Advancing the understanding and impact assessment of circular economy product legislation in the European Union

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Dissertation

von

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aus Aachen

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Acknowledgments

The present moment is filled with joy and happiness.
If you are attentive, you will see it.
- Thich Nhat Hanh

But wait! Have we not learned that meeting the needs of the present risks compromising those of the future? At least with a lively earth for humanity, this is truer than ever.

For the path to an academic title like a doctor, I see some similarities. You have to plan, you have to sit down, and you have to persevere. For all that effort in the present, you are unlikely to get any immediate rewards. You get wrong results, you have stressful days, and you face shattering criticism from reviewers.

For what it's worth, you look to the horizon and see authorship, a shining title, and true impact of your work. So you sit back down and you persist until the fog clears, and maybe you find a few sparks of joy along the way. A number falls into place, a positive comment, a paper published.

Full of pride, you turn around - only to realize that no one cites your paper, that titles are not a destination, and that your achievements are largely based on your privileges.

What remains are not two letters on a business card, but more importantly, the people along the way. And as Thich Nhat Hanh suggests, I want to be attentive by seeing some of those who have brought joy and happiness to the past and present without compromise.

My mom. Who saw me embarking on this journey and gave me everything I needed to walk through on my own.

My sister and my father, who continue to be an inspiration. And Federica, who put up with my workiness and continues to give me support and laughter every day.

My ISI colleagues, from whom I learn every day and without whom I would have been lost. First of all Antoine, who has been a fountain of ideas from day one and has always been there when I needed support. Clemens, as an ideally balanced supervisor, who brought me to Darmstadt and, together with Vanessa, gave me the right mix of guidance and freedom to strive.

My co-authors, friends, and family.

And finally, the serendipity of projects, reviewers, conversations, character traits (in myself and others), vacations, conferences, epidemics, rhetoric theories, Italian styles, and everything else that gave me a direction to embrace and steered my sails in a way that made this feel surprisingly fluid.

Grazie.

Kurzzusammenfassung

Produkte, vor allem energieverbrauchsrelevante, sind für große Umweltauswirkungen verantwortlich. Circular Economy wird als ein Weg zur Verringerung der Umweltauswirkungen angesehen, und Gesetzgebung kann als Hebel zur Implementierung von Circular Economy Maßnahmen dienen. Vor diesem Hintergrund ist diese Dissertation an der Schnittstelle der drei Bereiche Kreislaufwirtschaft, Gesetzgebung und (energieverbrauchsrelevante) Produkte angesiedelt, mit dem Ziel, das Verständnis für die Gesetzgebung im Bereich der Circular Economy, einschließlich der Analyse ihrer Umweltwirkungen, zu verbessern.

Die im Rahmen der kumulativen Dissertation durchgeführte Forschung kombiniert eine Vielzahl von Methoden wie qualitative und quantitative Textanalyse, systematische Literaturanalyse, politikwissenschaftliche Analyse unter Verwendung des Advocacy-Coalition-Frameworks mit qualitativer Inhaltsanalyse und Interviews sowie Szenariomodellierung mit einer Kombination aus Materialflussanalyse und Ökobilanz.

Die Ergebnisse zeigen, dass der Schwerpunkt der EU-Produktgesetzgebung in der Vergangenheit auf der Verringerung des Energieverbrauchs während der Nutzungsphase lag, dass aber die Umweltauswirkungen über die Nutzungsphase hinaus zunehmend an Bedeutung gewinnen und ein deutlicher Anstieg von Circular Economy verwandten Maßnahmen festgestellt werden konnte, wobei Unterschiede zwischen Produktfamilien und Kategorien von Maßnahmen bestehen.

Um die begriffliche Unklarheit rund um den Begriff "Circular Economy" zu verringern, wurde auf der Grundlage der Analyse der Ökodesign-Gesetzgebung und der EU-Batterieregulierungen eine Taxonomie von Circular Economy Maßnahmen entwickelt.

Die Batteriegesetzgebung wurde auch als Grundlage für die Analyse des politischen Entscheidungsprozesses verwendet, um die Ursachen für die Entwicklung hin zu einer stärkeren Berücksichtigung von Circular Economy Aspekten zu ermitteln. Es wurde die Hypothese aufgestellt, dass die Kreislaufwirtschaft der gemeinsame Nenner ist, der wirtschaftliche, ökologische und soziale Interessen in Einklang bringt.

Um die bestehenden Methoden der Analyse von Umweltwirkungen zu verbessern, wurde ein Scoping der Methoden zur Folgenabschätzung und eine systematische Literaturrecherche durchgeführt. Es zeigte sich, dass die Kombination von Materialflussanalyse und Ökobilanz für eine solche Analyse geeignet ist. Die Kombination ermöglicht es, die detaillierte ökologische Analyse der Ökobilanz zu erweitern, indem die Wechselwirkungen der Stoffströme über lange Zeiträume und große räumliche Bereiche der Materialflussanalyse berücksichtigt werden. Anschließend wurde ein Modell für die Ex-ante-Bewertung von Materialflüssen und Umweltauswirkungen von Circular Economy Maßnahmen entwickelt. Das Modell verwendet einen mehrdimensionalen Ansatz, bei dem die Material- und Umweltauswirkungen aus den Produktströmen über eine Produktdatenbank abgeleitet werden, die die physikalischen Eigenschaften der Produktvarianten definiert. Um seinen Nutzen zu demonstrieren, wurde das Modell in zwei Fallstudien zu Industriemotoren und Traktionsbatterien angewendet. Die Ergebnisse zeigen, dass ein breites Spektrum an politischen Anforderungen bewertet werden kann und dass die Ergebnisse mehrdimensional und produktspezifisch sind.

Diese Dissertation zielt darauf ab, das Verständnis und die Analyse von Umweltwirkungen der Circular Economy Gesetzgebung in der Europäischen Union voranzutreiben.

Abstract

Products, especially energy-related, are responsible for large environmental impacts along their supply chains. Circular economy is seen as a way to reduce environmental impacts and legislation can serve as a lever to facilitate the implementation of circular economy related measures. Against this background, this dissertation is located at the intersection of the three domains of circular economy, legislation and (energy-related) products, with the aim of improving the understanding of circular economy product legislation, including its impact assessment.

The research conducted as part of the cumulative dissertation combines a variety of methods such as directed content and keyword analysis, systematic literature analysis, policy analysis using the advocacy coalition framework with qualitative content analysis and interviews, and scenario modelling using a combination of material flow analysis and life cycle assessment.

The results show that while the focus of EU product legislation in the past has been on reducing energy consumption during the use phase, environmental impacts beyond the use phase are becoming increasingly important and a clear increase in circular economy related requirements could be identified, with differences between product families and circular economy categories.

In order to reduce the conceptual ambiguity around the term "circular economy", a taxonomy of circular economy requirements was developed based on the analysis of the Ecodesign implementing measures and the EU battery legislation.

The battery legislation was also used as a basis to analyse the policy making process for the underlying drivers of the evolution towards a stronger consideration of circular economy aspects. A hypothesis was derived that circular economy is the common denominator aligning economic, environmental and social interests.

The apparent increase of circular economy related aspects in product legislation requires an improvement of existing environmental impact assessment methods. A scoping of impact assessment methods and a systematic literature review showed the suitability of combining material flow analysis and life cycle assessment for such an impact assessment. The combination allows to extend the in-depth environmental assessment of life cycle assessment by considering the interactions of material flows over long time periods and large spatial scopes of material flow analysis. Subsequently, a model was developed for the ex-ante assessment of material flows and environmental impacts of a range of circular economy product policies. The model uses a layered approach where material and environmental impacts are derived from product flows via a product database that defines physical properties for product variants. To demonstrate its utility, the model was applied in two case studies of industrial motors and traction batteries. The case study results demonstrated the broad range of policy requirements that can be assessed and the multi-dimensional and product specific nature of the results.

Circling around circular means and promises this dissertation aims to advance the understanding and impact assessment of circular economy product legislation in the European Union.

Contents

Preamble to the cumulative dissertation

This cumulative dissertation consists of five scientific publications, constituting chapter two to six. Four of these publications have been published in international peer-reviewed journals. Paper five has been published as a peer-reviewed conference paper (see Figure 1 for the timeline).

1. **Robin Barkhausen**, Antoine Durand, Katharina Fick (2022): Review and Analysis of Ecodesign Directive Implementing Measures: Product Regulations Shifting from Energy Efficiency towards a Circular Economy. In: Sustainability.
   DOI: https://doi.org/10.3390/su141610318
2. **Robin Barkhausen**, Leon Rostek, Zoe Chunyu Miao, Vanessa Zeller (2023): Combinations of material flow analysis and life cycle assessment and their applicability to assess circular economy requirements in EU product regulations. A systematic literature review. In: Journal of Cleaner Production.
   DOI: https://doi.org/10.1016/j.jclepro.2023.137017
3. **Robin Barkhausen**, Katharina Fick, Antoine Durand, Clemens Rohde (2023): Analysing policy change towards the circular economy at the example of EU battery legislation. In: Renewable and Sustainable Energy Reviews.
   DOI: https://doi.org/10.1016/j.rser.2023.113665
4. **Robin Barkhausen**, Antoine Durand, Yan Yi Fong, Vanessa Zeller, Clemens Rohde (2024): Modeling stock, material and environmental impacts of circular economy product policies. Trade-offs between early replacement and repair of electric motors. In: Resources, Conservation and Recycling.
   DOI: https://doi.org/10.1016/j.resconrec.2024.107600
5. **Robin Barkhausen** (2024): Coherence of Novel Policies for Lithium-Ion Batteries for Electric Vehicles: A Multidimensional Analysis of Material Flows and Environmental Impacts. In: eceee 2024 Summer Study proceedings.
   DOI: 10.24406/publica-3334

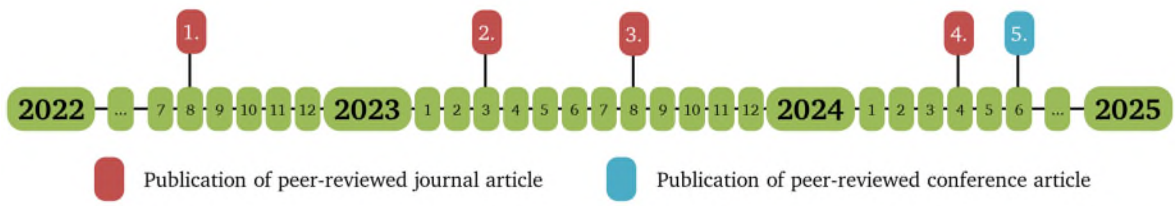


Figure 1 Timeline of the cumulative dissertation.

All illustrations in this dissertation are, if not otherwise stated, self-created.

1. Introduction

The following subchapters provide a brief introduction to the research area. First, the underlying motivation for the research is presented (1.1). Then the background is described (1.2) providing information and definitions for the three central domains in which the dissertation is situated: circular economy (1.2.1), legislation (1.2.2), and energy-related products (1.2.3). The following subchapter describes the state of scientific research on methodological approaches for impact assessment of circular economy product policies (1.3). Based on the literature review, research gaps and research questions to be addressed are defined, and an overview of the structure of this dissertation is provided (1.4).

1.1. Motivation

In 1966, US economist Kenneth Boulding introduced the term "Spaceship Earth" as a metaphor for the finite nature of our planet's resources. He describes the Earth as "a single spaceship, with no unlimited reserves, either for extraction or pollution" (Boulding 1966). In a report for the Club of Rome, a group of researchers confirmed the validity of this metaphor. Based on current growth trends, limits to population growth, food production and consumption of non-renewable resources could be reached within the next hundred years (Meadows et al. 1972).

To counter this trend, human development must become sustainable, defined as "development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (WCED 1991). But how can we analyse where we are on the path to sustainability?

While the assessment of sustainability has often been performed as a relative assessment (for example in relative impact assessment in environmental LCA), more recently sustainability assessment has focused on an absolute assessment of environmental impacts. Humanity's safe operating space has been outlined with the concept of planetary boundaries, which define thresholds that, if exceeded, would have potentially adverse effects on human life on Earth and lead to abrupt and non-linear environmental changes (see Figure 2) (Rockström et al. 2009; Steffen et al. 2015).

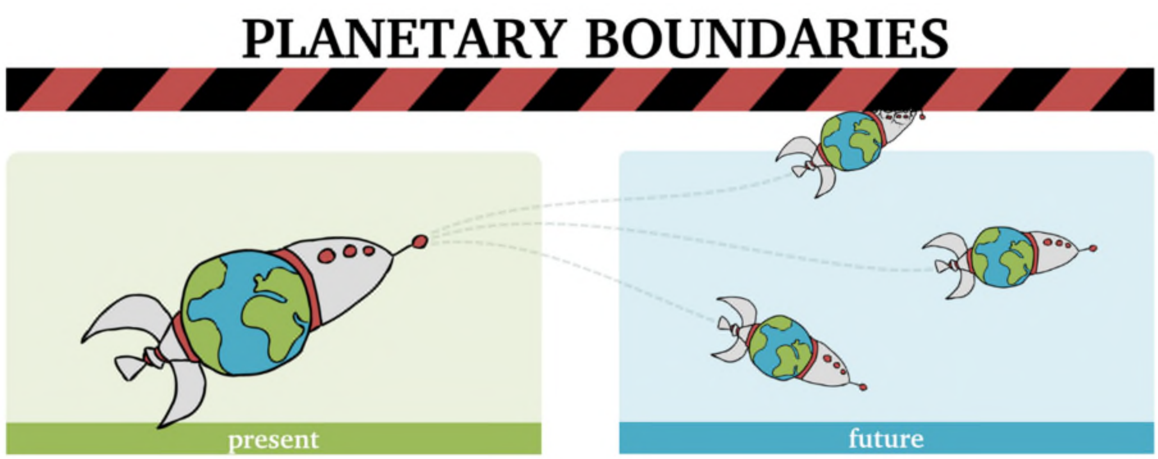


Figure 2 Spaceship Earth, sustainable development and planetary boundaries.

In the European Union (EU) context, the Joint Research Centre has developed the Consumption Footprint to assess the environmental footprint of representative products of the most consumed product groups in the EU (food, housing, mobility, household goods and appliances) (Sala et al. 2019). The results show that the per capita environmental impact of products in the EU exceeds the safe operating space for humanity in the impact categories of climate change, particulate matter, resource use, freshwater eutrophication and human toxicity-cancer (the original planetary boundaries have been translated by the Joint Research Centre into midpoint indicators). Appliances[1] alone account for almost half of the impacts in the impact dimension of mineral and metal resource use (Sala et al. 2019). Furthermore, waste generation in the EU is reaching alarmingly high levels, and quantities have not been effectively reduced (Eurostat reports 2.3 billion tonnes of waste in 2018 (Eurostat 2021)).

As a concept to bring the impact of products back into the safe operating space and also to reduce import dependencies, circular economy could play an important role. While large part of the economy still follows linear production patterns based on virgin raw materials (European Commission 2019c), moving away from a linear economic system towards a circular economy promises to reduce the environmental impacts associated with a wide range of products.

Product policy can act as a lever to foster the implementation of new concepts, such as the circular economy, into the production of products. However, it is crucial to assess the expected impacts of a policy measure on the different dimensions of sustainability in an evidence-based manner. Therefore, this dissertation is positioned in the research field of product policy, and in particular policy impact modelling. It will be applied to the emerging field of circular economy, where many new types of product policies are seen, especially at EU level, and where evidence on their impacts is needed.

1.2. Background

1.2.1. Conceptualisation of the circular economy

Kirchherr et al. (2017) identified 114 different definitions of how to describe a circular economy, suggesting that it is a vague concept. Because of this vagueness, many scholars try to circumvent the conceptual ambiguity by talking about material or resource efficiency instead of circular economy as more clearly defined concepts. Material efficiency can be described as "the ratio between the performance output of a product, service or energy system and the input of materials required to provide such output" (Cordella et al. 2019). Resource efficiency can be defined similarly, however it is slightly broader concept since not only materials, but all resources are considered (Di Maio et al. 2017).

Circular economy, in contrast, describes a concept that has material and resource efficiency at its core, but goes beyond this to represent a more holistic and systemic version for an economic system that strives for sustainable development with a particular focus on the environmental and economical dimension. It can be differentiated from circularity, a metric of the "percentage of the value of stressed resources incorporated in a service or product that is returned after its

[1] Appliances include the categories white goods (e.g. washing machines), appliances for basic functions to housing (e.g. air conditioners, whereas space heaters are part of the category housing), and entertainment and leisure (e.g. television) (Reale et al. 2019).

end-of-life" (Di Maio et al. 2017). In the following paragraphs, some further reflections on the concept of circular economy are presented.

Conceptually, circular economy can be distinguished from a linear economy, a view of the economy that seems to be a sufficient representation of human-centric reality, but only if the environmental system is ignored (Pearce and Turner 1990). However, the environmental system provides inputs (or resources) to the economy and acts as a repository for waste and emissions (e.g. carbon dioxide emissions to the atmosphere or solid waste in landfills). According to the first law of thermodynamics, there must be a balance between the energy and matter supplied to the economy (from the environment) and the outputs returned to the environmental system (Pearce and Turner 1990). According to the second law of thermodynamics, certain quantities of natural resources are irreversibly damaged (or degraded) and thus lost when they are used for economic activities (Georgescu-Roegen 1971). The environment has a limited assimilative capacity to deal with this economic waste and therefore circular economy aims to reduce the pressure on the environment to act as a sink for waste, either by reducing the amount of waste produced or by converting waste back into resources (recycling) (Pearce and Turner 1990). In line with this theoretical concept, the European Commission defines a circular economy as an economy, "where the value of products, materials and resources is maintained in the economy for as long as possible, and the generation of waste is minimised" (European Commission 2015). The benefits of such an economic system are potentially linked to all dimensions of sustainable development, with a focus on the environmental and economic dimensions (Kirchherr et al. 2017). Based on these findings, in this dissertation circular economy can be understood as having the aim of reducing waste and prolonging the effective use of materials and resources in order to contribute to sustainable development[2].

For the European Commission, circular economy is linked to the goal of achieving climate neutrality by 2050. The 2020 Circular Economy Action Plan aims to support the climate neutrality goal by decoupling economic growth from resource use and reducing the impact of resource extraction and processing on greenhouse gas emissions and biodiversity loss (European Commission 2020e).

This highlights that the concept of circular economy serves to achieve a specific goal (e.g. reduced environmental impact or reduced import dependency), and is not an end in itself[3], thus avoiding what Harris et al. (2021) call "circularity for circularity's sake".

1.2.2. Circular economy on the political agenda

The conceptual ambiguity of circular economy is reflected in the changing role that the concept has played on the policy agenda in recent decades, with a shift in focus from local waste management and pollution to global impacts such as climate change and critical material supply

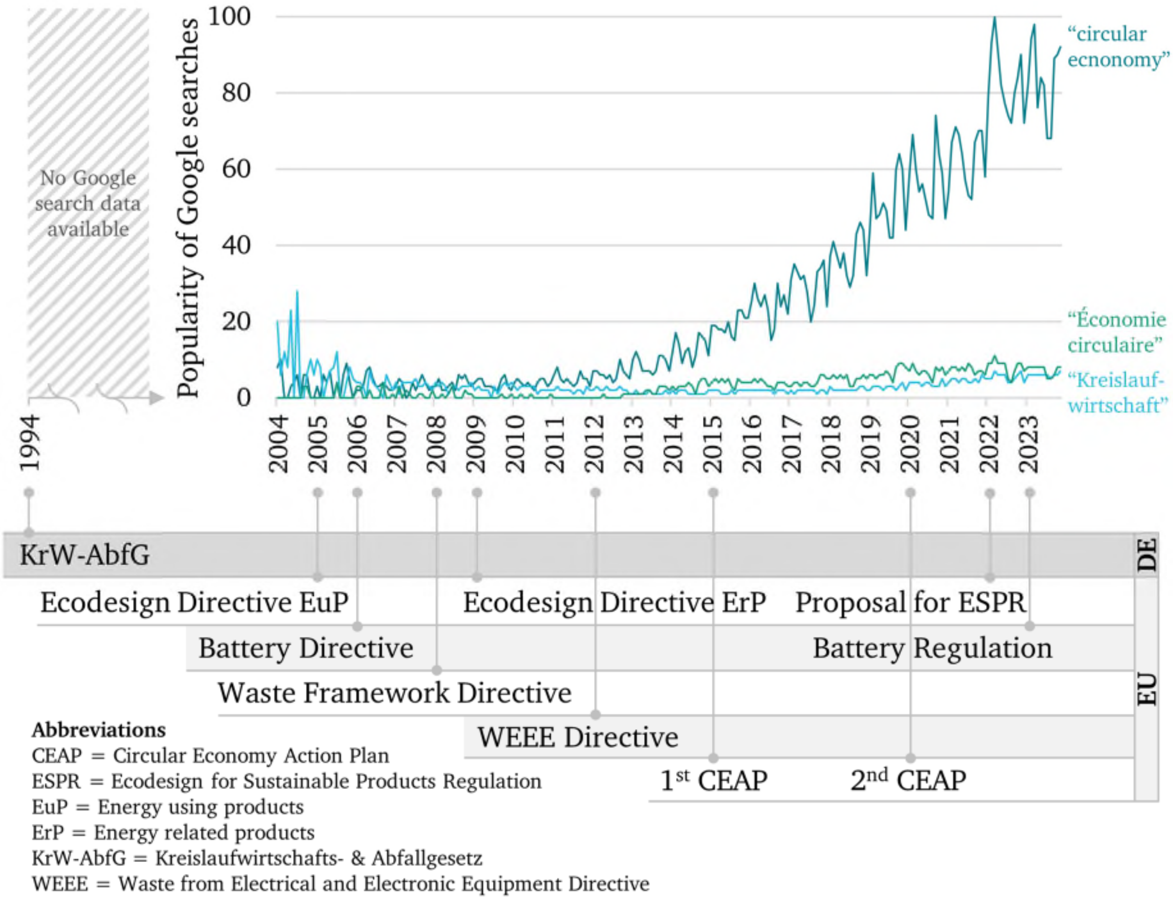
[2] Although this definition could be criticised for not meeting the hallmarks of a good definition (Figge et al. 2023) it is considered sufficient as a basis for this research.

[3] For example, maximising recycling efficiency to the limits of technical feasibility can disproportionately increase energy costs and cause more harm than good in terms of environmental impact.

risks. To provide an overview of the policy landscape, the following paragraphs highlight key legislation[4] related to circular economy, moving from the German to the EU level.

Under Angela Merkel as Environment Minister at that time, Germany passed a law specifically addressing the circular economy as early as 1994 with the Circular Economy and Waste Management Act (Kreislaufwirtschafts- und Abfallgesetz). This early interest in the circular economy was influenced by the oil crisis and concerns about local pollution from pesticides and landfills. While local resource scarcity played a role, the EU's dependence on imported materials and global environmental impacts such as climate change were not central (BMU 1996).

Public interest at EU level remained comparatively low and dwindled especially towards the end of the first decade of the 21st century (see Figure 3), despite the adoption of important legislation in the domain of circular economy.



Google search data based on (Google trends 2023). Data is normalised so that the largest entry is 100. Entries are also normalised to the total number of searches in each geographical location.
For some of the legislations listed, there are even earlier versions that are not included for simplicity.

Figure 3 Google search popularity of the term "circular economy" and its French ("Économie circulaire") and German ("Kreislaufwirtschaft") equivalent and (non-exhaustive) list of key circular economy product legislation.

[4] A legislation refers to a law or set of laws. The scope of this dissertation is the EU, where a legislation can either mean a regulation (directly applicable in all Member States) or a directive (has to be transposed into national law). Product group legislations drafted under the framework of the Ecodesign Directive are regulations, but also called implementing measures. A policy is understood as a broader framework and can refer to any guidelines or rules set an organisation.

As Figure 3 indicates, public interest in circular economy increased around the same time as the EU published its first Circular Economy Action Plan in 2015 (European Commission 2015). The focus increasingly shifted to the global implications of circular economy and its contribution to fighting the climate crisis and the global scarcity of mineral resources needed to defossilise industry. Although the Circular Economy Action Plan is only a communication and not a binding text, it was an early demonstration of the EU's political willingness to move towards a more circular economy.

Prior to the publication of the first Circular Economy Action Plan, the EU had already published important legislation, such as the Waste Framework Directive (European Commission 2008). Article 4 of this Directive defines a five-level waste hierarchy, from highest to lowest priority, with (1) prevention (or **r**educe), (2) preparation for **r**euse, (3) **r**ecycling, (4) other **r**ecovery (e.g. energy recovery) and (5) disposal (4Rs). The legislation also introduces the principle of extended producer responsibility, which means that the costs of waste management should be borne by the original waste producer or the current or former holders of the waste. While the Waste Framework Directive defines the waste hierarchy and acts as an overarching EU framework, it does not set circular economy requirements for individual product groups, leaving it up to Member States to decide what waste prevention measures to take. Leaving it up to Member states to take waste prevention measures, however, can lead to a situation with countries postponing major measures in order to expose other countries to risks or potential lock-ins first, as Wiesmeth (2021) points out.

To provide more product-specific[5] guidance and manage the rapid growth of electronic waste, the Directive on Waste from Electrical and Electronic Equipment was drafted and published in 2012 (amending the original 1996 version). The legislation introduces circular economy requirements such as mandatory collection rates and recovery targets (European Commission 2012a).

Several regulations complement the Waste Framework Directive and the Directive on Waste from Electrical and Electronic Equipment, but two in particular stand out.

The first is the Ecodesign Directive, published in its first version in 2005, which aims to improve the environmental performance of individual energy-consuming and, since 2009, energy-related product groups through binding EU regulations (so-called implementing measures) (European Commission 2005, 2009). The Ecodesign Directive allows a wide range of environmental aspects to be regulated in order to improve environmental performance along the entire supply chain. Initially, the implementing measures drafted under the Ecodesign Directive paid little attention to circular economy aspects and instead focused almost exclusively on energy efficiency in the use phase, although the scope of the Ecodesign Directive allowed to address them from the start (European Commission 2009). In March 2022, the EU published a proposal for a regulation for setting ecodesign requirements for sustainable products, which will replace the existing ecodesign framework and place greater emphasis on circular economy aspects such as durability, reparability and recyclability. The proposed legislation extends the scope beyond energy-related products (such as furniture and textiles), foresees the use of a

[5] A product can generally be described as the "result of a process" (ISO 9000:2000). In the context of this dissertation, a product is defined as any physical good, including both consumer goods and intermediate products.

digital product passport as an enabler of a circular economy, and includes measures to end the destruction of unsold consumer goods (European Commission 2022c).

The second important EU product legislation is the Batteries Directive, which, in contrast to the Ecodesign regulations, introduced circular economy aspects such as mandatory collection and recycling quotas at an early stage (European Commission 2006). In 2023, the Commission published a new Battery Regulation, containing very ambitious circular economy requirements (such as recycled content, digital product passport and material-specific recycling efficiency) (European Commission 2023a).

Based on the analysis of legislation, the implementing measures of the Ecodesign Directive and the Battery legislation seem to be suitable candidates to assess the role and potential of circular economy in EU product legislation.

1.2.3. Policy-making for energy-related products

When preparing new initiatives and proposals, the EU follows its Better Regulation toolbox. The toolbox suggests the use of impact assessments to "develop the Commission's policy response to a specific policy problem by providing the evidence base for - and the impacts of - different options". Policy options are usually compared with a benchmark reflecting a "no-policy-change" scenario, and multi-criteria analysis can be used when there are different criteria for comparing options (European Commission 2021b). One way of ensuring that impact assessments are carried out in line with the Better Regulation toolbox and that methodologies are sound is through the Regulatory Scrutiny Board, an independent body within the Commission that provides quality control of the legislative process by reviewing draft impact assessments (European Commission 2020d).

While impact assessment is carried out on a case-by-case basis in many legislations, the Ecodesign Directive provides a structured approach to analysing the potential impacts of all product groups regulated under its framework through the Methodology for Ecodesign of Energy-related Products (MEErP) (Kemna et al. 2011). At present, the core of the MEErP is a technical-economic and environmental assessment that evaluates life-cycle costs and environmental impacts. The environmental impact assessment is based on the EcoReport tool, a simplified LCA tool that assesses a limited number of environmental impact categories based on their mass in the final product. The environmental impacts for the different policy scenarios are then assessed using a stock model approach. Unlike the environmental assessment of individual products, where the EcoReport tool is provided as a dedicated and public Excel tool, there is no such tool for the environmental assessments of policy scenarios on EU market level, where only some basic theoretical requirements are outlined. The requirements define that the stock model should include, for example, annual sales, annual stock, average unitary impacts (only the significant ones) of the products sold and total impacts (stock x performance demand x unitary impact).

The MEErP and the EcoReport tool are currently being revised to update the data and categories of material and environmental impacts and to better reflect material efficiency aspects (European Commission 2021c). While there will be an updated version of the EcoReport tool with better integration of circular economy aspects, it is not expected that a dedicated tool will be provided to link the product level results with stocks or sales to assess impacts at EU level. The current stock model approach focuses on energy consumption in the use phase as the main impact category and does not provide a standard framework to adequately assess the impacts of circular economy policies beyond the impacts in the use phase. While some preparatory ecodesign studies (e.g. on batteries (Hettesheimer et al. 2019)) already extend the approach to individual material flows, a common approach to adequately consider circular economy aspects in policy impact assessments seems to be lacking.

The following chapter takes a closer look at existing scientific approaches that could be applied to the policy impact modelling process.

1.3. State of research

Policy impact assessments are a prerequisite for evidence-based policy making, and with the growth of circular economy policies, an appropriate adaptation of existing approaches is needed to assess the impacts and interactions of circular economy requirements.

Scientific articles dealing with the integration of circular economy aspects into product policy assessment date back to the early 2000s, and there are a variety of different approaches with different scopes of application. An initial search of the academic database Scopus for impact assessments related to product policy and circular economy identified six reviews dealing with this area of research. Most of them do not focus on policy assessment, but still provide a good overview of existing methodologies in the field of circular economy. A brief summary of the six reviews is provided below.

One review provides a very comprehensive overview of the field of circular economy research by analysing 565 articles (Merli et al. 2018). The authors find a large increase in publications on circular economy, with a particularly sharp increase after 2015, with the geographical focus of the studies largely concentrated in Europe and China. It was found that most articles focus only on the environmental or a combination of environmental and economic dimensions of sustainability, but the social dimension is increasingly included. Methods such as life cycle assessment (LCA) and material flow analysis (MFA) (including material flow cost accounting) are often used to model the impacts of circular economy, but may "fail to consider large-scale issues, like the scarcity of raw materials and the deterioration of products' value retention" (Merli et al. 2018). Indicators to assess circularity are found to be at an early stage of development. Overall, Merli et al. (2018) support the finding of (Kirchherr et al. 2017) (see 1.2.1) that circular economy has no clear conceptual boundaries, describing it as an "umbrella concept, associated with a variety of disciplines that define its roots".

Another review focuses on Italy, but also draws general conclusions. It analyses 609 scientific publications on national waste management in Italy and lists the available tools for evaluating circular economy policies (Camana et al. 2021). The authors find that the majority of the articles analysed focus on municipal solid waste (129), organic waste (112) and agriculture (61). And that the main assessment tools are material flow accounting, life cycle thinking and industrial ecology. In addition, the review highlights the importance of considering potential rebound effects and trade-offs in policy assessment (or design) and emphasises the role of the social dimension (e.g. acceptance of new technologies), which is not adequately addressed in many of the publications analysed. Life cycle thinking was identified as the most promising method, as it can take into account all life cycle phases and impact dimensions.

The role of the social dimension in particular is highlighted in the review by Padilla-Rivera et al. (2020), who identified 60 relevant studies and concluded that the social impacts of circular economy are receiving increasing attention in the scientific community, but that there is no agreed framework for assessing these social impacts. However, social LCA is identified as a promising methodology.

Two reviews are sector-based and deal specifically with electronic waste. One looks specifically at the potential of reuse as a circular economy strategy (Anandh et al. 2021), analysing 331 articles and identifying mobile phones and televisions as the most studied product groups. This review also notes that research on the environmental impacts and regulatory aspects of reuse is scarce, and identifies the most common tools and techniques used to assess reuse in waste electronic equipment as LCA for environmental impacts, MFA for understanding material flows, and game theory for analysing stakeholder interactions (e.g. consumer reactions). The second review looks in more detail at MFA (as well as substance flow analysis and product flow analysis), and assesses its role in waste electronic equipment management by reviewing 55 research articles (Islam and Huda 2019). It classifies the MFA approaches used in electronic waste research into static ("single-year evaluation of product and material flow with fixed product lifespan") and dynamic ("multi-year evaluation of product and material flow with variable product lifespan"). One finding for dynamic MFA is that Weibull distribution based product life modelling has become increasingly popular. Other approaches identified as promising are the cascade model proposed by Thiébaud et al. (2017) for performing dynamic MFA that provides information on reuse and storage stocks and flows, statistical entropy analysis when stock and flow information is available for different time periods (Laner et al. 2017), or a combination of MFA and LCA as used by Hischier et al. (2005) and Wäger et al. (2011). In the latter study, the authors compared the environmental impacts of scenarios involving incineration, disposal or recycling of waste electronic equipment in Switzerland, finding that the availability of data on the energy and resource requirements of the different end-of-life options proved to be a critical factor for the accuracy of the results.

Finally, a review examined the development of circularity metrics to measure the impact or benefits of circular economy (Corona et al. 2019). The review analysed 81 articles and identified seven measurement indices, nine assessment indicators and three assessment frameworks, but also found that "none of them are addressing the circular economy concept in full, potentially leading to undesirable burden shifting from reduced material consumption to increased environmental, economic or social impacts". Some of the findings of Corona et al. (2019) are summarised below.

The measurement indices represent the degree of circularity of a system (0 to 1), although the understanding of what is considered circularity varies. It was found that the focus is on the mass of materials as the basis for calculation, with few indices including economic value or duration in the equation.

To measure the value created by a circular system, there are circular economy assessment tools based on the underlying methodology of LCA, MFA or input-output analysis. In the field of MFA, authors have identified a framework for assessing the optimal level of circularity for different systems, called Complex Value Optimisation for Resource Recovery Evaluation (Lacovidou et al. 2017; Millward-Hopkins et al. 2018).

