constrained by data limitations, resource availability and stringent assumptions, provides an initial assessment of the expected impacts. Recognizing the need for a more comprehensive evaluation, future developments could encompass other policy options and scenarios, particularly those involving demand reduction and substitution with alternative materials. To enhance the depth of the analysis, upcoming work would involve extensive life cycle assessment studies of the entire value chain and its dynamic changes. The insights derived from such assessments hold the potential to play a significant role in shaping well-informed policy-making processes.

# 1 Introduction

This pilot study aims at providing an analytical framework to assess the environmental impacts of supply chain disruptions of raw materials and showcase its application in two real-world case studies: supply chain disruptions for (i) nickel and (ii) rare earth elements (REEs). The current policy debate around enhancing supply chain resilience for the EU has led to the implication of the concept of de-risking in the context of European reliance on other countries (EPRS, 2023). Several means and instruments are proposed as de-risking strategies which are predominantly grounded in geopolitical and economic considerations, with relatively limited attention to the potential environmental ramifications of these chosen responses. The particular objective of this study is to assess the environmental impacts of such policy driven responses to supply chain disruptions and provide evidence-based insights that could be considered when deliberating potential responses by policy makers. To achieve this goal, the study has developed a theoretical framework followed by a practical framework and has selected two case studies based on a set of relevance and feasibility criteria, including data and information availability, as explained in Section 2 of this report. The case studies (Sections 3 and 4) will provide insights into these materials supply chains and will support in assessing the applicability of the methodological framework, and further requirements for the framework to provide useful policy insights.

## 1.1 Background and Context

Over the last few years, concerns over European dependency on third party countries for critical resources and security of their supply has progressively become a defining issue of Europe's economic policy agenda, driven by a series of shocks: the COVID-pandemic, the war in Ukraine, the resulting energy crisis, and the overall increase in international tensions linked to the geopolitical decoupling between the United States, the EU, and other western countries with China. At the core of the issue stands the risk of supply chain disruption for critical products and commodities, such as vaccines during a pandemic, natural gas during a major energy crisis, or critical and strategic raw materials and clean technologies during the green transition.

When it comes to the risks associated with the green transition, the European Commission, after proposing in February 2023 a Green Deal Industrial Plan, unveiled in March 2023 two legislative proposals aimed at becoming the founding stones of a new EU policy framework in this space: the Critical Raw Materials Act (CRMA) and the Net-Zero Industry Act (NZIA).

The CRMA seeks to respond to the supply chain disruption risk in critical raw materials[1] mainly by boosting their domestic production, refining, and recycling. The Act identifies a list of 'strategic[2]' and 'critical' raw materials, which are crucial for the manufacturing of green, digital and defence technologies, and then sets benchmarks for domestic capacities along the raw materials supply chain and to diversify EU supply by 2030:

i. At least 10 % of the EU's annual consumption for extraction
ii. At least 40 % of the EU's annual consumption for processing
iii. At least 15 % of the EU's annual consumption for recycling
iv. Not more than 65 % of the Union's annual consumption for each strategic raw material at any relevant stage of processing from a single third country.

To achieve these goals, the CRMA fundamentally aims to make the issuing of permits to relevant industrial projects subject to a common EU time limit, with the aim of accelerating them. The proposed act also includes provisions on supply chain monitoring, stockpiling, and improving the recyclability of CRMs.

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1 CRMs are those that have high economic importance for the EU while associated with high supply risks.

2 According to CRMA, "strategic raw materials are crucial for key technologies important to Europe's green and digital ambitions, as well as for defence and space applications, while being subject to potential supply risks in the future".

Geological surveys and investigations of CRM contents in waste should identify domestic resources for mining and recycling of CRMs.

The CRMA acknowledges that while important, domestic actions will never make the EU self-sufficient in this space. As such, the Act also puts forward an international strategy aimed at diversifying the EU's imports of strategic/critical raw materials. On the one hand, the CRMA seeks to strengthen the EU's global partnerships with emerging markets and developing economies, notably in the framework of its Global Gateway strategy (European Commission, 2021). On the other hand, the CRMA seeks to "step-up EU's trade actions, including by establishing a Critical Raw Materials Club for all like-minded countries willing to strengthen global supply chains, strengthening the World Trade Organization (WTO), expanding its network of Sustainable Investment Facilitation Agreements and Free Trade Agreements, and pushing harder on enforcement to combat unfair trade practices".

The NZIA aims to tackle the supply disruption risk in clean technologies by undertaking the following actions. First, it lists the net-zero technologies that are strategic: solar photovoltaic and solar thermal technologies; onshore wind and offshore renewable technologies; battery/storage technologies; heat pumps and geothermal energy technologies; electrolysers and fuel cells; sustainable biogas/biomethane technologies; carbon capture and storage (CCS) technologies; grid technologies. Second, it adopts an overall headline target of reaching a manufacturing capacity for these technologies of at least 40% of the EU's annual deployment needs by 2030. In parallel, it also adopts a target for an annual injection capacity in $CO_2$ storage of 50 Mt $CO_2$ by 2030, so to spur the development of CCS. Third, to achieve these targets it puts forward a governance system based on the identification by member states of Net-Zero Strategic Projects (NZSPs) that will basically have to be:

i. Granted priority status at national level;
ii. Fast-tracked in permitting procedures, also to be given pre-set time limits by the EU;
iii. Fast-tracked in other administrative procedures, also through the identification of a one-stop-shop national authority in charge of these projects;
iv. Evaluated in public procurement procedures and auctions to deploy renewable energy sources also in view of a "sustainability and resilience" criteria, to be given 15-30 % award criteria weighting.

In the same vein, bids using equipment for which the country of origin provides at least 65 % of EU supply are disadvantaged. When it comes to the important issue of financing, the NZIA does not include any new EU-level initiative, and it also does not tackle the issue of simplification and streamlining of current EU-level tools. State aid is thus set to remain for now the main avenue of public support to clean tech manufacturing in Europe, unless the EU advances in the coming months the EU Sovereignty Fund idea.

The CRMA and the NZIA can be directly linked because the 'strategic' and 'critical' materials discussed under the CRMA are crucial for advancement of some of the major technology sectors covered by the NZIA.

## 1.2 Objectives and Scope

The recent crisis in the European energy markets caused by the war in Ukraine well illustrates, ex post, how large scale and systemic can be the effects, and the responses, to the disruption of the international supply of a primary energy commodity. Similarly, the interruption of the international supply of an industrial raw material, or significant price shocks in its market, can trigger a complex set of environmental, economic and social effects together with a range of industrial and policy responses aimed at limiting or eliminating those adverse effects.

The analytical framework and case studies presented in this work aim at providing a methodological approach to understand and analyse the potential environmental impacts of supply chain disruptions of two specific raw materials: **nickel and rare earth elements**. For the selection of these two materials, the following criteria were considered:

- Relevance to China and Russia dependency / Vulnerability of disruption
- Strategic importance for the EU
- Environmental impact data availability
- Trade data availability
- Recycling potential (time required to establish large scale recycling)

One important usage of REE is in the field of magnets. Permanent magnets constitute not only the majority of REE usage but also are collectively responsible for over 90 % of the total value of global REE consumption in 2022 (REIA, 2023). Arguably the two most essential REEs used in production of powerful REE permanent magnets are neodymium (a light REE) and dysprosium (a heavy REE). As such, for the purpose of this study, neodymium and dysprosium will be the primary focus of the REE case study considered.

In the analysis below, Europe is represented as the 'buyer country', while 'supplier countries' are identified based on resources availability and current position in international markets for those specific materials. Furthermore, this pilot study is conducted based on a series of assumptions and with consideration of specific limitations, which are explained in sub-sections 2.5 and 2.6 below.

# 2 A Framework and methodology for mapping the expected effects

## 2.1 Framing environmental effects within market dynamics: A theoretical framework

The environmental pressures of industrial materials are associated with their production (including mining, processing and refining), trade, consumption and waste management (such as disposal and recycling). Supply disruption induces complex changes of these processes and then of the associated environmental pressures at the international scale. Therefore, the analysis of expected changes in environmental pressure due to supply disruption must be framed into the ex-ante changes it induces in production, trade, consumption and recycling as well as in material saving innovations and material substitution processes.

For a specific industrial material or metal for which an international market with many supplying countries exists, a supply disruption can induce changes for: (i) buying country and supplier countries (both old and new suppliers); (ii) transit countries (pure trade transit or phases of the value chain, e.g. semi-finished inputs). The disruption can be quantity-related and/or price-related, that is: (i) flow interruption that can dynamically trigger price increases; (ii) shock in prices that can dynamically trigger quantity disruption (e.g., unsustainable raw materials costs). The reason for looking at quantity and price together is that, in international commodity/energy markets, a temporary or stable supply/demand shock translates into price changes, and price changes translate into changes in supply and demand as a standard dynamic sequence of search for new equilibria.

The associated change in environmental pressures must be informed by a differential approach, that is the effects in new supplying country, or self-supply with increased self-sufficiency of the buyer country, with respect to the old supplying country).

In this framework, the actual overall and environmental effect of a supply disruption can critically depend on the type and the pace of reaction by the countries involved, which can range from the short-term (with limited reaction possibilities) to the long term (with large reaction possibilities), with the medium term in between.

The **short term** is typically dominated by a limited elasticity of supply in both the world market of the material and domestically (primary and secondary supply) in the buying country, coupled with a limited elasticity of domestic demand (e.g., by substitution), and a limited possibility of deploying resource-saving innovations in the same country. Therefore:

- The immediate expected effects of a *quantity disruption* (flow interruption) without a reaction from a buying country are losses of value added and employment in both the buying and the supplying country. The reduction of production of the supplying country can generate a reduction of environmental pressures in both countries, provided all other variables and conditions are fixed and there is no domino effect in other sectors.
- If there is an increase in prices, the effects depend on the elasticity of demand to prices: if low, there can be increases in final good prices (up to inflation, if the material is economically important) in the buying country. In addition, it is relevant how the other suppliers in the world market can react to price increases by increasing supply (depending on supply elasticity to prices). The effects of the price increase can be, in any case, depressive on value added and employment in the buying country, as well as in the formerly selling country, which however can benefit from higher prices on the still active export flows. As far as there is a reduction of activity (quantities), there are lower environmental pressures. Even though the elasticity of supply can be low in the short term, a possible new supplier could benefit from entering the market attracted by higher prices.
- The effects on value added and employment depend also on how the increasing costs of inputs are transmitted to final prices (pass-through) and how final demand can react (decrease). If the demand moves towards substitutes, the latter can benefit from an increase in production. Consequently, also environmental effects depend on the elasticity of demand to prices (negative): if it is high, price increases can reduce demand (industrial activity) and environmental pressures, but the cross-effects on substitutes must be considered. In this same case, the high prices might attract new suppliers in the world market, which can alleviate the price increase (but their elasticity of supply could be low in the short term).

The expected effects in the **medium term,** are dominated by a higher elasticity of supply in the world market (procurement shift is more possible) and possibly by domestic supply activation (primary production, if the country is endowed with the resource, or secondary through higher recycling and fuller circularity), together with a possible reduction of domestic demand via substitution and adoption of resource-saving innovations.

In the **long term**, there is a higher possibility to put in place strategic reactions towards a permanent supply disruption for the material, up to achieving high self-sufficiency and re-shoring of those production phases that are subject to procurement risk. There is the possibility to adopt new industrial strategies to reshape the whole domestic value chain in which the material is used. There are also more possibilities to reduce domestic demand by fully deploying resource-saving innovations and demand-side shifts. Therefore:

- The buying country can achieve a full geographical shift of procurement, thus stabilising the trade flows with new reliable suppliers; the new suppliers in the procurement portfolio of the buying country can gain value added and employment, but losses on the environmental pressure side;
- The buying country can better implement measures to exploit and increase domestic supply potential (primary production and, especially, secondary raw materials), thus making steps towards self-sufficiency for the material.
- Self-sufficiency strategies of the buying country can cause increasing pressures on its domestic environment. Measures to mitigate these domestic environmental effects, as well as to respond to the NIMBY syndrome of households, communities and administrations (e.g. compensations), can be adopted; the higher the ambitions for self-sufficiency, the higher the concern and stronger the measures must be to avoid domestic environment disruption, or failures of the strategy caused by oppositions; the net global balance of this transfer of environmental pressures to the buying country depends on whether these pressures are lower (country environmentally more efficient) or instead higher than those (now missing) of the former supplier country (given the quantities).

- The buying country can fully exploit the potential industrial and environmental benefits of secondary domestic supply within a circular economy paradigm, which can deliver a third dividend through increasing self-sufficiency and resource independence; for some materials, the increase in circularity can be the only real option in front of a permanent supply disruption; at the same time, the buying country can fully deploy resource savings innovations and new industrial policies, up to full adaptation of the domestic value chains in which the material is, directly or indirectly, relevant.
- A higher self-sufficiency can induce higher domestic prices along the value chain if domestic production costs (extraction, transformation. etc.) are higher than those of the imported material (at higher prices), and 'imported inflation' might be substituted for by domestically generated inflation; this cost can be seen as a cost of insurance against the riskiness of the international market, but measures must be taken to avoid a possible unfair distribution of this cost (as reflected in higher consumer prices).

## 2.2 General data requirements: a preliminary exploration

A full implementation of the framework outlined above can be challenging for the majority of internationally traded raw materials. While data for several dimensions turn out to be readily available and reliable (e.g., bilateral trade flows), some critical gap remains (e.g., demand and supply short- and long-term elasticities, capacity utilization). In particular, issues of data limitation and quality can arise for the following dimensions.

**Demand and supply elasticities**: One of the most critical information for envisaging the market dynamics in response to a supply disruption, which are key to the environmental impacts, are demand and supply elasticities to prices (see e.g., Rosendahl & Rubiano, 2019). Indeed, a highly elastic demand for materials implies a substantial adjustment in quantities, while the same price shock for a fully inelastic demand has no impact on quantities. This is particularly important when assessing the environmental consequences of the shock, as they depend on physical quantities. The problem becomes even more challenging when disruptions are due to geopolitical tensions and retaliations (e.g., sanctions banning specific bilateral trade flows). In this case, supply and demand elasticity should consider the cross-price elasticities of supply or demand of other trade partners[3] (Fally & Sayre, 2018).

**Geographical distribution of resources and reserves**: Data on material extraction sites is crucial to understand the geographical distribution of both resources and reserves. A useful source is the United States Geological Survey (USGS) database on 'Major mineral deposit of the world by common geographic areas', available on their website(USGS, 2023). Another relevant source about the environmental implications associated with mine sites is, e.g., the EU-funded Fineprint project[4]. This includes data sets containing georeferenced polygons of more than 34,000 mines on a world-wide scale (based on satellite images), illustrating the land use of the global mining sector. Within the project also spatially explicit modelling of global material flows and related environmental impacts was carried out. Furthermore, the raw materials dashboard available on the Raw Materials Information System provides a comprehensive overview of the global and European reserves, production, export and import of critical and strategic raw materials (Joint Research Centre, 2023b).

**Capacity utilization of extraction facilities**: The presence of idle capacity in mining and extraction in existing sites of supplying countries which are unaffected by shocks represents a crucial 'degree of

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3 At the same time, however, bans to import/export between two countries could be interpreted as the extreme cases (i.e. corner solution) of infinite price for the importer and zero price for the exporter. In these corner solutions, elasticities are not defined and equilibrium quantities for the bilateral trade flow are zero).

4 https://www.fineprint.global/

freedom' to substitute suppliers hit by shocks in the short-medium term. Indeed, this would allow rapid compensation of supply shortages in global market. Information on capacity utilization of extraction facilities would be of great importance to build scenarios of possible reactions. Unfortunately, detailed site-specific data is hard to collect, while some estimate at the aggregate level would be possible by considering deviations in year-specific extraction flows compared to historical averages.

**Environmental impact of extraction and processing:** Environmental impacts of material's supply chain disruptions crucially depend on the difference in the environmental impact of extraction/mining between the site hit by the shock and the potential substitute site. In the absence of site-specific information, country-level life cycle assessment/life cycle inventory (LCA/LCI) data (e.g., Ecoinvent) [5] could represent a second-best solution.

**Stock of material embodied in products**: To estimate the potential recycling response to the disruption the amount of material embodied in products together with estimates of the expected lifetime of these products (e.g., few years for a smartphone, 20-30 years for a solar PV panel) are needed. Data on life-times of products is often used in dynamic material flow analysis (D-MFA) models which could be a good starting point for locating such information (IEF, 2017). Data on product-specific embodiment of materials could be found in LCA/LCI databases (e.g., Ecoinvent), while expected lifetime of products needs to be assessed product-by-product. In this respect, knowledge about the future availability of secondary raw materials will be developed by a Horizon Europe research project entitled FutuRaM[6], started in June 2022.

**Data on bilateral trade flows**: Data on bilateral trade flows (in quantity and monetary value) is widely available by narrowly defined product groups from several different databases (e.g., UN Comtrade, OECD, Eurostat ComExt, JRC Raw Materials Information System, EU RM Scoreboard, etc.). The challenge is twofold. First, the identification of one or more product groups for the (raw) material represents a crucial prerequisite for an unbiased description of the phenomenon. Second, the measurement of materials embodied in semi-finished and final products requires product-specific information on the material content of products. The overlap between classifications used for international trade data and classifications used for LCA/LCI databases needs to be assessed.

**Outlook of domestic and global demand trends**: Detailed data on the expected trends of material-specific global demand allow us to identify future gaps between demand and supply caused by potential supply chain disruptions. While material-specific forecasts could be hard to find and highly uncertain, more aggregate mega-trends could help to build outlooks for future patterns of demand. In this respect, the Joint Research Centre of the European Commission has developed some outlook for the demand of materials in the EU regarding strategic technologies and sectors (Carrara et al., 2023).

**Trade barriers**: Bilateral barriers to trade could hinder rapid shifts across different supplying countries. These barriers could be geographical, regulatory (e.g., tariff and non-tariff barriers), cultural, etc. Detailed information on export restrictions for raw materials is collected, for example, by the OECD (OECD, 2023). Where detailed data are not available or a more general measure is needed, estimates of bilateral barriers could be made by considering the drivers of bilateral trade flows (e.g., by means of gravity models) and using deviations from predicted values as proxies of these barriers.[7]

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5 https://ecoinvent.org/

6 https://futuram.eu/

7 The gravity model of international trade consists in estimating the drivers of bilateral trade flows by means of econometric techniques. The reference to the law of gravity comes from the consideration that trade flows between two countries depend positively on the economic size of the trading partners and negatively from their distance (Bergstrand, 1985).

## 2.3 Methodological approach: A practical framework

Based on the framework explained above, this section introduces a practical methodological approach to analyze the multifaceted environmental impacts of supply chain disruptions of the two critical raw materials selected for this study namely nickel and rare earth elements.

The aim of the proposed methodological approach is to display the wide range of potential effects and responses to supply chain disruption to a specific industrial raw material. The proposed approach has a distinct focus on environmental implications following a supply chain, mainly because the size of this project does not allow to fully capture the interactions with variables of economic and social relevance. At the general level, these are the variables that could be considered for such an analytical framework:

**Type of materials**: a specific industrial material or metal for which an international market exists; the material market is not monopolistic, even though specific countries may show a dominant position (>60% of global capacity) across any of the supply chain stages (e.g., mining, processing, recycling). For a comprehensive analysis, the supply of the selected material in the form of coproducts or by-products should be considered.

**Countries involved**: (i) buying country and supplier countries (both old and new suppliers); (ii) transit countries (pure trade transit or phases of the value chain, e.g., semi-finished products)

**Type of disruption**: temporary or permanent disruption; total (i.e., no flow) or partial (i.e., reduced flow) disruption. The framework does not differentiate the causes behind the supply chain disruptions, which could be due to either technical, economic or geopolitical reasons. They could also be triggered by natural disasters or pandemics.

**Type of responses**: what can be the expected response from the buying country in different time frames; responses include the search for alternative suppliers and the increase in domestic capacity across relevant supply chain stages (e.g., mining, processing and recycling). Other potential responses could be a reduction in demand or substitution by other materials and technologies. The CRMA is specifically intended to address such reactions to supply chain disruption (See Section 1.1 for more details about CRMA).

**Type of environmental impacts**: this includes environmental impacts for the buying country and for the rest of the World. Based on previous environmental assessment studies on raw materials supply chains (DIEH, 2010; Joint Research Centre et al., 2013), seven categories of environmental impacts could be considered for such an assessment:

(i) Air pollution: This includes emissions of greenhouse gases (i.e., $CO_2$, $CH_4$, etc.) but also hazardous emissions (i.e., mercury, lead, nitrogen, sulfur, etc.). Emissions occure all along the value chain.

(ii) Water pollution: This mainly occurs when cyanide and sulfuric acids are used to separate targeted minerals from ores.

(iii) Biodiversity depletion: Mining processes can have a wide range of impacts on the fauna and the flora affecting their richness, their abundance, and diversity which leads to consequences on the soil and water quality. The level of impact on biodiversity could eventually lead to long-term damage to ecosystems.

(iv) Land use: Extraction of materials from the ground necessitates huge amounts of land, especially when done with open pits. Deforestation and general loss of above ground ecosystems lying there is to be accounted for in this category of impact.

(v) Soil pollution: Soil pollution generally occurs when leaching of the tailings/waste, when improper treatment of drainage water, and when high levels of dust.

(vi) Geological instability: Can create negative environmental impacts (e.g., habitat destruction, affect water balance, soil erosion, etc.) and can occur either in excavation sites, in mines, or with tailings storage.

(vii) Waste creation: Waste rock and tailings are the common waste created in mining, processing and refining. Some of the tailings can be toxic, creating pollution as can be seen with soil pollution. Furthermore, tailings dams failures can lead to sever environmental damages and loss of life.

An additional variable to be considered in the framework is the application or end-use of the materials analyzed. This is key for the analysis of second-order effects. If, for instance, the materials are utilized in clean energy technologies, a disruption in the supply of those materials may have profound effects on the transition to renewable energy technologies. This could slow down the phase-out of highly polluting energy sources and the adoption of more sustainable renewable technologies. This type of effect can only be considered if the end-use of materials is included in the framework.

**Geographical distribution of the environmental impacts**: This will depend on the combination of geographical redirection (which is dependent on raw material reserves and production know-how of the new supplying country), international procurement, activation of domestic supply, substitutability, and increase in circularity of the material (recycling, reuse, etc.). Other aspects such as the implication of co-products to mining are also highly interconnected with the geographical redirection of mining activities and distribution of environmental effects.

