Environmental sustainability of NdFeB magnet recycling: foresight study on recycling systems and technologies

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Citation

Environmental sustainability of NdFeB magnet recycling: foresight study on recycling systems and technologies. (2024, November 1). *Environmental sustainability of NdFeB magnet recycling: foresight study on recycling systems and technologies*. Retrieved from https://hdl.handle.net/1887/4107092

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Colophon

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Ph.D. thesis, Institute of Environmental Sciences (CML), Leiden University, The Netherlands
1 November 2024
Keywords: Industrial Ecology; Recycling; NdFeB magnets; Recyclability; Technological innovation; Upscaling; Technological learning; Material flow analysis; Ex-ante LCA; Learning curves

ISBN: 9789051912333
Layout and cover design: Sander van Nielen
Printing: *Print&Bind* (www.printenbind.nl)

This research received funding from the European Union's *Horizon 2020 Research and Innovation Programme*, in the framework of SUSMAGPRO under grant agreement No. 821114.

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PROEFSCHRIFT

ter verkrijging van
de graad van doctor aan de Universiteit Leiden,
op gezag van rector magnificus prof.dr.ir. H. Bijl,
volgens besluit van het college voor promoties
te verdedigen op vrijdag 1 november 2024
klokke 10:00 uur

door
Sander Sebastiaan van Nielen

geboren te Gouda
in 1993

Promotores: Prof. dr. A. Tukker

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"When we try to pick out anything by itself,
we find it hitched to everything else
in the Universe."

– John Muir (1911)

Contents

Abbreviations

BEV	Battery electric vehicle
CRM	Critical raw material
DfR	Design for recyclability
DPP	Digital product passport
EV	Electric vehicle
EoL	End-of-life
EF	Environmental footprint impact assessment method
EU	European Union
EPR	Extended producer responsibility
FCC	Fluid catalytic cracking
GHG	Greenhouse gas
HDD	Hard-disk drive
HEV	Hybrid electric vehicle
HPMS	Hydrogen processing of magnetic scrap
HDDR	Hydrogenation–disproportionation–desorption–recombination
ICP-OES	Inductively coupled plasma optical emission spectrometry
KPI	Key performance indicator
LCA	Life cycle assessment
MRI	Magnetic resonance imaging machine
MFA	Material flow analysis
MIM	Metal injection molding
NdH_2	Neodymium hydride
NdFeB	Neodymium–iron–boron alloy
NiMH	Nickel-metalhydride
PV	Photovoltaic
PHEV	Plug-in hybrid electric vehicle
REE	Rare earth element
R&D	Research and development
RQ	Research question
§	Section
SSD	Solid-state drive
TRL	Technology readiness level
WEEE	Waste electric and electronic equipment

S Summary

This thesis investigated the potential for deploying magnet recycling technologies and their environmental consequences. These technologies are under development, aiming to provide a more sustainable alternative to magnets made from virgin rare earth elements (specifically neodymium-iron-boron (NdFeB) magnets). The market for rare earth magnets is experiencing great turmoil, with fluctuating prices and an exponentially growing demand driven by wind turbines and electric vehicles. As consumption increases, so do the waste volumes. Together with the limited mining capacity, this urges the development of a recycling system.

From an environmental perspective, the impact of magnet recycling depends on the availability of end-of-life (EoL) magnets, the efficiency of recovery and recycling technologies, and the effectiveness of the recycling system as a whole. The recycling processes have a low technology readiness level, therefore it is important to anticipate improvements in performance during their development. To study the trend of environmental impacts after introducing recycling on an industrial scale, a potential approach is to use environmental learning curves. However, a well-founded method is lacking for studying changes in environmental impact resulting from learning effects. We addressed these aspects of magnet recycling by investigating the following research questions.

1. What factors determine the success of recycling minor metals and their recycling systems?
2. What is the volume of NdFeB magnet waste in Europe? Which waste flows are most suitable for recycling?
3. What are the environmental impacts of industrial-scale recycling processes for NdFeB magnets?
4. How does technological learning affect environmental impacts, and how can life cycle analyses (LCA studies) address this effect?

Rare earth elements, used in magnets, are part of a larger group of minor metals that share several recycling challenges. To improve recycling rates, it is important to understand the bottlenecks and drivers in recycling value chains. Therefore, Chapter 2 analyzed existing recyclability frameworks and related recycling literature, revealing 113 factors that determine the success of recycling minor metals. These factors reflect a variety of perspectives, including economic, thermodynamic, and design perspectives. Together, they describe how product characteristics, recycling technologies, and societal conditions contribute to recyclability. These findings informed a novel recyclability assessment framework. The framework introduces a set of indicators, grouped by the stages of the recycling value chain, i.e. manufacturing, use & collection, preprocessing, metallurgical recovery and secondary marketing. Additionally, overarching factors are

addressed. Chapter 2 presented three case studies, in which the framework was applied to evaluate and compare the recyclability of minor metals in various products.

Chapter 3 quantified the amount of neodymium waste in European countries using material flow analysis, and assessed the recyclability of major EoL products using the framework from Chapter 2. In 2019, a waste flow of 2.8 kt neodymium was found, mainly consisting of NdFeB magnets. HDDs make up a large part of this waste flow, providing an attractive input for recyclers for several years. Other sources of secondary Nd, that are becoming increasingly significant, are the magnets used in industrial applications (pumps and robots) and in conventional cars. In the future, EoL electric vehicle motors and wind turbines may provide a source of neodymium with good recyclability. Given these dynamics of emerging magnet applications and waste flows, flexibility is required in waste collection and preprocessing. The recyclability assessment identified some bottlenecks in the recycling of EoL magnets. By addressing these, a robust recycling system can be established to handle the future growth in waste volumes. Future research could reduce uncertainty in the outcomes by mitigating reporting errors in production and trade data. Also, more detailed data on the market share of different magnet types would be beneficial.

Chapter 4 investigated the environmental impacts of new technologies for recovering, recycling and remanufacturing NdFeB magnets. The focus was on the direct recycling of NdFeB alloys, which recovers the materials as powders rather than separating the individual elements. The anticipated performance of large-scale recycling was modeled, and compared to the production of primary magnets. This analysis applied ex-ante LCA, combining input from measurements of pilot processes, expert technology forecasts, thermodynamic modelling, and equipment data from manufacturers. The results show that at pilot scale, recovered NdFeB powders have lower impacts than primary powders for almost all impact categories, demonstrating the environmentally benefit of recovering NdFeB alloys. Magnets from anticipated large-scale recycling have over 80% lower impacts than primary magnets in most of the impact categories analyzed. We examined the effect of four technology developments: process changes, size scaling, internal recycling, and optimization. All four types contributed to improving the performance from pilot-scale to industrial scale. The final configuration was validated by comparing it with an industrial reference and theoretical optimum configuration. Effective changes include reducing material losses during sieving, and improving the energy efficiency through size scaling. Furthermore, four magnet manufacturing routes were investigated: sintering, extrusion, metal injection molding (MIM), and bonding. Each has a distinct environmental profile, but all can progress to similarly low levels of impact. Extrusion and MIM are less mature, which results in higher initial environmental impacts. Since each magnet type has unique properties, the choice among routes should be based mainly on the functional requirements. Future LCA studies may incorporate cleaner electricity production and developments in primary magnet supply chains.

Chapter 5 fundamentally explored the effects of learning on the environmental impacts of industrially deployed technologies. We reviewed the theoretical foundations and empirical evidence of technological learning. We identified various learning mechanisms, some of which only affect production costs. Previous studies have mainly demonstrated reductions in impacts related to energy use. A key observation is that the results may vary by impact category, and certain impacts may not decline at all. Next, we developed a procedure for assessing learning effects in ex-ante and prospective LCA. We argue that learning involves operational or organizational changes, which are motivated by incentives. Therefore, environmental impacts may follow a learning curve if the origins of impacts coincide with where the main incentives are directed. By providing step-by-step guidelines to evaluate environmental learning effects and learning rates, we allow future studies to model the impact trends during industrial technology development. Applying the procedure to magnet recycling, significant learning effects are expected. Furthermore, we highlighted the need for expanding the evidence base for the environmental effects of technological learning. This can be achieved by reinterpreting datasets of existing technologies to determine their learning rates.

In conclusion, the average environmental footprint of NdFeB magnets in Europe can be reduced by improving and deploying magnet recycling technology. Deployment is limited by the availability of waste magnets. Recycling could at most fulfill 45% of the NdFeB demand in 2019, and this maximum share may decrease in the future due to the longer lifespans of emerging magnet applications. Still, recycling has a large growth potential, as it is currently not widely practiced. The environmental benefits of recycling compared to primary production will initially be limited, but may increase significantly as the processes improve. Recycled magnets can ultimately achieve over 80% lower impacts for most impact categories, as revealed through ex-ante LCA. The analysis of learning effects indicated that the environmental impacts may decrease by about 20% for each doubling of the cumulative recycling output.

This research supports previous studies that demonstrate the contribution of NdFeB magnet recycling to sustainable development. Recycling reduces the pressure on the primary magnet supply chain and increases the availability of raw materials for producing low-emission technologies such as electric motors. Moreover, recycling technologies that are implemented on a large scale have lower environmental impacts than primary production processes. Recycling alone cannot satisfy the demand for NdFeB magnets. However, the combination of multiple recycling technologies with effective waste collection systems can contribute to a significantly more circular magnet industry. Emerging technologies for magnet recycling enable successful and sustainable recycling systems in the future. These systems can be realized with circular business models and guiding policies based on this research.

This research focused on magnet recycling in Europe, making the results less relevant for other world regions. Other countries may differ in the demand for magnet applications, the societal context of recycling systems and the electricity mix. Depending on the growth rate of magnet demand, the maximum share of recycled feedstock may be higher or lower. When comparing the environmental impact of primary and recycled magnets within a non-European country, the preferred alternative is likely to be the same as in this study. However, a comparison with the global average of primary magnets requires a separate analysis.

Some aspects require further attention by researchers and recyclers. These include digital solutions for tracking and detecting magnets in products, collaboration to align the stages of recycling, co-recovery of magnets and other materials, and methods for promoting behavior that supports recycling. Recyclers can use insights in the various EoL magnet types and differences among countries to develop recycling activities. These recyclers can gain essential experience by learning-by-doing in an industrial setting.

S

Samenvatting

Milieuvriendelijkheid van de recycling van NdFeB-magneten – Verkennende studie van recyclingprocessen en -systemen

Dit proefschrift onderzocht het potentieel van de inzet van magneetrecyclingtechnologieën en de gevolgen voor het milieu van deze recycling. De technologieën zijn in ontwikkeling, met als doel een duurzamer alternatief te bieden voor magneten gemaakt van vers gedolven zeldzame aardmetalen (specifiek neodymium-ijzer-boor- of NdFeB-magneten). De markt voor deze magneten is onvoorspelbaar, met volatiele prijzen en een exponentieel groeiende vraag, vooral gedreven door windturbines en elektrische voertuigen. De toenemende consumptie veroorzaakt ook meer afval, en gezien de beperkte mijnbouwcapaciteit is er dringend behoefte aan de ontwikkeling van een recyclingsysteem.

Vanuit milieuperspectief hangt de impact van magneetrecycling af van de beschikbaarheid van gebruikte magneten, de efficiëntie van terugwinnings- en recyclingprocessen en de effectiviteit van het recyclingsysteem als geheel. Omdat de recyclingprocessen een lage marktrijpheid hebben, is het belangrijk om rekening te houden met prestatieverbeteringen tijdens de ontwikkeling ervan. Om de ontwikkeling van milieueffecten na de introductie van recycling op industriële schaal te bestuderen, lijken 'milieuleercurven' een mogelijke benadering. Echter ontbreekt het aan een solide methode om de gevolgen van leereffecten voor de milieu-impact te bepalen. We hebben deze aspecten van magneetrecycling onderzocht aan de hand van de volgende onderzoeksvragen.

1. Welke factoren bepalen het succes van het recyclen van technologie-metalen en hun recyclingsystemen?
2. Wat is de hoeveelheid NdFeB-magneetafval in Europa? Welke afvalstromen zijn het meest geschikt voor recycling?
3. Wat zijn de milieueffecten van recyclingprocessen voor NdFeB-magneten op industriële schaal?
4. Hoe beïnvloeden technologische leereffecten de milieu-impact, en hoe kunnen levenscyclusanalyses (LCA-studies) dit effect meenemen?

Zeldzame aardmetalen die worden gebruikt in magneten, maken deel uit van de groep technologie-metalen die meerdere recyclinguitdagingen gemeen hebben. Om de recyclingpercentages te vergroten, is het belangrijk om de knelpunten en aanjagers van de recyclingketens te begrijpen. Daarom analyseerde Hoofdstuk 2 bestaande studies naar recycleerbaarheid en aanverwante literatuur over recycling. Zo werden 113 factoren blootgelegd, die het succes bepalen van het recyclen van technologie-metalen.

Deze factoren vertegenwoordigen meerdere invalshoeken, waaronder het economische, thermodynamische en ontwerpperspectief. Samen beschrijven deze hoe productkenmerken, recyclingtechnologieën en de maatschappelijke context bijdragen aan recycleerbaarheid. Deze bevindingen vormden de basis voor een nieuw raamwerk voor de beoordeling van recycleerbaarheid. Het raamwerk introduceert een reeks indicatoren, gegroepeerd per stap van de recyclingketen: productie, gebruik en inzameling, voorverwerking, metallurgische terugwinning en opnieuw op de markt brengen. Ook overkoepelende factoren worden geanalyseerd. Hoofdstuk 2 presenteert drie voorbeeldstudies, waarin het raamwerk wordt toegepast om de recycleerbaarheid van technologie-metalen in verschillende producten te analyseren en te vergelijken.

Hoofdstuk 3 kwantificeerde de hoeveelheid neodymiumafval in Europese landen met behulp van stofstroomanalyse, en gebruikte het raamwerk uit Hoofdstuk 2 om de recycleerbaarheid van belangrijke afgedankte producten te bepalen. Voor 2019 werd een afvalstroom van 2,8 kt neodymium berekend, voornamelijk bestaande uit NdFeB-magneten. Harde schijven vormen een groot deel van deze afvalstroom en zullen daarom in de komende jaren een aantrekkelijke input zijn voor recyclers. Verder vormen magneten in industriële toepassingen (pompen en robots) en in conventionele auto's groeiende bronnen van secundair neodymium. In de toekomst worden ook oude elektrische automotoren en windturbines belangrijk als bron van neodymium, met redelijk goede recycleerbaarheid. Gezien deze dynamiek van opkomende magneettoepassingen en afvalstromen is flexibiliteit vereist bij de inzameling en voorverwerking van afval. De recycleerbaarheidsanalyse liet enkele knelpunten zien in de recycling van magneten. Door deze knelpunten aan te pakken kan een robuust recyclingsysteem worden opgebouwd, dat de toekomstige groei van de afvalvolumes aankan. Toekomstig onderzoek zou de onzekerheid in de uitkomsten kunnen verkleinen door rapportagefouten in productie- en handelsgegevens aan te pakken. Ook zouden gedetailleerdere gegevens over het marktaandeel van verschillende magneettypen nuttig zijn.

Hoofdstuk 4 onderzocht de milieueffecten van nieuwe technologieën voor het terugwinnen, recyclen en opnieuw produceren van NdFeB-magneten. De focus lag op de directe recycling van NdFeB-legeringen, waarbij het materiaal in poedervorm wordt teruggewonnen in plaats van de afzonderlijke elementen te scheiden. De verwachte prestaties van grootschalige recycling werden gemodelleerd en vergeleken met de productie van primaire magneten. Deze analyse maakte gebruik van ex-ante LCA, en combineerde meetgegevens van proefprocessen, technologische voorspellingen van deskundigen, thermodynamische modellen en apparatuurspecificaties van fabrikanten. Uit de resultaten blijkt dat NdFeB-poeders, teruggewonnen met een proefinstallatie, een lagere impact hebben dan primaire poeders voor bijna alle impactcategorieën. Dit toont het milieuvoordeel aan van de terugwinning van NdFeB. Grootschalige recycling van magneten heeft naar verwachting een ruim 80% lagere impact dan primaire magneetproductie in de meeste geanalyseerde impactcategorieën. We onderzochten ook het effect van vier technologische ontwikkelingen: procesveranderingen, schaalvergroting, interne recycling en optimalisatie. Alle vier de typen dragen bij aan het verbeteren van de prestaties van proefinstallatie naar industriële schaal. Het eindpunt van de aanpassingen werd gevalideerd door het te vergelijken met industriële referentieprocessen en een theoretisch optimale configuratie. Effectieve aanpassingen zijn bijvoorbeeld het verminderen van materiaalverlies tijdens het zeven en het verbeteren van de energie-efficiëntie door apparatuur op te schalen. Verder werden vier productieroutes voor magneten onderzocht: sinteren, extrusie, metaalspuitgieten (MIM) en binden met kunststof. Elke route heeft een eigen milieuprofiel, maar ze kunnen allemaal een vergelijkbaar laag impactniveau bereiken. Extrusie en MIM zijn minder volwassen, wat leidt tot een hogere initiële impact. Elk type magneet heeft unieke eigenschappen, daarom hangt de keuze tussen routes voornamelijk af van de functionele eisen. Toekomstige LCA-studies kunnen rekening houden met schonere elektriciteitsproductie en ontwikkelingen in de toeleveringsketens van primaire magneten.

Hoofdstuk 5 onderzocht het effect van leerprocessen op de milieu-impact van industrieel toegepaste technologieën op fundamenteel niveau. We beschouwden de theoretische grondslagen en het empirische bewijs van technologisch leren. Zo identificeerden we verschillende leermechanismen, waarvan sommige alleen de productiekosten beïnvloeden. Eerdere studies bewijzen vooral de vermindering van de impact die samenhangt met energiegebruik. Een belangrijke observatie is dat de effecten kunnen variëren per impactcategorie, en dat sommige impacts helemaal niet afnemen. Vervolgens hebben we een procedure ontwikkeld voor het beoordelen van leereffecten in ex-ante en toekomstgerichte LCA. Wij stellen dat leren operationele of organisatorische veranderingen met zich meebrengt, die gedreven worden door externe prikkels. Daarom kunnen milieueffecten een leercurve volgen als de oorsprong van de milieu-impact samenvalt met waar de belangrijkste prikkels aangrijpen. We hebben stapsgewijze richtlijnen ontwikkeld om leereffecten en leersnelheden op milieugebied te bepalen, zodat toekomstige studies de impacttrends van industriële technologie in ontwikkeling kunnen modelleren. Op basis van deze procedure verwachten we voor magneetrecycling aanzienlijke leereffecten. Verder benadrukt deze studie de noodzaak om het wetenschappelijke bewijs voor de milieueffecten van technologisch leren te vergroten. Dit is mogelijk door datasets van bestaande technologieën te herinterpreteren om de leersnelheden te bepalen.

Concluderend kan de gemiddelde ecologische voetafdruk van NdFeB-magneten in Europa worden verminderd door de magneetrecyclingtechnologie te verbeteren en uit te rollen. De uitrol wordt beperkt door de beschikbaarheid van gebruikte magneten. Recycling zou in 2019 maximaal 45% van de NdFeB-vraag kunnen vervullen, en dit maximale aandeel kan in de toekomst afnemen door de langere levensduur van opkomende magneettoepassingen. Toch heeft recycling een groot groeipotentieel aangezien recycling nu nauwelijks plaatsvindt. De milieuvoordelen van recycling ten opzichte van primaire productie zullen aanvankelijk beperkt zijn, maar kunnen aanzienlijk toenemen naarmate de processen verbeteren. Gerecycleerde magneten kunnen uiteindelijk een ruim 80% lagere impact bereiken voor de meeste impactcategorieën, zoals blijkt uit de ex-ante LCA. De analyse van leereffecten geeft aan dat de milieueffecten met ongeveer 20% kunnen afnemen bij elke verdubbeling van de cumulatieve productie van recycling.

De bevindingen bevestigen eerdere onderzoeken die lieten zien dat het recyclen van NdFeB-magneten bijdraagt aan duurzame ontwikkeling. Recycling vermindert de druk op de productieketen van primaire magneten, en maakt meer grondstoffen beschikbaar voor de productie van emissiearme technologieën zoals elektromotoren. Bovendien hebben recyclingtechnologieën die op grote schaal worden geïmplementeerd een lagere impact op het milieu dan primaire productieprocessen. Recycling op zichzelf kan niet voldoen aan de grote vraag naar NdFeB-magneten. Echter, door meerdere recyclingtechnologieën te combineren met effectieve afvalinzameling, kan een veel meer circulaire magneetindustrie worden bereikt. Opkomende processen voor magneetrecycling maken succesvolle en duurzame recyclingsystemen in de toekomst mogelijk. Deze systemen kunnen worden gerealiseerd met circulaire bedrijfsmodellen en richtinggevend beleid, gebaseerd op dit onderzoek.

Dit onderzoek richtte zich op magneetrecycling in Europa, wat de resultaten minder relevant maakt voor andere regio's in de wereld. Andere landen kunnen verschillen in de vraag naar magneettoepassingen, de maatschappelijke context van recyclingsystemen en de elektriciteitsmix. Afhankelijk van de groeisnelheid van de vraag naar magneten kan het maximale aandeel gerecyclede grondstoffen hoger of lager zijn. Bij het vergelijken van de milieu-impact van primaire en gerecyclede magneten binnen een niet-Europees land zal het voorkeursalternatief waarschijnlijk hetzelfde zijn als in dit onderzoek. Een vergelijking met het wereldwijde gemiddelde van primaire magneten vereist echter een aparte analyse.

Enkele aspecten vereisen verdere aandacht van onderzoekers en recyclers. Het gaat om digitale systemen voor het volgen en detecteren van magneten in producten, samenwerking en afstemming tussen alle partners in de recyclingketen, kansen voor het gelijktijdig terugwinnen van magneten en andere materialen, en methoden om recyclingvriendelijk gedrag te bevorderen. Recyclingbedrijven kunnen inzichten in de verschillende magneettypen en verschillen tussen landen gebruiken om hun recyclingactiviteiten te ontwikkelen. Voor deze bedrijven is het essentieel om ervaring op te doen door recycling toe te passen in een industriële omgeving.

1 Introduction

1.1 The industrial ecosystem of magnets

The discovery of NdFeB magnets was a seemingly small event with large consequences. In a modest scientific publication, Sagawa et al. (1984) disclosed a permanent magnet made of neodymium, iron and boron. NdFeB is the strongest permanent magnetic material known to date, owing to a fine-tuned production process and an optimized composition that includes other rare earth elements (REEs). It enabled the development of HDDs, tiny speakers in almost every gadget, compact electric motors, and efficient wind turbines. This wide range of applications resulted in rapid growth of magnet demand. However, innovation does not end here. New solutions are needed to address two major drawbacks: the pollution caused by REE mining (Arshi et al., 2018), and the vulnerability of the supply chain to disruptions such as geopolitical struggles (Sprecher et al., 2017).

The supply chains of neodymium and other REEs used in magnets are of large geopolitical importance for two reasons. First, NdFeB magnets are a key enabler of electrified renewable energy systems, due to their use in electric vehicles (EVs) and wind turbines. These low-emission technologies require large amounts of materials for their expansion. The economic importance of NdFeB magnets is further increased by their application in drones, robots, and military equipment (Pavel et al., 2020). Second, in recent decades, China has established a dominant position in the magnet supply chain, from REE mining and refining to magnet manufacturing (Smith et al., 2022; Tukker, 2014). This is problematic for other countries that rely on imports with a risk of logistic or political interruptions. REEs are not geologically rare; in fact, reserves are over 400 times larger than the annual mining rate (USGS, 2023). Still, few mines outside of China have opened due to long lead times, economic infeasibility and in fear of damaging the local environment (Filho, 2015; Liu et al., 2016). The risk of material shortages threatens the deployment of EVs and wind turbines if the REE supply cannot ramp up in time (Watari et al., 2020). The combination of high economic importance and high supply risk means that REEs are considered critical raw materials (Blengini et al., 2020; USGS, 2022).

To reduce the need for mining and mitigate supply risks, recycling is a promising solution. It diversifies the supply of magnets by making secondary resources available. Therefore, the European Union aims to obtain 25% of its REE demand from recycling by 2030, to improve its strategic autonomy (European Commission, 2023). Several technologies for magnet recycling have been investigated in laboratories, aiming to provide a more sustainable alternative to primary magnet production. The recent growth in commercial and political interest in this technology raises a new question, which is the main question of this doctoral thesis:

> *What is the potential and the environmental consequence of deploying large-scale NdFeB magnet recycling?*

To answer this question, three challenges have to be addressed. These challenges connect to the large themes of *Industrial Ecology*, as decribed in this chapter. Firstly, although recycling is conceptually simple, large-scale recycling systems can be complex to build and understand (see §1.3). Secondly, the wide range of magnet applications gives rise to an opaque societal metabolism, and tracing the waste flows is challenging (see §1.4). Thirdly, magnet recycling and other emerging technologies are constantly evolving, and therefore require a prospective approach to assessing their future sustainability benefits (see §1.5, §1.6). After summarizing the current status of magnet recycling in §1.2, the following sections introduce the research opportunities and research questions that guided the investigation into the recycling of NdFeB magnets.

1.2 Current status of magnet recycling technology

The recycling rates of NdFeB magnets are presently low due to various challenges in waste collection and recovery (Balaram, 2019; Jowitt et al., 2018; §1.3). Until recently, NdFeB magnets were rarely collected or sorted, consequently ending up in ferrous scrap after shredding (Bandara et al., 2014). This is how most small magnets, which are found in consumer electronics and vehicles, are lost. In most countries, the recovery of NdFeB material has until now been uneconomical (Binnemans et al., 2021), because labor-intensive recycling processes cannot compete with the price of primary production.

Figure 1.1 outlines various technologies that have been explored for the recovery and recycling of waste NdFeB (Yang et al., 2017). If devices are disassembled properly, magnets may be reused directly. The shortest recycling loop is through powder metallurgy, which recovers NdFeB powders from scrap magnets. Powder metallurgical recycling is also known as direct alloy recycling, because the magnet alloy is not separated into its constituent elements but used directly to make new magnets. A notable example is *hydrogen processing of magnet scrap* (HPMS), which uses hydrogen gas to pulverize magnets (Walton et al., 2015). Other recycling technologies, including hydrometallurgy (dissolution in acids) and pyrometallurgy (melting), purify and recover individual elements. However, this requires large amounts of either chemicals or energy. In contrast, HPMS is a promising technology from an environmental perspective because it operates at room temperature and uses no solvents (Ormerod et al., 2023; Yang et al., 2017).

During this doctoral research project, the development of magnet recycling has accelerated remarkably. The public awareness increased due to a growing media attention. News media reported on business efforts to recycle magnets, from the securing of funding to the opening of pilot plants (Figure 1.2). These business activities have been enabled by previous research efforts and are encouraged by policy plans (Koese et al., 2024). Specifically, if the EU Critical Raw Materials Act is implemented, it can stimulate the demand for recycled REEs. This combination of trends is likely to result in more widespread recycling in the future.

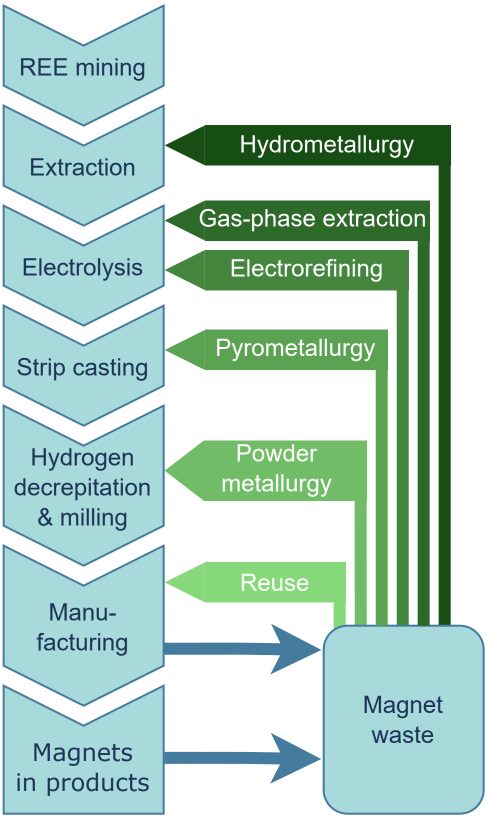

FIGURE 1.1: Overview of NdFeB recovery methods, and the point of entry of material into the primary supply chain. For completeness, direct reuse is included as a short-loop alternative to recycling. Reproduced from Koese et al. (2024).

This study aims to fill important knowledge gaps for the emerging magnet recycling sector. Scientific research can guide the advancement of new magnet recycling systems and technologies. An example of a new recycling initiative is the SUSMAGPRO project, which aimed to implement direct alloy recycling. SUSMAGPRO was a European Horizon-2020 research project[1] that lasted 4 ½ year and brought together a consortium of 19 organizations. This thesis was largely written in the context of SUSMAGPRO. The collaboration with SUSMAGPRO technology developers has created unique opportunities for interaction and data collection. This enabled us to model recycling technologies in great detail and gain more insights into the practical challenges of recycling. Chapter 4 describes SUSMAGPRO technologies in more detail.

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[1] Grant Agreement No. 821114. doi:10.3030/821114.

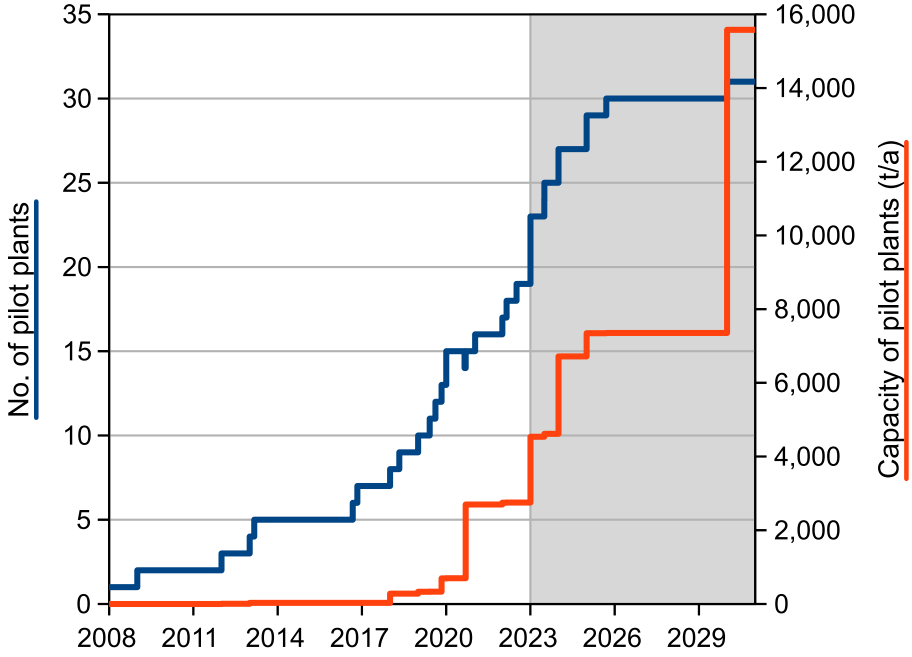

FIGURE 1.2: Number of new plants (blue) and their total recycling capacity (red) for NdFeB magnet recycling. The (expected) start of operation is displayed; when only the starting year was communicated, January 1st was assumed. For 7 of 28 facilities, no (estimated) capacity was released. The shaded area indicates announced facilities. Reproduced from Koese et al. (2024).

1.3 The role and complexity of recycling systems

Large-scale recycling is achieved through recycling systems, consisting of technical processes, physical infrastructure[2] and organizational structures. The societal benefits of metal recycling are much broader than waste management, as it contributes to a diversified metal supply, reduces price volatility, minimizes environmental impacts, and reduces pollution from landfills (Hagelüken & Goldmann, 2022; Hagelüken et al., 2016). In a circular economy, recycling is one of the strategies to close material loops; recycling enables the recovery of materials from EoL products that cannot be repaired or reused. Recycling and other circular strategies also compete for a limited supply of EoL products. Extending this concept, waste management was studied as a reverse market, where classical economic market forces apply (Stindt et al., 2017). The dynamics of reverse supply chains are shaped by the available resources and by interactions of involved stakeholders.

Improving a recycling system starts with measuring its current status and identifying

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[2] Technical processes refer to the specific methods and procedures involved in recycling. Physical infrastructure refers to supporting facilities, such as buildings, machinery, and transportation networks

any problems. Three common indicators are widely used to measure the success of recycling: end-of-life (EoL) recycling rate, recycled content, and old scrap ratio[3] (Graedel et al., 2011a). Recycling rates are low for many metals, especially for minor metals (Graedel et al., 2011a,b; Hagelüken & Meskers, 2010). This is a concern because minor metals such as REEs are key enablers of specialized technologies, and are therefore also referred to as technology metals or specialty metals (Graedel et al., 2011b). This motivates research into the recycling of these metals.

Previous research has documented several challenges for the implementation of effective recycling. Cramer (2018) categorized these as economic and business barriers (e.g. upfront investment costs for recycling infrastructure), regulatory barriers (e.g. standards and certifications), and social barriers (e.g. throw-away mentality). Minor metals face additional obstacles due to their dispersed use in numerous components and devices, and limited technology options for recycling emerging applications (Mathieux et al., 2018). In addition, tracing these metals can be challenging due to complex global trade networks (Nansai et al., 2014). The interpretation of recycling indicators for emerging minor metal waste flows requires caution (Hagelüken & Goldmann, 2022). While demand increases, high recycled content and old scrap ratios cannot be achieved. A low EoL recycling rate suggests that some parts of the recycling system have yet to be established, without specifying which parts. More specific indicators would help to identify the limiting factors of minor metal recycling. In fact, to establish effective recycling systems requires a holistic approach and an understanding of the role of all stakeholders (Reuter et al., 2013). Therefore, this thesis aims to answer the following research question (RQ).

RQ 1. *What factors determine the success of recycling minor metals and their recycling systems?*

This research question aims to look beyond current recycling rates, and create an overview of economic, technical and societal factors that play a role. The objective is to develop a recyclability assessment framework for minor metals in general, and later apply the insights to neodymium in NdFeB magnets. ‘Recyclability’ refers to the feasibility of recovering materials from discarded products (Henstock, 1988). To evaluate recyclability, multiple indicators are needed, addressing barriers and drivers throughout the recycling value chain.

1.4 Gauging the potential of magnet recycling

Conceptualizing recycling as a reverse supply chain (see §1.3) highlights the importance of secondary resources, namely the waste flows that serve as feedstock. For NdFeB, neodymium, and other minor metals, the availability of secondary resources can be a major limiting factor. In order to design and optimize reverse logistics of used NdFeB magnets, for instance using an optimization algorithm (Jin et al., 2018b), it is essential to quantify the waste flows. This is a challenge because NdFeB magnets are applied in a wide variety of products and components. For example, a passenger car has magnets in ten types of components, ranging from speakers to radiator fans (Restrepo et al., 2017).

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[3] The old scrap ratio describes the fraction of old scrap in the total recycling flow. Recycled content includes old scrap and pre-consumer scrap.

Moreover, in most electrical and electronic devices, magnets only account for a small fraction of the device's weight (Nansai et al., 2014). To address the potential for magnet recycling in European context, research question 2 was formulated.

RQ 2. *What is the volume of NdFeB magnet waste in Europe? Which waste flows are most suitable for recycling?*

This question can be answered by combining two approaches, material flow analysis (MFA) and recyclability assessment (see §1.3). MFA is introduced below. By combining these two quantitative assessments, a holistic view on the recycling system is pursued, covering material flows, recycling processes, and stakeholders.

Material flow analysis is the study of stocks and flows of materials and substances in society (van der Voet, 1996). To quantify the evolving flows of neodymium, a valuable approach is dynamic MFA. Dynamic MFA quantifies the flows into and out of the use phase, using either a stock-driven or an input-driven model (Müller et al., 2014). Stock-driven models calculate the flow into use based on observed or projected stock growth while accounting for discarded products. This approach is useful when precise information about the in-use stock is available, or when studying bulk metals (Fishman et al., 2014). Input-driven models start from product consumption and waste flows. Stocks are derived from the difference in in- and outflows, requiring an initial stock estimate and long time series. This approach is more suited for studying products that are weakly linked to population size, and for substances that only occur in specific product types. Both models typically calculate waste flows using lifespan distributions or averages, derived from e.g. surveys on electronic products (Forti et al., 2018; Thiébaud et al., 2018).

This thesis employs an input-driven MFA model to conduct a detailed analysis, aiming to disaggregate neodymium flows by product and country. To accurately model these material flows, this research accounts for two particularities of magnets. First, NdFeB magnets have been applied in an increasing number of applications over the last decades. It is therefore important to consider the changing market share of NdFeB magnets compared to other magnet types. Second, the wide scope of this study addresses the diversity of magnet types and applications, including those with a small associated Nd flow. In contrast, prior studies have either omitted certain magnet applications or assumed a constant market share of NdFeB magnets. The diversity of magnets is also acknowledged in the next section by considering recycling as a system with multiple inputs and outputs.

1.5 Environmental analysis of technological innovations

Recycling is often assumed to be environmentally beneficial. In industrial ecology, this assertion is tested by comparing different waste management options. Unfortunately, it was not possible to make a complete environmental comparison between NdFeB recycling and other types of waste treatment, because there are no characterisation factors for the extraction (resource depletion potential) and emission (toxicity) of REEs used in magnets (Edahbi et al., 2019; Fazio et al., 2018). A related question is whether the benefits of recycling outweigh the impacts of waste collection. However, the decision to collect waste is mostly motivated by the societal aversion of waste dumping, landfilling and incineration. From a life cycle perspective, the emissions of waste collection are therefore attributed to the use of a product rather than its recycling. Hence the scope can be limited to waste sorting and recycling.

Magnet recycling and other technological innovations contribute to the decoupling of environmental degradation from human wellbeing. As indicated by the $IPAT$ equation, innovations contribute to technological efficiency (T), and can be complemented by moderating population growth (P) and shifting from materialistic affluence to wellbeing (A) (Chertow, 2000). For technology developers, consumers and policy makers, it is important to understand the development prospects of a new innovation before its introduction to the commercial market. This foresight allows to direct investments towards the most promising technologies, and underlines the need for ex-ante life cycle assessment (LCA) (Cucurachi et al., 2018; van der Giesen et al., 2020).

For the magnet recycling processes introduced in §1.2, several studies have investigated the environmental impacts using LCA. Comparative studies indicated that recycling has lower environmental impacts than primary magnet production. Shorter recycling loops, as shown in Figure 1.1, generally perform better (Elwert et al., 2017; Wang et al., 2022). Recent advances in direct alloy recycling technologies justify a renewed investigation of environmental impacts, including an outlook on industrial-scale performance. This motivated the formulation of RQ 3.

RQ 3. *What are the environmental impacts of industrial-scale recycling processes for NdFeB magnets?*

This research question seeks to explore the role of technological changes that occur when direct alloy recycling develops from lab-scale and pilot-scale to industrial scale. By assessing various anticipated technological advancements and recycling routes, the potential for environmental impact reduction can be revealed.

The approach to answering RQ 3 builds upon the established LCA methodology, which allows to quantify the environmental impacts along the life cycle of a product, process or service. LCA studies quantify the contribution of a product's life cycle to global environmental issues, using a set of impact assessment methods (Guinée et al., 2002). Conducting an *ex-ante* LCA, i.e. before a technology is fully developed and implemented, allows to anticipate improvements in sustainability performance and identify areas of concern, guiding the technology's further R&D trajectory (van der Giesen et al., 2020). Therefore, ex-ante LCA is relevant for studying the developing magnet recycling technology. We aim for a detailed outlook by incorporating the latest information on technology performance from developers.

To conduct the ex-ante LCA, we can refer to existing guidelines. These guidelines provide general procedures, despite the wide variety of technological domains. Some approaches focus on inventory modelling (Tsoy et al., 2019), while others take a broader perspective (Langkau et al., 2023). The guidelines implicitly or explicitly draw from other forward-looking methods, such as cost projections, scenario building, and technology forecasting (Cho & Daim, 2013; Langkau et al., 2023). This research used scenario thinking and various unit process modeling techniques to explore the future impacts of magnet recycling.

Previous ex-ante LCA studies have primarily focused on the future technology performance (Jin et al., 2018a; Sprecher et al., 2014; Wang et al., 2022). It is much less explored how the environmental impacts would evolve during the technology development from small-scale to industrial scale. Such analysis would enhance the usefulness of ex-ante LCA, by explicitly highlighting the gap between small-scale and large-scale technology, and by indentifying the most effective improvements. We have pursued this approach, following the example of van der Hulst et al. (2020). Scenarios in ex-ante LCA typically consider external developments. This study only briefly touches on external developments, as extensively researched by my colleague (Miranda Xicotencatl et al., 2023).