Overall, Corona et al. (2019) state that "MFA is more suited than LCA and IO [input-output] methods to examine different scenarios over long periods of time. However, it does not consider the environmental or social impacts associated with the system". Input-output analysis was found to be well suited to assessing circular economy strategies at large scale (e.g., whole economy), but not ideal for product-level assessments due to data granularity and the difficulty of integrating new technologies. Common difficulties in circular economy assessments using LCA were found, for example, in burden shifting for open-loop recycling scenarios. However, this may be less of a problem when assessing the optimal state of the whole system and assuming a closed-loop recycling scenario for the product group under consideration, as might be the case when measuring the market-wide impact of circular economy policies rather than analysing individual products.

Summarising the available insights from the six reviews, several conclusions can be drawn:

- There is no harmonised definition of the "circular economy" and, as a consequence, there is no standard or one-size-fits-all assessment approach.
- The geographical focus of circular economy research is Europe and China, both of which were among the first to introduce comprehensive circular economy regulations or initiatives with China's 2008 circular economy Promotion Law (Standing Committee of the National People's Congress 2008) and the EU's Circular Economy Action Plan from 2015 and 2020 (European Commission 2015, 2020e).
- Many circular economy studies focus on the biological cycle (especially in agriculture).
- The social dimension is less studied, but is growing and can be integrated through life-cycle thinking and social LCA.
- Research on methods to adequately assess circular economy is growing rapidly.
- Common circular economy assessment methods (especially for techno-economic assessments) are LCA, MFA and input-output analysis, each of which has its strengths and weaknesses and combinations of which represent a promising field for further research.

The reviews have also underlined that the contribution of circular economy to sustainability must always be taken into account and that circularity should not be seen as an end in itself. For example, if the overall lifetime of a product is measured rather than its useful life, the increased lifetime may not be related to the increased use of the material or product, and there may be no or little benefit in terms of reducing environmental impacts (e.g. discarded smartphone in a drawer) (Corona et al. 2019).

The reviews provided a good overview of the field of studies on circular economy. However, most did not show a clear link to the policy-making process, did not provide a framework directly applicable to the multidimensional process of impact modelling, or did not focus on products.

To expand the literature analysis, the available reviews were supplemented by a separate and systematic search to identify all potentially relevant aspects of the policy impact assessment process and to map the diversity of existing scientific approaches. To this end, the following question was asked: What are the existing approaches for assessing the impacts of circular economy in the context of product policy?

A search was conducted in Scopus using the "TITLE-ABS-KEY" search function, which filters articles by keyword coverage in title, abstract or keywords. The search returned 2 391 results (see Table 1), of which eleven were duplicates. Due to the large number of publications, an initial screening was carried out using the title alone. This revealed that many articles focused on the biological cycle (especially in agriculture) or the building sector, and that many described sector or technology-specific solutions or improvements at the product design stage, without direct reference to the role of policy. In addition, many articles were found to evaluate only regional policies or initiatives, or to focus mainly on ex-post evaluation of regulatory instruments.

Table 1 Scopus search results (search conducted on 4th February 2022).

Subject	Keyword search	Scopus results
Policy focus	"policy" OR "policies" OR "regulation*" OR "requirement*" OR "legislation*" OR "instrument" OR "governance" OR "ecodesign" OR "directive*"	6 659 652
Impact assessment	"impact*" OR "assessment*" OR "evaluation*"	10 551 875
Focus on circular economy	"circular*" OR "circular economy" OR "resource efficien*" OR "material efficien*" OR "repairability" OR "recyclability" OR "spare parts"	518 668
Product focus	"product*"	6 779 509
	Combined[6]:	**2 391**

108 articles were shortlisted for closer examination. With the aim of expanding the toolkit of potential methods, the articles were then scanned to identify their underlying approach. Key approaches were included in a mind map of methods for assessing the impact of circular economy policies (see Figure 4).

The results show a highly multidisciplinary range of research fields and methods. Research fields include social or behavioural sciences, political science, innovation studies, economics and environmental studies. No universal approach was identified to integrate circular economy in its full multi-dimensionality into impact assessment, but several articles develop approaches that combine different methods such as environmentally extended input-output analysis, material flow cost accounting, dynamic MFA and LCA or techno-economic assessments.

The approaches are mostly sector or product specific or focus only on specific aspects such as indirect impacts of regulations, e.g. potentially negative impacts such as rebound effects on consumers (Shinde et al. 2022) or positive effects on innovation (D'Amato et al. 2021). There are several approaches to consider supply risks or material shortages (Rachidi et al. 2021) or to integrate dissipative material flows (Charpentier Poncelet et al. 2021).

The methods used to assess these aspects also vary widely, including machine learning approaches (Shinde et al. 2022), non-linear optimisation (Thakker and Bakshi 2021), matrix-based multi-level assessments (Kerdlap et al. 2022), institutional theory (Sadri et al. 2022), system dynamics (Saidani et al. 2021) or statistical entropy analysis (Parchomenko et al. 2020).

[6] Search phrase: "policy" OR "policies" OR "regulation*" OR "requirement*" OR "legislation*" OR "instrument" OR "governance" OR "ecodesign" OR "directive*" AND "impact*" OR "assessment*" OR "evaluation*" AND "circular*" OR "circular economy" OR "resource efficien*" OR "material efficien*" OR "repairability" OR "recyclability" OR "spare parts" AND "product*"

The wide variety of methods and tools suggests that the available data, system boundaries, geographical context, existing regulatory framework and objectives of the study are important factors influencing the choice of modelling approach. Holistic assessments run the risk of oversimplification or over-generalising, and thus produce generic results of limited usefulness. Similarly, assessments of only one specific aspect can lose sight of the larger system interactions and lead to false conclusions.

The combination of two well-established methods, LCA and MFA, has emerged as a promising area of research. MFA allows the effects of policies to be modelled over long periods of time in a bottom-up approach, while LCA components can add the crucial environmental assessment to optimise not the circularity of the system but the actual environmental performance.

The review paper by Islam and Huda (2019) also identified a research gap for combining MFA and LCA, but also mentioned the existing work in this area by Hischier et al. (2005). Another example of combining MFA and LCA was identified during the Scopus search as the work of Boldoczki et al. (2021). Therefore, both publications are examined in more detail.

Hischier et al. (2005) use waste material flows (from the component level of electronic equipment to the level of individual materials after recycling) provided by the Swiss take-back and recycling system for waste electronic equipment and combine them with a simplified LCA (taking into account the collection process and the different treatment and recycling routes). The system boundaries are set from the time the waste electronic equipment is collected to the time the recycled materials leave the recycling facility. The analysis therefore only covers the end-of-life phase and is therefore rather narrow. The quality of the results is highly dependent on the input data (environmental impacts of the different processes). The level of detail has been improved in a 2011 update that also includes more recent LCA data (Wäger et al. 2011).

The more recent publication by Boldoczki et al. (2021) combined dynamic MFA and LCA and applied it to the reuse of washing machines in Germany. They used a dynamic stock model and integrated what they call a “modular LCA” to assess environmental impacts. They evaluated product flows and did not include individual material flows in their analysis. As a result, they could not assess in detail the macroeconomic implications of different policies (e.g. how much of the demand can be met by recycling material flows and what the environmental and economic implications are). Boldoczki et al. (2021) also pointed to a number of existing studies in the area of hybrid LCA and MFA, suggesting that there is a high degree of inertia in the field.

However, there does not seem to be a complete overview of existing approaches in the form of a systematic literature review of the combination of MFA and LCA and its applicability to product policy impact modelling. Existing studies are often not applicable to products (e.g. when focusing on total (and product-undifferentiated) waste streams (e.g., Ismail and Hanafiah (2021)) or, conversely, focus only on product streams and do not go a step further to integrate material streams (e.g. Boldoczki et al. (2021)). Many studies are theoretical in nature and cannot be applied in the context of policy impact assessment because they represent the status quo and do not include elements of foresight such as scenario analysis (e.g., Farjana et al. (2021)).

Thus, a research gap is identified in combining elements of MFA and LCA, tailoring them to the circular economy policy impact assessment process and applying them to products.

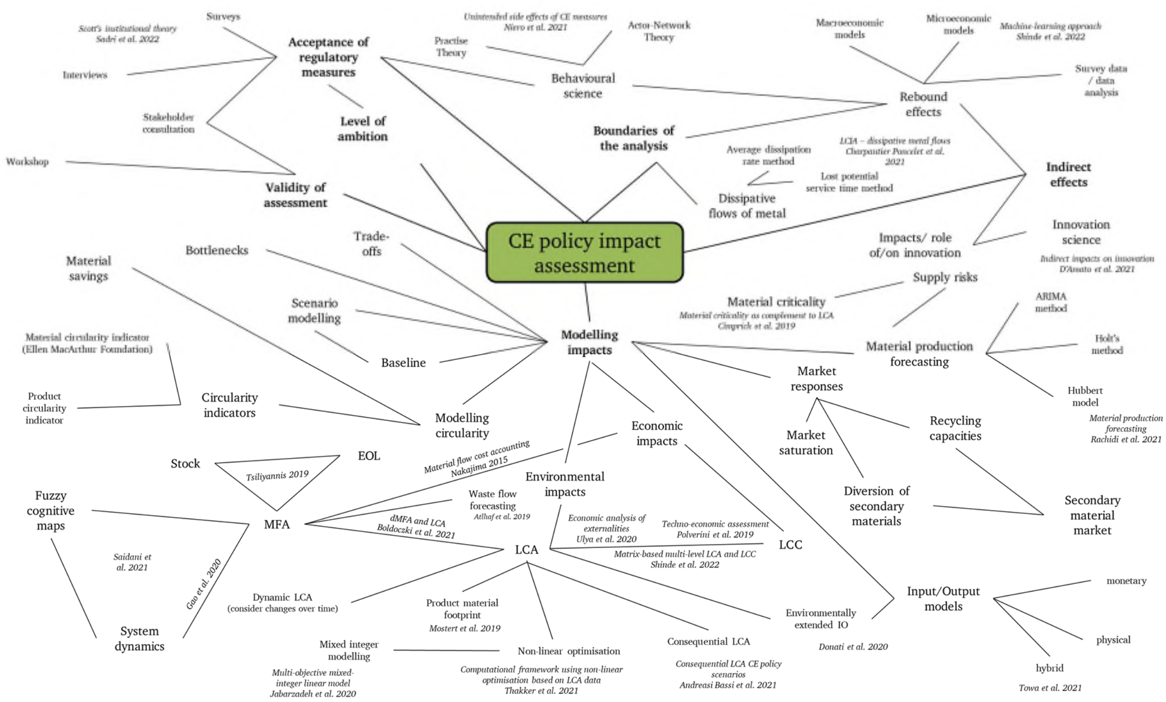


Figure 4 Mind map of circular economy (CE) policy impact assessment methods.

1.4. Research questions and thesis outline

This dissertation aims to improve the understanding of the intersection of the three distinct domains of circular economy, legislation and energy-related products. Within this intersection, the core methodological question deals with the integration of circular economy aspects into product policy impact assessments. An effective framework for assessing the environmental impacts of circular economy related policies and optimising the policy mix when designing or assessing product policies is currently lacking, which opens up a wide range of possible research questions.

The following paragraphs describe the core research questions (RQ) addressed in this dissertation.

The hypothesis is that the frequency, scope and depth of circular economy requirements have increased significantly over the last decade. While this seems evident in the case of EU battery legislation due to the numerous circular economy measures in the new battery regulation, the implementing measures of the Ecodesign Directive need to be assessed in more detail, which also allows for an analysis of the differences between product groups and types of requirements:

RQ1	**Has EU product legislation evolved towards a stronger circular economy focus and are there differences between product families and types of requirements?**

To improve policy impact assessment, it is important to understand the objectives that policy makers have pursued when developing circular economy requirements in recent legislation, such as that on batteries. In addition, a better understanding of the necessary framework conditions for the implementation of circular economy can be created:

RQ2	**What were the driving forces behind the integration of circular economy aspects in EU product legislation?**

Although circular economy is widely used as a buzzword, there is no common understanding of what a circular economy is, and many researchers avoid the conceptual ambiguity by talking about resource or material efficiency instead of circular economy. A clear categorisation of circular economy related requirements needs to be established as a basis for further research:

RQ3	**What type of circular economy requirements exist?**

With the increasing integration of circular economy requirements, it is important to have a robust methodology to assess the environmental impacts of circular economy requirements in the context of product policy impact assessments and to optimise the mix of circular economy requirements. In particular, the interdependencies and trade-offs between different circular economy requirements require an approach that goes beyond linear stock modelling. The initial literature analysis indicated that a combined MFA and LCA analysis may be well suited to model the impacts of circular economy related requirements. The central research question of this thesis can therefore be defined as follows:

RQ4	**How can material flow analysis be combined with life cycle assessment to conduct policy impact assessments for circular economy requirements?**

In the context of the previous question, conducting an impact assessment of different circular economy requirements for product-specific case studies can reveal strengths and weaknesses of the proposed approach, as well as empirical insights. It allows to assess the impacts of different circular economy requirements, such as lifetime extension or trade-offs between alternative or complementary requirements, such as recycled content, recycling efficiency and re-use. The analysis can also reveal differences between product groups, such as knowledge of the role of market maturity (e.g. mature markets such as industrial electric motors and growing markets such as traction batteries). The last question is therefore defined as:

RQ5	**What are impacts of and trade-offs between different circular economy requirements?**

To provide answers to these research questions, the dissertation is structured into seven chapters (see Figure 5), including this introductory chapter.

Chapter two sets the scene by analysing the uptake of circular economy requirements using the EU Ecodesign Directive as an example. The EU Ecodesign Directive was introduced as a framework to improve the environmental performance of energy-using and, later, energy-related products. From the outset, the Ecodesign Directive offered the possibility to consider not only the energy consumption of a product during its use phase, but also a wider range of environmental aspects throughout the life cycle of a product, including circular economy aspects. The chapter analyses the coverage of functional and informational circular economy requirements in the 27 product groups regulated by ecodesign implementing measures from 2008 to 2021 by conducting a content and keyword analysis of the legislative texts of 30 implementing measures and 16 amendments or repeals. It also proposes a taxonomy of circular economy related product requirements.

This taxonomy is extended in chapter three, which covers EU battery legislation from 1991 to 2022, analysing policy change in the subsystem and its underlying drivers. The advocacy coalition framework is used, with a mixed methods approach combining qualitative document analysis and interviews.

After this excursion into political science, chapter four builds on the preliminary literature analysis of the introduction (1.3) to conduct a systematic literature review of 44 prospective studies using the combination of MFA and LCA. The chapter examines the characteristics of existing combinations and their applicability to product policy impact assessment.

The systematic literature review forms the basis for the development of a new model in chapter five, which is suitable for the ex-ante assessment of the material flows and environmental impacts of circular economy product policies. The model uses a layered approach, where material and environmental impacts are derived from product flows via a product database that defines physical properties for product variants. The model is applied to a theoretical case study on electric motors, comparing motor repair with early replacement of inefficient motors (which may be seen as counterintuitive from a circular economy perspective) and with a base case to identify the strengths and weaknesses of the approach.

As a second case study besides industrial motors, chapter six applies the developed model to traction batteries. The sharp increase in electric vehicle sales observed today will lead to large quantities of lithium-ion batteries being retired in the coming decades. Against the background of the new battery regulation, with a stronger focus on circular economy related aspects, the model is used to analyse the material flows and environmental impacts of passenger car electric vehicle batteries in the EU-27 for the years 2011 to 2050. The analysis aims to provide insights into the impacts and trade-offs of novel policies, focusing on the impacts of a recycled content requirement and material-specific minimum recycling efficiencies. By assessing the market impacts, chapter six aims to contribute to the development of effective policies that promote energy efficiency and resource conservation in the context of battery management. The results of this chapter can inform decision-making processes and support the transition to a more sustainable electric vehicle industry.

Finally, chapter seven concludes this dissertation by providing an overview of the key findings and interdependencies, as well as a discussion of the results.

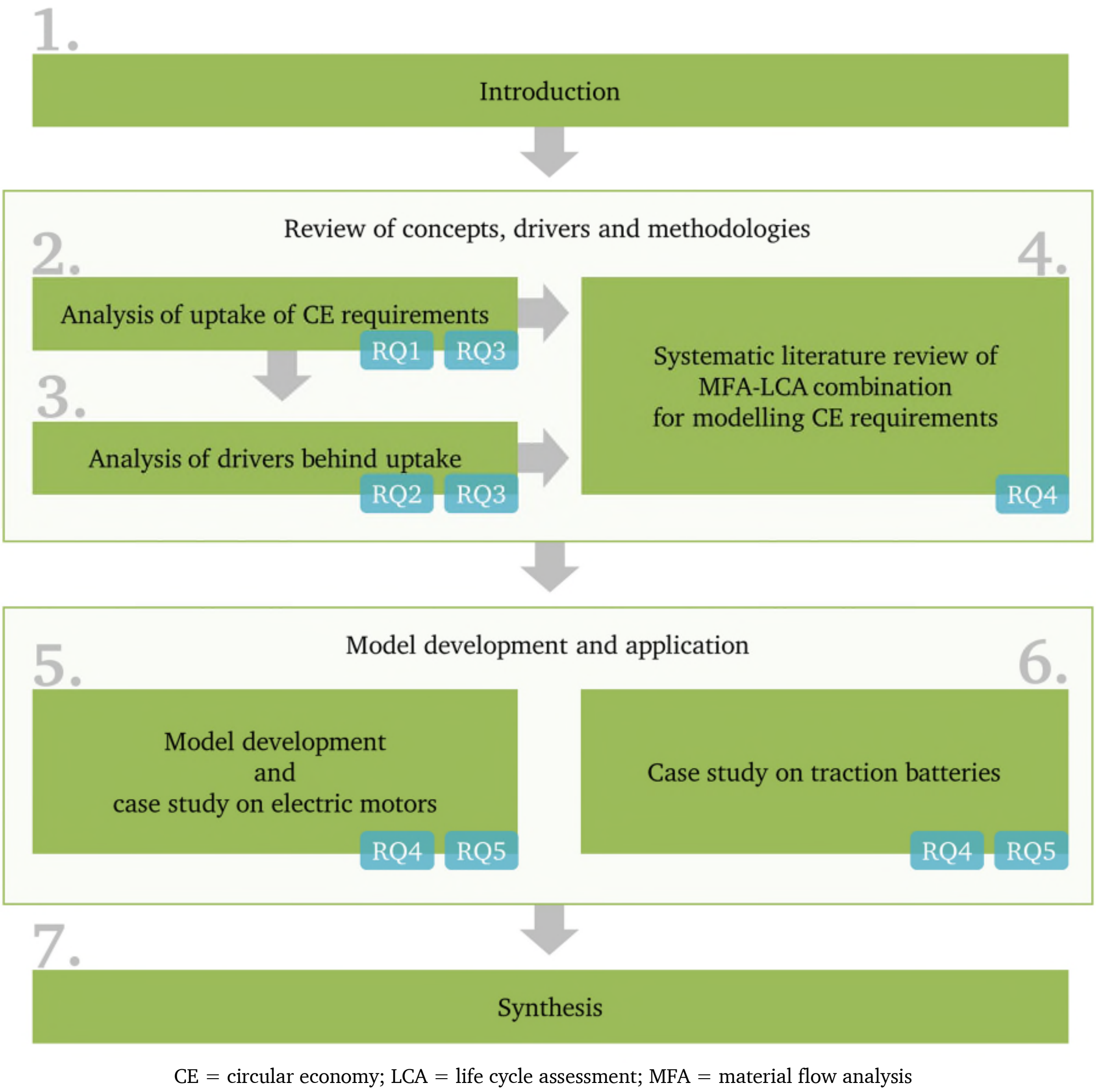


Figure 5 Dissertation outline. Chapter numbers in grey, research questions (RQ) in turquoise blue.

2. Review and analysis of Ecodesign Directive implementing measures: product regulations shifting from energy efficiency towards a circular economy

This chapter was published in August 2022 in the journal Sustainability. Table 2 provides more information about the publication.

Table 2 Publication information of Barkhausen et al. (2022).

Title	Review and Analysis of Ecodesign Directive Implementing Measures: Product Regulations Shifting from Energy Efficiency towards a Circular Economy
Authors	Robin Barkhausen, Antoine Durand, Katharina Fick
Publication date	19.08.2022
Journal	Sustainability (the research was first presented in June 2022 at the 11th International Conference on Energy Efficiency in Domestic Appliances and Lighting and could thereafter be submitted to a Special issue in Sustainability)
DOI	https://doi.org/10.3390/su141610318
Author contributions according to the Contributor Roles Taxonomy (CRediT 2023)	Robin Barkhausen: Conceptualization, Methodology, Investigation, Writing-Original Draft, Writing-Review & Editing, Visualization Antoine Durand: Conceptualization, Methodology, Writing-Original Draft, Writing-Review & Editing Katharina Fick: Investigation

2.1. Introduction

At the end of 2019, the European Commission announced the European Green Deal, an ambitious package of policy initiatives aimed at making the European Union (EU) carbon neutral by 2050 and decoupling economic growth from resource consumption (European Commission 2019c). As calculations for the EU's consumption footprint show, energy-using and energy-related products are responsible for large environmental impacts, especially in the impact category of mineral and metal resource use (Sala et al. 2019). In particular, raw material production is identified as a driver of the environmental impacts of household goods (Castellani et al. 2021).

To improve the environmental performance of appliances, the EU introduced product policies early on that focused on energy efficiency. The first date back to the 1970s oil crisis (Economidou et al. 2020), with requirements on the performance of space heating systems and the production of hot water in new or existing non–industrial buildings and on the insulation of heat and domestic hot-water distribution in new non-industrial buildings in Council Directive 1978/170/EEC (European Commission 1978). In 1997, the first labelling Directive (79/530/EEC) was published on the indication by labelling of the energy consumption of household appliances (European Commission 1979a). In the same year, Council Directive 79/531/EEC regulated electric ovens (European Commission 1979b). This was followed by Council Directive 1992/42/EEC of 21 May 1992 on efficiency requirements for new hot water boilers fired with liquid or gaseous fuels (European Commission 1992a). In the same year, an energy labelling framework was established with Council Directive 1992/75/EEC of 22 September 1992 on the indication by labelling and standard product information of the consumption of energy and other resources by household appliances (European Commission 1992b). The first directive on energy efficiency requirements for appliances was implemented in 1996 for household refrigerating appliances 96/57/EC (European Commission 1996) and, in 2000, for ballasts for fluorescent lighting 2000/55/EC (European Commission 2000). In 2005, the EU moved from drafting independent product-specific legislations to the Directive 2005/32/EC (Ecodesign Directive) (European Commission 2005), which established a framework for setting ecodesign requirements for energy-using products and amended Directives 92/42/EEC, 96/57/EC, and 2000/55/EC. The Ecodesign Directive was repealed 4 years later by Directive 2009/125/EC, which established a framework for setting ecodesign requirements for energy-related products in order to have a larger scope of application (European Commission 2009). Based on this directive and following MEErP, individual product groups are assessed in preparatory studies where potential ecodesign requirements are elaborated. These requirements can then be formulated in so-called implementing measures, EU regulations that set ecodesign requirements for products placed on the market. At present, 27 product groups are regulated by such implementing measures, covering a wide range of product groups, including lighting, white goods such as refrigerators and washing machines, information and communication technology (ICT) products such as computers, a variety of heating and cooling products, and industrial appliances such as welding equipment. In addition to the ecodesign implementing measures, the supplementing Directive 2010/30/EU (repealed by Regulation (EU) 2017/1369 (European Commission 2017)) establishes a framework for energy labelling energy-related products in order to pull the market towards more sustainable products (European Commission 2010).

It was estimated that EU ecodesign and labelling measures saved 10% of the EU27's primary energy consumption in 2020, and total financial savings in consumer spending were estimated at 60 billion euros (Wierda et al. 2021). In 2020, the average EU household used 70 regulated products (of which 30 were light sources and 25 electronic products) (Wierda et al. 2021).

Whereas both generations of the Ecodesign Directive (2005 and 2009) focused on energy efficiency requirements—with particular emphasis on reducing energy consumption during the use stage—they also allowed legislators to address many other aspects, all aimed at reducing the environmental impact of the products, including aspects related to the circular economy. Indeed, both directives emphasise in Article 15 on "implementing measures" that the preparation of an implementing measure must take into account "the life cycle of the product and all its significant environmental impacts". Furthermore, Annex I and II of the directives set out the method for setting generic and specific ecodesign requirements. From this perspective, the Ecodesign Directive takes a very holistic approach that goes far beyond energy efficiency alone.

Considering that it is not primarily technological but also cultural and market barriers that prevent industry actors from moving towards the circular economy, regulations can play a crucial role by pushing actors to overcome these barriers (Kirchherr et al. 2018). Design standards and norms are considered an essential tool to promote this development (Hartley et al. 2020). At the same time, the EU policy mix is still focused on waste management without promoting a circular economy holistically, and many product-related policies do not include resource efficiency aspects (Milios 2018).

However, EU policymakers have started to recognise this gap and the underlying potential, and in recent years there has been strong political advocacy for more consideration of circular economy aspects in product regulation. In its 2011 Roadmap to a Resource Efficient Europe, the EU proposed the goal of further improving the resource efficiency of products through the Ecodesign Directive (European Commission 2011). The first EU Action Plan for the Circular Economy (European Commission 2015) announced that future product requirements developed under the Ecodesign Directive should include circular economy-relevant requirements. Then, in 2015, the EC issued the groundbreaking standardisation mandate M/543 for material efficiency aspects within the Ecodesign Directive, with the aim of extending product lifetime, increasing component reuse, and recycling materials (Bertoldi 2021). This mandate led to a series of standards that were published between 2019 and 2020 Table 3.

Table 3 Material efficiency CEN-CENELEC standards developed under standardisation mandate M/543 (in addition to the listed standards, technical report CLC/TR 45550:2020 provides necessary definitions and CLC/TR 45551:2020 guidance on how to use the standards).

Reference	Title
EN 45552:2020	General method for the assessment of the durability of energy-related products
EN 45553:2020	General method for the assessment of the ability to remanufacture energy-related products
EN 45554:2020	General methods for the assessment of the ability to repair, reuse and upgrade energy-related products
EN 45555:2019	General methods for assessing the recyclability and recoverability of energy-related products
EN 45556:2019	General method for assessing the proportion of reused components in energy-related products
EN 45557:2020	General method for assessing the proportion of recycled material content in energy-related products
EN 45558:2019	General method to declare the use of critical raw materials in energy-related products
EN 45559:2019	Methods for providing information relating to material efficiency aspects of energy-related products

Whereas the standards were not published in time to be directly taken into account in the preparation of the implementing measures published in 2019, they do indicate a strong political will to increase the role of the circular economy within the Ecodesign Directive. The preparatory study and current proposal for the ecodesign implementing measure on mobile phones, cordless phones, and tablets proposed by the Commission in 2021 already covers many circular economy aspects and refers to the EN 4555X family of standards (Schischke et al. 2021). This trend is also underlined in the EU's latest Circular Economy Action Plan from 2020 (European Commission 2020e), which again refers directly to the Ecodesign Directive, but also announces the goal of going beyond energy-related products. This goal of expanding the product scope of the Ecodesign Directive is currently being developed in the Proposal for Ecodesign for Sustainable Products Regulation (European Commission 2022c).

2.2. Literature review

Several academic articles and policy evaluation studies have examined the apparent shift of focus in the Ecodesign Directive from energy efficiency to resource efficiency. The following paragraphs describe seminal work in the area, whereas the focus of this literature analysis is on whether the respective texts analyse the presence of circular economy requirements in published ecodesign regulations. For the most part, other aspects of the reviewed papers will not be discussed in detail.

Of the studies and papers assessed, several elaborated a methodology for effectively regulating circular economy aspects or identifying the potential or weaknesses of the current policy-making process (Ardente and Mathieux 2012; Mudgal et al. 2013; Bundgaard et al. 2017; Bracquené et al. 2018; Polverini and Miretti 2019; Schlegel and Akkerman 2019; Mathieux et al. 2020; Polverini 2021). A high potential for regulating circular economy aspects via ecodesign implementing measures is identified (Dalhammar 2014), but also the need to strengthen the role of resource efficiency aspects in preparatory studies (Bundgaard et al. 2017) and the lack of appropriate assessment methods and standards are highlighted, in particular, due to the product-specific character of circular economy aspects (Talens Peiró et al. 2020).

Stakeholder views on the role of the circular economy in ecodesign implementing measures vary, ranging from regarding it as a necessary development and a particularly positive attitude towards product durability and recycling requirements (Dalhammar 2016) to the view that the circular economy is a burdensome obstacle for original equipment manufacturers, e.g., due to costly monitoring (Egenhofer et al. 2018).

To test a methodology or elaborate a framework for a single product group, several studies have conducted case studies on the regulation of circular economy aspects under the Ecodesign Directive or other policy frameworks. The product groups studied include washing machines (Ardente and Mathieux 2012; Bracquené et al. 2018), LCD-TVs (Ardente and Mathieux 2012), electric motors (Dalhammar et al. 2014), vacuum cleaners (Bundgaard et al. 2017; Bracquené et al. 2018), and enterprise servers (Talens Peiró et al. 2020).

A number of studies define how to classify circular economy requirements, but with varying degrees of detail. Ardente and Mathieux (Ardente and Mathieux 2012) provide an extensive circular economy typology, which divides circular economy ecodesign requirements into a declaration of indices (such as recycling rates or recycled content), thresholds of those indices, design for recycling (e.g., reduction in contaminants), design for disassembly, availability of spare parts, warranty, indices for durability, dematerialization (e.g., lightweight design), declaration of substances, threshold of substances, marking/labelling/tracing, and provision of information. Several studies provide only a limited number of examples of possible types of circular economy requirements (Talens Peiró et al. 2020; Dalhammar 2016).

Bundgaard et al. (Bundgaard et al. 2017) define categories of resource efficiency aspects as reduction (e.g., use of resources during use), maintenance (e.g., maintenance instructions), reuse and redistribution (e.g., minimum lifespan), remanufacturing and refurbishment (e.g., easy to dismantle), and recyclability (e.g., information relevant for recycling). Polverini and Miretti (Polverini and Miretti 2019) propose a preliminary taxonomy of circular economy requirements as those aimed at increasing durability, repairability, and refurbishment capabilities, spare parts availability, recyclability via design for disassembly, information on material content and/or components marking, and promoting the reuse of secondary raw materials and/or components. Mathieux et al. (Mathieux et al. 2020) identify the criteria for resource efficiency assessments in a product policy context as reusability/recyclability/recoverability, recycled content, content of hazardous substances, and durability (related to reliability and ability to be repaired, upgraded, remanufactured). Polverini (Polverini 2021) adopts a preliminary taxonomy based on existing studies (Bundgaard et al. 2017; Polverini and Miretti 2019) and classifies circular economy aspects into the categories (1) durability, (2) repairability/refurbishment capacity/spare part availability, (3) recyclability,

(4) reusability of components, (5) consumables, and (6) circular economy requirements, differentiated into information and performance requirements.

Although many of the assessed studies provide examples of circular economy requirements in ecodesign implementing measures, only a few papers systematically investigate whether resource or material efficiency requirements are regulated in the existing regulations.

In a 2013 EU study on material efficiency under ecodesign and MEErP, the authors took a closer look at the coverage of material efficiency under the Ecodesign Directive in all already published implementing measures and also in the existing draft regulations at that time (Mudgal et al. 2013). They found a high number of material-related requirements, but these were almost exclusively information-based requirements. The most detailed study on circular economy coverage in implementing measures was published in 2017 by Bundgaard et al. (Bundgaard et al. 2017). In it, the authors evaluated a total of 23 adopted implementing measures and self-regulations developed under Article 17 of the Ecodesign Directive. They found 15 implementing measures that contained information requirements on resource efficiency, and only five that contained performance requirements. Based on the EU research programme REAPro, Mathieux et al. (Mathieux et al. 2020) briefly described the circular economy coverage in seven selected implementing measures from ecodesign regulations published in 2019. In 2021, Polverini (Polverini 2021) also focused on the 2019-generation of implementing measures but evaluated nine product groups and identified differences between the product families of business-to-consumer products (e.g., dishwasher), and business-to-business products (e.g., welding equipment), with clear commonalities in requirements identified for the former.

The following table Table 4 gives an overview of the different regulated product groups that have been assessed in the literature with regard to the coverage of circular economy aspects.

It should be noted that many of these studies do not assess systematically the shift towards more circular economy requirements within ecodesign, but rather how circular economy aspects can be integrated from a methodological point of view for specific product groups.

Our work differs from previous research in several aspects. First, our analysis extends the scope of previous work by systematically including all adopted ecodesign implementing measures, amendments, and repeals between 2008 and 2021 and showing what changes can be detected over time (excluding self-regulations developed under Article 17 of the Ecodesign Directive and energy labelling regulations). Second, the depth of analysis is increased by elaborating a clear classification of types and categories of circular economy requirements and delivering more nuanced results. Third, the differences between product families and generations of implementing measures are assessed and patterns are revealed.

By holistically assessing which circular economy requirements have been considered in ecodesign implementing measures and by analysing the differences between single product groups and product families (see 2.3.1 for definition), we aim to improve the understanding of the current state of circular economy in EU product policy and provide guidance for future product legislation. By summarising the coverage of circular economy measures over more than 20 years of the EU Ecodesign Directive, we seek to synthesise lessons learned and provide an outlook as well as recommendations for future development of product regulations. To reduce conceptual ambiguity, we propose a circular economy taxonomy tailored to ecodesign implementing measures.

The following research questions are explored: What are the links between different ecodesign implementing measures and how have they evolved? Has there been a quantifiable increase in circular economy requirements in ecodesign implementing measures over the years? What types of circular economy requirements are the most prevalent in implementing measures? Which product group regulations impose the most and the most stringent circular economy requirements on original equipment manufactuers? Are there differences in the type and quantity of circular economy requirements between product families?

Directed content analysis and quantitative keyword analysis of the legal texts are the methods used to answer these questions. The materials and methods section explains the methods used, defines a circular economy requirement, and describes the conceptual distinction between different types of circular economy requirements. It also explains the procedure used to conduct the content analysis and keyword analysis. The results section presents the findings of the analysis and the discussion section discusses the results, outlines the research limitations, identifies areas for further research, and draws conclusions.