**Timing of responses and effects**: The precise identification of relevant timeframes, across short-, medium long-term, is dependent on the type of material analyses and its supply chain dynamics, and—to a certain extent—arbitrary. As explained in the previous sub-chapter, the buyers will have limited response possibilities in the short-term, moderate response possibilities in medium-term and larger response possibilities in long-term. The difference in global environmental effects in this case will depend on the technologies and regulatory framework in the new supplying countries compared with the previous supplying countries.

## 2.4 An abridged framework for this study

In the current study, the analytical framework has been simplified in order to enable the analysis of two selected raw materials. As explained in the 'Assumptions' and 'Limitations' sections of this report (See Sections 2.5 and 2.6), the focus of this analysis is narrowed down to one potential response in each of the three time frames, excluding the second order effects that could be associated with any potential response and assuming that all other variables remain constant. These scenarios are outlined below:

| **Short term (2 years, 2023-2025)**<br><br>In the short term, the possible buyer's reactions can be limited. If there are alternative supplying countries, the supply chain can shift, and the environmental pressures will most probably shift to the new supplying countries. The difference in global environmental effects in this case will depend on the technologies and regulatory framework in the new supplying countries compared with the previous supplying countries. | |
|---|---|
| Key response | Establishment of trade relations with alternative suppliers |
| Key environmental impacts dependent on: | The difference in environmental impacts in this case will depend on the technologies, energy mix and regulatory framework in the new supplying countries compared with the previous supplying countries. |
| Potential scenarios for Europe | If there is a disruption in the supply of raw material X and demand in the buyer country stays fixed, can demand be met through supply from alternative countries? |

| | |
|---|---|
| | If demand can be fully satisfied by an alternative supplier, what is the relative environmental performance of supply from the 'old supplier' vs. the 'new supplier'? |
| | If demand can only be partially satisfied by an alternative supplier, what is the effect of the quantity disruption of the buyer country? (This scenario allows to take into account second-order effects) |
| **Medium-term (2-7 years, 2025-2030)**<br><br>The medium-term time frame is characterized by moderate response possibilities. In case of a permanent supply chain disruption, demand can be satisfied by alternative supplier countries, and an increase in domestic recycling. The overall environmental effects would depend mostly on the importance of the secondary value chain which is in turn dependent on policies and the maturity of the domestic recycling industry (EEA, 2022). For this scenario, it should also be considered the growth in the raw material demand. | |
| Key response | Growth in domestic capacity for recycling to satisfy demand in addition to establishing trade relations with alternative suppliers |
| Key environmental impacts dependent on: | Share of demand that can be satisfied via recycling rather than primary mining |
| | Relative environmental impact of recycling versus primary mining of the raw material |
| Potential scenarios for Europe | What share of internal demand can be met through domestic recycling? |
| | What is the relative environmental impact of recycling vs. primary extraction? |
| | Is it expected that alternative (primary and secondary) suppliers will enter the market and/or increase their supplying capacity? |
| **Long term** (7 – 17 years, 2030 - 2040)<br><br>In the longer term, the buying country can decide to design and implement more radical strategies of self-sufficiency of domestic supply with the development of fully integrated domestic primary (e.g., mineral extraction and refining) value chains. The overall environmental effects would depend mostly on the importance of the secondary value chain which is in turn dependent on policies and the maturity of the domestic recycling industry (EEA, 2022). The two response options of the short term and medium-term—alternative suppliers and increase in domestic capacity for recycling—are still valid for this scenario. | |
| Key response | Growth in domestic capacity for primary extraction, processing and refining to satisfy demand |
| Key environmental impacts dependent on: | Share of demand that can be satisfied via domestic extraction and processing |
| | Share of demand that can be satisfied via domestic recycling |
| | Relative environmental impact of domestic mining, processing and recycling |
| Potential scenarios for Europe | What share of internal demand can be met through domestic extraction, processing and refining? |

| | |
|---|---|
| | What is the relative environmental impact of domestic vs. foreign extraction, processing and refining? |

The dynamics and potential implications of these responses at a global level are briefly discussed in the conclusion part of the report.

To make this framework dynamic, it should be continuously monitored and updated to adapt to evolving factors, such as market conditions and technological advancements. Implementing such a comprehensive framework would require extensive data collection and modelling techniques. This is outside the scope of the current research.

## 2.5 Analytical assumptions

The framework needs to be built on assumptions based on the best knowledge available at a certain point in time. This of course brings limitations to the research as a consequence. The general assumptions are as follows:

- This research did not enable the researchers to further develop primary data and had to rely on data from previous research. The whole assessment is based on the assumption that the collected data represents a realistic state of knowledge.
- While a variety of other responses exists (as explained in Section 2.1, Theoretical Framework), the study has focused on three key scenarios (i) for the short-term scenario that in case of disruption, alternative countries are available for the EU to completely or partially shift its supply, (ii) for the medium-term scenario, that the technology will continue to develop and make recycling of nickel and REE more accessible and with a reduced energy demand, (iii) an for long-term scenario, that technology and investments enable the EU to increase its mining/refining capacities. In each timeframe and for each potential response it is assumed that all other variables and effects, including the demand in the buyer country (in this case Europe) remain fixed and the 'second order effects' are excluded from the scope.
- The focus of this study has been on the 'environmental' impacts of supply chain disruptions. To have a more comprehensive and realistic evaluation of disruptions, the environmental impacts should be studied in combination with 'economic and social' impacts.

## 2.6 Limitations

The primary limitations of this study stem from the availability of time and resources, as well as broader constraints related to the availability and accuracy of data for conducting a study of this nature. The principal limitations can be outlined as follows:

Data availability

- For most of the calculations made for this study, the aggregated environmental impacts for the supplying countries (old vs new) or for the sector (primary extraction/refining vs recycling) are compared. For a proper assessment of environmental impacts related to these processes, knowledge on the actual location of the production and recycling site is essential.
- The long-term assessment for REEs is based on LCA data available for the Nora Kärr deposit in Sweden, as no known LCA data are yet available for the recently discovered Per Geijer deposit, which will likely be an important ore source in the coming decades, once extraction there begins.
- While Russia serves as the largest provider of refined nickel to the EU, the LCA data of mining and refining in China was chosen for the assessment. This was primarily driven by the limited availability of LCA data on environmental impacts within the nickel sector in Russia (i.e. Ecoinvent database).

- Comparing data is limited by the absence of standardized and transparent information regarding mineral reserves, resources, production, and processing.
- The data on bilateral trade flows is widely available by narrowly defined product groups. However, the availability of country or site-specific trade data for each specific product category can be challenging.

Environmental assessment

- The assessment focuses on Production Environmental Impacts from the perspective of the EU only (excluding second order effects in other parts of the world). The 'system boundary' thus includes life-cycle stages in an attributional setting.
- Because of the global nature of the impacts assessed, the benefits on the environment are defined at the EU level and do not take into account the possibility that other countries fill the gap of the EU shifting their supply.
- Performing a comprehensive LCA covering every step of the value chain and all environmental hotspots proved unfeasible within this task. Therefore, this study narrowed its analysis to specific environmental impact categories such as climate change and ecotoxicity to water.

Material selection

- This report only focuses on nickel class I specifically for battery usage and does not look at nickel in general. Similarly, it focuses on dysprosium and neodymium, two of the most important elements for permanent magnets utilized in many green energy technologies, and not on REE in general. Covering all types of products and assessing the impact of processing and refining their coproducts or by-products could give a more comprehensive picture of the impacts.

## 3 Case study 1: Nickel supply chain disruptions

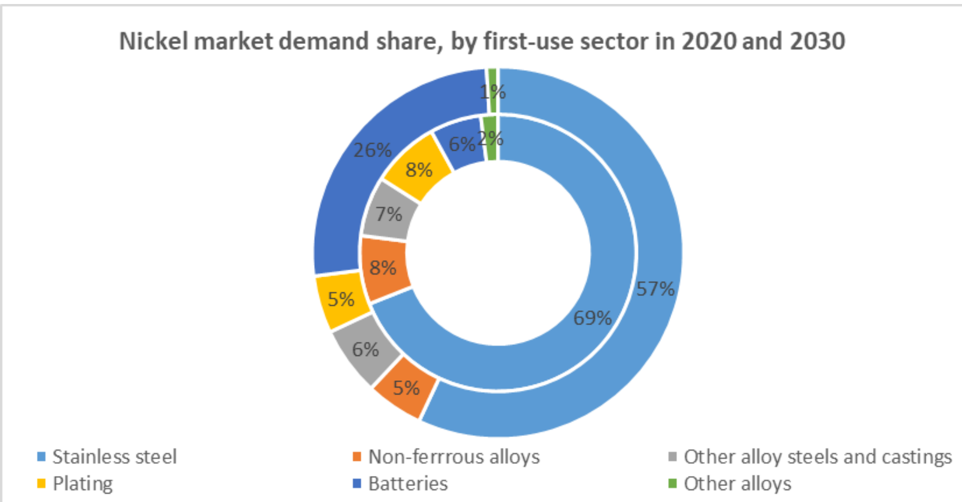
### 3.1 Nickel properties and applications

Nickel (Ni), the fifth most common element on earth (24th most abundant in the earth's crust), is a silver-grey metal with specific physical and chemical properties that make it an essential material in many products and for many applications (Nickel Institute, 2023c; Umicore, 2023). Among others, properties such as high melting point (1,435°C), flexibility, toughness, resistance to corrosion and oxidation, readiness to make alloys, highly ductile, magnetic at room temperature and catalytic properties can be highlighted.

In its various forms, nickel finds applications across a diverse range of products and uses. Traditionally, nickel has primarily found its purpose in the creation of industrial alloys, particularly stainless steel, where it plays a key role by enhancing corrosion resistance and workability in stainless steels (IEA, 2021; INSG, 2021). The stainless-steel industry consumes approximately 70 % of primary nickel production (SCRREEN Project, 2023a; Umicore, 2023). Its intrinsic ability to form alloys, coupled with its ferromagnetic characteristics, positions it as a crucial element in the production of super alloys. These alloys retain their strength under extreme stress and high temperatures, making them an essential material for strategic sectors like aerospace, military, petrochemicals, power, and energy (INSG, 2021; Umicore, 2023). Another key application of nickel is for plating and coating of base metal materials to improve their resistance to corrosion and wear, and increase the properties of hardness, superior strength and ductility (INSG, 2021).

On another level, nickel, due to its physical-chemical properties, have the potential to generate higher energy efficiency and greater storage capacity at a lower cost (Mistry et al., 2016; Nickel Institute, 2023b), a property that makes nickel containing cathodes an essential component for production of batteries and energy storage systems. It is estimated that around 10 % of nickel demand is used for various clean energy technologies, either for batteries or in the form of alloys for renewables and hydrogen (IEA, 2021). An emerging source of demand for nickel is for production of lithium-ion batteries (INSG, 2021). The most used types are nickel cobalt aluminum (NCA) and nickel manganese cobalt (NMC) lithium-ion batteries, with 80 % to 33 % of nickel respectively (Nickel Institute, 2023b). Nickel is also used in other types of batteries such as nickel-cadmium (NiCd) and nickel metal hydride (NiMH) batteries (INSG, 2021). Due to its critical role in battery production, the global demand for nickel and nickel products is expected to increase significantly in the coming years. Nickel market demand by sector in 2020 and the expected changes in share of various applications in 2030 are shown in Figure 3.1 (Fraser et al., 2021).

**Figure 3.1. Global nickel application by sector in 2021 (inner circle) and the estimated values in 2030 (outer circle).**




**Source:** Fraser et al., 2021

## 3.2 Nickel supply chain

Nickel ore deposits are formed in various geological settings and conditions such as magmatic, hydrothermal, sedimentary, and metamorphic processes and lateritic weathering. In these deposits nickel would mainly occur as oxides, sulfides and silicates. The most common nickel deposits can be categorized into two groups:

- **Sulfide Nickel Deposits:** These are typically associated with mafic and ultramafic intrusive rocks and are characterized by the presence of nickel and other metals (e.g., copper, cobalt, iron and occasionally PGEs) in form of sulfide disseminated in the host rock or as discrete ore bodies. Pentlandite, with a chemical composition of $(Fe,Ni)_9S_8$, is the most common nickel-bearing sulfide mineral that occurs in ore deposits. These types of deposits are usually found in less tropical zones such as Canada, Russia and Australia.
- **Lateritic Nickel Deposits:** These types of deposits, also known as oxide deposits, are formed through the weathering of ultramafic rocks in tropical or subtropical regions. Over time, prolonged weathering of ultramafic rocks causes leaching and accumulation of nickel and other elements in the soil, resulting in the formation of lateritic soil profiles. Lateritic deposits are classified into limonites and saprolites. Limonite is located in upper parts of a lateritic profile, while saprolite is a deeper layer soil (Nurjaman et al., 2021). Nickel is mainly absorbed in clay in the ore rather than in minerals (IEA, 2021). Lateritic nickel deposits are commonly found in tropical countries such as Indonesia, The Philippines, and New Caledonia.

Around 30-40 % of the global nickel production originates from sulfide deposits; the remaining 60-70 % from lateritic ores (INSG, 2021). The nickel concentration differs in various ore types. Generally, sulfidic deposits have a higher concentration of nickel, typically in the range of 1.5-3.0 %, than the lateritic ores. Saprolite has a relatively higher grade (1.5-2.5 %) than limonite (0.8-1.8 %) (INSG, 2021).

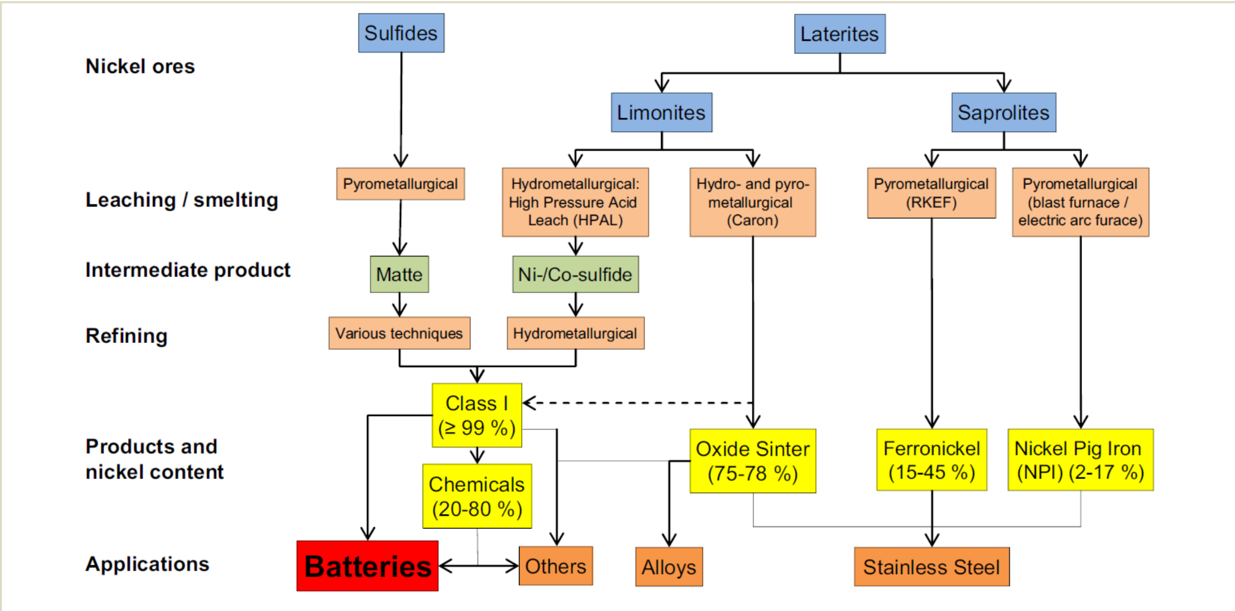
While in almost all nickel sulfide deposits, nickel is the main economic commodity, these deposits host significant amounts of copper. Depending on its concentration, the accompanying copper is normally extracted as either coproduct[8] or as by-product of nickel (Nassar et al., 2015). In addition to copper, most of the nickel deposits also host other by-products such as cobalt, PGEs and silver. In some cases, these companion metals (Nassar et al., 2015) can have a significant impact on the economies and value of the deposit.

Due to the differences in deposit types, their mineralogy and their nickel concentrations, there are different process methods and routes for primary nickel production. The most important processing routes, type of end products and their application are illustrated in Figure 3.2 (Schmidt et al., 2016).

Sulfide ores are normally located in deeper subterrestrial areas, requiring underground mining operations. In some cases, an early-stage open pit mining might be required. The mined ore is processed through various beneficiation processes, such as crushing, grinding and separation by floatation or magnets to generate concentrates (IEA, 2021; Schmidt et al., 2016) with relatively high (18 %) nickel content (Ali et al., 2023). These concentrates are suitable for pyrometallurgical (or smelting) processes, which in turn leads to generation of matte, an intermediate nickel-iron sulfide with 25-40 % nickel. In the next step, iron in the matte is oxidized, combined with a silica flux to form a slag. After drowning off the slag, a matte with 70 to 75 % nickel is produced. By implementing various refining techniques, such as pressure leaching, roasting, electrorefining or carbonyl refining, matte can be converted to class I nickel (Schmidt et al., 2016).

8 When mining metals together, the metal with the highest economic contribution is considered as host. Metals with similar economic contribution are coproducts and metals with contribution of around 20% or less of revenue are considered as by-products.

**Figure 3.2. The most common nickel primary production routes, intermediate and end products and their applications**




**Source:** Schmidt et al., 2016

The mining of lateritic ore is usually done by open pit mining methods to extract the nickel rich strata and discard the related waste materials or overburden (Norgate & Jahanshahi, 2011). Due to their higher moisture content and different chemical composition, the ore preparation phase for the lateritic deposits differs from the sulfidic types. In most cases, this includes crushing, screening, and drying processes (Ali et al., 2023). Limonite ore is mostly processed via hydrometallurgical processes and by implementing high-pressure acid leaching (HPAL), where sulfuric acid is used to leach nickel (and cobalt) under an average pressure of 4,500 kilo Pascal (kPa) (Ali et al., 2023). Then, the acid is neutralized including sulfide precipitation and as a result a mixed nickel-cobalt sulfide is produced (Ali et al., 2023). The intermediate product after hydrometallurgical process is nickel-cobalt sulfide and mixed hydroxides (Ali et al., 2023; Schmidt et al., 2016). Although the number of operating HPAL projects is currently small, several new projects are being built or are planned (IEA, 2021). Saprolite ore is generally processed through pyrometallurgical processes, via rotary kiln-electric furnace (RKEF), blast furnace or electric arc furnace (Schmidt et al., 2016).

Depending on the type of deposit, composition and concentration of nickel in the ore and processing routes and methods, three main types of refined nickel is produced (as shown in Figure 3.2 above):

- **Class I Nickel**, also known as 'nickel metal', contains more than 99.8 % nickel and can be produced through a combination of pyrometallurgical and hydrometallurgical processing of sulfide deposits, hydrometallurgical processing of limonite deposits or recycling of batteries and scraps (Mistry et al, 2016 and Schmidt et al, 2016). Currently the majority of the class I nickel is produced from nickel sulfide deposits (IEA, 2021). Class I products can be in the form of electrolytic cathode, briquettes and powder, and carbonyl powder and pellets (Fraser et al., 2021). This type of nickel is used in several applications such as for production of stainless steel, alloy steel, nickel alloys, electroplating and batteries (INSG, 2021; Schmidt et al., 2016).
- **Class II Nickel** contains less than 99.8 % nickel and includes nickel oxide sinter, ferronickel and nickel pig iron (NPI), with 75-80 %, 15-45 % and 2-17 % nickel content, respectively (Schmidt et al., 2016). These types of products are generally used for stainless steel production.

- **Nickel chemicals** include a wide range of compositions such as sulphates, acetates, carbonates, chlorides and oxides (Schmidt et al., 2016) and their nickel content varies between 20 to 80 %. Among other uses, nickel sulphate is an essential feedstock in the production of lithium-ion battery cathodes (Ali et al., 2023).

In complement to the common primary production route of nickel, recycling is also an important step in nickel's life cycle and an important source of this critical metal (Nickel Institute, 2023c). Nickel and nickel containing alloys can be effectively recycled without any loss of quality and properties. Based on a JRC report (2020), in 2016 and for Europe, the end-of-life recycling rate (EoL-RR) of nickel was reported to be 42 %, while the end-of-life recycling input rate (EoL-RIR) was around 16%(Torres De Matos et al., 2020).

Given the dominant use of nickel in production of stainless steel and other alloys (as shown in Figure 3.1 above), most of the nickel recycling activities are currently related to the stainless-steel industry (Torres De Matos et al., 2020). The recycling rate of nickel scrap recycling are estimated to be around 68 %, making it among the metals with the highest recycling efficiency (Nickel Institute, 2018). Production of stainless steel considers the use of recycled material, including stainless steels and other nickel alloys, mixed turnings, waste from primary nickel producers and re-melted ingot from processing nickel-containing slags, dusts, batteries and others. Although special alloys are preferred to be recycled as mono-material, different alloys and products may get mixed and blended to maintain quality(Torres De Matos et al., 2020). An alternative avenue to obtain secondary sources of nickel is through the recycling of batteries. Currently, several technologies exist for recycling batteries from different waste streams (Silvestri et al., 2020). These methods typically encompass a combination of four key steps, which include pre-processing, pyrometallurgy, mechanical processing, and hydrometallurgy (Boyden et al., 2016; Sheth et al., 2023).

The pre-processing stage involves disassembly, stabilization and sorting based on the type, chemical composition, shape, size and density (Sheth et al., 2023). In the pyrometallurgical process the metals and oxides in the battery are melted and converted into a copper, iron and nickel alloy, known as matte via redox reactions at approximately 500-600 $^{o}$C. Discharge and deactivation of batteries is done in the pre-processing and pyrometallurgical process (Sheth et al., 2023).

The metallurgical process is followed by mechanical processing and pre-treatment during which the battery components will be crushed and pulverized to increase the surface area of the component metals (such as iron, cobalt, copper, nickel, lithium, and aluminium). After this, a number of mechanical separation methods such as magnetic or density separation, froth floatation, sieving and vibrating drum screens are used. This separation process is normally followed by extractive hydrometallurgical methods either in form of solvent extraction based hydro treatment or hydrometallurgical based treatment process (Abdelbaky et al., 2021). The former involves leaching the final electrode material in a sulfuric acid finally extracting nickel (and copper) by pH value manipulation (Abdelbaky et al., 2021). The hydrometallurgical based treatment process, the extraction phase is performed by adjusting the pH value of the leaching acid medium with sodium hydroxide (NaOH) to co-precipitate targeted metals in the form of metal hydroxides. The advantage of the latter method is that metals can be efficiently recovered without using any solvents (Abdelbaky et al., 2021).