1.6 Anticipating environmental impact trends after commercial introduction

Next to the development from lab-scale and pilot-scale to industrial recycling, it would be great to extend the outlook to development during industrial operation. Current ex-ante LCA approaches help to explore future technology performance. It would be great to also have insight in the time it takes to reach anticipated future performance levels. While early technology innovation is unpredictable, research has shown that after the initial industrial introduction, costs follow a predictable trend, described by a learning curve. This inspired an investigation of the effect of learning on environmental impacts.

Learning plays a key role in the improvement of mature industrial processes (Feeney et al., 2023), therefore LCAs should consider learning when technologies are studied beyond their first commercial introduction. In their definition of ex-ante LCA, Cucurachi et al. (2018) refer to learning curves as a possible approach to derive technology development scenarios. Some studies have modeled environmental learning (Thomassen et al., 2020; van der Hulst et al., 2020) on the assumption that cost-based learning trends can be applied to the environmental domain. However, this assertion is not supported by, for example, identifying the drivers or mechanisms of learning. Furthermore, the possibility that learning may have different implications for different impact categories has largely been disregarded, with only one study indicating differentiated trends (Stamford & Azapagic, 2018). In short, there is a lack of conceptual underpinning and robust methods for the applicaton of environmental learning curves. This gap is addressed by RQ 4.

RQ 4. *How does technological learning affect environmental impacts, and how can LCA studies address this effect?*

By developing a method to assess environmental learning, this research aims to enable temporally-explicit outlooks in ex-ante LCA. The procedure is developed based on organizational learning theory, eco-innovations literature, evidence from case studies, and cost forecasting methods. The latter also inspired the use of contingency factors to improve future impact projections (see §5.3.6).

Thus far, learning curves have mainly been used in environmental sciences to study energy technologies, resulting in a focus on costs and CO_2 emissions. This contributed to insight into the development of renewable energy technologies, including their costs and efficiency. However, RQ 4 seeks to broaden this scope by aiming for a methodology that addresses all environmental impact categories. This extension aims to broaden the applicability of learning curves to a wider range of technologies.

1.7 Outline

Research questions 1–4 are addressed by the four ensuing chapters of this thesis. To gain insight into magnet recycling technologies and recycling systems at different levels, including opportunities for improvement, several complementary research methods are applied.

Chapter 2 addresses RQ 1 by conducting a literature review on factors that determine the recyclability of minor metals. The outcomes of the review are used to develop a framework for recyclability assessment, including indicators that enable quantitative comparisons.

Chapter 3 explores RQ 2 using a dynamic MFA approach to quantify the demand and waste flows of neodymium contained in magnets. This analysis provides insight into the composition of waste flows and their country of origin. A recyclability assessment provides additional information on the suitability of EoL products for recycling.

Chapter 4 delves into RQ 3 by determining the environmental imapacts of magnet recycling routes, for small-scale and for potential industrial-scale configurations. This chapter applies ex-ante LCA to identify hotspots of environmental impacts in the recycling chain, and compares recycled to primary magnets. Furthermore, the effect of techological progress is systematically assessed by grouping the different development mechanisms.

Chapter 5 addresses RQ 4, by presenting a method to account for learning effects in prospective LCA. This method builds on a review of organizational learning theories and evidence on environmental learning. These insights help to assess whether learning curves apply and to estimate the learning rate.

Chapter 6 synthesizes the main findings of Chapter 2–5. This final chapter answers the main research question by summarizing the environmental consequences of deploying large-scale NdFeB magnet recycling. It also provides a reflection on this research and its implications for the future.

Data availability

This thesis contains four appendices, corresponding to each of the main chapters and containing supplementary information and figures. Some supplementary data tables are unfit for printing. For these tables, the reader is referred to the Electronic Supplementary Information (ESI) on the webpage of the published article.

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2 Early-stage assessment of minor metal recyclability

This chapter has been published as: Van Nielen, S.S., Kleijn, R., Sprecher, B., Miranda Xicotencatl, B., Tukker, A., (2022). Early-stage assessment of minor metal recyclability. *Resources, Conservation and Recycling* 176, 105881. doi:10.1016/J.ResConRec.2021.105881.

Abstract

The growing demand for minor metals creates both pressure on the supply chain of these metals and challenges in waste management. Consequently, there is a wide interest in recycling opportunities. To identify these opportunities, it is key to understand bottlenecks and drivers in recycling value chains. Hence, we analyzed existing recyclability frameworks and related recycling literature, revealing 113 factors that determine the success of recycling minor metals. These factors were linked to the stages of the recycling value chain, i.e. manufacturing, use phase, waste collection, preprocessing, metallurgical recovery and secondary marketing. Based on the insights from the literature analysis, we propose a novel recyclability assessment framework. The framework indicates how properties of products, recycling technologies and society determine recyclability. The framework is suitable for assessing the recyclability of minor metals during the recycling technology development process. Therefore, it includes indicators that can be quantified easily, as demonstrated in three case studies. As such, it can be a useful tool to guide policy makers and technology developers towards closing material loops.

2.1 Introduction

2.1.1 Supply challenges of minor metals

Modern societies are expecting serious challenges in meeting future demand for minor metals, a group of metals that includes rare earth elements, precious metals, and specialty metals. The minor metals have in common a small market size compared to that of base metals, and they are often critical for modern technologies. This dependence is increasing, among others due to the rapid adoption of energy-efficient appliances and renewable energy technologies (Tercero Espinoza et al., 2019). Therefore, minor metals are often regarded as critical raw materials. In contrast with this demand-side growth, the production chain has limited possibilities for upscaling, because many minor metals are mined as by-products of base metals (Nassar et al., 2015). Moreover, the elevated environmental burdens associated with mining and refining many minor metals can provoke resistance to upscaling from local communities and governments (Conde, 2017) while inadequate waste treatment raises equally large concerns (UN, 2019).

It is widely recognized that for a sustainable supply of minor metals, recycling is essential (Reck & Graedel, 2012; Rombach & Friedrich, 2014). Recycling increases the availability of raw materials and the environmental impacts are often lower than those of mining (Mathieux et al., 2018). Nevertheless, most minor metals currently have very low recycling rates (Graedel et al., 2011; Mathieux et al., 2018). To close the gap between actual and potential recycling, a range of policy measures is available, targeting various stages of the recycling chain. These stages include manufacturing, collection, preprocessing, recovery and the secondary market. What policy interventions are effective is material-specific (Hagelüken et al., 2016) and requires knowledge of each element in the value chain. Studies of success factors and barriers for recycling are discussed in the next section.

2.1.2 Assessing recyclability

Recyclability has been investigated in various contexts. For metals in general, materials and WEEE, recyclability assessment methods have been proposed (Johnson et al., 2007; Mueller et al., 2017; Oguchi et al., 2011; Phillis et al., 2005; Sun et al., 2016; Villalba et al., 2002; Winterstetter et al., 2016; Zeng & Li, 2016). These methods typically address one or more aspects that enable or inhibit recycling, using quantitative or qualitative indicators. We are unaware of methods focusing on minor metals. Yet, the body of related literature shows that different approaches are possible and indicates methodological challenges. The differences in recyclability assessment methods appear to stem from different perspectives held by each actor in the value chain. Below, the perspectives that occur in the literature are discussed.

Before reviewing these studies, it is instructive to first define recyclability. As one of the first, Henstock (1988) introduced the concept of recyclability, which he defined as the technical ease and economic feasibility of recovering materials from products that would otherwise enter the waste stream. In contrast, Huisman et al. (2003) defined the concept from an environmental point of view, describing the extent to which a product's recycling can reduce environmental impacts. Zeng & Li (2016) give a thermodynamics-oriented definition based on statistical entropy. From a designer perspective, recyclability indicates whether product design facilitates recycling (Chen et al., 1994). In this paper, we adopts the definition proposed by Henstock, while explicitly considering all stages in the recycling value chain.

A macro-level insight into recyclability can be obtained using thermodynamic (Anctil & Fthenakis, 2013; Johnson et al., 2007; Zeng & Li, 2016) or economic (Villalba et al., 2002) indicators. These approaches are based on the principle that lower metal concentrations require more purification efforts (Reuter et al., 2013). This is a product-centric view (Reuter et al., 2013), which does not consider organizational aspects or the characteristics of recycling technologies.

Arguing from a waste collection perspective, Oguchi et al. (2011) categorized waste flows based on the number of products and product size. This aligns with the view of waste collection companies, who consider the product properties as a given. In contrast, products can be optimized for recycling from the designer's point of view. Product properties that hinder the preprocessing or recovery process can be formulated as design guidelines. This approach was taken by Hultgren (2012) and resulted in a set of design for recyclability (DfR) guidelines. A manufacturer can apply these guidelines as part of a circular business model, in which successful recycling yields financial benefits.

Other researchers based their approach on the similarity between geologic and technospheric resource mining (Mueller et al., 2017; Winterstetter et al., 2016). Both frameworks are intended as a tool in pre-feasibility or feasibility studies. This analogy was most thoroughly elaborated by Winterstetter et al. (2016), who adapted the UNFC-2009 framework to evaluate and classify secondary resources. The perspective adopted by Winterstetter et al. (2016) is that of a recycling company, seeking exploitable secondary resources. From this perspective, legislation and collection systems are seen as external factors. Here, the core factors relate to business economics, and support in-between go/no-go decisions. Both frameworks (Mueller et al., 2017; Winterstetter et al., 2016) comprise an elaborate set of aspects, while identifying limited data availability as a challenge for quantitative comparisons.

A gap in literature, addressed in this study, becomes apparent from the difference between technical and social studies. Papers with a technical perspective address aspects of preprocessing and recovery processes (e.g. Sun et al., 2016). This view emphasizes metallurgical and material-related aspects. Two technical approaches can be distinguished: material-centric and product-centric (Reuter et al., 2013). The material-centric approach focusses on the recycling rates of single materials, whereas the product-centric approach optimizes the recycling of all materials found in a product. In contrast, papers with a sociological approach argue that recyclability also depends on the societal setting (Gusmeroli, 2017; Lapko et al., 2019). Social aspects include behavior, networks and expectations of actors throughout the recycling value chain. Both the technology-centered view and the organizational perspective describe certain aspects of the recycling system. Sometimes, a holistic view is desired (Lapko et al., 2019), covering both technical and organizational aspects over the whole value chain.

2.1.3 Towards a novel framework for minor metals

Against the background of diverging perspectives on recyclability, the objective of this study is to provide a structured overview of the field. This overview highlights the applicability of previous findings to minor metals. Based on these insights, a novel framework is presented to conceptualize and assess the recyclability of minor metals. The framework aims to answer the question which economic, technical, and societal factors affect the recycling of minor metals. These insights are relevant to industrial actors throughout the product life cycle. Moreover, the framework supports policy makers to identify bottlenecks in the recycling chain as targets for circular economy policies.

As part of the proposed framework, this study provides a set of indicators for recyclability. Since the recycling industry for most minor metals is nonexistent or underdeveloped, the framework focuses on providing guidance in the pre-feasibility phase. At that stage, many alternatives are typically available while detailed information is limited. Therefore, simple indicators were preferred over rigorous evaluation methods. The framework addresses recyclability in the scope of developed countries.

2.2 Methods

The approach of this study builds on the insights from previous research to obtain a complete and consistent framework. This recyclability assessment framework was developed in three steps, i.e. the collection of relevant literature, the identification of aspects of recyclability, and the construction of a novel framework.

The first step aims to create an overview of recyclability research. In this step, available literature was collected using Google Scholar and the following initial search queries:

- (Recyclability OR "recycling potential") AND ("minor metal" OR "critical metal")
- ("Urban mining" OR "secondary resource") AND (classification OR indicators) AND framework
- Recycling AND (challenges OR issues OR barriers OR "success factors")
- ("design for recycling" OR "design for recyclability") AND metal*

The identified articles were screened on their relevance based on the title and abstract. We excluded articles that did not study recycling of metals or did not focus on challenges and success factors. After screening, 15 articles remained. This initial list was extended through the snowballing technique, i.e. by checking the references of articles yielded by the initial search. The resulting list comprised 25 articles on which the systematic literature analysis was based.

Using the iterative literature collection approach, we identified both articles that address recyclability explicitly, and articles that explore barriers and drivers for recycling in general. Note that several papers have a broader scope than minor metals only. The articles can be grouped into four typologies based on their approach:

4 overview articles discussing the recycling system and its challenges: Lundgren (2012); Reuter et al. (2013); Tansel (2017); Tanskanen (2013)

12 articles introducing a framework or indicators for recyclability: Habib (2019); Johnson et al. (2007); Lapko et al. (2019); Mueller et al. (2017); Oguchi et al. (2011); Phillis et al. (2005); Sun et al. (2016); Ueberschaar et al. (2017); Villalba et al. (2002); Winterstetter et al. (2016); Zeng & Li (2016); Zuo et al. (2019)

6 case studies of recycling: Anctil & Fthenakis (2013); Burkhardt et al. (2020); Gusmeroli (2017); Hagelüken (2012); REMANENCE (2017); Salim et al. (2019)

3 design-oriented approaches to recycling: Chen et al. (1994); Huisman et al. (2003); Hultgren (2012)

In the second step, the collected literature was summarized using a coding approach. This approach involved the listing of barriers, success factors and challenges for recycling as described in the articles. Similar phrasings were categorized as a common aspect. For articles describing a framework, we listed the aspects and indicators that were addressed by the framework. All coded aspects were categorized based on the stage of the value chain in which they appear. After coding, the occurrence frequency of aspects was counted to create an overview of their coverage in literature, as presented in §2.3.1. This section also discusses the identified aspects further.

The overview of aspects was the starting point for the final step, compiling a new framework. In the framework, the identified aspects are structured in clusters of factors. From groups of closely related aspects, the most relevant one was included to avoid repetition. For instance, 'ownership' was added while narrower 'ownership shifts' was not. Each factor was linked to one or more indicators. Indicators were derived as much as possible from the literature. When no corresponding indicator was found for a factor, a new indicator was added. New indicators were chosen to connect closely to the factors.

Once the framework was established, we tested its usefulness by applying it to three case studies. The cases were selected to reflect the diversity in the development of recycling chains: platinum from car catalysts (operational recycling), neodymium from TV speaker magnets (first industrial pilots), and indium from LCD screens (research stage).

2.3 Aspect identification and quantification

2.3.1 Aspect identification

A total of 113 distinct aspects were identified in the literature, all of which are listed in Appendix A.1. The aspects are clustered per stage of the value chain, thereby revealing how the research focus is distributed. In the analysis below, we addressed the occurrence frequency of aspects, defined as the sum of the number of unique aspects per paper.

In the reviewed articles, some value chain stages have received more attention than others. This is observed from Figure 2.1 and 2.2, both counting the occurrence frequency of aspects. Overarching aspects, not related to a particular stage, occur with high frequency. Besides, many aspects refer to the stages of collection and recycling. Possible reasons why these three stages are overrepresented are the scope of the studies or that these stages are most critical for recyclability. Either way, a focus on these stages only would give an incomplete picture.

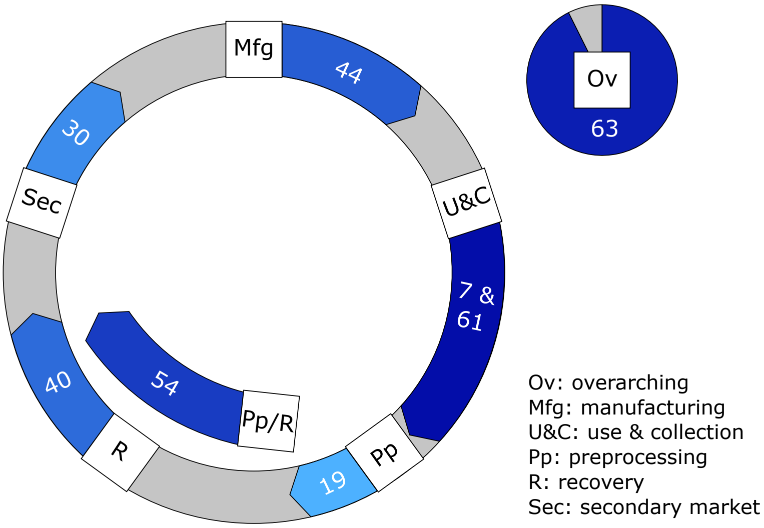


FIGURE 2.1: Overview of aspects identified in the literature analysis. For each value chain stage, the occurrence frequency of aspects is indicated. Aspects that could not be linked to either the preprocessing or recovery process in recycling are counted under 'Pp/R'.

A further insight is provided by Figure 2.2, with a division of aspects by publication type. The sets of framework and overview articles are similar in their focus, except that frameworks more often describe secondary market aspects. Frameworks address relatively few aspects of manufacturing and preprocessing. In contrast, design-oriented studies mainly focus on manufacturing and the links to preprocessing and recovery processes. Unsurprisingly, the four overview articles address a large number of aspects. In the three design-oriented articles, the occurrence frequency totaled only 17 aspects.

As highlighted in §2.1.2, recyclability can be approached from different angles, such as the business economic, thermodynamic, sustainability and policy perspective. Each angle yields different aspects and indicators, the major groups of which are discussed below.

2.3.1.1 Business economics

A frequently mentioned aspect is profitability. Moreover, eleven identified concepts are cost components. In liberal market economies, profit is indeed a dominant driving force and a profitable recycling process indicates a good recyclability. The result of a profit calculation depends on the scope of the business case. We distinguish between the business case of preprocessing and of recovery. Profitability is also a consideration for take-back schemes, when these are implemented by manufacturers under extended producer responsibility (EPR) frameworks. Consumers consider the financial aspects of their waste disposal options. In short, profitability is a relevant aspect of which is linked to various stakeholders.

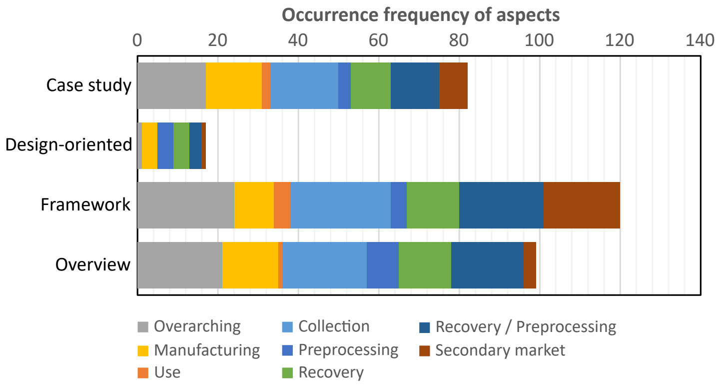


FIGURE 2.2: Overview of aspects, per value chain stage and publication type.

Although monetary indicators are suitable for aggregation and comparison, their quantification faces challenges. To quantify business cases, an elaborate financial assessment is required. Yet for recycling routes under development, the costs are unknown, uncertain or unsettled, complicating their assessment. For example, this challenge was encountered in a case study by Mueller et al. (2017).

2.3.1.2 Thermodynamics

An important barrier for the recycling of minor metals is their minor weight fraction in most products. This effect is exacerbated by the trend of miniaturization of components. A simple indicator for this effect is the metal concentration, as advocated by several studies (Habib, 2019; Johnson et al., 2007; Mueller et al., 2017; Oguchi et al., 2011; Winterstetter et al., 2016; Zuo et al., 2019). The concentration affects the recyclability in two ways: it increases the efforts needed to purify the metal and it increases the throughput required to recover a given mass of metal.

A related barrier is the complexity of products, which can be assessed with thermodynamic indicators for entropy. Products composed of many materials and metals require complex treatment processes, as the constituting materials are difficult to identify and their separation might require multiple steps. The literature analysis revealed that product complexity can quantified using statistical entropy, a concept originating from information theory (Rechberger & Brunner, 2002). This indicator is applied in recyclability assessments by Zeng & Li (2016) and Anctil & Fthenakis (2013). Statistical entropy analysis is also applicable in material flow analysis (MFA) studies to reveal how processes affect the dilution of materials (Rechberger & Brunner, 2002; Thiébaud et al., 2018).

Thermodynamic insights also form the basis for the element radar chart (Reuter et al., 2013). This tool indicates which metals can be separated from each other using metallurgical process, thereby identifying recoverable metals and contaminants. If contaminants accumulate in a metal or alloy, degradation can occur. An indicator for recyclability proposed by Villalba et al. (2002) is devaluation, because it reflects the reduced material value due to degradation.

2.3.1.3 Design

The 12 recyclability assessment frameworks that were reviewed in §2.1.2 did not address aspects related to product design and manufacturing. However, one fifth of all the analyzed articles stress that successful recycling depends on design (Burkhardt et al., 2020; Lundgren, 2012; Reuter et al., 2013; Salim et al., 2019; Tanskanen, 2013; Winterstetter et al., 2016). When manufacturers anticipate on the end-of-life (EoL) phase of their products, they apply eco-design or design for recycling, recyclability or disassembly. To help manufacturers in this process, several guidelines exist (Hultgren, 2012). These guidelines are formulated as design principles and do not indicate when DfR is considered successful, thus complicating the definition of indicators.

Nonetheless, an aspect that is fit for recyclability assessment are the joints (connections between materials and components) (Chen et al., 1994; Hultgren, 2012), because dismantling studies have revealed how joints influence the dismantling time (Desai & Mital, 2003; Kondo et al., 2003; Vanegas et al., 2018). Joints can be accounted for by classifying their separability (Reuter et al., 2013), or by counting their number or their dismantling times (Chen et al., 1994).

Another design-related aspect is the design variation of products and components. The product design variation was mentioned in multiple studies, some of which underlined the heterogeneity of products (Johnson et al., 2007; Lapko et al., 2019; REMANENCE, 2017; Reuter et al., 2013; Tanskanen, 2013) while others highlighted design changes over time (Salim et al., 2019; Ueberschaar et al., 2017). The design variation of components was only stated by Tansel (2017).

Some design guidelines aim to reduce the application of critical materials, e.g. by substitution (Chen et al., 1994). The benefit of these strategies is a reduced content of the metal in products or waste flows. At the same time, the recyclability of the remaining metal is reduced.

2.3.1.4 Uncertainty

Several studies identified aspects of recyclability that link to uncertainty, in particular uncertainty about the waste flow. The literature analysis shows that recycling requires knowledge about multiple aspects of waste: its composition (Lundgren, 2012; Reuter et al., 2013; Salim et al., 2019), its volume (Reuter et al., 2013), and product lifespan (Habib, 2019; Salim et al., 2019). Mueller et al. (2017) proposed an indicator for uncertainty, namely the confidence level of future flows.

From an organizational perspective, uncertainty can be reduced when stakeholders engage in information exchange (Gusmeroli, 2017; Lapko et al., 2019; Reuter et al., 2013; Salim et al., 2019; Tanskanen, 2013). Exchanges create more transparency and increase access to information, hence supporting recycling and well-informed decision making.

2.3.1.5 Social impacts

In all stages of the value chain, negative social effects can occur. More specifically, seven of the papers identify the issue of worker health hazards, caused by toxic substances in waste and in process chemicals (Chen et al., 1994; Hultgren, 2012; Lundgren, 2012;

Mueller et al., 2017; Salim et al., 2019; Tansel, 2017; Tanskanen, 2013). The risk level of substances is indicated by regulatory lists and limits, such as the European Union (EU) RoHS and REACH Directives (Hultgren, 2012; Lundgren, 2012). These hazards are critical in informal or illegal recycling (mostly in countries where the waste was not originally generated) (Huisman et al., 2015; Hultgren, 2012) and in virgin mining and refining operations. In both cases, inadequate worker protection is an issue. Although the labor conditions vary per company, the status per country can be estimated based on the human rights conditions or labor rights conditions (Lundgren, 2012; Mueller et al., 2017).

2.3.1.6 Environmental impacts

Recycling is often advocated for environmental reasons, and accordingly several aspects relate to the environment. The reviewed studies address four categories of environmental impacts: climate change (Mueller et al., 2017; Salim et al., 2019; Tanskanen, 2013), human health impacts (Lundgren, 2012; Mueller et al., 2017; Tansel, 2017; Tanskanen, 2013), resource depletion (Mueller et al., 2017; Salim et al., 2019; Zuo et al., 2019), and ecosystem degradation (Mueller et al., 2017; Salim et al., 2019; Ueberschaar et al., 2017). Climate change and resource depletion receive wide attention globally. Toxicity impacts are a specific concern in metallurgical processes, both mining and recycling (Lundgren, 2012).

The overall environmental performance is quantifiable with the life cycle assessment (LCA) methodology if sufficient process details are known. The related QWERTY approach compares the environmental impact of recycling (including primary material substitution benefits) with a worst-case alternative (Huisman et al., 2003). Alternatively, some studies quantify a specific aspect such as energy consumption (Zuo et al., 2019) or greenhouse gas (GHG) emissions (Salim et al., 2019).

2.3.1.7 Policy

Recycling is affected by various regulations, and recycling-friendly policies can remove barriers. Several policies aim to limit the social and environmental impacts discussed above, for example by moderating the export of waste (Reuter et al., 2013). Besides, some studies identify the policy option of economic incentives (Phillis et al., 2005; Reuter et al., 2013; Salim et al., 2019). Other policies address the waste collection process, which can be approached in several ways. Examples encountered in literature include take-back requirements (Lapko et al., 2019), recycling incentives (Gusmeroli, 2017), waste collection schemes (Winterstetter et al., 2016), EPR legislation (Gusmeroli, 2017; Lundgren, 2012; Reuter et al., 2013; Salim et al., 2019; Tanskanen, 2013) or the active implementation of a collection infrastructure.

When policies are created, several strategic considerations can play a role: the economic importance of the metal (Zuo et al., 2019), its supply risk (Sun et al., 2016), worker protection (Lundgren, 2012), or the employment effects (Salim et al., 2019). The former two aspects together determine the metal criticality, an important concept in resource policies (Schrijvers et al., 2020). When a metal is labeled as critical, Sun et al. (2016) and

Zuo et al. (2019) argue, its recycling is prioritized. In short, policies are shaped directly by political priorities and indirectly by the characteristics of the metal supply chain.

2.3.2 Aspect quantification and aggregation

A key feature of most frameworks is the use of quantitative indicators. In the 12 articles proposing a framework, 7 frameworks are centered around quantitative indicators. These indicators are for example metal concentration or product weight. Some indicators are derived with a calculation, e.g. ‘material grade’ (Zeng & Li, 2016) or environmental impact (Huisman et al., 2003). For both quantitative and qualitative frameworks, a more elaborate set of indicators can address more perspectives from §2.1.2, hence providing a more nuanced view.

Quantitative indicators are useful for comparisons and screening of waste flows. In comparisons, the reviewed studies interpret the indicator values relative to those of other waste flows. For example, Sun et al. (2016) assessed 11 waste flows to identify recovery opportunities for the recycling industry. A general challenge is the limited availability of quantitative information sources.

Whether aggregation is favored depends on the number of indicators. In frameworks with one or two quantitative indicators, no aggregation is applied; the results are conveniently presented in a scatter plot (Anctil & Fthenakis, 2013; Johnson et al., 2007; Oguchi et al., 2011; Villalba et al., 2002; Zeng & Li, 2016). When a framework comprises multiple indicators, their interpretation is often facilitated by grouping. Some authors combined all indicators into a single score, while others used two or three dimensions for grouping. Such aggregation was encountered in frameworks based on quantitative indicators. For instance, Sun et al. (2016) aggregate their indicators in a resource and a technology index. The framework by Sun et al. was adapted by Zuo et al. (2019), who added an environmental index. Mueller et al. (2017) defined accessibility as the overarching indicator, which aggregates multiple categories and subcategories.

Likewise, the qualitative frameworks also exhibit different structures. Some do not categorize their indicators (Gusmeroli, 2017; Lapko et al., 2019), whereas two frameworks define groups of related indicators (Ueberschaar et al., 2017; Winterstetter et al., 2016). Ueberschaar et al. (2017) based the structure of their framework on the aspects that the indicators relate to: the product, the recycling chain, and the economy. Winterstetter et al. (2016) distinguish between geological knowledge, technical feasibility and socioeconomic viability as a basis for qualitative classification. Although grouping gives a handhold for interpretation, no objective measure for a good structure exists.

When aggregating quantitative indicators, the need for weighting factors arises. Mueller et al. (2017) apply equal weight to all aspects, while recommending a more substantiated refinement. Burkhardt et al. (2020) introduce a multiplication factor for each aspect, based on expert judgement. Both Burkhardt et al. (2020) and Sun et al. (2016) calculate the product (rather than the sum) of indicators. This approach has two advantages. Indicators that are absolutely prohibitive for recycling can be assigned a value of zero, resulting in the minimum final score. Besides, weighting is only needed at the level of final scores. A drawback of multiplication is the nonlinear relation between the product and individual indicator values.

2.4 Proposed framework

2.4.1 Structure

Building on the insights from §2.3.1, we established a new framework. The framework structure is based on the recycling value chain stages, as shown in Figure 2.3. This visualization is inspired by the metal cycle as introduced by Reck & Rotter (2012), who mapped the anthropogenic metal flows. Figure 2.3 schematically represents the supply chain stages and their contribution to a metal's recyclability. Each stage features one or more aspects whose performance is scored using underlying indicators. In the diagram, the scores are indicated with a color gradient.

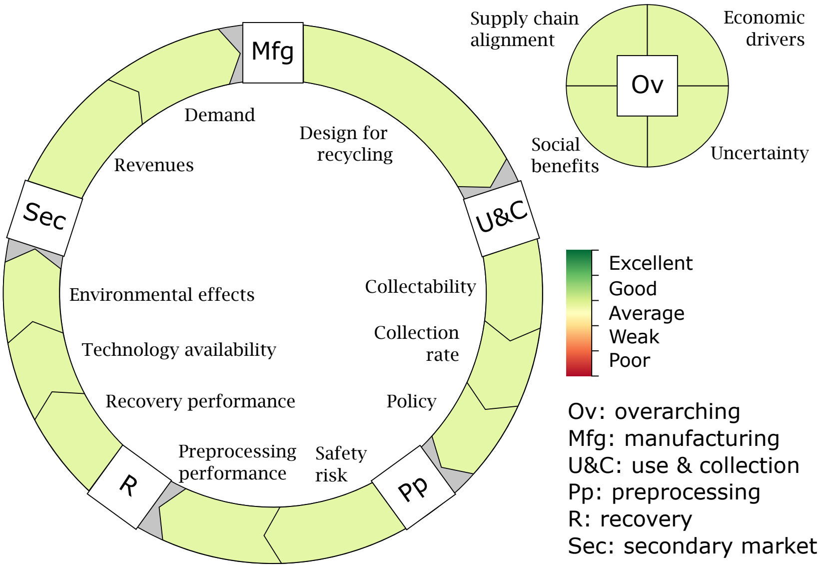


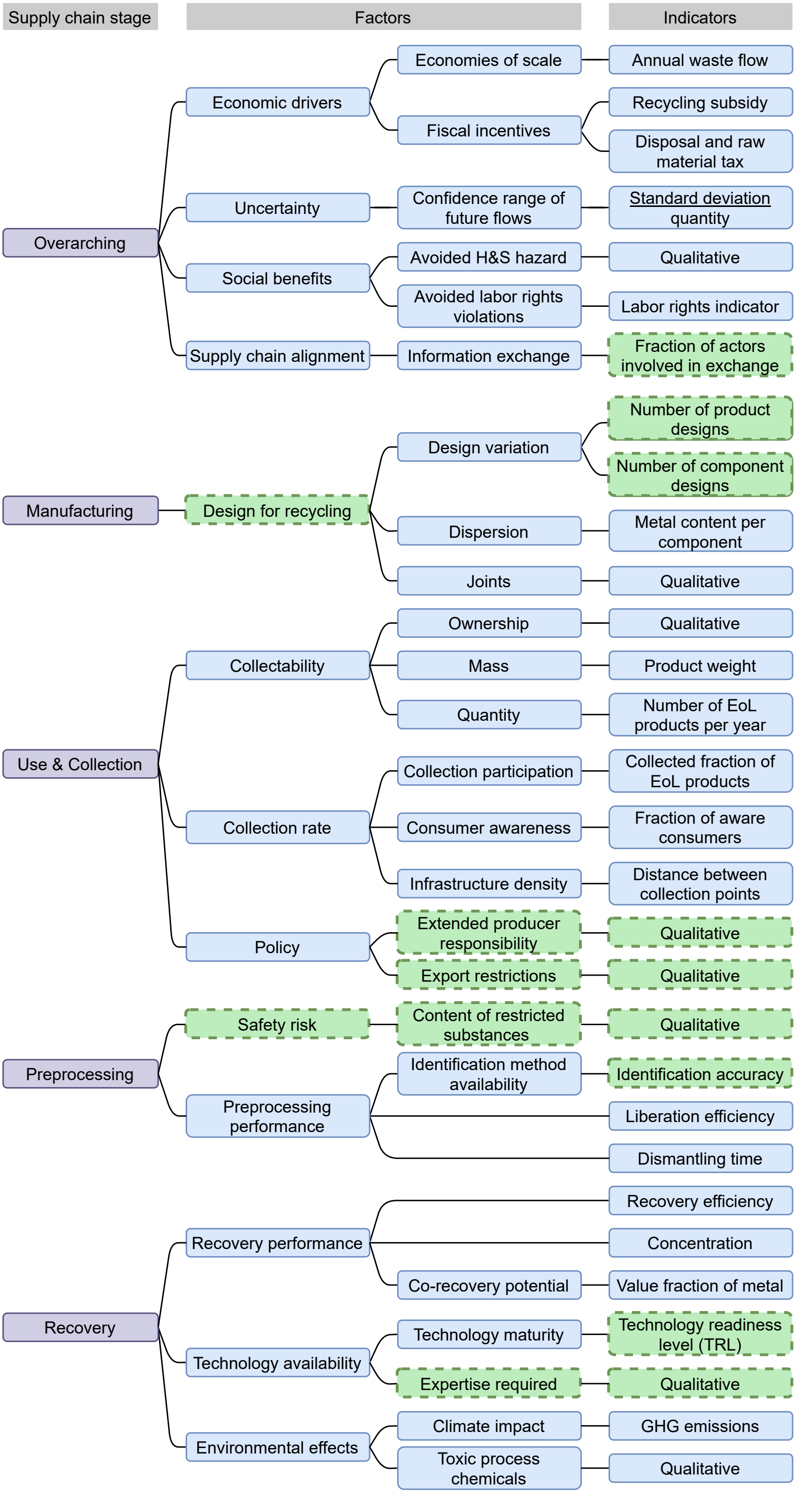
FIGURE 2.3: The framework structure, matching the stages of the recycling value chain.

The framework, including factors and indicators, is displayed in Figure 2.4. By means of a dashed outline, the diagram highlights factors and indicators that have not been used before in recyclability frameworks. In §2.4.2, the underlying factors are described, while §2.4.3 presents quantitative aspects of the approach.

2.4.2 Factors & Indicators

Overarching: A number of factors are not connected to any of the value chain stages in particular; these are classified as overarching factors. Two overarching factors, economic drivers and uncertainty, are both essential for making investment decisions. The other factors included here are the broader social benefits and supply chain alignment.

- The first of two economic drivers is economies of scale, which is measured as the mass of the annual waste flow. Transport costs and sorting capacity scale with this


Supply chain stage
Factors
Indicators
Overarching
Economic drivers
Economies of scale
Annual waste flow
Fiscal incentives
Recycling subsidy
Disposal and raw material tax
Uncertainty
Confidence range of future flows
Standard deviation quantity
Social benefits
Avoided H&S hazard
Qualitative
Avoided labor rights violations
Labor rights indicator
Supply chain alignment
Information exchange
Fraction of actors involved in exchange
Manufacturing
Design for recycling
Design variation
Number of product designs
Number of component designs
Dispersion
Metal content per component
Joints
Qualitative
Use & Collection
Collectability
Ownership
Qualitative
Mass
Product weight
Quantity
Number of EoL products per year
Collection rate
Collection participation
Collected fraction of EoL products
Consumer awareness
Fraction of aware consumers
Infrastructure density
Distance between collection points
Policy
Extended producer responsibility
Qualitative
Export restrictions
Qualitative
Preprocessing
Safety risk
Content of restricted substances
Qualitative
Preprocessing performance
Identification method availability
Identification accuracy
Liberation efficiency
Dismantling time
Recovery
Recovery performance
Recovery efficiency
Concentration
Co-recovery potential
Value fraction of metal
Technology availability
Technology maturity
Technology readiness level (TRL)
Expertise required
Qualitative
Environmental effects
Climate impact
GHG emissions
Toxic process chemicals
Qualitative

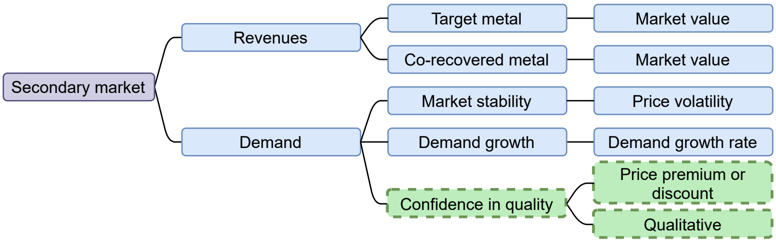


FIGURE 2.4: The recyclability assessment framework and its structure, clustering by value chain stages. New indicators—not found in existing frameworks—are indicated by a dashed outline. H&S: Health and safety; GHG: greenhouse gas.

indicator. Metallurgical recovery plants require a throughput of about 100 kt/a (Reuter et al., 2013). Large waste volumes could also be beneficial for other processes throughout the recycling value chain, through enhanced learning effects. A second economic driver are fiscal incentives, i.e. policies that increase the profitability of recycling compared to alternatives. Possible incentives are subsidies, or taxes on waste disposal and virgin material use.

- The major uncertainty of future flows is indicated by the standard deviation of the estimated waste quantity. For investments with a longer payback period, the uncertainty of flows in the further future is relevant. Therefore, the uncertainty up to the end of the investment horizon of recyclers is considered.

- Social benefits address the labor conditions along the primary metal supply chain. Issues arising here can incentivize recycling. On the one hand, incidents correlate with the health and safety hazard of the current practice, which is judged based on hazardous substances. On the other hand, incidents arise in poor labor rights conditions, as quantified by the labor rights indicator (Kucera & Sari, 2020) of the country that hosts the waste treatment or mining process.

- Supply chain alignment refers to the extent to which actors cooperate and engage in collective planning. This social factor is captured by information exchange, which refers to information relevant to recycling. It is quantified as the fraction of actors involved in exchanges.

Manufacturing: The manufacturer of a product plays a role by making design decisions that affect dismantling. This role is expressed in design for recycling, which comprises three important design parameters.

- Design variation is included because it influences how sophisticated the sorting and dismantling process should be. Heterogeneous waste flows are more difficult to process. The variation is assessed using two indicators, for product design variation and for component design variation. These indicators reflect the number of different designs of products and components available on the market. For certain products, these indicators can be connected.

- Dispersion refers to miniaturization and the small volumes of metal in products. The indicator used here is the metal content per component.
- The type of joints affect the dismantling effort. A preference order is used to rank the joints (based on Kondo et al., 2003; van Schaik & Reuter, 2012) from least to most preferable: Coating, paint, adhesive, glue, screws, encasing, plugs, snap fitting.

Use & Collection: The use and collection phase are closely linked because in both phases, the product owners play a key role. Their contribution to successful loop closure is expressed by the collection rate. Two factors that support collection are included as well: collectability (the ease of collection) and policy.

- Three product-related aspects are included to address the collectability: ownership, product weight and quantity. The type of product owner determines feasible collection network structures. When the manufacturer remains owner of the product in a service contract structure, their responsibility for EoL collection increases. When the users are companies, the collection process is relatively simple (Knemeyer et al., 2002), in particular for low numbers of companies. In the case of consumers, collection depends on the presence of a visible and extensive collection infrastructure, the creation of which requires significant efforts (Tanskanen, 2013). The structure of the collection network is largely determined by the combination of product weight and quantity. Heavy products are less likely to be disposed of incorrectly by consumers (Oguchi et al., 2011). For products that are disposed frequently, the investment in collection facilities is lower per product (Tanskanen, 2013).
- The collection rate depends on the collection participation, as indicated by the fraction of products collected. The prospects of future collection are based on consumer awareness and infrastructure density. Awareness is indicated by the fraction of consumers that is aware of separate waste collection infrastructure. The infrastructure density is characterized by the distance between collection points. These two indicators are mostly applicable to consumer goods.
- Collection is affected by policies, of which two major types are included here. EPR is a scheme that makes manufacturers responsible for correct collection and treatment of their products. Export restrictions aim to prevent undesired waste export to countries that lack effective recycling facilities. These restrictions are most effective in combination with law enforcement to counteract illegal export (Huisman et al., 2015).