Table 4 Scope of comparative studies. “x” means implementing measure was assessed towards circular economy, “/” means only examples of circular economy aspects within the product regulation are provided, and “c” means there is no systematic assessment, but a case study. Green cells indicate that a circular economy requirement was detected.

Product Groups Regulated via Ecodesign Implementing Measures / Source	(Ardente and Mathieux 2012)	(Mudgal et al. 2013)	(Dalhammar et al. 2014)	(Dalhammar 2014)	(Dalhammar 2016)	(Bundgaard et al. 2017)	(Egenhofer et al. 2018)	(Bracquené et al. 2018)	(Polverini and Miretti 2019)	(Schlegel and Akkerman 2019)	(Talens Peiró et al. 2020)	(Mathieux et al. 2020)	(Kishita et al. 2021)	(Polverini 2021)
Standby and Off Mode Electric Power Consumption[...]		x				x								
Simple Set-Top Boxes		x				x								
Non-directional Household Lamps		x		/	/	x	/				/		/	
Fluorescent Lamps [...]		x				x	/				/		/	
External Power Supplies [...]		x				x								
Household Refrigerating Appliances		x				x						x		x
Electronic Displays and Televisions	c	x			/	x						x		x
Circulators [...]		x				x								
Electric Motors		x	c			x								
Household Dishwashers		x		/		x						x		x
Household Washing Machines and Washer-Dryers	c	x				x		c	/			x		x
Fans Driven by Motors [...]		x				x								
Air Conditioners		x				x								
Water Pumps		x				x								
Household Tumble Driers		x				x								
Directional Lamps, Light Emitting Diode Lamps [...]		x			/	x	/				/			
Computers and Computer Servers		x		/		x			/	/	xc	x		x
Vacuum Cleaners		x			/	xc	/	c	/		/		/	
Space Heaters [...]		x				x								
Water Heaters [...]		x				x								
Domestic Ovens and Range Hoods						x								
Small, Medium, and Large Power Transformers														x
Residential Ventilation Units														
Solid Fuel Local Space Heaters														
Local Space Heaters														
Solid Fuel Boilers [...]														
Professional Refrigerated Storage Cabinets														
Air Heating Products, Cooling Products [...]														
Welding Equipment												x		x
Refrigerating Appliances with a Direct Sales Function												x		x
Light Sources and Separate Control Gear														x

* Analysis of reparability criteria, without a focus on the Ecodesign Directive.

2.3. Materials and methods

This section starts by defining the circular economy and the different types and categories of circular economy requirements and then describes the method for conducting the content and quantitative keyword analysis based on these definitions.

2.3.1. Definition of circular economy in the context of the Ecodesign Directive

As described in the Ecodesign Directive "an ecodesign requirement means any requirement in relation to a product, or the design of a product, intended to improve its environmental performance, or any requirement for the supply of information with regard to the environmental aspects of a product" (European Commission 2009).

To assess whether there has been a shift towards circular economy requirements under the Ecodesign Directive, the concept of a circular economy needs to be defined. There is no harmonised understanding of what a circular economy is (Kirchherr et al. 2017), and many scholars try to circumvent this conceptual ambiguity by discussing material or resource efficiency instead. Material efficiency can be described as "the ratio between the performance output of a product, service or energy system and the input of materials required to provide such output" (Cordella et al. 2019). The circular economy, however, describes a concept that has material efficiency at its core, but goes beyond this and aims to establish a more holistic and systemic economic system based on sustainable development.

Conceptually, a circular economy can be distinguished from a linear economy, a view of the economy that seems to represent reality, but only if the environmental dimension is ignored (Pearce and Turner 1990). However, the environmental dimension provides inputs (or resources) to the economy and functions as the repository of waste products (e.g., car-bon dioxide emissions or solid waste in landfills). According to the first law of thermodynamics, there must be a balance between the energy and matter supplied to the economy (from the environment) and the outputs returned to the environmental system (Pearce and Turner 1990). Furthermore, according to the second law of thermodynamics, certain natural resources are irreversibly damaged (or degraded) and thus lost when used for economic activities (Georgescu-Roegen 1971). The environment has limited assimilative capacities to deal with this economic waste and therefore a circular economy aims to reduce the pressure on the environment to act as a sink for waste, either by reducing the amount of waste produced or by converting waste back into resources (recycling) (Pearce and Turner 1990). In line with this theoretical concept, the EC defines a circular economy as an economy, "where the value of products, materials and resources is maintained in the economy for as long as possible, and the generation of waste is minimised" (European Commission 2015). The benefits that such an economic system brings are potentially linked to all dimensions of sustainable development, with a particular focus on the environmental and economic dimensions (Kirchherr et al. 2017).

There are different frameworks defining how to achieve a circular economy. According to Article 2 of the Chinese Circular Economy Promotion Law (Standing Committee of the National People's Congress 2008), "circular economy" is defined as a term for reducing, reusing, and recycling activities (3Rs). In the EU, Article 4 of the EU Waste Framework Directive defines a five-level waste hierarchy, from higher to lower priority with (1) prevention, (2) preparing for reuse, (3) recycling, (4) other recovery (e.g., energy recovery), to (5) disposal (4Rs) (European Commission 2008). The model of circular economy strategies can be extended to 9Rs, namely refuse (R0), rethink (R1), reduce (R2), reuse (R3), repair (R4), refurbish (R5), remanufacture (R6), repurpose (R7), recycle (R8), recover (R9) (Potting et al. 2017).

In our study, we adopt the EU definition of a circular economy with the aim of reducing waste and prolonging the effective use of materials and resources in order to contribute to sustainable development (European Commission 2015). In our taxonomy of circular economy requirements, we start from a 3R framework: reduce, reuse, and recycle (Table 5). Based on these circular economy strategies, circular economy categories are defined to classify the circular economy requirements found in ecodesign implementing measures.

The categorisation of circular economy requirements results from the literature review, the Circular Economy Action Plan (European Commission 2020e), the EN Standards on material efficiency, and a preliminary screening of ecodesign implementing measures.

Table 5 Circular economy taxonomy adopted for ecodesign implementing measures (based on the 3R framework).

Circular Economy Strategy	Circular Economy Category	Circular Economy Requirements
Reduce	Durability	• Minimum lifetime/warranty • Availability of updates • Installation/maintenance instructions
Reuse	Reusability/ Repairability	• Repair/disassembly instructions • Information/warning on non-repairability • Spare parts • Design for disassembly or repair • Secure data deletion
Recycle	Recycling Critical sourcing Hazardous substances	• Disassembly/dismantling instructions • Design for dismantling, recycling and recovery • Marking of components • Information on how to dispose • Information on critical raw material content • Information on hazardous material content • Ban of materials

In the Circular Economy Action Plan, the EU sets itself the goal of regulating (among other aspects) the durability, reusability, upgradability, repairability, the presence of hazardous chemicals, remanufacturing, and recycling of products. This corresponds to the circular economy categories found in the literature. Circular economy requirements can be classified into measures that increase the effective lifetime, either by improving the durability of the product or components (reduce) (Ardente and Mathieux 2012; Mathieux et al. 2020) or by increasing reusability/repairability and thus postponing disposal due to malfunction or technical failure (reuse) (Ardente and Mathieux 2012; Bundgaard et al. 2017; Mathieux et al. 2020). Once products have reached their end-of-life, recycling recovers materials and/or components and uses them as input for the production of new products (Ardente and Mathieux 2012; Bundgaard et al. 2017). Accurate information about the presence of hazardous substances and also critical raw materials (critical sourcing) is essential for recyclers to develop efficient and safe processes (Ardente and Mathieux 2012; Bundgaard et al. 2017).

There are some additional circular economy categories and types of requirements, but the first screening of ecodesign implementing measures revealed that they are not used in current ecodesign regulations. These include mandatory recycled contents, which is becoming increasingly important, as shown by its inclusion in the proposal for a Batteries Regulation in 2020 to replace the Directive 2006/66/EC on batteries (European Commission 2020c) (the legislation on batteries is not regulated under the ecodesign framework, even though the preparatory study for this regulation was carried out in line with the MEErP (van Tichelen et al. 2019)). The proposal also explicitly refers to the repurposing of batteries for a second life and introduces the so-called battery passport, a digital product passport that could improve the traceability of batteries and thus increase recycling inflows. Relevant life cycle data such as charging cycles to facilitate repurposing can also be recorded in the digital product passport. The Batteries Directive from 2006 already included minimum collection and recycling rates to increase the number of recycled materials (European Commission 2006). Other new approaches can be found in the EU's Circular Economy Action Plan, which explicitly refers to measures against premature obsolescence, a ban on the destruction of unsold but durable goods, and the concept of "product-as-a-service" (European Commission 2020e). In the future, some of these aspects may find their way into ecodesign implementing measures, so it may be necessary to include them in further analysis and to integrate other new circular economy approaches into the circular economy taxonomy for ecodesign implementing measures.

When assigning circular economy requirement types to the defined categories, the definitions of the EN standards on material efficiency are taken as a starting point. Therein,

- Durability is defined as the ability to function as required, under defined conditions of use, maintenance and repair, until a limiting state is reached. The degree to which maintenance and repair are within the scope of durability varies between products
- Repair is defined as the process of returning the product to serviceability
- Reuse is defined as any operation by which products or components are used again for the same or another purpose for which they were conceived
- Remanufacturing is defined as an industrial process that produces a product from used products or used parts where at least one change is made that influences the safety, original performance, purpose, or type of the product
- Recycling is defined as a recovery operation of any kind by which waste materials are reprocessed into products, materials, or substances whether for the original or other purposes excluding energy recovery
- Hazardous substance is defined as a substance that has, according to defined classification criteria, the potential for adversely impacting human health and/or the environment (not defined under standardisation mandate M/543 but in IEC Guide 109:2012 (IEC Guide 109:2012))
- A critical raw material is defined as a material that, according to a defined classification methodology, is crucial due to its economic importance and its supply risk.

In our definition, durability is defined as the useful life of the product without including repair measures, which are considered a separate category. Thus, requirements that are classified under durability include, for example, premature failure rate, survival factor, rated lifetime, minimum number of switching cycles for lamps, the minimum number of loading cycles for batteries, the minimum oscillations of the hose of vacuum cleaners, as well as mandatory warranty and updates on firmware, software, or safety. As maintenance is usually regulated together with measures related to repair—in the following we will use the term repairability—(e.g., "access to repair and maintenance information" in (EU) 2019/2019 on household refrigerating appliances), both are classified under repairability. Reusability is grouped together with repairability as both extend the time the product is used for the purpose for which it was designed. Requirements on reusability/repairability include, e.g., information on non-destructive disassembly, secure data deletion, warning about non-repairability, or requirements related to spare parts, such as availability, order procedure, maximum delivery time, or the replaceability of parts with commonly available tools. Requirements that promote recycling include, for example, design for recycling, marking of specific material fractions (e.g., plastic components), and information about how to return or dispose of the product or design for recycling. The requirements on design for recycling are normally combined with others as design for dismantling, recycling, and recovery, and thus also apply to the circular economy category repairability. In our analysis, we include requirements with reference to dismantling in the category of recycling to avoid double counting (exemption: non-destructive disassembly for maintenance purposes).

Information regarding hazardous materials as well as critical raw material content support the recycling process by indicating necessary safety measures or the presence of high-value materials. Requirements regarding hazardous materials and critical raw material include, for example, the provision of information on hazardous material content such as mercury, lead, cadmium or refrigerant gas, the ban of materials such as halogenated flame retardants, and the provision of information on critical raw material content such as cobalt. Figure 6 gives an overview of the product value chain stakeholders in the linear or circular economy and the material flows that are influenced by the different circular economy categories.

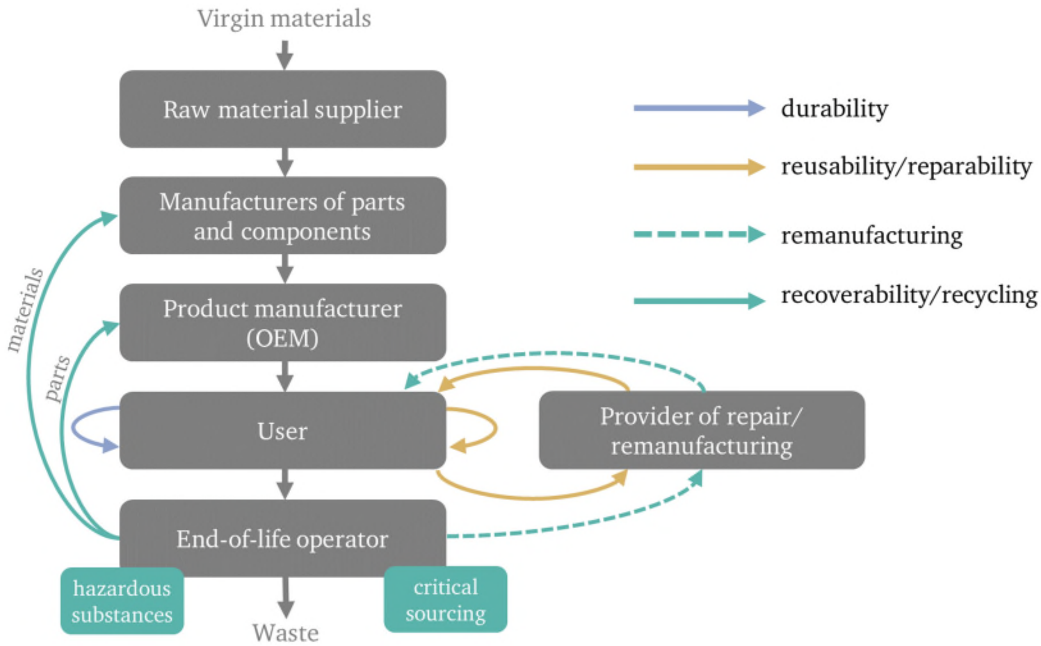


Figure 6 Stakeholders, circular economy categories, and material flows along value chain. Adapted with permission from Polverini (2021). Copyright 2021 Elsevier.

2.3.2. Type and scope of text analysis

Our text analysis covers the 27 product groups regulated by ecodesign implementing measures from 2008 until 2021, including the legislative texts of 30 original implementing measures and 16 amendments or repeals. Three product groups (space heaters, electrical lamps, and luminaires) have subgroups that are regulated independently with separate implementing measures. Therefore, we considered them individually in our analysis, resulting in 30 product (sub)groups. On 1 October 2019, three separate implementing measures on different lighting sources were repealed by one implementing measure that covers all previous lighting sources. Thus, from 2019, we consider only 28 regulated product (sub)groups.

Voluntary agreements and energy labelling regulations were not part of the analysis.

First, a directed content analysis was carried out. The rationale for choosing this method is the different style of the regulatory texts, with changing sections and structures which require manual analysis. In addition, circular economy requirements are very product-specific, as also noted by Talens Peiró et al. (2020), who pointed out that the "durability of lamps can be measured by lumen maintenance factor, whereas the durability of vacuum cleaners is assessed according to the fatigue life testing of motor and hose." This individuality requires manual identification of requirements.

Changes to existing implementing measures take the form of amendments and repeals. Amendments are considered in the analysis when they introduce new circular economy requirements or when existent circular economy requirements are replaced or amended. Due to the nature of EU legislation, single amendments often affect several implementing measures at once, with a distinction between the individual implementing measures in the text. Repeals can also replace several implementing measures at once, as illustrated by the example of (EU) 2019/2020 on lamps, which replaced (EC) No. 244/2009, (EC) No. 245/2009, and (EU) No. 1194/2012 (all of which regulated different types of lamps). In order to provide a general overview of the development of ecodesign framework, each initial implementing measure, amendment, and repeal was marked with its official publication date. In contrast, each individual circular economy requirement was documented with its specific date of entry into force.

To classify circular economy requirements, the Ecodesign Directive distinguishes between specific and generic ecodesign requirements (Article 2). A specific ecodesign requirement is defined as a "quantified and measurable ecodesign requirement relating to a particular environmental aspect of a product, such as energy consumption during use, calculated for a given unit of output performance". A generic ecodesign requirement is defined as "any ecodesign requirement based on the ecological profile as a whole of a product without set limit values for particular environmental aspects". In our analysis, we additionally classified generic requirements as hard or soft, depending on the level of ambition and detail. In addition, we distinguished functional (generic and specific) from informational requirements, which we subclassified as hard, medium, or soft (Table 6). Hard informational requirements demand the provision of specific performance values for the product, whereas medium informational requirements demand specific actions from manufacturers, e.g., providing a list of spare parts on the company's website. Finally, soft informational requirements are non-specific and allow original equipment manufactuers greater leeway in meeting the requirements. An example of this is Regulation (EC) No 640/2009 on electric motors, which includes the very unspecific information requirement of providing "information relevant for disassembly, recycling or disposal at end-of-life" without further specifications. Changes to existing requirements through amendments that increase the level of ambition but do not change the type of requirement (specific functional, generic functional, informational) or the year of entry into force are not considered new requirements, as quantifying the "strength" of a circular economy requirement beyond our proposed classification is likely subjective.

Table 6 Requirement classification and examples.

	(Functional) Specific	(Functional) Generic		Informational		
		Hard	Soft	Hard	Medium	Soft
Explanation	Set limit values	No set limits, but specific	No set limits and unspecific	Specific numbers	Specific instructions	Unspecific
Example	Rated lamp lifetime ≥ 1 000 h	Availability of spare parts	“Components can be dismantled” without further details	Rated lifetime or material content; also plastic or cadmium marking	Dismantling instructions	“Information relevant for disassembly, recycling or disposal at end-of-life”

Deciding what counts as a single requirement can be challenging, due to the aforementioned individuality of the legal texts and the specificity of product groups. When estimating the number of requirements per implementing measure, sub-items were grouped together as one requirement if they specify the same requirement for different operating states or product variations. For example, in Regulation (EC) No. 244/2009 on non-directional household lamps, the requirement in Annex II on the minimum number of switching cycles was counted as one requirement even though it sets two different conditions for different lamp types.

For some implementing measures, different implementation levels exist for the same requirement. In this case, each new implementation level (if the date of entry into force differs) was considered a replacement of the original requirement. For example, Regulation (EC) No. 244/2009 on non-directional household lamps introduced minimum lamp functionality requirements in 2009, but the requirements were increased in 2013. Therefore, when creating our database of circular economy requirements in ecodesign legislations, there are two entries, one for the first requirement, and (at the time of the amendment) one entry for the inclusion of the updated requirement which replaces the original requirement.

Within the implementing measures, there can be exemptions from the requirements for certain product types or variants (for example, in (EC) No. 245/2009 on fluorescent lamps in Annex I: “The following lamps shall be exempted from the provisions of this Regulation: [...]”). These exemptions from the requirements for certain product types were not considered in the analysis.

Each requirement was assessed to see if it addresses circular economy aspects and then assigned to a circular economy category (see Table 5); allocation was not always straightforward. Two examples illustrate this process. The first one is Regulation (EC) No. 245/2009 on fluorescent lamps, where some requirements are clearly related to the circular economy, such as the lamp survival factor, which is defined as “the fraction of the total number of lamps which continue to operate at a given time under defined conditions and switching frequency” and related to durability. However, this is less clear for the lamp lumen maintenance factor. If we consider the functional unit of the lamp to be the amount of light (lumens) provided and assume that consumers will replace the lamp if the lumens emitted decrease over time, then this requirement does extend the lifetime of the product and accordingly, we consider it as a circular economy requirement. The second example is Regulation (EU) 2019/2020 on light sources, which states that “the energy consumption of the product and any of the other declared parameters shall not deteriorate after a software or firmware update”. The so-called “other declared parameters” technically include the parameter lifetime, but as the requirement relates mainly to energy consumption, it is not considered a circular economy requirement. Circular economy aspects that are mentioned in the "subject matter and scope" section of regulations were not considered in the content analysis. For example, Article 1 part 3 in Regulation (EU) 2019/1783 on transformers, which states that transformers with a replaced core or windings must be reassessed for conformity with the regulation, was not considered. However, it should be noted that the legal status of refurbished products is a necessity for a complete circular economy framework.

To differentiate between similar product groups, we defined the product families lighting, white goods, heaters/coolers, ICT, and other as follows:

- Lighting combines the regulation of non-directional household lamps, fluorescent lamps, directional lamps, and the combined regulation of light sources and separate control gears.
- White goods include the regulations on household refrigerating appliances, household dishwashers, household washing machines and washer-dryers, household tumble driers, domestic ovens and range hoods, professional refrigerated storage cabinets, and refrigerating appliances with a direct sales function.
- Heaters/coolers include the regulations on fans driven by motors, air conditioners, space heaters, water heaters, ventilation units, solid fuel local space heaters, local space heaters, solid fuel boilers, and air heating products.
- ICT includes the regulations on simple set-top boxes, external power supplies, electronic displays and televisions, and computers and computer servers.
- All other product groups are classified as "other".

Due to the chosen approach in the content analysis of defining what counts as an individual requirement on a qualitative case-by-case basis, the number of requirements identified should be treated with caution. Nevertheless, on an aggregated level, the results still reveal the main trends among ecodesign implementing measures. To reduce the ambiguity of the directed content analysis and create a more quantitative layer of analysis, a keyword analysis was also carried out. Based on an initial screening, appropriate keywords were defined and assigned to each circular economy category (according to our previously defined circular economy taxonomy) and the number of mentions of these keywords was counted. If necessary, the words were reduced to their word stem, such as "dismantl" for both "dismantle" and "dismantled". The table with the results of the keyword analysis can be found online in the Supplementary Information (Table S1, accessible online, see 2.7).

The word "replac*" was excluded from the analysis because, beyond its circular economy-related meaning, it is often used in amendments to refer to the replacement of articles from the original implementing measure. Although the keyword analysis provided more quantitative results, it has several limitations. Amendments and repeals often refer to several regulations and may combine amendments or repeals from different product groups. Therefore, differentiation between product groups and product families is not possible, and only the overall development across all ecodesign regulations can be presented.

The following chapter presents the results of the content and keyword analysis.

2.4. Results

In this section, we describe the chronological development of the implementing measures with their respective amendments and repeals and in more detail, the development of the types and categories of circular economy requirements based on the content analysis. Finally, we present the results of the keyword analysis.

2.4.1. Evolution of ecodesign regulations and their circular economy coverage

The first ecodesign implementing measure was published on 17 December 2009 for standby and off-mode electric power consumption, followed by eight other product groups (and several amendments) all based on the 2005 Ecodesign Directive on energy-using products. The first regulation referring to the 2009 Ecodesign Directive was published on 10 November 2010 on household dishwashers. New regulations were published frequently from 2008 to 2016, whereas no implementing measure, amendment, or repeal was added between 2016 and 2019, before a total of 13 regulations were published in 2019 (thereafter referred to as the "2019-generation"), 12 of which were published on 1 October 2019 (on 15 March 2019: computers and computer servers; on 1 October 2019: standby and off-mode electric power consumption, external power supplies, household refrigerating appliances, electronic displays and televisions, circulators, electric motors, household dishwashers, household washing machines, small, medium, and large power transformers, welding equipment, refrigerating appliances with a direct sales function, light sources, and separate control gears). The latest ecodesign regulation was published on 23 February 2021, amending several of the regulations from the 2019-generation. Figure 7 shows the chronological evolution of ecodesign product regulations. As can be seen in the figure, lighting regulations were repealed in 2019 by a single regulation covering all lighting products. Note that several of the amendments and repeals address more than one regulation. A table listing the assessed legislations is included in the supplementary information (accessible online, see 2.7). One product group that stands out is standby, off-mode electric power consumption of electrical and electronic household and office equipment, which can be interpreted as a horizontal regulation covering many product groups (compared to a vertical regulation covering one product group). This overarching character can explain the high number of amendments seen in Figure 7, as the regulation has been amended each time that products falling within its scope were regulated with their own vertical regulations (and exempted from the horizontal regulation).

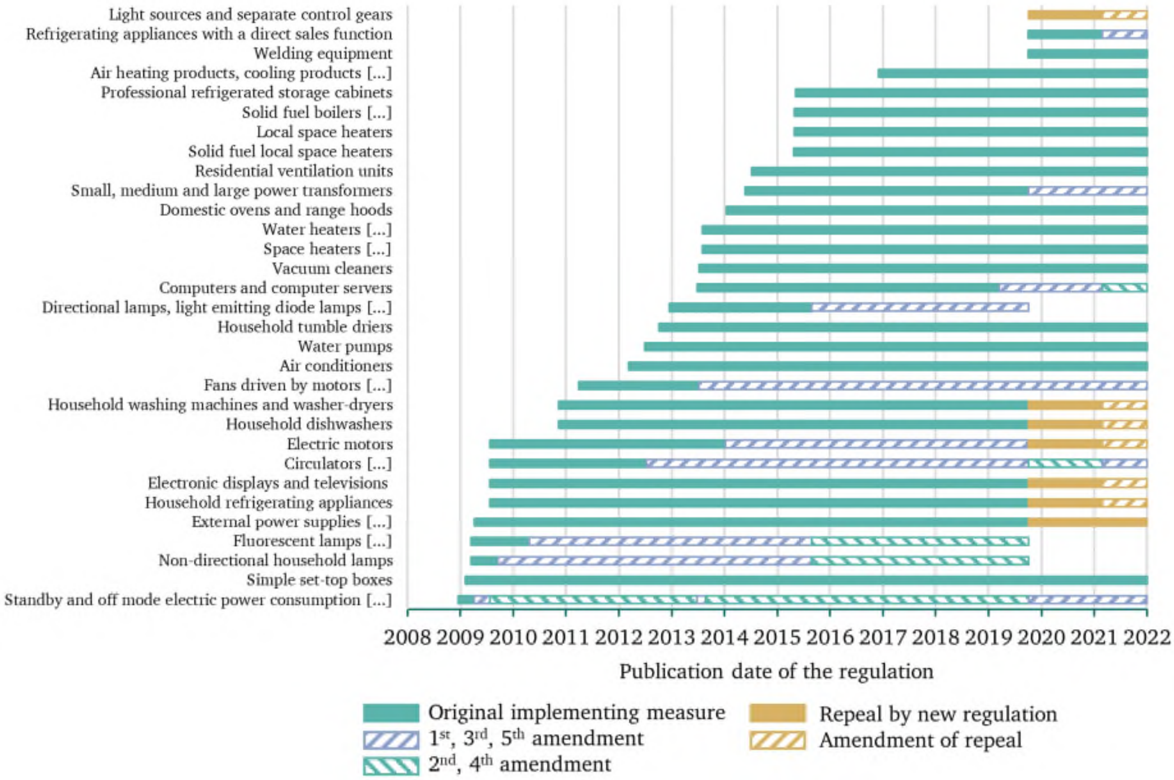


Figure 7 Evolution of ecodesign regulations.

Figure 8 shows the regulations identified by the content analysis that include circular economy aspects according to our circular economy taxonomy. As can be seen in the figure, no circular economy requirements could be detected for only six product groups, namely standby power consumption (Regulation (EC) No. 1275/2008 and amendments), simple set-top boxes (Regulation (EC) No. 107/2009), external power supplies (Regulation (EC) No. 278/2009), air conditioners (Regulation (EU) No. 206/2012), household tumble driers (Regulation (EU) No. 932/2012), and power transformers (Regulation (EU) No. 548/2014 and amendment). Today, 21 out of 28 regulated product groups (75%) have active circular economy requirements.

Comparing our results to the literature, the findings are largely consistent with the assessments of (Mudgal et al. 2013; Bundgaard et al. 2017; Mathieux et al. 2020; Polverini 2021). Differences were found in three cases. Bundgaard et al. (Bundgaard et al. 2017) identified informational resource efficiency requirements in the 2010 regulation on domestic dishwashers and both specific and informational requirements in the 2010 regulation on household washing machines. The corresponding requirements are not specified, but most likely relate to water consumption during the use phase, which is specified in both regulations. In our analysis, the special case of water consumption during the use phase is not included in the circular economy taxonomy, and thus circular economy requirements are only identified in the 2019-generation of household dishwashers and washing machines (e.g., availability of spare parts). Similarly, Polverini (Polverini 2021) identified one circular economy requirement in Regulation (EU) 2019/1783 on power transformers, which explicitly mentions transformers with "replaced core and one or more of the complete windings replaced". However, as this definition is only mentioned in the "subject matter and scope" section of the regulation, which we did not assess, it was not included in our analysis.

It should be noted that Figure 8 does not indicate the extent to which circular economy requirements are integrated into the regulation, and several product groups have only a very low circular economy coverage. For example, Regulation (EC) No. 640/2009 on electric motors only contains one non-specific informational requirement in Annex 2.2, stating that product information “relevant for disassembly, recycling or disposal at end-of-life” must be provided.

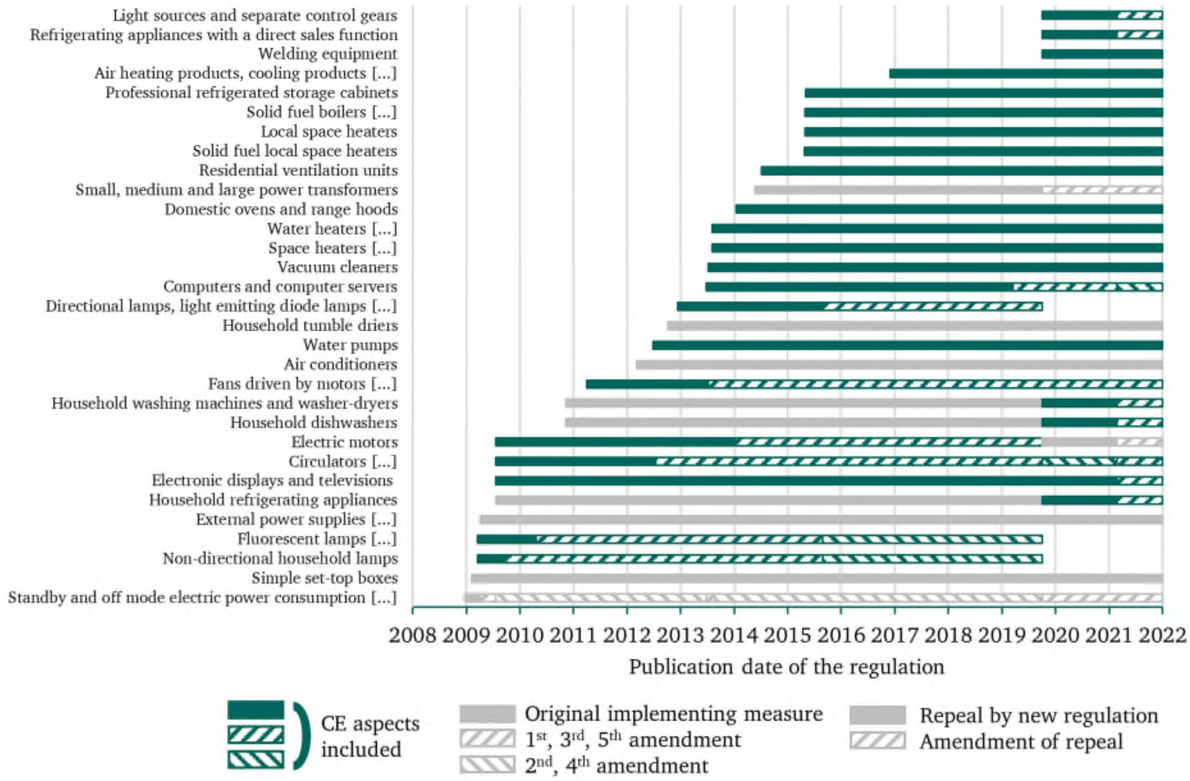


Figure 8 Circular economy coverage of ecodesign regulations.

2.4.2. Types of circular economy requirements

When assessing in more detail which kind of circular economy requirements are set in the regulations, Figure 9 provides further insights. Here, it should be noted that the number of circular economy requirements (left axis) refers to the date of entry into force of the individual requirement, whereas the number of regulated product groups (right axis) refers to the publication date of the legal document. The chronological evolution shows a sharp increase in circular economy requirements caused by the three lighting regulations published in March 2009 and December 2012, and then only a moderate increase in mainly informational requirements (correlated with the increase in the number of regulated product groups) until the 2019-generation, which caused a second significant increase in specific and informational circular economy requirements and also set the first generic requirements. A line chart that was normalised to the number of regulated product groups can be found online in the supplementary in-formation (Figure S1, see 2.7).

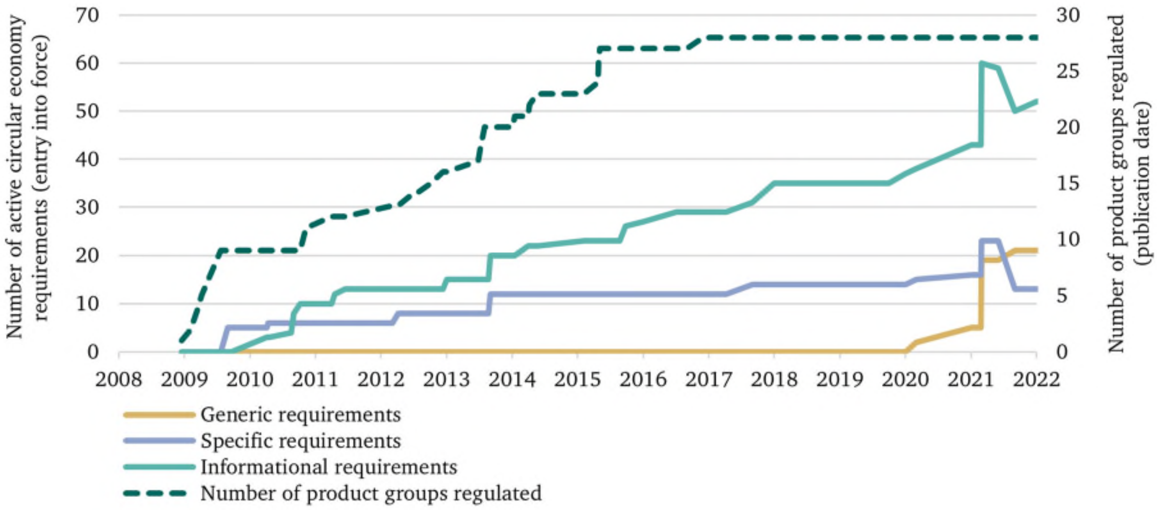


Figure 9 Types of circular economy requirements included in ecodesign product regulations over time.