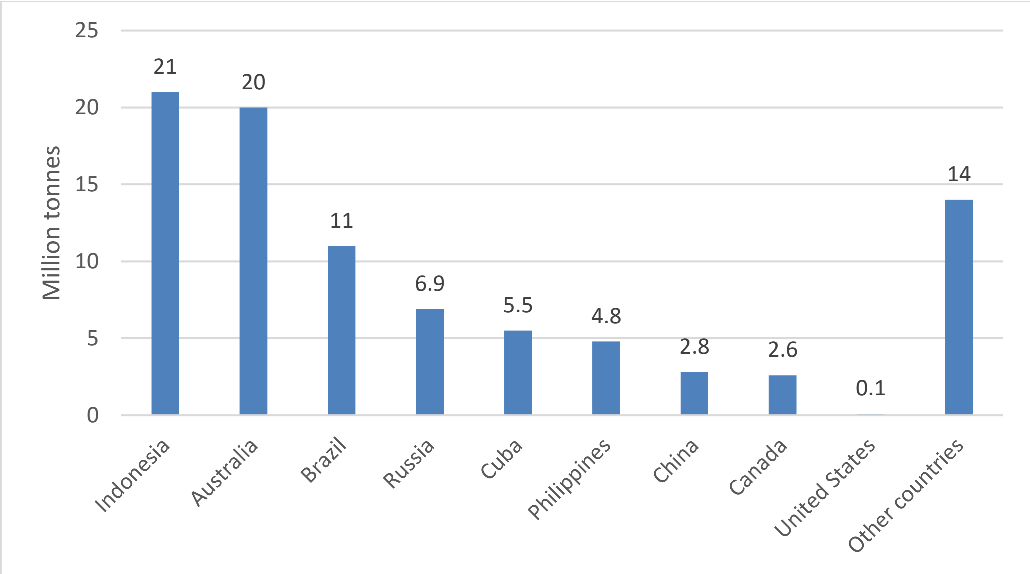
## 3.3 Overview of global and European supplying countries

While studying global production or trade of nickel it is crucial to distinguish between various types of nickel products, which will in turn have an effect on their market demand and trade. As such in this chapter the production and trade data are provided for nickel reserves and mine products, intermediates, refined (class I nickel), nickel sulphate and secondary (recycled) nickel.

Global nickel reserves and nickel mine production

The largest **nickel reserves**[9] in the world are reported to be in in Indonesia, Australia, Brazil, Russia, Cuba, Philippines, China and Canada (Figure 3.3) (INSG, 2021). While the nickel ores in Russia, Canada, Australia and China are mainly sulfide deposits, the ones in Indonesia, Brazil, the Philippines, Cuba are mainly lateritic type (IEA, 2021).

**Figure 3.3. World nickel reserves estimation, million tonnes**

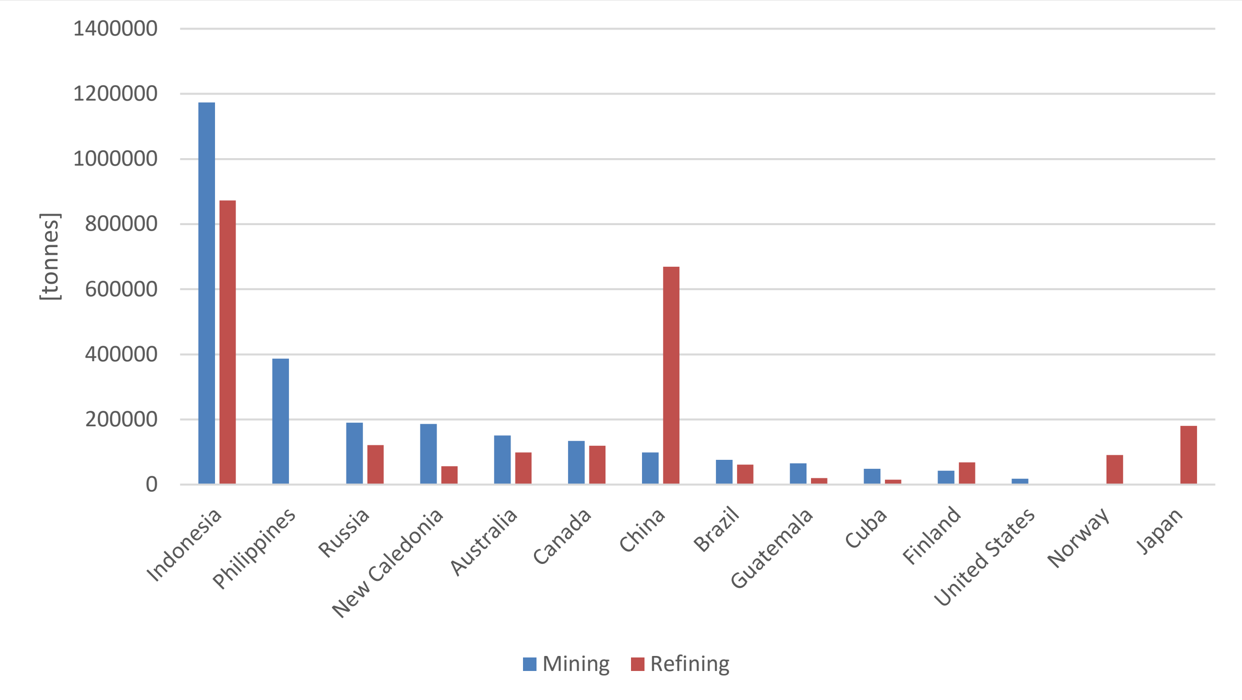



**Source:** INSG, 2021

Based on the International Nickel Study Group (2021), in 2020, there were 27 countries with nickel mining activities in the world. Figure 3.4 below shows the top nickel mining countries and their importance in the refining stage. Currently, Indonesia and the Philippines represent around 55 % of global mine production (Idoine et al., 2023). The largest active nickel-producing mines in 2021 are reported to be in Russia (Kola MMC Mine and Sorowako Mine), followed by the mines located in the Philippines (Taganito Mine and Rio Tuba Mine) (GlobalData, 2021).

9 Reserves is defined as: Identified resources of mineral fuel-bearing rock from which the mineral or fuel can be extracted profitably with existing technology and present economic conditions. The concept can be used in global, regional, or local senses, or applied as a measure of the remaining effective life of an individual mine.

**Figure 3.4. Origin of the mined and refined nickel supplied to the world, 2021, in tonnes**




**Source**: Idoine et al., 2023

In 2021, the **EU-27** accounted for only 1.7 % of global nickel primary production (Joint Research Centre, 2023). Within the EU-27, Finland and Greece are the major mining countries, contributing respectively more than 87 % and 11 % of the total EU production in 2021 (Joint Research Centre, 2023a). Poland's contribution amounts to approximately 1 % of the overall nickel production within the EU (SCRREEN Project, 2023a). In Finland nickel is mined from sulfide ore deposits and production is carried out by two companies, namely Terrafame and Boliden. Both of these mining companies process their mine production into intermediate products, which are then refined elsewhere. The lateritic nickel ore in Greece is mined by the Greece's General Mining and Metallurgical Company (LARCO) which operates the mine and produces ferronickel, which is mostly exported to Spain, Italy and Belgium (Fraser et al., 2021). Between 2020 and 2040, the EU-27 mine production is expected to rise by around 0.5 % per year and this is mainly due to increased output from the mine in Finland for its integrated nickel sulphate production. There are no advance exploration projects or indication of new capacity in the medium to long-term within the EU-27. As such, based on the current knowledge of nickel reserves in the EU-27, the share of EU-27 in global mine supply is forecasted to decline by 2030 and 2040 (Fraser et al., 2021).

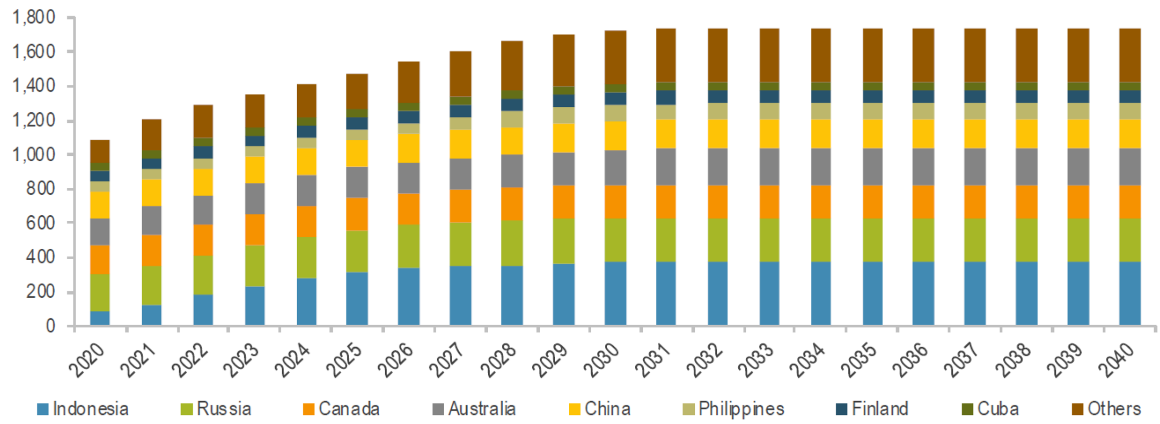
Intermediate products

In term of **intermediates**, Russia is currently the world's largest producer, followed by Canada, Australia, China, Philippines, Finland, and Cuba. The total intermediate production in 2020 is estimated to be 1.1 Mt and this is expected to grow to 1.7 Mt by 2040. The global intermediate production, major countries and expected trend in production by 2040 are shown in Figure 3.5 (Fraser et al., 2021). Indonesia is expected to be the main contributor to this growth due to their plans in establishing and improving pyrometallurgical processing (HPAL plants) for their laterite ores to produce intermediates suitable for battery production (Fraser et al., 2021; IEA, 2021).

The share of the **EU-27** in the global production of intermediate products is estimated to be 5.7 %. With 95 % share of the production in the EU, Finland is the main producer of nickel intermediates, followed by Germany, Poland, and Sweden. The majority of nickel intermediate production in Finland comes from operations in Boliden and Terrafame companies, and to a much lesser extent from crude nickel sulphate from Mondo (Fraser et al., 2021).

The Boliden's Harjavalta smelter in southwest Finland is the only nickel smelter in Western Europe, producing nickel matte. The nickel concentrate needed for matte production is partially sourced from their own mines in Finland supplemented by concentrates from other external mines, mainly Canada, South Africa, Norway and small quantities from Russia (Boliden Company, 2023; Fraser et al., 2021). The company has announced plans for increasing its matte production capacity (Fraser et al., 2021). The Terrafame company produces mixed sulfide precipitate (MSP), one of the key forms of nickel intermediate products (Fraser et al., 2021).

**Figure 3.5. Expected total intermediate production by country, 2020-2040 (kt)**




*Source:* Roskill, 2020.

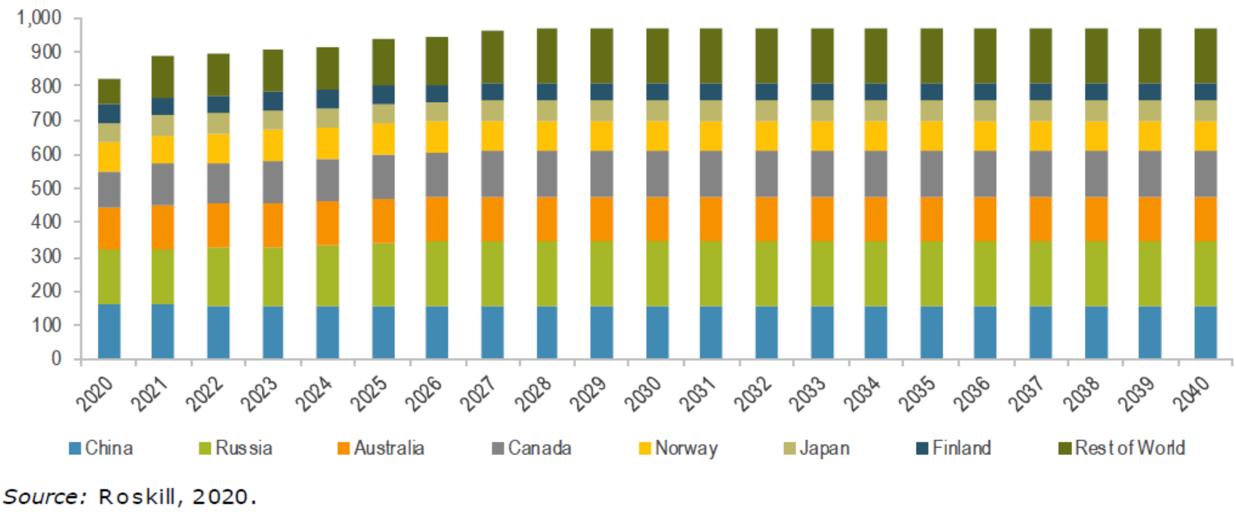
**Source:** Fraser et al., 2021

Global refined nickel

In 2021, the main **refined nickel producers** were China and Indonesia refining close to 60 % of the total refined nickel worldwide (see Figure 3.4).

Looking specifically at **Class I nickel**, the major producers in 2019 were China, Russia, Australia, Canada, Norway, Japan, Finland (Fraser et al., 2021; USGS, 2018). Among various types of class I products, nickel cathode, briquettes and carbonyl powders and pellets are the main types produced in these countries (Fraser et al., 2021). Based on the forecast study conducted by Roskill (Fraser et al., 2021) and as shown in Figure 3.6, class I production is expected to increase by 2030 and might be stabilized in the period between 2030 and 2040. Currently several countries are planning to expand their class I nickel refining operations. However, lack of new major nickel sulfide discoveries in recent years can lead to low production growth rates in the future (Fraser et al., 2021).

**Figure 3.6. Expected total Class I production by country, 2020-2040 (kt)**




**Source:** Fraser et al., 2021

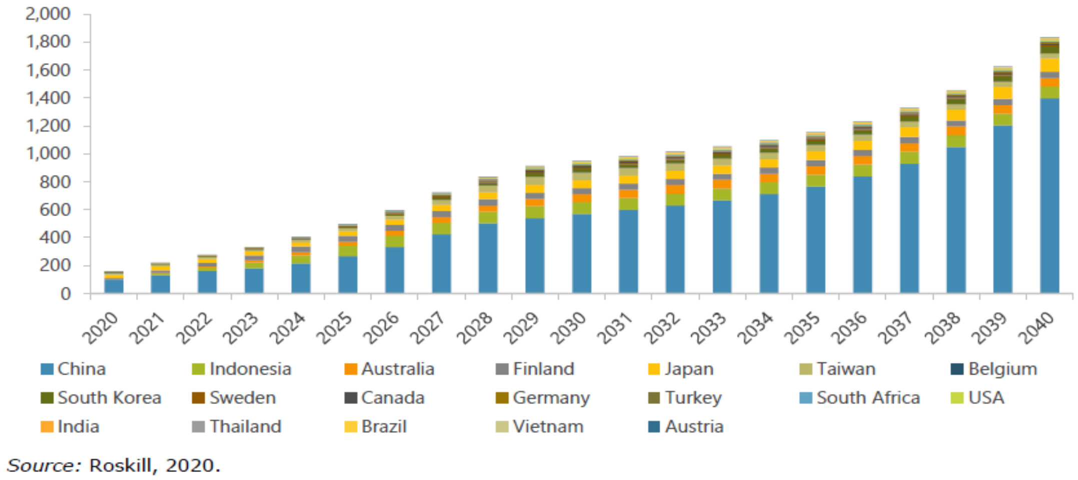
Nickel sulphate

Nickel sulphate, which is a chemical product from high-purity refined nickel, is necessary for the manufacturing of lithium-ion batteries. In 2021, the main producers of nickel sulphate were China, with more than half of the global production, followed by Japan, Finland, Taiwan, Indonesia, Belgium, Australia, South Africa and USA. In this year, the EU is projected to account for roughly 10 % of the total nickel sulphate production, with significant contributions from Finland, Belgium, and Germany (Fraser et al., 2021).

Nickel sulphate is produced by integrated or non-integrated producers. Based on a recent study (Fraser et al., 2021), nickel sulphate production from integrated producers is expected to grow by about 10 % per year between 2020 and 2040. This growth is especially expected in the coming years because existing producers are expected to either operationalize their new projects or introduce new expansion plans. Furthermore, intermediate producers (such as Indonesia) are expected to shift their focus toward integrating nickel sulphate production into their operations. Production from non-integrated producers is expected to grow strongly at about 15 % per year by 2040. The majority of growth is expected to come from conversion of intermediate and recycled material feedstock to nickel sulphate (Fraser et al., 2021).

Outlook for nickel sulphate production by country is shown in Figure 3.7 (Fraser et al., 2021). China is expected to remain the main producer of nickel sulphate owing to expansion from existing producers and growth in non-integrated producers. Indonesia, Finland and Australia are expected to become the next largest producers (Fraser et al., 2021).

**Figure 3.7. Outlook for nickel sulphate production, by country 2020-2040, kt**

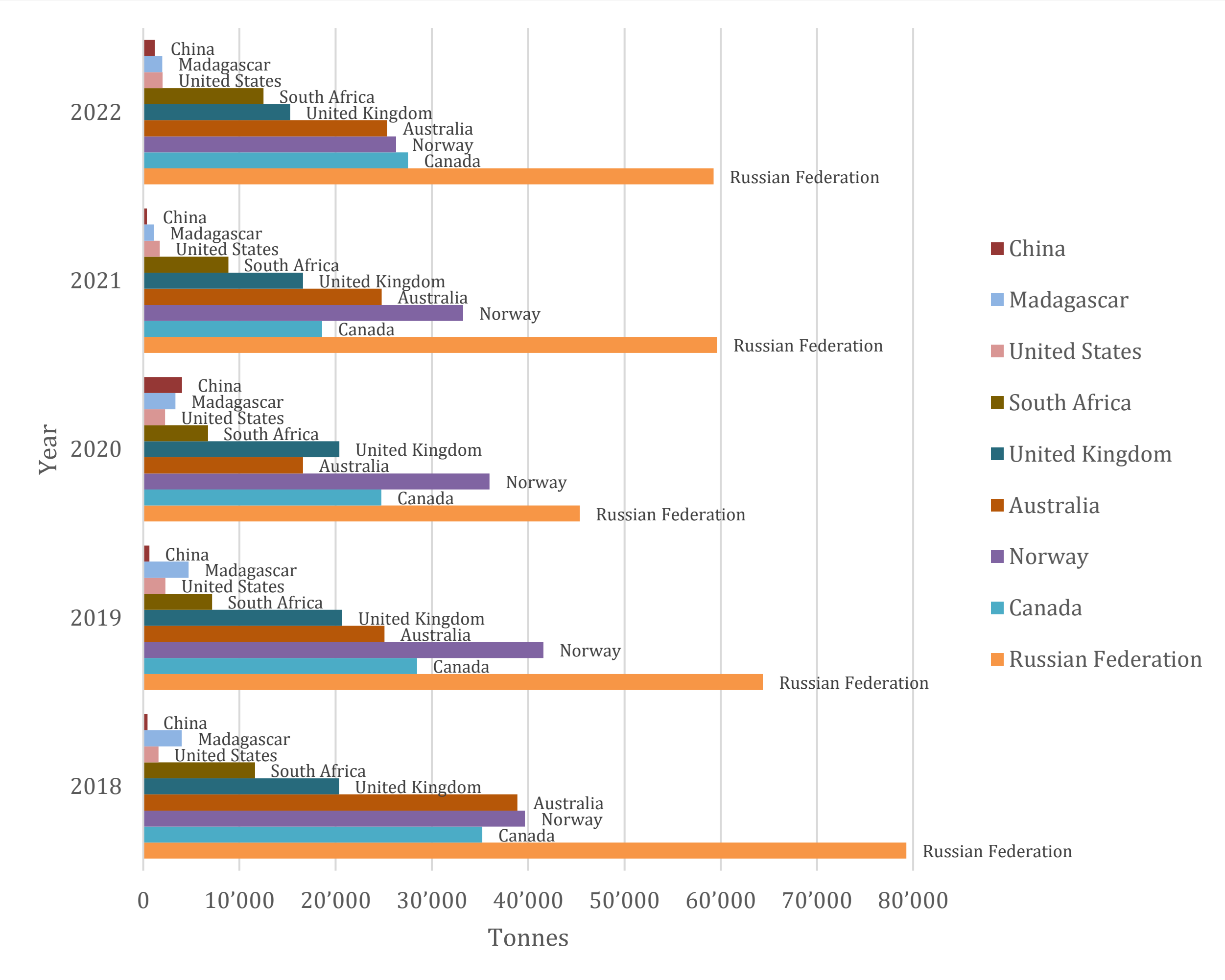



**Source:** Fraser et al., 2021

Supply at the EU level

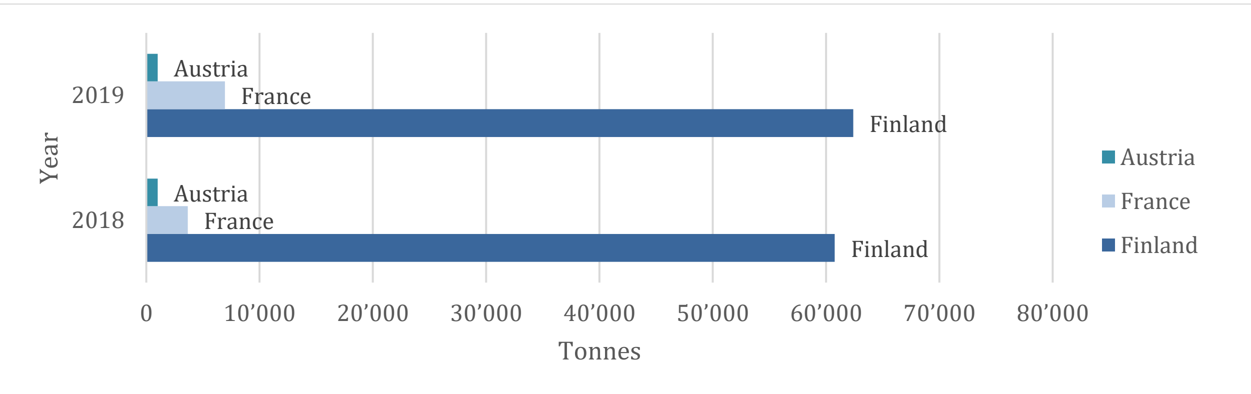
At the EU level, the supply of refined nickel Class I (including salts and oxide) is largely dominated by Russia (Figure 3.8) and Finland (Figure 3.9). Although those two countries cover more than 60 % of the total EU supply, the rest (40 %) is covered by a big diversity of other nations. The most important ones being Canada, Norway, Australia, United Kingdom, South Africa, United States, Madagascar, and China (Figure 3.8 and Figure 3.9). As detailed in Figure 3.8, Finland is an important source of refined nickel Class I for the EU. The data after 2019 is missing but this shows the already strong reliance of the EU on internal refined nickel.