Preprocessing: This stage comprises all manual and mechanical processing steps in a sorting or recycling plant. It is characterized by its safety risk and its technical performance.

- The safety risk is indicated by the content of restricted substances in waste. Toxic or harmful substances create H&S risks, in particular in the preprocessing stage.

This calls for safety measures to protect workers. In addition, safety regulations (RoHS and REACH in the EU) demand that a recycling company invests in e.g. certification and permitting procedures.

- The preprocessing performance is assessed by three indicators. First, the identification accuracy indicates the maturity and availability of methods to identify components or devices in the waste flow. This is an essential preprocessing step when only specific components contain the metal of interest (Burkhardt et al., 2020; Habib, 2019; Ueberschaar et al., 2017). The second indicator is the liberation efficiency, i.e. the fraction of target metal that enters the recovery process. A trade-off exists between the liberation efficiency, purity after liberation and preprocessing efforts (Reuter et al., 2013). Third, dismantling needs labor inputs from either humans or robots. Both can have a substantial effect on the economic feasibility. A good indicator, even in a pilot phase, is the dismantling time, because it indicates the complexity of the task.

Recovery: Recovery refers to the metallurgical processes that separate and purify the metal of interest. Similar to preprocessing, also here one of the factors is technical performance. In addition, the technology availability and the environmental performance are considered.

- The performance of the recovery is indicated by its efficiency, which can be quantified in lab or pilot experiments. Besides, an indicator is included to address the performance potential, i.e. the metal concentration in the recovery input (after preprocessing). When the concentration is low, high recovery is either impossible or costly. A third factor of performance is the potential co-recovery of metals within a product, which can be assessed using an element radar chart (Reuter et al., 2013). This tool indicates which metals are recoverable in each metallurgical process. An indicator for co-recovery is the fraction of metals by value that have compatible extraction. This accounts for product compositions and differences in metal values (Reuter et al., 2013; Zuo et al., 2019).
- In particular for recovery, sufficiently developed technologies are often missing. Therefore, the technology availability is assessed using the indicators of technology readiness level and expertise required. When more expertise is required, it is more challenging to find competent personnel and to operate the process correctly.
- The estimated environmental effects of recycling can be known from prospective LCA studies (Arvidsson et al., 2018). The environmental burdens are addressed by two factors: climate impact and toxic process chemicals. It is advised to focus on the environmental impacts of the recovery stage, because this stage is a hotspot for emissions and is most variable.

Secondary market: The final link that closes a supply chain loop is the secondary market, where manufacturers purchase recovered metals. This transaction is characterized by the revenues. A second factor is the demand, which addresses the extent of 'pull' from the market.

- Revenues are generated by selling the recovered target metal and optionally co-recovered metals. An indication of each is obtained from the average market price. The indicator is expressed per ton of waste, reflecting that the waste input determines the processing capacity.
- The demand is assessed by three factors. The first, market stability, is characterized by a low price volatility. Many minor metal markets show a high price volatility (DERA, 2019), resulting in uncertain recycling business cases uncertain and reluctant investors. Price volatility intensifies the investment risk created by long payback times. On the other hand, more mature recycling chains offer the advantage of a more constantly priced resource. The second aspect is the demand growth rate (Lapko et al., 2019), because it increases the risk of temporary shortages whereas recycling offers a more constant supply. In addition, an expanding market offers a growing number of potential clients. Third, the confidence of clients is indicated by the price premium or discount. For emerging recyclers, it is challenging to gain the trust of potential clients, mainly related to the recycled product quality (Salim et al., 2019). On the other hand, clients might be willing to pay a premium if they value the sustainability of recycled resources.

2.4.3 Quantification and aggregation

The proposed framework intends to facilitate the comparison and ranking of different waste flows, and to this end it includes an approach for determining recyclability scores. These scores are calculated for each factor based on the corresponding quantitative or qualitative indicators, as illustrated in Figure 2.5. For qualitative indicators, we use a rating scale, in which 0–1 indicates a poor and 4–5 indicates an excellent performance. These scales can benefit from a frame of reference based on different cases. Quantitative indicators are normalized relative to minimum and maximum possible indicator values, and then translated to a 0–5 scale as well.

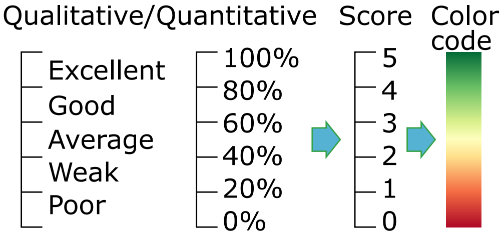


FIGURE 2.5: Example of mapping indicator values to scores.

It is possible to determine the score of aspects higher in the hierarchical structure of the framework (see Figure 2.4) through aggregation. Aggregated scores are calculated as the average of underlying indicators. It is possible to apply differentiated weighting factors based on the relative importance of factors, although for simplicity the present study does not differentiate.

In practice, the information to determine indicator values can be unavailable. These undefined indicators are disregarded when determining the average scores. For instance, when the collection participation is unknown, the score for collection rate is only based on the scores for consumer awareness and collection infrastructure density.

2.5 Case studies

2.5.1 Case study scope

This section features three case studies to illustrate what insights can be derived when the framework is applied. The case studies and their scope are outlined in Table 2.1. To support the interpretation of the findings, they are presented in the form of a diagram in Figure 2.6.

TABLE 2.1: Definition and scope of case studies.

Case study metal	Product category	Recycling technology	Geography
a) Neodymium	TV speakers	hydrometallurgy	EU-28
b) Indium	LCD screens (TVs, monitors, laptops)	hydrometallurgy	Switzerland
c) Platinum	car catalytic converters	plasma arc furnace smelting using iron collection	EU-28

2.5.2 Case study data

For all three case studies, we collected data to evaluate the indicator set. Data sources include MFA studies, indicator reports (DERA, 2019; Kucera & Sari, 2020) and qualitative information from literature. Scores on a 0–5-scale were calculated according to the proposed method and formulas (§2.4.3, Appendix A.2.2). The results are described in §2.5.3, while underlying data and assumptions are detailed in ESI 3[1].

2.5.3 Case study results

Figure 2.6 illustrates that stages with higher scores indicate drivers, while low scores indicate bottlenecks for recycling. The visual representation with color coding allows to compare the case studies and identify those with a high recyclability.

We find that the Nd recycling case has on average the lowest indicator scores (Figure 2.6a). In each stage, there is room for improvement, with the most notable barriers found among the overarching factors. Potential drivers for successful recycling are the social benefits and the recovery and preprocessing performance.

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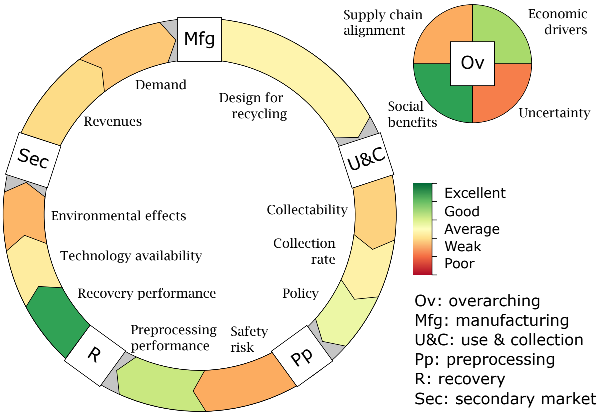
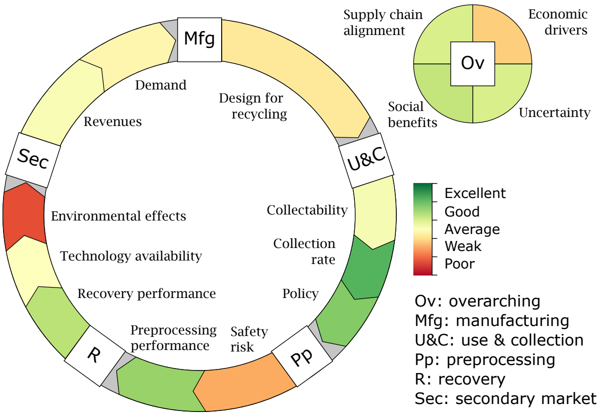
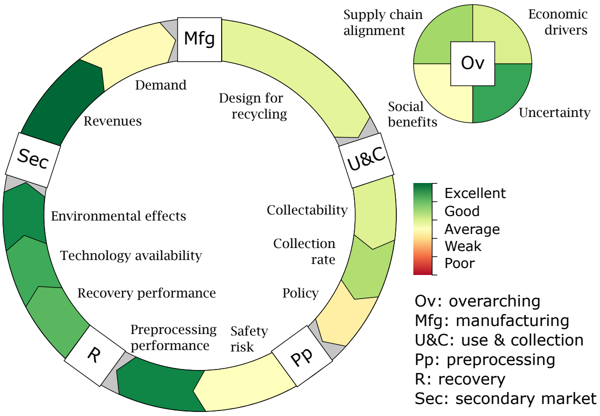
[1] https://www.sciencedirect.com/science/article/pii/S0921344921004900#sec0026



(A) Neodymium from TV speakers



(B) Indium from LCD screens



(C) Platinum from catalytic converters

FIGURE 2.6: Recyclability scores of three case studies.

In contrast, the indium case study (Figure 2.6b) reveals three major bottlenecks while other aspects are moderately positive. One bottleneck is the environmental impact, as several studies indicate that indium recycling causes a higher impact than primary mining (Böni et al., 2015). This barrier is linked to the low concentration of indium in screens. The other barriers are the safety risk from hazardous substances, and limited economies of scale.

Figure 2.6c shows that platinum recycling from car catalysts has almost no barriers, which is in line with the worldwide EoL recycling rate of over 50% (Graedel et al., 2011). Given the maturity of platinum recycling compared to emerging recycling industries, policy incentives are less relevant. While e.g. subsidies are useful for emerging technologies, they are no longer essential when a recycling system has been established. Room for improvement is identified in the export regulations, because this policy can be enforced better (Mehlhart et al., 2017).

2.5.4 Case study discussion

Generally, in the case studies we successfully quantified most indicators using publicly available data. As expected for early-stage assessments, some input data for case a) and b) entailed high uncertainty. For example, no quantitative data was found for the information exchange and the fraction of aware consumers. Still, the indicator set allowed to draw an indicative picture of the recyclability in each case.

Case b) indicated the benefit of an iterative approach. Within Switzerland, a limited economy of scale can be attained, which can be improved by enlarging the geographic scope of the analysis. With the scope extension, other indicators also change, so understanding the net effect on the recyclability requires a re-assessment. In addition, each further iteration can refine the quantitative input data.

Because the selected case studies are diverse, ranking them based on overall recyclability was not our goal. A ranking would require to trade-off the multitude of factors. In contrast, a ranking of alternatives will be easier in practical settings, where alternatives only differ on some aspects. An example is the comparison of wastes as inputs for a platinum recycling plant. These wastes will have similar scores for the factors related to demand and recovery technology. Another example is the comparison of robotic and manual dismantling, in which case the preprocessing and recovery factors are relevant. In both examples, a subset of indicators is used which simplifies the interpretation.

The comparison of the three case studies shows a high variability in recyclability characteristics for different metals and different applications. The scores ranged from mostly good to predominantly weak. In addition, differences were revealed as to which value chain stages present bottlenecks for recycling. Therefore, we conclude that a low recyclability can have several causes, which require different actions to overcome.

2.6 Discussion & Conclusions

This paper aimed to improve the conceptual understanding of minor metal recyclability. A systematic screening of available literature provided and overview of barriers and drivers for recycling. Based on this overview a framework was proposed, which provides a structured view on the factors that determine recyclability. To our knowledge, this framework is unique in addressing minor metals specifically, as well as addressing all stages of the recycling chain. The focus on minor metals is reflected by the indicator set. Compared to other materials and metals, minor metals have a high *uncertainty* of future flows, a high degree of *dispersion* and a low *concentration* in products. Because of these properties, important process-related factors are the *identification accuracy* and the *expertise required*. Finally, the relatively high metal price volatility is included under *market stability*.

Next to providing conceptual understanding of recycling systems, the framework is a step towards an operational assessment of recyclability. The framework is useful for analyzing various recycling systems, as demonstrated in three case studies. The framework can be applied for comparative analyses between minor metals, waste flows, or between recycling technologies. For these comparisons, a geographic scope must be defined, since several factors are location-dependent. It is also possible to compare recyclability between countries, which could help to identify a location with favorable conditions for recycling.

Contrasting the framework to other recently published frameworks, a few differences stand out. One difference is in the type of the final outcome. Other frameworks yield an outcome that is aggregated to one (Mueller et al., 2017), two (Sun et al., 2016) or three (Zuo et al., 2019) dimensions. These overall scores allow for prioritization of numerous alternatives, but do not allow to pinpoint value chain steps with bottlenecks. The latter is a strength of our framework, due to the grouping of factors by value chain step. The hierarchical grouping shows the underlying factors as in Figure 2.3, enabling to identify bottlenecks.

A notable difference with most other frameworks is the absence of monetary indicators. This choice is motivated by the uncertainty in business cases, which is particularly high in the case of minor metals (see §2.3.1.1). Instead, the proposed framework does include important cost drivers. This approach is similar to the way in which Sun et al. (2016) used size reduction as a proxy for preprocessing costs.

The proposed indicator set is applicable to end-of-life recycling in most regions. The framework is less applicable to pre-consumer scrap, for which indicators of use and preprocessing could be irrelevant. Besides, some adjustments might be needed for application in developing countries. This limitation stems from a bias in the reviewed literature. Although none of the studies states the geographic demarcation, they are geared towards developed countries. Consequently, the social aspects of recycling might need different interpretation. For instance, a lack of H&S regulations influences the social benefits of recycling. Note that the proposed indicators are primarily applicable in early stages of development. For more advanced recycling systems, more detailed indicators can be added. These indicators use information from e.g. financial assessments and stakeholder surveys.

Several factors of recyclability are interlinked, as it is impossible to isolate independent factors. These interdependencies and causal loops can be investigated in future research using system dynamics models (Glöser et al., 2013). This enables to determine weighting factors that express the importance of each indicator. It is therefore recommended to investigate the system dynamics, using the presented framework as a basis.

Another future research opportunity is to use the recyclability assessment in parallel with MFA. The latter highlights current flows and losses and provides quantitative input to some indicators. This was demonstrated in the case studies, that referenced MFA studies. In turn, the recyclability indicators highlight the relevance of flows for recycling. In this way, both analyses complement and enrich each other.

The framework is particularly helpful to close the cycles of minor metals, as it facilitates recyclability assessment in both the recycling industry and by policy makers. The presented framework and indicator set can be used as guidance for three main decisions: what waste flow to address, which technologies to apply and where to locate recycling operations. Besides, the framework assists policy makers to identify and resolve bottlenecks in recycling systems. To conclude, this framework paves the way for a more circular economy for metals that might be minor in volume, but major in economic importance.

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3 Analysis of the availability and recyclability of European waste flows

This chapter has been published as: Van Nielen, S. S., Sprecher, B., Verhagen, T. J., & Kleijn, R. (2023). Towards neodymium recycling: Analysis of the availability and recyclability of European waste flows. *Journal of Cleaner Production*, 394, 136252. doi:10.1016/J.JClePro.2023.136252

Abstract

The world is facing a growing neodymium demand, creating the need for developing a recycling system to handle future waste flows. Recycling technologies are emerging, but the recycling system around them can only be established with knowledge about available end-of-life (EoL) products. Therefore, this study quantified neodymium waste in European countries using material flow analysis, and assessed the recyclability of major EoL products. For 2019, we find a waste flow of 2.8 kt Nd, consisting mostly of NdFeB magnets. HDDs represent a large current waste flow, while the demand for magnets in industrial applications is increasing. In the future, electric vehicle motors and wind turbines likely provide a source of neodymium with good recyclability. Consequently, there will be different product groups that determine the future waste volumes. To manage the changing waste flows, a neodymium recycling system should be developed with the product properties of future waste flows in mind. Meanwhile, the recyclability of products can be improved by addressing bottlenecks in the recycling chain.

3.1 Introduction

3.1.1 Neodymium: a critical material

Neodymium is a strategically important resource and an essential element in modern societies. It is a key enabler of the energy transition due to its application in electric motors and wind turbines (Constantinides, 2018). If global climate ambitions are realized, the neodymium demand could increase tenfold (Alves Dias et al., 2020; Elshkaki, 2021). The European Union (EU) is a frontrunner in the energy transition, and therefore the challenge of neodymium waste treatment is expected to arise here first (Alves Dias et al., 2020). Consequently, the EU has developed strategies to create a circular economy for neodymium and other critical raw materials (Schäfer et al., 2020).

Recycling reduces the criticality and narrows the gap between supply and demand (Binnemans et al., 2013). Several neodymium recycling technologies are under investigation (Yang et al., 2017). Three major technologies are pyrometallurgical recovery, hydrometallurgical processing, and direct recycling of rare earth alloys (Walton et al., 2015). The latter technology is particularly attractive due to the low environmental impact (Miranda Xicotencatl et al., 2023; Sprecher et al., 2014). However, these recycling technologies have not yet been applied on large scale. To further develop neodymium recycling technologies through practical experience, suitable neodymium-containing wastes need to be identified and captured.

3.1.2 Research on neodymium flows

Recycling is enabled by material flow studies (MFAs) that map neodymium flows in society. Regarding the flows of neodymium in the EU, four recent studies have contributed to an understanding at aggregated level. The historic flows were mapped by Guyonnet et al. (2015) for 2010, while Ciacci et al. (2019) modeled flows in the past until 2016. Future flows were explored by Reimer et al. (2018), drawing attention to the expected large waste flows of electric vehicle motors. Finally, the neodymium flows associated with batteries, electronics and passenger vehicles were mapped by Huisman et al. (2017). These studies show diverging results due to differences in scope. For example, only one study included industrial motors (Reimer et al., 2018). A comparison with global neodymium flows indicates that motors (other than traction motors) and pumps could be a significant source of secondary neodymium (Schulze & Buchert, 2016), but several motor types were not considered in the studies with EU-focus.

Neodymium is used in neodymium-iron-boron (NdFeB) magnets, NiMH batteries and catalytic materials. All three of these components compete with alternative components, e.g. ferrite magnets and Li-ion batteries. Therefore, MFA studies should account for the market share of neodymium-containing components, which shows both changes over time and regional variation. For conventional cars, the content of NdFeB magnets was found to increase (Restrepo et al., 2017). Regional differences were observed for example in passenger cars, that contain larger amounts of NdFeB magnets in Japan than in the United States (Nguyen et al., 2019). For end of life (EoL) magnetic resonance imaging machines (MRIs) in Europe, two sources report very different waste flows: 10 t NdFeB was reported in 2016 (Ciacci et al., 2019) and 1000 t NdFeB in 2018 (Reimer et al., 2018). This difference can partly be traced back to the assumed market shares of 10% and 100%, respectively. For other products, snapshots of their composition and NdFeB market share are provided by waste analysis studies (e.g. Böni et al., 2015; Dańczak et al., 2018; Lixandru et al., 2017; Menad & Seron, 2017).

To set up an effective waste collection and processing system, it is key to know the type of available waste products. Since various products contain Nd, the ease of recycling can vary markedly between products (Habib, 2019; Yang et al., 2017). These differences are assessed in only few studies, with a focus on technical characteristics. A comparison of some waste electric and electronic equipment (WEEE) types identified hard-disk drives (HDDs) as a suitable waste flow for Nd recovery (REMANENCE, 2017). Besides, Habib (2019) discussed how product properties affect recycling.

The information on total volumes of neodymium flows in previous MFAs is insufficient for finding recycling opportunities for three reasons. First, inconsistencies in market shares and scope definition create uncertainties. Second, no country-specific analysis of waste flows is available, although Huisman et al. (2017) provide a starting point by analyzing certain waste flows in Europe. Third, the focus on total volumes ignores many important factors that determine the viability of recycling. Factors like product design and technology availability often prevent recycling from being implemented (van Nielen et al., 2022).

3.1.3 Insights to support recycling

As argued above, existing literature on neodymium flows in society is patchy at best and a systematic overview on the level of EU members states and product groups is lacking. This is problematic for policy makers and industries that aim to build a recycling system within the EU. In this study we address this by providing detailed information about past and present waste flows. We aim to evaluate the potential for neodymium recovery, and to identify promising EoL products for recycling. To that end, neodymium waste was quantified per product group in EU member states. The analysis used a dynamic material flow model that accounts for the lifespan distribution of products, diffusion of new products and the time-dependent market share of neodymium-containing components. In addition, the barriers and drivers for recycling were evaluated. Major waste flows were compared regarding their recyclability using a framework for recyclability assessment (van Nielen et al., 2022). This framework addresses product properties, policies and other aspects that determine the recyclability of neodymium.

3.2 Materials & Method

3.2.1 Overview & Scope

Material flow analysis and a recyclability assessment framework were used to quantify secondary neodymium resources over time. The waste quantities were calculated using a 4-step approach, as described in the sections below and shown in Figure 3.1. First, the production and trade flows were derived from statistics (*Apparent product consumption*). Next, input data were collected for the *Market trends* and *Product composition*, allowing to calculate the European consumption of Nd. Finally, *Waste generation* was modeled using a distributed lifespan approach. For five products with major Nd waste flows, the recyclability was assessed using a *Recyclability assessment* framework.

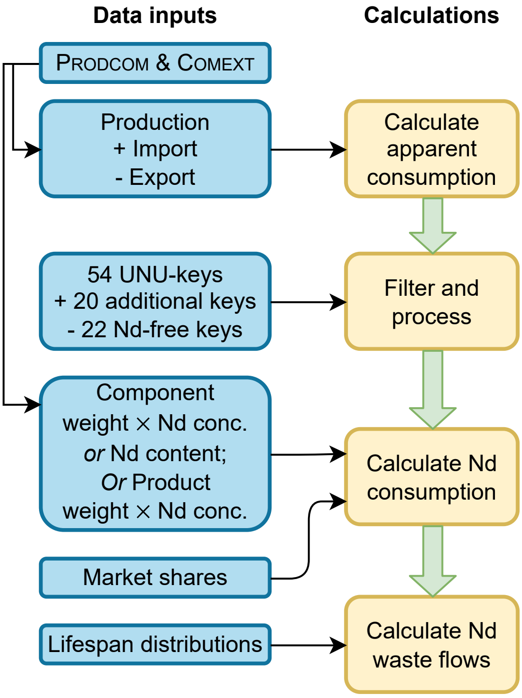

FIGURE 3.1: Overview of calculation steps in this study.

The geographical scope is limited to the current EU-27 and the UK (EU-28). The temporal scope ranges from 1990 to 2018, since for this range EU-wide statistical trade data is available. Given the focus on recycling, this study is limited to the flows that enter and leave the use phase.

For the recyclability assessment, the geographic scope is in line with the MFA, and the reference year is 2019. As recycling technology, we considered hydrogen decrepitation of sintered magnets, an environmentally benign option (Miranda Xicotencatl et al., 2023; Sprecher et al., 2014). This process recovers Nd alloys and is applied in magnet-to-magnet recycling (Zakotnik & Tudor, 2015) and hydrogen processing of metallic scrap (HPMS) (Walton et al., 2015).

3.2.2 Apparent product consumption

As a first step, the consumption of Nd-containing products over time is quantified. For this step, we followed the dynamic material flow approach of the 'Waste over Time' script developed in ProSUM (Huisman et al., 2017). Both the original source code and modified script are available online (van Straalen et al., 2016).

The 'Waste over Time' script implements the apparent consumption method, implying that the consumption in a country is calculated as the sum of production and imports minus exports. The data inputs were obtained from Eurostat statistics on production (Eurostat, 2019c) and international trade (Eurostat, 2019a). Since some statistical data points are missing or erroneous, the script makes several modifications (van Straalen et al., 2016). The calculation involves six steps.

1. Several PRODCOM data points were confidential and withheld by Eurostat. These gaps are filled using interpolation from other years and countries.
2. Missing data for product weight are derived from the number of products and the average product weight.
3. Outliers are replaced by interpolations.
4. CRT monitor data are derived from the PC sales volumes.
5. Data for before 1995 are extrapolated with a linear trend starting at the introduction year of each product.
6. Data for 2019 are extrapolated from the trend in recent years.

To match the scope of the present research, four adjustments were made to the input files and calculation procedures. Additional goods were included, statistical data covering 2016–2018 were included, consumption data from other sources were added, and average product weights were updated. The products were grouped using the UNU-Key classification system (Forti et al., 2018). Within this system, 54 UNU-Keys cover all WEEE. The classification was extended with 18 keys, covering vehicles, industrial applications and wind turbines. Although the UNU-Keys are intended to have homogenous material compositions (Forti et al., 2018), we considered 8 keys as inhomogeneous with respect to Nd content. Therefore, these categories were divided in subcategories, indicated by a suffix in Table 3.1. Ultimately, our analysis included 53 product categories, which we believe covers almost the whole range of neodymium-containing products.

The statistical dataset was extended to recent years, using the same sources as the original method (Eurostat, 2019a,c). For three products, more specific sales data were obtained from dedicated sources: MRIs (OECD, 2019), passenger vehicles (ACEA, 2021; ICCT, 2018), and wind turbines (EurObserv'ER, 2019; Eurostat, 2019b).

The product weights were updated to better capture the change over time. For each product type and year in the COMEXT dataset, the product weight was determined as the median value of the weight in all countries. These updated weights were used in step 2 to derive mass flows from numbers of products. Whenever possible, this conversion used weights specific for each year.

3.2.3 Market trends

The neodymium flows associated with each product category were derived from the apparent consumption while accounting for market penetration dynamics. The demand for Nd D_{Nd} was calculated by multiplying the demand for a product, D_p, by both the Nd content of the product M_{Nd} and the market share of Nd-containing products s_p (Eq. 3.1). This calculation is applied to each product, country and year.

$$D_{Nd} = D_p \times M_{Nd} \times s_p \tag{3.1}$$

In Eq. 3.1, s_p is a crucial factor because of the existence of substitutes to Nd-containing components. In fact, either a NdFeB magnet or a ferrite magnet can be used in many applications, and NiMH batteries exist next to several other battery technologies.

As a basis, we created an overview of market shares of Nd-containing components in different years and applications based on scientific literature and technical reports. For some applications, this overview contained sufficient data points to interpolate the market share evolution over time. For applications with no or few data points, the market share was assumed to follow the average permanent magnet market trend depicted in Fig. 3.2 (see Appendix B.1, B.2).

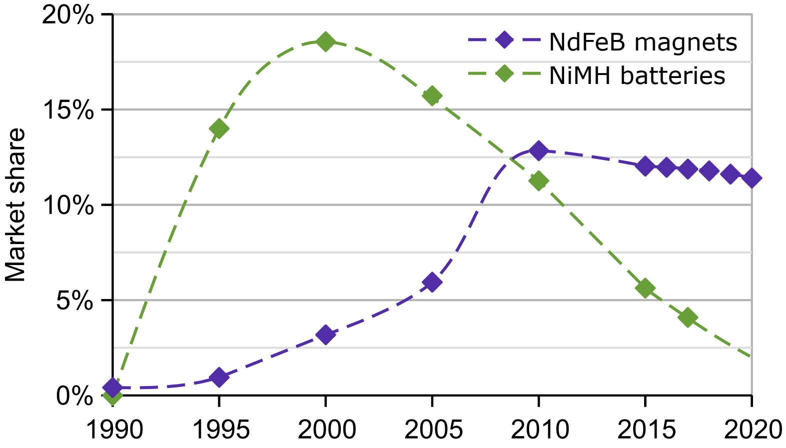

FIGURE 3.2: Volumetric market share of NdFeB magnets in the global permanent magnet market, and NiMH batteries in the portable battery market (Pillot, 2018). See Appendix B.1 for data sources.

In the model, we explicitly modeled the diffusion of solid-state drives (SSDs), smartphones and tablets. These consumer electronics have shown a large sales growth, which is untraceable in statistic reports. These electronics are aggregated with products with a significantly different NdFeB magnet content (HDDs, mobile phones and laptops, respectively), justifying a disaggregation based on market share data. The market share trend for HDDs and SSDs was based on global shipment data (Alsop, 2017, 2019; Robinson, 2020; Sprecher et al., 2014). This trend is approximated by Eq. 3.2. For the data storage market excluding datacenters (Alsop, 2019), Eq. 3.3 was derived. A full overview of the data is given in Appendix B.3. As a proxy for the share of smartphones in the mobile phone market, the penetration of smartphones in the United States was used (ComScore, 2017). The trend after 2015 was extrapolated using a fitted logistic function, see Eq. 3.4. The ratio between tablet and laptop sales were obtained from global sales figures (Ubrani & Nataraj, 2019).

$$S_{HDD,all} = \frac{0.7}{1+e^{0.4\cdot(y-2018)}} + 0.3 \tag{3.2}$$

$$S_{HDD,PC} = \frac{1}{1+e^{0.4\cdot(y-2018)}} \tag{3.3}$$

$$S_{smartphones} = \frac{0.92}{1+e^{-0.57\cdot(y-2011.66)}} \tag{3.4}$$

3.2.4 Product composition

The demand for each product category induces a neodymium demand, which was calculated using the average application properties. Eq. 3.1 indicates the need for data on the Nd content per product M_{Nd}, which was obtained from literature and manufacturer information. Since the Nd content is not directly available for most products, additional data were gathered on Nd-containing components. We distinguish between three of these components: NdFeB magnets, NiMH batteries and catalytic materials. When only the component weight M_c was known, we used the Nd concentration in the component $C_{Nd,c}$ to calculate the Nd content (Eq. 3.5). When the Nd weight fraction $C_{Nd,p}$ was available, it was multiplied by the category-average product weight M_p [1] to obtain the Nd content (Eq. 3.6). The product compositions are listed in Table 3.1.

$$M_{Nd} = M_c \times C_{Nd,c} \tag{3.5}$$

$$M_{Nd} = M_p \times C_{Nd,p} \tag{3.6}$$

TABLE 3.1: Average product properties per product group. The listed product weights are weighted averages.[b]

Product group	Product weight M_c (g)	Nd in product $C_{Nd,c}$ (g/g)	Component weight M_p (kg)	Nd in component $C_{Nd,p}$ (g/g)	Data source
NdFeB magnets					
Central heating (household)	30.85		200	0.2231	[1]; [2]
Dishwashers	45.49		35	0.228	[3]; [4, 5]
Washing machines	71.39	$1.11 \cdot 10^{-4}$	130	0.228	[6]; [3]; [4, 5]
Dryers	43.19		102	0.228	[3]; [4, 5]
Heating and ventilation (household)	12.14	$1.23 \cdot 10^{-4}$		0.228	[6]
Fridges	38.18	$1.04 \cdot 10^{-4}$	112.5	0.228	[6]; [3, 5]; [4, 5]
Freezers	43.91	$1.69 \cdot 10^{-5}$	112.5	0.228	[6]; [3, 5]; [4, 5]
Air conditioners	26.7		152	0.228	[5, 7, 8]
Other cooling	41.7		200	0.228	Own assumption

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[1] The product weight was derived from COMEXT as described in §3.2.2.

Product group	Product weight M_p (g)	Nd in product $C_{Nd,p}$ (g/g)	Component weight M_c (kg)	Nd in component $C_{Nd,c}$ (g/g)	Source
Cooling (professional)	102.82		350	0.228	[7]
Microwaves	20.52	$2.20 \cdot 10^{-5}$	110	0.228	[6]; [4]
Fans	1.97	$2.75 \cdot 10^{-5}$	0.6	0.228	[6]
Vacuum cleaners	3.27	$3.16 \cdot 10^{-5}$	16.8	0.228	[6]
Personal care	1.89	$4.18 \cdot 10^{-4}$	1	0.228	[4]; [4]
Shavers	5.52	$6.20 \cdot 10^{-4}$	1	0.228	[4]; [4]
Toothbrushes	0.55		1	0.228	[4]
HDDs	0.49	$5.65 \cdot 10^{-3}$	15.03	0.2868	[9]; [9, 10, 11, 12]; [4, 9]
Desktop PCs	9.33	$4.35 \cdot 10^{-4}$	18.86	0.2698	[6]; [13, 14, 15]; [4, 5, 16, 17]
Laptops	1.81	$1.77 \cdot 10^{-3}$	11.4	0.27	[6, 17]; [14, 15, 17, 18]; [4, 5, 17, 18]
Tablets	0.5		2.4	0.2245	[19]
Printers	9.12	$3.63 \cdot 10^{-4}$	15	0.228	[6]; [15]
Telecom	0.58		0.48	0.2105	Assumed identical to mobile phones
Mobile phones	0.10	$3.85 \cdot 10^{-3}$	0.48	0.2105	[17, 20]; [4, 17, 18, 21, 22]; [4, 17]
Smartphones	0.10	$1.10 \cdot 10^{-3}$	0.2008	0.2245	[6, 23]; [17, 19]; [4, 17]
Flat screen monitors	5.32		3.95	0.1648	[24]; [24]
Small consumer electronics	0.39	$4.69 \cdot 10^{-4}$	2.10	0.177	[6]; [21]
Headphones, earphones	0.09	$5.59 \cdot 10^{-3}$	0.93	0.177	[20]; [14]; [17]
Music instruments, radio, HiFi	3.88	$8.07 \cdot 10^{-3}$	1.33	0.3305	[6]; [5, 20]
Video players	3.51	$1.13 \cdot 10^{-3}$	1.21	0.3305	[6]; [4, 5]; [4, 5]
Speakers	2.45	$4.41 \cdot 10^{-5}$	47.19	0.2330	[6]; [15, 17]; [16, 17]
Cameras	0.54		0.20	0.2245	Assumed identical to smartphones
Flat screen TVs	10.46		11.48	0.1795	[14, 25, 24]; [24]
Game consoles	0.48		10	0.3305	[15]
MRIs	16000	$6.87 \cdot 10^{-4}$	1.6E+6	0.2290	[6]; [4, 8, 16]; [4, 16]
Cooled vending machines	92.22		112.5	0.2280	
Cars	1293	$1.25 \cdot 10^{-5}$	273	0.26	[6]; [5, 13, 26–28]; [5]

Product group	Product weight M_p (g)	Nd in product $C_{Nd,p}$ (g/g)	Component weight M_c (kg)	Nd in component $C_{Nd,c}$ (g/g)	Source
BEVs	1131		2000	0.1964	[4, 8, 16, 29, 30–33]; [2]
PHEVs	1801		2000	0.1964	[4, 8, 16, 29, 30–33]; [2]
HEVs	1510		1316	0.2275	[16, 30, 34, 35]; [5, 16]
Snowmobiles, golf cars etc.	3779	$1.72 \cdot 10^{-4}$		0.26	[6]; [16]
Trucks	3475		39.8	0.26	[3, 36]
Buses	3705		39.8	0.26	[3, 36]
Motorhomes	2349		39.8	0.26	[3, 36]
Electric bikes	33.60		266	0.2320	[7, 8, 13, 15, 16, 21, 37]
Industrial machines & motors	3468	$9.57 \cdot 10^{-5}$	175	0.2513	[6]; [38]; [5, 16]
Industrial pumps	9.28	$7.42 \cdot 10^{-5}$	350	0.2231	[6]; [1]; [2]
Lifting and conveying machines	134.62	$1.87 \cdot 10^{-4}$		0.2513	[6]
Shaping machines	657.68	$4.21 \cdot 10^{-4}$		0.2513	[6]
Wind turbines, onshore, low speed [a]	38.22		625	0.294	[8, 16, 39, 40]; [16, 39, 41]; [2]
Wind turbines, onshore, high speed [a]	38.22		134	0.294	[8, 16, 39, 40, 42]; [16, 39, 41]; [2]
Wind turbines, offshore, low speed [a]	38.22		625	0.294	[8, 16, 39, 40]; [16, 39, 41]; [2]
Wind turbines, offshore, high speed [a]	38.22		134	0.294	[8, 16, 39, 40, 42]; [16, 39, 41]; [2]
Industrial robots	30.81		1990	0.27	[5]; [5]
Catalysts					
Catalytic converter			1.114	0.0157	[43]; [44]
FCC catalyst		$1.95 \cdot 10^{-3}$	1	0.0035	[6]; [45, 46]
NiMH batteries					
Other small household: wrist-watches	1.1		0.5	0.0109	[c]
Vacuum cleaners	3.27		490.5	0.0109	[47]; [c]
Personal care	1.89			0.0109	[c]
Shavers	0.312		78	0.0109	[47]; [c]
Toothbrushes	0.55		137.5	0.0109	[47]; [c]
Telecom	0.582		105	0.0109	[47]; [c]
Mobile phones	0.099			0.0109	[c]
Cameras	0.545		62	0.0109	2 AA batteries; [c]
Power tools	2.53	$9.57 \cdot 10^{-5}$	598	0.0109	[47, 48]; [c]

Product group	Product weight M_p (g)	Nd in product $C_{Nd,p}$ (g/g)	Component weight M_c (kg)	Nd in component $C_{Nd,c}$ (g/g)	Source
Tools (professional)	23.17	$9.57 \cdot 10^{-5}$	598	0.0109	[c]
Toys	0.45		93	0.0109	3 AA batteries; [c]
HEVs	1510	$2.02 \cdot 10^{-4}$	38467	0.0109	[28, 34, 35, 49, 50]; [28, 35, 51]; [c]

BEV: battery electric vehicle; FCC: fluid catalytic cracking; HEV: hybrid electric vehicle; PHEV: plug-in HEV.

[a] Quantities are reported per MW installed capacity. [b] In the model, a product weight specific for each year and country was used when possible. The values listed here are the weighted averages. [c] Nd content in NiMH batteries was calculated as the mean of reported values (Fishman et al., 2018; GEUS & D'Appolonia, 2017; Guyonnet et al., 2015; Restrepo et al., 2017; Rombach & Friedrich, 2014; Schüler et al., 2011; Sommer et al., 2015; Yano et al., 2016).

[1] Personal communication, Grundfos, 2020, [2] (SUSMAGPRO, 2020), [3] (Seo & Morimoto, 2014), [4] (Habib et al., 2014), [5] (Sekine et al., 2017), [6] (Nansai et al., 2014), [7] (Morimoto et al., 2019), [8] (Schulze & Buchert, 2016), [9] (Dańczak et al., 2018), [10] (Tecchio et al., 2018), [11] (Sprecher et al., 2014), [12] (Auerbach et al., 2017), [13] (Yang et al., 2017), [14] (Hobohm & Kuchta, 2015), [15] (Glöser-Chahoud et al., 2016), [16] (Reimer et al., 2018), [17] (Böni et al., 2015), [18] (Buchert et al., 2012), [19] (Manhart et al., 2016), [20] (REMANENCE, 2017), [21] (Ciacci et al., 2019), [22] (Bandara et al., 2014), [23] (Wu et al., 2008), [24] (Lixandru et al., 2017), [25] (Thiébaud et al., 2018), [26] (Peck et al., 2017), [27] (Restrepo et al., 2017), [28] (Fishman et al., 2018), [29] (Constantinides, 2018), [30] (Elwert et al., 2017), [31] (Hofmann et al., 2013), [32] (Gutfleisch, 2013), [33] (de Haan & Zah, 2013), [34] (Bauer et al., 2010), [35] (Yano et al., 2016), [36] (Widmer et al., 2015), [37] (Habib & Wenzel, 2014), [38] (Buchert et al., 2014), [39] (Viebahn et al., 2015), [40] (Goodenough et al., 2018), [41] (Fishman & Graedel, 2019), [42] (Barteková, 2015), [43] (Belcastro, 2012), [44] (Thermo Fisher Scientific, 2012), [45] (Topete, 2014), [46] (Hsu & Robinson, 2019), [47] (Sommer et al., 2015), [48] (Pillot, 2018), [49] (GEUS & D'Appolonia, 2017), [50] (Moss et al., 2013), [51] (Davies, 2006).

3.2.5 Waste generation

After calculating the consumption of products as described above, we derived the waste flows. The waste flows were modelled using a distributed lifespan approach to account for differences in the service life of products. This approach is preferred over a constant lifespan assumption, since most applications under study have a changing demand over time. The approach was applied to obtain the waste flow of both applications and incorporated Nd.

The lifespans of products were modelled as Weibull functions ($L(t)$), which approach practical lifespan distributions (Forti et al., 2018). The lifespan distribution function in Eq. 3.7 describes the expected lifespan using parameters α and β. Then for each model year t and product p, the waste flow $W(t)$ is determined from the consumption D in all previous years n:

$$L(t) = \frac{\alpha}{\beta^{\alpha}} \cdot t^{\alpha-1} \cdot e^{-(t/\beta)^{\alpha}} \tag{3.7}$$

$$W(t) = \sum_{n=t_0}^{t} D(n) \cdot L_p(t-n) \tag{3.8}$$

The Weibull parameters and data sources are provided in Appendix B.4.