The results are in line with the findings of Bundgaard et al. (Bundgaard et al. 2017) who similarly identified that is was mostly the lighting regulations that included specific circular economy requirements in the early years of the Ecodesign Directive.

Figure 10 shows the types of circular economy requirements that were in force for each product group at the time of the latest amendment. It shows, that the product family lighting has the highest number of specific requirements per individual product regulation.

This is followed by white goods and ICT, although there are strong differences within these product families. There are almost no circular economy requirements for household tumble driers, domestic ovens and range hoods, and professional refrigerated storage cabinets, and this finding also applies to simple set-top boxes and external power supplies in the ICT product family. The product group of electronic displays and televisions has the highest number of circular economy requirements since its 2019 amendment. Before 2019, only hazardous substances were regulated (mercury or lead content), but in the 2019 amendment, several requirements were added: a ban on halogenated flame retardants, spare parts availability and maximum delivery time, availability of software updates, and several informational requirements.

Very few and weak circular economy requirements were found for heating and cooling products and other products such as industrial equipment. In the 2019-generation, mainly white goods (household refrigerating appliances, household dishwashers, household washing machines, and refrigerating appliances with a direct sales function), ICT (electronic displays and televisions, and computers and computer servers), the new combined lighting regulation, and the regulation on welding equipment (grouped into the product category "Other") were responsible for almost all new circular economy requirements. Our results confirm the finding of other studies that the 2019-generation achieved a strong increase in circular economy requirements. However, not all product group regulations that are active today contain circular economy requirements: in the case of ICT regulations, simple set-top boxes were already regulated in 2009 and have not been amended or repealed since. In the preparatory study, environmental optimisation focuses mainly on energy consumption, and although improvements in recyclability can be detected in a reduction in flame retardants, this requirement does not find its way into the regulation (Harrison and Jehle 2007). Examples of regulations from the 2019-

generation that contain no or only a few circular economy requirements include those on external power supplies and electric motors. For motors, Dalhammer et al. (Dalhammar et al. 2014) identified several improvement options related to the use of rare earth elements, such as design for recycling to facilitate the extraction of rare earth elements at the end-of-life, or the provision of information on key materials and their location. Despite these recommendations, none of them found their way into the 2019 update of the electric motor regulation. In the product family of white goods, an increasing number of circular economy requirements were integrated into the 2019-generation of regulations. No circular economy requirements are included in the regulation on household tumble driers (2012) and only a few in the regulations on domestic ovens and range hoods (2014) and in professional refrigerated storage cabinets (2015), all of which have not been amended since.

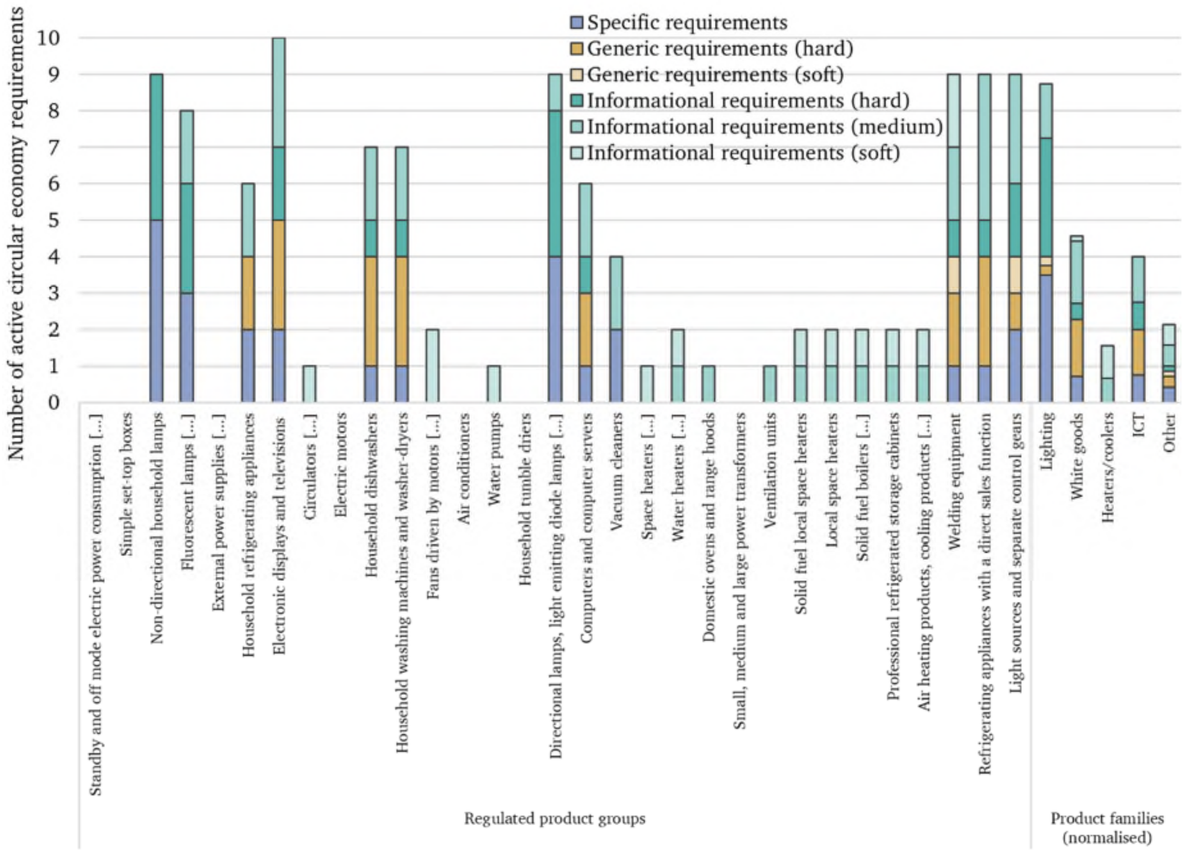


Figure 10 Types of circular economy requirements according to individual products and product groups.

Whereas the findings shown in Figure 9 and Figure 10 illustrate the shift towards increasingly stringent circular economy requirements over time, the question remains as to which circular economy strategies from the 3Rs framework and which circular economy categories are most pronounced.

2.4.3. Categories of circular economy requirements

Figure 11 shows the focus of the circular economy requirements within the ecodesign framework. There was a strong focus on durability (reduce) in the early ecodesign regulations, almost exclusively due to the three lighting regulations, whose requirements entered into force between 2009 and 2013. These included durability requirements such as lamp survival factor, lumen maintenance, rated and nominal lamp lifetime, the minimum number of switching cycles, and maximum premature failure rates. The lighting regulations also explain the decrease in the number of durability requirements and the slight decrease in requirements on hazardous chemicals in 2020, because the three existing lighting regulations were replaced by a single overarching lighting regulation. Despite this, the number of product groups remained constant in 2019, as two new product groups and one lighting regulation replaced the three former lighting regulations. The only other product groups subject to durability requirements before 2019 were the 2013 regulation on computers and computer servers, which includes informational requirements on the minimum number of loading cycles of the batteries, and the 2013 regulation on vacuum cleaners, which sets specific durability requirements for the minimum oscillations of the hose and for motor lifetime (requirements entered into force in 2017).

The findings correspond to those of Mudgal et al. (Mudgal et al. 2013), who state that requirements on minimum lifetime or warranty have not been widely extended to other product groups after its introduction for lamps.

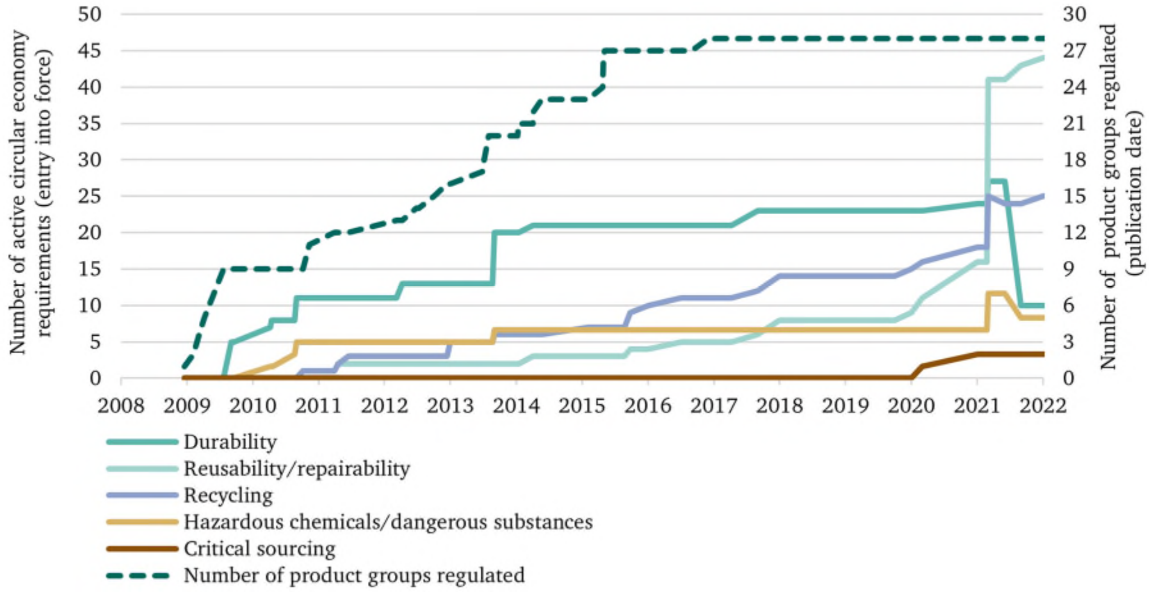


Figure 11 Categories of circular economy requirements included in ecodesign product regulations over time.

Over the years, recycling requirements have increased steadily, but this development is mainly due to often non-specific informational requirements (asking to provide “information relevant for [...] recycling, recovery and disposal at end-of-life”), e.g., in the regulations on vacuum cleaners, domestic ovens and range hoods, space heaters, water heaters, or ventilation units. The next real increase in specific requirements is in the 2019-generation, which mainly addressed aspects of repairability and recyclability. For example, Regulation (EU) 2019/2021 on electronic displays and televisions introduced design for recycling aspects (“[...] fastening or sealing techniques do not prevent the removal, using commonly available tools [...]”) and marking of components. The sharp increase in repairability requirements was caused by requirements related to spare parts, especially within the white goods product family, such as in the regulations on household refrigerating appliances, household dishwashers, and household washing machines, but also in non-white good requirements such as in the regulation on welding equipment.

In the supplementary information (accessible online, see 2.7), Figure S2 shows the results of Figure 11 normalised to the number of regulated product groups.

Mathieux et al. (Mathieux et al. 2020) noted that it is mainly recycling that is the focus of attention in the EU compared to reuse or remanufacturing. However, they do acknowledge that six of the Ecodesign regulations published in 2019 include requirements on reparability and lifetime extension, a number that corresponds with our findings.

When assessing differences between product groups, Figure 12 shows a high share of durability aspects in the lighting regulations and a similarly high focus on reusability/repairability in the white goods product family. Requirements related to critical sourcing are a more recent and still rare aspect and were only found in Regulation (EU) 2019/1784 on welding equipment (information on the presence of critical raw materials and amounts) and in Regulation (EU) 2019/424 on computers and computer servers (weight range of cobalt in the batteries and neodymium in the hard disk drive), corresponding to the findings of other researchers (Mathieux et al. 2020; Talens Peiró et al. 2020).

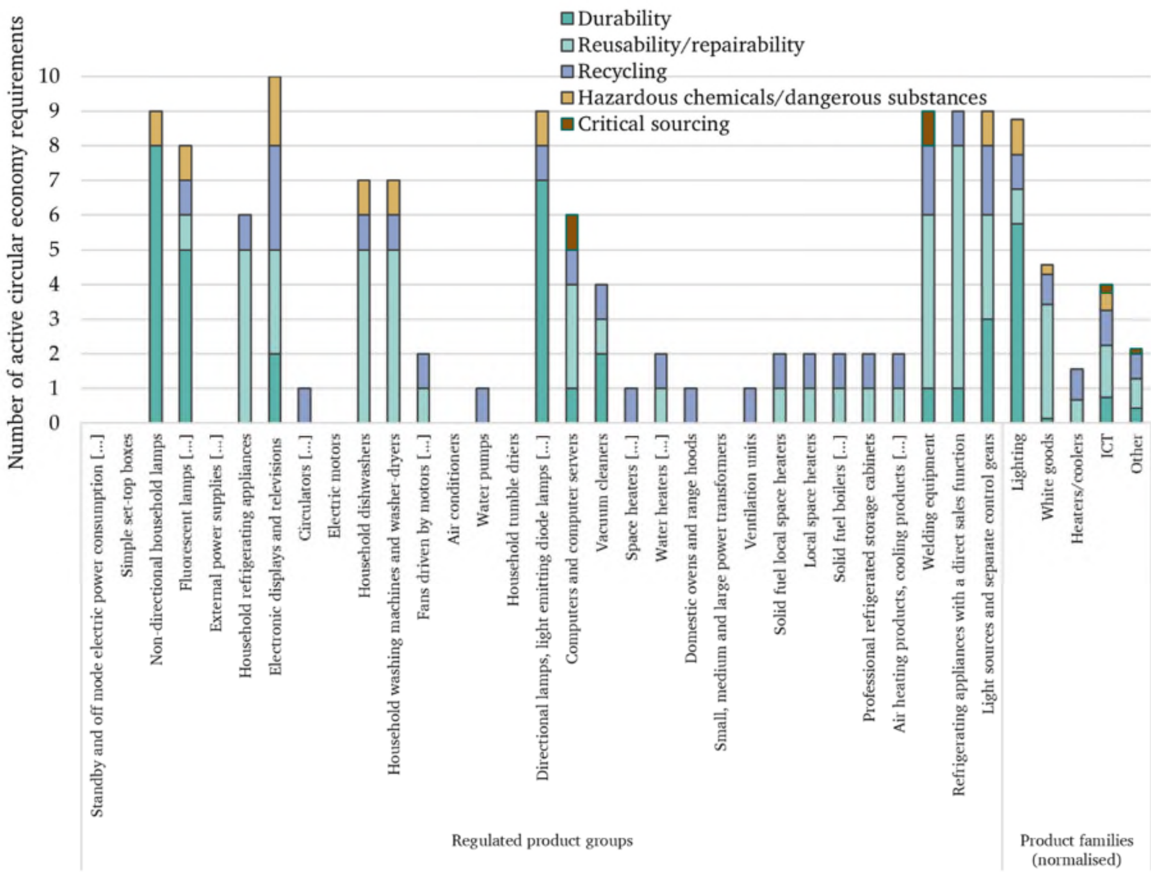


Figure 12 Circular economy requirements according to circular economy category for products and product groups.

2.4.4. Keyword analysis

Finally, Figure 13 shows the results of the keyword analysis. Unlike the content analysis, the keywords refer to the publication date of the regulation. Furthermore, keywords included in repealed regulation are not subtracted, which is why the increase in circular economy keywords is stronger and the curves do not drop in 2019. Taking these factors into account, the resulting graph mirrors the findings of the content analysis. There is an increase in circular economy keywords with the lighting regulations in 2009 and 2012 and a sharp increase due to the 2019-generation of ecodesign regulations. The increase in 2014 can be explained by the high frequency of the keyword “recover” in Regulation (EC) No. 1253/2014 on ventilation units, which here refers to the recovery of heat or moisture and can thus be recorded as an anomaly with no direct link to the circular economy.

Keywords with the highest frequency are “lifetime” with 47 mentions for durability aspects found in particular in the lighting regulations (e.g., Regulation (EC) No. 244/2009 and Regulation (EC) No. 1194/2012), “repair” with 168 mentions for reusability/repairability aspects and a high frequency in the 2019-generation (e.g., in Regulation (EU) 2019/1784, Regulation (EU) 2019/2019, Regulation (EU) 2019/2021, Regulation (EU) 2019/2022, Regulation (EU) 2019/2023, Regulation (EU) 2019/2024), “recycl*” with 65 mentions for recycling aspects (e.g., in Regulation (EU) 2019/2021 or Regulation (EU) 2016/2281), “mercury” with 82 mentions for hazardous chemicals, mainly in the lighting regulations (e.g., in Regulation (EC) 244/2009, Regulation (EC) No. 245/2009, and Regulation (EC) No. 1194/2012), and “critical raw materials” with 11 mentions for critical sourcing aspects (e.g., in Regulation (EU) 2019/424 and Regulation (EU) 2019/1784).

The results table and additional graphs based on the keyword analysis can be found in the supplementary information (accessible online, see 2.7).

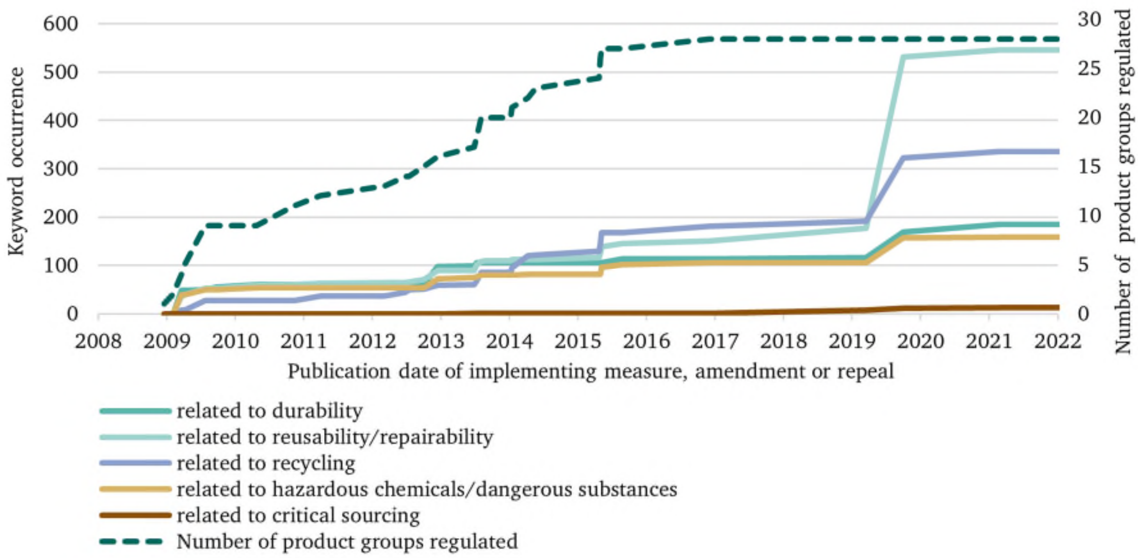


Figure 13 Keyword analysis of circular economy requirements according to circular economy category.

2.5. Discussion

In its 2005 version, the Ecodesign Directive already provided all the legal means needed to regulate circular economy aspects (see Article 15 4(a)).

Overall, our analysis showed a constant increase in the number of regulated product groups under the ecodesign framework, which only was interrupted between 2016 and 2019, and an overall increase in the number of circular economy-related requirements, but with significant variations between product groups.

The first lighting regulations (2009 and 2012) were already integrating a large number of circular economy requirements and focused especially on durability (reduce). However, the lighting regulations proved to be outliers and relatively few durability requirements were set for other product groups.

As discussed by Mudgal et al. (Mudgal et al. 2013), the specific policy measure of minimum lifetime or warranty has high relevance to improving material efficiency but has not been widely extended to other product groups after its introduction for lamps. A prerequisite for its application is the availability of testing standards, which become more difficult (more time-consuming and costly) to implement as product complexity increases (Mudgal et al. 2013; Bobba et al. 2016). Since lamps are a comparably simple product, this can explain why it was possible to include durability requirements in the regulations ahead of other product groups. Compared to many other regulated product groups, lights have no moving parts and have unique usage patterns. Once installed, there is usually no direct contact with the user, who only switches the product on and off, so the risk of incorrect operation during the product's lifetime is low. Accordingly, durability requirements were easier to set and more likely to be accepted by manufacturers for lighting products than for other product groups.

For more complex ecodesign products, the strategy was initially to regulate only individual components, such as the durability of the hose and minimum motor lifetime for vacuum cleaners in 2013. Further durability requirements of more complex products were implemented in the 2019-generation of ecodesign: for example, the availability of software and firmware updates for electronic displays, televisions, computers, and computer servers, or informational requirements on warranty (refrigerating appliances with a direct sales function) or the minimum number of battery loading cycles (computers and computer servers). The standard EN 45552:2020 on the assessment of the durability of energy-related products, despite its generic character, might lead to more (and more ambitious) functional durability requirements in the future.

The focus on durability for lighting can also be explained by the difficulty of repairing lighting products. In the three lighting regulations prior to 2019 we only found one requirement related to repairability (informational requirement on maintenance instructions for fluorescent lighting). Improvements in repairability are most likely not caused by missing spare parts, but instead by technical limitations or safety concerns. In 2009, the incandescent lamp faced competition from compact fluorescent lamps, which proved to be much more efficient and durable, but contained the toxic substance mercury (Elijošiutė et al. 2012). Accordingly, there were product-specific reasons why the ecodesign requirements should address durability, e.g., to avoid the release of toxic substances. In addition, the requirements guaranteed the cost-effectiveness of the compact fluorescent lamps technology compared to cheaper incandescent lamps.

Whereas regulating repairability or recyclability at market entry is difficult for lighting products beyond informational requirements, requirements on recycling at the end of life are covered by the horizontal Directive on waste electronic equipment. Therein, Annex V defines minimum recovery targets for lighting (European Commission 2012a).

If lighting regulations are regarded as an outlier, then the observed increase in circular economy requirements in the 2019-generation confirms the findings of other studies that a standardised approach to integrating circular economy aspects into the ecodesign policy-making process has only recently received greater attention (Polverini and Miretti 2019; Mathieux et al. 2020; Polverini 2021). This trend can be seen for example in the policy-making process for the 2019 regulation on data servers with its strong focus on circular economy aspects (Talens Peiró et al. 2020). We found electronic displays and televisions since their 2019 amendment to be the single product group with the highest number of circular economy requirements. This strong emphasis on circular economy in the 2019 amendment goes in line with the methodological work of the REAPro Research programme by the Joint Research Centre, where a material efficiency assessment method was developed that was then applied to the policy formulation process of several product groups including that of electronic displays (Mathieux et al. 2020). The overall increase in circular economy requirements thus goes hand in hand with the improvement or development of methodological approaches on assessing material efficiency (Mathieux et al. 2020), a process that is still ongoing (Polverini 2021). The revision of the MEErP is another important component for improving the circularity of products. In the revision, the systematic inclusion of material efficiency aspects in the modelling process was identified as one area for improvement (European Commission 2021c).

Whereas the EN 4555X series of standards on resource efficiency now exist, the standards are very generic and it remains to be seen how well they can be integrated into upcoming ecodesign regulations. The individuality of the product groups will most likely require the development of individual vertical product standards, as also pointed out by Polverini (Polverini 2021). Nevertheless, development is well underway also outside of the EU ecodesign context, as evidenced also by national legislation such as the French repairability index, which was implemented in 2021 (Ministère de la Transition écologique 2021). Our results also highlight the differences between product families, with a focus on durability for lighting regulations and a high share of reuse requirements for white goods. Although the low product complexity of lighting products might have facilitated the use of durability requirements, white goods would require more complex and time-consuming test standards (Stamminger et al. 2020). On the contrary, the repair of white goods is likely more attractive than lighting due to higher product costs, which has led to a greater role of requirements related to spare parts (reuse) in 2019. Those requirements can reduce the barrier to repair through lack of spare parts availability (Tecchio et al. 2019).

Heating and cooling products were found to have very few circular economy requirements, which might be explained by their design; these products often have almost no moving parts and are of robust construction. Furthermore, heating and cooling products receive little attention in the academic publications regarding circular economy aspects, compared to other product groups that are assessed in dedicated case studies, like for example electric motors (Dalhammar et al. 2014), computers and computer servers (Talens Peiró et al. 2020), or vacuum cleaners (Bundgaard et al. 2017; Bracquené et al. 2018). Another explanation for their low number of circular economy requirements is the fact that they were not among the products amended in 2019 and thus did not benefit from the advancements in material efficiency assessment methods (Mathieux et al. 2020).

The hypothesis that the least preferable circular economy option in the 3R framework (recycle) is often the focus of research and policy efforts (Mathieux et al. 2020) is not supported by our findings. Although it is true that the highest priority option of durability is almost exclusively addressed in the lighting regulations—which can be considered an exception within the ecodesign framework—both the content and the keyword analysis showed a stronger increase in reuse requirements compared to those related to recycling. In particular, the numerous requirements for spare parts in the 2019-generation indicate a strong political will to promote the reuse of products.

The circular economy requirements we have identified in this work are largely consistent with the literature (Mudgal et al. 2013; Bundgaard et al. 2017; Mathieux et al. 2020; Polverini 2021) and based on our circular economy taxonomy we could furthermore differentiate between the strength and type of requirements. After having conducted the analysis, few adjustments to our initial taxonomy might be relevant: “consumables” could be added in the “reduce” strategy to account for the reduction in resource consumption during the use phase, for example, the use phase water consumption for domestic dishwashers and household washing machines. Furthermore, circular economy requirements can mainly regulate the product at the time of market entry, therefore repairability and recyclability can be regulated more easily than actual repair or recycling operations (Polverini 2021). Instead of “recycle”, the term “recyclability” will be more adequate in the taxonomy.

A common taxonomy with a clear classification of circular economy requirements is essential for advancing circular economy product regulations in the EU. Our proposed taxonomy can serve as a starting point. Similarly, further enhancement of standards and testing methods is needed to allow for the integration of more and stronger circular economy requirements in upcoming ecodesign legislation (Mathieux et al. 2020). Work is well underway (Mathieux et al. 2020; Polverini 2021) and is focussing on a variety of different approaches, including a scoring system for the repairability of products comparable to the French repairability index (Cordella et al. 2019; Spiliotopoulos et al. 2022).

Many other possible circular economy requirements are not yet part of current ecodesign regulations. These include recycling and collection rates, for instance, which are already applied in other EU regulations such as the Batteries Directive (European Commission 2006), or digital product passports and recycled contents (see also (Polverini 2021)), which have been introduced in the proposal for a new Batteries Regulation (European Commission 2020c). Other novel concepts are product-as-a-service, bans of unsold goods, or measures against premature obsolescence, which are mentioned in both the Circular Economy Action Plan 2020 (European Commission 2020e) as well as in the Impact Assessment for the Sustainable Products Initiative and the Proposal for Ecodesign for Sustainable Products Regulation (European Commission 2022c). Critical sourcing requirements were only detected in two regulations, although this aspect may become increasingly important due to the high demand for critical raw materials needed for low-carbon technologies, such as cobalt and lithium. It is expected that all the above-mentioned aspects will play a greater role in future ecodesign regulations or in legislation based on the upcoming Ecodesign for Sustainable Products Regulation. Both the Ecodesign Working Plan 2016–2019 (European Commission 2016) as well as its successor, the Ecodesign Working Plan 2022–2024 (European Commission 2022a) refer explicitly to the circular economy, indicating that ecodesign will continue to evolve from an energy efficiency focus towards a stronger emphasis on circular economy aspects.

Although our results indicate that there has been an increase in the number and diversity of circular economy requirements, the actual impact of market-entry requirements on reuse and recycling rates can only be verified via ex-post evaluations, as also suggested by (Mathieux et al. 2020). Furthermore, many of the existing impact assessment studies do not consider material efficiency aspects, and if they do, they often do so only qualitatively or semi-quantitatively, as noted by (Polverini 2021). Thus, a stronger focus should be placed on assessing the impact of circular economy requirements, and in this respect, it will be very crucial to follow the impact of the 2019-generation ecodesign regulations on the market (e.g., does the number of repairs or spare part purchases increase significantly, and will the average lifetime of the products increase?).

In terms of the general development of the circular economy in the EU, the increasing inclusion of circular economy requirements in ecodesign can be seen as a positive indicator, helping to overcome cultural and market barriers and supporting the political advocacy to put more emphasis on resource efficiency.

Our analysis has several limitations: the robustness of our content analysis is limited due to its qualitative nature. In particular, the number of requirements should be treated with caution, not only because of the qualitative classification but also because of the different styles (wording) of the ecodesign regulations, which vary greatly in terms of the number and level of detail of the requirements per regulation. The keyword analysis added a layer of reliability and supported the trend identified by the content analysis.

We covered a wide range of legislative texts and identified overall trends within ecodesign. At the same time, this holistic approach and the focus of the legislative texts reduced the depth of the analysis for individual product groups and the policymaking process in general.

2.6. Conclusions

Our results show that the product groups covered by the Ecodesign Directive have expanded greatly over the years, with frequent amendments or repeals of existing legislation. Our taxonomy of circular economy requirements provided nuanced results, differentiating requirements by type and strength. We could show a clear increase in circular economy requirements over time, especially in the 2019-generation of implementing measures, where the number of specific and generic function requirements increased significantly. Prior to 2019, mainly informational requirements were implemented, with the exception of lighting regulations.

Lighting was identified as an outlier, with many requirements focussing on durability before implementing measures for other product groups including circular economy requirements. The focus on durability for lighting remains an exception also in the 2019-generation, where we did find a significant increase in measures, but mainly related to reusability/repairability and to a lesser extent recycling.

Electronic displays and televisions were found to be the product groups that include most circular economy requirements, in equal parts functional and informational requirements.

In terms of product families, lighting contains by far the highest and most stringent circular economy requirements, followed by white goods (with a high share of repairability requirements) and ICT (being the only product family that started to regulate critical sourcing). The product family Heaters and Coolers showed consistently very few circular economy requirements and mainly informational ones. For the remaining products (product family Other), it was only vacuum cleaners and welding equipment that include several circular economy requirements.

Although our results revealed the trend of product policies towards circular economy aspects, the drivers behind this development are yet to be identified. Which advocacy coalitions influence the product policy-making process, what are the advantages and disadvantages for the main stakeholders involved, and what role is played by macroeconomic considerations at the EU level? A robust method for integrating circular economy aspects into policy impact assessments and ex-post evaluations is needed to justify and steer the uptake of circular economy requirements in product policy regulations. The EN 4555X series of standards is already an important step in the right direction but needs to be extended by product group-specific (vertical) standards to improve the implementability of circular economy requirements. Another important building block is the revision of the MEErP with the aim of systematically including material efficiency aspects into the ecodesign impact assessment (European Commission 2021c). Finally, the extension of ecodesign to non-energy-related products under the Proposal for Ecodesign for Sustainable Products Regulation (European Commission 2022c) offers the opportunity to integrate the circular economy into framework legislation from the outset and to increase the scope and environmental benefits of ecodesign. The large number of individual methodological advancements, standards, and initiatives show great inertia in the field of circular economy product policies and can be a great leap forward.

2.7. Supplementary information

The following supporting information can be downloaded at:
https://www.mdpi.com/article/10.3390/su141610318/s1

- Figure S1: Number of circular economy requirements (entry into force) normalised to number of regulated product groups (publication date) according to type and year;
- Figure S2: Number of circular economy requirements (entry into force) normalised to number of regulated product groups (publication date) according to category and year;
- Figure S3: Examples of keyword occurrence in ecodesign regulations over the years;
- Figure S4: Number of circular economy keyword occurrence normalised to number of regulated product groups according to categories in ecodesign requirements;
- Table S1: Results of the keyword analysis;
- Table S2: Over-view of assessed product (sub)groups and legislative texts.

3. Analysing policy change towards the circular economy at the example of EU battery legislation

This chapter was published in August 2023 in the journal Renewable and Sustainable Energy Reviews. Table 7 provides more information about the publication.

Table 7 Publication information of Barkhausen et al. (2023a).

Title	Analysing policy change towards the circular economy at the example of EU battery legislation
Authors	Robin Barkhausen, Katharina Fick, Antoine Durand, Clemens Rohde
Publication date	31.08.2023
Journal	Renewable and Sustainable Energy Reviews
DOI	https://doi.org/10.1016/j.rser.2023.113665
Author contributions according to the Contributor Roles Taxonomy (CRediT 2023)	Robin Barkhausen: Conceptualization, Methodology, Formal Analysis, Data Curation, Writing-Original Draft, Writing-Review & Editing, Visualization Katharina Fick: Conceptualization, Methodology, Formal Analysis, Investigation, Data Curation Antoine Durand: Conceptualization, Methodology, Data Curation, Writing-Review & Editing Clemens Rohde: Writing-Review & Editing, Supervision

3.1. Introduction

Batteries play a key role in the decarbonisation of road transport and as such have become a technology of key interest to the EU economy with its strong focus on the automotive sector. However, almost all mining production and reserves of key battery materials (e.g. lithium and cobalt) are located outside of Europe (USGS 2022) and the majority of battery manufacturing takes place in Asia (Olivetti et al. 2017). To support the development of a competitive European battery industry, the European Commission launched the European Battery Alliance in 2017 (European Commission 2020b) and presented its Strategic Action Plan on Batteries in 2018 (European Commission 2018). In the context of Europe's dependence on imported materials, the EU developed a strategy to increase the adoption of a circular economy with its Circular Economy Action Plan, which identifies batteries as one of seven key product value chains (European Commission 2020e). Due to the strategic importance of batteries for the EU economy, it is relevant to analyse how the product group of batteries is regulated.

Batteries have been regulated at EU level since 1991 and the legislation has been replaced once in the last three decades. It is currently undergoing another review process, indicating significant movement in this policy area (Fick 2022).

Council Directive 91/157/EEC was the first piece of legislation to deal specifically with batteries, focusing on the "recovery and controlled disposal of those spent batteries [...] containing dangerous substances" (Council of the European Communities 1991). The legislation set restrictions on the mercury content of alkaline manganese batteries, but was criticised for its limited scope in terms of battery type coverage and depth of measures (BIO Intelligence Service 2003). Following a public consultation and an impact assessment study (BIO Intelligence Service 2003), a new legislation covering a wider range of batteries and accumulators was proposed and published in 2006 as Directive 2006/66/EC. While other energy-related product groups have only recently started to include more circular economy aspects (Barkhausen et al. 2022), the Battery Directive was comparatively progressive, introducing requirements such as minimum collection rates and recycling efficiency as early as 2006.