**Figure 3.8. International supplying countries for the EU for refined nickel Class I (including salts and oxides).**




**Source**: Joint Research Center, 2023.

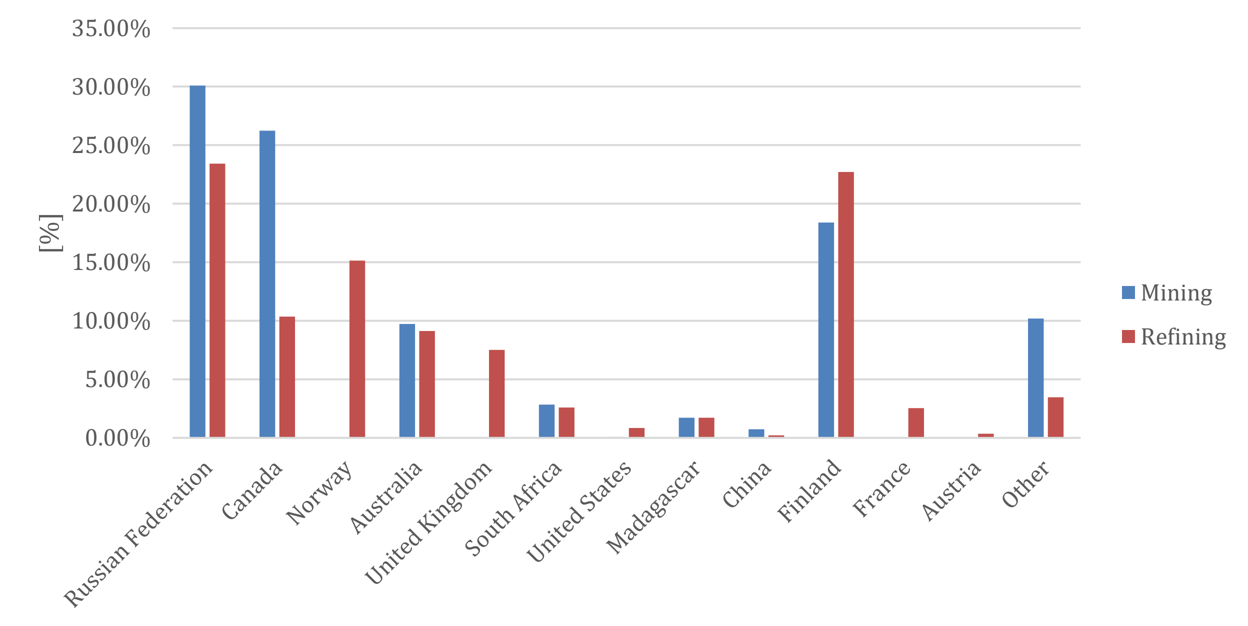
**Figure 3.9. EU supplying countries for the EU for refined nickel Class I (including salts and oxides), in tonnes**




**Source**: Joint Research Center, 2023.

In terms of the origin of the mined ore, nickel coming in the EU is mostly mined in Russia, followed by Canada, Finland and Australia (Figure 3.10). China is among the smallest supplying country of mined ore for the EU.

**Figure 3.10. Origin of the mined and refined nickel Class I, supplying the EU, 2019**



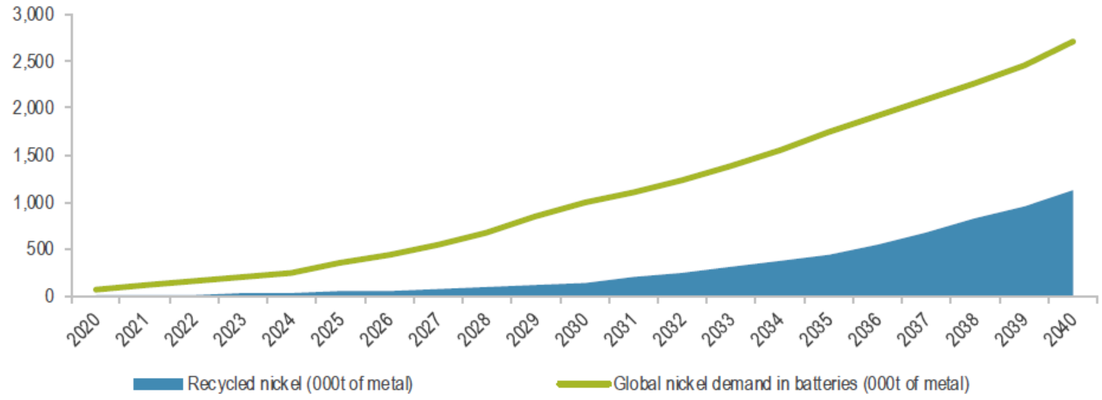

**Source**: Joint Research Center, 2023.

<u>Secondary sources (recycled nickel)</u>

As mentioned in the previous chapter, using **secondary sources** of nickel for stainless steel production is a common practice in this industry. Many stainless steel producers get their nickel input from stainless steel scrap, rather than primary nickel. It is predicted that the amount of nickel sourced from stainless steel scrap will increase in the coming years. However, since this type of secondary nickel is directly used by stainless steel industry it is not available to other sectors such as for batteries. In the EU-27, the key stainless-steel producers such as Finland, Germany, Italy, Spain and Sweden are reported to be net importers of scrap (Fraser et al., 2021; SCRREEN Project, 2023a).

As predicted in several studies (Abdelbaky et al., 2021; Fraser et al., 2021; IEA, 2021) and shown in Figure 3.11 below the nickel supply from secondary sources (from both battery and non-battery sources) is expected to increase in the coming years.

**Figure 3.11. Predicted availability of secondary nickel from end-of-life battery recycling compared with the global nickel demand in batteries, 2020-2040, kt**




**Source:** Fraser et al., 2021

## 3.4 Environmental impacts of nickel production

Nickel production from primary resources is reported to be one of the most energy intensive processes and is associated with significant water and consumption and climate impacts (Norgate & Jahanshahi, 2011). Based on a recent life cycle assessment study conducted by Sphera for Nickel Institute (Nickel Institute, 2023a), the main environmental impact categories for production of three key types of nickel products are listed in Table 3.1. This assessment is based on the input from members of the Nickel Institute for the year 2017, representing 52 % of the global production of Class 1 nickel; 47 % of the global production of ferronickel and 15 % of the global production of nickel sulphate and focusing on the impacts of cradle to gate production of nickel sulphate (Nickel Institute, 2023a).

**Table 3.1. Summary of impact categories for the three main primary nickel products**

| Impact category | 1 kg Class I Ni* | 1 kg Ni in FeNi** (27% Ni in FeNi) | 1 kg NiSO4*** |
|---|---|---|---|
| Global Warming Potential [kg $CO_2$ eq.] | 13 | 45 | 4 |
| Acidification Potential [kg $SO_2$ eq.] | 1.4 | 0.17 | 0.26 |
| Primary Energy Demand [MJ] | 236 | 592 | 68 |
| Eutrophication Potential [kg Phosphate eq.] | 5.2E-03 | 0.016 | 1.5E-03 |
| Photochemical Ozone Creation [kg Ethane eq.] | 0.055 | 0.010 | 0.010 |
| Blue water consumption [kg] | 106 | 924 | 49 |

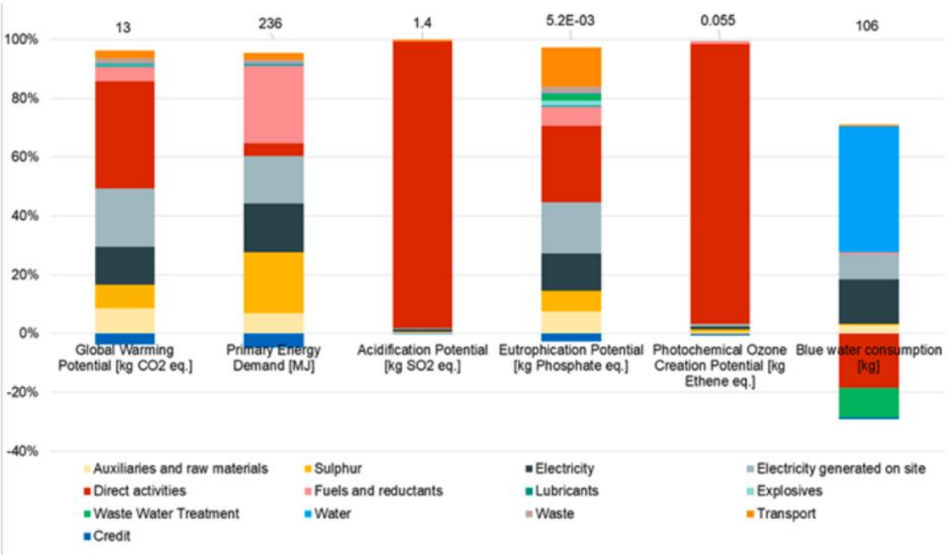
*1 kg of Class 1 nickel (>99.8), **1 kg nickel in ferro nickel (with a reference Flow of 3.7kg ferronickel based on 27% nickel content), *** 1 kg of nickel sulphate hexahydrate (nickel sulphate) (22% nickel content)

**Source:** Nickel Institute, 2023a

The metallurgical stages of the nickel production processes are the main contributors to environmental impact of all nickel products (Nickel Institute, 2023a). These processes are energy intensive, as the chemical bonds in the nickel-containing mineral must be broken to liberate the metal (Mistry et al., 2016). A life cycle study (LCA) study on the environmental impacts of nickel sulphate production reveals that in term of value chain stages, the beneficiation step represents up to 68 % of the environmental impact, the primary extraction and processing up to 40 % and refining and creation of nickel sulphate up to 44 % of the total environmental impact (Mistry et al., 2016).

As an example, the contribution of various sources to impact categories for production of 1 kg Class I nickel is illustrated in Figure 3.12.

**Figure 3.12. Contribution of various sources to impact categories for production of 1 kg Class I nickel**




**Source:** Nickel Institute, 2023a

Despite the lower estimated energy intensity of the mining process for lateritic (oxidic) ore deposits (Mistry et al., 2016), the life cycle-based energy consumption for metallurgical processing (especially when the pyrometallurgical process is involved) of these ores are reported to be higher compared to sulfide ores (G. M. Mudd, 2010; Norgate & Jahanshahi, 2011). Several factors contribute to the higher energy demands for processing lateritic ores, including (Norgate & Jahanshahi, 2011; Schmidt et al., 2016):

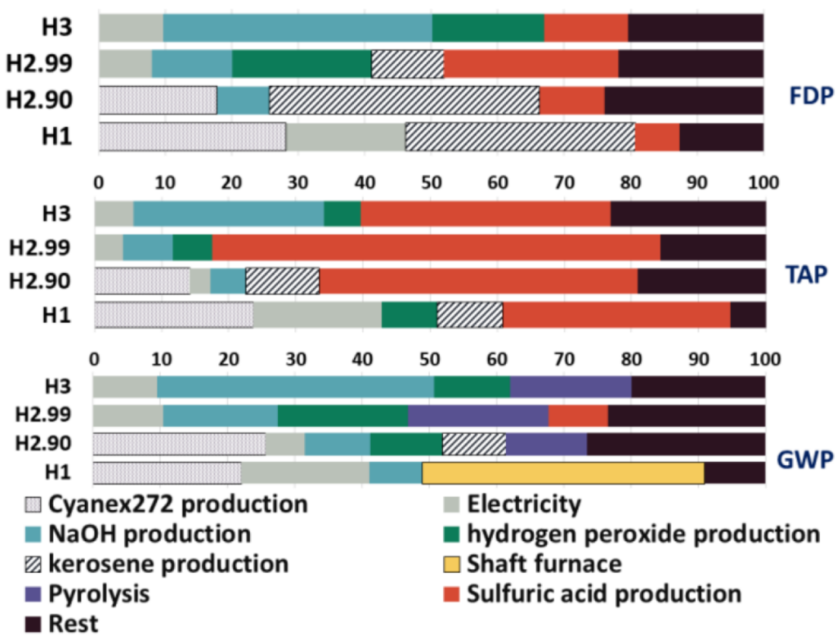
- Since laterites have lower concentration of nickel, the metal extraction stage is more energy intensive than sulfidic ores. For example, for ferronickel production, the ore should go through the rotary kiln/electric furnace for ferronickel production;
- Due to lower grades, larger amounts of laterite ores are required to be processed to yield an equivalent quantity of nickel metal;
- While in sulfide ores, sulfur acts as fuel source, in laterites with lower sulfur and higher moisture content, more energy is required for smelting;
- The location, depth and shape of the ore deposit can also have a major impact. Deposits in remote or relatively inaccessible areas require more infrastructure, specialized equipment and greater energy consumption.

The need for higher energy demand for processing lateritic deposits is an important point to consider because due to the ageing of existing sulfide deposits and lack of new ones, the extraction of nickel from laterites is expected to grow in the coming years (IEA, 2021; G. Mudd & Jowitt, 2014; Schmidt et al., 2016). Additionally an essential consideration is that for a proper assessment of environmental impacts related to the production processes, knowledge on the actual location of the production site is crucial (Schmidt et al., 2016).

The environmental impacts of production of secondary sources of nickel are investigated in several LCA studies. More specifically, the impact of lithium-ion battery recycling has been the attention of researchers in recent years. For hydrometallurgical based recycling, most of the GWP impacts are generated in the hydrometallurgical treatment stages, whereas most of the avoided burdens happened in the dismantling stage where 34 % of the battery mass could be recovered (Abdelbaky et al., 2021).

The results of a comparative life cycle assessment for three generic recycling routes of nickel-cobalt-manganese (NMC111) li-ion batteries are shown in Figure 3.13 (Abdelbaky et al., 2021). The studied processes are a three stage pyrometallurgy based treatment process (marked as H1), a solvent extraction based hydro treatment process (H2.90 and H2.99[10]) and a hydrometallurgical based treatment process (H3). The study reveals that the key environmental hotspots of battery recycling include extraction solvent, water emissions from the wastewater treatment process and electricity consumption (Abdelbaky et al., 2021; Silvestri et al., 2020). These impacts are higher when using hydrometallurgical processes, specially solvent extraction based hydro treatment process (Abdelbaky et al., 2021; Silvestri et al., 2020). The reason is the massive chemical consumption that characterizes this technique, or the generation of waste that will require further reprocessing stages (Silvestri et al., 2020).

**Figure 3.13. Contribution of different processes in battery recycling to the environmental impacts from fossil depletion potential (FDP), terrestrial acidification potential (TAP) and global warming potential (GWP).**




**Note:** H1= pyrometallurgy based treatment process, H2.90= solvent extraction based hydro treatment process with 90% solvent reused, H2.99= solvent extraction based hydro treatment process with 99% solvent reused, H3= hydrometallurgical based treatment process

**Source:** Abdelbaky et al., 2021

Despite the negative environmental impacts associated with nickel recovery from various battery recycling processes, the LCA studies show that these impacts are generally less than the impacts from battery production using virgin materials (Dunn et al., 2015; Silvestri et al., 2020). Based on a study by Silvestri et al (2020) on recycling technologies of nickel-metal hydride (NMH) batteries, the best benefits of recycling

[10] H2.90 refers to a scenario suggested in this study where 90 % of solvent can be reused, whereas H2.99 refers to a case where 99 % of the solvents can be reused (Abdelbaky et al, 2023).

processes are reported to be abiotic depletion and human toxicity. However, the global warming potential (GWP) is reported higher than battery production, due to higher demand for energy in recycling processes (Silvestri et al., 2020).

## 3.5 Environmental effects of supply chain disruptions in short-, medium- and long-term

With the ongoing growth in electric mobility, there will be an increasing market demand for nickel products. The worldwide nickel market demand share for batteries is expected to increase from 6 % in 2020 to 50 % in 2040 (Fraser et al., 2021). At global level, Russia is the main producer of mined nickel and China serves as the primary manufacturer of essential intermediate and refined nickel components required for battery production. Should there be any shifts in Russia's or China's economic or trade policies that lead to a potential deficiency in the demand from the buying country, an examination of three distinct scenarios across varying time spans is undertaken by delving into the potential environmental impacts of these scenarios, as outlined in Sub-section 2.4 and explained below.

**Short term (2023-2025) – Change to an alternative supplier**

In this scenario it is assumed that the total or a portion of the EU demand for nickel that was met by China, could be replaced in short term by an alternative supplier such as Canada. Although Russia is the biggest supplier of refined nickel to the EU (Figure 3.10), China was selected for this pilot study because of the Ecoinvent data availability on environmental impacts of the nickel production in the country. No Ecoinvent data was available for Russia. As China also refines all of the nickel mined domestically, this eased the calculations for the impacts of the mining and refining stage. Canada was selected as alternative for this scenario due to its major role in the global mined, intermediate and refined nickel production (see Sub-section 3.3). Additionally, all the nickel refined in Canada is sourced from domestic mines, therefore the impacts of the mining stages can be used for the quantities of nickel refined in the country. Furthermore, the availability of data for making the assessment was an important criterion for the selection of this alternative supplier. The assumptions made for this chapter are detailed in sub-section 2.5 (Analytical assumptions).

To conduct an environmental assessment, the environmental hotspots for the nickel value chain were considered. Based on the literature review and as mentioned in the previous chapter, the hotspots' level of impact may vary from one country to another or from one process to another. The steps which can have significant impacts on the environment are (Ali et al., 2023):

- The beneficiation step, which can represent up to 68 % of the total environmental impact.
- The primary extraction and processing: which can go up to 40 % of the total environmental impact.
- The refining and creation of sulphate step which can represent up to 44 % of the total environmental impact.

The assessment was performed using the Ecoinvent database and the ReCiPe 2016 v.03, midpoint (h), as impact assessment method. Given the availability of the data, the chosen impact categories were:

i. Climate change also seen as global warming potential in kg $CO_2$-Eq,
ii. Energy resources non-renewable, referring to the non-renewable energy used for the functional unit in MJ,
iii. Ecotoxicity to water, including freshwater, marine water, and terrestrial water in kg 1,4 dichlorobenzene (1,4-DB) eq.

The following processes were selected from Ecoinvent:

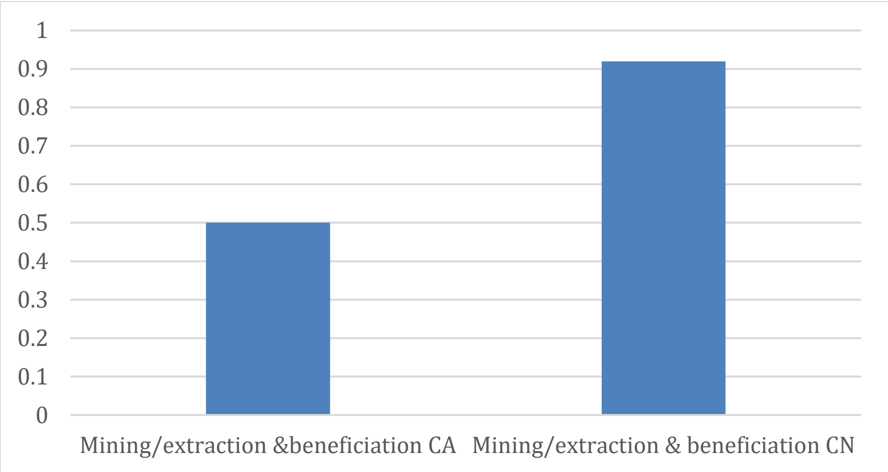
- Data for Canada (Quebec): [nickel mine operation and benefication to nickel concentrate, 16% Ni | nickel concentrate, 16% Ni | APOS, S – CA-QC]
- Data for China: [nickel mine operation and benefication to nickel concentrate, 7% Ni | nickel concentrate, 7% Ni | APOS, S – CN]

- Data for a global average of refining stage: [market for nickel sulphate | nickel sulphate | APOS, S – GLO]

The preliminary results of the assessment for the three chosen impact categories are shown in Figures 3.14 and 3.19.

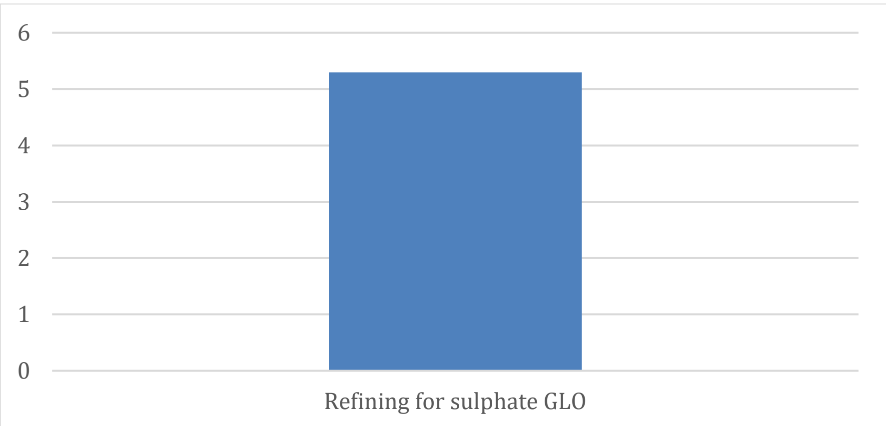
As illustrated in Figure 3.14, the global warming potential of mining and beneficiation process in Canada is 45 % lower than the impact caused by these processes in China. Furthermore, the analysis reveals that the intensity of GWP of the refining process for production of nickel sulphate (global average) is almost five times higher than the impact for nickel mine production in Canada and China (Figure 3.15)

**Figure 3.14. Impact on climate change of the mining and beneficiation stages (comparison between Canada (CA) compared with China (CN)). FU: 1kg of nickel concentrate.**




**Source:** Ecoinvent data, 2023

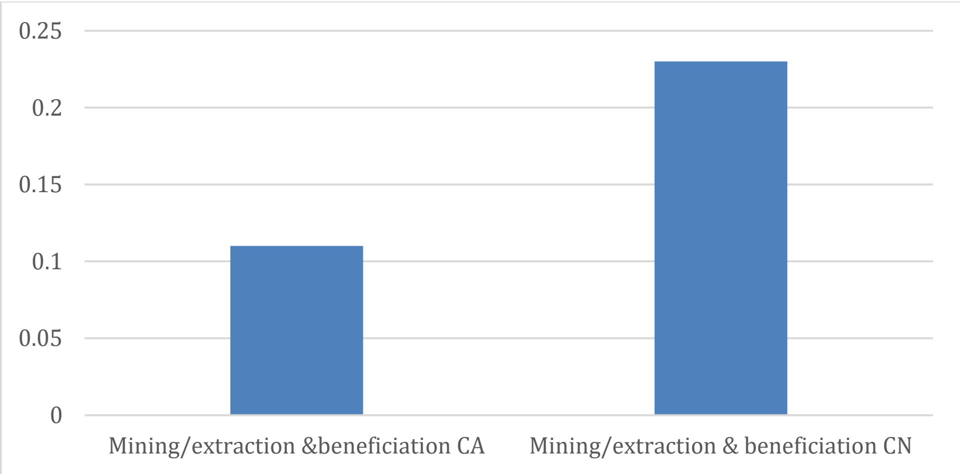
**Figure 3.15. Impact on climate change of the refining stage on a global average. FU: 1kg of nickel concentrate.**




**Source**: Ecoinvent data, 2023

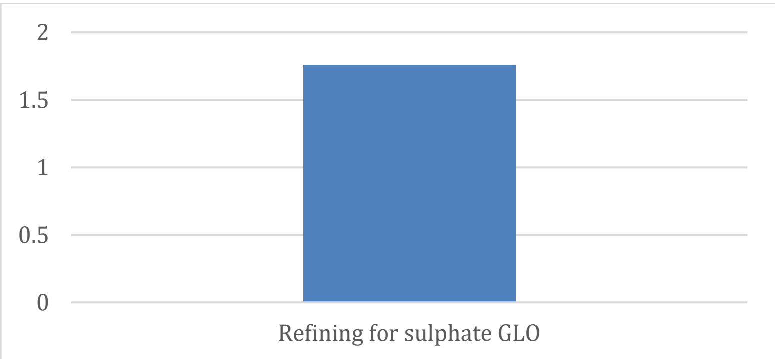
A similar pattern can be observed when comparing non-renewable energy consumption across various countries and stages of the value chain (Figures 3.16 and 3.17). In terms of value chain stages, the energy resources needed for mining and beneficiation are significantly lower compared to the refining process for nickel sulphate production. Moreover, a comparison between the two countries reveals that Canada has a more energy-efficient mining and beneficiation process compared to China. The use of non-renewable energy resources would be reduced by 52 % in mines in Canada in comparison with those in China.