Flows were quantified per country because waste can be transported freely within national borders, and waste policies differ. The average Nd waste density was calculated per square kilometer, using land area (FAOSTAT, 2021).

3.2.6 Recyclability assessment

A further assessment of the suitability of products for recycling was performed for five selected flows. This selection was based on the magnitude of current and future waste volumes, which are large for battery electric vehicle (BEV) motors, HDDs, speakers, wind turbines, and industrial pumps. For these waste flows, the recyclability was assessed using a framework recently developed by van Nielen et al. (2022). That paper has reviewed existing methods to assess the recyclability of materials and developed a combined framework, which covers the major factors that determine the recyclability of minor metals in waste flows (van Nielen et al., 2022). The framework provides 35 indicators, grouped by steps in a product's life cycle (Table 3.2).

The indicators are evaluated using various data sources. For indicators that refer to material flows, numbers from this study were used. When possible, product properties were obtained from waste dismantling studies (Habib et al., 2014; Lixandru et al., 2017; Talens Peiró et al., 2020, e.g.). A full overview of data sources in provided in Appendix B.5. To allow for a better overall comparison of waste flows, the results were aggregated by taking the simple mean of indicator values.

TABLE 3.2: Indicators to assess recyclability, grouped by step in the product life cycle. Adapted from van Nielen et al. (2022).

Life cycle step	Factors	Indicators
Overarching	Supply chain alignment	Fraction of actors involved in exchange
	Uncertainty	Standard deviation of future waste flow
	Economic drivers	Annual waste flow; Disposal and raw material tax; Recycling subsidy
	Social benefits	Avoided health and safety hazard; Labor rights indicator
Manufacturing	Design for disassembly	Material joint types; Metal content per component; Number of component designs; Number of product designs
Use & collection	Collectability	Annual waste generation; Ownership; Product weight
	Policy	EPR legislation; Export restrictions
	Collection rate	Collected fraction of EoL products; Distance between collection points; Consumer awareness
Preprocessing	Preprocessing performance	Dismantling time; Identification accuracy; Liberation efficiency
	Safety risk	Content of restricted substances
Recovery	Environmental effects	GHG emissions; Toxic process chemicals
	Recovery performance	Concentration after preprocessing; Recovery efficiency; Value fraction of recoverable metals
	Technology availability	Expertise required; Technology readiness level
Secondary market	Demand	Demand growth rate; Price premium or discount; Price volatility
	Revenues	Target metal value; Co-recovered metal value

3.3 Results

3.3.1 Neodymium demand

This section analyzes the amount of neodymium contained in consumed products. The demand for neodymium in the EU-28 was found to grow rapidly, due to both rising consumption and market penetration of Nd components. In 2000, only 1.7 kt neodymium was consumed, which grew by 300% to 5.3 kt in 2010. The growth has continued since then, up to a demand of 7.7 kt in 2019. Most of the neodymium was consumed in the form of NdFeB magnets, accounting for over 97% of the total demand. The combined demand for other applications amounted to 90–100 t/a in the last decade.

For NdFeB magnets as the major neodymium consumption driver, Figure 3.3 shows a disaggregation of demand. This diagram shows that new magnet applications emerged over time, creating successive *waves* of neodymium demand. Initially, magnet demand was driven by consumer electronics, such as HDDs and game consoles. From 2004 onwards, NdFeB magnets rapidly penetrated the industrial market, with applications in pumps and robots, and also in cars. A third wave appears to emerge in recent years,

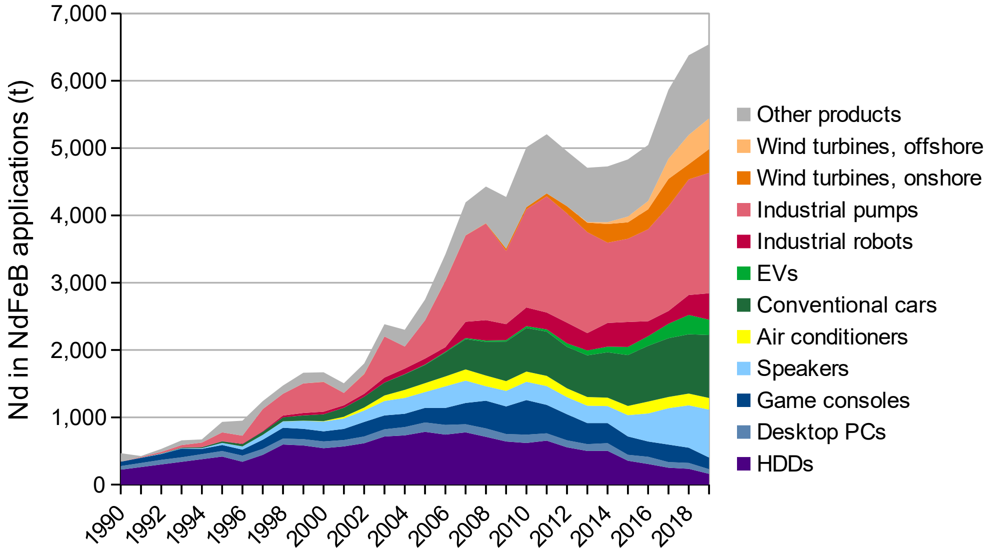


FIGURE 3.3: Increasing demand for Nd in EU-28 countries, grouped by application types.

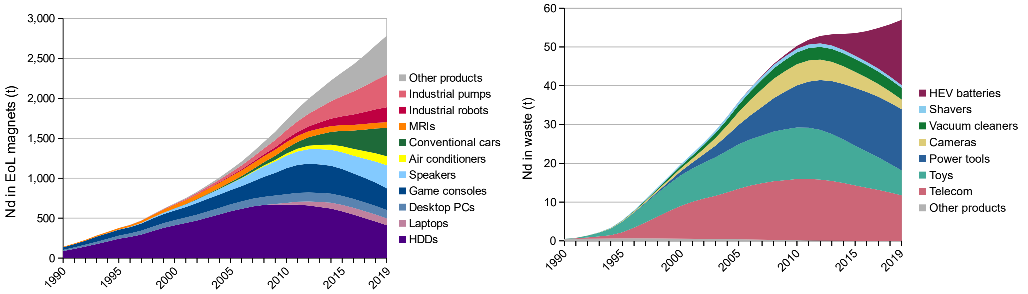
consisting of clean energy applications, i.e. EVs and wind turbines. The combined effect of these waves is an almost continuous and rapid demand growth trend.

3.3.2 Neodymium waste flows

In contrast to the demand flow, the dominant source of Nd in waste is consumer electronics (Figure 3.4a). In 2019, 15% of the Nd waste flow originated from HDDs. Yet this amount has been declining from its peak in 2011, a trend also found for laptops, desktops and game consoles. Another important waste flow concerns speakers, which includes various audio devices such as professional loudspeaker systems, home audio, and smart speakers. Contrary to other consumer electronics, the demand for Nd in speakers has increased in recent years, with waste following with delay.

The last few years show an increase in Nd flows associated with EoL industrial robots and pumps. These waste flows represented 7% and 15% of all Nd waste in 2019. Due to the longer lifespan of these products as well as the increasing demand, these waste types are expected to continue to grow in the future. Over the whole range, a shift to product groups with longer lifespans is observed. Automotive and industrial applications are in use longer than consumer electronics, therefore the stock increases even with constant influx.

Compared to EoL NdFeB magnets, other Nd-containing components contribute only little to the European Nd waste flows. Figure 3.4b shows a waste flow of 57 kt Nd in 2019, which represents 2% of the total in that year. Among the battery and catalyst applications, an increasing amount of Nd originated from hybrid electric vehicle (HEV) batteries. This concerns NiMH batteries found in older generations of HEVs. Since the



(A) Neodymium contained in EoL magnets.

(B) Neodymium contained in EoL batteries and catalysts.

FIGURE 3.4: Neodymium waste generated in EU-28 countries, grouped by application type.

market for HEVs and BEVs is overtaken by Li-based batteries (Pillot, 2018), this increase is only temporary.

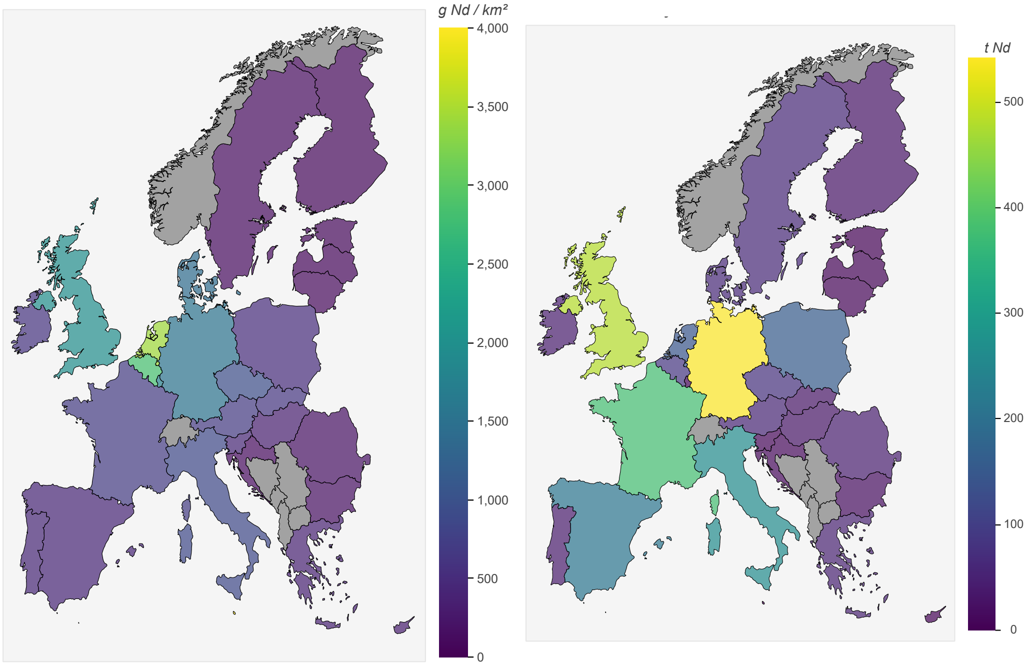
While the emergence of wind turbines and EVs is apparent in Figure 3.4a, these categories are virtually absent in the overview of waste flows. Based on the long lifespan of wind turbines and EVs, significant waste flows are expected several years in the future. Such a delay due to the lifespan was also observed for Nd from conventional cars.

3.3.3 Country comparison

A comparison of annual Nd waste flows of European countries revealed large differences, as illustrated by Figure 3.5. The waste density varies with almost two orders of magnitude, amounting to 75 g/km^2 for Latvia and 6.5 kg/km^2 for Malta. Besides Malta, high waste densities are found in other densely populated countries, Belgium and the Netherlands. The variation is also high for the absolute volume of Nd waste, coinciding with differences in population size (Figure 3.5b). Unsurprisingly, the total Nd waste flow was largest in countries with a large population, such as Germany, France and the UK.

3.3.4 Neodymium stocks

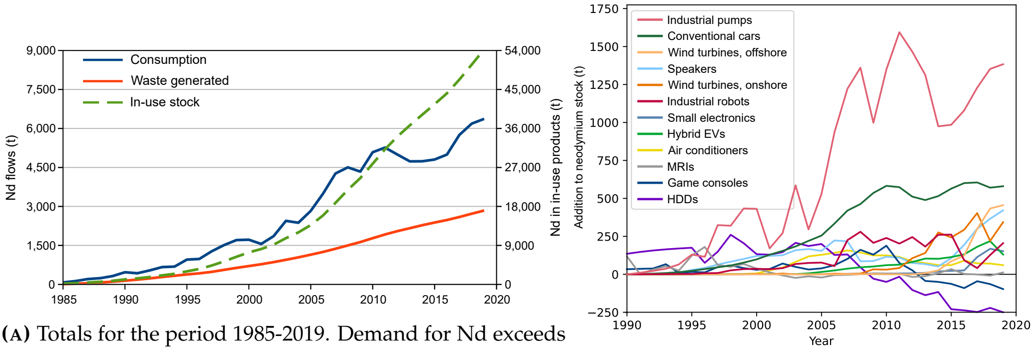
Lastly, Nd stocks and flows were examined from a macro-level perspective. Since 2000, Nd flows increased annually by on average 112 t for waste and 225 t for consumption. Besides, Figure 3.6 reveals an accumulating in-use stock, which results from a growing demand of long-lived products. The stock accumulation is mostly driven by industrial and automotive applications, see Figure 3.6b. This figure illustrates a contrast between stock build-up in various emerging technologies, and a net outflow of HDDs. In 2019, the stocks contained almost two thirds (63%) of all Nd resources consumed from 1985 onwards. The historic Nd waste flows are most likely dissipated to slags and steel alloys (Guyonnet et al., 2015; Thiébaud et al., 2018).



(A) Waste flow density.

(B) Total waste flow.

FIGURE 3.5: Geographic distribution of neodymium in waste per European country in 2019. Appendix B.6 provides a data table. *Note to editor: an interactive version of these figures is available and can be included in the online version of the article.*



(A) Totals for the period 1985-2019. Demand for Nd exceeds Nd in waste flows.

(B) Net addition to Nd stock, for products with a stock change greater than 150 t/a.

FIGURE 3.6: Neodymium stock and flow trends for EU-28 countries.

3.3.5 Recyclability assessment

The recyclability assessment framework (§3.2.6) was applied to the product groups of BEV motors, HDDs, speakers, wind turbines and industrial pumps. By evaluating a set of indicators, drivers and barriers for recycling were identified, as described below. Table 3.3 summarizes the results per life cycle stage. All data and sources used in the assessment are reported in the ESI[2].

TABLE 3.3: Recyclability of selected waste flows. Aspects scores are normalized to the range $0 - 5$, where 5 is most favourable for recycling.

Life cycle stage		EV motors	HDDs	Industrial pumps	Speakers	Wind turbines
Overarching	Supply chain alignment	1.3	1.0	0.5	0.5	3.0
	Uncertainty	3.3	4.0	1.7	1.6	4.5
	Economic drivers	1.4	1.9	3.0	3.0	2.4
	Social benefits	4.4	4.4	4.4	4.4	4.4
Manufacturing	Design for disassembly	1.6	2.8	2.0	2.3	4.2
Use & collection	Collectability	2.2	1.6	3.5	2.2	3.6
	Policy	3.0	2.3	0.5	2.3	3.8
	Collection rate	3.1	3.5	0.8	3.0	4.5
Preprocessing	Preprocessing performance	4.3	4.0	2.9	3.2	5.0
	Safety risk	3.5	3.0	5.0	4.5	4.5
Recovery	Environmental effects	4.8	4.8	4.8	4.8	4.8
	Recovery performance	4.9	3.9	4.4	4.9	4.4
	Technology availability	2.3	2.3	2.3	2.3	2.3
Secondary market	Demand	1.9	1.9	1.9	1.9	1.9
	Revenues	5.0	5.0	4.2	3.6	3.9

Overarching The recyclability of all five products is supported by the overarching social benefits. These benefits stem from avoiding primary mining of rare earths, and the associated safety hazards (Bailey, 2019; Yang et al., 2017) and labor rights violations (Kucera & Sari, 2020). A distinguishing economic driver is economies of scale. This limits the recyclability of emerging applications (wind turbines and BEVs) at present, whereas sufficient waste quantities are available from products used in mature markets. Financial barriers can be overcome by economic policy incentives, which do exist but not for magnet recycling specifically. All five products have a low supply chain alignment, lowering the recyclability score. A mixed picture is seen for uncertainty, with a remarkably high uncertainty for the future waste flow of industrial pumps and speakers and low uncertainty for wind turbines.

Manufacturing None of the products is fully designed with recycling in mind, as can be seen from the way components are integrated. Magnets are often glued in place in pumps and speakers (Lixandru et al., 2017), while at best metal casings and screws are

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[2] https://www.sciencedirect.com/science/article/pii/S0959652623004109#appSB

used in the other product designs (Talens Peiró et al., 2020; Walachowicz et al., 2014). On the other hand, many components have very similar designs, like HDDs and wind turbine rotors, which is favorable for routinized disassembly.

Use & Collection The collection rate of EoL products varies a lot. It is high for wind turbines (Reimer et al., 2018), as expected from the high collectability. Collection of wind turbines as well as pumps is enabled by their weight and company ownership. Speakers, HDDs and BEVs have a moderate collection rate. While their collectability is lower and uncontrolled export is a threat (Huisman et al., 2015), this is counterbalanced by strong waste policies. The lowest collection rate was found for industrial pumps, which were until recently excluded from the WEEE directive (Andersen, 2022; European Commission, 2012). Although some pump manufacturers offer take-backs [3], extensive collection systems are missing.

Preprocessing Wind turbines have the most effective preprocessing, since the magnets are large, can be identified with ease and liberated effectively. For speakers and pumps, the challenge is to distinguish units with NdFeB magnets, as these products have a high market share of ferrite magnets [4]. All five waste flows have the advantage that they are free from hazardous substances. However, the waste treatment of HDDs is often subject to data protection regulations.

Recovery Since the hydrogen decrepitation process is assumed to be used, the recovery performance is almost identical for all waste flows. Hydrogen decrepitation has several advantages, such as a high recovery efficiency, no toxic process chemicals (Walton et al., 2015), low GHG emissions (Miranda Xicotencatl et al., 2023), and no interference with recovery of other metals. On the other hand, the technology needs expertise and has intermediate maturity. Hydrometallurgical and pyrometallurgical technologies have a similarly high recovery. These alternatives are more mature, but also have higher environmental impacts.

Secondary market All recycled magnets share the same secondary market. This market is characterized by a steady demand growth (Elshkaki, 2021) and a high price volatility (Bastian, 2020). The latter presents a barrier for long-term business planning. Recovered magnet alloys can generate relatively large revenues, even though a weak price premium is expected on top of the commodity price. The products differ regarding the co-recovered metals, which are valuable for HDDs and EVs and smaller for speakers.

Contrasting the five EoL products, significant differences exist in both product characteristics and value chains. These differences translate to diverse bottlenecks or barriers for recycling. As also observed in the MFA, the waste flows differ in annual quantity. Besides, big differences exist in product weight, design variation, and waste collection policies.

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[3] Grundfos, personal communication

[4] Personal communication, B&C Speakers and Grundfos.

Aside from these differences, common drivers and barriers for Nd recycling can be identified. Prominent bottlenecks appear in the overarching factors, notably supply chain alignment. Additional bottlenecks are the metal price volatility and product design. A common driver is the recovery step, owing to the good performance and compatibility.

3.4 Discussion

3.4.1 Promising waste flows

This study has shown that a large share of neodymium is currently used for magnets in industrial applications. We identified pumps and robots as significant and growing contributors to Nd waste flows, as opposed to most previous studies on the EU level. Also, HDDs were identified as a major source for the near future, which is in line with earlier work (Thiébaud et al., 2018). We calculated a total Nd demand of 7.7 kt in 2019, while a previous study (Huisman et al., 2017), which disregarded industrial applications, found a much lower value. For other application groups such as vehicles, the results are in agreement.

Surprisingly, the recyclability assessment revealed that the recovery process—although immature—was not the main bottleneck for recycling. Instead potential bottlenecks are in design, waste collection and other steps that are product-dependent. This implies that the experience gained with one product might not be transferable to other products. Further R&D efforts on recycling should define target products and focus on these to develop relevant solutions.

In line with previous findings, a good recyclability was found for wind turbine magnets (Habib, 2019). Wind turbines scored above average for many recyclability indicators, except for the current waste flow.ăThis implies that if the growth of wind energy follows the political ambitions, it becomes a highly attractive source of EoL magnets.

The rapid growth of Nd demand observed in this study has two major implications. First, the growth creates a large gap between Nd demand and the potential secondary supply (Fig. 3.6a). Even when Nd recycling is expanded to its full potential, virgin input is needed to meet the demand. Therefore, Nd recycling is unlikely to be hindered by limits to recycled content in magnets. Second, the in-use stock is growing, especially for long-lasting products. This stock creates an urgency to establish a recycling system soon, thereby avoiding dissipation through improper waste treatment.

Countries differ considerably in terms of total Nd waste volume and waste density (kg/km^2). This implies a trade-off for recycling industries, with the outcome depending on business characteristics. Pilot plants could best be established in countries with a high Nd waste density such as Belgium and the Netherlands. When upscaling opportunities are important, a country with a large total volume available would be preferred. Having access to material within a single jurisdiction avoids permitting procedures for waste export. Further upscaling is possible by importing waste flows from other countries.

3.4.2 Limitations & uncertainties

The dynamic MFA approach delivered results with a high level of detail, but also has some limitations. The strength of the followed approach, based on trade statistics, is the ability to model dynamic markets. Compared to a stock-driven approach, it allows to study products that are weakly linked to population size, e.g. industrial equipment. The statistical data were processed to correct for reporting errors. Still, some errors could remain, presenting a source of uncertainty.

The results are sensitive to the market penetration of NdFeB magnets, therefore this study considers market shares to be time-dependent. This approach leads to results that differ from other studies, as highlighted in a comparison for auxiliary vehicle motors (i.e. outside the drivetrain). These motors are identified as the presently largest source of EoL NdFeB magnets by two previous studies (Ciacci et al., 2019; Reimer et al., 2018). For 2018, Reimer et al. (2018) reported $\sim$ 2600 t EoL auxiliary vehicle motor magnets (or 776 t Nd). Ciacci et al. (2019) calculated a similar amount for the year 2016. Our value for auxiliary motors is smaller (310 t in 2018), because we correct for the market share of NdFeB magnets based on Restrepo et al. (2017) and Habib et al. (2014). In fact, cars manufactured before 2000 only contain minor Nd quantities (Restrepo et al., 2017).

3.4.3 Future research

Market shares of permanent magnets are also expected to differ between countries, although it has been little researched. Only for passenger vehicles and wind turbines, regional differences were indicated (Carrara et al., 2020; Nguyen et al., 2019). Among EU countries, wealthier countries are expected to consume more high-end (‘premium’) versions of applications, featuring rare earth magnets more often. Tightening energy efficiency standards are expected for pumps and other motors, triggering a wider use of lightweight and strong NdFeB magnets. To understand the differences between countries, further research is needed.

This study investigated recyclability from a value chain perspective. As a next step, recyclers could investigate specific technical characteristics of waste flows. Relevant characteristics for NdFeB magnets include the type of coating and corrosion level (Burkhardt et al., 2020). In addition, feasibility studies could address the prevalence of sintered and bonded NdFeB magnets, since the magnet type determines what recycling technologies are applicable (Yang et al., 2017).

3.4.4 Improving the recycling system

The recyclability assessment complemented the MFA insights by broadening the perspective. The framework comprises both key material flow indicators and indicators of product properties and organizational structures. Besides, it allows to identify possible improvements to the recycling system of products. During the further development of neodymium recycling, the recyclability framework is useful for monitoring changes in recycling technologies, product properties and societal settings. Then trends can be identified that make recycling more worthwhile.

To optimize the recycling system, sufficient scales are needed and can be achieved through consolidation, i.e. combined processing of waste flows. The need for consolidation is most apparent for consumer electronics. In this waste flow, a shift is ongoing from HDDs to a diverse mix of appliances. Combined collection and sorting seems feasible in existing recycling centers, although each application needs different handling. For some industrial products, growing volumes could justify a dedicated collection system in the future. Such B2B take-back has organizational advantages.

Consolidation of disassembly is highly challenging due to the identified design variation and evolution. This could explain why disassembly pilots focus on single products (Baba et al., 2013; Bast et al., 2014; Zakotnik & Tudor, 2015). A possible solution is design standardization, which would facilitate e.g. EV motor recycling (Bast et al., 2014). Alternatively, an adaptive process could be developed, using sensing and detection tools. Metallurgical recovery processes are already capable of combined processing. To ensure the output quality, it is essential to characterize and monitor the inputs.

3.5 Conclusions

This research underlines the urgency for developing a neodymium recycling system in Europe, both to meet the growing demand for raw materials and to avoid resource losses. Our analysis identified a rising Nd consumption, which leads to growing waste flows in the future. We identified HDDs as a major Nd waste flow, providing attractive input for recyclers for some years. Nd waste associated with industrial pumps and robots, and conventional cars are becoming more significant. In one to two decades from now, it is expected that EoL EVs and wind turbines will become the major source of secondary Nd. These large and/or growing waste flows should be targeted first to achieve significant recycling.

The growth of NdFeB magnet waste flows enables new business opportunity due to economies of scale. However, the application of neodymium is dispersed over many products, which complicates standardized recycling. The recycling industry should adapt to the expected changes in quantities of EoL products. Most flexibility would be needed in sorting and disassembly facilities. For emerging products, appropriate dedicated recycling routes need to be developed in time to accommodate for their future waste flows. This involves the development of processes for each part of the recycling system, and can be informed by a recyclability assessment. To improve on recyclability, it is suggested to focus on the design and secondary market steps of the recycling chain.

This study has explored the neodymium waste flow dynamics in European countries and the recyclability of EoL products, thereby providing insight into the recycling potential and differences between countries. Our research has shown that new magnet applications emerged over time, creating successive waves of neodymium demand. Consequently, there will be different product groups that determine the future waste volumes. The availability and product properties of these future waste flows should be kept in mind when developing neodymium recycling systems.

Meanwhile, current waste flows provide an opportunity for the developing neodymium recycling industry. EoL products with good recyclability can be used to gain experience in Nd recovery. Before starting, companies should choose consciously which waste flows to process and in which countries to start, as large differences exist between them. These findings support the development of effective recycling technologies and systems to close neodymium resource loops.

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4

Ex-ante LCA of magnet recycling: progressing towards sustainable industrial-scale technology

This chapter has been published as: Van Nielen, S. S., Miranda Xicotencatl, B., Tukker, A., & Kleijn, R. (2024). *Journal of Cleaner Production, 458*, p.142453. doi:10.1016/J.JClePro.2024.142453.

Abstract

To alleviate the pressure on the rare earth supply chain, new technologies are under development for recovering, recycling and remanufacturing NdFeB magnets. In this study, the anticipated environmental performance of large-scale recycling is investigated and compared to the production of primary magnets. To do so, this ex-ante life cycle assessment combines input from measurements of pilot processes, expert technology forecasts, thermodynamic modeling, and equipment data from manufacturers. We examined the effect of four technology developments: process changes, size scaling, internal recycling, and optimization.

The results show that at pilot scale, recovered NdFeB powders have lower impacts than primary powders for almost all impact categories. This demonstrates that the recovery of NdFeB alloys is environmentally beneficial. Magnets from anticipated large-scale recycling have over 80% lower impacts than primary magnets in most of the impact categories analyzed. All four investigated types of technology development contributed to this improved performance. The final configuration was validated by comparison with an industrial reference and theoretical optimum configuration. Four magnet manufacturing routes (sintering, extrusion, metal injection molding, bonding) have distinct environmental profiles, but all can progress to similarly low levels of impact. The choice among routes should be primarily based on the functional requirements.

4.1 Introduction

4.1.1 Recycling technology development

To achieve a net-zero society, we need not only huge investments in clean technologies and new infrastructure, but also a sound circular economy for all materials involved. This requires new recycling technologies, complementing repair and reuse. For example, end-of-life (EoL) NdFeB magnets arise in increasing quantities from applications in rapidly growing markets like electric vehicles (EVs), data centers and wind turbines (van Nielen et al., 2023). Recycling is currently limited and small-scale, signaling a potential waste treatment issue and an opportunity for new recycling activities (Jowitt et al., 2018). Anticipating the introduction of industrial-scale NdFeB recycling, it is important to understand the technology and its environmental impacts.

Magnets can convert electric energy into motion and *vice versa*, e.g. in electricity generators, motors, and speakers. High performance magnets are made from NdFeB, consisting of iron, boron and about 32% REEs, including Nd, Pr and Dy. The high magnetic strength results from a fine-tuned composition of the magnet alloy and its microstructure, and is achieved by sintering, i.e. fusing fine NdFeB powders at high temperatures (Brown et al., 2002). Bonded magnets are composed of NdFeB powder and a polymer binder. Bonded magnets can be produced in complex shapes, but have lower magnetic strength (Brown et al., 2002).

Several types of magnet recycling technologies have been investigated: direct alloy (or short-loop) recycling, pyrometallurgy and hydrometallurgy (Binnemans et al., 2013; Yang et al., 2017). Direct alloy recycling recovers NdFeB material directly without separating its constituents, often by pulverizing magnets under hydrogen atmosphere. Pyrometallurgy employs high-temperature melting, while hydrometallurgy involves dissolving magnets. These technologies have the advantage of removing impurities, but require large amounts of energy and chemicals, respectively (Ormerod et al., 2023). Previous life cycle assessment (LCA) studies have indicated the environmental benefits of direct alloy recycling (Elwert et al., 2017; Jin et al., 2018, 2020; Sprecher et al., 2014; Walachowicz et al., 2014; Wang et al., 2022; Zakotnik et al., 2016) and hydrometallurgical recovery (Bailey, 2019; Beylot et al., 2020; Schulze et al., 2018), see Table 4.1. This study focusses on direct alloy recycling, as it shows lower impacts.

Direct alloy recycling consists of multiple processes, further explained in Section 4.2.1. The key stages are waste pre-processing, alloy recovery, and magnet manufacturing. Previous LCA studies have focused on the recovery stage, assuming a single processing pathway for waste pre-processing and manufacturing. However, a successful recycling system can handle a variety of waste inputs, and can manufacture multiple products depending on the needs of the market. It is important to model the entire recycling chain up to a new product, because this allows to assess the effectiveness of substituting primary magnets.

In all stages—pre-processing, alloy recovery and magnet manufacturing—recent efforts in research and development (R&D) have resulted in significant technology advancement. Experimental work has explored new ways to automate the liberation of magnets from waste (Burkhardt et al., 2023) and their recovery through hydrogen processing of magnetic scrap (HPMS) (Jönsson et al., 2020). Remelting opens a new route

Table 4.1: LCA studies and results (as greenhouse gas (GHG) emissions) for recycling of NdFeB magnets using direct alloy recycling (top) and hydrometallurgy (bottom). Not all numbers are comparable due to differences in scope.

Study	Used magnet source	Final product	GHG emissions of recycled product (kg CO_2-eq/kg)
Sprecher et al. (2014)	HDD	Sintered magnet	3.3–10
Jin et al. (2016)	HDD	Sintered magnet	12.5
Jin et al. (2020)	HDD	Sintered magnet	26.1
Bailey (2019)	EV motor	Sintered magnet	6.0
Walachowicz et al. (2014)	EV motor	Sintered magnet	178
Jin et al. (2018)	EV motor	Sintered magnet	18–41 (US), 25–56 (China)
Wang et al. (2022)	Small magnets	Sintered magnet	8.4
Bailey (2019)	EV motor	Sintered magnet	13–42
Schulze et al. (2018)	EV motor	Nd-Pr alloy	13–59
Walachowicz et al. (2014)	EV motor	Rare earth oxides	12.2–15.9
Beylot et al. (2020)	HDD	Rare earth oxides	5.93–6.55

for alloy recovery from partly oxidized magnets, as it removes metal oxides as slag (SUSMAGPRO, 2019). Hydrogenation–disproportionation–desorption–recombination (HDDR) yields powders suitable for bonded magnets (Gutfleisch & Harris, 1996; Lixandru et al., 2017). The ‘shaping–debinding–sintering’ approach produces magnets directly in the final shape, hence improving the material efficiency of magnet manufacturing (Gonzalez-Gutierrez et al., 2018). Even traditional sintered and bonded magnets have a relatively short history (Brown et al., 2002) and may also be developing efficiency-wise. These developments justify a renewed look at the environmental performance of NdFeB magnet recycling and manufacturing.

4.1.2 Towards industrial deployment

The maturity of recycling processes can be expressed using the commonly used technology readiness level (TRL) (ISO, 2013). At TRL 1, the basic principles of a technology are observed, and at TRL 9 the production is fully operational (EARTO, 2014). TRLs help to define the scope and approach of an LCA (Bergerson et al., 2020; Thomassen et al., 2019). In this paper, we use TRLs to indicate the maturity of processes, and ‘small scale’ refers to the maturity level at the time of data collection.

On the path towards industrial-scale recycling of NdFeB magnets, technology developers will face fundamental decisions regarding process design and the overall layout of the recycling system. It is unknown how these choices affect the future industrial operation and its environmental performance. Current R&D efforts are mainly focused on smoothing the way to commercial implementation, whereas this may entail unforeseen but profound consequences for future operations.

4.1.3 Environmental impacts of future magnet recycling

To calculate the anticipated future environmental impacts of an emerging technology at large-scale, the method of ex-ante LCA has been developed (Cucurachi et al., 2018). Ex-ante LCA acknowledges that inventories of small-scale processes are not representative for industrial operation, since fundamental process changes, technology optimization, and changes in the wider economy are expected. Multiple studies have contributed to the conceptual development of methods for upscaling from small to industrial scale (Balgobin & Evrard, 2020; Buyle et al., 2019; Cucurachi et al., 2018; Langkau et al., 2023; Piccinno et al., 2016; Tsoy et al., 2020; van der Giesen et al., 2020; van der Hulst et al., 2020; Villares et al., 2017). Methods include process simulation, physics-based models, proxy technologies, participatory methods, and scaling relations. The preferred approach depends on the case at hand and the data availability. When small-scale data is available, the upscaled technology performance can be estimated, as demonstrated by case studies on e.g. chemicals (Piccinno et al., 2016), photovoltaics (Blanco et al., 2020), energy technologies (Caduff et al., 2014), and steel slag (Buyle et al., 2021). For some process types, upscaling guidelines are available, but this is not the case for powder metallurgy and magnet production.

This research aims to quantify the environmental impacts of industrial-scale magnet recycling, based on information available from current small-scale processes and envisioned technology developments. We apply ex-ante LCA to compare recycled magnets to magnets from primary materials, as well as to pinpoint specific areas of concern within the recycling chain. By identifying environmental hotspots, we aim to support technology developers in developing more sustainable solutions. To validate the results, we aim to compare the projected impacts with an industrial reference configuration and a theoretical, thermodynamic minimum impact. Moreover, we aim to find the kind of changes that contribute most to the improvement of environmental performance, by systematically assessing different mechanisms of technology development. This may provide research priorities for other ex-ante LCA studies, while also paving the way for a sustainability-focused R&D agenda.

As opposed to previous research, we aim to study recycling that starts from a range of waste flows and includes the manufacturing of various new magnet types. The processes that are part of the recycling chain were modeled in collaboration with technology developers participating in the SUSMAGPRO project (SUSMAGPRO, 2019). We determined the future performance of each process by considering fundamental process changes, size scaling, internal recycling and process optimization (as described in Section 4.2.4). Moreover, we compared secondary and primary production at the level of NdFeB powders and of finished magnets.

4.2 Methods

4.2.1 Goal & Scope

This ex-ante LCA studied the environmental performance of developing processes for direct alloy recycling of NdFeB magnets. The recycling chain is defined as spanning from waste sorting to secondary magnet manufacturing, resulting in the foreground system

TABLE 4.2: Definition of demonstrator magnets.

	Type of magnet	Application	Dimensions (mm)	Shape	Coating
1	Extruded	EV drive rotor	25 × 14 × 4	Rectangular	Epoxy
2	MIM	Sensor	18 × 16.5 × 2.5	Disk	Phosphate + epoxy
3	Sintered	EV drive rotor	30 × 30 × 50	Block	Epoxy
4	Bonded	Water pump	46.5 × 11 × 2.7	Rectangular	None

depicted in Figure 4.1. We evaluated the life cycle impacts for recovered alloy powders and for magnets made from recycled material (demonstrators). We defined two functional units and investigated alternative processing routes for each. These routes yield products that differ in shape and functionality, making some products incomparable. The first functional unit is 1 kg NdFeB alloy powders, an intermediate output produced by fine-tuning (Figure 4.1b). Six alternative sources of waste magnets (listed in Table C.1) were compared, to assess the effectiveness of recovery. To evaluate magnet remanufacturing, the second functional unit is an amount of magnets containing 1 kg NdFeB, at factory gate. Table 4.2 presents the product alternatives. Note that the reference flow for bonded magnets weighs 1.125 kg including the polymer binder, and the weight of coated magnets also exceeds 1 kg.

The recycling feedstock consists of EoL hard disk drives (HDDs), EV rotors, loudspeakers, industrial pumps, TV speakers and wind turbine magnets (see Appendix C.1, Table C.1). These waste flows were selected for their prominent contribution to NdFeB magnet consumption and waste production (Appendix C.1; van Nielen et al. (2023)). The recycling feedstock is assumed to comprise equal shares of waste magnets from these six sources.

The demonstrator magnets in Table 4.2 represent magnets for specific applications, e.g. an EV drive rotor, as produced and tested in pilot settings. Each demonstrator is the product of a distinct production route in Figure 4.1c, yielding 1) extruded 2) metal injection molding (MIM), 3) sintered and 4) bonded magnets. The studied processes had a TRL of around 4–6, and approached TRL 7 or 8 at the end of the project. Routes 1 and 2 have a lower TRL than the other routes. After extrusion or injection molding, both follow a similar procedure of debinding and sintering.

4.2.2 General inventory data

This study focused on magnet recycling in Europe, and assumed the European average market mix in 2018 for all inputs, as modeled in Ecoinvent 3.8. In Europe, waste is already collected independently of magnet recycling. Therefore, waste collection was excluded from our scope (while magnet liberation was included). Changes in the background system were not considered, as the focus is on changes in the foreground technology. To establish a baseline for comparison, we calculated the environmental impacts of primary magnets produced in Europe. The main raw materials for primary magnets are rare earth metals, for which we adopted the global average market mix (see Section 4.2.5).

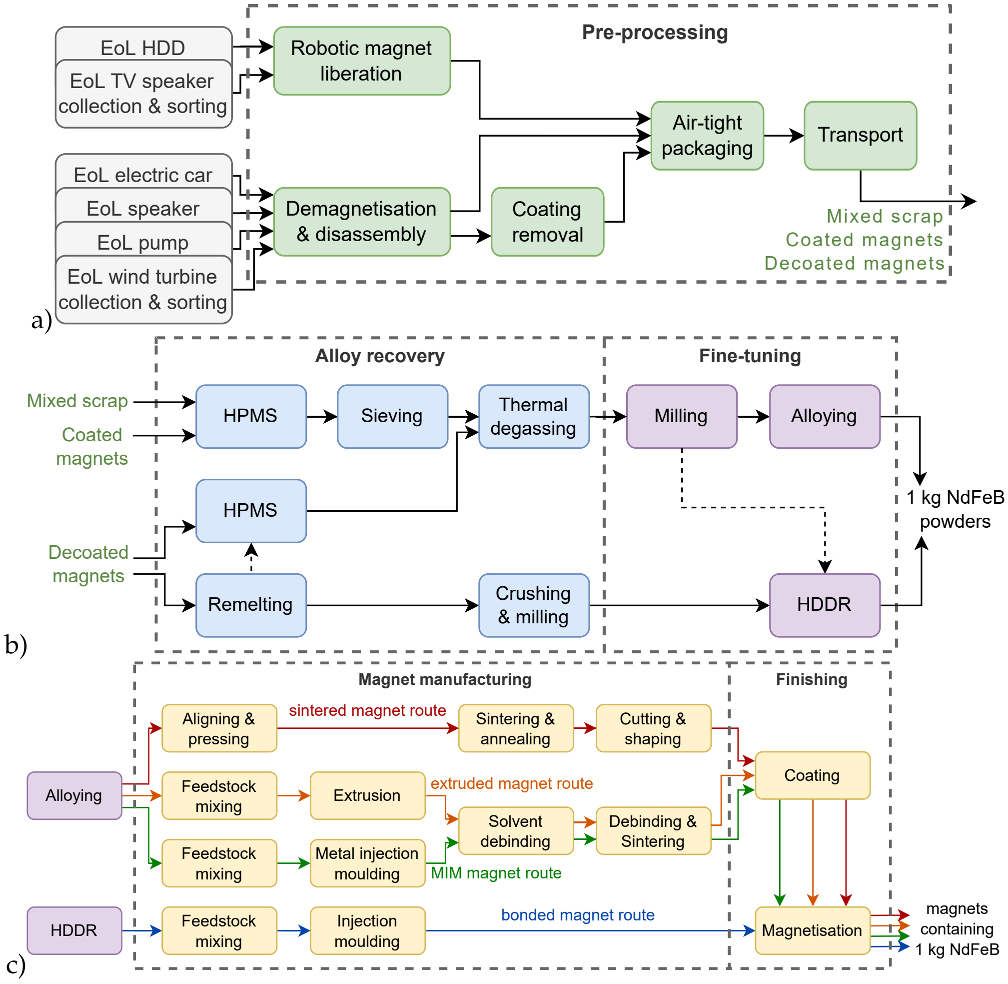

FIGURE 4.1: Flowcharts of the studied recycling routes. The functional units (on the right) are 1 kg NdFeB powders and magnets containing 1 kg NdFeB. The system boundaries include five recycling stages, indicated by dashed outlines. Dashed arrows indicate feasible routes not fully explored in this study. HPMS: hydrogen processing of magnetic scrap; HDDR: hydrogenation–disproportionation–desorption–recombination.