In line with the growing political interest in the battery sector over the last decade, the revision of the Battery Directive was launched at short notice, with a preparatory study on batteries being launched under the Ecodesign framework. This can be seen as a surprise, as there was no mention of batteries in the Ecodesign Working Plan for 2016-2019 (European Commission 2016), and the overall progress of the Ecodesign legislative package has been slow, as evidenced by an open letter to the European Commission in 2018 in which 55 organisations criticised the delays (55 European organisations 9/26/2018). Despite the apparent slow progress for many product groups, a study for an Ecodesign preparatory study on batteries was commissioned in 2018, the same year as the open letter (European Commission 2021d). This study led to the Commission's proposal for a new regulation on batteries in December 2020, which aims to cover the entire life cycle of batteries and create a competitive and sustainable European battery industry (European Commission 2020c). Compared to the 2006 version, there is a major shift in the scope and type of requirements. The legislation changes from a Directive to a Regulation, and the new proposal explicitly mentions lithium-ion and traction batteries and includes several previously unseen circular economy requirements, such as mandatory recycled content, material-specific recycling efficiencies, legal specifications on second battery life, and the battery passport as a digital product passport. Consultations and negotiations based on the proposal have been ongoing since 2020, and a provisional agreement between the European Council and Parliament was reached in December 2022 (European Commission 2022b) (see Figure 14).

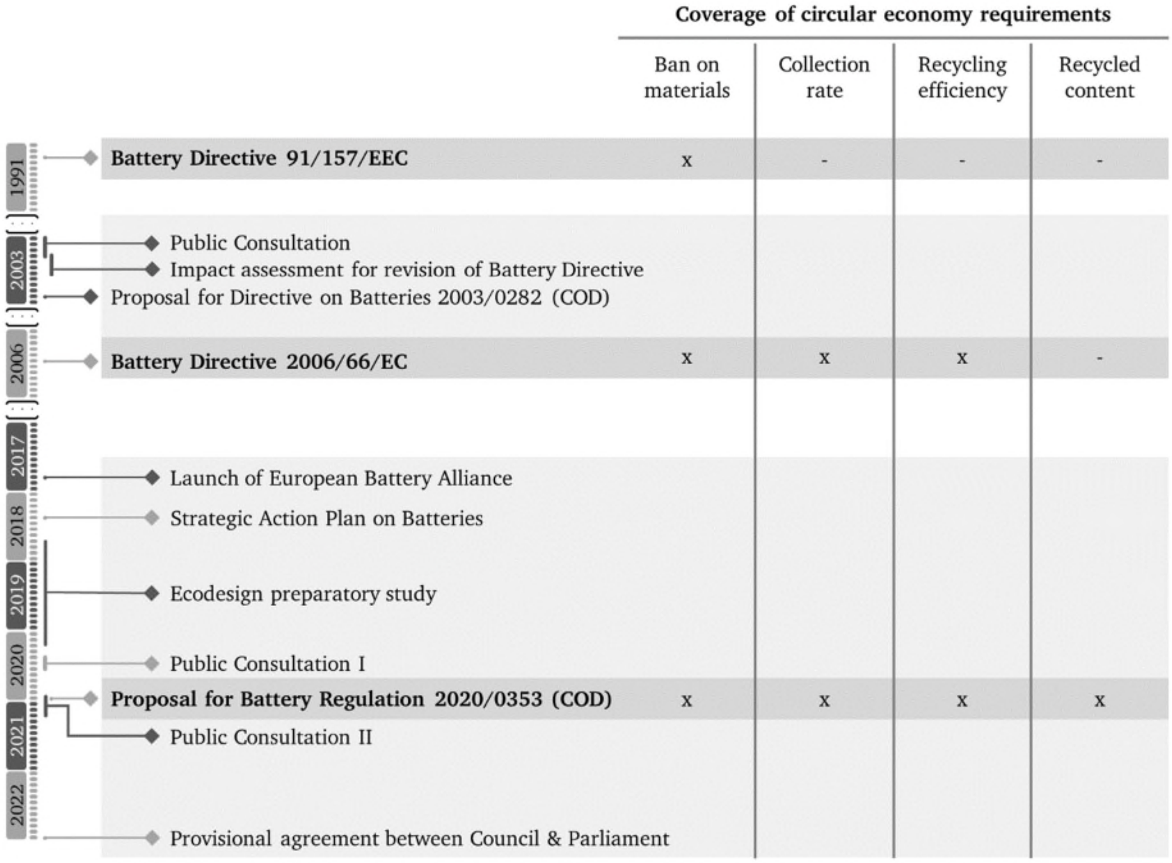


Figure 14: Timeline of EU battery legislation and related events, and coverage of circular economy requirements.

Given the apparent evolution of the EU battery legislation towards a stronger focus on circular economy aspects, it is of interest to analyse this trend and its underlying drivers. The key research question of the study is how the policy change in the battery policy subsystem occurred, and which internal (actor-driven) and external (system-driven) factors have contributed to it. In particular, it aims to assess how a change in stakeholder groups and their motivation (belief systems) influenced the evolution of EU battery legislation towards a stronger integration of circular economy aspects.

The paper begins with the theoretical framing of the applied framework and an overview of studies in the field of battery policy change. It then details the mixed methods approach, explaining the methods and means of the interviews and document analysis. The results are then presented in four parts. First, the circular economy aspects of EU battery legislation are analysed, which requires a definition of circular economy. Second, the stakeholders that are or have become part of the policy subsystem are examined. Third, potential coalitions and their influence on policy change are examined. And fourth, the impact of external events is assessed. The final chapter discusses the implications of the findings and provides a brief summary and critique of the results.

3.2. Theoretical framing

In 1988, Paul Sabatier proposed the Advocacy Coalition Framework (ACF) as a conceptual framework for explaining policy change in a policy area (or sub-system) over time (Sabatier 1988). In order to provide a reasonably accurate picture of policy change, time spans of at least a decade are usually assessed (Sabatier 1988). The ACF considers all actors within the policy subsystem and assumes that they can be grouped into so-called advocacy coalitions, "composed of people from various organisations who share a set of normative and causal beliefs and who often act in concert" (Sabatier 1988). The beliefs of actors within the policy subsystem are grouped along a declining susceptibility to change into deep core (fundamental normative axioms), policy core (policy positions to achieve the deep core normative axioms), and secondary aspects (the multitude of instrumental decisions required to implement the policy core) (see Figure 15).

Policy change occurs within a policy subsystem, and Jenkins-Smith et al. describe four ways in which such change can be triggered as (1) external shocks or events (outside the control of actors in the policy subsystem), (2) internal shocks, (3) policy-oriented learning, or (4) negotiated agreements (Figure 15) (Jenkins-Smith et al. 2014). We will focus in particular on the coalition-building process and the role of external shocks in the policy subsystem of EU battery legislation. External factors or shocks (e.g., changes in socio-economic conditions, public opinion, or in cross-subsystem policies) can play an important role in policy change, and while they alone may not be sufficient to change the core policy features of a government programme, they may be a necessary condition (Sabatier 1988; Sabatier and Weible 2015). In the case of the EU battery legislation, cross-subsystem impacts could include the EU strategy towards a circular economy and the electrification of passenger road transport (European Commission 2019c, 2020e); a strategy, in which batteries play a central role. In addition, the growing importance of batteries for the transport sector may have led to an increase in the number of actors involved and possible shift in coalitions.

There is considerable variation in how advocacy coalitions are identified when applying the framework (Pierce et al. 2020), but most studies conduct qualitative case studies using methods such as interviews and/or document analysis (Pierce et al. 2017).

The ACF is most commonly applied to high-salience policy issues (e.g. nuclear power plants (Nohrstedt 2005)), but has also been found to be appropriate for less controversial, low-salience policy issues (Giordono 2020), such as batteries (often with less clearly defined results). In recent years, there have been several policy studies on batteries, such as the implications for future legislation due to end-of-life batteries (Giosuè et al. 2021), in particular from electric vehicles (Malinauskaite et al. 2021). Other studies have looked at battery recycling policies (Turner and Nugent 2016) or the impact of the proposed EU regulation at a global level (Melin et al. 2021). However, the analysis of battery policy change often does not focus on the impact of internal, actor-based factors (e.g. actor beliefs and coalitions) (Fick 2022). Studies that do consider such internal dynamics iclude the work of Häge, who analysed coalitions in the EU Council during the development of Directive 2006/66/EC (Häge 2013), Gupta et al., who analysed stakeholders in the context of battery waste (Gupta et al. 2018), or Bonsu, who assessed drivers for the transition to electric vehicles in the UK (Bonsu 2020). To the best of knowledge, no study has been conducted applying the ACF to the EU battery policy subsystem.

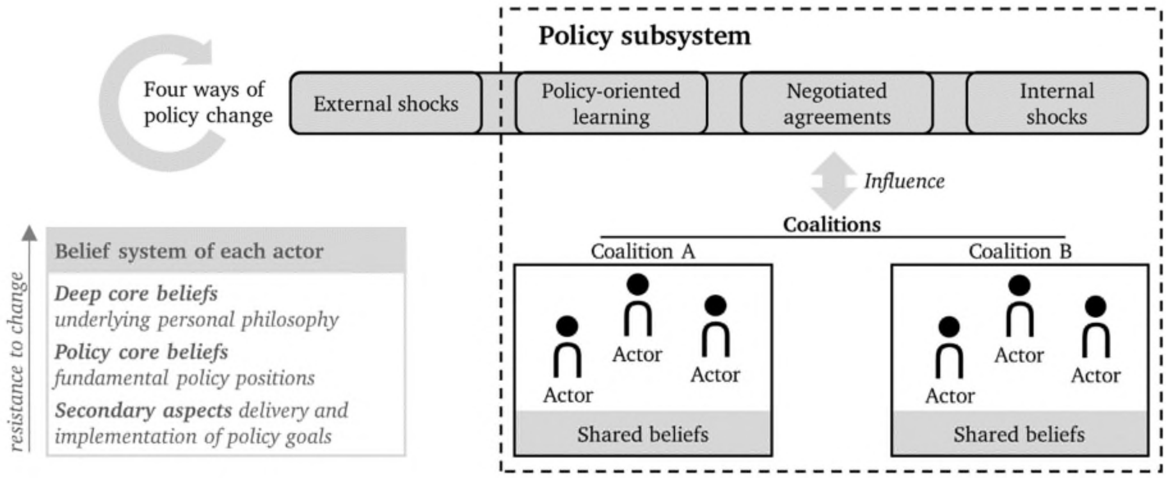


Figure 15: Coalitions and policy change in the Advocacy Coalition Framework.

3.3. Material and methods

A mixed method approach was chosen, combining qualitative document analysis and interviews. Before conducting the interviews and analysing stakeholder comments, the current and historical EU battery legislation was analysed for its coverage of circular economy aspects. This included the definition of circular economy policy measures. Based on this, the legislative texts were scanned, the findings were integrated into a taxonomy of circular economy policy measures and the coverage of circular economy aspects was compared with other EU product legislation. This legislative analysis was also used to select the circular economy aspects for the subsequent document analysis of stakeholder comments.

Following the publication of an official document detailing the aim or proposal of EU legislation (e.g. consultation document, inception impact assessment, or proposal for a regulation), interested stakeholders (e.g. companies or interest groups) are given several weeks to submit comments (one to several pages each). Such consultation rounds are open to the public and submitted comments are available on the Commission's website. In the document analysis, such comments from the public consultation on Directive 2006/66/EC and two consultation rounds on the current proposal for a Battery Regulation (2020/0353 (COD)) were analysed (see Table 8). Stakeholder comments on the consultation document for the revision of the 2006 Battery Directive could be submitted until 28 April 2003 (European Commission 2003), for the inception impact assessment of the most recent battery regulation proposal until July 2020 and for the proposal for a regulation until March 2021 (European Commission 2020a, 2021a). A total of 368 stakeholder positions from the three consultation rounds were extracted from the Commission websites or a web archive (see Table 8), then the data were cleaned and converted to tabular format. Stakeholders were then grouped into the categories of academia, national or regional authorities, individual economic operators, joint industry and trade associations and non-governmental organisations (NGOs). After an initial analysis to identify key actors and topics of discussion during the consultation, a total of 39 positions were selected for detailed analysis. The focus was set on comments from joint industry and trade associations and NGOs, each representing a large number of stakeholders. This approach was chosen to limit the number of positions for detailed analysis. In order to classify the stakeholder positions on the identified circular economy policy measures, a blind rating of the comments on a seven-point scale ranging from agreement to disagreement was carried out. Three of the authors rated each comment independently, without knowledge of the stakeholder, and the results were averaged.

Table 8: Stakeholder comments during the EU battery legislation public consultations.

Reference legislation	2006/66/EC	2020/0353 (COD)	
Year of public consultation	2003	2020	2021
Number of stakeholder positions	129	104	135
Source	Extracted from internet archive	(European Commission 2020a)	(European Commission 2021a)

Although the majority of stakeholders were individual economic operators, it was considered appropriate to focus on joint position statements as they reflect the agreed view of a large number of individual stakeholders. The reader should bear in mind that the comments of individual stakeholders vary and that the positions analysed represent the common ground.

To complement the document analysis, a total of six interviews (30 to 45 minutes) were conducted between May and July 2022. Two semi-structured interviews were conducted with researchers in the field of EU battery legislation in order to identify the most relevant stakeholders in the policy subsystem and to generate hypotheses on the drivers for policy change. Based on the stakeholder scoping, four semi-structured interviews were conducted with representatives of several of the identified stakeholder groups (recyclers, battery component suppliers, environmental organisations and consumer organisations) to explore actors' belief systems, coalition building and external factors at play in the policy subsystem. It was beyond the scope of this study to conduct interviews with representatives of all identified stakeholder groups.

The Interviews were transcribed using AmberScript and qualitative analysis software MAXQDA was used for contextual categorisation into themes such as circular economy aspects, external factors, beliefs or coalitions.

Figure 16 provides an overview of the interrelationship between the research methods of interviews and document analysis.

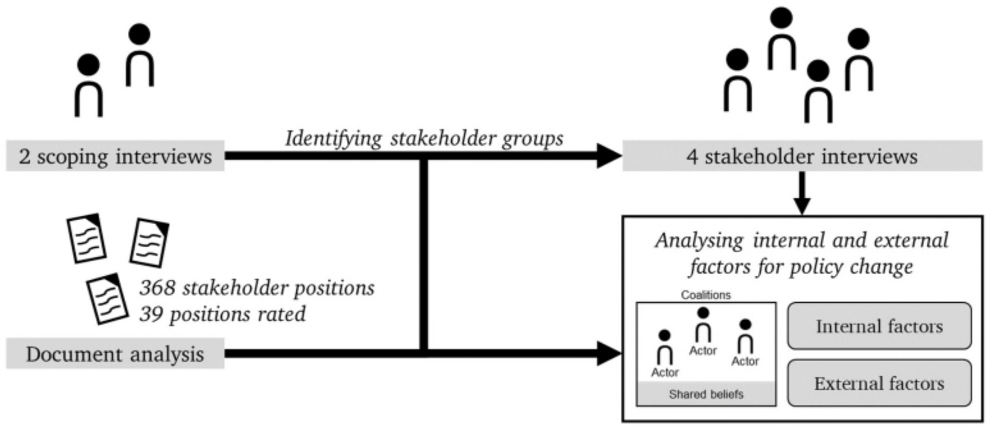


Figure 16: Overview of the mixed methods approach.

3.4. Results

3.4.1. Coverage of circular economy requirements

The analysis of the EU battery legislation showed a strong consideration of circular economy aspects, going beyond the requirements of other EU product legislation (in particular compared to the 27 product groups covered by the framework of the Ecodesign Directive (Barkhausen et al. 2022)) and extending the scope beyond entry-to-market requirements.

Here, the definition of a circular economy as an economy "where the value of products, materials and resources is maintained in the economy for as long as possible, and the generation of waste is minimised" was adopted (European Commission 2015), by replacing the "'end-of-life' concept with reducing, alternatively reusing, recycling and recovering materials" and with "the aim to accomplish sustainable development" (Kirchherr et al. 2017). Following this definition, the taxonomy of circular economy requirements proposed by (Barkhausen et al. 2022) for ecodesign product groups has been extended with additional measures introduced in the EU battery legislation Figure 17. A minimum collection rate and recycling efficiencies for specific battery types, as well as the concept of producer responsibility, were already introduced in Directive 2006/66/EC, and information on and mandatory levels of recycled content and legal clarifications on second-life batteries are new measures in the current legislative proposal 2020/0353 (COD). The current proposal also includes the Battery Passport (Digital Product Passport) as a horizontal measure affecting more than one of the circular economy strategies "reduce, reuse and recycle" (for simplicity and due to overlaps between circular economy strategies, the classification in Figure 17 has been based on the 3Rs framework, while recognising that the 3Rs framework can be extended, e.g. to the 9Rs (Potting et al. 2017)).

Figure 17 also shows that the EU battery legislation and its current proposal include a large part of the circular economy requirements used in the product regulations of the Ecodesign Directive, such as material bans or disassembly instructions.

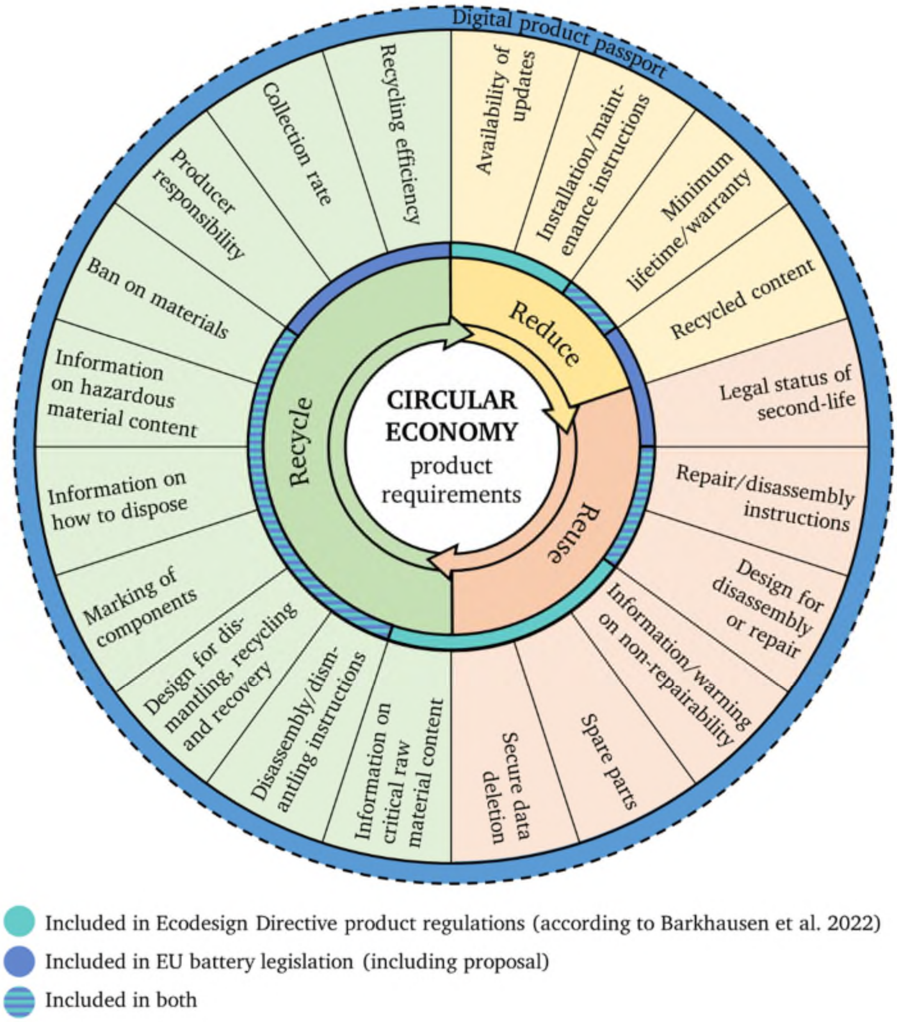


Figure 17: Taxonomy of circular economy product requirements.

Other new requirements in proposal 2020/0353 (COD) are the mandatory carbon footprint declaration and supply chain due diligence. While both contribute to sustainable development (the aim of the circular economy), they are not inherently linked to material flows or the end-of-life phase. Therefore, they are not classified as circular economy requirements.

Three of the circular economy requirements are particularly noteworthy because they each represent a novel and potentially controversial aspect in one of the three generations of battery legislation/ legislative proposals: ban on materials, recycling efficiency and recycled content (Table 9). The restrictions on mercury content were already part of legislation 91/157/EEC and were extended to cadmium in legislation 2006/66/EC; the minimum recycling efficiencies were introduced in legislation 2006/66/EC on a total product weight basis and are tightened up in proposal 2020/0353 (COD) to material specific values for cobalt, nickel, lithium, copper and lead; the mandatory declaration or level of recycled content is the most novel of the three requirements, appearing for the first time in proposal 2020/0353 (COD).

Table 9: Evolution over time of three circular economy requirements in EU battery legislation.

	91/157/EEC	2006/66/EC	2020/0353 (COD) (proposal)
Ban on materials	Restrictions of mercury content (0.025-0.05% by weight) for alkaline manganese batteries (Article 3)	Restrictions of mercury (0.0005% by weight) and cadmium (0.002 % by weight) content (Article 4)	Restrictions of mercury (0.0005-0.1%) and cadmium (0.002-0.01%) content depending on application
Recycling efficiency	Not included	Recycling efficiencies from 2011 onwards, 50-65% by average weight depending on battery type (Article 12)	Increased recycling efficiencies for lead-acid and Lithium-Ion batteries and material specific values for Cobalt, Nickel, Lithium, Copper, Lead
Recycled content	Not included	Not included	Mandatory declaration or levels of recycled content (e.g. Cobalt, Nickel, Lithium, Lead)

The analysis of coalition building and its impact on the policy-making process focuses in particular on the three circular economy requirements mentioned above.

3.4.2. Stakeholders

Stakeholders were identified on the basis of two scoping interviews and an initial screening of stakeholder consultation documents and can be grouped into (1) industry associations (raw material suppliers, battery component suppliers, battery manufacturers, automotive manufacturers, non-automotive product manufacturers, battery recyclers), (2) NGOs (environmental, consumer), (3) public authorities (Commission, Parliament and Council representing the governments of the Member States) and (4) other stakeholders (e.g. media or academia). This classification is broadly in line with that proposed during the recent impact assessment process (Stahl et al. 2021).

Interviewees (I1, I2) indicated that the stakeholders involved in the current legislative process have become much broader due to the extended scope of the regulation, reflecting the increased economic importance of batteries (mainly due to electric mobility). Indeed, while the total number of stakeholder comments in each consultation round has remained relatively stable (Table 8), in 2003 only three (2%) of the comments during the public consultation were submitted by companies or associations from the automotive sector, compared to 14 (10%) in 2021.

Looking at the development of the battery market, the strong dynamism of the sector is evident (Figure 18). In 1991 (the year Directive 91/157/EEC was published), the EU battery market was dominated by lead-acid batteries, a rechargeable battery used mainly as a starter battery for internal combustion engines. In 2006, lead-acid batteries were still by far the most produced battery type. Although rechargeable lithium-ion batteries have been sold since 1991, their market share was small compared to other battery types in the early 2000s and they were therefore not mentioned in Directive 2006/66/EC. It is only in the last decade that we have seen an exponential growth of lithium-ion batteries, mainly due to their use in electric vehicles, and today they have replaced lead-acid batteries as the battery type with the highest production demand (Figure 18). Between the Battery Directive 2006/66/EC and the current proposal 2020/0353 (COD), there has been a dramatic evolution of the battery market and related industries. In a relatively short period of time, batteries have become one of, if not the core component for automotive companies, making the battery sector increasingly important for the EU economy as a whole.

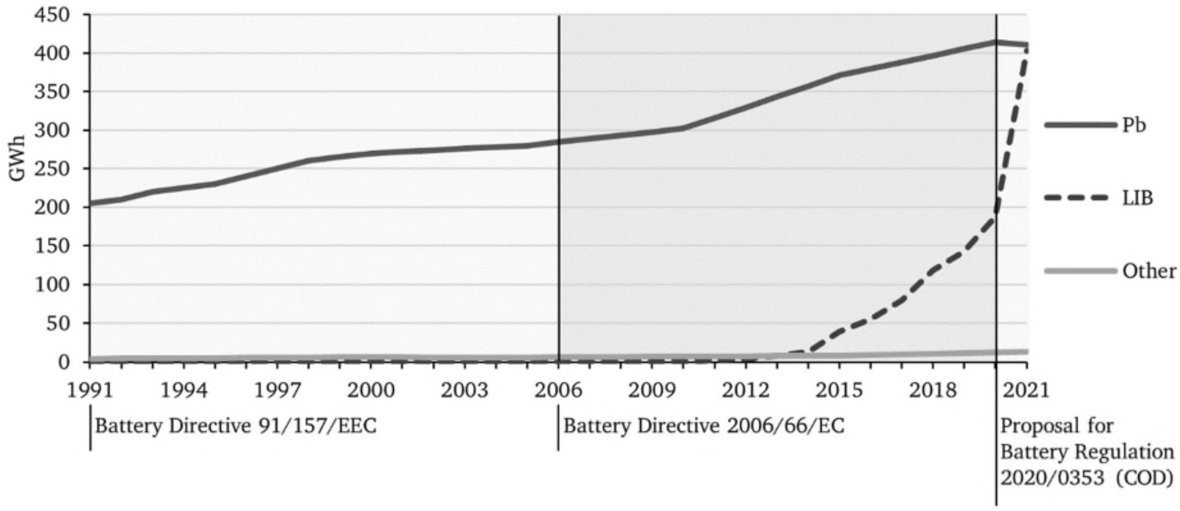


Figure 18: Development of the global battery market demand based on (Thielmann et al. 2017), updated in January 2023 with internal market study data from Fraunhofer ISI (Pb = lead-acid battery; LIB = lithium-ion battery; "Other" includes nickel-cadmium, nickel-metal hydride, or redox flow batteries, among others).

3.4.3. Coalitions

The convergence of actors' beliefs, together with observed coordination and shared resources, can indicate the existence of coalitions. Several joint position papers were published in the public consultation rounds, e.g. by industry (battery and product manufacturers), environmental NGOs and consumer organisations. The importance of all dimensions of sustainability (environmental, social, economic) for the battery policy subsystem (I1) suggests that deep core beliefs could be distinguished between material (economic) and purposive (environmental, social) beliefs. Material beliefs could be competitiveness and maintaining jobs in an open EU market (I1, I3) (industry coalition) versus purposive beliefs such as the protecting the environment and human health (I4) (NGO coalition). In terms of translating core beliefs into policy core beliefs, interviewees indicated that the industry coalition tended to minimise the level of regulation, while the NGO coalition tended to call for more stringent obligations (I4, I6). One interviewee from a national consumer organisation (I6) noted that there was little coordination between environmental and consumer organisations, despite their presumably shared beliefs. However, it is expected that consumer organisations at EU level will have a greater degree of interaction in the policy subsystem.

While there has been a broadening of the range of stakeholders involved, in particular due to the increased involvement of the automotive sector, coalitions appear to be relatively stable during the legislative process (I4, I5). However, we see changes over time in the discussion of circular economy aspects, as exemplified by the development around three measures: ban on materials, recycling efficiency and recycled content (see Table 9 for legislative texts).

For a cadmium ban, there was a clear division between industry and NGOs. Industry stakeholders even went so far as to produce a cartoon depicting a bleak world after a ban on nickel-cadmium batteries and distribute it to Members of the European Parliament before the vote (Häge 2013). When analysing the opinions of associations and NGOs during the public consultation in 2003, a clear distinction between the presumed coalitions can be seen (Figure 19 a)). The exception is one product manufacturer that does not oppose the ban on cadmium.

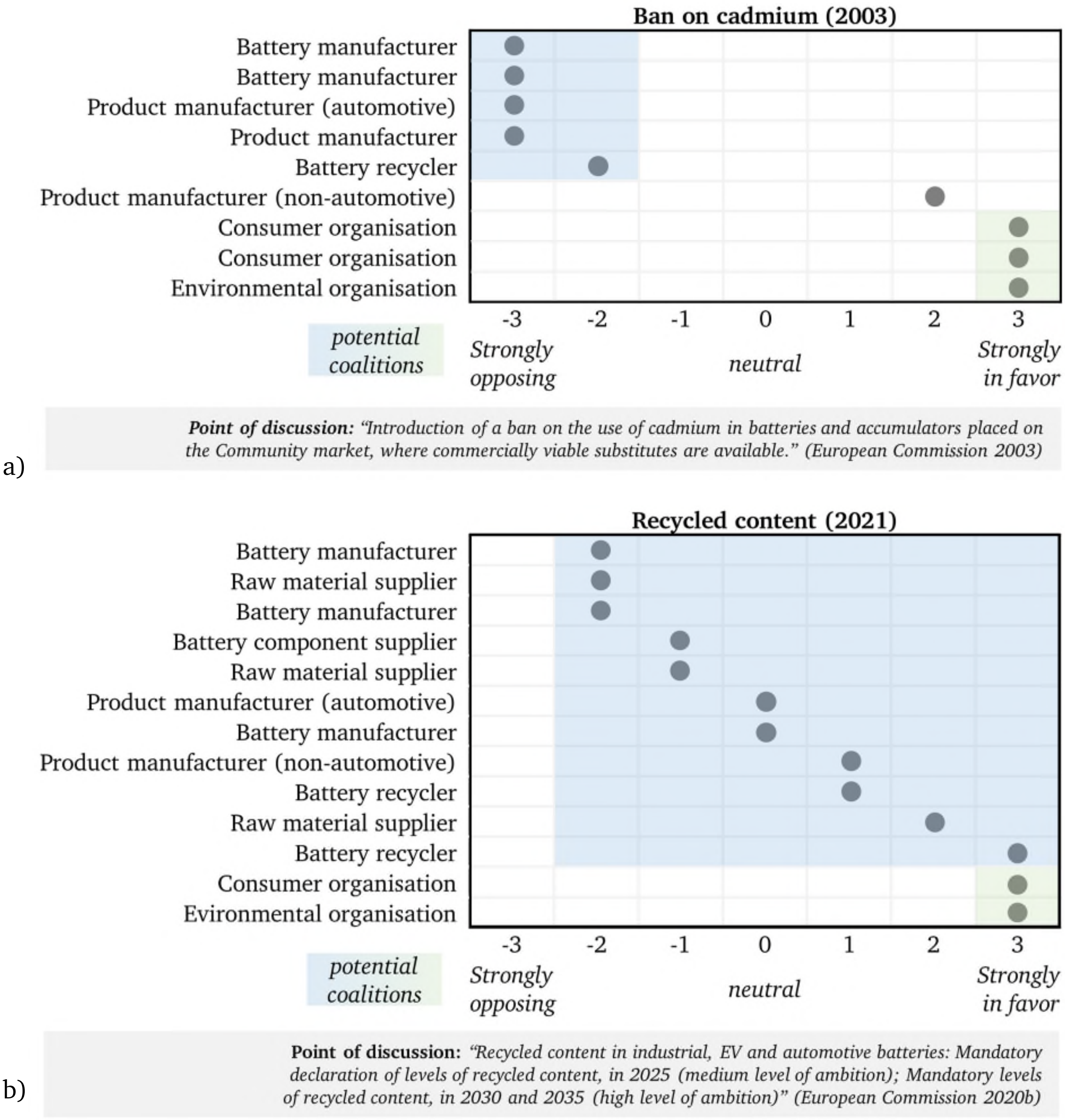


Figure 19: Stakeholder positions on circular economy aspects. a) ban on cadmium in the 2003 public consultation and b) recycled content in the 2021 public consultation.

The extent to which different coalitions influenced the policy-making approach is difficult to assess, but the policy outcome can provide an indication. While the European Parliament initially adopted an approach in favour of a ban on cadmium, this was dropped from the proposal after a second reading (Häge 2013).

In the debate on recycled content, changes are evident (Figure 19 b)). Firstly, there has been an increase in the number of stakeholders that have taken a position on this issue. As explained in 3.4.2, there is a higher proportion of automotive stakeholders involved and an increase in the number of associations. Secondly, similar to the discussion on the cadmium ban, there seems to be a clear purposive-driven coalition of environmental and consumer organisations. They are unambiguously in favour of strict and ambitious recycled content requirements. Thirdly, the opinions of industry stakeholders are more mixed, ranging from opposition to support for the proposed recycled content measure and even for a mandatory character, with no clear differentiation between industry stakeholder groups. Only battery recyclers tend to be generally supportive - not surprising given that the recycled content requirement will increase the demand for, and presumably the price of, recycled materials.

Recycling efficiency is an example of a circular economy measure that was included in the discussion leading to the 2006 legislation and in the current legislative proposal. It should be noted that the measure itself has changed significantly. From a recycling efficiency target per total battery weight (2006) to material-specific targets in the current proposal. Furthermore, the introduction of a new measure (such as recycling efficiency in 2006) is different from increasing the level of ambition of an existing obligation. Nevertheless, the analysis of stakeholder positions shows that in both the 2003 and 2021 consultation rounds the issue was less controversial, with less pronounced coalitions and less clear commitment against the measure. Stakeholder comments showed that the measures themselves were not questioned as much as the method of calculating them. What the analysis of recycling efficiency shows well is that the stakeholder pool has changed. Many of the stakeholders who participated in 2003 did not leave any comments in 2021 and many new stakeholders entered the scene. A graph showing this development can be found in the annex.

It was reported that cross- and inter-coalition interaction have increased over the years (I1, I3). This is evidenced by an increase in the number of joint statements issued by several actors. One interviewee also reported cooperation between environmental and human rights organisations and industry members (mainly battery companies) (I4). Evidence of this can be seen in an open letter from industry representatives (Northvolt et al. 2021) and NGOs (European Environmental Bureau 2021) with similar or identical wording warning the European Commission against unnecessary delays to the new battery legislation in 2021.