**Figure 3.16. Comparison of the non-renewable energy consumption for the mining and beneficiation stage in Canada (AC) and China (CN). FU: 1kg of nickel concentrate**




**Source**: Ecoinvent data, 2023

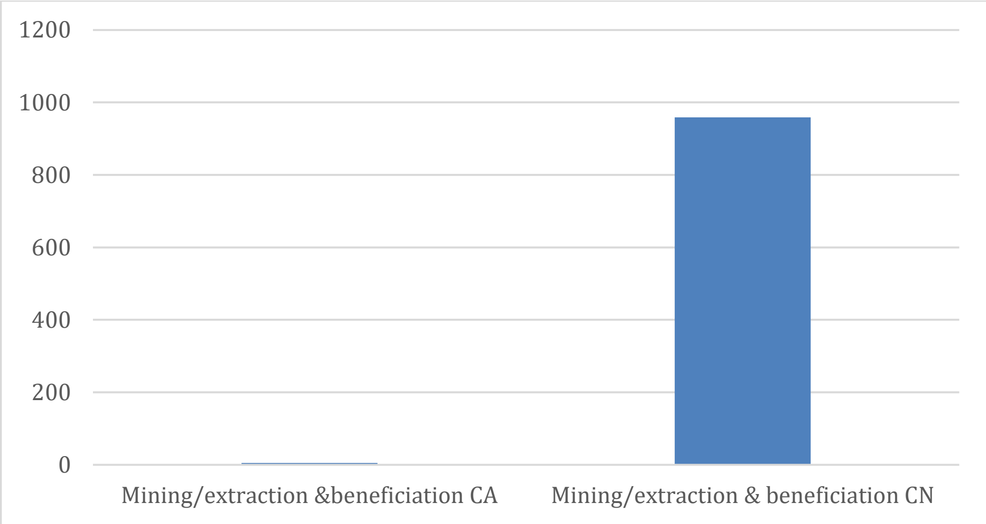
**Figure 3.17. Non-renewable energy consumption for the refining stage on a global average. FU: 1kg of nickel concentrate.**




**Source**: Ecoinvent data, 2023

Another environmental impact studied here is the ecotoxicity to water (fresh, marine, terrestrial) at different stages and different regions. As shown in Figure 3.18, mining and beneficiation of nickel in China is causing huge ecotoxicity to water when compared with Canada. The impact is estimated to be 99% smaller in Canada compared with China. This is even higher than the ecotoxicity impact caused by the refining process (calculated based on the average global data).

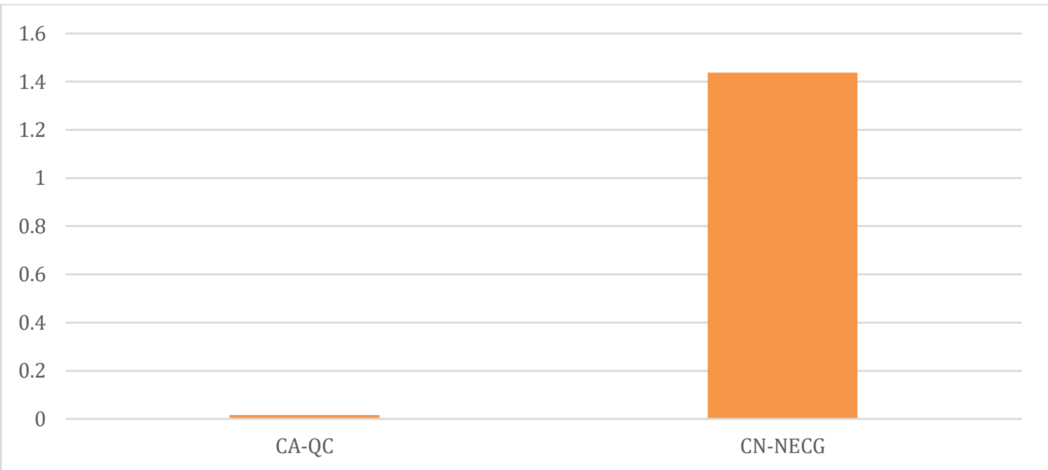
**Figure 3.18. Impact in terms of ecotoxicity to all types of water for the mining and beneficiation stage (comparison between Canada and China). FU: 1kg of nickel concentrate**




**Source**: Ecoinvent data, 2023.

Along the production chain of nickel, the consumption of electricity is significant. It has been estimated that China's nickel production consumes between 2.6 % and 6.9 % of the total electricity consumption in the country (Guohua et al., 2021). As such, for the purpose of the current assessment, the impact of the electricity grid in China and Canada are compared and presented in Figure 3.19.

**Figure 3.19. Comparison of the impact on climate change for the production of 1kWh of electricity in Canada (Quebec, QC) versus in China (North-East-Chinese-Grid, NECG).**




**Source**: Ecoinvent data, 2023.

The electricity mix data was taken from the following Ecoinvent data:

- For Canada: electricity, high voltage, production mix | electricity, high voltage | APOS, S – CA-QC
- For China: electricity, high voltage, production mix | electricity, high voltage | APOS, S – CN-NECG

As mentioned in Sub-section 2.6, there are limitations in the methodology used in this study combined with the lack of comparable data for performing an in-depth environmental impact assessment. Taking these limitations into consideration, the analysis reveals that if as a short-term reaction, the EU sources nickel from Canada instead of China, this could lead to reduced environmental impacts.

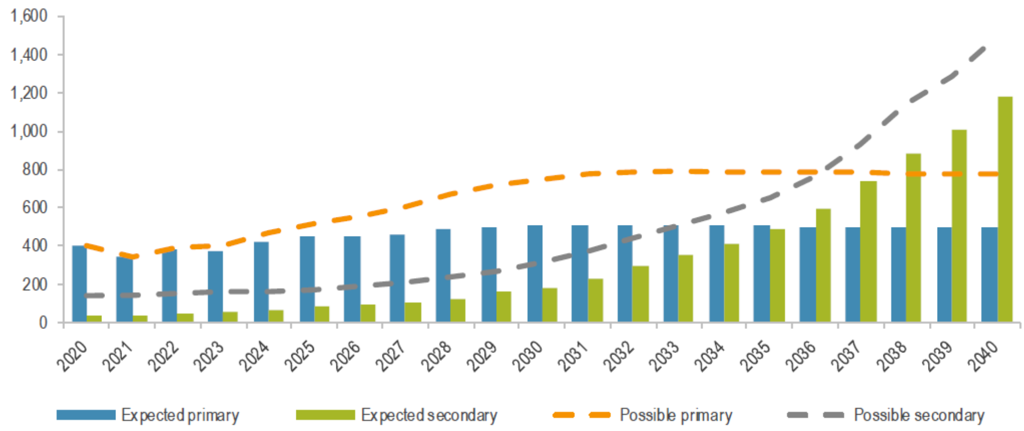
Considering the current EU supply mix for mined and refined nickel Class I (battery grade), it is estimated that 2,383 tonnes of refined nickel Class I imported in the EU is mined in China. Although this represents only 1.3 % of the total mined nickel Class I used in the EU, a total shift of sourcing of this amount from China to Canada has the potential of reducing the GWP by 1,000 t $CO_2$-Eq. A shift to mines in Canada would also represent a reduction of consumption of close to 300 tonnes oil-Eq. An additional big impact avoided would be on the ecotoxicity to water which would avoid the emissions of 2,273,382 tonnes 1.4 DCB-Eq. For a 50 % shift from China to Canada, the estimated potential of reducing GWP is of 500 t$CO_2$-Eq, energy resource of 142 tonnes oil-Eq and ecotoxicity to water of 1,136,214 tonnes 1.4DCB-Eq. For a 25 % shift from China to Canada, the estimated reduction in GWP is of 250 t $CO_2$-Eq, in energy resources of 71 tonnes oil-Eq and in ecotoxicity to water of 567,630 tonnes 1.4 DCB-Eq.

**Medium term (2025-2030) – Improving nickel recycling from end-of-life batteries**

The assumptions made for this chapter are detailed in chapter 2.5 (Analytical assumptions).

As predicted in many studies (Abdelbaky et al., 2021; Fraser et al., 2021; IEA, 2021) and shown in Figure 3.20, secondary sources of nickel (recycled from end-of-life batteries) is expected to grow significantly at global level in the coming years.

**Figure 3.20. Outlook for feedstock availability, primary versus secondary sources, 2020-2040, kt**

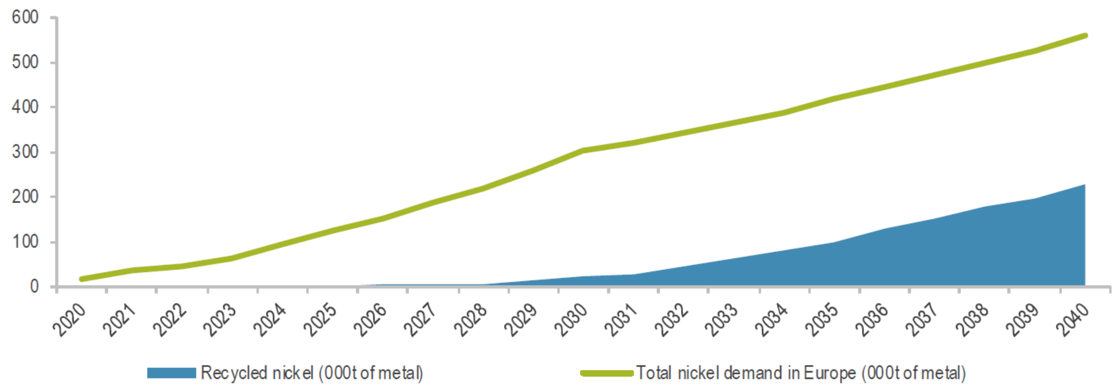



**Source:** Fraser et al., 2021

More specifically, after 2030, it is predicted that the secondary feedstocks will become increasingly important for non-integrated sulphate producers, where majority of this secondary nickel feedstocks will be sourced from EOL battery recycling (Fraser et al., 2021; IEA, 2021).

Figure 3.21 below presents the availability of nickel from battery recycling in the EU compared with the nickel demand in this region (Fraser et al., 2021). Improving the battery recycling industry in the EU can be a solution to meet the demand in long term, although this would need a significant increase in recycled material as the overall demand will increase steadily.

**Figure 3.21. Available nickel from EOL electric vehicle (EV) battery recycling in EU-27 compared with the nickel demand in Europe**




**Source:** Fraser et al., 2021

The share of annual electric vehicles sales in the EU is forecasted to reach 23 % of global sales by 2030 (Abdelbaky et al., 2021; IEA, 2021) and this will automatically lead to an increase in availability of end-of-life batteries from these vehicles in the EU. However, the availability of EOL batteries for recycling depends on various factors, such as battery useful life, reuse, collection rates, recycling rates and chemistry (Fraser et al., 2021). For example, an increase in reuse of batteries or prolonging their useful life could reduce or delay their availability for recycling processes. While reuse and extension of product lifetimes are preferred options from a circular economy perspective, to improve the recycling capacity in the EU and establish an economically feasible and sustainable recycling industry, it is crucial to ensure the availability of a critical mass of EOL batteries for the industry at the point in the lifecycle where batteries can no longer serve their original purpose (Fraser et al., 2021).

As mentioned in the previous sub-chapter, the recovery of secondary nickel from battery recycling has in general less environmental impacts than the impacts related to battery manufacturing using virgin materials (Dewulf et al., 2010; Dunn et al., 2015; Silvestri et al., 2020). However, based on the LCA study conducted by Silvestri et al (2020), the global warming potential (GWP) of battery recycling is 22 % higher than battery production, due to higher demand for energy in recycling processes. As such, any advancements in the battery recycling industry in the EU should consider the use of renewable energy sources or implementing energy-efficient technologies in the recycling processes to reduce the GHG emissions to a lower level than those emitted during the primary production. Considering the revised EU Battery Regulation which sets new targets (i.e. recycling efficiency target for nickel-cadmium batteries of 80 % by 2025), it is expected that the European battery recycling sector will go towards less emissions and a high environmental benefit per recycled batteries compared to their primary production (European Council, 2022).

The environmental impacts of battery recycling remain discussed and uncertain as other findings, such as a report commissioned by the Finnish company Recser Oy (producer organization active in the recycling of batteries in Finland), show that the production of recycled metals present in batteries can reduce between 50 % and 98 % the GHG emission in comparison with the use of primary metals resulting from mining. The research based on the findings of Van der Voet et al., 2018, show that the GHG emissions per tonne of nickel drop from 22,300 kg $CO_2$-Eq to 513 kg $CO_2$-Eq between primary production and secondary production.

Assuming that the European battery recycling sector will indeed follow a path of decreased emission for those activities and taking the number from the industry (study mandated by Recser Oy) of 50 % reduction of GHG, and based on the forecast that in 2030 10 % of the nickel demand for batteries in the EU could be covered by recycling, this would represent a 5% reduction of GHG emissions (estimated 334,000 tonnes of $CO_2$-Eq).

Additionally, it should be noted that some countries (e.g. USA) have planned to develop research in substitution for nickel in batteries which could result in batteries containing less nickel in the future (FCAB, 2021).

**Long term (2030-2040) – Improving domestic refining processes**

The assumptions made for this chapter are detailed in chapter 2.5 (Analytical assumptions).

Currently there are no advanced exploration projects or indication of new capacity in the medium to long-term within the EU-27. As such the share of EU-27 in global mine supply is forecasted to decline by 2030 and 2040 (Fraser et al., 2021). Even if there were any potential deposits, nickel mines have a long lead time at 17.5 years, when compared with other commodities[11]. Owing to these challenges, other alternative strategies should be considered to secure the supply in the long term.

As previously indicated, the demand for nickel products will significantly increase for battery production. Since the EU may not be able to improve its domestic primary mine production, a potential strategy could be to improve its refining capacity for production of class I nickel and nickel sulphate needed for battery production (Fraser et al., 2021).

Currently, the majority of refined nickel production is class I nickel (85 % vs 15 % nickel sulphate). Due to the lack of domestic primary mine products needed for production of class I nickel, there are ongoing plans in the EU to improve its nickel sulphate production. As such it is expected that by 2040, the share of nickel sulphate production to class I production will increase to 44 % (Fraser et al., 2021). The expansion plans for Terrafame nickel sulphate plant in Finland is one of the main reasons for this increase (Fraser et al., 2021)

The Terrafame company claims that based on an externally verified life-cycle analysis the carbon footprint of their nickel sulphate production process is 60 % lower than the industry average mainly due to using bioleaching method, where microbes separate the metal from the ore. For this method, which may take about five years, there is no need for crushing or grinding of the ore (as fine as in the traditional process) and no high temperature metallurgical processes are used in further processing. As such less energy will be consumed than other methods[12]. The company claims to achieve carbon neutrality by 2039[13], although these claims have not been verified independently.

Currently 22 % of the EU supply of refined nickel Class I comes from Finland (Figure 3.9) amounting to 63’000 tonnes. Considering an increase in refining capacity in Finland of 50 % by 2040 (95,000 tonnes), this would mean the share of supply coming from Finland would increase from 22 % to 34 %. Based on the global average impacts for refining for nickel sulphate (highlighted in Figure 3.15 and Figure 3.17) and on the Terrafam company that they will have achieved carbon neutrality by 2040, this would translate in a GWP reduction of 15 %.

Additionally, Finland currently sources 66 % of its nickel from domestic mines. The rest being imported from Russia mostly (28 %) and Canada (4 %) (Figure 3.22). The reduction of impacts only looks at an increased refining capacity in Finland but does not include a change in sources of nickel ore. It stands to reason that although increasing domestic refining capacities has the potential to reduce the overall impact

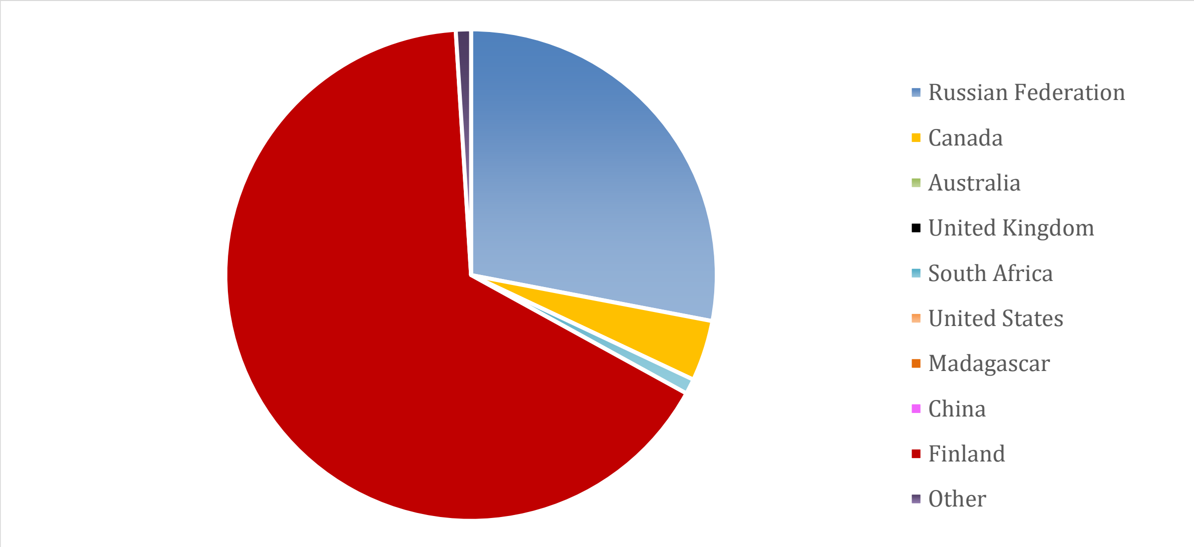
11 https://www.spglobal.com/marketintelligence/en/news-insights/research/discovery-to-production-averages-15-7-years-for-127-mines

12 https://www.terrafame.com/offering/battery-chemicals.html

13 Terrafame Sustainability Review 2022: https://issuu.com/terrafame/docs/terrafame_sustainability_review_2022-web

of the value chain, the sourcing of primary nickel should be done from countries with the least impactful mining sector as this will also affect the overall impact of the nickel value chain.

**Figure 3.22. Origin of the mined nickel ore that is refined in Finland. Average between 2018-2022.**




**Source**: Joint Research Center, 2023.

**Summary of findings – Nickel Case Study**

| Scenario | Short-term (2023-2025) | Medium-term (2025-2030) | Long-term (2030-2040) |
|---|---|---|---|
| GWP (kg $CO_2$-Eq) | Between **18 %** and **45 %** potential reduction depending on the level of supply shift from China to Canada (from 25 % to full shift). | Potential of **5 %** reduction by 2030 considering 10 % recycling input rate. | Potential of **15 %** reduction with an increased refining capacity of 50 % in Finland compared to current status. |
| Energy Resources (kg oil-Eq) | Between **13 %** and **52 %** potential reduction depending on the level of supply shift from China to Canada (from 25 % to full shift). | | |
| Ecotoxicity to water (kg 1.4 DCB-Eq) | Between **24 %** and **99 %** potential reduction depending on the level of supply shift from China to Canada (from 25 % to full shift). | | |

# 4 Case study 2: Rare Earth Elements (REE) supply chain disruptions

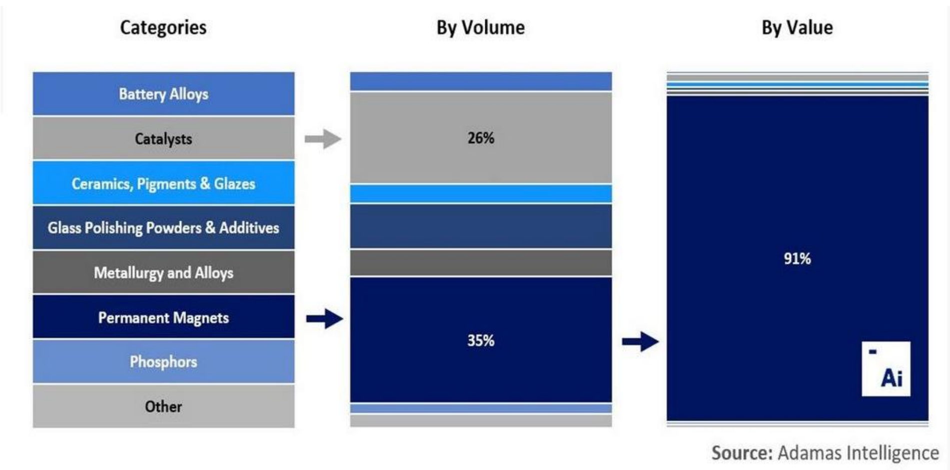
## 4.1 REE properties and applications

Rare Earth Elements (REE) are a series of 17 elements found on the periodic table. These include the 15 elements from atomic number 57-71 in the lanthanide series, from lanthanum (La) to lutetium (Lu) plus two additional elements of scandium (Sc) and yttrium (Y), which are included given their similar elemental behaviours. They have a distinctive chemistry that is defined by their electronic configuration and their "lanthanide contraction" where the atomic radii decrease with increasing atomic number. The group is often categorized into light rare earth elements (LREE) from $^{57}$La to $^{63}$EU, and heavy rare earth elements (HREE), from $^{64}$Gd to $^{71}$Lu. Despite their similarities, there are some notable behaviour characteristics that make each REE distinctive.