To allow for the evaluation of all possible combinations of process alternatives, a parametrized model was built in ActivityBrowser version 2023.03.03 (Steubing et al., 2020) using Ecoinvent 3.8 (cut-off version) as background database (Wernet et al., 2016). The various waste inputs, recycling routes, output magnets, and technology developments were implemented as flow-scenarios.

4.2.3 Data on small-scale recycling

Data on the current performance of recycling technology was obtained from SUSMAGPRO partners [1] through site visits, measurements, and interviews with technology experts. This resulted in a good understanding of the pilot unit processes and their interdependencies. Experimental process trials allowed to identify technically feasible processes and conditions. Processes with poor performance were discarded and promising processes were developed further. Next, interviews and workshops were conducted, again involving technology developers, to explore potential changes and improvements towards large-scale operation.

4.2.4 Projecting industrial-scale recycling

Impacts of lab-scale technology are not representative for industrial operation (Tsalidis & Korevaar, 2022). Therefore, we aimed to estimate life cycle inventory data for a recycling chain at industrial scale, based on experience with lab-scale processes and initial tests at pilot-scale. This addresses an important and challenging step in prospective technology assessment, as technological reconfigurations are most likely during this phase of development. Given the current maturity of magnet recycling technology, our approach is mostly based on van der Hulst et al. (2020).

We calculated the environmental pressures for a base case and four anticipated configurations. The base case represents recycling at pilot scale, which is close to the present process implementations but with higher operating hours. The anticipated configurations were derived from the pilot set-up by accounting for four developments towards industrial recycling (van der Hulst et al., 2020): process changes, size scaling, internal recycling (a process synergy), and optimization (combining technological learning and economies of scale) (Figure 4.2). The definition and the general approach for each configuration are outlined below, along with a general indication of data sources.

Pilot process is a pre-industrial prototype process, with a TRL around 6. This is close to the TRL of processes for which data was collected, although some processes had a different TRL. The equipment is operated 8 hours per day and 240 days per year. For processes that take more than 8 h per batch, 240 batches per year were assumed. The lifespan of machines was estimated by technical experts, and varies between 8 and 30 years. *Process change* includes fundamental changes of the process or materials. Process changes were identified in exchanges with SUSMAGPRO technology experts. Information on the energy consumption and the weight of some equipment was obtained from equipment manufacturers. *Size scaling* involves increasing dimensions of equipment (upscaling), to achieve higher throughput. The general target capacity for equipment

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[1] A list of the SUSMAGPRO participants is available at https://doi.org/10.3030/821114.

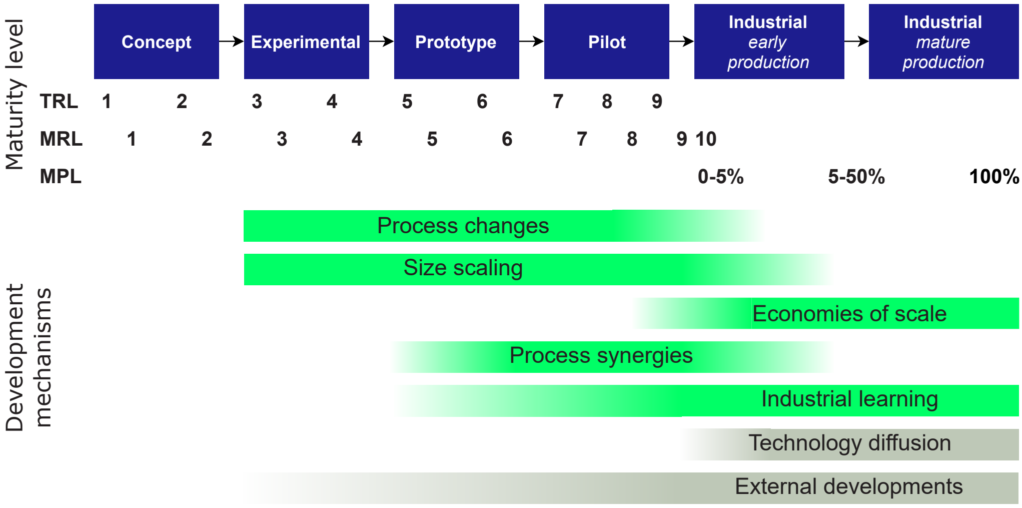

FIGURE 4.2: Mechanisms for technology development, adapted from (Buyle et al., 2019; van der Hulst et al., 2020). Mechanisms in green are addressed in this study; economies of scale and industrial learning are assessed jointly. Mechanisms in grey apply to a broader level of analysis. TRL: technology readiness level; MRL: manufacturing readiness level; MPL: market penetration level.

is 200 t/a. Further scaling beyond 200 t/a is achieved mostly by parallel processing, therefore it would only marginally change the process performance. 16 operating hours per working day are assumed. In some cases, the pilot process performance was extrapolated using scaling relations. *Internal recycling* refers to recycling of waste flows, like solvents and inert gases, and production of NdH_2 from recovered materials. Other process synergies were not considered. *Optimization* accounts for small or difficult improvements, achieved through ongoing industrial learning and economies of scale. The optimized processes operate 24 h per day.

Furthermore, two reference configurations were evaluated. The *industrial reference* processes are mostly modeled after a similar unit process from a comparable sector. For example, MIM of steel powders was used as a proxy for MIM of NdFeB powders. Internal recycling is assumed in the magnet industry. The *theoretical optimum* describes a thermodynamically ideal process, with an energy efficiency of 100%, and no material loss. For all processes with high operating temperatures, a thermodynamic model was constructed to calculate the energy use at large scale and at the theoretical optimum.

All development steps were modeled as cumulative improvements. When no changes were expected for a certain flow or parameter, the performance of the preceding step was applied. The data and assumptions are detailed per process in Appendix C.2 and ESI 1[2]. Using the pedigree matrix by Weidema (1998), we determined data quality indicators between 1.6 for pilot processes and 3.6 for the optimized configuration, see Appendix C.2.1.

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[2] https://www.sciencedirect.com/science/article/pii/S0959652624019012#appSC

4.2.5 Data on primary REE and magnet production

Primary rare earth elements (REEs) are used for two processes: for the production of neodymium hydride (NdH_2) (added during alloying [3]), and for the production of primary magnets (the baseline). The primary supply chain of REEs is modeled after Miranda Xicotencatl et al. (2021). As REE ore sources, we assume the average market mix for 2021–2022, consisting of 10% monazite from Australia, and 82% bastnäsite–monazite and 8% ion adsorption clays from China (USGS, 2023). For simplicity, NdFeB alloy is assumed to consist of 27% neodymium, 72% iron pellets and 1.3% boron carbide (Sprecher et al., 2014). In reality, the alloy also contains other REEs co-produced with Nd such as Dy and Pr.

4.2.6 Impact assessment

We calculated the environmental impacts for 16 impact categories. All except one were calculated with the Environmental Footprint v3.0 impact assessment method (EF) (Fazio et al., 2018), as listed in Appendix C.3. We only deviate from EF for water use impacts, because water extractions and emissions are regionalized in EF but not in Ecoinvent. Water use was assessed using the characterization factors in Appendix C.4.

4.3 Results

4.3.1 Hotspots in magnet recycling at pilot scale

Analysis of the pilot-scale recycling system shows that most impacts arise from alloy fine-tuning and magnet manufacturing. Figure 4.3 shows the environmental hotspots for the sintered demonstrator magnets for EVs. The contribution of fine-tuning stems from jet milling and primary neodymium, added in the form of NdH_2 to ensure good magnetic properties. Other burdens are linked to the production of electricity (36% of climate change impacts) and equipment (52% of human cancer effects). Besides, significant material losses occur during sieving (35%) and cutting magnets to shape (21%). These hotspots were used as guidance for identifying areas of technology improvement. Although pre-processing and finishing have limited impacts, process developments were also investigated within these stages.

4.3.2 Projected impacts of recovering magnet alloys

NdFeB powders can be produced by jet milling, by vibratory milling and by HDDR. Each process yields powders with distinct characteristics; e.g. HDDR powders are suited best for bonded magnet production. Their greenhouse gas (GHG) emissions are compared in Figure 4.4. The impacts are plotted for six waste sources, revealing that the effect of starting material is only small.

All recycled powders have lower GHG emissions than primary powder already at pilot scale. The impacts of jet milling, mainly stemming from inert gases, are closest

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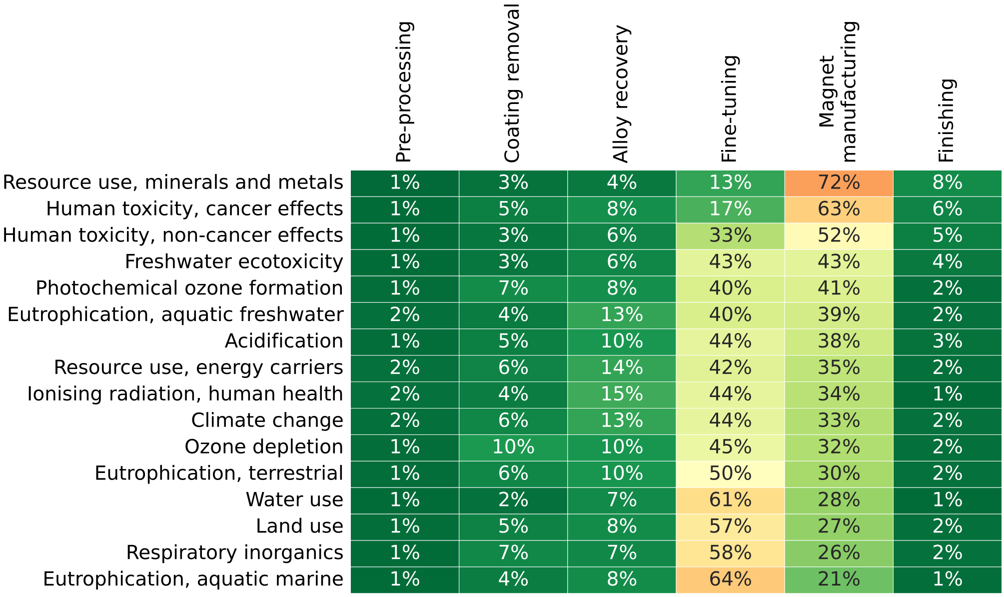
[3] Alloying means mixing metal powders to form an alloy.



	Pre-processing	Coating removal	Alloy recovery	Fine-tuning	Magnet manufacturing	Finishing
Resource use, minerals and metals	1%	3%	4%	13%	72%	8%
Human toxicity, cancer effects	1%	5%	8%	17%	63%	6%
Human toxicity, non-cancer effects	1%	3%	6%	33%	52%	5%
Freshwater ecotoxicity	1%	3%	6%	43%	43%	4%
Photochemical ozone formation	1%	7%	8%	40%	41%	2%
Eutrophication, aquatic freshwater	2%	4%	13%	40%	39%	2%
Acidification	1%	5%	10%	44%	38%	3%
Resource use, energy carriers	2%	6%	14%	42%	35%	2%
Ionising radiation, human health	2%	4%	15%	44%	34%	1%
Climate change	2%	6%	13%	44%	33%	2%
Ozone depletion	1%	10%	10%	45%	32%	2%
Eutrophication, terrestrial	1%	6%	10%	50%	30%	2%
Water use	1%	2%	7%	61%	28%	1%
Land use	1%	5%	8%	57%	27%	2%
Respiratory inorganics	1%	7%	7%	58%	26%	2%
Eutrophication, aquatic marine	1%	4%	8%	64%	21%	1%

FIGURE 4.3: Contribution of recycling stages to the environmental impacts of 1 kg sintered magnets for an EV drive rotor, as calculated for the pilot-scale processes. Stages are defined in Figure 4.1. Coating removal is part of pre-processing.

to primary production, with higher impacts for ionizing radiation and freshwater eutrophication (see ESI 6[4]). When improvements up to internal recycling are implemented, recycling performs better on all environmental indicators. Drivers for impact reduction include energy-efficient equipment and recovery of inert gases.

Vibratory milling and HDDR cause lower impacts than jet milling. The lowest impacts are achieved by combining remelting and HDDR. This is largely because small-scale HPMS uses more energy than remelting, and because no alloying additions are needed for HDDR powder. For comparison, the combination of HPMS and HDDR has impacts similar to remelting with HDDR after size scaling (4.9 and 4.6 kg CO_2-eq.), indicating that the upscaled HPMS process performs similar to remelting. The optimized large-scale recovery processes all have comparable impacts ($\sim$ 3.4 kg CO_2-eq.).

Having analyzed alloy recovery, we now examine the effect of technology development on the whole magnet recycling chain. These further analyses assume equal shares of scrap magnets from each of the six EoL applications, and only consider HDDR applied to remelted material.

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[4] https://ars.els-cdn.com/content/image/1-s2.0-S0959652624019012-mmc2.xlsx

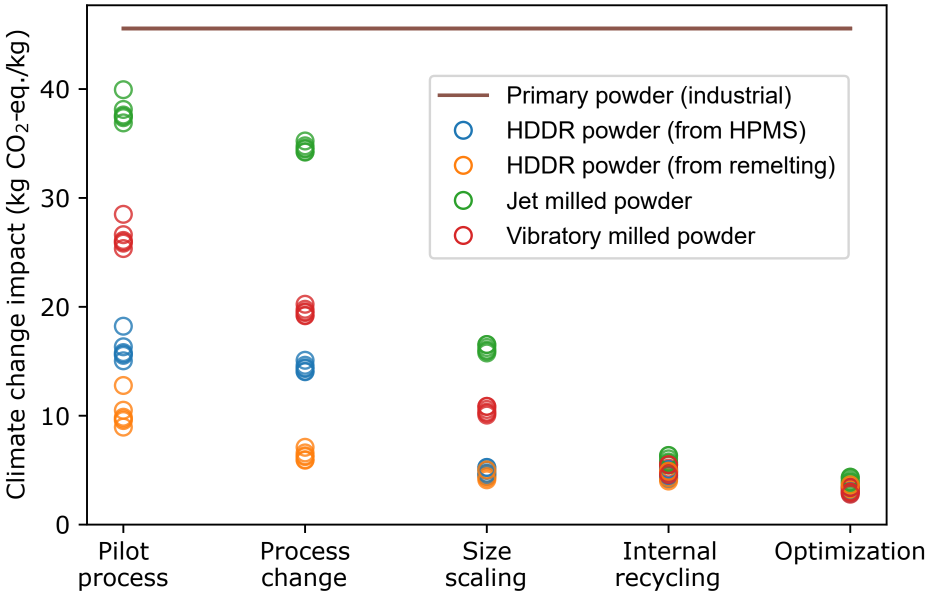

FIGURE 4.4: Climate change impacts of NdFeB powders, from primary origin and recovered from any of six waste sources. Recovery impacts are shown for six waste sources, for the pilot scale and after four cumulative technology improvements (described in Appendix C.2). All outputs are 1 kg of fine powders, hence milled powders have been alloyed with NdH_2.

4.3.3 Projected impacts of magnet recycling

The climate change impacts of recycled magnets are compared on a mass basis in Figure 4.5. For all demonstrator magnets, technology improvements can lower the emissions, possibly even below the industrial reference performance. At pilot scale, the highest GHG emissions are associated with the production of extruded EV demonstrator magnets. MIM magnets are second, followed by sintered magnets. Finally bonded pump magnets have the lowest emissions. The demonstrators with the highest emissions at pilot scale also have the greatest potential for impact reduction. In a large-scale, optimized plant, MIM and extruded magnets have emissions close to those of sintered magnets (10.6, 8.8, and 7.4 kg CO_2-eq. respectively).

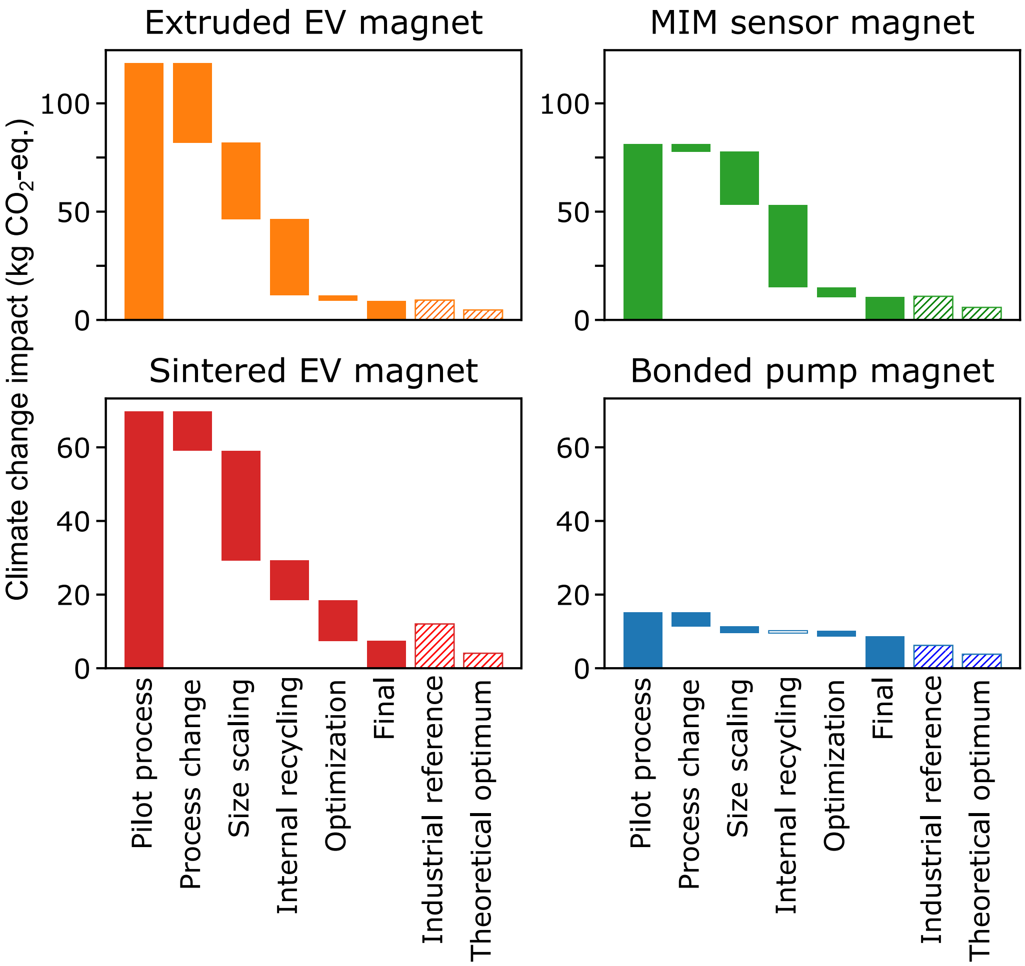

FIGURE 4.5: Climate change impacts of 1 kg recycled demonstrator magnets. Impacts are presented for the pilot scale, after four cumulative technology improvements, and two reference configurations (dashed bars). Note that the *x*-axes are shared.

Figure 4.6 shows that recycled magnets can achieve lower environmental impacts than primary magnets. The reference values for primary magnet production are only plotted for sintered and bonded magnets, because the other two types are currently not produced industrially. For at least 6 of 16 impact categories, recycling already performs superior at pilot scale, e.g. for freshwater ecotoxicity and land use.[5] However, pilot-scale recycling has high climate change impacts, and several improvements are needed to reduce impacts to below primary levels.

Figure 4.6 also illustrates that the results for some impact categories look different from climate change effects. Although extruded magnets have the highest GHG emission at pilot scale, their impact is similar to sintered magnets for freshwater ecotoxicity and land use. Moreover, extruded magnets show a significant decline in impacts as technology improves, highlighting the potential for optimization in this less mature magnet production route.

All types of technology development can contribute to lower impacts (Figure 4.6 and 4.7). The largest reductions are achieved by size scaling and internal recycling, although this depends on the demonstrator magnet. For sintered magnets, a remarkable drop in water use (−88%) is observed due to process changes. This was achieved by improving the insulation of the sintering furnace and thus reducing the cooling water use. Although most process changes are required to enable upscaling, some cause little impact reduction. Specifically for extruded magnets, a major process change is the reduced energy use for degassing of feedstock.

Sintered magnets use more water than other types, especially at pilot scale. Most water is used directly as cooling water, which can be reduced by better insulation and water recirculation. Land use impacts are low for bonded magnets and are very similar for the other three demonstrators. Land use is mostly related to electricity production and REE mining.

Bonded magnets have low impacts in every impact category. This is because bonded magnet manufacturing does not require sintering and annealing, which are very energy-intensive processes. The manufacturing of bonded magnets is already well-developed and leaves little room for improvement. Moreover, bonded magnets do not require additions of (virgin) NdH_2. Most improvements occur in the recovery and fine-tuning (remelting and HDDR). Bonded magnets are produced via the remelting route, which causes less impacts than the HPMS route at pilot scale. After upscaling and optimization, the HPMS route had similar impacts, due to improvements in jet milling, optimization of the HPMS motor and replacement of virgin NdH_2 inputs. These and other major drivers of emission reduction are analyzed in Section 4.3.4.

4.3.4 Key strategies to reduce impacts

This section examines the changes that contribute most to impact reduction and the effect of recycling technology development on four impact categories. At the level of inflows and outflows, impact reduction is mostly driven by electricity, inert gases, and NdFeB

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[5] Depending on the demonstrator, impacts are lower in 6–15 categories. For recycled sintered magnets, the impacts are higher at pilot scale for: ionizing radiation, freshwater eutrophication, human toxicity (cancer effects), energy use, water use, resource use (minerals and metals), and climate change. See also ESI 6.

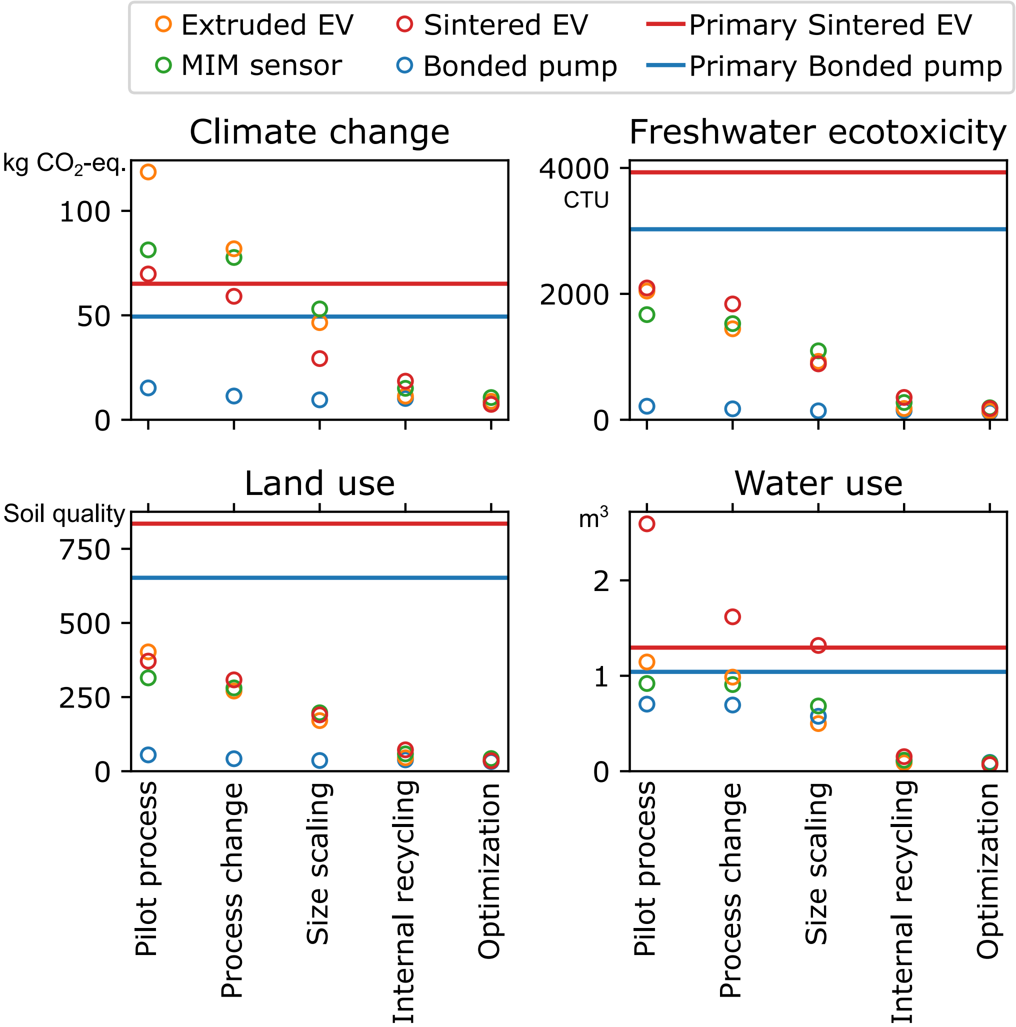

Figure 4.6: Environmental impact changes due to cumulative technology developments in recycling, for four impact categories. The impacts of industrial primary magnets are shown as lines for comparison. The reference flow of each demonstrator magnet contains 1 kg NdFeB. The *x*-axes are shared.

losses. Electricity is used in all processes, and energy savings are often possible by applying optimization measures (see ESI 1). For instance, the HPMS vessel can be rotated at lower speed or less often. Furnaces are more energy-efficient if they are larger or have better insulation. Secondly, inert gases prevent oxidation of NdFeB powder. Recovering these gases, notably from jet milling, avoids energy-intensive gas production and saves 15.9 kWh. Some fresh gas remains needed, to compensate for leakage and for contaminant removal.

Thirdly, a key process improvement is the reduction of NdFeB losses. The highest losses occur in sieving (although the loss is uncertain) and shaping the magnet. Powder sticking to coating residues and too fine powders are lost. These losses can be reduced by optimizing preceding processes. Regarding shaping, the MIM and extrusion routes have a clear advantage: because the shaping process occurs before sintering, internal recycling of shaping losses is easy and no excess material is sintered. Further significant improvements address the use of raw materials, notably NdH_2 by using less or using recycled NdH_2, and solvents by distillation. The NdH_2 content can be minimized without compromising the magnet's performance. To a lesser extent, higher utilization rates of machines reduce toxicity impacts related to their production.

Some process improvements had only marginal environmental benefits. For instance, low impacts are associated with ICP-OES measurements, QR-code scanning (both aim to determine a magnet's composition), magnetization, coating of magnets, and H_2 use. Consequently, additional measurements are worthwhile if they help to reduce material losses. Changing the mentioned processes may bring some benefits, but would not change the overall outcomes.

For each recycled magnet type, the trend of impact categories is remarkably similar (Figure 4.7). This demonstrates that all impacts are reduced by the process improvements. For the bonded magnet demonstrator, the implementation of internal recycling of cooling water has mixed effects. While the water use is reduced (-82%), the impacts of all other indicators increase, albeit slightly. The effect of cooling water recycling also affects other demonstrators, although less visible in Figure 4.7 because of the positive effect of other internal recycling solutions.

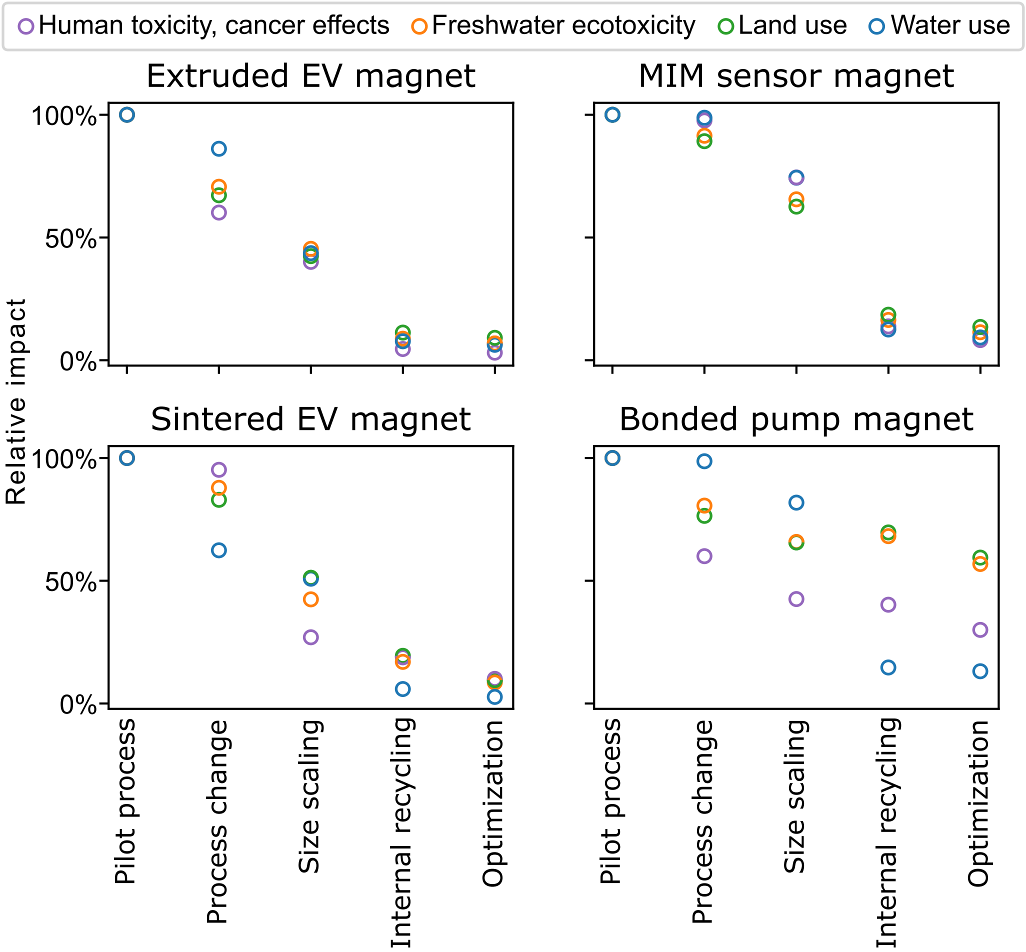

FIGURE 4.7: Environmental impact changes due to projected technology developments for recycled demonstrator magnets, for selected impact categories. Other impact categories are provided in Appendix C.5. The values are plotted relative to the impacts at pilot scale and the *x*-axes are shared.

4.4 Discussion

4.4.1 Validation

The high impact of pilot-scale recycling is due to inefficient small-scale processing, as observed throughout manufacturing (Gutowski et al., 2017, 2009). After implementing all projected technology developments, the emissions of recycling are close to the industrial reference values and still above the theoretical optimum (see Figure 4.5). This comparison suggests that the projections provide a realistic endpoint. However, the final GHG emissions of recycled sintered EV magnets fall below the industrial reference. This may indicate overly optimistic assumptions, e.g. regarding cutting and shaping losses.

In line with the low impacts of bonded magnets, these demonstrators also have the lowest theoretical optimum impact in all impact categories. This is due to the low number of high-temperature processes needed for recycled bonded magnets (see Section 4.2.4).

4.4.2 Uncertainties and limitations

The characterized results are in agreement with previous studies for recycled sintered magnets. Using pilot-scale data, 1 kg recycled sintered magnets cause 70 kg CO_2-eq. emissions. When implementing anticipated process changes, size scaling and internal recycling, the impact reduces to 22.9 kg CO_2-eq./kg. With further optimization, 8.9 kg CO_2-eq./kg may be achieved (Figure 4.5). The latter two numbers are in line with the values in Table 4.1. Although some previous studies were not explicit about the technology scale, it seems that they assumed industrialization. It was not possible to identify the origin of the large difference with the findings by Walachowicz et al. (2014).

Waste collection is excluded from the scope of this study. This is justified for the environmental impact assessment, because waste is already collected in Europe, independent of magnet recycling. Only in identification and disassembly (covered by this study), additional efforts are needed. In other regions, waste collection needs to be set up before recycling is possible. This results in additional impacts, that should be divided over the recovered materials.

To address the inherent uncertainty associated with an emerging technology, this research presented the effect of different developments separately. Not all changes might be implemented as anticipated, therefore the final performance could deviate from the impact after the ‘optimization’ step. Although ‘process changes’ are fundamental and uncertain, several changes have been tested successfully in SUSMAGPRO pilots. Size scaling comes with the challenge to guarantee consistency throughout a batch. This could, for example, limit the scale of HDDR. Internal recycling is often only feasible for large-scale facilities. Optimization might take more time and effort, particularly for processes with a low TRL.

This study evaluated changes in recycling processes, not in the wider economy. The effect of background system changes is illustrated by Appendix C.5, which shows that a switch to electricity from other countries can significantly increase environmental impacts. Additionally, it shows the important contribution of renewable electricity to cleaner recycling (further investigated by Miranda Xicotencatl et al. (2024)). Since recovery relies more on electricity than REE mining, recycling benefits most, and the advantage over primary magnets remains.

A relatively uncertain part of the LCA model is the equipment. While the best available estimates for the weight of machines were used, their production capacity and technical lifespan are uncertain for the novel processes considered here. Besides, the Ecoinvent data for machine compositions may not be representative. For the shaping process, a metal working machine was assumed. This machine contains 10% copper by weight, whereas an unspecified industrial machine (1.4% copper) was deemed more representative for other processes. These uncertainties mainly affect abiotic resource depletion and to a lesser extent toxicity impacts.

Uncertainties also exist in the production of primary magnets, mainly related to the source of REEs. Different rare earth deposits vary in their environmental impacts, as shown by previous research (Bailey et al., 2020; Marx et al., 2018; Miranda Xicotencatl et al., 2021). This research assumed the current market mix of REE production, but shifts in this mix can significantly alter the future environmental profile.

4.4.3 Methodological reflection

Notwithstanding the large number of processes involved in magnet recycling and manufacturing, we obtained a comprehensive insight in the development prospects. The modeling of processes became structured because the same types of technology developments were assessed for each process. At the same time, the grouping by type of development facilitated the interpretation of results.

The reliability of the outcomes was improved by combining different approaches for estimating process performance: lab-scale data, pilot process measurements, industrial reference values, and thermodynamic models. Lab-scale and pilot-scale data were useful for determining the focus of further analysis and for calibrating upscaling models. Industrial proxies supported estimations of the optimization potential. Thermodynamic models helped to identify the drivers for energy use, somewhat similar to exergy analysis (Dincer & Rosen, 2013; Granovskii et al., 2008). By contrasting process data from all three sources, inconsistencies were identified and corrected. This allowed for selective collection of additional data.

Environmental assessments have a rather different approach to dealing with uncertainty compared to cost studies. An environmental assessment model usually assumes a ‘flawless’ process operation. Material losses are accounted for, but this is not the case for equipment down-time, energy use during idling, additional steps, or safety measures. All these unexpected setbacks are typically accounted for in cost calculations by a contingency factor. The less mature a technology is, the higher the contingency costs. For a small pilot plant (TRL 6), a contingency factor of 20–35% is recommended (AACE, 1991). In the case of NdFeB magnet recycling, two occurrences could negatively influence the process performance. Firstly, some batches might be discarded because the quality criteria are not met. This can significantly lower the net output. Secondly, safety measures are needed to handle magnetized magnets and fine pyrophoric NdFeB powders. These effects can be included in future studies for a more complete environmental profile.

The relative effect of various technology developments differs from the results in a case study on photovoltaic laminate (van der Hulst et al., 2020). In the present study, all types of technology developments contributed to improvements in environmental performance. The relevance of each differs per demonstrator and impact category. van der Hulst et al. (2020) concluded that process changes have the largest effect. Hence no generalizations from one ex-ante LCA study to the next can be made. Future studies of other technologies should therefore assess all development mechanisms.

4.4.4 Recommendations for recycling technology

This study yielded new scalable unit process models, applicable beyond the case of magnet recycling. Specifically, thermodynamic models were created for jet milling, feedstock mixing (mixing powders with polymer binders), and solvent debinding (see ESI 1). Hereby, we extended the available set LCA models of powder metallurgical processes (Azevedo et al., 2018; Raoufi et al., 2020).

This research contributed to the development of NdFeB magnet recycling by providing guidance for more sustainable process improvements. Section 4.3.4 provided guidance for technology developers to define a focus for further improvements. Most improvements apply to multiple magnet production routes, signaling opportunities for knowledge cross-over. For example, the large-scale pelletizing process used for bonded magnets can be adapted and adopted to improve the feedstock preparation for MIM and extruded magnets. Sintering is applied in three manufacturing routes, and although the settings depend on the presence of a binder, best practices could be exchanged for energy-efficient design and operation.

The four routes for manufacturing NdFeB magnets have distinct environmental profiles. At current technology levels, recycled bonded magnets offer the largest environmental benefits. The demonstrator that has the highest impact is different per impact category. Due to the anticipated technology developments, the impacts of the manufacturing routes will converge. Consequently, the industrial-scale versions of all routes perform significantly better than their primary production counterparts. The choice for either manufacturing route should not be based on the current performance, but rather on the functional requirements and on the expected ease of improvement.

Future research could investigate the profitability of recycling, while considering fluctuating REE prices. Profitability is likely to increase through upscaling and efficiency gains. Prospective assessments suggest that hydrometallurgical recycling is cost-competitive (Beylot et al., 2020; Chowdhury et al., 2021; Elwert et al., 2017).

4.5 Conclusion

This study investigated how advancements in magnet recycling technology can reduce its environmental impacts, and compared these impacts to those of the primary production route. Recycled NdFeB powders were shown to have lower environmental impacts than powders from primary sources, already for pilot-scale recovery. For recycling and magnet manufacturing combined, all improvements together can result in 80% lower environmental impacts compared to primary magnets for most impact categories. The industrial-scale performance is achieved by upscaling and optimizing the unit processes, as quantified in this study.

The most effective identified improvements address three environmental hotspots: electricity use, inert gas use, and losses of NdFeB material (Section 4.3.4). Therefore, large impact reduction can be achieved through internal recycling and by minimizing losses of NdFeB material. Size scaling effects contribute to lower heat losses, significantly minimizing the energy consumption. Process changes and optimization also contributed to lower impacts. While the major improvements address the magnet manufacturing stage, the relative importance of pre-processing increases in optimized recycling systems.

This case study shows that innovation and emerging technology development can result in large reductions of environmental impacts. Although most changes are motivated by efficiency and costs, the environment also benefits. An exception could be internal recycling measures, which require additional investments in recovery equipment, and only become financially viable at larger scales.

Based on our findings, the rare earth permanent magnet industry can reduce its environmental impacts in three ways. First, by incorporating more recycled materials in magnets. Second, by investing in process innovation for cleaner production and recycling. Third, by upscaling and applying more resource-efficient manufacturing routes like MIM and extrusion when suitable. With these focus points, NdFeB magnets can continue to enable clean electricity production and consumption.

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5

Accounting for learning in prospective LCA: Theory and practical guidance

This chapter has been submitted for publication as: Van Nielen, S. S., Tukker, A., & Kleijn, R. (2024). Accounting for learning in prospective LCA: Theory and practical guidance. *Journal of Industrial Ecology (under review).*

Abstract

Learning is important for the development of industrially deployed technologies, and learning curves have been used to determine future production costs. Although the effect of learning on costs has been extensively studied, very little evidence exists for its effect on environmental impacts, and a conceptual underpinning is lacking. Based on a review of theoretical foundations and empirical evidence, this study presents a procedure for assessing learning of industrial processes in ex-ante and prospective life cycle assessment (LCA). We argue that learning involves operational or organizational changes, which are motivated by incentives. Therefore, environmental impacts may follow a learning curve trend if the origins of impacts coincide with dominant incentives. A key observation is that the results may vary by impact category, and certain impacts may not decline at all. We developed guidelines to evaluate environmental learning effects and rates, and illustrated these with examples in an LCA context. Further research is needed to expand the evidence base for environmental effects of learning, by re-interpreting datasets of existing technologies to determine their learning rates.

5.1 Introduction

5.1.1 Anticipating future environmental performance

New technologies are rapidly being developed, which is crucial for shaping a sustainable society. Especially for breakthrough technologies with a large deployment potential, it is important to understand potential efficiency gains and the associated timeframe resulting from learning, optimization, and upscaling. These insights direct investments towards environmentally more promising technologies, aiding investment planning and decision-making (Sandén & Karlström, 2007).