3.4.4. External events

Several externalities have affected the policy subsystem. First and foremost is the change in the battery market, which has changed the actors involved, as we have seen in 3.4.2. The underlying effect for this market change can be seen in the EU goal of carbon neutrality, which is embodied in a shift away from fossil fuel road transport towards electric mobility (I2, I6). The importance of batteries for the transport sector and the EU economy as a whole has suddenly increased dramatically.

Although it happened after the publication of the current legislative proposal, the war in Ukraine was identified as another potential external factor that increased the focus on raw material dependency throughout the policy-making process of the current battery regulation (I2), a factor particularly relevant for lithium-ion batteries, which are dependent on critical raw materials such as lithium and cobalt (I1). The current crisis may have reinforced the role of import dependency which was already high on the political agenda, as evidenced by the update of the Circular Economy Action Plan in 2020, which identifies batteries as one of the key product value chains.

The media can be seen as another external factor influencing the subsystem. There has been a lot of public attention on the material sourcing and social concerns along the supply chain (I2), making it important for the regulation to address the sustainability issues of batteries (I2).

It could be argued that due to the increasing politicisation of many areas related to batteries, and also due to technological change, this area has received more attention (moving from a low to a high-salience policy issue), potentially attracting new actors who bring new knowledge that can influence existing beliefs.

3.5. Discussion and conclusion

A strong coverage of circular economy aspects in the EU battery legislation was found, which goes beyond other EU product regulations and has increased over time. The issue of resource efficiency and circular economy aspects in EU product regulation has been studied by a number of other researchers and the evolution towards a stronger integration of circular economy aspects in addition to mere energy efficiency aspects has been clearly shown (Bundgaard et al. 2017; Barkhausen et al. 2022). What is interesting about the analysis of EU battery legislation is that it goes beyond the circular economy policy measures applied to the 27 product groups regulated under the Ecodesign Directive. Few publications were found that focus specifically on the development of battery policy in the EU, and none that specifically analyse the drivers of policy change or use the advocacy coalition framework. Instead, publications often focus on the impact of proposed legislation (Giosuè et al. 2021; Bobba et al. 2019; Malinauskaite et al. 2021) and those that consider the internal dynamics of policy change were published before the proposal for a new regulation was published (Häge 2013). It can be assumed that the novelty of the new Battery Regulation (provisional agreement between the European Council and Parliament on the proposal was reached in December 2022) is the reason for the lack of comparative literature.

The subsystem of battery policy is a multifaceted domain, not only belonging to the field of environmental policy, but also increasingly becoming an integral part of the EU's industrial strategy due to external factors such as the transition of the transport sector towards electric mobility. The results indicate a differentiation between a material-driven industry coalition and a purposive-driven NGO coalition, as well as less clear fronts on secondary aspects over time and a diversification of stakeholders in the legislative process related to proposal 2020/0353 (COD) (mainly increased participation of the automotive industry). The growing importance of the automotive sector for battery technologies is also supported by a review of publications in the field of batteries, where a strong focus on traction batteries can be observed (Malinauskaite et al. 2021; Baars et al. 2020).

When differentiating between actor groups, clear coalitions on the ban on cadmium were found, with strong opposition from industry actors, which is likely to have influenced the failure to include a ban in the 2006 legislation. Coalitions on recycled content and recycling efficiency are less clear, with mixed positions among industry stakeholders. While the controversial nature of an issue is a necessary condition for the emergence of opposing coalitions, the results also suggest that the industry coalition evolved through new actors and external events that brought positions much closer to those of the NGOs. It could be argued that circular economy policies have become the common denominator, combining economic, environmental and social interests. Circular economy measures such as requirements on recycled content and recycling efficiency requirements, promise to reduce the environmental impact of material production, the social concerns associated with raw material extraction, but also the EU's import dependency with far-reaching economic implications.

Thus, this paper shows an increased cross-coalition interaction and a general convergence of positions on circular economy issues. Circular economy aspects can be seen as a common denominator for environmental concerns (climate neutrality of the transport sector) and economic interests (reduced material import dependency).

A limitation is the small sample size of interviewees, which reduces the generalisability of the findings. In addition, it was difficult to find information on the policy-making process in the early 2000s, when information flows were less digitised.

Applying the ACF to this policy sub-system is a step towards deepening the understanding of policy change in highly technical and dynamic areas - looking beyond mere policy inputs and outputs. As the provisional agreement between the European Council and Parliament was reached in late 2022, it will be interesting to follow the publication of the legislation in its final version in light of the findings.

3.6. Supplementary information

A table with the background of interview partners' organisation and additional figures can be found online at:
https://www.sciencedirect.com/science/article/pii/S1364032123005221#appsec1

4. Combinations of material flow analysis and life cycle assessment and their applicability to assess circular economy requirements in EU product regulations. A systematic literature review

This chapter was published in March 2023 in the journal of Cleaner Production. Table 10 provides more information about the publication.

Table 10 Publication information of Barkhausen et al. (2023b).

Title	Combinations of material flow analysis and life cycle assessment and their applicability to assess circular economy requirements in EU product regulations. A systematic literature review
Authors	Robin Barkhausen, Leon Rostek, Zoe Chunyu Miao, Vanessa Zeller
Publication date	31.03.2023
Journal	Journal of Cleaner Production
DOI	https://doi.org/10.1016/j.jclepro.2023.137017
Author contributions according to the Contributor Roles Taxonomy (CRediT 2023)	Robin Barkhausen: Conceptualization, Formal Analysis, Investigation, Visualization, Writing-Original Draft, Writing-Review & Editing, Visualization Leon Rostek: Conceptualization, Writing-Original Draft, Writing-Review & Editing Zoe Chunyu Miao: Conceptualization, Writing-Original Draft, Writing-Review & Editing Vanessa Zeller: Supervision, Writing-Review & Editing

4.1. Introduction

In its Green Deal, the European Commission set itself the overarching goal of reaching carbon neutrality by 2050 and decoupling economic growth from resource consumption (European Commission 2019c), a goal where the transition to a circular economy is expected to play a crucial role (European Commission 2020e). Increasing consumption is responsible for many environmental problems, and one of the ways to reduce the high environmental impacts associated with products (Sala et al. 2019) is through stringent policy. For energy-related products, the EU Ecodesign Directive offers a framework for setting ecodesign requirements, and circular economy-related requirements have increased in recent years (Barkhausen et al. 2022). Different approaches to assess future product policy impacts exist, including MFA and LCA. Both are independent methodologies, but with clear overlaps. While LCA inherently includes an assessment of the material flows during the life cycle inventory, the presented literature review focuses on studies going beyond this classical practice. When combined, these two methodologies are assumed to be suitable for evaluating the long-term environmental impacts of circular economy policies (Elia et al. 2017; Merli et al. 2018; Anandh et al. 2021), but a concise overview of how they are combined is to-date missing.

The following subchapters provide further information on MFA and LCA (4.1.1), an overview of MFA and LCA usage in EU policy and specifically in the Ecodesign Directive context (4.1.2), and more information about the motivation, objective and contribution of our systematic literature review (4.1.3).

4.1.1. Material flow analysis and life cycle assessment

In response to the need for extensive evidence-based quantitative information regarding the use structure, losses and circularity of material systems (Graedel 2019), MFA has been developed as a methodology for quantifying the stocks and flows of materials within a specified system. Systems are defined by spatial and temporal boundaries and by the type of material, which ranges from single substances to complex physical goods. The main principle of MFA is the conservation of mass, which makes it possible to apply mass balance to any process and stock of the system by accounting for inflows and imports with outflows and exports (Brunner, Rechberger 2017).

In static MFAs, the mass balance is performed within one time increment, while dynamic MFAs track changes over multiple time intervals. MFA models can be stock-driven, where stocks and their material composition are exogenous input parameters while inflows are calculated endogenously, or inflow-driven, where it is the other way round (Wiedenhofer et al. 2019). In both cases, the mass balance enables the analysis of stocks and flows of special interest but unknown quantities due to inaccessible or not practicable physical accounting. There is no international norm or official standardisation for MFA, but there are some national norms, such as a norm for MFA in Austria (ÖNorm S 2096). The absence of a standardised MFA procedure has the advantage of high flexibility in each case-specific application, but it can impede the comparability and interpretation of results. Therefore, extensive documentation of each methodological procedure is indispensable for any MFA study to achieve transparency and reproducibility. Brunner, Rechberger (2017) proposed a generic standard iterative procedure for MFA with four steps: Problem definition, system definition (determination of system boundaries and relevant materials and processes), determination of material flows and stocks (collection of input data, balancing of materials, consistency checks and uncertainty evaluation), and illustration and interpretation of results.

MFA studies have a wide variety of goals and objectives, ranging from the micro to the macro level. The spatial scope comprises single processes, companies or cities as well as regions, countries or even the whole globe. The period of time can cover historical and recent flows in retrospective studies or future flows in prospective analyses combining MFA with scenario analysis (Baars et al. 2022).

MFA simulations can be conducted in universal environments like programming languages (e.g. Python, R), system dynamics software (e.g. Vensim) or spreadsheet programs (e.g. Excel). In addition to these universal environments, there are also MFA-specific frameworks such as STAN (Technische Universität Wien 2012) or ODYM (Pauliuk and Heeren 2020). Exogenous variables such as data and additional assumptions serve as the input for any MFA model. Common data sources include statistics or surveys from governmental agencies (e.g. geological surveys), intergovernmental organisations (e.g. UN), industry associations and private companies or consultancies (c.f. (Cullen et al. 2012; Rostek et al. 2022; Bertram et al. 2017)). Uncertainties embedded in exogenous parameters and methodology are unavoidable and it is essential to examine these using uncertainty evaluation and sensitivity analysis as an integral step within MFA.

Since the early 1990s, MFA has been applied in various domains at regional, country and global levels. Socio-economic metabolism and metal cycles comprise an early and ongoing objective for MFA studies (Graedel 2019), while additional studies on construction minerals and specific products have emerged more recently (Baars et al. 2022). MFA has become an established methodology in resource management, environmental management, and waste management (Brunner, Rechberger 2017), and monitoring and simulating the circular economy has become an additional important field of application for MFA (Jacobi et al. 2018; Gao et al. 2020).

With regards to environmental impact assessment, the forerunner of LCA can be dated back to the 1970s when different methods were used (Klöpffer 2014). After decades of development, LCA has now become a standardised methodology under International Organization for Standardization (ISO) 14040/44 (ISO 14040:2006; ISO 14044:2006) to assess the environmental impact associated with a product or service by addressing the entire life cycle from raw material extraction to end-of-life management. An LCA study comprises four phases: goal and scope definition, inventory analysis, impact assessment, and interpretation, and is performed in an iterative manner to optimise processes, identify hotspots and trade-offs in various environmental aspects, and support decision making. The inventory phase analyses the input and output data of the system, which includes the material flows and therein follows the approach of an MFA. Open source and commercial LCA databases are available, providing transparent high-value data sets of regional and international industrial activities, and LCA specific software was developed with a broad range of functions, such as uncertainty analysis.

Besides ISO standards, several deliverables and guidelines have been published to assist practitioners in conducting consistent LCAs. However, the ambiguity in terminology and definitions has been causing methodological inconsistency (Schrijvers et al. 2016; Köhler and Pizzol 2019; Plevin et al. 2014; Zeller et al. 2020; Finkbeiner 2021; Heijungs et al. 2021; Suh and Yang 2014).

In LCA, a case-specific system boundary needs to be defined based on a set of criteria (ISO 14040:2006; ISO 14044:2006) and for many aspects, such as spatial and temporal (Tillman et al. 1994). The geographical boundary not only represents the production site and supply chain, but also indicates the sensitivity of the local receiving environment (Tillman et al. 1994). The temporal boundary of a study can be differentiated into many types: In their review, (Lueddeckens et al. 2020) identified three time horizons (one defined in goal and scope, one for the inventory, and one for the impact assessment). When considering the time horizon in life cycle inventory, time-dependent or dynamic LCA can then be understood as integrating parameters such as supply delay to achieve higher granularity of the system over time, either over short-term technology optimisations or long-term operational and environmental changes (Tiruta-Barna et al. 2016).

Specific LCA approaches are applied for particular purposes. For example, when assessing the futuristic state of emerging technologies, prospective LCA, a synonym for anticipatory (Wender et al. 2014) or ex-ante LCA (Cucurachi et al. 2018) is applied with adaptations in the methodological choices to reflect the research goal (Arvidsson et al. 2018).

LCA methodology and life cycle thinking are also applied to policy design and implementation. Despite criticism (Finkbeiner 2014; Pedersen and Remmen 2022), the Product Environmental Footprint, as a harmonized methodology of LCA, was developed under the initiative of The Single Market for Green Products (European Commission 2012b).

In addition to the information provided above, a table summarising the characteristics of MFA and LCA can be found in the supplementary information (accessible online, see 4.6).

An example of the use of life cycle thinking and material flow based approaches in policy is the development of product regulations under the EU Ecodesign Directive, which is discussed in more detail in the following section.

4.1.2. Application of material flow analysis and life cycle assessment to evaluating circular economy requirements using the example of the EU Ecodesign Directive

The EU Ecodesign Directive (Directive 2009/125/EC and prior Directive 2005/32/EC) aims to enhance the environmental performance of energy-using and energy-related products by imposing ecodesign requirements, including informational and performance requirements on energy efficiency. Via specifying these measures in so-called implementing measures, EU regulates products sold in the EU market (Barkhausen et al. 2022). Prior to establishing these requirements, the potential environmental impact of the requirements shall be evaluated.

Within the context of the Ecodesign Directive, there is a structured approach to analyse the environmental improvement potentials (policy impact assessment) of product groups using the MEErP (Kemna et al. 2011). Currently, at its core, the MEErP comprises a techno-economic-environmental assessment that evaluates life cycle costs and the environmental impacts of representative products and design options. The EcoReport tool has been developed for the environmental impact assessment. This is a public Excel-based tool containing a database of unit impact indicators for close to 100 materials and processes, to statically evaluate selected and weighted environmental impact categories based mainly on the bill-of-materials of the final product.

In the scenario analysis phase of the MEErP, the impacts of product design options on the market are assessed using a basic stock model approach (including annual sales, stocks, and average unitary impacts of the most significant environmental impacts). Furthermore, the MEErP proposes to consider changes for the global warming impact of the electricity mix when conducting the scenario analysis (Kemna et al. 2011).

In the past, impact assessments under the Ecodesign Directive framework focused mostly on energy consumption during the use phase as the most important environmental impact category and placed less emphasis on aspects of material efficiency (Bundgaard et al. 2017), but especially since 2019 there has been a marked increase in the number of circular economy related measures (Barkhausen et al. 2022). Furthermore, the EU recently published its proposal for Ecodesign for Sustainable Products Regulation, which will expand the scope of products beyond energy-related one's (European Commission 2022c). As the MEErP mainly uses a static environmental assessment focussing on the environmental impacts of material and energy consumption in the use phase, it may come to its limits with the new product groups and when assessing the long-term implications of circular economy policies on a wide range of environmental impact categories. To address these shortcomings, the MEErP is currently undergoing a revision process to ensure that it is still fit for its purpose. Within this revision the systematic inclusion of material efficiency aspects in the modelling is one of the central goals (European Commission 2021c).

According to the current draft of the updated version, there will be the possibility to use dynamic stock models with lifetime distribution functions (allowing to e.g. consider improvements in product durability). The EcoReport tool is expected to be aligned to the impact categories of the Product Environmental Footprint, and the modelling of product end-of-life phases (e.g. recyclability or recycled content) will adopt the allocation methods of the Circular Footprint Formula, where the environmental burdens and credits of end-of-life operation are allocated between supplier and user of recycled material (Gama Caldas et al. 2021).

The proposed changes to the MEErP and the EcoReport tool would improve the inclusion of circular economy aspects when assessing the products in the preparatory studies, but it remains to be seen how well the new approach will allow dynamic analyses and how well it is suited to modelling the impacts of circular economy measures on a system level (going beyond simply comparing product variations). The current approach under the MEErP can be described as a simplified LCA that is coupled with a stock modelling approach to assess the environmental benefits of a market roll-out of single product improvements, following the premise of practicality to offer a widely applicable approach with sufficient but not excessive level of detail. By analysing the existing combinations of MFA and LCA, we want to assess whether there is an alternative or complementary approach for modelling circular economy aspects in product policy impact assessments using the example of the Ecodesign Directive.

4.1.3. Motivation, objective and contribution

As shown in section 4.1.1, MFA and LCA have distinct characteristics. LCA shows a high potential to assess the environmental performance of circular economy strategies, but is mainly applied at product level (even though results can be scaled up to system level) (Corona et al. 2019). In contrast, MFA is well suited for macro scale system analysis with different scenarios over long periods of time, but does not account for the related environmental impact (Corona et al. 2019; Elia et al. 2017). It is therefore of scientific interest to examine the possibilities of combining the two approaches, although this is not a trivial task. MFA is a method with a non-standardised procedure, while LCA is conceptually more defined through the ISO 14040 and ISO 14044, but has many different adaptations and interpretations (e.g. EU Product Environmental Footprint) (EC-JRC 2010).

Several review studies have pinpointed the potential and need for further research on combining MFA and LCA (Withanage and Habib 2021; Islam and Huda 2019; Sakai et al. 2017; Perminova et al. 2016). Baars et al. published a semi-structured literature review (using citation searching) of 33 studies on improving prospective MFA as a decision support tool by combining it with other methods including LCA as a so-called “integrated MFA” (Baars et al. 2022). Due to the large number of assessed methods, the review has a wide scope, and does not go into detail for each method combination. Therefore, it is of interest to extend their research, zoom in, and conduct an entire literature review on the combination of MFA and LCA and its applicability to the circular economy policy evaluation based on the example of the EU Ecodesign Directive impact assessment. To the best of our knowledge, no previous study has systematically and exclusively examined this combination. This paper reviews the research that has been conducted on the combination of MFA and LCA with two main objectives: First, to examine and characterise the existing types of combinations of MFA and LCA, and second, to identify how different forms of combination can be applied to the circular economy policy-making process and specifically to the impact assessment of the EU Ecodesign Directive. In doing so, we contribute to research by providing an overview of the characteristics and a systematisation of methodological variations when combining MFA and LCA in future-oriented studies. We also add value for practitioners by assessing how this combination can support decision making.

4.2. Method

A systematic literature review was performed following the procedure proposed in the PRISMA statement that promotes higher transparency and completeness in review studies (Page et al. 2021). The search query was conducted on July 7th 2022 in the scientific databases Scopus and Web of Science, searching for publications that include the term "life cycle assessment" or "life cycle analysis" and "material flow analysis" or "substance flow analysis" (or the abbreviations "MFA" and "LCA") in either title, abstract or keywords. The search query was deliberately open and did not include a sector or field of application in order to obtain a large number of results and only then to assess the applicability of the method to certain sectors or applications. Only peer-reviewed articles (excluding reviews, conference papers, book chapters etc.) were included; no limit was set on the year of publication. After identifying this initial set of records, the bibliographic information was extracted from Scopus and Web of Science, combined, and cleaned for duplicates and incomplete entries (see flow diagram in Figure 20). For the remaining articles, the title and abstract were screened and those excluded (sequentially), that first, did not utilise a combination of LCA and MFA, second, did not apply the method to a case study (merely theoretical framework), or third, did not perform a future-oriented study. The combination of MFA and LCA is hereafter defined as studies whose modelling uses both MFA and LCA, as opposed to studies that

- use only one of the two methodologies,
- compare both approaches without modelling, or
- do not have modelling interlinkages between the two methodologies (e.g., different study object).

Theoretical frameworks that do not include case studies were excluded, because they are the clear minority and, depending on the level of detail, can be difficult to compare to studies that include case study applications. For example, they do not normally define a spatial or temporal scope of analysis.

According to the temporal dimension of the analysis, the publications can be grouped in retrospective or prospective. Retrospective studies are used to assess past policies and historical developments, but normally do not consider future developments. Retrospective studies are not deemed appropriate for ex-ante assessments of the impact of policies on the market. Therefore, and to limit the scope, we only included prospective studies, as these are considered directly applicable to the policy impact assessment process that deals with scenario-based evaluations of future development. In our study we define prospective studies as those that are future-oriented, by modelling future material flows and sometimes by considering future points in time in the life cycle inventory (foreground and background data). Within the prospective studies (which are all included in the analysis), we furthermore differentiate according to the temporal modelling approach between static and dynamic MFA (assessing one time increment compared to tracing changes in material flows over multiple time steps) and static and dynamic life cycle inventory data (keeping the inventory data constant over the study period or varying it and considering changes over time). It was not always easy to identify such prospective studies. Often, it is not clear from the title and abstract alone whether a study is retrospective or prospective. Some articles track material flows across the product lifetime without indicating the time horizon or year of analysis (e.g. Aryapratama and Pauliuk (2022), van Stijn et al. (2022)). If a present day analysis calculates the results over a 50 year span of product lifetime, it can be argued that the study is to a certain degree future-oriented. For such studies we assumed that they are prospective and included them in the review. Filtering studies according to their prospective character made a case-by-case evaluation necessary, and includes a risk of bias due to the qualitative nature of the assessment. Limiting the scope to prospective studies might exclude studies that are non-prospective but methodologically relevant, but we made a deliberate decision to limit the scope at this point.

To expand the spectrum, the final papers (particularly introduction and methodology section) were reviewed for additional citations that met the search criteria.

Full records were retrieved and reviewed according to the modelling approach (or type of combination). Furthermore studies were assessed according to the categories spatial scope, object of study, temporal scope, data and software usage, and by assessing the link between the study and the circular economy policy context and its applicability to the EU Ecodesign Directive impact assessment process.

4.3. Results

Of the 767 identified records (of which three papers were identified via citation searching), 43 were retrieved for the review process (see Figure 20). The following subchapters summarise the findings according to the bibliometric information of the articles, object of study, the identified types of combinations, variations in spatial and temporal scope, the data and software usage, and their applicability for the assessment of circular economy policy strategies.

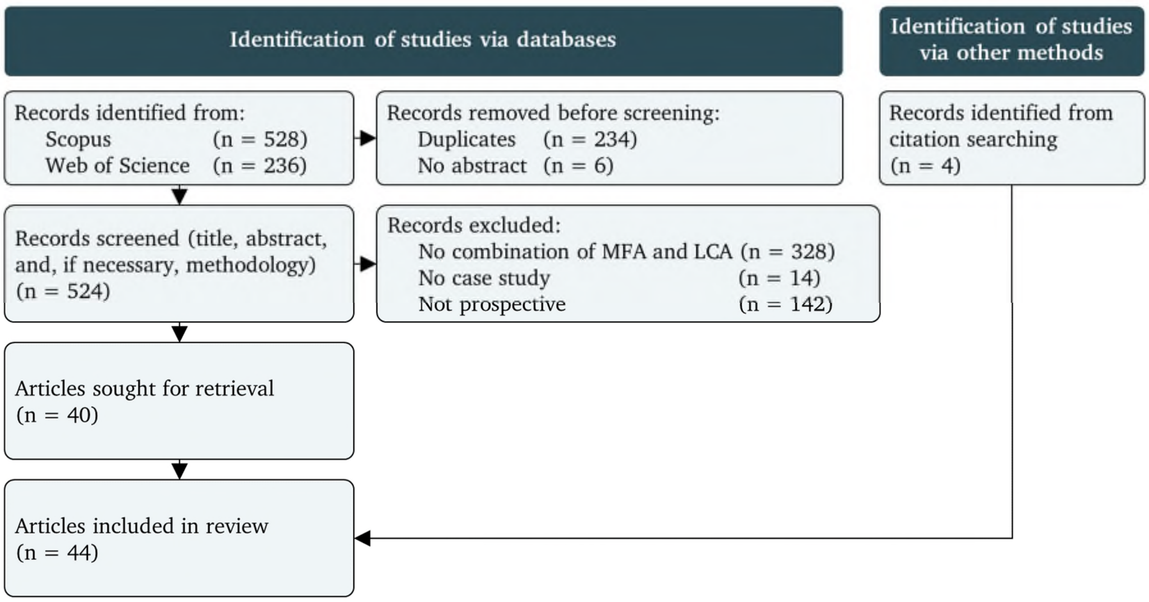


Figure 20 Flow diagram of systematic review.

4.3.1. Bibliometric and spatial analysis and object of study

The earliest identified prospective study that applied the combination of MFA and LCA was published in 2003 (Yokota et al. 2003), but the large majority (80%) of articles were published in the three years prior to our analysis 2020 (11), 2021 (11), and 2022 (13), indicating that the combination of MFA and LCA has only recently received wider attention in the research field. Additional figures showing e.g. the number of publications can be found in supplementary information (accessible online, see 4.6). We found articles being published in 18 different journals, with almost half of the publications in three journals, namely the Journal of Industrial Ecology (11), Resources, Conservation and Recycling (7), and Journal of Cleaner Production (4) (Figure B.3 in supplementary information, accessible online, see 4.6). Prospective articles applying the combination of MFA and LCA were published by a total of 161 authors, of which 22 published more than one paper, and only five more than two.

Figure 21 shows that, for the majority of analysed studies, the spatial scope comprises one entire country, such as China, where most country-scale assessments were conducted (Zhang et al. 2022; Chu et al. 2022; Di Dong et al. 2022; Ryter et al. 2021; Liu et al. 2020a; Liu et al. 2020b), or Japan (Morimoto et al. 2020; Kayo et al. 2019; Yokota et al. 2003). Within Europe, Norway (Lausselet et al. 2021; Sadeleer et al. 2020; Pauliuk et al. 2013; Sandberg and Brattebø 2012; Venkatesh et al. 2009) and Switzerland (Wiprächtiger et al. 2022; Wiprächtiger et al. 2020; Heeren and Hellweg 2019; Mehr et al. 2018) are the most frequently assessed study areas. The spatial scope is influenced by the location of the authors or research groups, and thus all the studies in the Norwegian context are (at least partly) driven by researchers affiliated with the Industrial Ecology Programme and the Department of Hydraulic and Environmental Engineering at the University of Science and Technology in Trondheim. Similarly, the studies on the Swiss context were conducted by researchers affiliated with the Institute of Environmental Engineering at ETH Zurich.

Second most common are region- or city-based assessments, especially for studies on the building sector or on waste management, e.g. a study on residential building stocks in Leiden (Netherlands) (Yang et al. 2022b), on the built stock in Melbourne (Australia) (Stephan et al. 2022), on solid waste management in Davao City (Philippines) (Olalo et al. 2022), or on the impacts of incineration phase-out on waste management in Madrid (Spain) (Istrate et al. 2021). Less frequent are studies on a global scale, e.g. (D'Amico et al. 2021; Pauliuk et al. 2021), and only one study was identified that focused on a production process (Ardolino et al. 2020).

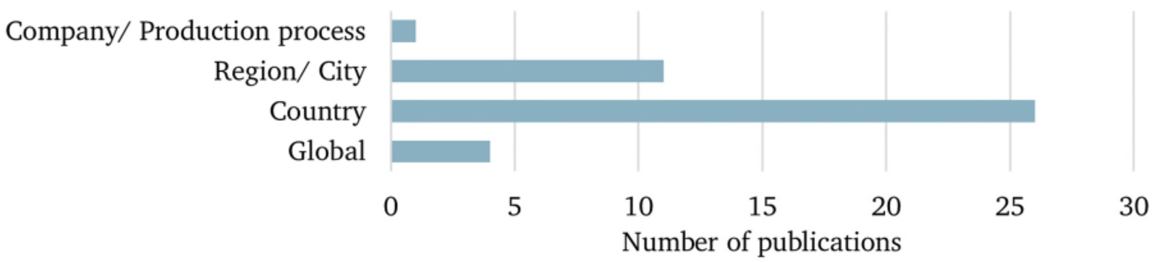


Figure 21 Number of prospective studies applying the combination of material flow analysis and life cycle assessment (only those studies included where spatial scope was clearly defined). Studies on the EU are counted under the category country.

The study object could most often be allocated to the construction or manufacturing sector, followed by the waste management sector. Within the construction sector, most studies deal with building stocks (e.g. Yang et al. (2022a), Göswein et al. (2021), Lausselet et al. (2021)), and some with building components or materials (e.g. Malabi Eberhardt et al. (2021), Heeren and Hellweg (2019)). For manufacturing, a wide range of products are assessed by the studies, including traction batteries (Bobba et al. 2020; Nguyen-Tien et al. 2022), refrigerators (Velásquez-Rodríguez et al. 2021; Glöser-Chahoud et al. 2021), washing machines (Boldoczki et al. 2021), mobile phones (Glöser-Chahoud et al. 2021), and air conditioners (Yokota et al. 2003; Zhang et al. 2020). All of these products are either energy-consuming or energy-related, and thus fall under the scope of the Ecodesign Directive (even though batteries are regulated under a separate Directive (2006/66/EC) in the EU context). Interestingly, among studies that trace the flows of a single material across sectors, there is a strong focus on copper. Six studies were conducted on copper (e.g. Zhang et al. (2022), Di Dong et al. (2022)), and only one on a different material, namely steel (Milford et al. 2013).

Further information on the study object and fields of application can be found in supplementary information (accessible online, see 4.6).

4.3.2. Modelling approach

Different ways of combining MFA and LCA are conceivable and our systematic review confirmed that there is no one-fits-all approach, but instead a wide spectrum of combination types.

Based on our analysis, we classified modelling approaches of combined MFA and LCA according to their focus and level of integration into the three broad categories "Systematic Flow Extended LCA", "Environmentally Extended MFA" and "Integrated Modelling" (Figure 22). Needless to say, the boundaries of this classification are open and approaches may have characteristics that could fit into different categories. We propose this classification merely to provide a better understanding of the characteristics and variations within the assessed studies and thereby to facilitate a discussion.

It should be noted that many authors are rather inconsistent when using the term LCA, also referring to it when conducting an environmental assessment that is not entirely in accord with the ISO standard (e.g. coupling unit process data from LCA databases with the results of an MFA). Thus, the term LCA in Figure 22 can denote both a LCA conducted according to the ISO standard as well as different forms of simplified environmental impact assessments based on life cycle thinking (e.g. when items or requirements from the ISO standard are missing or it is unclear whether they have been considered, e.g. no specific notice is made of the four LCA phases or their concluding items, or the functional unit is not clearly defined).

These simplified environmental assessments are exemplified in the category of **Environmentally Extended MFA**, where the MFA results are complemented by a basic environmental assessment. No full LCA is conducted, and only environmental inventory data (or emission factors) are used to add an additional environmental dimension to the results. The clear focus of this approach is on the MFA. Without the LCA (or environmental impact assessment) part, the study would still function, but be less informative. In its simplest form, the MFA delivers the material flows, which are then linked with the background inventory data of LCA databases. The inventory data can refer to, e.g. material-related emissions (Lausselet et al. 2021; Yang et al. 2022b; Chu et al. 2022), energy-induced emissions (such as emissions of the electricity mix) (Glöser-Chahoud et al. 2021; Yang et al. 2022b; D'Amico et al. 2021), or process-induced emissions that contain several material or energy flows (Liu et al. 2020a; Milford et al. 2013). The environmental results often target greenhouse gas emissions and other impact categories are not part of the analysis. Furthermore, such analyses normally use static LCA inventory data without temporal changes (see 4.3.3).

Systemic Flow Extended LCA is similar to the previous category, but focuses on the LCA side. Here, the stand-alone results of the LCA are the primary goal of the study. It should be noted that an LCA according to the ISO standard (ISO 14040:2006) includes the data collection of relevant material and other physical inputs as part of the life cycle inventory. Practically speaking, every standard LCA includes an analysis of the material flows (compiling the inventory), i.e. an MFA. Under this premise, we could consider every LCA which uses dynamic and prospective material flows (e.g. via variable parameters in the modelling process) as a combined MFA and LCA. However, we only look at those studies that provide a clear distinction between the two methodologies. Systemic Flow Extended LCA under our definition is thus characterised by sequential modelling, in which the MFA provides the input for the inventory of the LCA and thereby a more systemic overview of material flows and easier parameterisation. For example, (Aryapratama and Pauliuk 2022) perform an inflow driven dynamic stock modelling of wood use in Indonesia, and use the results as the input for the LCA model. All the research questions are related to environmental impact, so the objective of the study is clearly focused on the LCA part. A reversed but also sequential approach is used by (van Stijn et al. 2022; Malabi Eberhardt et al. 2021). Here, the authors first conduct the LCA to calculate the environmental impacts of different design options (over the lifetime), and then use the inventory data to do a more detailed MFA that provides macroeconomic implications such as the necessary imports and exports.

The highest level of integration is achieved, when the two methodologies are combined in one model, as **Integrated Modelling**. The majority of analysed studies fall under this category. Articles that follow this approach were marked by combined results and/or same system boundaries and/or using the same functional unit or flow object in both MFA and LCA. Without the respective counterpart methodology, the study would not (or less) function, and often the modelling is (at least partly) implemented outside of traditional MFA or LCA software (see 4.3.4 for software usage). Integrated Modelling can be done similar to the Systemic Flow Extended LCA in a sequential manner, where the MFA is conducted and provides the flows (of materials, energy, products etc.) which are fed into the inventory building of the LCA. In contrast to Systemic Flow Extended LCA, the results of the MFA are an integral and independent part of the results and are reported separately from the LCA results. For example, (Velásquez-Rodríguez et al. 2021) first conducted an MFA to estimate the number of end-of-life refrigerators in Colombia, which serves as the input to calculate the impact of different end-of-life routes with LCA. Both the result of the MFA as well as the result of the LCA are presented. Similarly, (Venkatesh et al. 2009) used a sequential modelling approach for wastewater pipeline networks in Oslo, Norway, in which the MFA served as the input for the LCA, but both MFA and LCA deliver independent results.