The REEs have unique chemical, magnetic, catalytic, and luminescent properties that have enabled their increasing incorporation into many green energy applications. In fact, given this explosion of utilization in technology, these elements have been dubbed 'the vitamins of modern industry' (Balaram, 2019). Some of the most important usages of REEs are in renewable energy components such as in electric vehicles (EVs), solar cells, wind turbines and rechargeable batteries. For example, solar cells utilize the unique electronic configuration of REEs and their ability to convert light photons into different wave lengths, thus allowing for more light absorption of the cell (Dushyantha et al., 2020). Other usages for REEs are in the field of optics, lasers, medical applications and X-ray generation and as alloys in aerospace engineering.

One important usage of REE is in the field of magnets. Powerful REE permanent magnets, composed mostly of neodymium (Nd) and the HREE dysprosium (Dy), are used in 'direct drive' wind turbines and in the majority of the electric car drive motors (Tazi et al., 2023). These technologies use Nd-Iron (Fe)-Boron (B) magnets, with small admixtures of praseodymium (Pr), and the HREEs gadolinium (Gd), terbium (Tb) and especially Dy to provide an increased reliability and require less maintenance as compared to those turbines using non-REE magnets (Alves Dias et al., 2020; Dushyantha et al., 2020). The HREEs and in particular Dy provide crucial stability at higher operating temperatures to prevent from demagnetization (Binnemans et al., 2013). REE magnets are also found in many computers hard disk drives. In fact, permanent magnets constitute not only the majority of REE usage (Figure 4.1) but also are collectively responsible for over 90% of the total value of global REE consumption in 2022 (REIA, 2023). Therefore, Nd and Dy are increasingly becoming the two most important REEs. The majority of the following discussion will thus be focused on REE for usage in magnets, given their current importance in this sector.

**Figure 4.1. The 2022 distribution of the various categories and sectors where they are utilized and by both volume and value**



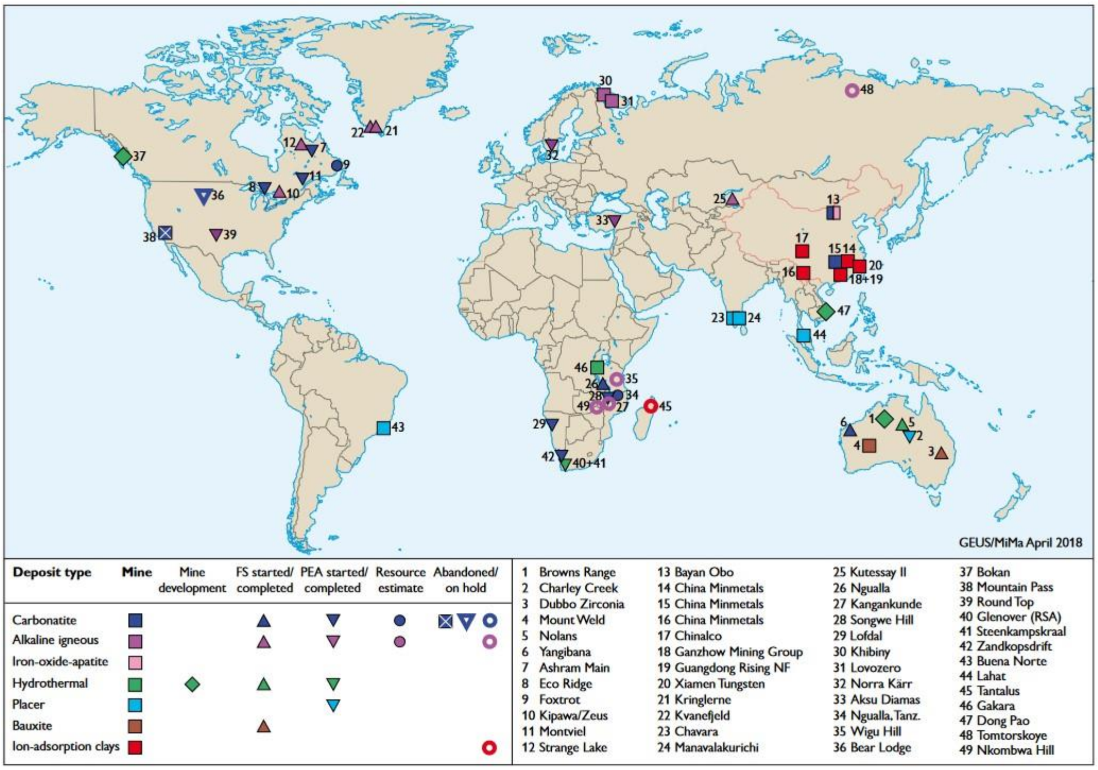

**Source:** REIA, 2023

REE magnets are among the most critical materials in the EU (EC 2023). These magnets and motors play a crucial role in various sectors, such as automotive, electronics, and renewable energy. The European Raw Materials Alliance recognises them as the most critical value chain for many EU industrial ecosystems; therefore, there is a need to strengthen the domestic production and supply of REE magnets and motors to reduce dependency on imports and ensure the competitiveness of EU industries.

Despite their name, REEs are not as rare in the earth's crust as compared with other elements utilized in modern applications. For example, cerium (Ce), La, Y and Nd have a higher abundance in the earth's crust than some other well know critical raw materials (CRM) such as cobalt (Co) and tin (Sn) as well as even copper (Cu), all minerals important for current green and digital technologies (Jaireth et al., 2014; Weng et al., 2013). They are only rarely found in mineable concentrations high enough to be of economic value and or found as a native element, instead incorporated into the crystal structure of other minerals such as oxides, carbonates, phosphates, silicates and halides (Dushyantha et al., 2020; Gupta & Krishnamurthy, 2005).

Given the occurrence of REE in a wide variety of geologic environments, the ore deposits are divided into primary and secondary ore types. The primary REE ores are those formed by magmatic, metamorphic or hydrothermal processes and include some mineral deposits such as bastnäsite, monazite, eudialyte and xenotime. Bastnäsite is also the most common REE bearing mineral found in the important carbonatite type deposit. Secondary deposits are those formed from processes like erosion and weathering of primary ores (Balaram, 2019) and include ion adsorption clays (IAC), laterites and placer deposits.

**Figure 4.2. Map showing the distribution of REE mines or known project by type of geologic deposit.**




**Note:** The newly discovered REE deposit at Kiruna Mine in northern Sweden and numerous exploration projects in Canada not shown

**Source:** Kalvig & Machacek, 2018

Globally, the most important REE reserves are found in bastnäsite, monazite, xenotime and REE bearing IACs. Figure 4.2 shows the global distribution of known REE mines or deposits and by geological host rock type. The majority of the REEs are found in carbonatites or deposits bearing the mineral monazite which are found in Australia, Brazil, China, and the US among other locations (Dushyantha et al., 2020). Ion adsorption clays are found mainly in southern China in addition to other SE Asian deposits for example in Myanmar, Thailand and the Philippines (Sanematsu et al., 2013; Wu et al., 2023). While all REEs are found in the same ore type, generally, LREEs tend to concentrate more in the bastnäsite bearing carbonatites or monazite bearing host rocks. Globally, monazite ores can be more problematic, as trace amounts of radioactive thorium (Th) and uranium (U) are found together with the REEs. The HREEs tend to concentrate more in the IAC deposits, with more than 90% of HREEs coming from IACs, with China supplying the majority of HREEs (Van Gosen et al., 2017; Wu et al., 2023). The HREEs are also of higher value as they are generally less abundant. They also do not contain the radioactive elements associated with other REE deposit types. The mineralogy of these ore deposits varies greatly, which can have a significant impact on the ease of the processing chain.

## 4.2 REE supply chain

**REE Supply Chain**

Given the current and growing importance of magnets, the value chain is divided into mining of the ores, the processing steps such as beneficiation, cracking or leaching, refining, transportation, logistics and finally recycling and will focus more on REE magnets. The following section describes the most environmentally impactful steps of the permanent magnet value chain.

**Mining**

The REE supply chain begins with mining the ore. Of the numerous global deposits, there are a handful that are either currently sourcing REEs or are large deposits in the development stages towards exploitation. Particularly important are the deposits found in China, the USA, Australia, and Sweden. These notable deposits are described below.

Bayan Obo China is considered the world's largest REE deposit. Located in inner Mongolia, the host rock consists of mainly numerous bastnäsite and monazite bearing carbonatite dykes. Bayan Obo reserves are estimated at over 57.4 million tonnes with estimated Nb reserves of over 2 million tonnes (Dushyantha et al., 2020). The ore is mined at Bayan Obo via open pit mining. This deposit is particularly enriched in LREEs (Dushyantha et al., 2020).

Mountain Pass, CA, USA is considered the second largest REE global deposit. It consists of mostly bastnäsite and minor amounts of monazite minerals hosted in carbonatites. While this deposit has been mined since the 1950s, it is still estimated to contain over 18 million tonnes of reserves (Burron, 2023). This deposit tends to be enriched in LREEs relative to HREEs and the mine operators focus on producing high purity Nd and Pr oxides (Burron, 2023). The ore is mined via open-pit and is processed and refined on-site (NS Energy Business, 2023). The mined ore now undergoes beneficiation and processing at a facility recently opened near the mine.

Another important deposit is found in Australia at Mount Weld. This deposit hosts both a carbonatite complex sitting below a lateritic deposit. The majority of the REE are found in this overlying deposit (Jaireth et al., 2014). The primary ore mineral in both the carbonatite and the laterite consists of mainly monazite (Lottermoser, 1990). Mt Weld is host to approximately 19.7 million tonnes of known REE reserves from its central lanthanide deposit (CLD), in addition to other deposits (Burron, 2022). It is reported that the CLD has one of the highest grade REE ores of known global deposits (Dushyantha et al., 2020). This deposit is enriched in LREE found mostly in monazite with lesser amounts of HREE (Voncken, 2016). Today only the CLD is being mined using conventional open pit operations and the ore is sent to the Lynas Advanced Materials Processing (LAMP) plant in Malaysia (Haque et al., 2014).

Found predominantly in southern China is a very important and unique deposit of REE that supplies the majority of the global HREE. Here the REEs are hosted on an ion adsorption clay (IAC) laterite and are formed from weathering of igneous rocks around the tropics in conditions of high humidity and temperatures (Borst et al., 2020; Dushyantha et al., 2020). While the grade of ore in the IACs of southern China are low (Deng & Kendall, 2019), the REEs are weakly sorbed onto the clay minerals making their extraction via heap- or an *in-situ* leach mining process relatively easy (Deng & Kendall, 2019). These IAC deposits are particularly concentrated in HREE, specifically Dy (Ling & Yang, 2014).

In the last several years, Myanmar has grown to become one of the most important REE producing countries. In the mountainous corner of the country in the state of Kachin on the border with China there are IAC deposits enriched in HREE. The mining in this region is however poorly understood as it is said that since the 2021 military coups, extraction is controlled by militias. What is known is that the ores are artisanally mined using low technology and in situ leaching, where solutions of ammonium sulphate are injected into holes drilled directly into the deposit. The HREEs are incorporated into the solution, which is then collected into surface ponds and the HREEs are precipitated out of solution (Global Witness, 2022). Given China's intensified efforts to clean up the environment around its own HREE mines, it is reported that HREEs from the Kachin region are exported directly to China, though the exact amount is unclear (Global Witness, 2022). In 2021, the US Geological Survey estimated Myanmar as the third largest producer of REE (Global Witness, 2022; Sadan et al., 2022; USGS, 2023).

In the last decade there have been discoveries of REE deposits in Sweden that have the potential to become an important global source in the coming years. A particularly well studied deposit is that of the Norra Kärr region of southern Sweden, which contains a large deposit of REE found primarily in the eudialyte-mineral group as well as some other REE-bearing minerals (Schreiber et al., 2016; Sjöqvist et al., 2013). The deposit is estimated to have probable mineral reserves of approximately 23.5 million tonnes (Goodenough et al., 2016). These eudialyte REE deposits also have low concentrations of radioactive elements, giving a potential advantage to the utilization of such deposits (Schreiber et al., 2016). The primary REEs found in the Norra Kärr deposit are La, cerium (Ce), Nd as well as the important HREEs (Voncken, 2016). Though the deposit is not currently being mined, the prefeasibility study conducted by the developer described the plan to extract the resources via open pit methods (Short et al., 2015).

At the beginning of 2023 it was announced that a large new deposit of REE had been discovered in northern Sweden above the artic circle, near the existing underground Kiruna iron ore mine (LKAB, 2023). The iron ore body consists of a massive ore body hosted in altered metavolcanics (Westhues et al., 2017) and these so-called Per Geijer REE deposits occur together with the mineral apatite (LKAB, 2023). The deposit is currently estimated to contain about 1.3 million tonnes of REEs (Johnson, 2023), and contains both in LREEs and HREEs, though is more enriched in the previous. The LKAB, Sweden's state owned mining company, is the principle owner of the mineral rights and in November of 2022 partnered with a Norwegian rare earth extraction technology firm REETec, who will utilize a high efficiency, lower $CO_2$ emission, low energy processing technology to extract the REEs (LKAB, 2022). Their processing methodology reportedly recycles all process inputs. REETec will build its first processing facility in Herøya, Norway, to be operational by the second half of 2024 and has plans to build a second plant to be operational by 2026 (LKAB, 2022). The facilities are aiming to help break the global dependence on China for REE processing. The LKAB is currently in the permitting process for mining of the Per Geijer REE deposits near Kiruna; however, if the deposit can be deemed a 'strategic project' by the EU under the proposed new Critical Raw Materials Act, this complicated permitting process could be expedited (Johnson, 2023).

Finally, Canada is also increasing its investment into its own domestic REE production capacity. While it is currently not mining REE, it has numerous exploration projects and is reportedly home to some of the largest REE reserves and resources worldwide (Government of Canada, 2023). One strategically important project is the Nechalacho deposit in the Northwest Territories, which is enriched in both LREE and the crucial HREEs in multiple zones of mineralization. This deposit is expected to be mined via open pit and underground operations with initial beneficiation and hydrometallurgical refining on-site (Micon International Limited, 2013). The Nechalacho project has reportedly begun mining in 2021, however, it is unclear if these initial extractions are focused only on LREE or also HREE.

**Processing and Refining**

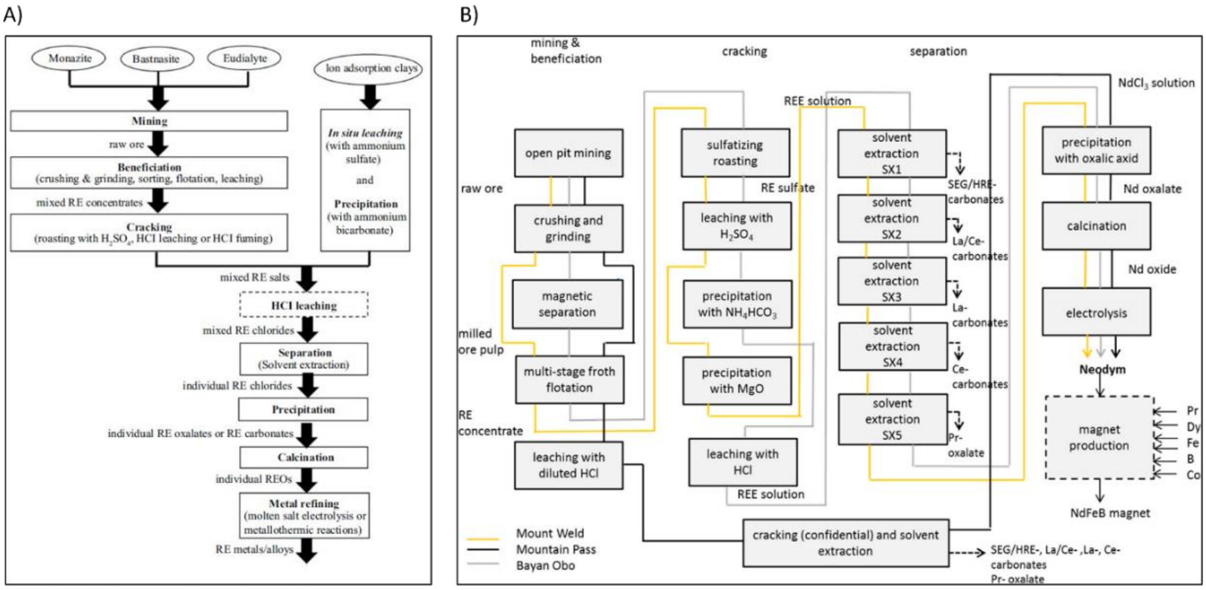
The usage of REEs in technology often means that the purity of the individual elements is very important. Given the mineralogy of ore deposits, their extraction from the respective crystal structure and host rock followed by the processing and refining into the individual elements is often deposit specific, extremely complex and utilizes large volumes of chemicals for the extraction and separation of the individual REEs.

Figure 4.3 A) below displays simplified extraction procedures for the primary and secondary REE ores (Zapp et al., 2022). While the bastnäsite, monazite, eudialyte differ in mineralogy and host rock, as can be seen their extraction and refining process is similar. While the IAC deposits undergo a different initial process of in situ extraction from the host rock, their additional extraction, separation, and refining steps are similar to that of the other minerals. Figure 4.3 B) below shows the more complex Nd extraction routes and the various chemical reagents used for the major deposits found at Mount Weld, Australia, Mountain Pass, California, USA, and Bayan Obo, China (Marx et al., 2018).

**Ores of monazite, bastnäsite, eudialyte**

- **Ore preparation and beneficiation** – otherwise known as concentration, beneficiation is a process which separates the ore from minerals of no value. The processes employed during beneficiation are displayed in Figure 4.3 A) (Jordens et al., 2013; USDOE, 2020). Considerable costs are associated with the beneficiation of these ores, as a large volume of rock is processed, and it consumes large amounts of chemical reagents and energy.
- **Cracking (roasting with acids**) – Following beneficiation, the cracking process converts the REE concentrates into soluble REE salts, allowing for easier subsequent separation steps. The specific chemical reagents used depend on the original mineralogy of the REE ore, but the two main procedures are either roasting with sulfuric acid (H2SO4) or roasting with hydrochloric acid (HCl). During the acid roasting of bastnäsite with H2SO4 the highly toxic hydrogen fluoride (HF) gas is produced, which has to be first removed during ventilation (Zapp et al., 2022). To avoid HF production, bastnäsite ores can also be roasted using HCl (Zapp et al., 2022). The exact cracking process used at Mountain Pass, CA is subject to confidentiality (Zapp et al., 2022). Overall, cracking uses a high volume of chemicals and is energy intensive.
- **Leaching** – After the cracking process, additional minerals or other secondary elements (such as radioactive Th and U) are separated from different solutions via a process of leaching and precipitation, which uses again large quantities of chemicals. The subsequent sludge waste produced often has varying levels of radioactivity which can be a problem for waste disposal (Zapp et al., 2022). This process generates significant negative impacts to the environment via pollution and waste production.
- **Solvent extraction, precipitation, calcination** – Until this point the REEs have not been separated into the respective individual elements. Their separation is very complex, given the similar chemical properties of the elements. The most common method is using solvents for their separation followed by their subsequent precipitation of the separated element into intermediate compounds which then undergo a process of calcination using an oven or kiln (Zapp et al., 2022). These processes again use high volumes of chemicals, both organic and inorganic reagents, and are energy intensive.
- **Metal refining** – Finally the refining of the ores into individual REEs uses an electrolysis process. It uses a high volume of chemicals such as various electrolytes and is energy intensive (Zapp et al., 2022)

**Figure 4.3. A) The simplified comparison of the process chain of REEs from the primary ore types, and B) The detailed process route for the Nd ores from Mount Weld, Australia, Mountain Pass, California, USA, and Bayan Obo, inner Mongolia, China**




**Note:** In A, dashed lines represent optional steps in the processing chain
**Source:** A) Modified after Zapp et al., 2022, B) Modified after Marx et al., 2018

**Steps: ion adsorption clay deposits:**

**In situ leaching**

As previously mentioned, IACs undergo a process of in situ leaching to be extracted from the clays and requires no beneficiation. The leaching is followed by the precipitation using ammonium bicarbonate (Zapp et al., 2022), which is then subjected to a similar separation and refining process as the bastnäsite, monazite or eudialyte ores. Thus overall, the processing and refining step of IAC ores uses a high volume of chemical reagents and is also energy intensive.

**Recycling**

Finally, separate to the process for virgin ore refining is the reutilization of end of life (EoL) spent REE products such as existing magnets. However, globally the recycling of EoL REE products is in its infancy. Currently it is estimated that the recycling rate for REEs is less than 1 % globally (Eggert et al., 2016). One of the main hindrances is the availability of recycling input material, though EoL magnets supply is expected to increase in the coming years with EV battery and wind turbine magnets reaching their EoL (Rizos et al., 2022). Given the mounting demand for REEs and concerns for the future availability of virgin materials, these gaps in recycling capacity are just beginning to be addressed. Only recently have the REEs in phosphors and batteries started to be recycled (Binnemans & Jones, 2015; ERECON, 2015).

**Recycling Methodology**

There are many challenges with recycling of REE bearing waste that have hindered the development of this sector. To reach the EU's recycling targets (Critical Raw Materials Act) detailed knowledge of the waste input is first required. Regarding the recycling of REE magnets, the specific type of magnet, the exact composition of REEs present and the presence of any material coating the outer surface is necessary for proper processing (Rizos et al., 2022). Once this is known, there are then a few main processes for their recycling.

**Conventional hydrometallurgical techniques** – also known as 'wet' recycling, involves dissolution, chemical leaching and precipitation (Binnemans et al., 2013; Rizos et al., 2022), which requires a large amount of both water and chemical reagents and generates a large volume of liquid wastes that require treatment (Binnemans et al., 2013; Rizos et al., 2022).