Technological learning is the improved performance of a product or process over time due to increasing experience and knowledge. It has been identified as one of the mechanisms responsible for decreasing environmental impacts, mainly for mature technologies (Junginger et al., 2008). Technology forecasting guidelines (NETL, 2013; Roussanaly et al., 2021; Rubin, 2019) distinguish a formative phase with fundamental technology changes, followed by an industrial deployment phase with incremental and more predictable development driven by learning. The industrial phase typically starts at technology readiness level (TRL) 9. Learning at the level of individuals, teams, companies, and sectors is a key to achieving sustainability goals (Feeney et al., 2023). For instance, technological learning reduced emissions for the production of solar panels (Kavlak et al., 2018; Louwen & van Sark, 2020), metals (Gutowski et al., 2013), and chemicals (Ramírez & Worrell, 2006). This paper addresses these learning effects.

Learning can significantly improve industrial technologies, but solid evidence only exists for the economic implications. In business economics, learning curves refer to the phenomenon of decreasing costs of production. The downward trend of costs as a function of cumulative production is strongly supported by empirical evidence, both for companies and for industrial sectors (Dutton & Thomas, 1984). It has been postulated that the effect of learning on environmental impacts (‘environmental learning’) resembles the effect on costs (Faber et al., 2022; Thomassen et al., 2020; van der Hulst et al., 2020). However, evidence and justification are lacking, as explained below. For either learning curve, tracing causality is difficult (Grubb et al., 2021; Kavlak et al., 2018).

5.1.2 Missing insight in environmental learning curves

To assess the future environmental impacts of novel technologies, several ex-ante life cycle assessment (LCA) modelling practices have been developed (Arvidsson et al., 2018; Buyle et al., 2019; Thomassen et al., 2019; Tsoy et al., 2020; van der Giesen et al., 2020). Good guidelines exist for the formative phase of technology innovation, extrapolation of lab-scale data to large scale, and for modelling background systems (Thonemann et al., 2020; Tsoy et al., 2020). Yet, the subsequent industrial phase, including learning and scaling effects, has been addressed by only few LCA-related studies (Buyle et al., 2019; Caduff et al., 2012; Thomassen et al., 2020; van der Hulst et al., 2020), and a comprehensive approach is lacking for predicting these effects.

A major uncertainty is if and how cost-based learning curves can be translated to environmental learning curves (Buyle et al., 2019). Bergesen & Suh (2016) and Thomassen et al. (2020) suggest that technological development is the main driver for both costs and impacts, as illustrated by the middle column in Table 5.1. However, this is challenged by examples of purely cost-oriented learning effects (Table 5.1, left column) and design trade-offs between costs and emissions (Table 5.1, right column). These cases indicate a disconnect between costs and emissions. Furthermore, it is uncertain whether learning can be generalized to other products, industries, or impact categories. Various environmental impacts often arise from different processes, each process with a distinct learning trend. Therefore, extrapolating learning curves from one impact category to another requires careful consideration. Thomassen2020 provided guidelines for technological learning without differentiating between impact categories, leaving unclear what environmental impact is affected. van der Hulst et al. (2020) constructed a learning curve for greenhouse gas (GHG) emissions, questioning if other impact categories would show a similar correlation.

TABLE 5.1: Learning effects, grouped by economic and environmental consequences.

Factors that decrease costs with minimal environmental effects [a]	Technology changes with both environmental and financial benefits	Technology changes with environmental benefits, possibly increasing costs
Budget overruns due to delays Cost of capital (interest payments) Regulatory fees (permitting) Commercial and legal risk mitigation Insurance costs Overhead costs Marketing costs for new products Single orders rather than bulk purchases	Improved resource and energy efficiency of processes (Bergesen & Suh, 2016) Improved design of process equipment (Arundel et al., 2008) Improved use-phase efficiency (Weiss et al., 2010) Higher equipment utilization and lower depreciation (Rubin, 2019)	End-of-pipe solutions (Frondel et al., 2004) Product improvement to meet the preferences of environmentally conscious consumers (Cai & Li, 2018) Shift to inputs [b] or suppliers with lower impacts (Bergesen & Suh, 2016) Material or process substitution (Ferioli et al., 2009)

[a] Source: Roussanaly et al. (2021); Santhakumar et al. (2021)

[b] Inputs include feedstock materials, energy carriers, and equipment.

5.1.3 Research aim and approach

The objective of this paper is to investigate how learning effects influence environmental impacts, and to develop an approach to assess environmental learning in product systems. We combined a review of empirical results with an investigation of the theoretical foundations and driving mechanisms of environmental learning. Based on these insights, we aim to develop recommendations to guide forward-looking LCA studies. Specifically, our guidance addresses two questions: 1) Is a learning curve expected for a given environmental impact category? 2) How can the learning rate be quantified?

This study considers learning effects on the market-average environmental impacts of specific products, rather than product categories. The scope is limited to industrial goods, manufactured products with a TRL ≥ 9, and commodities, thus excluding services. We focus on patterns at a sector-level rather than the level of individual firms or economy-wide dynamics.

This paper argues that environmental learning curves do not always follow the trend as costs, but can be explained by technology changes that are motivated by external incentives. §5.2 provides an overview of the concept of learning, learning mechanisms and their role in technology development. §5.3 presents a systematic procedure to account for learning in ex-ante LCA studies, illustrated by examples in §5.4.

5.2 Technological learning theory and evidence

5.2.1 Overview of technology development

Research into the future impacts of emerging technologies has revealed several technology development mechanisms, that operate in both the formative and industrial phase. van der Hulst et al. (2020) identified five mechanisms: process changes, size scaling, process synergies, industrial learning (or technological learning) and external developments. Buyle et al. (2021) additionally introduces technology diffusion. Based on the literature, Figure 5.1 illustrates the various mechanisms and the maturity levels at which they become active.

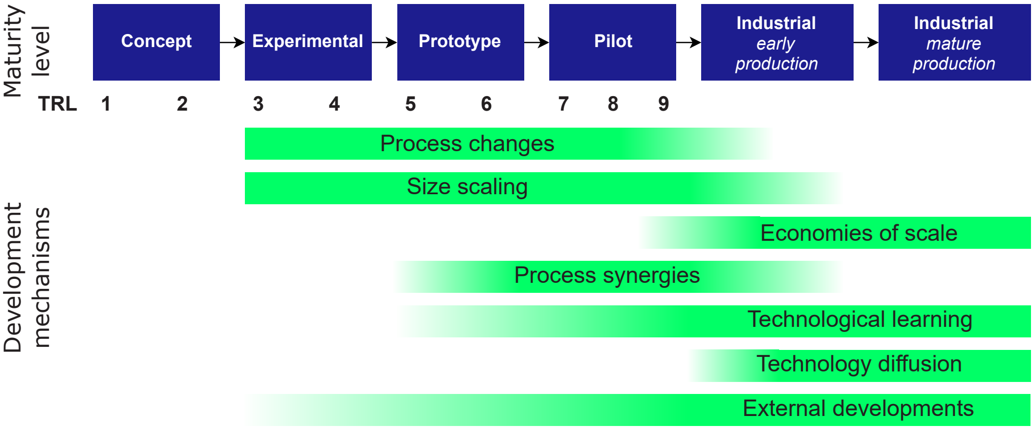

FIGURE 5.1: Stages and mechanism in technology development. Technological learning is also referred to as industrial learning. TRL, technology readiness level. Based on Buyle et al. (2019); van der Hulst et al. (2020).

Except for technology diffusion and external developments, each mechanism implies changing the process or its operation by workers. These changes become more incremental as the maturity increases, partly because the low-hanging fruit for optimization has been picked and partly due to tighter integration with the industrial ecosystem. Figure 5.1 distinguishes between size scaling of unit processes and economies of scale due to company size. These scaling effects are often included in the learning effect because they occur simultaneously and are interconnected (Dutton & Thomas, 1984; Kavlak et al., 2018).

Besides classifying technology developments as process-level changes (Figure 5.1), they can also be differentiated by organizational mechanisms. These include technology adoption (installing improved equipment), novel knowledge combinations, imitation and reverse engineering (Arundel et al., 2008), and knowledge spillovers from other companies (Clarke et al., 2008). Technological learning mechanisms include learning-by-doing, learning-by-using, learning-by-interacting, and R&D (Junginger et al., 2008).

5.2.2 The learning curve

In cost-based learning studies, the mechanisms of technology development during the industrial phase are combined into a continuous trend. This trend is most often described by Equation 5.1 (Dutton & Thomas, 1984). According to the equation, production costs per unit C decline from the initial level C_0 as a function of the cumulative production P (in units or kg). Here, P_0 is the initial production, and exponent a defines the slope of the learning curve (negative values indicate a downward trend) (Dutton & Thomas, 1984). Eq. 5.2 defines the learning rate (LR) as the percentage cost reduction after a doubling of cumulative production.

$$C = C_0 \cdot \left(\frac{P}{P_0}\right)^a \tag{5.1}$$

$$LR = 1 - 2^a \tag{5.2}$$

Eq. 5.1 allows to extrapolate historical trends, with greater accuracy if more data points are available. The log-linear relation applies from TRL 9 onwards, as underlined by C_0 which is the cost of the first unit produced (Rubin, 2019).

Historic data are often unavailable for emerging technologies, therefore alternative approaches have been developed. Santhakumar et al. (2021) recommend to combine a bottom-up cost model (to identify cost drivers) and a component-based learning curve. This implies that the costs are broken down by constituent parts or processes, each of which is modelled by its own learning curve (Ferioli et al., 2009). The component learning rates can be based on similar technologies (proxies). Alternatively, the decomposition approach introduces a separate learning curve for each upstream process, which sum up to a combined learning trend (Kavlak et al., 2018; Nadeau et al., 2010). As Eq. 5.1 indicates, processes with a longer production history tend to progress slower for a given increase of cumulative production. Decomposition is based on the actual supply chain, whereas component-based learning looks at similar technologies and components. Both approaches provide a detailed view of the learning effects on components. For more developed technologies, other approaches exist (see Appendix D.1.1).

5.2.3 The technological learning process

Learning in industrial and organizational context is a change in behavior that aims for improved performance (Sterman, 2002). The concept of learning originates from behavioral sciences, where it describes how individuals and organizations gain experience over time (Fiol & Lyles, 1985; Lapré et al., 2000). Organizational learning is defined as the process of creating, acquiring and transferring knowledge and changing behavior accordingly (Fiol & Lyles, 1985).

During the learning process (Figure 5.2), technology changes—whether radical or incremental—are implemented in response to perceived effects and the operator's motivations. Although inventions and opportunities for improvement may emerge serendipitously, their adoption is a deliberate decision (Chandler & Hwang, 2015). Even if unconscious changes occur, they are influenced by stakeholder demands and market pressures, shaping the proliferation of technologies. Technology change involves new working practices, implying organizational or behavioral change (Dutton & Thomas, 1984). These changes require a motivation to overcome the inertia of entrenched practices.

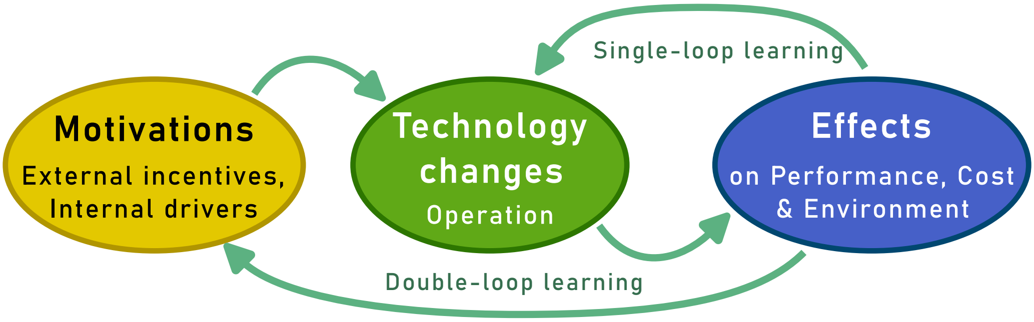

FIGURE 5.2: Schematic representation of learning in technology development, inspired by the conceptualization of organizational learning by Argyris & Schön (Fiol & Lyles, 1985).

Given this context, to know the direction and rate of changes from learning, it is important to understand the prevailing incentives. As Figure 5.2 illustrates, the effects of a technology are evaluated against an actor's motivations, resulting in a decision to change or not. Motivations for individual employees consist of recognition and reward policies and their personal drivers. Motivations for companies and sectors are market selection criteria and other stakeholder pressures, as these determine the success and proliferation of companies (Henderson & Stern, 2004). Stakeholder pressures can incentivize cost reductions but also enhance other indicators like aesthetics, safety and performance (McCollum et al., 2017). These motivations can often be translated to management performance indicators (KPIs). §5.2.4 elaborates on each element in Figure 5.2.

5.2.4 From motivations to environmental effects

5.2.4.1 Motivations

Due to learning, products or processes can develop lower environmental impacts if this aligns with the dominant motivations. Environmental effects can result either directly (intentional impact reduction) or indirectly (as side-effect of pursuing other goals). In many mature markets, motivations for cost reduction dominate. This explains the high predictive value and wide applicability of cost-based learning curves. Gradual trends emerge due to relatively constant prices and cost pressures. In almost all decisions in industrial companies, costs are considered (Merchant & Shields, 1993). However, not all decisions lead to lower cost, because other product attributes also play a role (McCollum et al., 2017).

On longer time scales, motivations can change as a result of double-loop learning, i.e. learning with respect to goals and values (Fiol & Lyles, 1985). As Figure 5.2 indicates, double-loop learning involves changing motivations, sometimes mediated by shifting stakeholder pressures. Double-loop learning, also referred to as *transformative learning*, applies to systems involved in a societal transition (Feeney et al., 2023; Lankester, 2013). These transitions are difficult to predict, adding uncertainty to the analysis.

Sections 5.2.4.2 and 5.2.4.3 review the types of motivations, as uncovered by eco-innovations literature. Although eco-innovation is different from environmental learning (the former emphasizes the novelty of changes), the motivations for both most likely overlap. We distinguish between external incentives (from stakeholder pressures) and internal drivers. External incentives are most relevant for industry-wide trends of learning, while internal drivers help to explain the differences among organizations.

5.2.4.2 External incentives

In general, company decisions—also those affecting environmental impacts—are influenced by market pressures, e.g. competitive forces (Porter, 1979) and institutional pressures (DiMaggio & Powell, 1983). For eco-innovations, these incentives have been classified based on various theoretical backgrounds. Nevertheless, different studies have identified broadly similar sets of drivers (Hojnik & Ruzzier, 2016). Based on established theoretical frameworks (Cai & Li, 2018; Hojnik & Ruzzier, 2016), five groups of external incentives can be distinguished: customer preferences, competitive pressure, restrictive environmental regulation, market-stimulating regulation, and investor and partner preferences.

These pressures are exerted by stakeholders on a producing company. Normative pressure is exerted by customers, and also emerges in institutional forms from investors, value chain partners and societal organizations. Organizations have to comply with societal norms if they want to ensure their legitimacy and access to resources (Hojnik & Ruzzier, 2016). Besides, shareholder value may be created by avoiding environmental risks and liabilities (Sarkis et al., 2010). Governments can apply restrictive or market-stimulating regulation. Competitive pressure can encourage adoption of technologies used by others (DiMaggio & Powell, 1983).

The importance of different drivers was determined by several empirical studies. For instance, eco-innovation was found to correlate with competitive pressure, market-stimulating regulation, and customer green demand for Chinese companies (Cai & Li, 2018). Five major drivers were reported for eco-innovation in Germany: existing and expected regulations, voluntary codes and industry agreements, customer demand, and competitor moves (Horbach et al., 2012). These incentives affected various environmental impacts differently. Cost savings are an important motivation for reducing energy and material use, while regulations mostly encourage lower air, water and noise emissions, and customer requirements influence impacts related to waste and hazardous substances (Horbach et al., 2012).

5.2.4.3 Internal drivers

Next to external incentives, learning is influenced by company-internal factors (Hojnik & Ruzzier, 2016). Already in 1984, it was noted that "the learning rate is neither fixed nor automatic, leading to the question how it is managed best" (Dutton & Thomas, 1984). Cai & Li (2018) distinguish two types of internal drivers: organizational capabilities and technological capabilities. Managers are responsible for defining sustainability KPIs and integrating them into business models (Schaltegger et al., 2012), supported by (physical) resources and environmental management systems (EMSs) (Sarkis et al., 2010). Differences in management, intrinsic motivation and internally-defined performance targets lead to deviations from the average market trend.

For energy efficiency in manufacturing companies, empirical studies confirm the importance of organizational and management variables (Solnørdal & Foss, 2018). A relevant finding for sector-level learning trends is that "innovation breeds innovation" (Cai & Li, 2018), i.e., organizations can use the capabilities gained during a learning process to capture further opportunities. Therefore, more innovative sectors may respond faster to incentives and exhibit higher learning rates.

5.2.4.4 Technology changes

In response to the motivations described above, operational or technological changes can be made that reduce environmental impacts (see Figure 5.2). Learning can alter any operational parameter. Parameters with environmental effects include energy use, material use, waste or discards, equipment use, waste treatment or use-phase efficiency. Some effects are unintentional, for example, manufacturers may replace a material for aesthetic reasons, while the new material has lower or higher environmental burdens. Examples of technology changes are listed in Table 5.1.

A specific technology change is the upscaling of production capacity. In line with Figure 5.2, the pursuit of economies of scale (motivation) leads to upscaling (technology change), resulting in cost reductions (effect). Scaling typically provides environmental benefits, although diseconomies of scale also occur (Wilson, 2012). Upscaling and learning are intertwined, because the experience gained from demonstration plants is essential to develop a process at larger scale, as exemplified by wind turbines (Caduff et al., 2012). Additionally, faster material processing due to learning would increase the production capacity, particularly for modular technologies (Wilson et al., 2020).

5.2.4.5 Effects on performance, cost, and environment

Technology changes in products and processes can have both positive and negative environmental effects. Manufacturers may respond to stakeholder pressures, including preferences for certain product characteristics. Some characteristics align while others conflict with sustainability performance. For instance, the consumer preference for larger passenger vehicles and refrigerators has resulted in decreasing energy efficiency during certain time periods, whereas the efficiency increased during other periods (Dahmus, 2014). Besides, process innovations may have diverging effects across environmental domains (Chen et al., 2022; Kammerer, 2009).

In specific cases, the prevailing incentives may lead to increased emissions. For example, customers can prioritize a low price, neglecting external costs. Alternatively, customers can prefer higher performance (better resolution, larger products). Competitive pressures can provoke mimicry of competitors and a return to conventional practices. Local regulation can cause burden shifting to other regions or impact categories. Burden shifting may also result from substitution of raw materials or processes (Yu et al., 2016). For instance, a raw material can be replaced by a cheaper but more polluting alternative, while process automation may reduce waste and space occupation, but requires more energy and machines (Moreau et al., 2021).

Some learning effects only or predominantly influence costs, while economic and environmental benefits can also be achieved simultaneously (Table 5.1, left and middle column). E.g., cost-driven industries with high energy intensity invest more in energy efficiency (Kalantzis & Niakaros, 2020), boosting environmental learning for fuel-related emissions.

5.2.5 Evidence for environmental learning

Thomassen et al. (2020) reviewed 105 studies on technological learning, identifying only four studies that used or derived a learning rate for the environmental impact: Görig & Breyer (2016); Louwen et al. (2016); Stamford & Azapagic (2018); and Yuan et al. (2018). We identified eight more studies[1] using Google Scholar and the keywords in Appendix D.2, and review these below. The sample size of 12 papers is sufficient for this conceptual paper, as it is not a systematic review.

Eight of twelve reviewed studies assessed only a single environmental indicator. The most common indicator was energy consumption (‘cumulative energy demand’ or ‘specific energy consumption’, in 7 studies), followed by GHG or CO_2 emissions (3 studies). Only Stamford & Azapagic (2018) systematically studied a range of impact categories using LCA, and found a different trend for each impact category. Except for CO_2 absorption, all technologies had a TRL above 9. Besides, it is remarkable that five studies investigated photovoltaic (PV) technology.

The learning rates varied widely, as shown in Figure 5.3 for the reviewed studies. Even for a single indicator, reported learning rates vary significantly, ranging from 3% to 29% for energy use across products. Interestingly, this range largely coincides with the learning rates reported for use-phase energy consumption, 18 ± 9% (Weiss et al., 2010). The differences partly arise from the functional unit definition (area or rated power of PV panels) and whether the thermodynamic minimum is considered (Ramírez & Worrell, 2006) (see §5.3.5).

For material use and waste generation, the learning rate varies depending on the material. PV manufacturers have reduced the need for silver more than the need for silicon (Louwen & van Sark, 2020). These results align with the conceptualization of learning discussed above. For the early PV industry, strong incentives existed for cost reduction, therefore the more valuable silver input was reduced more strongly. Additionally, improved power production efficiency contributed to a decrease in costs per functional unit (the peak power output) (Kavlak et al., 2018; Louwen & van Sark, 2020).

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[1] Bergesen & Suh (2016); Brucker et al. (2014); Gutowski et al. (2013); Hettinga et al. (2009); Lapré et al. (2000); Louwen et al. (2020); Ramírez & Worrell (2006); Rochedo & Szklo (2013)

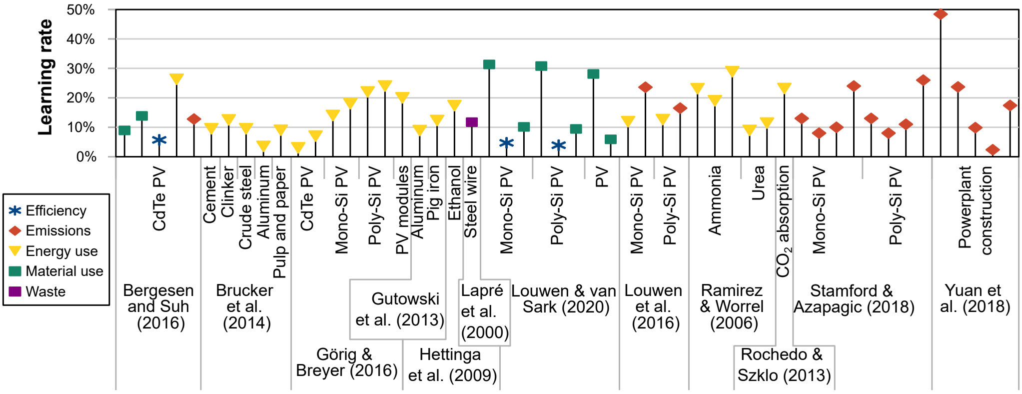

FIGURE 5.3: Learning rates for environmental indicators, as derived in literature case studies. The learning rate is the impact reduction after a doubling of cumulative production (see Eq 5.1).

5.3 Accounting for learning in LCA

Drawing from the insights in §5.2, we developed a procedure for evaluating environmental learning effects. Figure 5.4 outlines the six involved steps, which are elaborated below. As in any LCA, the first step is a scope definition. The learning rates of supply chain processes may vary (§5.2.2), while environmental hotspots differ across impact categories. Therefore, step 2 identifies these hotspots. Learning is driven by incentives arising from stakeholder pressures (§5.2.3), as analyzed in step 3. Step 4 compares these incentives to environmental hotspots, to determine if environmental learning is expected. Step 5 quantifies the learning rate, enabling LCA practitioners to incorporate technological learning effects in prospective assessments. Estimates of future impacts involve uncertainty, which is assessed in the final step.

5.3.1 Define the scope

The scope is defined by choosing an industrial product and an impact category. Since processes with a short production history progress faster, these should be included in the foreground system. Note that the appropriate scope of a process varies from case to case. It can be a single unit operation if the operations have diverging TRLs, or a whole manufacturing line.

5.3.2 Assess environmental hotspots

Next, LCA is used to calculate the product's life cycle environmental impacts. For technologies with low TRL, other ex-ante LCA methods should be used first to estimate the performance at industrial scale (TRL 9). Then it is possible to identify the environmental hotspots and their link to a foreground process or its inputs. The hotspot analysis determines which environmental impacts have common origins. If the burdens of multiple

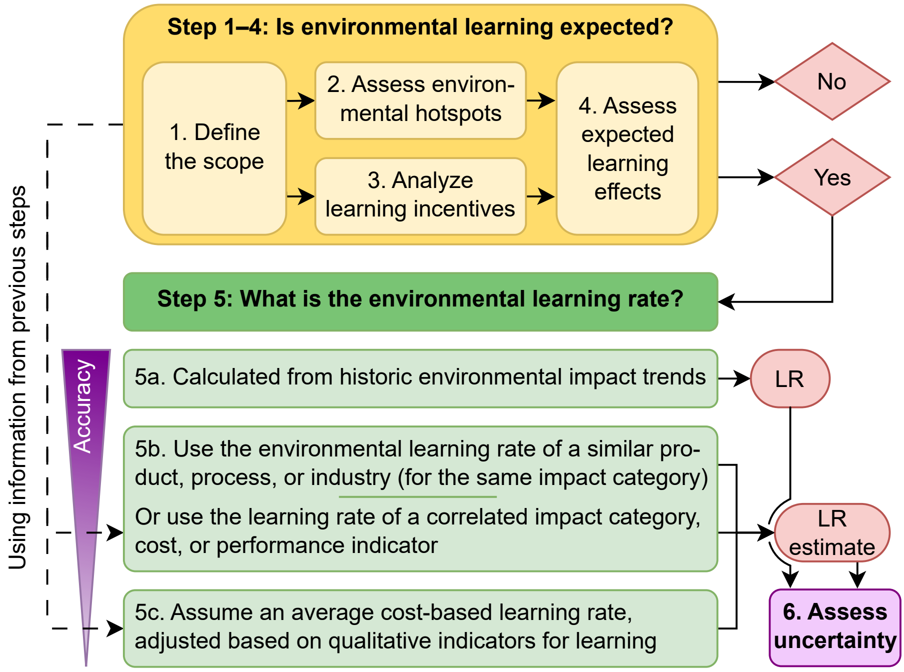

FIGURE 5.4: Overview of the procedure for evaluating environmental learning effects in LCA. The diagram shows connections between the 6 steps and, for step 5, the preference order of alternative approaches are shown. LR: environmental learning rate.

categories originate from a shared process (e.g. fuel combustion) or the same material (e.g. the supply of a metal), their trend will be similar.

5.3.3 Analyze learning incentives

Stakeholders that influence the foreground system can be identified by considering six groups: consumers, regulators, investors, competitors, societal organizations, and value chain partners. Then, the main incentives for the operator of the foreground process are determined, ideally through stakeholder interviews. Relevant drivers were described in §5.2.4.2. If changing motivations (§5.2.4.1) introduce uncertainty, multiple scenarios may be developed.

5.3.4 Assess expected learning effects

This step inspects the link between environmental hotspots (from step 2) and incentives experienced by the responsible actors (step 3). If both are linked to different processes or inputs, no environmental learning is expected. If both are linked to the same processes or inputs, environmental learning is expected to occur. If incentives or KPIs derived from them represent a trade-off with environmental impacts, a negative learning rate is expected (as discussed in §5.2.4.5). If technology diffusion and size scaling introduce economies of scale for hotspot processes, the environmental learning rate is positive. Extensive analyses may repeat this step for additional hotspots or stakeholders.

5.3.5 Estimate learning rate

If step 4 concludes that learning effects are plausible, the learning rate can be estimated. The estimation can use one of the following three approaches, depending on data availability. These are similar to approaches for cost-based learning. The estimates can be adjusted upwards or downwards based on the learning incentives analyzed in step 3.

a. Using historical data

The most straightforward approach uses historical data to determine the environmental learning rate. Records of past impacts are fitted to Eq. 5.3, allowing to extrapolate future trends. Analogous to Eq. 5.1, I_0 is the environmental impact of the initial production volume P_0, and the time-dependent variables are cumulative production P and environmental impact I (e.g. emission per kg product).

$$I = I_0 \cdot \left(\frac{P}{P_0}\right)^a \tag{5.3}$$

Some impacts are bounded by thermodynamic limits, e.g. the minimum land area or energy needed. In those cases, Eq. 5.3 should be adjusted to enforce the lower limit of impacts I_{min} (Ramírez & Worrell, 2006), as in Eq. 5.4.

$$I = I_{min} + I_0 \cdot \left(\frac{P}{P_0}\right)^a \tag{5.4}$$

Novel upstream processes should either be considered as part of the main process or be addressed by a decomposition approach, introduced in §5.2.2. This is because the main process can contribute to significant demand growth of a material, driving upstream learning and development (Bergesen & Suh, 2016). This effect was observed in the PV supply chain (Kavlak et al., 2018).

b. Using a proxy

Secondly, the environmental learning rate of a similar product, process, or industry can be assumed, e.g. based on §5.2.5. Alternatively, the learning rate of an impact category can be estimated by referring to another impact category, cost, or performance indicator (e.g. energy efficiency), if the two are correlated. The correlation is revealed in step 2 and 4. Groups of correlated impact categories may occur (Esnouf et al., 2019), although the distinct learning rates will vary (Stamford & Azapagic, 2018).

As a cost-based proxy, over hundred industry-specific learning rates are available (Balasubramanian & Lieberman, 2010) Appendix D.3.6). Alternatively, aggregated averages are $19 \pm 8\%$ for industrial products, $16 \pm 9\%$ for energy supply technologies, and $18 \pm 9\%$ for use-phase energy consumption (Weiss et al., 2010). However, cost-based learning is unsuitable as proxy if cost reductions can be primarily attributed to purely financial factors (Table 5.1). Besides, such analysis should correct for material price fluctuations (Thomassen et al., 2020).

Extrapolated learning curves address both learning and scaling effects. However, if the scaling behavior differs between the target and proxy technology, scaling effects should be modelled separately using empirical scaling laws or theoretical models (Caduff et al., 2012, 2014; Piccinno et al., 2016), see Appendix D.1.2.

c. Using sectoral averages

The third approach obtains a rough estimate based on sector-specific characteristics that tend to foster a high learning rate: agility (exemplified by the sector's ability to innovate in the past), repetitiveness of manufacturing, potential for automation or economies of scale, and substitutability of the hotspot process or input (see §5.2.4.4). Also, higher learning rates are expected when the incentives are more aligned with (un)intentional impact reduction (step 4). Based on these factors, the learning rate can be adjusted towards the lower or higher end of reported ranges (Weiss et al., 2010). Although this is the least preferred option, it might be the only method for emerging technologies.

5.3.6 Assess uncertainty

Learning curves are a type of extrapolation, associated with uncertainties. Prior to extrapolation, step 1–4 aim to verify if an environmental learning curve is expected. Due to limited availability of empirical data (see §5.2.5), proxies for environmental learning should be used with caution.

To assess uncertainties, environmental learning can be included in scenario-based LCA studies. Scenario narratives becomes more detailed by integrating stakeholder pressures, incentives (described in §5.2.4) and technological changes (§5.2.4.4). Learning rates can be adjusted in line with other scenario assumptions. Further guidance on addressing uncertainty in ex-ante LCA is provided by van der Giesen et al. (2020).

Uncertain technology performance demands for contingency measures. Therefore, ex-ante cost evaluations often introduce contingency factors, that increase the bare cost estimate by a fraction depending on the maturity (AACE, 1991; Rubin, 2019). Contingency costs decrease with experience, and account for e.g. failed batches, unexpected down-time, and additional steps or safety measures. These unplanned deviations are rarely included in (ex-ante) LCA studies, therefore this approach could contribute to more realistic environmental assessments.

5.4 Illustrative examples

5.4.1 Implications for specific product types

Bulk goods and mass products are mostly subject to cost reduction incentives. By definition, this applies to commodities (Merriam-Webster, 2023). If energy is the main expenditure, fuel-related emissions are likely to decline (e.g. Gutowski et al., 2013). If materials are the main cost driver, impacts related to the material production could decrease.

Environmental impacts of waste management may not exhibit learning effects, unless impacts of waste are internalized.

For consumer products, customer preferences influence the learning process. Important aspects are price, product quality and ease of use (Jayasinghe, 2016). If these preferences induce material substitution, burden shifts are expected. If most impacts originate from a material, while consumers prefer more material-intensive products (e.g. larger smartphones or refrigerators) (Dahmus, 2014; Kasulaitis et al., 2015), negative learning may occur. If the main criterion is the product price, the trends of bulk goods apply. However, if the production costs are largely associated with labor, environmental learning may be absent. Since consumer preferences are variable, learning rates in this sector are more uncertain.

Manufactured products consisting of many elements are typically developed to meet consumer needs of functionality, usability, and durability, next to investor incentives for cost minimization. This leads to minimization of manufacturing time. One way is automation, resulting in higher impacts related to metals production for machines and possibly lower losses of feedstock materials (Moreau et al., 2021). Another way is greater labor efficiency, leading to lower impacts from operating manufacturing buildings.

Agricultural products (food and biofuels) are expected to display learning with respect to land use, due to a high cost of labor and land and due to a focus on productivity (Taramuel-Taramuel et al., 2023). Agriculture faces a clear trade-off between efficient land use (high crop yield) and nutrient emissions. In many countries, the costs of fertilizers outweigh the costs of land use, resulting in increasing nutrient emissions (Zhang et al., 2015). In the case of high fertilizer costs or strong policy incentives, emissions of nutrients and pesticides might decrease.

Different trends are expected for small pollutants not related to energy production, such as heavy metals or carcinogens. Price-driven industries will not naturally address these substances. Regulatory, societal or supply-chain pressure can induce material substitution or end-of-pipe solutions.

5.4.2 Procedure applied to copper

Box 5.1 demonstrates the procedure in Figure 5.4 for the case of copper production, to determine the expected environmental impact trends. Copper production is an established process, hence sufficient data is available for the analysis. The main data source is ecoinvent 3.9.1 (cut-off version) (Wernet et al., 2016). This database provided unit process data and prices in 2005-Euros. The functional unit is 1 kg copper from the global "market for copper, cathode". An LCA was conducted using Brightway 2.4.3 (Mutel, 2017). The life cycle impact assessment methods were obtained from Environmental Footprint method 3.0 (Fazio et al., 2018). In step 3, costs are calculated based on the LCA model. Note that ecoinvent assigns an economic value of zero to waste. In reality, waste treatment entails levies or taxes. Still, this analysis is based on ecoinvent data for illustrative purposes.

Box 5.1: Example of environmental learning assessment applied to copper. The LCA approach is described in the main text.

1. **Define the scope:** The global market mix of copper, and its impact in seven impact categories (Climate change, Acidification, Freshwater ecotoxicity, Freshwater eutrophication, Human toxicity: cancer, Human toxicity: non-cancer, and Tropospheric ozone).

2. **Assess environmental hotspots:** The LCA results in Figure 5.5 show several hotspots depending on the environmental impact category. Smelting is a hotspot for human toxicity and acidification, mine tailings are a hotspot for freshwater eutrophication and ecotoxicity, and electricity contributes most to climate change.

3. **Analyze learning incentives:** The main stakeholders are shareholders of mining companies and wholesale purchasers. Copper is a commodity, and the main driver for the supply chain is cost reduction. The calculated costs of €3.74 per kg copper are broken down in Figure 5.5.

4. **Assess expected learning effects:** Cost reduction aligns most with reduced climate change impacts. For both costs and climate change, important drivers are electricity and fuels. Therefore, learning is most likely to reduce climate change impacts.
   Copper smelting is a hotspot for acidification and human toxicity effects (cancer and non-cancer), while the process causes around 5% of the costs. There are no cost incentives for smelting to reduce the emissions to air and water. Moreover, technologies to avoid pollution by tailings and exhausts are more costly (Lèbre et al., 2017). Hence no learning curve is expected for human toxicity impacts. This may change when pressures arise from e.g. local communities or governments to address pollution. Similarly, no learning effects are expected for freshwater ecotoxicity and eutrophication. These impacts originate primarily from mine tailings, which are disposed of at very low costs.
   Finally, waste recycling bears significant costs, while causing minimal environmental burdens. Therefore, it is expected that recycling will increase if the availability of copper scrap allows, leading to greater added value and reduced environmental impacts. The recycled content is not solely determined by learning processes, since it is limited by physical constraints.

5. **Estimate learning rate:** Historical data is available for GHG emissions of primary copper production (Rötzer & Schmidt, 2020). Although ore grades have declined over time, we assume a constant grade of 1.7% to make a functionally equivalent comparison. These data are combined with the cumulative production (Porter et al., 2018). Finally, the parameters of Eq. 5.3, are fitted to the data, yielding a learning rate of 23% for GHG emissions.
   No historical data are available for tropospheric ozone formation, but it may exhibit learning due to some alignment with costs. We use the sector-average learning rate for costs. The relevant sectors are primary and secondary smelting and refining of nonferrous metals, corresponding to SIC 333 and 334,

which have respective learning rates of 24.1 ± 5% and 21.4 ± 5% (Balasubramanian & Lieberman, 2010). Because of the partial overlap with cost reduction, the estimated learning rate is towards the lower bound, at 18%.

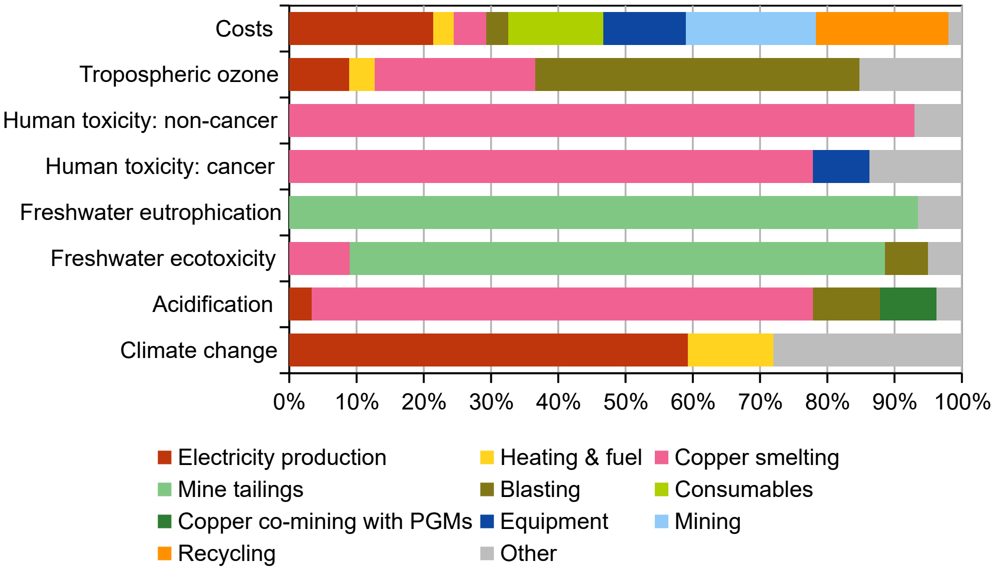

FIGURE 5.5: Contribution analysis of environmental impacts and costs of copper production. PGMs: platinum group metals.

5.5 Discussion

5.5.1 Limitations

Like cost-based learning, environmental learning curves are subject to several limitations, most of which apply to both. Both projections are sensitive to cumulative production estimates (Junginger et al., 2008) and to what is defined as the first industrial plant or unit. These aspects are important for the temporal pinpointing of future environmental impacts. To avoid double counting, studies should specify if economies of scale are included in the scope.

Learning is a multi-scale phenomenon, that takes place at the level of individual workers, teams, organizations, sectors, and society as a whole. This research focused on learning at a sector level, while encompassing learning on lower levels. Our approach does not account for knowledge spillovers that emerge at societal level (Clarke et al., 2008). To study this assimilation of technologies developed in other sectors, an integrated, economy-wide model would be required.

5.5.2 Methodological contributions

Bergesen & Suh (2016) and Thomassen et al. (2020) postulated that technological change is the central driver for both costs and emissions. Our approach, based on reviewed literature, emphasizes the guiding role of incentives for organizational and technological change. This conceptualization enables a structured analysis of environmental learning, decoupled from cost trends.

Our approach takes a neutral perspective on learning that is very suitable for explorative or predictive scenarios. It acknowledges that impacts could either increase or decrease due to learning, and it also covers unintentional effects on impacts. In contrast, eco-innovation literature focusses on drivers and incentives for impact reduction specifically, reflecting a normative perspective. Depending on the research aim, learning curves are one of many scenario generation methods (Arvidsson et al., 2018; Cho & Daim, 2013). If a technology assessment revealed potential environmental impact reductions, the proposed analysis helps to indicate the likelihood of realizing those reductions and the expected rate of change. Therefore, we support the recommendation to combine a bottom-up and top-down approach (Santhakumar et al., 2021; Thomassen et al., 2020) for technology scenarios and roadmaps.

5.5.3 Implications

Understanding the ubiquity of cost-based learning curves can accelerate learning. Cost incentives are predictable, unavoidable, and relatively constant. Moreover, costs can be translated to KPIs that are supported by managers and understood by investors and operators. Environmental learning will be stimulated if more stakeholders would reward environmental performance and would make long-term commitments. Currently, the incentives involved in environmental learning appear less constant. Consequently, both emerging and existing industries can quickly alleviate environmental pressures once appropriate incentives are implemented.