The more integrated type of modelling intertwines MFA and LCA. There is an identical or clear overlap in the system boundary of MFA and LCA, and/or the final results or goal of the study are a combination of the integrated modelling, based on the results of both MFA and LCA. Some examples are provided hereafter. (Mehr et al. 2018) conducted a study in which they combine MFA with a modular LCA. In their study, a high-level MFA depicted wood use in Switzerland over a 200 year time horizon, and provides input and output flows of the processes (or modules), which are then combined with the LCA in a product-process-matrix. This matrix is then used as the basis for an algebraic optimisation model. Some of the same authors were also part of two later publications (Wiprächtiger et al. 2022; Wiprächtiger et al. 2020) (research group at ETH Zurich, see 4.3.1) that applied a similar matrix based modelling approach, in which the MFA estimated the flows between processes and the LCA the environmental impacts of those processes. The final result, e.g. the optimal wood use strategy (Mehr et al. 2018), was a result of the integrated modelling. Within integrated modelling, the environmental assessment can, but does not have to, extent the scope of processes and material flows. Integrated modelling can be performed with identical system boundaries in a layered approach, in which the study object is disaggregated from e.g. the product to material, energy, and emission level. Each layer provides additional dimensions of information while results are consistent due to the common derivation (Pauliuk and Heeren 2021). E.g. by tracking first the product flows and then disaggregating the information at material level by classifying material intensities for certain types of the assessed product (e.g. (Zhang et al. 2020)). For the building sector, a similar multi-layered approach can be used, in which material intensities are defined for different building archetypes (Lausselet et al. 2021).

Another approach was followed in a study from 2021, where (Boldoczki et al. 2021) applied a combined MFA and LCA approach to assess washing machine reuse in Germany. The MFA is conducted to get the stocks of washing machines for each efficiency class per year. Then those results are combined with a modular LCA, in which each process is modelled separately. Both MFA and LCA have the same system boundary. However, the MFA excludes flows of resources and energy to/from the processes except the flow of washing machines. The results show the evaluation of different reuse targets and represent a symbiosis of the environmental impacts from the LCA and the dynamic tracking of flows (in this case products) over long periods of time derived from the MFA.

While it was not always possible to understand how closely MFA and LCA are integrated in a study, a clear indication of an intertwined modelling is provided if MFA and LCA are modelled combined in only one software application. E.g., (Istrate et al. 2021) build an integrated model using MATLAB, to evaluate the municipal solid waste management of Madrid, Spain (more information on software usage in 4.3.4).

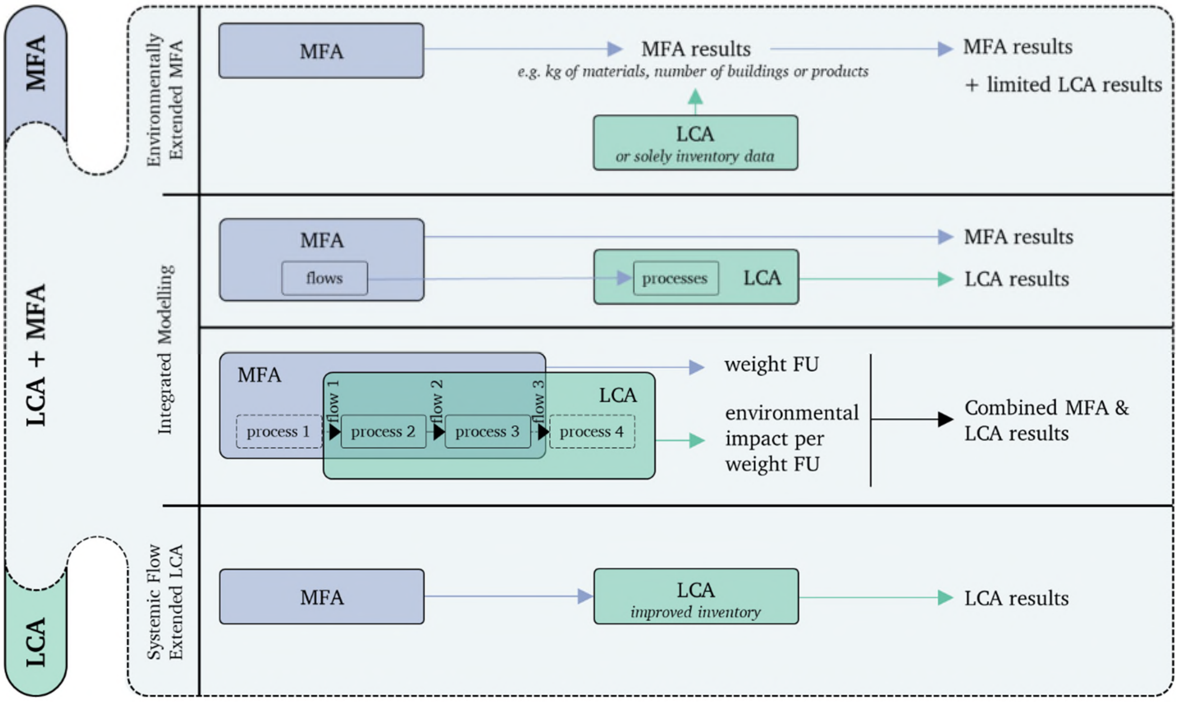


Figure 22 Modelling approaches of prospective studies applying the combination of material flow analysis (MFA) and life cycle assessment (LCA) (referring here to both ISO standard environmental LCA as well as simplified environmental impact assessments based on life cycle thinking). FU = functional unit.

Several publications use multi-method analyses that included MFA and LCA, but additional methods as well, such as economic analysis (Brignon 2021; Liu et al. 2020a; Liu et al. 2020b; Ryter et al. 2021), interviews to validate assumptions (Ryter et al. 2022), mathematical optimisation (Mehr et al. 2018; Olalo et al. 2022), regression analysis (Ciacci et al. 2020), nested systems theory (Stephan et al. 2022), or criticality assessments (Bobba et al. 2020).

4.3.3. Temporal scope

One of our selection criteria was to only consider prospective studies (see 4.2 for definition). On a side note, our analysis showed that most studies who conduct a future-oriented analysis do not use the term "prospective".

Prospective modelling of material flows and life cycle inventory can be done static with discrete time increments (static MFA/ static life cycle inventory data) or continuously over multiple time steps (dynamic MFA/ dynamic life cycle inventory) (see Figure 23).

While it is more common to assess dynamic material flows, there are studies that use static material flow, like (Istrate et al. 2021), who compared the state of the waste management system in Madrid in 2017 to two prospective scenarios for the year 2025. More studies were identified using dynamic material flow while keeping the life cycle inventory data static. (Pauliuk et al. 2013) conducted a study on the Norwegian dwelling stock (until 2060), in which they assume the upstream impacts to be constant over time. While acknowledging that "the electricity mix in particular may change drastically in the future", they calculate with the current energy mix mainly due to uncertainties about future development. (D'Amico et al. 2021) follow a similar argumentation in their study on material substitution in building construction. They assumed static carbon coefficients over time, referring to "equally likely" scenarios for the increase and decrease of global carbon emissions until 2040. In their study on Swiss construction materials with a time horizon until 2055, (Heeren and Hellweg 2019) kept the life cycle inventory data constant over time arguing that technological innovation was outside the study scope.

The most uncommon temporal modelling approach is to keep the material flows static while utilising dynamic life cycle inventory data. Studies using this approach are likely to be LCA-based studies that were not identified in our search query. However, the study of Ardolino et al. (2020) can be understood as a representative of this group. The authors calculated the static material flows for different thermochemical processes for the treatment of municipal solid bio-waste and used scenario-based variations of the electricity mix until 2030 when interpreting the results, while keeping the material flows constant. Finally, the combination of dynamic material flows and dynamic life cycle inventory data continuously models the material flows and the variations of the environmental inventory over time. Studies differ by which part of the life cycle inventory data are dynamic. In the simplest form, only one parameter of the background inventory data is dynamic. Glöser-Chahoud et al. (2021), for example, coupled their MFA of different consumer products with dynamic greenhouse gas emission data of the average European electricity generation over time, and Zhang et al. (2020) analysed air conditioning in a Chinese city, creating scenarios with different annual decrease rates of the power grid emission factor. Environmental inventory data are also modelled dynamically by Di Dong et al. (2022) to assess China's potential for reducing primary copper demand and the associated environmental impacts until 2100. Here, the authors considered changes in the electricity mix (background system) as well as in process efficiencies (foreground system).

The theoretically possible four different approaches of temporal modelling when combining MFA and LCA are illustrated in Figure 23. Theoretically, it is possible to combine static MFA and static life cycle inventory data in a prospective manner, but this would always require at least two discrete time states to allow for comparison. With additional time steps, the approach can soon be classified as dynamic modelling.

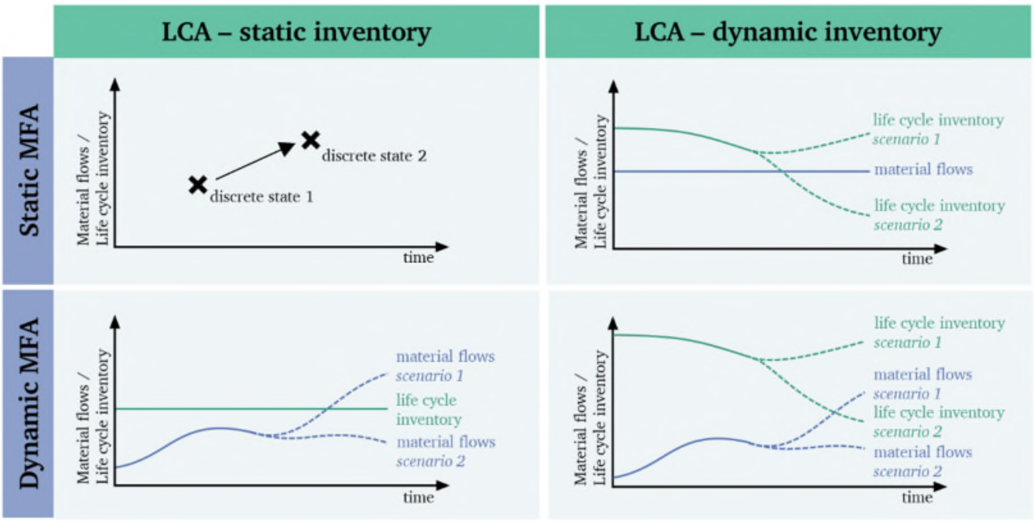


Figure 23 Variations in static and dynamic temporal modelling of prospective studies applying the combination of material flow analysis (MFA) and life cycle assessment (LCA). Life cycle inventory refers to specific (e.g. process or material related) emissions in the background inventory data of the LCA. The curves are only indicative.

Figure 24 shows that the study period of the material flows ranges from comparing individual years to time spans of more than 100 years. There are several studies that (in addition to the prospective modelling) include a retrospective assessment to generate a production-based build-up of stocks or to have a base case and to ground scenario assumptions. Especially when assessing long-lived products, tracing the age of cohorts is decisive for determining the failure probability to estimate end-of-life flows, which constitute the stocks outflow. Velásquez-Rodríguez et al. (2021), for instance, designed their scenarios for the volume of end-of-life refrigerators in Colombia (as well as the lifetime function) based on historical waste collection data, and Liu et al. (2020a) based their estimations of end-of-life vehicles on the retirement of vehicles on the base year 2018.

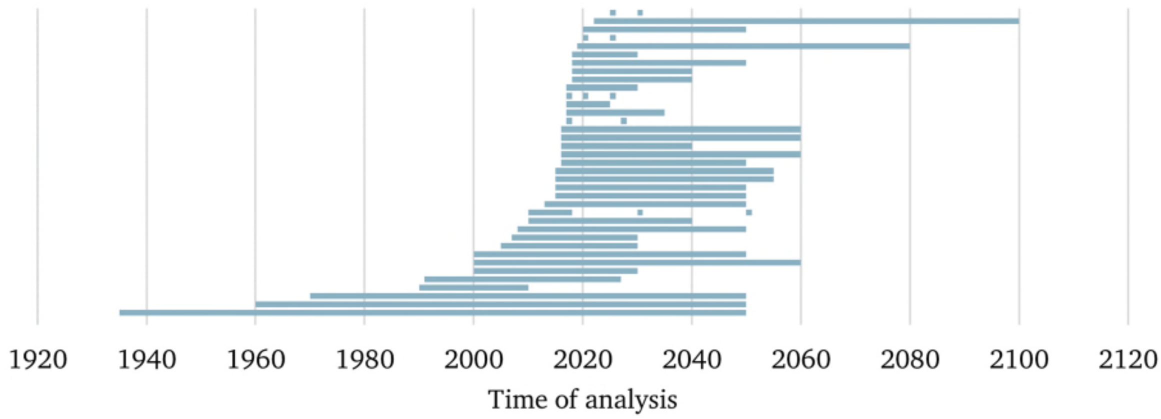


Figure 24 Analysed time period of the material flows in prospective studies applying the combination of material flow analysis and life cycle assessment (excluded in the figure are studies that model scenarios using material/product lifespans without indicating a specific year).

How much the results depend on individual modelling parameters can be quantified using sensitivity analysis. Sensitivity can be defined as the fraction of relative change in the results over the relative change in the input parameter (Sadeleer et al. 2020).

For prospective scenario-based analysis, the results are highly dependent on the modelling assumptions. As expected, we found that the majority of studies (27) use and describe a sensitivity analysis to assess how changes in certain parameters influence the results (such as changes to the electricity mix, or process efficiencies over time), e.g. Boldoczki et al. (2021), Wiprächtiger et al. (2020). Most assessed studies conduct a scenario analysis in which they vary individual model parameters, such as product lifetimes (Glöser-Chahoud et al. 2021), collection or reuse rates (Di Dong et al. 2022), material or design choices (Göswein et al. 2021) or sociodemographic changes such as population growth (Pauliuk et al. 2013).

4.3.4. Data and software usage

In our analysis, software refers to the programming environment in which the analysis was conducted, and data refer to the exogenous modelling inputs. Foreground data (for both MFA and LCA) are normally case specific and can be derived from a multitude of sources, such as field sampling, previous research studies or national statistics. In this study, we mainly focused on the background life cycle inventory data to analyse data usage.

Turning to the software used, we found that around half of the studies do not specify which software was used for the modelling process. Those that do specify the software can be differentiated in LCA-specific software, MFA-specific software, software that allows combined modelling, or combinations thereof. Several studies use LCA-specific software, such as SimaPro (Boldoczki et al. 2021; Ryter et al. 2022), openLCA (van Stijn et al. 2022; Malabi Eberhardt et al. 2021), or Brightway (Wiprächtiger et al. 2020; Wiprächtiger et al. 2022). In contrast to LCA, the wider conceptual flexibility of MFA allows for more variation in modelling software, and thus purely MFA-specific tools, such as STAN (Wiprächtiger et al. 2020), are rarely mentioned in the studies. Instead, it is common to conduct the modelling in a flexible software solution, such as spreadsheets (Mehr et al. 2018; Olalo et al. 2022), MATLAB (Istrate et al. 2021), or Python (Boldoczki et al. 2021; Velásquez-Rodríguez et al. 2021; Aryapratama and Pauliuk 2022). Several studies implement modelling (at least partly) outside of LCA-specific software by utilising modular LCA (Boldoczki et al. 2021; Bobba et al. 2020; Wiprächtiger et al. 2020; Mehr et al. 2018). Modular LCA is characterised as not modelling the entire system, but individual life cycle stages or processes (= modules), and only later combining those modules in a variable manner (Steubing et al. 2016). These modules can have wider system boundaries than single unit LCA processes. However, this requires a sequential approach where the environmental impacts of a module are first modelled in LCA-specific software and are then exported to another software environment to be integrated into the MFA. Using only unit processes extracted from LCA databases may allow modelling in a single-software solution.

Due to its more flexible nature, modular LCA can reflect the "broader level of abstraction often found in MFA models" (Mehr et al. 2018), and is "capable of reducing the effort involved in performing scenario analyses and optimization when several key choices along a product's value chain lead to many alternative life cycles" (Steubing et al. 2016). Technically, LCA-specific software allows the calculation of different alternatives using parameters, but the modelling flexibility for a high number of scenarios with varying multiple parameters over long time scales is limited, and researchers therefore have shifted to other modelling environments.

In terms of data usage, many studies did not specify or did not use data from life cycle inventory databases. Of those that did, the large majority (20 studies) used some version of the Ecoinvent database. Sometimes the Ecoinvent data were complemented, e.g. with other inventory databases such as EPiC (Stephan et al. 2022), reports from intergovernmental organisations such as the Intergovernmental Panel on Climate Change or the International Energy Agency (Ciacci et al. 2020; Olalo et al. 2022), or governmental statistics (Sevigné-Itoiz et al. 2015; Olalo et al. 2022).

4.3.5. Applicability for the assessment of circular economy policy strategies

Before discussing the results in terms of their applicability to the ecodesign impact assessment process, it is interesting to see whether the studies assessed are already being used in the circular economy policy-making process, either by evaluating the effect of potential policies, or by providing policy recommendations based on the study results.

(Di Dong et al. 2022), for example, state several specific policy implications of their results and the analysis by (Ryter et al. 2021) estimates different regional policy change scenarios. Circular economy requirements can be differentiated into the circular economy strategies of reducing, reusing and recycling (3Rs), all of which are present in current Ecodesign regulations (Barkhausen et al., 2022). In our analysis, we identified studies that consider all three strategies (e.g. (Di Dong et al. 2022), but found a clear focus on the assessment of measures related to recycling (38 studies), followed by reuse (16 studies) and reduce (10 studies). Examples of assessing primarily recycling strategies include (Velásquez-Rodríguez et al. 2021), who assess the roll-out of a nation-wide refrigerator-recycling program in Colombia called "Red Verde", and (Sadeleer et al. 2020), who assess different recycling rates of household food waste. Studies evaluating the impacts of reuse strategies include (Boldoczki et al. 2021), who conduct their study on washing machines in Germany to measure "long-term implications of policy decisions", such as the effect of implementing a 5% target for preparation for reuse. The least number of studies assess options related to reducing material consumption (e.g. sufficiency strategies or sharing concepts), and this is often only assessed in addition to measures related to recycling and/or reuse. A study that can be classified as reducing was carried out by (Glöser-Chahoud et al. 2021), who assess the influences of extending product lifetime on the environmental performance of refrigerators and mobile phones. However, when considering lifetime extension due to repair and not design choices, this could be counted as a reuse strategy. Based on our findings in the previous sections, we will now turn to their implications for the ecodesign policy-making process.

4.4. Discussion

Already in 2000, (Haes et al. 2000) stated that MFA and LCA are independent research communities, "separated rather by historical than by methodological lines". Since then, a variety of prospective studies that apply the combination of MFA and LCA have been published, with large variations in the modelling approach used. Our analysis provides an overview of existing approaches, in particular their applicability to product policy impact modelling, and also classifies existing approaches into three distinct categories. Environmentally Extended MFA was found to be a simple and practical approach, because it avoids going through all the formal phases of an LCA study, and can be conducted without LCA expertise. However, this carries the risk of a lack of proper understanding of the environmental data of flows and processes, which might lead to oversimplification and wrong conclusions (and in turn reduce trust in the reliability of environmental assessments). Such analysis should always be based on a thorough understanding of LCA theory and the reliability of the life cycle inventory data. The problem of LCA studies not conforming to the ISO standards, or ambiguities within the standards, has been highlighted before (Schaubroeck et al. 2022; Heijungs et al. 2021; Schrijvers et al. 2016)). MFA, in turn, is a less standardised methodology, and other researchers have pointed out that the definition, including the methods used under the term MFA, are not always clear in the literature (Baars et al. 2022).

Systemic Flow Extended LCA can be a way to enhance the LCA by considering wider system boundaries and systemic interactions of material flows. And Environmentally Extended MFA can be a way to add an environmental layer to physical material flows. A similar concept has been framed by (Pauliuk and Hertwich 2016) as extended dynamic MFA, where MFA is extended with product life cycles in a multi-layered way.

Integrated modelling approaches are most frequent, which simultaneously apply MFA and LCA (or simplified environmental impact assessments based on life cycle thinking) within one combined modelling environment and can provide multi-dimensional results. We found different hybrid software solutions but also a trend towards the use of flexible software solutions. Integrated modelling approaches seem to benefit from conducting the analysis outside of LCA specific software by using e.g. Python instead. This offers higher modelling speed and flexibility to combine changing processes with dynamic flow parameters, and to easily modify the study design. When conducting such a study not using LCA specific software, using modular LCA can help to avoid the modelling constraints of traditional LCA software.

Regarding the spatial scope, our analysis provides detailed insights and shows that studies at the country or regional level are most frequent. This is in line with the observation of (Baars et al. 2022), who used the broader metric of micro and meso level to group integrated MFA studies. The majority of the studies they analysed fall into the meso scope, which includes the country level and local regions such as cities and neighbourhoods. This observation may stem from the characteristics of MFA and LCA regarding spatial scope. LCA is a function-based methodology that considers all the flows and processes required to provide the service or function of interest, therefore normally the geographical scope follows the function. MFA, on the other hand, is often conducted within a clear geographical boundary. In a combined MFA and LCA, the LCA often uses a pre-defined spatial boundary. With regard to the evaluation of circular economy policies, upscaling and the clear geographical boundary are both needed to assess the implications of certain measures for e.g. one defined country.

The temporal dimension was found to be a critical demarcation point between studies. Most of the reviewed studies model the material flows dynamically (e.g. (Velásquez-Rodríguez et al. 2021; Boldoczki et al. 2021; Huang et al. 2020)), but within the LCA part, there are large differences in terms of accounting for changes over time.

If the LCA foreground and background data are static (remain unchanged over time), then it is easier to combine MFA and LCA in a decoupled manner (e.g. (D'Amico et al. 2021; Sadeleer et al. 2020; Heeren and Hellweg 2019)). However, if changes to material flows over time are combined with changes in e.g. process emissions over time, then an integrated MFA and LCA study is needed to evaluate scenario results, as MFA and LCA results are interdependent (e.g. (Ryter et al. 2022; Di Dong et al. 2022; Sevigné-Itoiz et al. 2015)). The tendency to not discuss temporal aspects in the inventory when conducting LCA has been noted by (Resch et al. 2021), who used a dynamic LCA approach to model environmental impacts in the building sector. Considering dynamic environmental impacts such as lower future emissions due to technological progress can provide more multi-faceted results at the cost of greater uncertainty. Furthermore, dynamic (time-dependent) inventory data is commonly not available in standard LCA datasets such as Ecoinvent and thus can require substantial additional data collection efforts and might be prone to higher uncertainties. Data collection efforts are a relevant concern, and the problem of low quality and quantity of inventory data is a known issue also noted by (Islam and Huda 2019). Especially in the field of MFA, there are few inventory databases compared to LCA (Birat 2020), and our results confirm this: where secondary databases are mentioned, the majority are LCA based (mostly some version of Ecoinvent),

Corona et al. (2019) identify MFA and LCA as two of the three "backbone" frameworks for circular economy assessment (input-output analysis being the third). Going beyond this, our results showed that the combination of MFA and LCA is also suitable and commonly used in the context of circular economy policy-making. Combined MFA and LCA approaches are applied across sectors, including energy-consuming and energy-related products that fall under the scope of the Ecodesign Directive.

Several requirements can be defined for assessing the potential impacts of circular economy measures under the EU Ecodesign Directive, all of which can be well covered by combining MFA and LCA (see Table 11): Stock modelling needs to move from the product to the material level, and environmental assessment should cover life cycle stages beyond the use phase to evaluate recycling strategies. The analysis needs to be applicable to energy-using and energy-related products, cover the spatial scope of the EU (and beyond if certain life cycle stages require it) and be prospective to assess scenarios over long time periods.

As our analysis has shown, existing approaches are able to fulfil these requirements. In addition, the use of data and software for impact assessments should be simple and comprehensible and allow the use of the same modelling environment for different product groups. As complexity increases (e.g. due to the consideration of additional life cycle phases, extended lifetimes or the tracking of individual materials), the effort required for data collection increases. We have seen a variety of modelling approaches to combine MFA and LCA, making it difficult to generalise about the complexity of modelling. The approaches are often case study specific and there is a correlation between the level of detail of the analysis and the complexity. In principle, the combination of MFA and LCA provides a toolkit for assessing aspects of the circular economy in the context of ecodesign product regulations that combines the individual strengths of MFA and LCA.

Table 11 Ability of methods (considering current usage) to meet requirements for impact assessments of circular economy requirements under the Ecodesign Directive framework (well (++), moderately (+) or not (-) meeting requirements).

	Requirements for impact assessments in the Ecodesign Directive context	MFA	LCA	Combined MFA+LCA
Circular economy requirements	Tracing individual material flows across product lifetime	++	+	++
	Estimating environmental performance (beyond the use phase)	-	++	++
Spatial scope	Clear geographic boundary (regarding product sales in EU market)	++	+	++
Temporal scope	Dynamic and prospective (scenarios over long periods)	++	+	++
Data and software usage	Flexible and transparent, with readily available software and datasets (currently Excel-based)	+	+	+
Sectors of application	Sector not explicit, energy-using and energy-related products, and beyond under the proposal for Ecodesign for Sustainable Products Regulation	++	++	++

The combination of MFA and LCA is well suited to compare not only the environmental impacts of static product design alternatives, but also to consider the impacts of circular economy measures on product and material flows over time with high granularity. Combined approaches allow optimisation at system level by combining the detailed function-based approach of LCA with the broader MFA perspective. Integrated modelling can increase flexibility and provide multi-dimensional results with high granularity, moving away from the sequential approach in ecodesign of conducting an environmental assessment of the design options with the EcoReport tool and an independent stock model. An adaptation of the current approach is needed to assess the requirements of the circular economy by moving from the product to the material level and focusing more on the life cycle phases beyond the use phase. The diversity of integrated modelling approaches provides the opportunity to meet the changing requirements.

Dynamic LCA inventory is expected to greatly impact long-term scenario results and it seems pertinent to at least consider its usage for certain parameters. There is, however, a trade-off between the level of detail versus modelling complexity and data collection efforts, which could be mitigated to some extent by conducting upfront sensitivity analyses to identify the most relevant study parameters and only keep these dynamic.

It remains to be seen how well the update of the MEErP can deal with the new requirements that are created by the stronger consideration of circular economy requirements in EU product regulations.

A number of important limitations need to be considered. First, our systematic literature review could have missed publications that do not refer to the terms MFA and LCA. Studies that combine MFA with environmental impact assessments without conducting a full LCA might have not used the keyword LCA and consequently could have been missed by our systematic search. Conversely, our findings showed that many studies use the term LCA very liberally, even when it is not apparent that the required steps of the ISO standard have been followed. The spectrum of what is considered LCA varies widely among the studies. Therefore, it must be highlighted that in our classification of combined LCA and MFA approaches, LCA refers to both ISO standard analysis and simplified (life cycle thinking based) environmental assessments. It can only be emphasised that a clear understanding of what constitutes LCA should be a prerequisite for labelling one's analysis LCA. As our analysis has shown, the fields of LCA and MFA are increasingly merging which makes it even more important to be consistent in nomenclature and classifications to avoid misunderstandings.

Secondly, due to the high number of identified sources and manual sorting, studies that meet the selection criteria (combine MFA and LCA, are prospective, and include a case study) could have been falsely sorted out if this information was not clear from their title and abstract. In addition, articles were sorted out that did not meet the selection criteria (e.g. not prospective), but which could have been relevant from a methodological point of view.

Thirdly, the classification of the articles by categories such as spatial scope or sector of application contains ambiguities, as some studies have cross-category characteristics. The corresponding figures should therefore be treated with caution. Our analysis focused on the combination of MFA and LCA, and while we identified several studies that combined these two approaches with additional methodologies such as economic analysis, we did not assess these combinations in detail. Of particular interest could be to investigate the combination of MFA and LCA with input-output analysis (e.g., environmentally extended input-output analysis), for top-down assessments of material flows and environmental impacts across economic sectors. Thinking beyond mere environmental impacts, life cycle sustainability assessment comes into consideration. Especially when evaluating circular economy policies, the social and economic impacts could be considered and included in further analysis.

Researchers should also look at how to improve the flexibility in LCA-specific software or how to develop software that includes modelling interfaces for both MFA and LCA.

4.5. Conclusions

Our systematic review identified 44 scientific publications of prospective studies using the combination of MFA and LCA, with varying levels of integration of the two methodologies, geographical and temporal scope and fields of application. We found sequential modelling approaches that lean either on the LCA side if the inventory modelling in LCA is improved by conducting an upfront MFA, or on the MFA side if the results of the MFA are extended by a basic environmental assessment. However, the majority of studies use integrated modelling where both MFA and LCA are essential for the study's objective and the modelling is often implemented in one software. Most studies are conducted in the European context, and there is a focus on country-scale assessments. The time scale varies greatly from comparing discrete years to continuous modelling of periods longer than 100 years. Most studies use dynamic MFA (material flows change over time) and it is less common to see a dynamic approach in the life cycle inventory. Prospective studies that combine MFA and LCA are applied across sectors, including construction (buildings), manufacturing (especially energy consuming or related products) and waste management. Almost all studies have a clear link to the policy making process, and the combination of methodologies features many characteristics that make it suitable for impact assessment modelling under the EU Ecodesign Directive.

Our research is novel and to the best of our knowledge provides the first in-depth overview of studies combining MFA and LCA and their applicability to the circular economy policy context based on the example of the Ecodesign Directive. We revealed the different combinations used and, based on our results, suggest that researchers in this field should further explore integrated one-software modelling with dynamic material flow and dynamic life cycle inventory.

Our results have important implications for researchers and policy makers. The combination of MFA and LCA shows suitable characteristics to assess the multidimensional domain of circular economy policy, especially in the context of the Ecodesign Directive. It can scale-up the robust environmental assessment of LCA by considering the complex interactions of material flows over long time periods and large spatial scopes of MFA. Compared to the alternative of conducting separate MFA and LCA, integrated approaches create more interdependencies that can reduce flexibility and increase uncertainty due to the complexity of modelling and data collection. In return, the integrated approaches promise increased information value, improved consistency of results, and a more robust basis for decision-making.

4.6. Supplementary information

Supplementary information can be found online at:
https://www.sciencedirect.com/science/article/pii/S0959652623011757#appsec1

5. Modeling stock, material and environmental impacts of circular economy product policies. Trade-offs between early replacement and repair of electric motors

This chapter was published in April 2024 in the journal Resources, Conservation and Recycling. Table 12 provides more information about the publication.

Table 12 Publication information of Barkhausen et al. (2024).

Title	Modeling stock, material and environmental impacts of circular economy product policies. Trade-offs between early replacement and repair of electric motors
Authors	Robin Barkhausen, Antoine Durand, Yan Yi Fong, Vanessa Zeller, Clemens Rohde
Publication date	09.04.2024
Journal	Resources, Conservation and Recycling
DOI	https://doi.org/10.1016/j.resconrec.2024.107600
Author contributions according to the Contributor Roles Taxonomy (CRediT 2023)	Robin Barkhausen: Conceptualization, Data Curation, Formal Analysis, Investigation, Methodology, Software, Validation, Visualization, Writing-Original Draft, Writing-Review & Editing Antoine Durand: Conceptualization, Validation, Methodology, Writing - review and editing Yan Yi Fong: Data Curation, Investigation Vanessa Zeller: Supervision, Writing-Review & Editing Clemens Rohde: Supervision, Writing-Review & Editing

5.1. Introduction

The transition to a more circular economy is a pillar in the EU's strategy for achieving carbon neutrality and decoupling economic growth from resource use, as shown by the inclusion of the second Circular Economy Action Plan as one of the main pillars of the EU Green Deal (European Commission 2019c). Products, and appliances in particular are responsible for significant environmental impacts, leading the EU to trespass its global share of planetary boundaries in impact categories such as climate change and resource use (Sala et al. 2019).

While acknowledging the conceptual ambiguity of the term circular economy (Kirchherr et al. 2017), there is a clear upward trend in circular economy-related policy requirements for energy-related product groups regulated under the Ecodesign Directive, which previously focused on energy efficiency in the use phase (Bundgaard et al. 2017; Mathieux et al. 2020; Mudgal et al. 2013; Polverini 2021; Barkhausen et al. 2022). A similar trend can be noticed in the EU's battery regulation (Barkhausen et al. 2023a).

When drafting new legislation, EU policy makers need sound impact assessments to support their decision making. This is particularly true for circular economy policies, as they are a means to an end and not an end in themselves, making it necessary to assess whether they actually deliver the promised reductions in environmental impacts. Traditionally, ex-ante environmental and economic assessments of policy impacts on energy-related products in the EU have been carried out using a streamlined environmental assessment at the individual product level (EcoReport tool) and a subsequent detached stock model logic (where no tool is provided) based on the structured approach described in the MEErP (Kemna et al. 2011).

With the ongoing shift in EU product regulation from a focus on energy efficiency in the use phase to a stronger consideration of circular economy aspects, new modelling challenges arise (Barkhausen et al. 2022; Dalhammar et al. 2014; Mathieux et al. 2020). The MEErP is currently being revised to better reflect circular economy aspects. As far as is known, the focus is on updating the EcoReport tool with newer data and changes that allow the assessment of circular economy related measures at the level of individual products (European Commission 2021c). The EcoReport Tool will be more closely aligned to the Product Environmental Footprint, which, despite criticism (Finkbeiner 2014; Pedersen and Remmen 2022), provides a detailed and streamlined LCA methodology. No dedicated tool is provided by the EU for the scenario analysis, and so far it is not clear whether an approach or model will be provided to scale up individual product results in their multidimensionality to the stock level. The adoption of circular economy policies greatly increases the complexity of modelling, and a consistent modelling approach is crucial for harmonizing EU product policy-making across product groups.

In academic research, there has been a focus on assessing product policies related to recycling (Barkhausen et al. 2023b). However, in addition to end-of-life measures such as recycling, measures that affect product lifetime are also relevant but increase the complexity of the model, especially if they affect only a proportion of the stock and are active for a limited timescale. Measures affecting lifetime can extend a product’s life, e.g., through repair, but can also reduce it through early replacement with a more efficient product. While early replacement may seem counterintuitive from an environmental perspective, it can still be relevant e.g. for products whose energy consumption during the use phase is their dominant environmental impact and if the replacements offer significant performance improvements.

In order to model the environmental impacts of different circular economy product policies at the stock level and to support decision making, the dimensionality of the modelling needs to be extended. Not only product flows but also material flows need to be calculated and environmental impacts at the individual product level need to be disaggregated to the stock level (Elia et al. 2017; Merli et al. 2018; Anandh et al. 2021). A combination of MFA and LCA could be suitable for this purpose (Barkhausen et al. 2023b).