**Pyrometallurgical recycling** – also known as 'dry' extraction, this aims to reduce the amount of chemicals used and liquid wastes produced by involving the remelting of alloyed input material (e.g. magnets with coatings) at high temperatures for further extraction of the REEs (Rasheed et al., 2021; Rizos et al., 2022). This process tends to have higher energy consumption than the conventional hydrometallurgical technique.

**Short Loop Hydrogen Processing (HPMS)** – also known as 'closed' or 'magnet to magnet' recycling, this highly efficient and new technique directly remanufactures EoL NdFeB magnets into new magnets. Developed through the University of Birmingham, UK, the technique reduces permanent magnets to a alloy powder when exposed to hydrogen (HyProMag GmbH, 2022). This process thus allows the mechanical separation of non-REE components without the use of toxic chemicals. (HyProMag GmbH, 2022). After further processing the powder then can be repressed and re-magnetized into new magnets. The process has demostrated to save up to 90 % energy and 98 % less human toxicity compared to virgin production (Sprecher et al., 2014)).

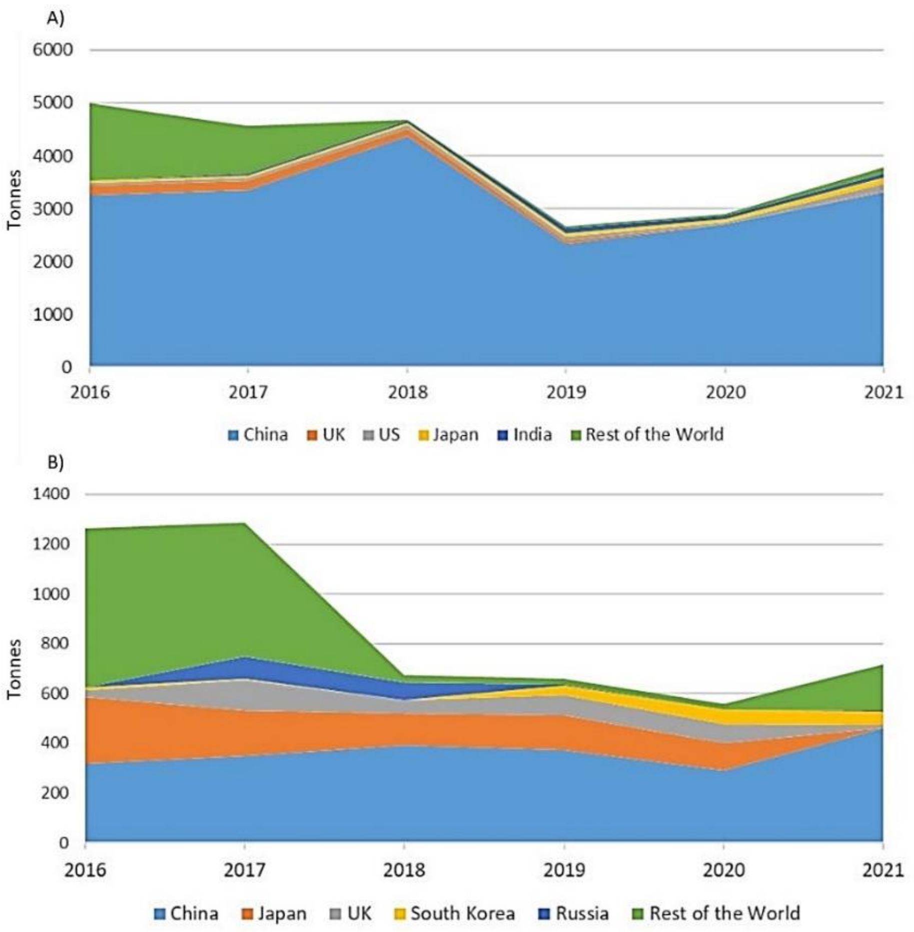
Given the current and future importance of REE, there are now a few projects or commercial companies popping up around the globe that have the potential to be a reliable source of REEs in the near future, should supply chain disruptions occur. Some of these are:

- Two EU funded projects SusMagPro and REEsilience developed the HPMS technology for the efficient recycling of NdFeB magnets. Now licensed to HyProMag, Ltd (August 2023 acquired by Maginito Ltd) start up from University of Brimingham, UK, in the process of scaling up technology, based at Tyseley Energy Park in Birmingham; and Hypromag GmbH (subsidiary) in Pforzheim, Germany. HyProMag technology uses the lowest energy and shortest loop recycling of sintered NdFeB magnets. HyProMag (Maginito Ltd) at Tyseley Energy Park production is targeting late 2023 for first production with HyProMag GmbH in Germany targeting first production in 2024;
- The company GeoMega Resources Inc., based in Montreal, Canada is currently developing a rare earth refining and recycling demonstration plant, with some funding from the Canadian government. The investment in this operation is a part of Canada's net zero by 2050 strategy. GeoMega plans the processing facility to recycle 1.5 tonnes of REE-magnets per day using its Innord subsidiary's ISR technology, an organic solvents free, aqueous chemistry process that separate high purity REEs one at a time, and the chemical reagents are recovered and reused (GeoMega Inc., 2023). The plant is expected to be operational in 24 months (Voloschuk, 2023).

## 4.3 Overview of global and European supplying countries

China dominates the mining production of REE ores at about 70 %, followed by the US at 14 % and Australia at about 6 % (Liu et al., 2023). While China may be the dominate country mining REEs, given the known global deposits, it actually only makes up about 37 % of the global reserves (USGS, 2023). This suggests that there may be options for alternative sourcing countries, should supply chain disruptions occur in the future.

**Figure 4.4. The EU imports of A) LREEs (code CN 28469010) and B) HREEs (code CN 28469020) in tonnes and by country from 2016 to 2021.**

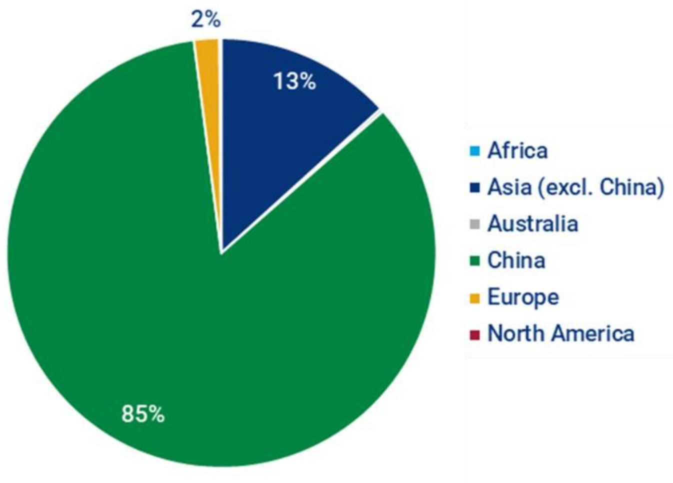



**Source:** Modified after SCRREEN Project, 2023b and based on Eurostat 2022

Overall, given this REE mine production market domination from China has resulted in the EU's heavy reliance on China for both LREEs and HREEs. As can be seen in Figure 4.4 above, the EU imports the majority of its REE supply from China (SCRREEN Project, 2023b). According to these data the import reliance on China for LREEs increased from 2016 to 2021, from a reported 65 % of total imports to 88 % of total imports in 2021, with imports from the USA in a much distant second at 4 % total (SCRREEN Project, 2023b). Similarly, the EU import reliance on China for HREE also increased from 2016 to 2021, from a reported 25 % of total to 64 % of total HREE imports (SCRREEN Project, 2023b).

The global REE processing, separation and refining is similarly complicated, as this is also dominated by China, and demonstrates that should mine production output increase elsewhere, this would not necessarily assure independence from China or a larger supply of some REEs, especially those that will be in high demand in the near future (IEA 2021). Globally, China dominates the processing and refining of REE ores at about 85 % (see Figure 4.5 below). There are four main processing facilities operating outside of China, these are found in Malaysia (the LAMP facility), India (plants operated by IREL (India) Limited) and in Europe with a small processing and refining capacity in France (Solvay in La Rochelle) and Estonia (NMP Silmet, a subsidiary of Neo Performance Materials) (IEA, 2021).

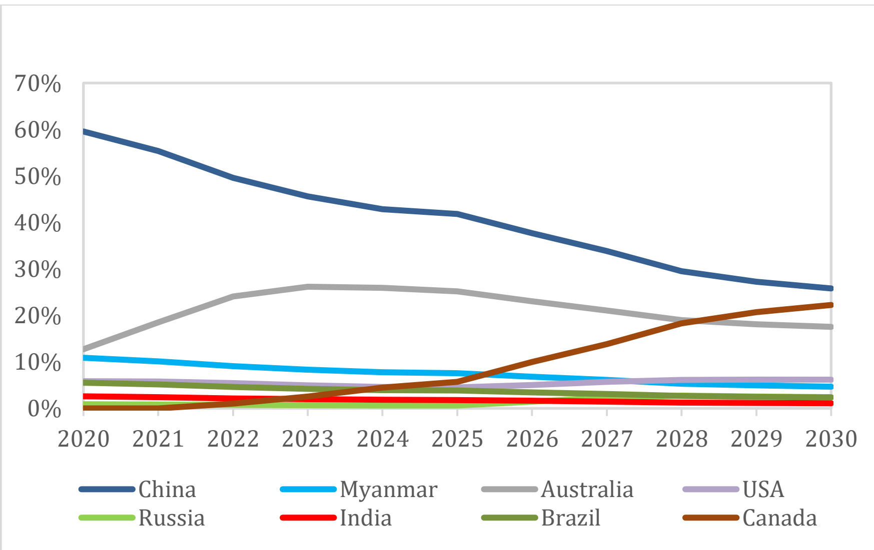
**Figure 4.5. Estimations for percent global refined REE supply by region for 2021**




**Source:** Embleton & Merriman, 2021

With regards to European REE processing capacity, Solvay announced in September 2022 that it is investing heavily in a new rare earths processing plant for the EV and wind turbine market, specifically targeting the production of rare earth oxides for permanent magnets, to its existing facility, though a start date has yet to be announced (Kinch, 2023). The aim is to create a European minerals hub to ease reliance on China. It is reported that Solvay signed a non-binding memorandum of understanding (MoU) together with Western Australia's Hastings Technology to supply carbonatite hosted REEs at 2,500 mt/year for Nd to be used in magnets (Kinch, 2023).

**Figure 4.6. An estimated, short- to medium-term projection of the geographic distribution of REE permanent magnet production rate as a percentage of total global production.**




**Note:** Data is a cumulative sum for Nd, Dy, Pr, Tb and does not include any potential REE production from Greenland.
**Source**: JRC (Joint Research Centre et al., 2020)

The Joint Research Center (JRC) projections for geographical distributions of REE mining production for the short term are displayed in Figure 4.6 above. These estimations were based on nominal levels of output at each operating mine in the respective countries and were based on 2019 production rates. Projections were calculated with an estimated number of mine production years remaining at full production capacity and assuming an estimated average recovery (Joint Research Centre et al., 2020). Developing projects were estimated using anticipated production start-up dates and an assumed ramp-up trajectory to reach full capacity (Joint Research Centre et al., 2020). As can be seen, it is expected that in the coming years there is a diversification of REE sources to the global market. While Chinese output is expected to remain steady, this diversification is projected to reduce their overall market share while those of other countries, such as Canada and to some degree the US, will see an increase (Joint Research Centre et al., 2020).

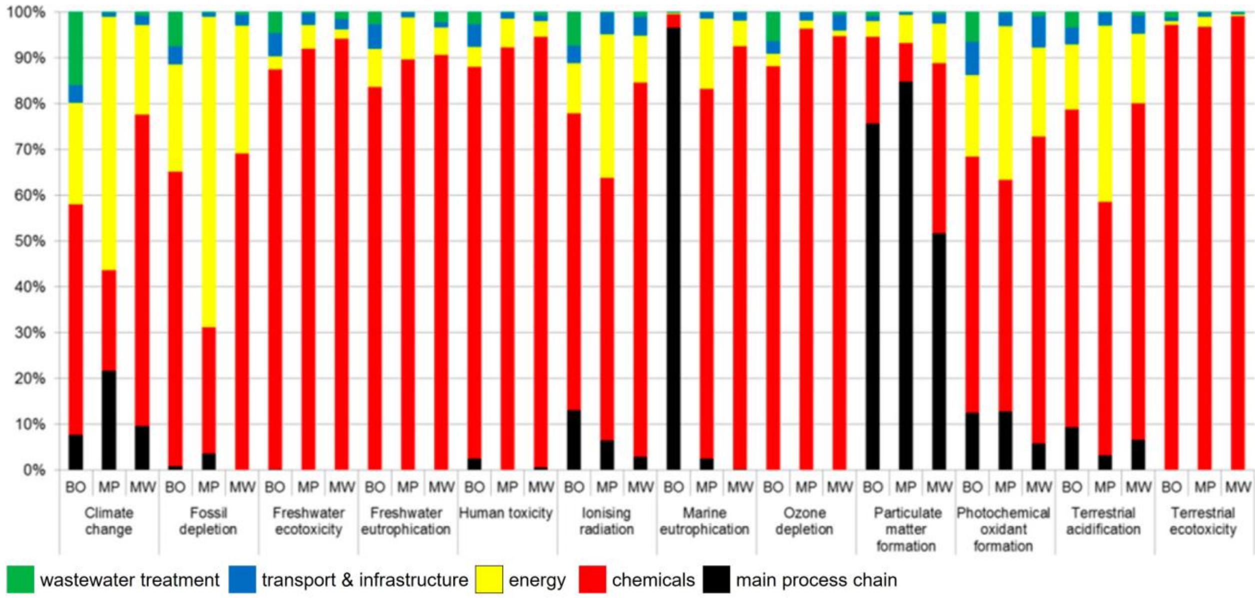
## 4.4 Environmental impacts of REE production

There are numerous environmental impacts associated with the exploitation of various REE resources and their subsequent processing. Because of the variety and complexity of REE extraction, these impacts vary depending on ore mineralogy, method of extraction, local energy production source and the overall environmental protection and regulatory requirements of the countries where the resources are found. Mining activities such as rock blasting, drilling, transportation and processing can release significant amounts of REE- and other toxic or heavy metal-containing dust into the air, polluting surface waters and soils, which could lead to significant impacts to region's people and wildlife (Balaram, 2019). In addition, the land use impacts of the visible scars of open pit mining can often stir-up the local population's fears of environmental contamination and biodiversity loss and lead to project development slow-downs or stalls for detailed environmental impact assessments. For example, the 2016 change in status of the Norra Kärr deposit, as a result of the Swedish supreme court downgrading the mining permit to 'exploration status' as it is near an EU protected, Natura 2000 classified area (Cater & Zimmermann, 2022). There are now environmental impact assessments underway.

There are many known impacts to the environment around REE deposits and with their extraction and processing. For example, elevated levels of REEs have been found in the street dust of an urban industrial city in China (Sun et al., 2017). In addition, it is possible that dust containing radioactive U and Th could come from mines extracting monazite ores. Not to mention that waste from the monazite ore processed at Malaysia's LAMP facility creates significant issues of proper disposal, as the sludge is radioactive (Burton, 2023; Jaireth et al., 2014). In early 2023, the Malaysian government closed the cracking and leaching operations of the plant until July, for LAMP to remove the conditions that lead to high levels of radioactivity in the processing waste (Burton, 2023). This was extended to December 31, 2023, after an appeals process in April. Thus, the status of the cracking and leaching aspect to LAMP's processing operations in 2024 and what effect this might have on the global REE supply chain remains unclear. Furthermore, it has been calculated that not only does REE ore mining and processing currently consume large amounts of energy and water but this consumption is anticipated to increase, in all major sourcing countries analyzed (Golroudbary et al., 2022).

Mining and extracting of HREEs is particularly damaging to the environment in China and Myanmar. Although the HREEs are weakly adsorbed to the IAC deposits and thus allows for easy recovery via in situ or heap leaching, the impact the extraction has on the environment is significant. It is estimated that for 1 tonne of HREE, an estimated 300 $m^2$ of vegetation and topsoil must be removed, leading to significant land use degradation (Yang et al., 2013). This leads to issues of soil compaction, increased runoff and soil erosion as well as an increase in frequency and magnitude of flooding. Furthermore, the injection of the alkaline solution causes an increase in pH of the ground- and surface water, an increase in electrical conductivity and total dissolved solids, as well as sulphate and other water pollutants that cumulatively lead to negative impacts to stream and ecosystem biodiversity (Yang et al., 2013).

**Figure 4.7. The share of total environmental impacts of 1 kg of Nd production from Bayan Obo (BO), Mountain Pass (MP) and Mount Weld (MW) deposit**



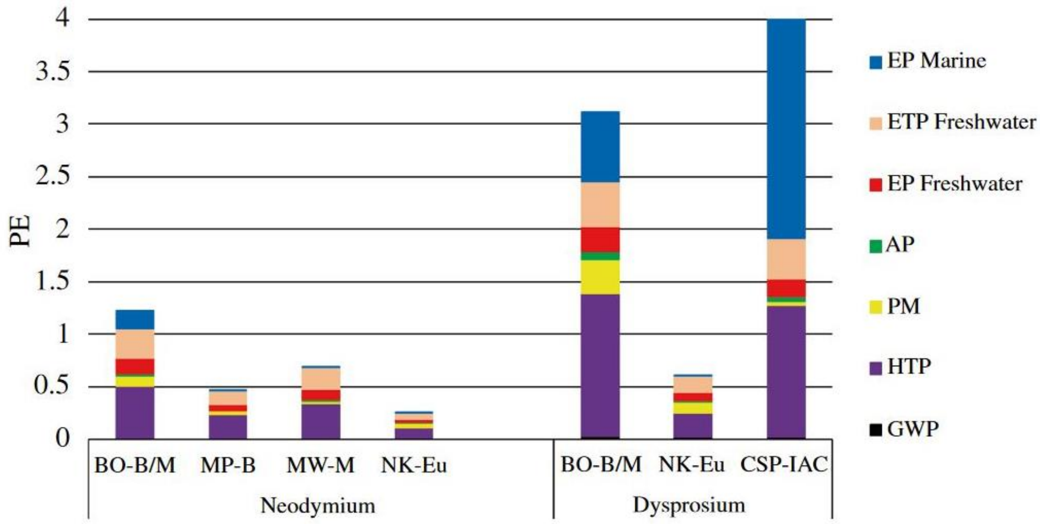

**Source:** Modified after Marx et al. 2018.

Several life cycle assessments and reviews that have been conducted for various aspects of the REE value chain to assess exactly the extent of environmental impacts of REE mining and processing or specifically REE magnet production (Bailey et al., 2020; Deng & Kendall, 2019; Marx et al., 2018; Navarro & Zhao, 2014; Schreiber et al., 2016; Zapp et al., 2022). These studies have all pointed to the processing chain having the overwhelming largest impacts on the environment when it comes to issues of global warming potential (GWP), human, freshwater- and terrestrial ecotoxicity, and freshwater and marine eutrophication to name a few. In a 2018 study, researchers conducted a comparative LCA analysis looking at the differences in the environmental impacts of 1 kg Nd production for magnets from the Bayan Obo (mixed bastnäsite and monazite deposit), Mountain Pass (bastnäsite deposit) and Mount Weld (monazite deposit) mines (Marx et al., 2018). The results of the share of total environmental impacts from the individual steps in the value chain are shown in Figure 4.7. As can be seen, in almost every impact category, the largest environmental impacts come from the chemicals used in the separation of the REEs, followed by the impacts from energy usage and the main (mining) processing chain (Marx et al., 2018).

Similarly, a more recent study compared the overall environmental performance of the generalized REE production pathways from the Bayan Obo, Mountain Pass and Mount Weld deposits LREE (as represented by Nd) and HREEs (as represented by Dy). The results of the comparison are shown in Figure 4.8. The authors found that for all environmental impacts, the REE processing chains from Chinese ores show the poorest performance (Zapp et al., 2022). In general, the Dy ores show higher impacts as HREEs tend to be less abundant. Furthermore, HREEs sourced from IACs have the highest environmental impacts due in part to the process of in situ leaching, which particularly influences the marine eutrophication potential (EP). The results agreed with other previous LCAs on HREEs also showing best performance from the processes associated with eudialyte bearing ores (Zapp et al., 2018). The differences are less pronounced for LREEs when comparing Mountain Pass and Bayan Obo.

Of the currently active mines, Mountain Pass had the best performance in terms of environmental impact of Nd, which was attributed to the various measures currently employed, such as recycling of processing water to reduce chemical usage, cleaner power source, a reported cracking process that does not use roasting (Zapp et al., 2022), and others have also attributed similar good performance of the mine to the environmental measures and legal restrictions in which it operates under (Marx et al., 2018). The overall lowest impacts from Nd processing was displayed from the Norra Kärr eudialyte ores, though since this is not currently an active mine, the results are based on laboratory data (Zapp et al., 2022).

**Figure 4.8. The normalized environmental impacts in person equivalents (PE) of 1 kg Nd and Dy comparing the Bayan Obo bastnäsite/monazite deposit (BO-B/M), Mountain Pass bastnäsite deposit (MP-B), Mount Weld monazite deposit (MW-M), Norra Kärr eudialyte deposit (NK-Eu) and Chinese southern provinces, ion adsorption clays (CSP-IAC).**




**Note:** PEs are normalized data that set each impact relative to the same impact induced by an average person per year. Impacts are - Eutrophication Potential (EP), Ecotoxicity Potential (ETP), Acidification Potential (AP), Particulate Matter (PM), Human Toxicity Potential (HTP), and Global Warming Potential (GWP)

**Source**: Zapp et al., 2022

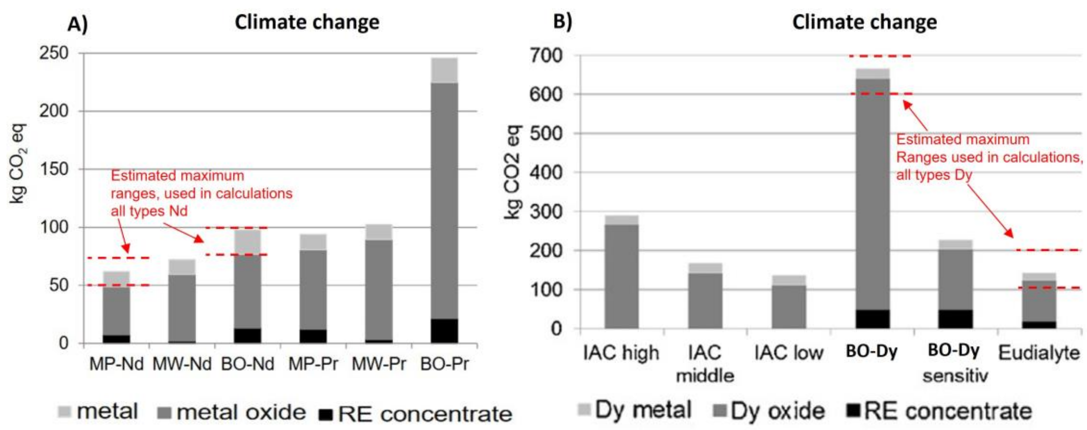
## 4.5 Environmental effects of supply chain disruptions in short-, medium- and long-term

In the case of supply chain disruption, the EU has potential alternatives for sourcing these REEs. The environmental and economic effects are however highly dependent on the time frame of the disruption, for example the promising developments in terms of domestic REE recycling and virgin deposits in Sweden are both possible alternatives found in medium to long term scenarios. A recent analysis of major mines that came online between 2010 and 2019 showed that it took about 16.5 years on average to develop projects from discovery to production, though exact duration varies by commodity and location (Manalo, 2023). In the short term (defined here from 2023 to 2025), it is expected that REEs will face tight supply as demand rises, while in the medium term (defined here as 2025-2030) demand will likely surpass the expected supply from existing mines and projects under construction (IEA 2021). This demand, however, could be supplemented by recycling, should such projects progress in a timely manner and are systematically implemented. Long term scenarios are more promising but are highly dependent on the increased development of domestic mining and processing capacity. These short-, medium-, and long-term scenarios are described in more detail below.