There is a selection bias in empirical emission data for some pollutants. Organizations that report their emissions are more inclined to reduce emissions (Christensen et al., 2017; Downar et al., 2021). Monitoring and disclosure are intermediate steps between awareness and action. Concurrently, this implies that reporting can support environmental learning.

5.5.4 Recommendations for future research

Future research on environmental learning in LCA could benefit from integrating insights from eco-innovation studies, which provide valuable quantitative evidence. However, it is challenging to apply the findings of these studies in LCA, as eco-innovation outcomes are reported in binary terms (Horbach et al., 2012), as absolute numbers (Wang et al., 2018) or per economic output (as in EU assessments (Al-Ajlani et al., 2022; Kemp et al., 2019)). To strengthen the connection, eco-innovation outcomes should be reported as relative impacts (e.g. energy use per functional unit), preferably per (sub)sector. Note that LCAs define the functional unit precisely, whereas eco-innovation studies include more diverse or variable products. Greater synergies are therefore expected for (sub)sectors with a more uniform product, such as paper or asphalt.

Besides the studies cited in §5.2.5, additional evidence for environmental learning may be obtained. Future research may expand the evidence by re-evaluating existing datasets from a learning curve perspective. For example, environmental learning of agricultural products can be studied using FAO statistics on fertilizer use efficiency. Manufacturing companies and industry associations may apply our procedure to their historical data on process efficiencies or emissions. Only if more data becomes available, the uncertainties for extrapolation will be reduced to an acceptable level. Moreover, empirical research can help to refine the approach and possibly identify more variables that influence environmental learning. Future research could explore learning in the context of services. Service-providing activities have greater flexibility in terms of input and functional output (Torugsa et al., 2018), adding complexity to the analysis. This extension is relevant because many products are ultimately used to provide a service.

5.6 Conclusion

Learning curves have been applied extensively as empirical trends to extrapolate costs. This study investigated the fundamental mechanisms of learning, to develop an approach for environmental learning in LCA. Technologies and production processes often evolve in response to incentives created by stakeholder pressures. Therefore, learning affects environmental impacts if these impacts are interlinked with prevailing business performance indicators.

A few studies have examined the influence of learning on resource use and emissions trends. These studies have primarily found reduced energy consumption over time, suggesting that environmental impact reduction is more likely for energy-intensive emerging technologies. Other impact categories remain less explored. Building upon reviewed theories, we proposed a procedure to anticipate environmental learning effects. This procedure complements guidelines for ex-ante LCA in two ways: by addressing the industrial development phase, and by providing a temporal outlook. These insights support decision-making regarding emerging technologies by informing scenarios, roadmaps, and investment strategies.

Data availability

The data that supports the findings of this study are available in the supporting information of this article, which may be found in the online version of the article at the publisher's website. The data that support the findings of § 5.4.2 are available from Ecoinvent. Restrictions apply to the availability of these data, which can be accessed at www.ecoinvent.org.

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6 Discussion & Conclusion

6.1 The need for recycling magnets

The demand for NdFeB magnets is rapidly growing, due to their use in key technologies such as hard disks, electric motors and wind turbines. However, the primary production of these magnets raises concerns due to its environmental impacts and the concentrated supply chain. In response to these concerns, magnet recycling technologies are under development. This thesis aimed to determine the potential and environmental consequences of deploying large-scale recycling of NdFeB magnets. This was achieved by investigating the material flows of recyclable magnets, the environmental impacts of recycling processes, and the prospects for future recycling technology development including learning effects.

The previous four chapters have described the environmental implications of the emerging magnet recycling system. We started with a system-level perspective on recycling in Chapter 2, surveying the factors that determine the recyclability of minor metals. Next, the potential for magnet recycling was assessed in Chapter 3 by quantifying the waste flows of end-of-life (EoL) magnets. Chapter 4 presented an ex-ante LCA study on direct alloy recycling technology for magnets, which determined the environmental performance of current and potential future process configurations. Chapter 5 investigated the effect of environmental learning on technology advancement after commercial deployment.

This general discussion synthesizes the results and insights from previous chapters, and provides a critical reflection on the limitations and implications. §6.2 presents the key insights regarding NdFeB magnet recycling technology and recycling systems, and discusses methodological contributions. The main research question is answered in §6.3. §6.4 summarizes the limitations of this research, together with recommendations for future research. Finally, §6.5 provides an outlook by discussing the implications of the findings.

6.2 Key findings and contributions

6.2.1 Recyclability of EoL magnets

RQ 1. *What factors determine the success of recycling minor metals and their recycling systems?*

Chapter 2 presented a novel framework to assess the recyclability and the potential for successful recycling of minor metals. The framework features a set of indicators that addresses all stages in the recycling chain, allowing to identify bottlenecks for recycling. Recyclability indicators characterize economic factors (such as revenues, economic incentives), technical factors (recovery efficiency, technology maturity), societal factors (supply chain alignment, social benefits), and characteristics of the waste flow (design, collectability, waste volume). The recyclability also reflects the extent of change needed in a recycling system to make its deployment successful. The framework is unique in addressing minor metals specifically. Compared to other materials and metals, minor metals have a high uncertainty of future waste volumes, a high degree of dispersion and a low concentration in products. With these features, the framework has enabled an extensive and coherent study of the NdFeB magnet recycling system.

A circular economy for magnets requires that all parts of the recycling system are established and functioning. Chapter 3 identified a number of underdeveloped aspects in the magnet recycling chain that may be improved, notably product design, alignment between recycling chain partners, and willingness to pay for recycled magnets. Some bottlenecks vary by product type, requiring differentiated solutions. For consumer electronics, it is typically challenging to collect and to liberate magnets. For electric vehicles (EVs) and wind turbines, the waste volumes are currently too small to justify dedicated recycling facilities, although these volumes will increase rapidly in the future.

The recyclability assessment framework helps to interpret recent developments in magnet recycling. During this research, a rapid increase of announcements for commercial activities and pilot plants emerged, see Figure 1.2. While technology continued to develop, there were no technological breakthroughs that could explain this sudden interest. Instead, some key limiting factors, as identified by the recyclability framework, have diminished. Primarily, the available volume of waste increased, enhancing the economies of scale. In addition, the price of rare earth elements (REEs) contained in magnets increased (China Magnets Source, 2023). More EoL products emerged with favorable properties for recycling, such as electric bikes and industrial motors. In the future, the recyclability of NdFeB can improve further with the emergence of larger magnets in EVs and wind turbines. In contrast, recycling magnets from other devices remains challenging due to miniaturization, material reduction, and a higher number of materials per product (see §1.3; Mathieux et al., 2018).

6.2.2 Availability of magnet waste flows

RQ 2. *What is the volume of NdFeB magnet waste in Europe? Which waste flows are most suitable for recycling?*

Chapter 3 identified the NdFeB waste flows that are potentially available for recycling in Europe, using dynamic material flow analysis and recyclability assessment. Magnets are used in a wide range of products and new magnet applications are emerging over time, resulting in a rapid growth of demand and waste streams. In 2019, the total consumption required 7.7 kt neodymium, and EoL magnets contained 2.8 kt neodymium. and this amount is growing each year. At present, most EoL NdFeB magnets in Europe can be found in electronic waste. These small and dispersed magnets pose a challenge for collection, magnet identification and liberation. To recycle these magnets, existing WEEE collection channels can be used as a basis. Alongside, magnet demand has been growing for industrial applications: robots, motors, and pumps. These applications have recycling challenges similar to consumer electronics, although to a lesser extent. In the future, waste flows from EVs and wind turbines are expected to grow, resulting from the currently rising demand. These applications contain much larger magnet assemblies, and their large size and numbers may justify dedicated collection systems. However, current EV motor designs and disassembly techniques are largely incompatible.

Given the described changes in waste flows, the magnet recycling system should be able to adapt. The optimal design of recycling processes—in particular collection and pre-processing—depends on the EoL product characteristics. To anticipate on the emergence of new magnet-containing wastes, product manufacturers can improve design for recycling and recyclers can experiment with disassembly. These actions ensure that dedicated recycling systems are established in time for future waste flows.

The quantity and composition of NdFeB waste varies substantially between European countries. Higher population densities result in more EoL magnets per square kilometer in e.g. the Netherlands and Belgium. Besides, the demand for industrial magnets depends on the industrial structure in each country, causing further divergence. Countries with a large population generate the greatest amount of waste magnets. These insights help recyclers to select favorable recycling markets.

The scale of magnet recycling processes is an important parameter for its environmental sustainability. Large-scale processes have lower environmental impacts (Chapter 4) and enable economies of scale. However, it is challenging to operate at the scale of other metallurgical processes, due to the limited volumes of waste magnets. To still minimize costs, a high tendency emerges to centralize magnet recycling, implying that magnets would be recycled in only a few countries. The same trend has occurred for primary NdFeB magnets: rare earth and magnet production are centralized (Smith et al., 2022), and Chinese magnet supply chains have recently been strongly consolidated (Chang, 2022). Centralized supply chains are vulnerable to disruptions, therefore it is important to find a balance between scale and resilience.

6.2.3 Environmental performance of magnet recycling processes

RQ 3. *What are the environmental impacts of industrial-scale recycling processes for NdFeB magnets?*

Chapter 4 concluded that direct alloy recycling of magnets can have lower environmental impacts compared to producing primary magnets. In most of the impact categories analyzed, magnets from anticipated large-scale recycling have over 80% lower impacts than primary magnets. A comparison of NdFeB powders showed that recycling has lower environmental impacts than primary powder production, already for pilot-scale recovery. The industrial-scale performance can be achieved by upscaling and optimizing the unit processes of recycling.

The systematic assessment of technological changes towards industrial-scale recycling has revealed which changes deliver the largest environmental improvements. The most significant improvements occur in the stages of alloy recovery and magnet manufacturing. Generally, the improved performance stems from lower material loss and higher heating efficiency. All types of technology developments contributed: process changes, size scaling, internal recycling and optimization, with some differences among impact categories and recycling routes. Although it is difficult to achieve complete optimization, i.e. with performance close to thermodynamic optimum, the ex-ante LCA revealed potential improvements for each unit process.

The recycling processes for different types of EoL magnets have comparable impacts. This is because waste pre-processing uses comparatively little energy, and represents only a small portion of the total recycling impacts. Therefore, while the recycling of small magnets may be costly, it would be justified from environmental perspective, even if more intensive preprocessing were required. Meanwhile, the type of EoL magnets determines the design of the magnet recycling system (i.e. collection schemes and disassembly methods). Also, the composition of waste magnets determines the recycled magnet quality and hence the possible applications. For instance, EoL magnets with a high Co or Dy content are ideally recycled into magnets which need a high thermal stability.

This research demonstrated that ex-ante assessments can greatly benefit from inputs provided by technology developers and experts. As Tsoy et al. (2020) pointed out, LCA is more than a computational exercise, and interaction with technology developers is valuable for the goal & scope definition and data collection. For Chapter 4, two participatory methods were used: workshops and meetings with experts. These methods contributed to an accurate system definition and unit process models, and feasible scenarios for technology development. Future ex-ante LCA studies are recommended to select an appropriate participatory method from the available options (Cho & Daim, 2013; Slocum, 2003). The contribution of LCA is an overarching environmental insight that places the impact of different unit processes into perspective.

Another methodological innovation in Chapter 4 is the use of reference values to validate the results of projected technology performance. In our study on magnet recycling, this indicated that the environmental impacts of the final optimized configuration were higher than the theoretical optimum and were close to an industrial reference configuration (based on the performance of existing processes). Thermodynamic modelling helped to validate the results and helped to uncover inefficiencies in pilot-scale processes.

6.2.4 General effects of environmental learning

RQ 4. *How does technological learning affect environmental impacts, and how can LCA studies address this effect?*

Chapter 5 presented an approach for evaluating environmental learning effects in LCA studies on industrial technologies. Given the uncertainties involved in extrapolations to the future, the procedure first verifies if a learning curve is expected and then searches a representative learning rate based on similarity. Chapter 5 further argued that environmental impacts decrease faster if the environmental hotspots coincide with the incentives for learning, e.g. when both costs and emissions mostly arise from the same process. Acknowledging that environmental hotspots vary across impact categories, each impact category follows a different trend, and some may not decline at all. The approach extends existing ex-ante LCA methods, by describing technology development during the industrial deployment phase.

The notion that environmental learning is driven by stakeholder pressures implies that the learning process can be accelerated by these stakeholders. If, for example, customers or regulators articulate and highlight their preference for sustainable production, manufacturers will be more motivated to improve their environmental performance. Consequently, the rate of environmental progress depends on industrial players as well as societal developments. Given the importance of economic incentives, it is promising for short-loop magnet recycling that most technology improvements examined in Chapter 4 reduce both costs and environmental impacts.

The insights from Chapter 4 and 5 could accelerate the development of recycling technology by identifying significant process improvements and learning mechanisms. These chapters respectively decomposed technological progress and technological learning, providing guidance for the otherwise difficult to grasp innovation process. In Box 6.1, the guidelines for environmental learning are applied to magnet recycling, indicating significant reductions in environmental impact. While several insights may apply to other similar technologies, it is important to acknowledge the unique features of each technology.

Environmental learning alone cannot achieve sustainable development. While it is promising that learning mechanisms can reduce the environmental impacts of existing technology, these incremental improvements tend to level off over time. Therefore, profound decoupling of human well-being from environmental impact requires changes in other parts of the *IPAT* equation (see §1.5), such as lifestyle changes.

Box 6.1: Evaluation of learning effects in magnet recycling, based on the environmental assessment in Chapter 4 and the procedure in Chapter 5.

Evaluating learning effects in magnet recycling

We first review the learning incentives and environmental hotspots. Magnet recyclers are incentivized to compete with primary magnets on quality and costs. Therefore, key metrics are energy efficiency and recovery efficiency. The most significant consumable in terms of expenditure at pilot scale is electricity, which also has a high contribution to air pollution and climate change. This underlines the double advantage provided by energy savings. Besides, higher recovery efficiency is rewarded by higher revenues while reducing the environmental impacts per functional output. If material losses require treatment as hazardous waste, the high involved costs further encourage material efficiency.

Other technology changes have a less pronounced environmental effect. Labor is the largest cost driver (van Nielen et al., 2023), therefore a shift to automation is likely. This entails a higher need for machines and associated impacts. Labor costs can also be reduced by size scaling, which improves the energy efficiency and reduces multiple environmental impacts. When equipment costs are reduced, the effect is mostly economic, not environmental. The quality of recycled magnets presents a trade-off with environmental impacts, as virgin NdH_2 can be added to recovered alloys to enhance their magnetic properties. However, Chapter 4 argued that optimization through growing experience can reduce the need for virgin additions. To conclude, most incentives for magnet recycling are aligned with environmental impact reduction.

After verifying that a learning curve is expected for magnet recycling, the next step is to estimate the learning rate. No historical data was found, nor were environmental trends in related processes or sectors. The closest approximation is the cost-based learning rate for SIC sector 334, *Secondary Smelting and Refining of Nonferrous Metals*, which averages 21.4% (Balasubramanian & Lieberman, 2010). Since both potential technology improvements (Chapter 4) and more efficient recovery and electricity savings (described above) have a favorable effect across a wide range of impact categories, we can infer that the estimated learning rate is applicable to all of these categories. The low historic recycling rates imply that the cumulative production may double more than once in the coming years, resulting in large impact reductions. A more detailed calculation would differentiate the learning rate by process, e.g. applying sectoral averages for shaping (19.4%), heat treatment (12.3%), and casting (26.9%) of metal parts.

Finally, competition among recycling technologies may affect the sector's impacts. Under economic pressure, hydrometallurgical magnet recycling may prevail over direct recycling. Hydrometallurgy can also process polluted waste fractions and therefore requires less costly preprocessing. It may salvage magnet scrap that is unsuitable for direct alloy recycling. However, this recycling route has relatively high environmental impacts (Elwert et al., 2017; Wang et al., 2022). Therefore, when hydrometallurgical recycling expands by replacing direct alloy recycling, the environmental efficiency of the magnet industry decreases.

6.3 The environmental consequences of deploying large-scale magnet recycling

The combined findings from Chapter 2–5 enhanced the understanding of the magnet recycling system and recycling technology, as well as their sustainability implications. Drawing from the key findings, this section answers the main research question, *"What is the potential and the environmental consequence of deploying large-scale NdFeB magnet recycling?"*

The development and diffusion of magnet recycling technology can reduce the average environmental footprint of NdFeB magnets significantly. This effect can be quantified for the European NdFeB magnet market, using the calculated NdFeB material flows and LCA results. Generally, the trend of a metal's production impact is determined by changes in the recycled input rate (equivalent to recycled content, p.12, or percentage of recycled magnet feedstock), the environmental impact of recycling, and the energy mix (Harpprecht et al., 2024). These elements are depicted in Figure 6.1 and discussed below. Due to NdFeB demand growth, the maximum share of recycled feedstock is below 50%, see Figure 3.6a. Waste volumes are expected to increase, but at a slower rate than the demand due to a shift to products with longer lifespans. Consequently, the maximum share of recycling will decrease, from 45% in 2019 to lower levels. Still, the *real* share

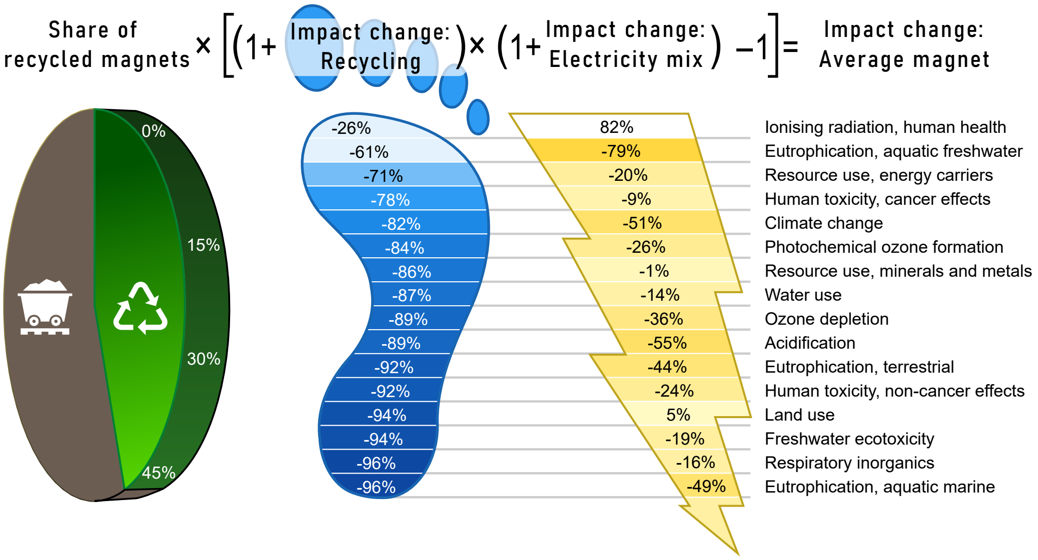

FIGURE 6.1: Calculation approach to determine the environmental impact reduction achieved by recycling NdFeB magnets. The calculation uses the share of recycled magnet feedstock, the change in specific impacts of recycling using industrial reference technology (compared to primary magnets), and the impact change from replacing European electricity by the Swedish electricity mix. Data sources: Figure 3.6a, Appendix C.5 and ESI 6.

has a large growth potential given the currently low recycling rates. Based on Chapter 4, the environmental benefits of recycling will initially be limited, but may increase significantly as the technology advances. For most impact categories, recycled magnets can ultimately reach over 80% lower impacts than their primary counterparts. The most significant change in the wider economy is cleaner electricity production, which further reduces the impact of recycling (Miranda Xicotencatl et al., 2024). Primary magnet production also benefits from cleaner electricity and internal technological progress, but to a lesser extent (Langkau & Erdmann, 2021; Lee & Wen, 2018; Wang et al., 2017).

Figure 6.1 quantifies the potential and environmental consequences of magnet recycling in the EU by accounting for the above effects. For multiple impact categories, it shows the effect of substituting primary magnets with recycled magnets and of replacing the European electricity mix with electricity from Sweden (which represents a potential future electricity mix with a low emission intensity). For example, assuming a conservative 10% recycling share, the climate change impacts can decrease by 9% (as calculated using the equation in Figure 6.1). With a recycling share of 40%, the magnet industry could even achieve 36% lower impacts. This illustrates the multiplier effect of increased waste collection, eco-efficient recycling processes, and clean energy production.

In addition, ex-ante LCA and learning curves (Chapter 4 and 5) together provide a foundation for a roadmap of magnet recycling technology development. The ex-ante LCA outlines the direction and potential endpoint of development. The rate of progress towards the endpoint depends on resources invested in the technology and the environmental learning rate, as derived in Box 6.1. Collectively, these insights indicate when certain environmental performance levels are expected.

6.4 Limitations and Recommendations

The indicators included in the recyclability assessment framework (Chapter 2) were obtained through a literature review. The framework can be strengthened by empirical research on the aspects that correlate most with a high or increasing recycling rate. This approach would determine if some aspects are more important than others. Alternatively, a survey among actors in the recycling value chain may help to prioritize barriers and drivers for recycling. This approach may use comparable participatory techniques as those used for the ex-ante LCA (see §4.2.3).

The accuracy of material flow analysis (MFA) results depends on the quality of input data. Chapter 3 relied on statistical data and applied several data cleaning steps. Besides, a large amount of information was collected on product attributes, such as weight, composition and lifespan. Most of this information is also needed for studying the material flows of other metals and materials. Therefore, a common database for material flow data would benefit the industrial ecology research community, in much the same way as LCA databases have accelerated LCA research.

As discussed in §3.4.3, some studies suggest significant variations in the products sold in different countries. These differences stem from both the composition of products and the market share of magnet types. It is crucial to acknowledge these differences, especially when studying NdFeB flows across the globe. Therefore, it is recommended to model compositions and market shares separately for each region.

The LCA in Chapter 4 had a technology-centric approach, focusing on the impacts and possible improvements of recycling technology. Some studies suggest to consider societal and market mechanisms for additional insights (Bergerson et al., 2020; Moni et al., 2020). Table 6.1 discusses the effect of the most influential factors. Although these factors may influence the LCA outcomes, the main conclusions remain unchanged.

TABLE 6.1: The anticipated effects of factors that were not (fully) addressed by the comparative ex-ante LCA of recycled magnets in the EU (see Chapter 4).

Influencing factor	Explanation	Effect
Rebound effects	While recycling reduces the environmental impact of magnets, it may stimulate an increased consumption.	Rebound effects are probably small, because magnets have minor impacts on the price and weight of products. However, lower supply constraints may accelerate the demand for electrified versions of bikes, skateboards, boats etc.

Impact allocation	Due to price fluctuations of individual REEs, economic allocation may assign different impacts to primary NdFeB magnets in the future.	Allocation decisions mostly affect the impacts of primary magnets and slightly affect recycled magnets. For recycling, allocation applies to disassembly processes that produce several scrap fractions. However, the contribution of these processes is limited.
Changes in reference technology	REE mining and primary magnet production may improve (see p. 125).	The benefits of recycling would decrease compared to the baseline, but still persist.
Background systems	The recycling location determines the energy mix and the origin of chemicals used.[1]	Appendix ap:rel-impacts highlights the importance of the electricity mix for the results. This research also accounted for diversity in EoL magnets and the type of produced magnets.
Product lifespans	The lifespan of magnets should be considered to compare them on a functional basis.	To date, no studies have measured the longevity of recycled magnets. However, magnets rarely limit the lifespan of current applications.

While Chapter 5 outlined the fundamental driving mechanisms of environmental learning, further empirical research is needed to make environmental learning curves more robust. A promising avenue is the study of environmental impact trajectories of mature industries. As discussed in §5.5.4, existing datasets may be re-evaluated to determine historical rates of environmental learning at product group level. This evidence is essential for applying a learning curve approach in environmental assessments, and for general insight into environmental progress trends.

This thesis addressed environmental sustainability, while there are also concerns about social sustainability in magnet supply chains. This calls for methods to evaluate social impacts, and ultimately integrated life cycle sustainability analysis. This is a major challenge, especially due to data limitations (Costa et al., 2019; Guinée, 2015). At the same time, policy makers are already mandating supply chain transparency and due diligence reports (Bond et al., 2023). These reports may provide detailed data for future research on social sustainability.

6.5 Outlook

6.5.1 Solutions for more sustainable magnet recycling

Chapter 2 and 3 investigated issues for the recycling system of magnets. Solutions to these issues can be supported by several societal trends. Below, we highlight the role of digital tools, social innovation, and supply chain collaboration.

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[1] (Langkau & Erdmann, 2021; Miranda Xicotencatl et al., 2024)

Digital tools can support recycling and reuse at various stages of the chain, as described by the concept ‘Recycling 4.0’ (Blömeke et al., 2020). These tools help to optimize logistics, sort waste accurately, and characterize recovered materials. Two particularly promising tools are digital product passports (DPPs) and artificial intelligence. DPPs contain detailed data about a product’s composition, supply chain, and sustainability aspects. These passports are increasingly explored by companies and EU policy-makers (as part of circular economy regulations, specifically the *Ecodesign for Sustainable Products Regulation* (European Commission, 2023b)) as a means to inform consumers and provide supply chain transparency. However, the trade-off between data disclosure and protection may result in a decentralized system (Ducuing & Reich, 2023), prioritizing transparency within supply chains over public transparency. In addition, DPP databases are prone to the same problems as trade statistics, including inconsistent classification and reporting errors. In conclusion, the usefulness of DPPs for research on the industrial metabolism of magnets depends on their implementation.

To advance magnet waste collection, social innovation should complement technical innovations. Recycling rates depend strongly on the recycling behavior of individuals and organizations, which, in turn, is influenced by public attitudes, awareness, and motivation (Hagelüken & Goldmann, 2022). Therefore, it is recommended to engage end-users and to make waste sorting behavior easy (Hagelüken & Goldmann, 2022). To achieve a broader supply chain alignment (see §3.3.5), it is important that stakeholders from across the recycling value chain collaborate. This implies that the Chinese magnet sector has a competitive advantage in recycling, due to the high level of integration and cooperation among magnet manufacturers, equipment manufacturers and rare earth producers (Miranda Xicotencatl et al., 2023). The SUSMAGPRO project also demonstrated how collaboration accelerates the identification of mismatches between recycling steps as well as solutions.

The digital tools discussed above can potentially revolutionize NdFeB magnet recycling by facilitating customized treatment approaches. This would result in specialized niches for distinct recycling techniques. Each niche could focus on processing specific types of waste magnets and develop dedicated collection channels. The implications in terms of efficiency and profitability remain to be studied. On one hand, the industry may gravitate toward multiple smaller recycling paths, pursuing efficiency through specialization. On the other hand, recyclers may seek economies of scale through consolidation (see §6.2.2). Simultaneously, specialized recycling facilities may be threatened by obsolescence due to superior alternatives or the evolving composition of waste flows.

6.5.2 Policy recommendations

Policy makers have played an important role in the development of magnet recycling by supporting R&D (Koese et al., 2024). Recycling plays an essential role in strategic raw materials policies, as it secures materials from local secondary sources, and allows to maintain renewable energy systems that rely on critical raw materials. The applied research methods provided insights into the challenges of the magnet recycling system, leading to the following policy recommendations.

In general, policy should find a balance between regulatory burdens and effect. Given the increased maturity of recycling technologies, policy should shift emphasis from supporting research to fostering market creation. Support should primarily focus on lesser developed parts of the recycling system, identified by recyclability assessment. General bottlenecks for magnets include design for recycling, demand for recycled materials, and alignment of supply chain actors. Recyclability indicators allow to monitor the developments over time. Other indicators may be used as well, as simple indicators (such as EoL recycling rate) provide a complementary, overarching insight. The MFA showed that magnets are dispersed over a wide range of applications. Therefore, waste collection requires coordination and enforcement, including take-back and collective collection schemes. Export of magnet-containing waste should be disincentivized, to avoid environmentally harmful practices abroad and to stimulate domestic circular supply chains. DPPs can play a supporting role, therefore policy makers should explore the opportunities of digitalization.

Ensuring the success of NdFeB magnet recycling, regulation should acknowledge the diversity of recycling technologies. As discussed in §6.5.1, direct alloy recycling and other recycling methods may fulfill complementary functions. Regulations should avoid prescribing a single recycling method, recognizing that the effectiveness of technologies depends on the local availability of magnet scrap, the types of EoL products, and the desired characteristics of newly produced magnets. A successful magnet recycling industry is capable of processing feedstocks from diverse sources and EoL products, which can be achieved by combining complementary recycling technologies. In summary, technology-agnostic regulation fosters a robust, competitive and material-efficient recycling system.

Some regulations, such as the EU Critical Raw Materials Act (European Commission, 2023a), focus on single-material recovery, which may lead to suboptimal recycling solutions. Instead, we advocate for a product-centric recycling approach, as proposed by Reuter et al. (2013, 2019). This approach adopts a holistic view on resource conservation, emphasizing the maximum recovery of all materials. It encourages researchers and recyclers to explore opportunities for co-recovery of multiple materials. As discussed in §2.4.2, this goal can be achieved through design for recycling and by extended producer responsibility. Intelligent, interoperable design of products and recycling systems is needed to maximize the circularity of material cycles.

6.5.3 Advancing recycling technologies further

Our findings inform and support industrial magnet recycling efforts. Chapter 4 highlighted the substantial improvement potential for magnet recycling technology. To realize this potential, both R&D investments and practical production experience are needed (Chapter 5). The key to further advancement is learning-by-doing in an industrial setting; these activities can be informed by the promising waste flows identified in Chapter 3. Even if these waste flows require different pre-processing than future waste flows, learning on shared recycling steps already addresses the largest environmental hotspots.

To achieve large-scale magnet recycling, new business models are key. For companies wishing to implement the novel recycling processes, the challenge is to integrate them into a circular business model. Circular business opportunities can be identified by creative thinking and by following good practices applied to other critical metals.

For instance, recycling of rhenium and platinum are successful due to long-term relationships with end users and manufacturers, and the high metal price (Hool et al., 2022). These business models offer protection against volatile prices and unstable metal supply.

Extending the perspective beyond magnet recycling, this thesis informs an outlook on a more sustainable rare earth magnet industry by identifying four ways to reduce its environmental impacts. First, by operating production processes using renewable energy (Miranda Xicotencatl et al., 2024), preferably generated locally. Second, by investing in process innovation and stimulating learning for cleaner operations. Third, by applying and upscaling more resource-efficient manufacturing routes like MIM and extrusion when suitable. Fourth, by incorporating more recycled materials in magnets. While a truly circular system requires stabilization of magnet demand, the growing amount of waste magnets offers ample opportunities for recycling.

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Acknowledgements

I have enjoyed working on my PhD research for four years. I would like to thank everyone who contributed to this enjoyable time. Starting with Benjamin Sprecher and René Kleijn, who designed a sound research agenda as part of the SUSMAGPRO project plan. R, thank you for giving me plenty of room for my own input and responsibility as a day-to-day supervisor. The various groups you recommended and set up provided a useful context and support, especially during the COVID pandemic. I felt at home in these groups, from the *magnets update* to the ex-ante LCA meetings and the Future Resources Hub. I also enjoyed playing an organisational role in the latter. Of course, these groups would have been nothing without the input of many colleagues, to whom I am very grateful for their feedback and inspiration. In short, the CML has provided an environment in which research can flourish.

Furthermore, I would like to thank the partners of the SUSMAGPRO project for their fruitful collaboration. An industrial ecologist cannot succeed without data, so it was great that the SUSMAGPRO partners were able to provide it – as far as the information was available. Visiting several partners greatly enhanced my understanding of magnet recycling, and provided memorable experiences abroad. Special thanks to Brenda. We complemented each other well during the work for SUSMAGPRO and we succeeded in producing six informative deliverables. Standing in front of the whiteboard together, we managed to find our way in the world of magnets and LCAs.

The fifth and final year of the research was a tough time for me. It is thanks to the moral and practical support of several people that this thesis was finally completed. For that support, I am grateful to several colleagues, especially Brenda, Chen, Janneke, Kai, Maarten, Tales, Teun, and Yanan. As fellow PhDs, we could share frustrations about publishing and also celebrate successes. As a change from my 'own' research, it was refreshing to work on collaborative papers with Brenda, Carina, Chen, Gloria, Marc, Maarten and even Jessie from TU Delft. Family and friends provided the necessary distraction and perspective, occasionally pulling me out of my proverbial cave. Training with triathletes from ZVL-1886 was also a good way to clear my head and make room for new ideas.

I would especially like to thank my supervisors. René, thank you for your optimism and your unwavering belief in a good outcome. I appreciate the many suggestions and tips you gave me, without them being mandatory requirements. You facilitated exchanges, solved practical problems and answered many of my questions, which made this doctoral journey inspiring and successful. Arnold, without your sharp eye and attention to detail, this dissertation might never have been completed. Sometimes you were better able to grasp the core of my research than me, even *on top of* all your other ongoing research matters.

Dankwoord

Vier jaar lang heb ik met plezier aan mijn promotieonderzoek gewerkt. Ik wil iedereen bedanken die heeft bijgedragen aan dat werkgenot. Te beginnen met Benjamin Sprecher en René Kleijn, die een goede opzet van de onderzoeksagenda hadden gemaakt als onderdeel van het SUSMAGPRO projectplan. René, bedankt dat je mij als dagelijks begeleider veel ruimte gaf voor eigen inbreng en eigen verantwoordelijkheid. De verschillende groepen die je aanraadde en opzette vormden een nuttige context en houvast, vooral tijdens de COVID-pandemie. Ik voelde me thuis binnen die groepen, van de *magnets update* tot de ex-ante LCA meetings en de Future Resources Hub. In die laatste heb ik dan ook met veel plezier een organisatorische rol vervuld. Deze groepen waren natuurlijk niks zonder de inbreng van vele collega's, wie ik zeer dankbaar ben voor hun feedback en inspiratie. Kortom, het CML bood een stimulerende en inspirerende omgeving.

Verder wil ik de partners van het SUSMAGPRO project bedanken voor de goede samenwerking. Een Industrieel Ecoloog komt niet ver zonder data, dus was het geweldig dat de SUSMAGPRO-partners die konden aanleveren – voor zover de informatie beschikbaar was. Het bezoek aan verschillende partners heeft bijgedragen aan een beter begrip van magneetrecycling én memorabele ervaringen in het buitenland. In het bijzonder bedank ik Brenda. Wij konden elkaar goed aanvullen tijdens het werk voor SUSMAGPRO, en konden zo zes informatieve rapporten opleveren. Staand voor het whiteboard lukte het samen om de weg te vinden in de wereld van magneten en LCA's.

Het vijfde en laatste jaar van het onderzoek was een zware tijd voor mij. Het is dan ook te danken aan de mentale en inhoudelijke steun van diverse mensen dat dit proefschrift er toch gekomen is. Dat waren diverse collega's, in het bijzonder Brenda, Chen, Janneke, Kai, Maarten, Tales, Teun, and Yanan. Als PhD's konden we frustraties over publiceren delen, en ook successen vieren. Ter afwisseling van het 'eigen' onderzoek was het verfrissend om aan gezamenlijke papers te werken, met Brenda, Carina, Chen, Gloria, Marc, Maarten, en zelfs Jessie van de TU Delft. Familie en vrienden zorgend voor de nodige afleiding en relativering, door mij af en toe uit mijn spreekwoordelijke schrijfhol te trekken. Ook de trainingen met triatleten van ZVL-1886 waren een goede manier om mijn hoofd leeg te maken, zodat er weer ruimte kwam voor nieuwe ideeën.

In het bijzonder dank ik mijn promotoren. René, bedankt voor je optimisme en het rotsvaste vertrouwen in een goede afloop. Ik waardeer de vele suggesties en tips die je hebt gegeven, zonder dat het dwingende eisen waren. Je hebt uitwisselingen gefaciliteerd, praktische problemen opgelost en vele vragen van mijn kant beantwoord, wat dit promotietraject inspirend en succesvol maakte. Arnold, zonder jouw scherpe blik en oog voor details was dit proefschrift misschien wel nooit afgerond. Soms wist jij beter de kern te vatten van mijn onderzoek dan ik, en dat *naast* alle andere lopende onderzoekszaken.

Curriculum Vitae

Experience

Sander van Nielen was born on February 11th, 1993, in Gouda, the Netherlands. After attending Waldorf school, he pursued his interest in quantitative sciences and sustainability, and chose to study in Leiden. Here, he studied *Molecular Science & Technology*, followed by a master's in *Industrial Ecology*. He graduated in 2016, with a thesis on the role of solid oxide fuel cells in future energy systems. Both his bachelor's and master's degrees were joint programs offered by Leiden University and TU Delft.

From 2017 to 2019, Sander worked as a sustainability advisor at the Engineering department of *Gemeente Rotterdam*, a job focused on Green Public Procurement and renewable energy systems. At the same time, he participated in a traineeship programme to develop essential professional skills.

In 2019, he was accepted as a Ph.D. candidate at the *Institute of Environmental Science* (CML) at Leiden University. He subsequently returned to Leiden, where he applied analytical Industrial Ecology methods to investigate the recycling of magnets. The Ph.D. research was supervised by Prof. Dr. Arnold Tukker and Prof. Dr. René Kleijn. His research contributed to the SUSMAGPRO project in several ways, including presentations at project meetings, workshops and reports. In addition, in 2024 he worked on the *Getting the Data Right* project.

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Appendix to Chapter 2

A.1 Literature analysis

Table A.1: Overview of aspects of recyclability found in the literature, grouped by value chain stage.

Value chain stage	Group	Aspect
Overarching	Economies of scale	Quantity
		Economies of scale
		Investment inertia
	Environmental benefits	Environmental effects
		Human health impact
		Resource depletion
	Social benefits	Social externalities
		Labor conditions; Labor rights conditions
		Societal stability
		Job creation
	Supply chain alignment	Supply chain organization
		Supply chain cooperation
		Information exchange
	Uncertainty	Confidence level of future flows
		Future waste composition
		Uncertain waste quality
		Uncertain waste quantity
		Product information available
	Profitability	Net present value
	Costs	Awareness campaign costs
		Collection costs
		Transport costs
		Dismantling costs
		Investment costs
		Operating costs
		Processing costs
		Recycling costs
		Energy costs
		Labor costs

		Landfill costs
Manufacturing	Design approach	Eco-design
		Manufacturers vision
	Product design	Product complexity
		Product design variation
		Variety of components
		Compact design
		Dispersion
		Dismantability
		Joints; Fixation
Use & Collection	Collectability	Size
		Stock
		Property rights; Eligibility
		Ownership
		Ownership shifts
		Product lifespan
		Dissipation
	Collection participation	Collection rate
		Collection rate trends
		Consumer awareness
		Consumer attitude
	Infrastructure	Collection infrastructure
		Consumer facilitation
		Distance between collection points
		Geographic dispersion
	Policy	Extended producer responsibility (EPR) legislation
		Collection/recycling rules
		Collection enforcement
		Waste export
		Export regulation
		Informal/illegal recycling
Preprocessing	Preprocessing performance	Component identification; Material identification
		Contaminant detection
		Shredding time; Size reduction
		Dismantling time
		Liberation efficiency
		Separability after liberation
Recovery	Compatibility	Compatibility
		Co-recovery
		Contamination
		Coating
	Recovery performance	Recovery efficiency

		Concentration
		Variety of materials
		Metallurgical complexity
		Expertise required
		Technology availability
Preprocessing / Recovery	Environmental effects	Emissions to environment
		Toxic materials
		Toxic/hazardous process chemicals
		Energy consumption; Energy savings
		Process safety
	Policy	Economic incentives
	Recycling performance	Statistical entropy
		Material grade
	Technology availability	Technology maturity
		Current recycling rate
		Processing capacity
Secondary market	Criticality	Supply risk of metal
		Price volatility
		Depletion time
		Economic importance of metal; Importance of metal for emerging industries
		Resource independence
	Demand	Demand growth
		Market stability
		Confidence in recycled product quality
		Degradation; Purity of recycled materials
	Revenues	Recovered metal value
	Competition	Competing recycling processes

A.2 Proposed framework

A.2.1 Novel factors and indicators

The following factors and indicators have not been integrated in recyclability assessment frameworks before. The factors are mentioned in the literature, but not included in any framework. The indicators are novel ways to quantify an aspect.

- Factors
    - Design for recycling
    - Component design variation
    - EPR legislation
    - Export restriction
    - Safety risk
    - Expertise required
    - Confidence in quality
- Indicators
    - Fraction of actors involved in exchange
    - Number of product designs
    - Number of component designs
    - Identification accuracy
    - Technology readiness level
    - Price premium or discount

A.2.2 Score calculations

TABLE A.2: Formulas to calculate the score belonging to the value of quantitative indicators. Outcomes > 5 or < 0 are replaced by the maximum or minimum score.The formulas have been designed to return a number within the 5-point scale, by adjusting to the value ranges observed for minor metals. If an indicator and recyclability are inversely related, a minus sign was added. In the case that a range spanned several orders of magnitude, a log function was used.

Indicator	Unit	Formula
Annual waste flow	t/a	$\frac{x}{200} \cdot 5$
Recycling subsidy	\$/t	$x/20$
Disposal and raw material tax	\$/t	$x/20$
Standard deviation / future waste flow	%	$5 \cdot (1 - 2\|x\|)$
Labor rights indicator	–	$\frac{x}{2}$
Number of product designs	№	$5 - \frac{x}{4}$
Number of component designs	№	$5 - \frac{x}{4}$
Metal content per component	g/component	$\log(10 \cdot x)$
Product weight	kg/unit	${}^{5}\log(5 \cdot x)$
Annual waste generation	units/capita/a	${}^{4}\log(800 \cdot x)$
Collected fraction of EoL products	%	$5 \cdot x$
Fraction of aware consumers	%	$5 \cdot x$
Distance between collection points	km	$5 \cdot (1 - \frac{x}{100})$
Identification accuracy	%	$5 \cdot x$
Liberation efficiency	%	$5 \cdot x$
Dismantling time	h/t	$5 - \frac{x}{10}$
Recovery efficiency	%	$5 \cdot x$
Concentration after preprocessing	wt%	${}^{5}\log(20,000 \cdot x)$
Value fraction of recoverable metals	%	$5 \cdot x$
Technology readiness level	TRL	$5 \cdot (x - 1)/8$
GHG emissions	% of virgin	$5 \cdot (1 - x)$
Target metal value	\$/t	$x/20$
Co-recovered metal value	\$/t	$x/400$
Price volatility	%	$5 \cdot (1 - \dfrac{x - 5\%}{25\%})$
Demand growth rate	%	$15 \cdot x + 1.75$

Quantities in tonnes (t) refer to waste input.

B Appendix to Chapter 3

B.1 Permanent magnet market

The market share of NdFeB magnets in the permanent magnet market was estimated on the basis of time series from three sources [1–3]. For ranges where these time series overlapped, the data were averaged.

TABLE B.1: Global permanent magnet production. Totals are calculated average of three market reports (1 in 4; 2; 3). NdFeB magnet amounts are based on Constantinides [5].

Year	Total (t)	NdFeB (t)
1990	306,865	1,254
1995	476,954	4,515
2000	552,104	17,560
2005	663,327	39,384
2010	627,255	80,525
2013	812,487	83,535
2014	877,587	90,057
2015	913,387	109,868
2016	981,423	117,437
2017	1,052,726	125,006
2018	1,125,414	132,576
2019	1,207,195	140,145
2020	1,295,094	147,714

B.2 Market share data

TABLE B.2: Market share of Nd-containing components. BEV: battery electric vehicle; FCC: fluid catalytic cracking; HEV: hybrid electric vehicle; PHEV: plug-in HEV.

UNU-Key	Product group	Assumption or sources
0001	Central heating (household)	Proxy: 1204
0102	Dishwashers	Proxy: 0104
0104	Washing machines	[6; 7]

UNU-Key	Product group	Assumption or sources
0105	Dryers	Proxy: 0104
0106	Heating and ventilation (household)	Proxy: 0201b
0108	Fridges	[6; 8; 9]
0109	Freezers	Proxy: 0108
0111	Air conditioners	[10; 11] EU market share is half that of Japan
0112	Other cooling	25% of 0111
0113	Cooling (professional)	Proxy: 0108
0114	Microwaves	50% of permanent magnet market share
0201b	Fans	Permanent magnet trend with 5% max.
0204	Vacuum cleaners	[11]
0205	Personal care	[11]
0205b	Shavers	Proxy: 0205
0301b	HDDs	See §B.3
0302	Desktop PCs	See Eq. 3 in main text
0303a	Laptops	idem
0303b	Tablets	100%
0304	Printers	Permanent magnet trend with 1% max.
0305	Telecom	Half of the market share of 0306a
0306a	Mobile phones	[8]
0306b	Smartphones	[8]
0307	Professional IT	Permanent magnet trend with 1% max.
0309	Flat screen monitors	Proxy: 0408
0401	Small consumer electronics	Half of the market share of 0401b
0401b	Headphones, earphones	[12; 13]
0403	Music instruments, radio, HiFi	Permanent magnet trend with 80% max.
0404b	Video players	[10]
0405	Speakers	[12; 14], [1]
0406	Cameras	[12]
0408	Flat screen TVs	[13; 15]
0702	Game consoles	Proxy: 0301b
0802b	MRIs	[16]
1002	Cooled vending machines	Proxy: 0108
1101	Cars	[11; 17]
1102a	BEVs	[6; 18]
1102b	PHEVs	100%
1103	HEVs	100%
1104	Snowmobiles, golf cars, etc.	Half of the market share of 1101

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[1] Personal communication, B&C speakers

UNU-Key	Product group	Assumption or sources
1105	Trucks	75% of the market share of 1101
1106	Buses	75% of the market share of 1101
1107	Motorhomes	Half of the market share of 1101
1108	Electric bikes	[19]
1201	Industrial machines & motors	Permanent magnet trend with 2% max.
1202	Industrial pumps	Permanent magnet trend with 10% max.
1203	Lifting and conveying machines	Permanent magnet trend with 5% max.
1204	Shaping machines	0% [a]
1205a	Wind turbines, onshore	[20]
1205b	Wind turbines, offshore	[20]
1206	Industrial robots	Permanent magnet trend with 90% max. [a]
	Catalysts	
1101	Car catalytic converter	Nd phase-out between 1990–2000. [a]
1301	FCC catalyst	[21], Nd phase-out between 1995–2010. [a]
	NiMH batteries	
0204	Vacuum cleaners	Linear growth of products with a battery to 15.9% in 2012 [22].
0205	Personal care	Half of the products contain a battery. NiMH share as in Fig. 2.
0305	Telecom	Half of the products contain a battery. NiMH share as in Fig. 2.
0306a	Mobile phones	[23]
0406	Cameras	Twice the average market share of NiMH batteries.
0601	Power tools	Linear growth of products with a battery to 2% in 2009 [24].
0602	Tools (professional)	Half the value of 0601.
0701	Toys	Follows the average market share of NiMH batteries.
1103	HEVs	[23; 25; 26]

[a] Personal communication, Gareth Hatch (2020).

B.3 Data storage market

The market shares of HDDs and SSDs were based on market reports [27–30] reporting global unit shipments. Logistic curve fitting yielded an inflection point at 2018 and an ultimate market share of 85% for SSDs.

TABLE B.3: Global market data and forecasts for HDD and SSD sales, in million units per year.

Year	HDD sales [27]	SSD sales	SSD sales source	HDD market share	Fitted HDD market share
2003	251				
2004	295				
2005	368				
2006	424				99%
2007	486				99%
2008	521	2.0	[28]	100%	98%
2009	548	11.0	[28]	98%	98%
2010	632	14.0	[28]	98%	97%
2011	600	17.3	[28]	97%	95%
2012	548	39.0	[28]	93%	93%
2013	510				90%
2014	520				86%
2015	425	105	[29]	80%	80%
2016	377	140	[29]	73%	74%
2017	355	190	[29]	65%	66%
2018	310	232.2	[30]	57%	58%
2019	279	317.8	[30]	47%	49%
2020	249	354.3	[30]	41%	41%
2021	223	408.2	[30]	35%	35%
2022	182	451.8	[30]	29%	29%
2023	168	488.2	[30]	26%	25%
2024	158	508.9	[30]	24%	22%

B.4 Product lifespan

TABLE B.4: Weibull parameters describing the lifespan distribution of applications.

UNU-Key	Product group	scale (β)	shape (α)	Source
0001	Central heating (household)	14.21	2	[31]
0102	Dishwashers	12.12	1.64	[31; 32]
0104	Washing machines	13.6	2.2	[31; 32]
0105	Dryers	14.6	2.58	[11; 31]
0106	Heating and ventilation (househ.)	13.47	2	[31]
0108	Fridges	16.71	2.2	[31]
0109	Freezers	18.55	1.28	[31; 33]
0111	Air conditioners	14.52	2.69	[31]
0112	Other cooling	13.36	2.36	[31]
0113	Cooling (professional)	15.36	1.6	[31]
0114	Microwaves	17.99	2.07	[31]
0201b	Fans	7.97	1.22	[31]

UNU-Key	Product group	scale (β)	shape (α)	Source
0204	Vacuum cleaners	8.7	1.45	[31; 32]
0205	Personal care	8.09	1.2	[31]
0205b	Shavers	9.5	1.5	[31]
0301b	HDDs	5.91	1.25	[31]
0302	Desktop PCs	8.95	1.58	[31]
0303a	Laptops	6.57	1.6	[31]
0303b	Tablets	6.8	1.6	
0304	Printers	9.31	1.88	[31]
0305	Telecom	7.22	1.24	[31]
0306a	Mobile phones	6.26	1.56	[31]
0306b	Smartphones	6.26	1.56	[31]
0307	Professional IT	7.78	1.46	[31]
0309	Flat screen monitors	7.39	2.33	[31]
0401	Small consumer electronics	9.87	1.3	[31]
0401b	Headphones, earphones	6.15	1.3	[31]
0403	Music instruments, radio, HiFi	15.54	2.09	[31]
0404b	Video players	8.33	1.14	[31]
0405	Speakers	11.5	1.49	[31; 34]
0406	Cameras	6.75	1.19	[31]
0408	Flat screen TVs	11.75	2.01	[31]
0702	Game consoles	4.78	1.14	[31]
0802b	MRIs	14	2.5	[31; 35]
1002	Cooled vending machines	15	2	[31]
1101	Cars	15.5	3.6	[10; 36]
1102a	BEVs	15.5	3.6	[10; 36]
1102b	PHEVs	15.5	3.6	[10; 36]
1103	HEVs	15.5	3.6	[10; 36]
1104	Snowmobiles, golf cars, etc.	15.5	3.6	[10; 36]
1105	Trucks	23	3.3	
1106	Buses	23	3.3	
1107	Motorhomes	17	1.8	[32]
1108	Electric bikes	10	1.8	[11; 32]
1201	Industrial machines & motors	16.7	1.5	[37]
1202	Industrial pumps	27.1	2.2	[38]
1203	Lifting and conveying machines	16.7	1.5	[32]
1204	Shaping machines	16.7	1.5	[32]
1205a	Wind turbines, onshore	24.7	3	[11; 12; 38]
1205b	Wind turbines, offshore	24.7	3	[11; 12; 38]
1206	Industrial robots	13.6	2.7	[10; 38]
1301	FCC catalyst	2.3	1.5	[39]

B.5 Recyclability assessment

B.5.1 Data sources

The following data sources were used to quantify each of the quantitative indicators of the recyclability framework.

- Labor rights indicators from Kucera & Sari [40] were weighted using the shares in global production of rare earth elements.
- Price volatility was obtained from DERA [41].
- Target metal value, Co-recovered metal value: based on metal prices [41] and product composition data (§B.5.2).
- Value fraction of recoverable metals: §B.5.2.
- Annual waste flow, Annual waste generation: from MFA.
- Metal content per component, Product weight: Table 1 in main text.

B.5.2 Value of recoverable metals

The recovered metal value is calculated as the product of the recovery efficiency, the target metal content after preprocessing and the target metal price. The target metal, neodymium, has an average metal price of \$61.53 (€50.45) [41].

For a view on the recoverable metal value fraction, a similar calculation was used. Price data were again obtained from Bastian [41], with data for iron from Focus-Economics [42]. For printed circuit board contained in HDDs, pumps and wind turbine nacelles, it was assumed that 25% of the value is recoverable.

Table B.5: Value of metals contained in EoL products.

EoL product	Composition source [b]	Total metals (\$/t)	Neodymium (\$/t)	Co-recovered metals (\$/t)
EV motors	[43]	\$ 3,296	\$ 387	\$ 2,883
HDDs	[44]	\$ 7,159	\$ 464	\$ 6,308
Industrial pumps	[45]	\$ 2,812	\$ 315	\$ 1,560
Speakers	[46]	\$ 1,273	\$ 276	\$ 997
Wind turbines [a]	[47]	\$ 3,301	\$ 969	\$ 1,341

[a] Nacelle is taken as reference. [b] Metal content of printed circuit board was taken from Holgersson et al. [48] and Wang et al. [49].

B.6 Nd content of waste flows

Table B.6: Nd content of waste flows in 2019, per country. This data is plotted in Fig. 5 in the main text.

Country code	Nd in waste (t)	Nd in waste (g/km^2)	Land area (km^2)
AUT	55.69	664	83,879
BEL	86.28	2,826	30,530
BGR	17.55	158	111,000
CYP	3.01	326	9,250
CZE	74.66	947	78,870
DEU	541.86	1,515	357,580
DNK	60.63	1,413	42,920
ESP	211.00	417	505,953
EST	4.01	88	45,340
FIN	28.43	84	338,460
FRA	378.26	689	549,087
GBR	482.37	1,980	243,610
GRC	49.97	379	131,960
HRV	10.91	124	88,073
HUN	31.54	339	93,030
IRL	40.69	579	70,280
ITA	260.58	863	302,068
LTU	7.09	109	65,290
LUX	4.54	1,752	2,590
LVA	4.81	75	64,594
MLT	2.08	6,490	320
NLD	142.20	3,423	41,540
POL	156.44	500	312,690
PRT	39.39	427	92,230
ROU	43.55	183	238,400
SVK	31.33	639	49,030
SVN	8.92	435	20,480
SWE	63.14	119	528,861

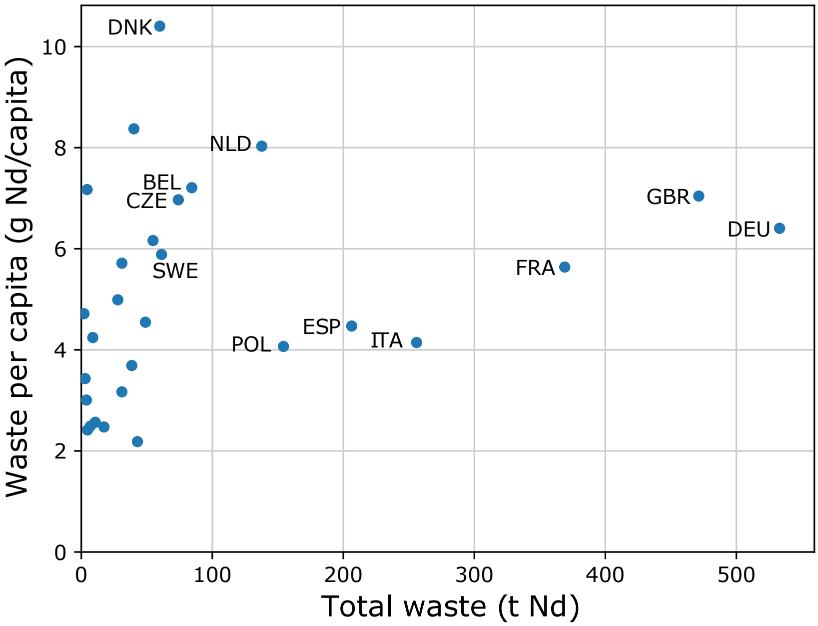

FIGURE B.1: Per capita Nd waste generation per country, in 2019.

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C Appendix to Chapter 4

C.1 Information on magnet wastes and applications

TABLE C.1: Weight of magnets and magnet assemblies in EoL products.

EoL product	Assembly weight (kg)	Magnet weight (g)	Source
EV drive rotor	10.9–14.6	1500–2000	[1]
HDD	0.488	15	[2]
Industrial pumps	1.6	132	INS*
Speaker assembly	4	500	B&C*
TV speaker	0.037	5	STENA*
Wind turbine drive (10 MW)	$149{\cdot}10^3$	$11.88{\cdot}10^3$	[3]

* SUSMAGPRO project partner.

TABLE C.2: Breakdown of demand and waste flows for NdFeB magnets by application, in the EU and the UK in 2019 (based on Chapter 3).

Application	Waste share	Demand share	Magnet type
HDDs	14.7%	2.5%	sintered
Speakers	10.4%	11.3%	sintered
Other small electronics	21.8%	9.8%	sintered
Electric vehicles	0.6%	3.6%	sintered
Wind power	0.4%	12.8%	sintered
Conventional cars	12.7%	14.8%	sintered, bonded
Industrial applications	21.6%	34.7%	sintered
HVAC	6.1%	4.3%	sintered, bonded
Large electric devices	8.5%	4.2%	sintered, bonded
Other	3.2%	2.0%	sintered, bonded

C.2 Unit process descriptions

C.2.1 General approach

As introduced in Chapter 4, we consider the following configurations: pilot process, process changes, size scaling, internal recycling, optimization, industrial reference process, and the thermodynamic or theoretical optimum.

For size scaling, equipment with an approximate capacity of 200 t/a was selected. For reference, the target capacity of both HyProMag plants is 100 t/a. Further scaling beyond 200 t/a is achieved mostly by parallel processing. Therefore it will only marginally change the process performance.

The equipment is operated 8 hours per day and 240 days per year. For processes that take more than 8 h per batch, 240 batches per year were assumed. The lifespan of machines was estimated by technical experts, and varies between 8 and 30 years. The upscaled process has 16 h per day, and optimized 24 h per day.

For comparison, also the industrial process performance and the thermodynamic or theoretical optimum were determined. The industrial reference is often a proxy, i.e. a similar unit process from a comparable sector. For example, metal injection moulding (MIM) of steel powders was used as a proxy for MIM of Nd-Fe-B powders. The theoretical optimum describes an ideal process, with an energy efficiency of 100%, and no material loss.

Table C.3 provides a data quality score for each process configuration, based on the pedigree matrix by Weidema [4].

C.2.2 Unit processes

The description of all unit processes can be found online at dx.doi.org/10.1016/J.JClePro.2024.142453, in Electronic Supplementary Information (ESI) 1.

C.2.3 Effective emission reduction

Table C.4 summarizes the contributions of improvements to lowering the environmental impacts of recycled magnets.

TABLE C.3: Data quality indicators for the life cycle inventory of each process configuration.

Indicator	Pilot process	Process change	Size scaling	Internal recycling	Optimization	Industrial reference
Reliability	1	2	2	3	4	1
Completeness	4	5	5	5	5	2
Temporal correlation	1	1	1	2	3	3
Geographical correlation	1	1	1	1	1	3
Further technological correlation	1	1	2	3	5	5
Average score	1.6	2	2.2	2.8	3.6	2.8

TABLE C.4: Process improvements grouped by their effectiveness for impact reduction.

Improvements with significant effect	Improvements with moderate effect (or process level only)	Processes with marginal effect
Reduced loss at sieving	Choose binder with low melting point	ICP-OES analysis
Use recycled NdH2 from HDDR	Switch to ultrasonic sieving	QR-code scanner
Reduce the amount of NdH$_2$	Switch to induction demagnetization	Coating the magnet
Internal recycling of inert gases	More magnets per mould (MIM process)	Magnetization
Solvent distillation and reuse	Cooling water recycling	Parallel cutters for HDDs
Less rotations of HPMS vessel		
Size scaling of furnaces (e.g. sintering)		
Optimized sandblasting		

C.3 Life cycle impact assessment

The applied set of impact categories is listed in Table C.5. The corresponding characterization factors and methods are adopted from Fazio et al. [5].

TABLE C.5: Life cycle impact categories and indicators, as recommended by the EF 3.0 scheme.

Impact category	Indicator	Unit
Climate change	Global warming potential (GWP100)	kg CO_2 eq
Ozone depletion	Ozone depletion potential (ODP)	kg CFC-11 eq
Respiratory inorganics	Human health effects associated with exposure to $PM_{2.5}$	Disease incidences
Photochemical ozone formation	Tropospheric ozone concentration increase	kg NMVOC eq
Acidification	Accumulated exceedance (AE)	mol H^+ eq
Eutrophication, terrestrial	Accumulated exceedance (AE)	mol N eq
Eutrophication, aquatic freshwater	Fraction of nutrients reaching freshwater end compartment (P)	mol P eq
Eutrophication, aquatic marine	Fraction of nutrients reaching marine end compartment (N)	mol N eq
Ionising radiation, human health	Human exposure efficiency relative to ^{235}U	kBq ^{235}U
Human toxicity, cancer effects	Comparative toxic unit for humans	CTUh
Human toxicity, non- cancer effects	Comparative toxic unit for humans	CTUh
Freshwater ecotoxicity	Comparative toxic unit for ecosystems	CTUe
Resource use, energy carriers	Abiotic depletion potential (ADP): fossil fuels	MJ
Resource use, minerals and metals	Abiotic depletion potential (ADP): elements (ultimate reserves)	kg SB eq
Land use	Soil quality index (aggregating biotic production, erosion resistance, mechanical filtration, and groundwater replenishment)	–
Water use	Water depletion potential (not from EF, see Table C.6)	m^3

C.4 Characterization factors for water use

Water use was calculated as the depletion of surface water. The extraction of water from water bodies contributes to depletion, whereas emission of water (not to air) is counted as restoration. All quantities are in units of square meter.

TABLE C.6: Characterization factors for water use. The contributions are in cubic meters.

Biosphere exchange	Exchange categories	Contrib.
Water, cooling, unspecified natural origin	natural resource::in water	+1
Water, in air	natural resource::in air	+1
Water, lake	natural resource::in water	+1
Water, river	natural resource::in water	+1
Water, salt, ocean	natural resource::in water	+1
Water, salt, sole	natural resource::in water	+1
Water, turbine use, unspecified natural origin	natural resource::in water	+1
Water, unspecified natural origin	natural resource::in ground	+1
Water, unspecified natural origin	natural resource::in water	+1
Water, unspecified natural origin	natural resource::fossil well	+1
Water, well, in ground	natural resource::in water	+1
Fresh water (obsolete)	water::surface water	−1
Water	water::surface water	−1
Water	water::ground-	−1
Water	water::fossil well	−1
Water	water::ground-, long-term	−1
Water	water	−1

C.5 Relative impacts

Supplementing Figure 4.7 of this thesis, Figure C.1 below shows the relative impact of all 16 impact categories (listed in Table C.5). The gray data series indicate that the impacts mostly follow the trend of selected impact categories (in color).

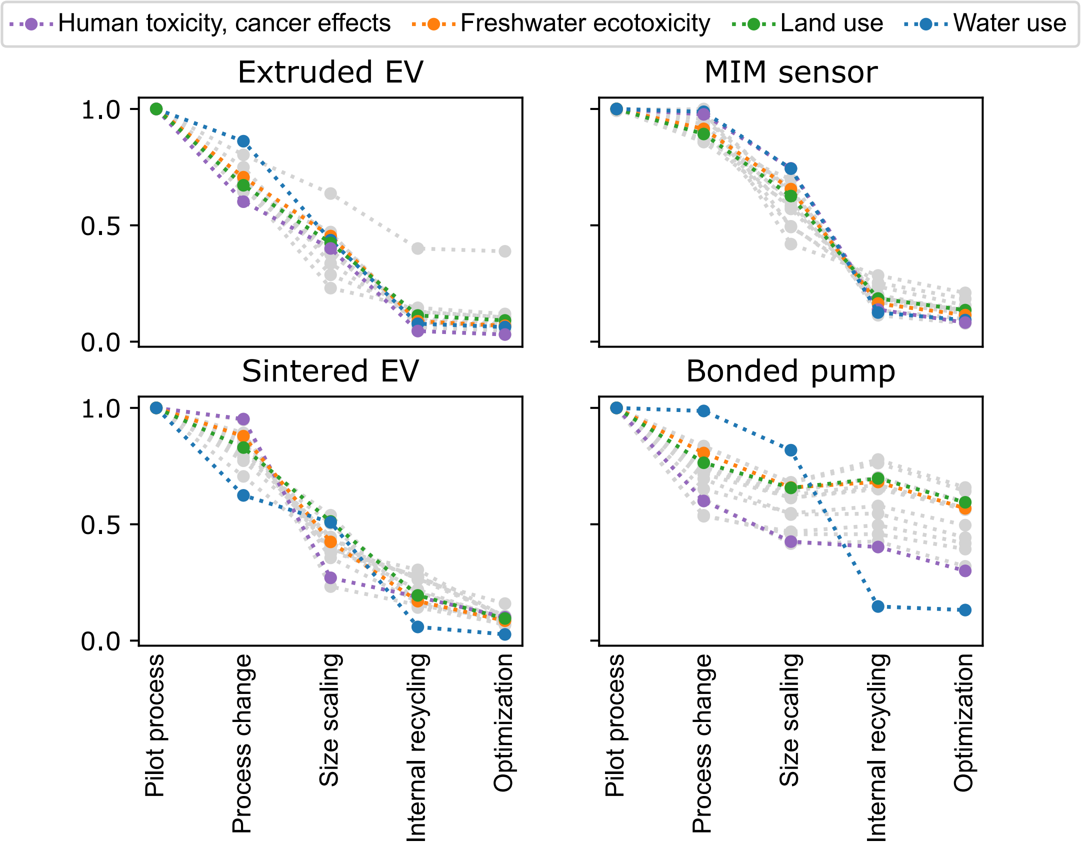

FIGURE C.1: Environmental impact changes due to projected technology developments for recycled demonstrator magnets, for all 16 impact categories. The values are plotted relative to the impacts at pilot scale.

To show the sensitivity of the LCA results to the electricity mix, a sensitivity analysis was conducted for the scenario of recycled sintered magnets for EV motors, produced from a mix of EoL magnets using industrial reference technology. The effect on different impact categories is shown in Figure C.2. In the default scenario, recycling processes use the average European electricity mix. In China and Poland, a large share of electricity is generated by coal-fired power plants, resulting in higher impacts in several categories. On the other hand, France uses mostly nuclear energy, therefore the ionizing radiation impacts are higher while most other impacts are below the base case.

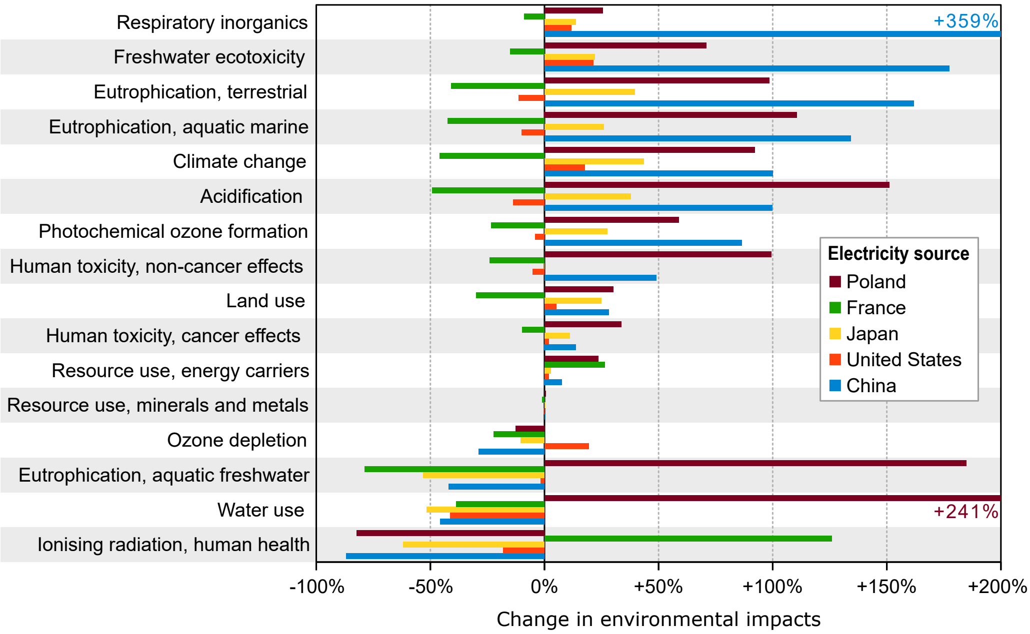

FIGURE C.2: Change in environmental impacts resulting from different electricity mixes. The base case is 1 kg recycled sintered magnets for EV motors, produced from a mix of EoL magnets using industrial reference technology and the European electricity mix.

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D Appendix to Chapter 5

D.1 Modeling approaches

D.1.1 Modeling learning effects

TABLE D.1: An overview of methods to estimate future costs of technologies, as presented by Santhakumar et al. [1]. For each development phase, they recommend to use a different experience curve approach, complemented with bottom-up modeling of the production process and optionally other approaches.

Development phase	Experience curve approach	Complementary approaches
Prototype and Demonstration	Component-based model utilizing learning assumptions	Qualitative approaches (TIS theory)
Initial build-up phase	Single-factor model utilizing empirical observations	Qualitative approaches (TIS theory)
Upscaling and Growth phase	Multi-factor model	Technology diffusion curves, Scenario analysis
Saturation	Multi-factor model or single-factor model for aggregated learning rate	

TIS: technological innovation system.

D.1.2 Modeling size scaling

Size scaling of products or processes can be modeled using scaling relations, as presented by Caduff et al. [2, 3] and Piccinno et al. [4]. Two approaches are described below.

Empirical scaling laws are derived from empirical observations and data collected during the scaling of similar processes or technologies. These laws help to extrapolate technology performance indicators, such as energy consumption or production efficiency, to larger scales.

Equation D.1 gives a scaling law that relates the production capacity (or size) of equipment to environmental impact. Here EI_2 is the environmental impact of equipment

2; EI_1 is the environmental impact of equipment 1; X_2 is the capacity factor of equipment 2; X_1 is the capacity factor of equipment 1 and b is the size scaling factor.

$$I_2 = I_1 \cdot \frac{X_2}{X_1}^{b} \tag{D.1}$$

Reported cost scaling factors are typically between 0.5 and 1. If no specific data are available, a scaling factor of 0.6 is recommended [3].

Theoretical models are based on thermodynamic frameworks or mathematical models to predict how changes in size or scale influence the inputs and outputs of a process. These models typically integrate knowledge from physics, chemistry, and engineering.

D.2 Empirical data

To find studies with empirical evidence, we compiled search queries using the keywords in Table D.2. Each query included keywords from all columns.

TABLE D.2: Keywords used to search literature on empirical evidence for environmental learning.

Technological learning	+	Industrial technology	+	Environment
experience curve		industrial process		carbon intensity
learning	+	manufacturing	+	ecotoxicity
learning curve		manufacturing industry		efficiency
progress ratio		process		embodied energy
technological learning		technology		emissions
				energy
				environmental
				eutrophication
				historical emissions
				material inputs
				materials, resources
				ozone
				SOx
				waste reduction

Empirical data from previous studies are listed in Table D.3. These are environmental learning rates; cost-based learning rates are provided in Table D.4.

TABLE D.3: Learning rates for environmental indicators, supporting Figure 3 in the main text. The learning rates for efficiency are negative, because the efficiency increases. Note that only a selection of the results from Stamford & Azapagic [5] are listed; see the source publication for full details.

Product [b]	Indicator [b]	LR	σ	R^2	Source
Ammonia, BAT	SEC [c]	28.9%		0.997	Ramírez & Worrell [6]
Ammonia, BAT	SEC	19.0%		0.951	Ramírez & Worrell [6]
Ammonia, average	SEC	23.1%		0.925	Ramírez & Worrell [6]
Urea, BAT	SEC [c]	11.4%		0.856	Ramírez & Worrell [6]
Urea, average	SEC [c]	8.9%		0.724	Ramírez & Worrell [6]
Ethanol from corn	SEC	17.3%		0.825	Hettinga et al. [7]
Pulp and paper	SEC, primary	9.0%	1.5%	0.84	Brucker et al. [8]
Clinker	SEC, primary	12.5%	3.0%		Brucker et al. [8]
Cement	SEC, primary	9.5%	2.0%	0.93	Brucker et al. [8]
Crude steel	SEC, primary	9.5%	1.5%	0.8	Brucker et al. [8]
Primary aluminum electrolysis	SEC, primary	3.5%	1.0%		Brucker et al. [8]
Poly-Si PV	GHG emission	16.5%	2.3%		Louwen et al. [9]
Mono-Si PV	GHG emission	23.6%	1.9%		Louwen et al. [9]
Poly-Si PV	CED	12.6%	0.9%		Louwen et al. [9]
Mono-Si PV	CED	11.9%	1.0%		Louwen et al. [9]
CdTe PV	GHG emission	12.8%	0.3%		Bergesen & Suh [10]
CdTe PV	Efficiency (W/m^2)	-5.7%			Bergesen & Suh [10]
CdTe PV	Electricity use	26.3%			Bergesen & Suh [10]
CdTe PV	CdTe use (g/cell)	8.9%			Bergesen & Suh [10]
CdTe PV	CdTe use (g/W)	13.8%			Bergesen & Suh [10]
Steel wire	Waste rate	11.7%			Lapré et al. [11]
PV	Si use (g/cell)	5.9%	0.8%	0.85	Louwen et al. [12]
PV	Ag use (g/cell)	28.1%	1.7%	0.95	Louwen et al. [12]
Mono-Si PV	Efficiency (W/m^2)	-4.7%	0.5%	0.93	Louwen et al. [12]
Poly-Si PV	Efficiency (W/m^2)	-3.9%	0.1%	0.99	Louwen et al. [12]
Mono-Si PV	Si use (g/W)	10.1%	1.3%		Louwen et al. [12]
Poly-Si PV	Si use (g/W)	9.4%	0.9%		Louwen et al. [12]
Mono-Si PV	Ag use (g/W)	31.3%	2.2%		Louwen et al. [12]
Poly-Si PV	Ag use (g/W)	30.8%	1.8%		Louwen et al. [12]
Mono-Si PV	GHG emission	10%			Stamford & Azapagic [5]

Product [b]	Indicator [b]	LR	σ	R^2	Source
Mono-Si PV	ODP (kg CFC11-eq.)	24%			Stamford & Azapagic [5]
Mono-Si PV	ADPe (kg Sb-eq.)	13%			Stamford & Azapagic [5]
Mono-Si PV	FAETP (kg DCB-eq.)	8%			Stamford & Azapagic [5]
Poly-Si PV	GHG emission	11%			Stamford & Azapagic [5]
Poly-Si PV	ODP (kg CFC11-eq.)	26%			Stamford & Azapagic [5]
Poly-Si PV	ADPe (kg Sb-eq.)	13%			Stamford & Azapagic [5]
Poly-Si PV	FAETP (kg DCB-eq.)	8%			Stamford & Azapagic [5]
Pig iron	SEC	12.3%		0.950	Gutowski et al. [13]
Aluminum smelting	SEC	8.9%		0.983	Gutowski et al. [13]
PV modules	CED (MJ/W)	20%			Görig & Breyer [14]
Mono-Si PV	CED (MJ/W)	18%			Görig & Breyer [14]
Poly-Si PV	CED (MJ/W)	24%			Görig & Breyer [14]
CdTe PV	CED (MJ/W)	7%			Görig & Breyer [14]
Mono-Si PV	CED (MJ/m^2)	14%			Görig & Breyer [14]
Poly-Si PV	CED (MJ/m^2)	22%			Görig & Breyer [14]
CdTe PV	CED (MJ/m^2)	3%			Görig & Breyer [14]
Thermal powerplant construction	CO_2 (kg/MW)	2.4%		0.951	Yuan et al. [15]
Hydropower construction	CO_2 (kg/MW)	48.4%		0.954	Yuan et al. [15]
Nuclear powerplant construction	CO_2 (kg/MW)	23.7%		0.955	Yuan et al. [15]
Wind construction	CO_2 (kg/MW)	17.4%		0.985	Yuan et al. [15]
Solar construction	CO_2 (kg/MW)	9.9%		1	Yuan et al. [15]
CO_2 absorption	Energy use (GJ/t)	23.2%		0.539	Rochedo & Szklo [16]

[a] LR: learning rate, σ: standard deviation, R^2: coefficient of determination.

[b] BAT: best available technology, CED: cumulative energy demand, SEC: specific energy consumption, GHG: greenhouse gas, ODP: ozone layer depletion potential, ADPe: abiotic depletion potential—elements, FAETP: freshwater ecotoxicity potential, DCB: *1,4*-dichlorobenzene.

[c] Curve fitting accounted for the theoretical minimum impact I_{min}.

D.3 Supplementary examples

The following sections provide examples belonging to the steps of the proposed procedure. Environmental hotspots for step 2, stakeholders and incentives for step 3, technology changes and economic effects for step 4, and learning rates for step 5.

D.3.1 Environmental hotspots

- Process energy use
- material use
- waste or discard rate
- waste treatment
- equipment use
- use-phase material or energy use

D.3.2 Stakeholders

- Customers and consumers
- Investors
- Value chain partners (suppliers)
- Societal organizations (NGOs, labor unions)
- Governments and regulators
- Competitors

D.3.3 Incentives

External forces that influence company decision on technological change:

- Customer preferences (normative pressure)
- Competitive pressure (mimetic pressure)
- Restrictive environmental regulation (coercive pressure)
- Market-stimulating regulation
- Preferences of investors and value chain partners (normative pressure)

D.3.4 Technology changes

- Improved resource and energy efficiency of processes
    - Accumulation of experience by workers and machine optimization [17]
    - Optimization of logistics, operating parameters, feedstock compositions, etc.
    - Learning about redundant steps or superfluous safety measures [18]
    - Increased equipment productivity due to better tuning and more operating hours
    - Process automation: shift from human labour to electric equipment [19]

- Improved design of process equipment [20]
    - Less conservative design (no oversizing, lower safety margins)
    - Less spare or redundant equipment
    - Automation of equipment production
    - Maintenance and repair for service life extension
- Product improvement for better performance during the use phase [21], e.g. higher energy efficiency, material efficiency, or longer lifespan.
- Implementation of end-of-pipe solutions [22]
- Size scaling to increase production capacity [3; 4]
- Product improvement to meet the preferences of environmentally conscious consumers [23]
- Shift to inputs[1] or suppliers with lower impacts [10]
- Material or process substitution [24]

D.3.5 Economic learning effects

The following financial parameters decrease with growing experience and larger scales, thus contributing to cost reduction [1; 18; 25]:

- Budget overruns due to delayed construction or production
- Cost of capital (interest payments)
- Regulatory fees (permitting)
- Commercial and legal risk mitigation,
- Insurance costs
- Overhead costs
- Marketing expenditures for new products
- Single orders rather than bulk purchases

Besides, outsourcing to low-income countries reduces labor costs [19], while the environmental impacts may increase due to lower environmental standards in these countries. Raw material price fluctuations may also distort the observed trend.

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[1] Inputs include feedstock materials, energy carriers, and equipment.

D.3.6 Learning rates

Table D.4: Mean learning rates for production costs at company-level for US manufacturing industries, by two-digit standard industrial classification (SIC) code. Learning rates for lower-level SIC sectors and standard deviations are provided in the source publication [26].

SIC	Description	Mean learning rate
20	Food and kindred products	0.361
22	Textile mill products	0.256
23	Apparel	0.285
24	Lumber and wood products	0.223
25	Furniture and fixtures	0.261
26	Paper and allied products	0.235
28	Chemicals and allied products	0.417
29	Petroleum refining	0.350
30	Rubber and misc. plastic	0.282
31	Leather and leather products	0.154
32	Stone, clay, concrete	0.290
33	Primary metal industries	0.229
34	Fabricated metal (excl. machinery)	0.257
35	Machinery (incl. computers)	0.286
36	Electrical and electronic equipment	0.304
37	Transportation equipment	0.263
38	Measuring instruments	0.356
39	Misc. manufacturing industries	0.272
	All	0.288

D.4 Illustrative example for copper

D.4.1 Cost decomposition

Costs of copper production were broken down by inputs to the main processes, as shown in Figure D.1 below. The added value of each process was calculated as the difference in costs between inputs and output. Added value is the combination of labour costs and profits, and may also include taxes and levies. In the case of mining, added value could be spend on mining taxes or the acquisition of land.

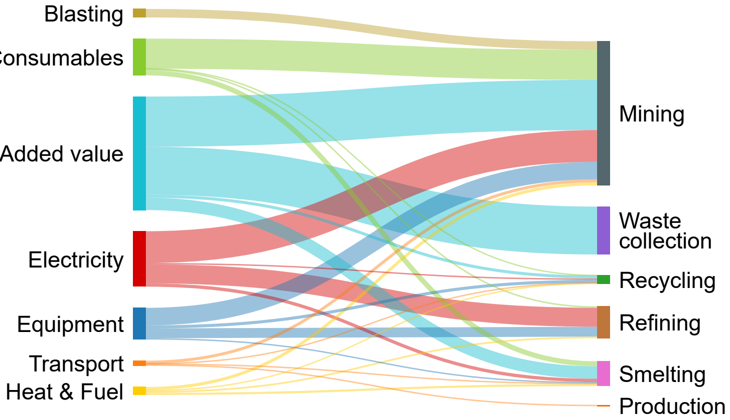

FIGURE D.1: Cost breakdown for the production of copper, by input (left) and by process (right).

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