**Short term (2023-2025) – Change to an alternative supplier**

Short term supply chain disruption scenarios are the most critical in terms of economic shock and thus present more challenges for the EU. In this scenario, it is assumed that the EU demand for REE will need to be met by an alternative supplier. Existing country supply projections show diversification in virgin resources in the short and medium term (see Figure 4.6). However, in terms of the environmental impacts of such a disruption, an immediate switch to an alternate supplier has the potential to have a net positive environmental influence on at least the LREE supply (see Figure 4.9). Given the current overwhelming reliance on China coupled with their overall poor environmental performance for sourcing both LREEs and HREEs, a switch to an alternate supplier would have immediate environmental benefits. While the EU is currently importing only about 4 % LREEs from the USA, it could feasibly increase this percentage to counterbalance any shortages from the EU's current reliance on China. It is also a positive development in terms of environmental impact given that Mountain Pass now has the capabilities to process the ore in-house using cleaner technologies. In addition, is anticipated that Canada will become a leading supplier to the global REE market (Figure 4.6), as the production from Nechalacho and other deposits increase in the coming years. In contrast, any immediate increase of imports from Mount Weld, Australia would be less promising given the issues with radioactive waste and the uncertainty surrounding the LAMP facilities in 2024.

**Figure 4.9. LCA results for estimated impacts to climate change (GWP) for A) Nd production from Mountain Pass, California USA (MP) versus Bayan Obo, China (BO); and for B) Dy production from Eudialyte deposits from Nora Kärr, Sweden vs Bayan Obo, China**




**Source:** A) Modified after (Marx et al., 2018); B) Modified after (Zapp et al., 2018).

Immediate avoided impact calculations can be estimated using the values in Table 4.1, where the average EU consumption amount of LREEs (as represented by Nd) from 2016-2020 as well as the average EU suppliers (processed and refined) are listed (Eurostat 2022 data after (SCRREEN Project, 2023b)). Looking at the previous results of selected LCA analyses comparing Mountain Pass, California USA with Bayan Obo, China production (see Figure 4.9 A) after Marx et al., 2018) and considering these consumption and refined supply rates, we can calculate a simplified, approximate range of avoided impacts for selected impact categories for LREEs. For example, for impacts to climate change, Marx et al., (2018) estimates the share impacts of Nd for combined metal, metal oxide and RE concentrates from Mountain Pass between 50-75 kg $CO_2$ eq and Bayan Obo between 75-100 kg $CO_2$ eq per kg of Nd production. If it is considered that the EU consumes approximately 119 tonnes Nd per year and approximately 80 % of that is supplied by Chinese producers and refiners, an estimated 25-33 % immediate reduction of $CO_2$ emissions can be avoided by completely changing suppliers to the USA. At a 50 % reduction of reliance of imports from China, the avoidance of $CO_2$ impacts reduces to about 12 to 16 %, while about a 25 % reduction of reliance of imports from China could avoid approximately a mere 6-8 % of $CO_2$ emissions.

The environmental implications of the short term scenario for an alternate supplier of HREEs is less clear, with the picture unlikely to change in near term. While supplying HREE from the Norra Kärr deposit looks promising when comparing the environmental impacts presented in Figure 4.8, this deposit will not be available in the short-term scenario. Given Chinese (including production from Myanmar) current dominance in the HREE market, the immediate solution to any supply chain disruption would need to be the diversification of its smaller producing sources of virgin HREEs. The environmental impact of such diversification is however unclear as impact analyses would need to consider the role transporting less overall HREE ore from a larger number of countries. This would likely have negative environmental implications, principally on the climate and GWP. Canadian HREE production has the potential to alleviate some of this dependency, should the Nechalacho project be mining HREEs currently or in the near future. If diversification of smaller HREE suppliers is not possible, then the EU will need to consider reducing consumption. The substitution of HREEs in permanent magnets, specifically for wind turbines and EVs, is a topic of ongoing research and has produced thus far some promising results (Pavel, Lacal-Arántegui, et al., 2017; Pavel, Thiel, et al., 2017; Smith & Eggert, 2018). However, unless significant advances in substitution are rapidly made, the EU will likely continue to require HREEs in magnets or need to compensate with utilizing non-REE magnets, for example in wind turbines that require more maintenance.

**Table 4.1. Net import values of REE supply and demand for the EU in metric tonnes, average from 2016-2020 (in metal content).**

| REE | Global Production (in tonnes) | EU Consumption | EU Suppliers (refined) | Import Reliance (%) |
|---|---|---|---|---|
| Nd | 26845 | 119 | China (80%)<br>Other countries (11%)<br>UK (3%)<br>USA (2%) | 100 |
| Dy | 708 | 1.13 | China (60%)<br>Japan (18%)<br>Russia (21%)<br>UK (8%) | 100 |

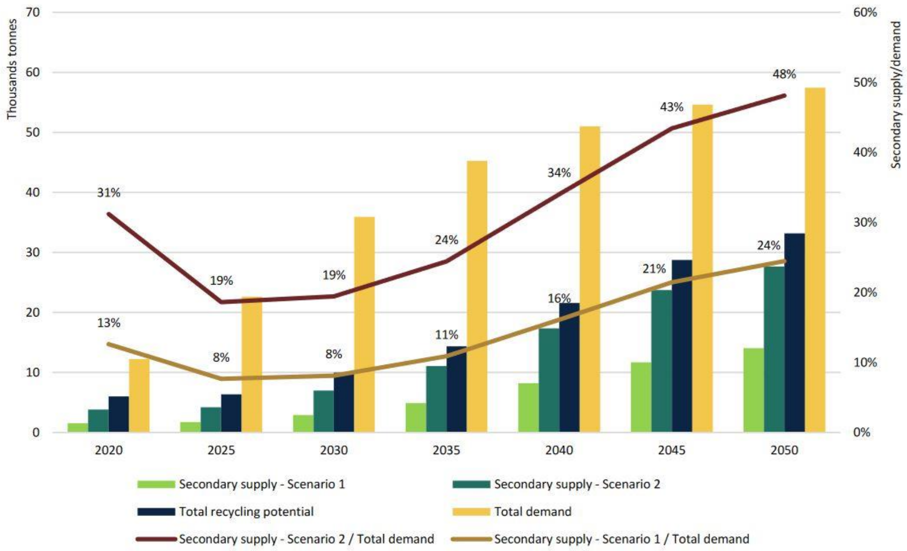
**Note:** EU Suppliers for Nd represent average values for all LREE;
**Source**: Eurostat 2022 data, SCRREEN Project, 2023b

It remains possible that recycling activities could begin to supply the EU with both HREE and LREE in the short term. This is highly dependent on the progress of the recycling facilities of, for example, HyProMag in the UK and in Germany, with a reported production start of late 2023 and 2024, respectively. As with the recycling of e-Waste, the question remains open if these facilities will be able to secure the input materials necessary to supply the entire EU block in the short term. Thus, recycling will be discussed in greater detail in the medium term scenario.

**Medium term (2025-2030) – Improving REE recycling potential**

The EU potential, future capacity for NdFeB magnet recycling as an environmentally friendlier option of sourcing of both LREE and HREE looks more favourable in the medium term. In a recent study, researchers examined model of two scenarios of EoL NdFeB magnet availability with the assumption that a viable magnet recycling chain is established. Their results are shown in Figure 4.10 below. The researchers modelled that it is reasonably possible that recycling could provide 8-19% of overall NdFeB magnet demand in the medium term between 2025-2030 (Rizos et al. 2022).

**Figure 4.10. Two modelled scenarios for NdFeB magnet aggregate demand in the short to long term, based on a theoretical (hypothetical) recycling potential (TRP), secondary supply and secondary supply demand ratio in the EU**




**Source:** Rizos et al., 2022

Not only does the future prospects for recycling to supply some REEs look promising in the medium term, but importantly the environmental impacts of replacing primary mining with recycling would be significantly reduced. In a previous comparison LCA exploring the impacts of the production of NdFeB from virgin versus recycling materials, the differences were significant. The results are shown in Table 4.2. As can be seen, the study found that the recycling of NdFeB magnets has lower impacts in all environmental impact categories as compared to those of magnet production from virgin materials (Jin et al., 2016). The authors calculate that up to about 55 % $CO_2$ eq per kg Nd magnet of emissions can be avoided by recycling. Given this study is from 2016, it was conducted assuming the magnet recycling methodology based on mixing recycled with virgin materials (Jin et al., 2016; Zakotnik & Tudor, 2015). Such environmental impacts may likely be even further reduced if calculations were to include the newer HPMS technology.

**Table 4.2. Comparison of LCA environmental impact results between virgin and recycled NdFeB magnets**

| Impact category | Unit | Virgin | Recycled |
|---|---|---|---|
| Global Warming | kg $CO_2$ eq | 27.602 | 12.453 |
| Acidification | $H^+$ moles eq | 20.524 | 11.320 |
| Carcinogenics | benzene eq | 0.069 | 0.035 |
| Non carcinogenics | toluene eq | 249.382 | 136.075 |
| Respiratory effects | kg PM2.5 eq | 0.124 | 0.059 |
| Eutrophication | kg N eq | 0.011 | 0.004 |
| Ozone depletion | kg CFC-11 eq | 1.25E-06 | 4.89E-07 |
| Ecotoxicity | kg 2,4-D eq | 94.285 | 45.345 |
| Smog | kg $NO_X$ eq | 0.109 | 0.034 |

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**Source:** Jin et al., 2016

Unfortunately, this does not solve the scenario should the EU's supply from China be completely disrupted and a complete alternative supplier be necessary. Thus, in the medium term, diversification of virgin sources along with increased sourcing from recycling will need to be considered. As such, the scenarios presented in the short term also apply to the medium term, with Canada projected to be an important player in the global REE supply chain in the medium term, with a tangible uptick in production around 2025 (Figure 4.6 above).

**Long term (2030-2040) – Domestic primary production**

There are several advanced and promising new exploration domestic projects in virgin REE mining on the horizon that have the potential to secure a longer-term source for the EU in the coming decades. These include the recently announced Per Geijer deposits and known resources at Norra Kärr in Sweden, but also those known deposits found in Greenland at Kvanefjeld. There are currently no known environmental LCA analyses yet conducted on the rather recently discovered Per Geijer deposits. However, given the developmental plans of LKAB and REETec to build an efficient extraction and processing plant that consumes lower energy and has lower $CO_2$ emissions (LKAB, 2022), one could use the environmental impacts calculated for Norra Kärr as a proxy (Figure 4.8). Any domestic production of Nd and Dy is promising to be much less polluting for the EU than the current production in China (Bayan Obo mine). Overall, it has been shown for Norra Kärr that this deposit could achieve environmental advantages over the current Bayan Obo production. This is due to a planned better emission control, as well as waste and sludge treatment caused by stricter environmental legislation in Sweden (Schreiber et al., 2016). Normalized total values of the effects per kg LREEs can be reduced by approximately 60 %, and for HREEs, a reduction of approximately 80 % can be reached in comparison to the Bayan Obo production (Schreiber et al., 2016). Such reduced environmental impacts could also be expected from the Per Geijer deposit given the promises of extraction plans. Furthermore, the increased investment in domestic refining capacity suggests the EU could become partially if not wholly self-sufficient in the long term. Within the EU, a growth in processing and refining capacity is expected, for example with the ongoing facility upgrades at Solvay in France and at NMP Silmet in Estonia together with the development of greener processing facilities for the Per Geijer ores from LKAB in Sweden, operating together with REETec.

Indeed, in simplified calculations utilizing the information in Table 4.1 and estimates for Dy production in Figure 4.9 B), there is again significant impact reductions for climate change (GWP). For example, conservative estimates for the share impacts for combined Dy metal, Dy metal oxide and RE concentrates from Norra Kärr, Sweden are between 100-200 kg $CO_2$ eq and Bayan Obo between 500-600 kg $CO_2$ eq per kg of Dy production. If it is considered that the EU consumes approximately 1.13 tonnes Dy per year and approximately 65 % of that is supplied by Chinese producers and refiners, an estimated 72-83 % immediate reduction of $CO_2$ emissions could be avoided by completely changing suppliers to an eventual domestic EU source. At a 50 % reduction of reliance of imports of HREEs from China, the avoidance of $CO_2$ impacts reduces to about 32-38 %, while about a 25 % reduction of reliance of imports from China avoids approximately 16-19 % $CO_2$ emissions (Zapp et al., 2018).

Nevertheless, investments into magnet and other REE recycling capabilities need to continue in order to lessen any impacts of future supply chain disruptions. It is estimated that in the late 2030s, there will be an increased availability of EoL magnets from EVs currently in use and potentially coupled to a slowdown in their demand (Rizos et al., 2022). This suggests that recycling will be increasingly important in the medium to long term. Previous estimates show that by 2040, between 16 % and 34 % REE demand could be met by recycling currently in use products and this number only increases by 2050 potentially reaching near 50 % (Rizos et al., 2022). As with other recycling sectors, it will be imperative in the coming years that the EU make advances in securing REE bearing recycling input material to ensure a future viable and reliable REE recycling sector.

**Summary of findings – REE Case Study**

| Scenario | Short-term (2023-2025) | Medium-term (2025-2030) | Long-term (2030-2040) |
|---|---|---|---|
| GWP (kg CO2-Eq) | Between **6 %** and **33 %** potential reduction depending on the level of supply shift from China to USA (from 25% to full shift). | Up to **55 %** potential reduction can be avoided by recycling Nd magnets | Between **16 %** and **83 %** potential reduction depending on the level of supply shift from China to a domestic source, like Sweden (from 25% to full shift). |

# 5 Key findings and conclusions

The methodological approach developed in this report, reflects on three core responses to supply chain disruptions (i) Establishment of trade relations with alternative suppliers, (ii) Growth in domestic capacity for recycling to satisfy demand and (iii) Growth in domestic capacity for primary extraction, processing and refining to satisfy demand.

## 5.1 For Europe

The key findings relevant for Europe and for each case study are summarized below.

<u>Case study nickel</u>

- Short term scenario (2023-2025): where it is assumed that the total or a portion of the EU demand for nickel that was met by China, could be replaced in short term by an alternative supplier such as Canada. The environmental impact assessment reveals that in terms of GWP, energy resource (non-renewable) and ecotoxicity to water, mining nickel in Canada has lower impacts than mining in China. The global warming potential of mining in Canada appears to be 45 % lower than the one in China.
- Medium term scenario (2025-2030): where it is assumed that improving the battery recycling industry in the EU can be a solution to meet the nickel demand in the long term. The available LCA studies show that recovery of secondary nickel from battery recycling has in general less environmental impacts than the impacts related to battery manufacturing using virgin materials. However, the global warming potential (GWP) of battery recycling is higher than battery production, due to higher demand for energy in recycling processes. This current result is expected to change in the coming decades as the EU's electricity mix will shift towards more renewable solutions.
- Long term (2030-2040): where improving refining capacity for production of class I nickel and nickel sulphate in the EU can be considered as a solution to a potential supply deficiency. The assessment shows that Finland is currently one of the biggest suppliers of refined nickel class I to the EU and has plans to increase its refining capacity by 50 %. This would mean the share of supply coming from Finland would increase from 22 % to 34 %. Increasing the supplies from Finland could greatly decrease the carbon footprint of this industry as refiners in Finland claim to achieve carbon neutrality by 2039.
- Currently 22 % of the EU supply of refined nickel Class I comes from Finland (Figure 3.10) amounting to 63,000 tonnes. Considering an increase in refining capacity in Finland of 50 % by 2040 (95,000 tonnes), this would mean the share of supply coming from Finland would increase from 22 % to 34 %.

<u>Case study rare earth elements</u>

- Short-term scenario (2023-2025): this would likely be the most critical in terms of market shock. There are currently viable alternatives for LREEs that would provide some immediate reduction in environmental impacts, such as shifting suppliers. It is estimated that between 25-33 % of $CO_2$ emissions could be avoided by completely changing suppliers to the USA. Given the current dominance of China and Myanmar in terms of HREE, the short-term scenario for HREEs looks less clear.
- Medium-term scenario (2025-2030): this looks more favourable for recycling to alleviate some pressures. Some estimates show that recycling could significantly reduce the environmental impacts, for example 55 % $CO_2$ eq per kg Nd magnet of emissions can be avoided by recycling. As with the short-term scenario, for this to be a viable alternative, investments into development of the REE recycling sector need to continue.

- Long-term scenario (2030-2040): appear the most promising for the EU to wean its dependency on REE imports with the development of domestic sources. The recently discovered Per Geijer REE deposit in Sweden could alleviate all (given the current estimations provided by LKAB and based on current consumption) of this reliance and would also provide significant reductions in many environmental impacts of the current REE supply chain. It is estimated that up to 72-83 % of $CO_2$ emissions could be avoided by wholly changing suppliers to an eventual domestic EU source.

## 5.2 Potential implication of this analysis for the world or at global level

Unforeseen supply chain disruptions can have global consequences for economies but also for impacts to the environment. The inter-dependency of supply chains (as in the nickel case), and the lack of diversity in suppliers, processing and manufacturing (as in the REE case) of materials makes the markets even more vulnerable. As disruptions cannot be foreseen in advance, it is difficult to assess exactly the global ramifications. Any benefit to the EU doesn't necessarily mean avoided impacts worldwide, as this is highly dependent on several factors, including global demand and supply dynamics, future market expectations and the applications of raw materials in relation to clean energy technologies and the broader green transition.

For the sake of simplification, short-term global environmental effects caused by the switch to an alternative, 'cleaner' supplier on the side of Europe depend on whether the demand gap would be filled by a third country. In the case of nickel, for instance, this report has found out that procuring nickel from Canada rather than China may decrease the global warming potential by 45 %. However, would this 45 % reduction in warming potential be truly 'global'? To answer this question, a main factor to consider is the consequences on the Chinese supply. In a scenario where global demand was not able to absorb the 'excess' supply from China, this could translate into lowered environmental impact globally. If instead, demand from third countries was in the position to absorb the Chinese supply, there would be no effects, leading to a zero-sum game.

As mentioned above, nickel is currently used for various clean energy technologies and due to its critical role in battery production, the global demand for nickel and nickel products is expected to increase significantly in the coming years. Due to this market projection, it appears unrealistic at this stage to estimate that China would decrease its supply due to the decreased demand from Europe. The same case appears to hold, even with stronger evidence, to the case of REE. Therefore, for raw materials where increased demand in the foreseeable future is expected, effects for the rest of the world appear to be irrelevant.

It should be mentioned that to have a comprehensive assessment of environmental consequences at global scale, several additional factors would need to be included in the analysis. One such factor is the application of the specific raw material. Both nickel and REE are currently considered key raw materials for the uptake and scaling up of clean energy technologies. If a decrease in global supply leads to a slowing down of the clean energy transition, for instance, the phasing out of fossil fuel systems might be delayed, leading to much wider consequences in relation to global climate change.

The international response and fallout from previous geopolitical events, such as the REE crisis of 2010-2011 when China suspended its exports as a result of tensions between it and Japan, shows that such disruptions can cause market uncertainties and lead to global resource price spikes and inventory stockpiling. What is clear is that a disruption could bring about both positive or negative repercussions in terms of environmental impacts, depending on investments made in securing these value chains now. For example, there are existing data for both Ni and REE that show recycling avoids significant emissions of $CO_2$ to the atmosphere. Thus, shifting focus to more sustainable sources for Ni or REEs could assist nations at reaching their own specific targets and goals, like achieving net zero or improvements in domestic material processing capacity.

In this study, emphasis has been placed on the supply side, operating under the assumption that the demand within the buyer country remains constant. It is crucial to highlight other potential policy responses, such as reducing domestic demand, embracing resource-saving innovations, or substituting materials with lower environmental impacts. This is particularly significant concerning nickel, given that future global supply of this material is expected to be influenced by advancements in processing lateritic ores in Indonesia and the Philippines, which are linked to higher environmental impacts during processing and refining compared to sulfidic ores.

## 5.3 Applicability of the framework & next steps

The methodological approach outlined in Sub-section 2.4, along with the wide range of potential variables, effects, and responses introduced, can serve as a template to evaluate the environmental impact of supply chain disruptions for any other materials. The key variables to considered for any future assessments include the material type, countries involved, nature of the disruption, response strategies, environmental impact types and their geographical distribution, as well as the timing of responses and effects. Depending on the size and complexity of the task, the assessment can be achieved by incorporating additional and interconnected variables into the study. The findings from such assessments could have a significant role in shaping well-informed policy-making processes.

# 6 List of abbreviations

| Abbreviation | Name | Reference |
|---|---|---|
| Dy | Dysprosium | |
| EEA | European Environment Agency | www.eea.europa.eu |
| EOL | End-of-life | |
| EoL-RIR | End-of-life Recycling Input Rate | |
| EoL-RR | End-of-life Recycling Rate | |
| EU | European Union | |
| FU | Functional Unit | |
| GWP | Global Warming Potential | |
| HREE | Heavy Rare Earth Elements | |
| JRC | Joint Research Center | |
| LCA | Life Cycle Assessment | |
| LCI | Life Cycle Inventory | |
| LREE | Light Rare Earth Elements | |
| NCA | Nickel-Cobalt-Aluminium | |
| Nd | Neodymium | |
| Ni | Nickel | |
| NMC | Nickel-Manganese-Cobalt | |
| NMH | Nickel-Metal Hydride | |
| REE | Rare Earth Elements | |

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