MFA and LCA are independent methodologies in different fields of research. MFA was developed to quantify material stocks and flows within defined spatial and temporal boundaries based on mass balance (Brunner, Rechberger 2017). It is particularly suitable for long-term macroscale system analysis (Corona et al. 2019; Elia et al. 2017). LCA is a standardized methodology (ISO 14040/44) for assessing the environmental impacts of products or services (ISO 14040:2006; ISO 14044:2006). While any LCA inherently includes an assessment of material flows in the life cycle inventory, LCA is typically performed at the product level without considering material stocks (ISO 14044:2006). When combined with MFA to assess the impact of long-term scenarios on entire product stocks, LCA can holistically assess the environmental impact of product-specific circular economy policies. Several researchers have highlighted the potential benefits of combining MFA and LCA for decision making (Withanage and Habib 2021; Islam and Huda 2019; Sakai et al. 2017; Perminova et al. 2016), and particularly the general suitability of combined modeling to assess the impact of circular economy requirements in product regulations (Barkhausen et al. 2023b; Elia et al. 2017; Merli et al. 2018) (supplementary information for additional information on combined MFA and LCA modeling accessible online, see 5.6).

With the aim of improving science-based decision making, this study therefore seeks to answer the following question: How can MFA and LCA be combined to assess the market-wide material flow and environmental impacts of circular economy-related product policies?

This paper consists of five chapters, including this introductory one. Chapter two describes the overall structure of the developed model and the links between the model components, and then considers the theoretical dimensions of each modeling step. It also outlines a case study on electric motors. The third chapter briefly presents some results of the case study, and the fourth chapter discusses their general implications as well as the insights gained and limitations of the theoretical model. Finally, the conclusion provides a brief summary and critique of the findings.

5.2. Materials and methods

5.2.1. Model overview

Based on the analysis of existing combinations of MFA and LCA (see 5.1), we developed a model that combines three components: a conventional stock model, a product database and an environmental assessment. By combining these components, the model can assess the environmental impacts, product and material flows of circular economy policy measures over time and for entire markets. A *circular economy policy measure* here refers to an assumed effect on individual products, product sales or product stocks induced by a specific product policy, such as reparability requirements leading to extended lifetime. We define *product stock* as the cumulated products that have been sold and are in use, and a *product variant* as one design variation within the same product group with unique characteristics (e.g., material composition, lifetime, and possibly other characteristics such as energy consumption).

The policies that the model can evaluate were chosen based on an existing taxonomy of circular economy product policy requirements (Barkhausen et al. 2023a) and fall into three categories (see bottom of Figure 25). Those that affect production or design, those that affect the use phase and particularly lifetime, and those that affect the end-of-life, each with its own modelling challenges. Lifetime changes include extensions due to repair or improvements in the production or design phase and reductions due to early replacement (of inefficient) products. End-of-life measures include material-specific recycling efficiencies, and production/design requirements include material-specific recycled content.

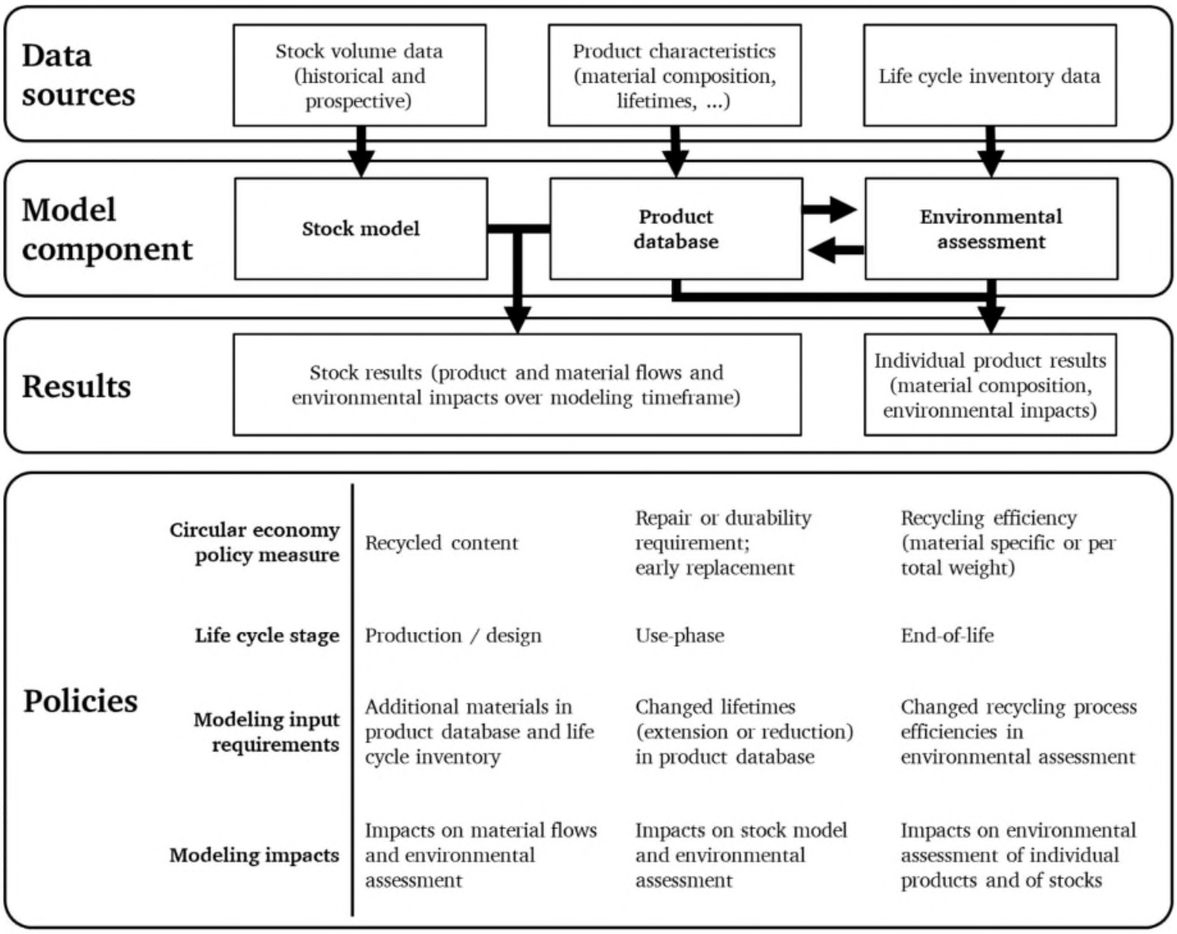


Figure 25 Model overview.

Figure 26 shows which product and material flows are affected by these policy measures. It also shows the system boundary of the stock and material modeling and the environmental assessment. The environmental assessment is done for one point in time and one product variant, and includes all life cycle stages from cradle to grave. The stocks and materials are modeled over time, starting when the products enter the stock, considering potential lifetime changes for certain shares of the stock, and stopping when the products leave the stock.

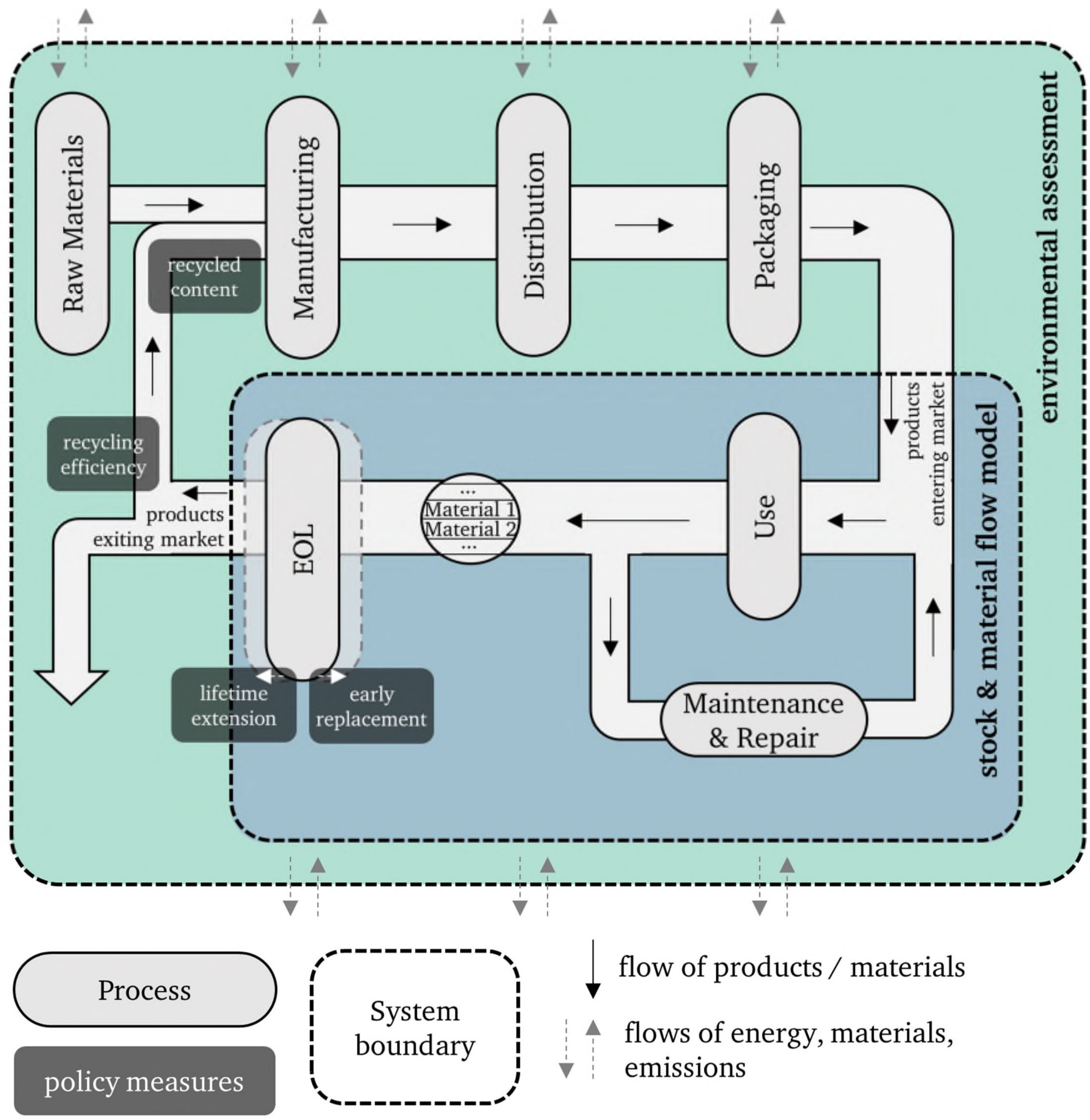


Figure 26 Flowchart of model product and material flows, system boundaries, and policy measures.

Figure 25 and Figure 27 show how the components (or modules) of the model are linked. The model works with a given stock as an input value (stock-driven), which remains unchanged. Therefore, lifetime changes may change product sales, but the stock remains the same (further information in 5.2.2).

The interconnector of the model is a product database that defines material intensities and other characteristics (energy consumption, efficiency, etc.) of the product variants. The product database serves as the input for the exogenous environmental assessment, and feeds back the environmental results for predefined impact categories into its database. This information, together with product sales and stocks, can then be used to assess material flows and environmental impacts over the modeling timeframe.

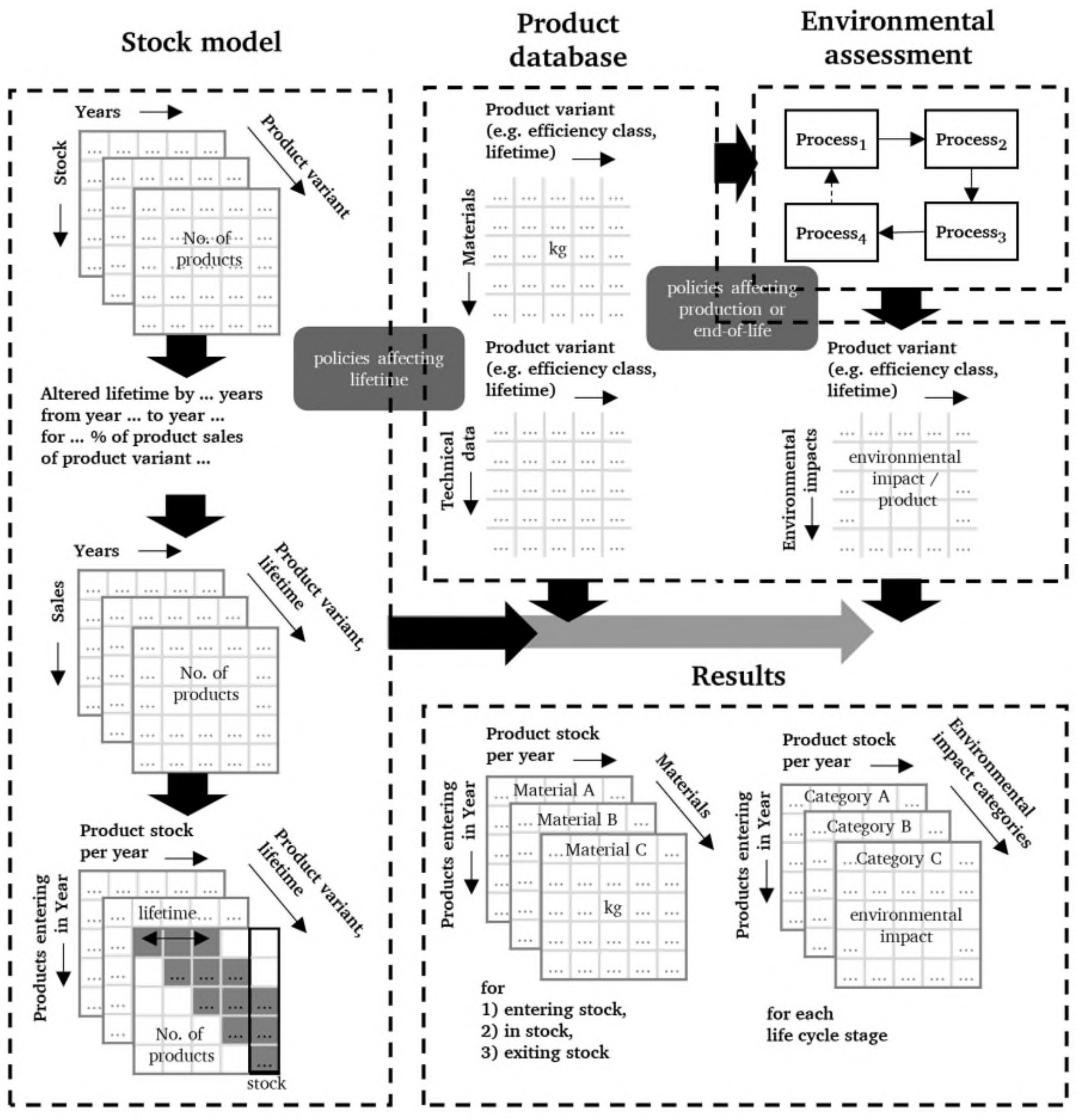


Figure 27 Schematic representation of the modeling logic.

The following subchapters provide more detailed information on the general characteristics of the stock model, the product database and the environmental assessment. As there is a well-established environmental assessment approach for the impact assessment in the Ecodesign Directive context in the form of the EcoReport tool (see 5.1), the focus of this study is on explaining the stock model, where lifetime changes impacting the product stock pose new modelling challenges. Finally, a theoretical case study on electric motors demonstrates how the model can be used to analyze the impact of policies inducing lifetime changes.

5.2.2. Stock model

We use a stock-driven model with an exogenously given stock level (demand-driven) from which inflows and outflows are calculated endogenously based on average lifetimes (B. Müller 2006; Wiedenhofer et al. 2019). It is assumed that the demand for products is not affected by the analyzed policy measures, since they do not alter the function of a product nor do they induce behavioral changes (in contrast to sufficiency measures, for example). The product stock as the given input value includes shares of product variants. The model is future-oriented and historical figures are used to build up the current stock. Historical data (actual or estimated) must be available so that the calculated stock is equal to the actual stock in the first result year minus the maximum lifetime of the product.

In addition to historical stocks, scenarios for prospective stock development have to be estimated. Scenarios are part of future and foresight science and can be described as "coherent descriptions of alternative hypothetical futures that reflect different perspectives and past, present and future developments" (van Notten 2005). The underlying inputs for the scenarios must be reported transparently, clearly stating the origin of external data (e.g., author, year of creation, geographical scope) and the reasons for any assumptions made.

As discussed earlier, the system boundary of the stock model is from when products (and materials) enter the stock until they leave it. The time unit is a full year.

Stocks and sales are calculated in a layered structure. First, the sales are calculated inversely to match the stock, which is the given input value and necessary condition (see formulas 1 and 2 for the calculation of sales and stock in a given year y). Second, these calculated sales are taken as a new initial reference for the model, to which changes in sales due to policy measures are applied. Third, the results (materials, environmental impacts) are calculated by pairing the sales and stocks per product variant with the material composition and environmental impacts of each product variant from the product database.

$$sales_y = stock_y - stock_{y-1} + sales_{y-lifetime} \quad (1)$$

$$stock_y = \sum_{t=y-lifetime+1}^{t=y} sales_t \quad (2)$$

In its current form, the logic is sufficient to model the impact of policies that make changes only to new product sales (e.g. improved energy efficiency). Now, if we consider policies beyond market-entry requirements (beyond the scope of past Ecodesign requirements), such as policies that change the lifetimes of products in stock (such as early replacement or repair), modelling complexity increases. Since we use a stock-driven approach, changes in lifetime should not impact the overall number of products in stock. Therefore additional considerations are needed. Although policies that change the lifetime are applied to the product stock and not the sales, it is modeled as if they were known when the products entered the stock. Therefore, the lifetimes of past product sales are modified to achieve the desired effect. For example, for a two-year earlier replacement in 2030 and an original lifetime of ten years, the products sold in 2022 have a reduced lifetime of eight years. As a result, the products exit the stock in 2030, two years earlier than they would normally. Similarly, for a two-year lifetime extension due to repair, the lifetimes in 2020 are extended to twelve years.

For the stock, as the given constant for the model, to remain the same (see formulas 3 and 4 for sales and stock in year y for different policy measure scenarios), the sales of products with normal lifetimes are adapted accordingly (see Figure 28). First, when the products with extended or shortened lifetimes enter the stock, sales of products with normal lifetimes decrease by the same amount. Second, during the period in which the measure is active, and products leave the stock earlier (early replacement) or stay in the stock longer (repair), the number of sales of normal-lifetime products is reduced (for the repair measure) or increased (for early replacement) in line with the proportion of affected products. This reduction or increase is only temporary, since the sales of the products are only shifted into the future. The change in sales together with the different lifetimes create oscillations that must be compensated by increases or decreases in sales over time.

$$sales_{y,scenario} = stock_y - stock_{y-1} + sales_{y-lifetime(y,scenario)} \quad (3)$$

$$stock_y = \sum_{t=y-lifetime(y,scenario)+1}^{t=y} sales_t \quad (4)$$

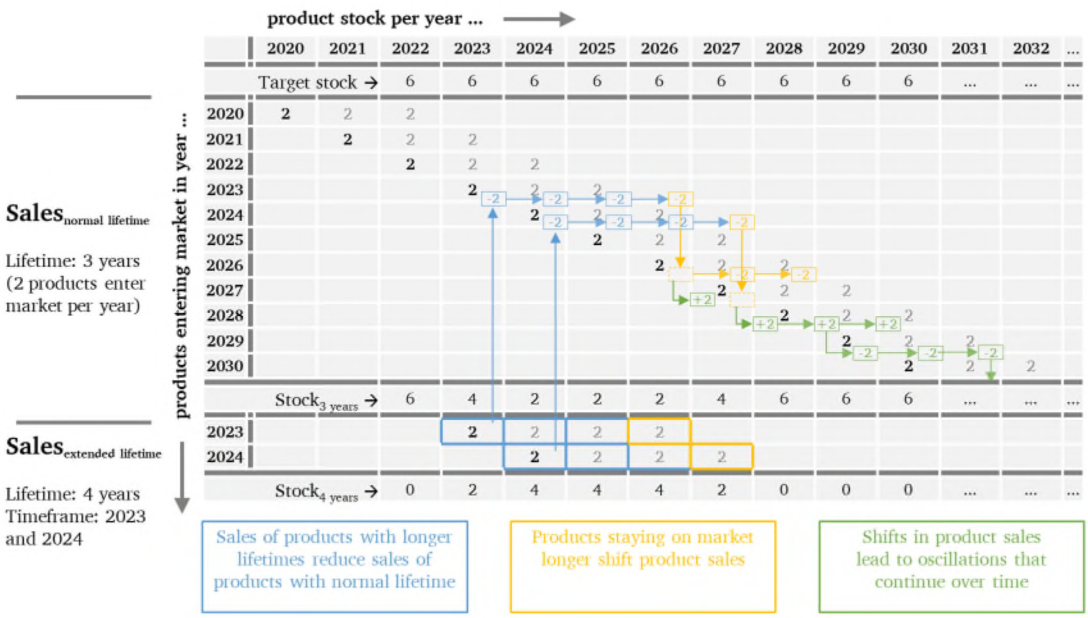


Figure 28 Exemplary visualization of lifetime changes on stock modeling. Example for product lifetime extension from three to four years of products entering stock in 2023 and 2024.

5.2.3. Product database

The product database serves as the integrator between the stock model and the environmental assessment. Here, the bill of materials is defined for each product variant. The intersection of the bill of materials and product flows allows the calculation of material flows. The bill of materials together with other foreground life cycle inventory data (energy consumption, efficiency, etc.) serves as input for the environmental assessment, which returns its result to the product database and can then be matched with the product flows to upscale the environmental impact of individual products to the entire stock.

Depending on the product group, there may be a significant number of product variants. Consider an electric motor (see case study) with five power ranges (each captured in the product database by a representative product variant), and six efficiency classes. In this example, there would be 30 product variants listed in the product database, each with a unique material composition and performance characteristics. The environmental assessment must be performed for each potential lifetime change. Taking into account normal lifetime, shortened lifetime due to early replacement, and extended lifetime (due to repair) results in 90 product variants (30 x 3).

5.2.4. Environmental assessment

The environmental dimension enters the model through the product database. There, material composition is defined for each product variant, which serves as input for the environmental modeling, together with other relevant data such as energy consumption or efficiency.

In principle, any environmental assessment can be linked to the product database. Regardless of the method chosen for the environmental assessment, it is important to remember that the analysis is likely to be carried out for a large number of product variants. While the environmental assessment itself can be carried out independently outside the stock model software (exogenous), as in the case study, it may be advisable to link or integrate the assessment directly.

The results of the environmental assessment are linked back to the stock model results as follows: The environmental impacts for the life cycle stages raw materials, manufacturing, distribution, packaging, and maintenance & repair are all considered in the year the products enter the stock. Sales in a given year for a given product variant are multiplied by the environmental impacts of that product variant. For the use phase, the environmental impacts are evenly distributed over the entire product lifetime. The distributed environmental impacts per year are calculated in the product database based on the results of the environmental assessment. The formula multiplies these values by the sum of the stock in a given year for each product variant.

5.2.5. Case study

Motivation

A case study on industrial electric motors in the EU was used to demonstrate the model. Data on product stocks and motor type-specific characteristics (lifetime, material composition, etc.) were taken from the preparatory study conducted by the ISR University of Coimbra, which formed the basis for the first ecodesign implementing measures for electric motors in 2009 (Commission Regulation (EC) No 640/2009) (Almeida et al. 2008). The analyzed policy measures are limited to those that affect the use phase by inducing lifetime changes and include a repair measure (extending the lifetime of 10% of the total motor stock by two years from 12 to 14 years) and an early replacement rate (5% of IE2 motors replaced by IE4 motors two years before their normal end-of-life - 10 instead of 12 years), each compared to a situation with no active policy measure. The policy measures are assumed to be active from 2025 to 2030.

Motors are a particularly interesting case study because early replacement of old inefficient motors (reducing lifetime) is expected to have a significant beneficial effect, which supposedly outweighs the material and production savings from extending the lifetime of the motor, e.g., through repair measures (extending lifetime) (Almeida et al. 2023).

It should be noted that the available data are from 2009 or before and therefore outdated, and the geographical scope is limited to the EU-15 countries. The calculations are only meant as a theoretical example to validate the approach and implementation of the model.

In terms of software use, the goal was to develop an accessible model that enables the assessment of the environmental impacts of different policy options. Therefore, it was decided to implement the stock model in Microsoft Excel without using any programming language extensions such as Visual Basic.

Stock model

The stock is initiated from a hypothetical value of zero to match the first actual data in 1992, and missing values are linearly interpolated. A dataset from the University of Coimbra was used, which also includes estimates of future stocks up to 2030, which are used for linear extrapolation to create a scenario up to 2050. See 5.2.2 for the stock model logic.

Product database

The motors assessed are three-phase induction motors (highest market share in terms of units sold in the EU) in the power output range of 0.75 to 7.5 kW (used, e.g., in fans, pumps or compressors) (Almeida et al. 2008; UN Environment 2017). In the absence of data on the share of efficiency classes in the motor stock, values were derived from the EU minimum energy performance standards. These require motor sales with a rated power output between 0.75 and 1 000 kW to be at least IE3 efficient by July 2021 (European Commission 2019a). For simplicity, only three efficiency classes were considered: IE2, IE3 and IE4 motors (IE = International Efficiency, higher meaning more efficient). Assuming a lifetime of 12 years (UN Environment 2017), IE2 class motors are gradually replaced by IE3 and IE4 motors and disappear from the stock in 2033 (supplementary information with further information accessible online, see 5.6).

Environmental assessment

The EcoReport tool was used as an exogenous approach to assess environmental impacts. As described in the introduction, the tool was developed for the European Commission to facilitate environmental impact analysis within the elaboration of EU product regulations for energy-using or energy-related products (under the MEErP). The EcoReport tool is an Excel-based, simplified life-cycle based tool that was first released in 2005 (Kemna et al. 2005). It is a relatively simple calculation tool where methodological choices have been made, such as system boundaries and included parameters (Wesnaes and Hansen 2021). It was updated in 2011 and 2013 to include improved end-of-life modeling (Kemna et al. 2011; Mudgal et al. 2013) and is currently undergoing another revision with significant changes being made.

While not the focus of this case study, policies related to production/design (recycled content) or end-of-life (recycling efficiency) can be assessed at the product level in the EcoReport tool and then be scaled up to the overall stock level using the proposed model. In the case study, the environmental assessment is implemented using the 2013 version of the EcoReport tool, which is the most recent publicly available version. Within the EcoReport tool, the results can be normalized to the total impacts generated in the EU in the respective impact category. The material composition of the different motor efficiency classes was provided by the University of Coimbra and can be found in the supplementary information (accessible online, see 5.6) (personal communication, November 2022). Since electric motors can be considered energy converters, transmitting mechanical energy to the end-use device (Almeida et al. 2008), only energy losses are considered for the environmental assessment since the remaining consumed energy is being transmitted as mechanical power. Other assumptions required for the environmental assessment are described in the supplementary information (accessible online, see 5.6).

5.3. Case study results

The results of the case study show the modeling results without policy measures and the changes induced by the policy measures repair and early replacement.

Figure 29 shows some of the results that can be derived from the model for a baseline analysis without any of the policy measures. The model provides information at the individual product level as well as results at stock level.

At product level, the material composition (input value) is shown for the three motor efficiency classes analyzed (Figure 29 a) as are the environmental impacts per year (normalized to EU totals) according to the efficiency class and motor lifetime (normal lifetime of 12 years, extended lifetime of 14 years and reduced lifetime of 10 years) (Figure 29 b)). The more efficient motors require significantly more materials, mainly due to an increase in electrical steel and copper (over 30% mass increase per efficiency class improvement). At the same time, these more efficient motors use significantly less energy, so the energy consumption (per year of lifetime) is reduced by more than 20% for each efficiency class (not shown in Figure 29). The other environmental impact categories vary greatly when changing from an IE2 to an IE3 motor (12-year lifetime, impacts per year), and range from a 19% increase in persistent organic pollutants to a 17% reduction in volatile organic compounds. The large difference can be explained by the life cycle stages to which environmental impacts are attributed. Persistent organic pollutants are mainly generated in the raw materials stage, where the amount of material increases due to the more efficient motors, as mentioned above. Volatile organic compounds, on the other hand, are mainly caused by energy consumption in the use phase, where the more efficient motors lead to significant energy savings.

At the stock level, Figure 29 c) shows the materials demand for motors in the EU stock, and Figure 29 d) shows the environmental impacts (including for each respective year the raw material and production impacts of products entering the stock, use phase impacts (related to energy consumption) and impacts of products reaching end-of-life). The material amounts correspond to the growth of the motor stock, with an even stronger increase from 2022 when all sales are the more material-intensive IE3 and IE4 motors. The environmental impacts at stock level extrapolate the individual motor results, with a shift between impact categories as the more efficient motors start to enter the stock.

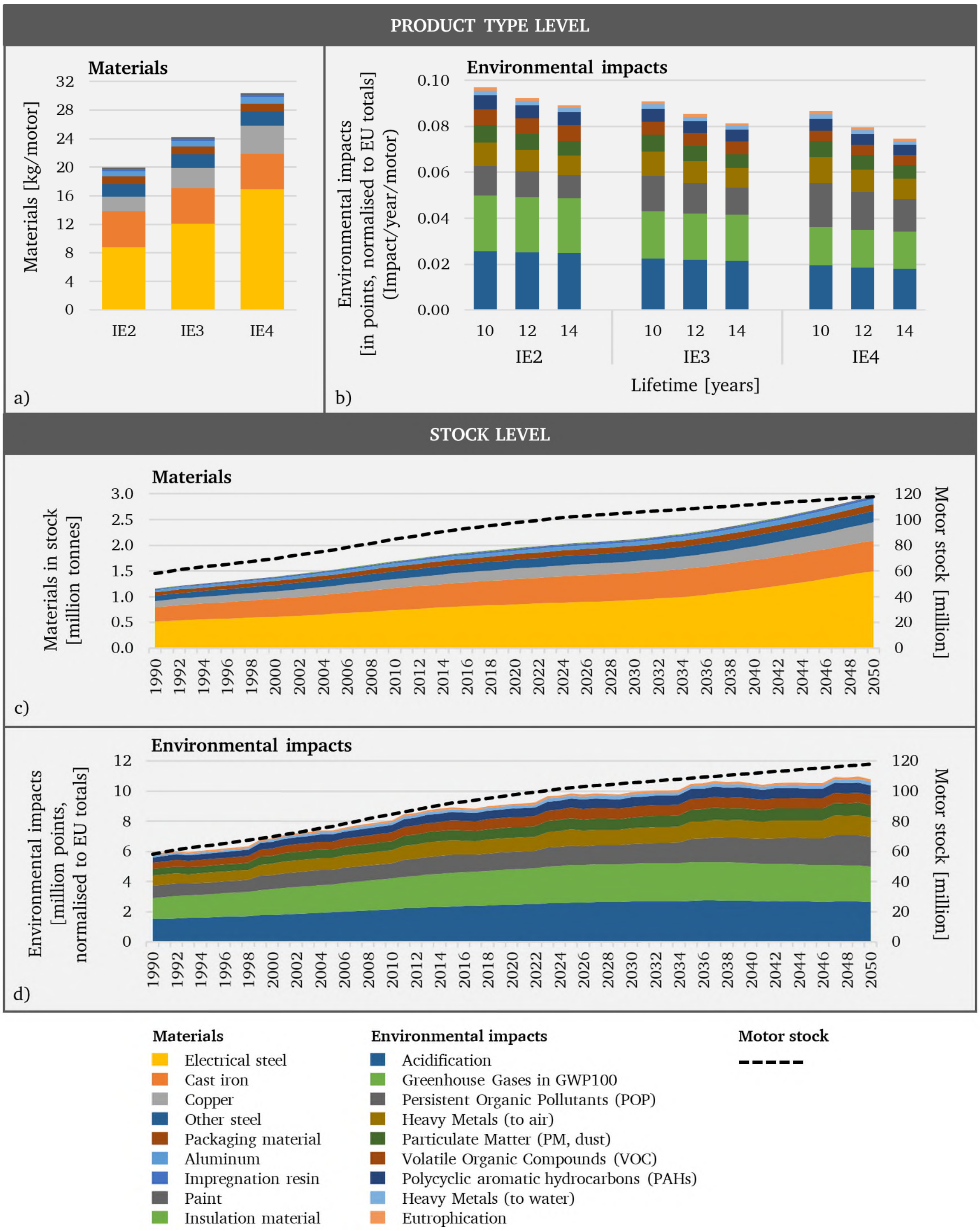


Figure 29 Exemplary case study results (without activated policy measures). a) Material composition for efficiency classes, b) Normalized environmental impacts per efficiency class and lifetime, c) Materials in the stock 1990-2050, d) Environmental impacts of motors 1990-2050.

Next, exemplary results for activated policy measures are presented. The model delivers the number of motors with different lifetimes entering the stock in a given year (Figure 30 a)). It also allows the comparison of the material entering the stock (material demand) and the material leaving the stock (maximum theoretical recycling potential) (Figure 30 b)), as well as the environmental impacts (Figure 30 c)).

Looking at the quantities of copper entering and leaving the stock in motors, Figure 30 b) shows the oscillations in motor sales. When the policy measures are active, the change in sales together with the different lifetimes trigger oscillations in the stock that must be compensated by an increase or decrease in sales over time in order to retain the original stock value as the input value. Because this is an effect caused by the logic of the model (see 5.2.2), results for individual years may be outliers and results should be considered over a longer period of time that averages out the fluctuations. For example, Figure 30 c) shows the results for the global warming potential, which has the second highest impact in the normalized results per individual motor. The results are shown for the 2025 to 2030 timeframe and the 2025 to 2035 timeframe - with clear differences in the results.

As might be expected, for the 2025 to 2035 timeframe, the largest reductions are achieved by the early replacement policy measure (significant reduction in energy consumption due to the more efficient motors). However, this pattern is not yet visible in the 2025 to 2030 timeframe. Here, the repair measures result in the highest savings (due to reduced material production). With a lifetime of 12 years, the more efficient motors that enter the stock in 2030 as a result of early replacement remain in the stock and trigger savings until 2042.

As already pointed out, the case study results are limited by the data availability and are mainly meant to demonstrate how the model functions, which is discussed in the following section.

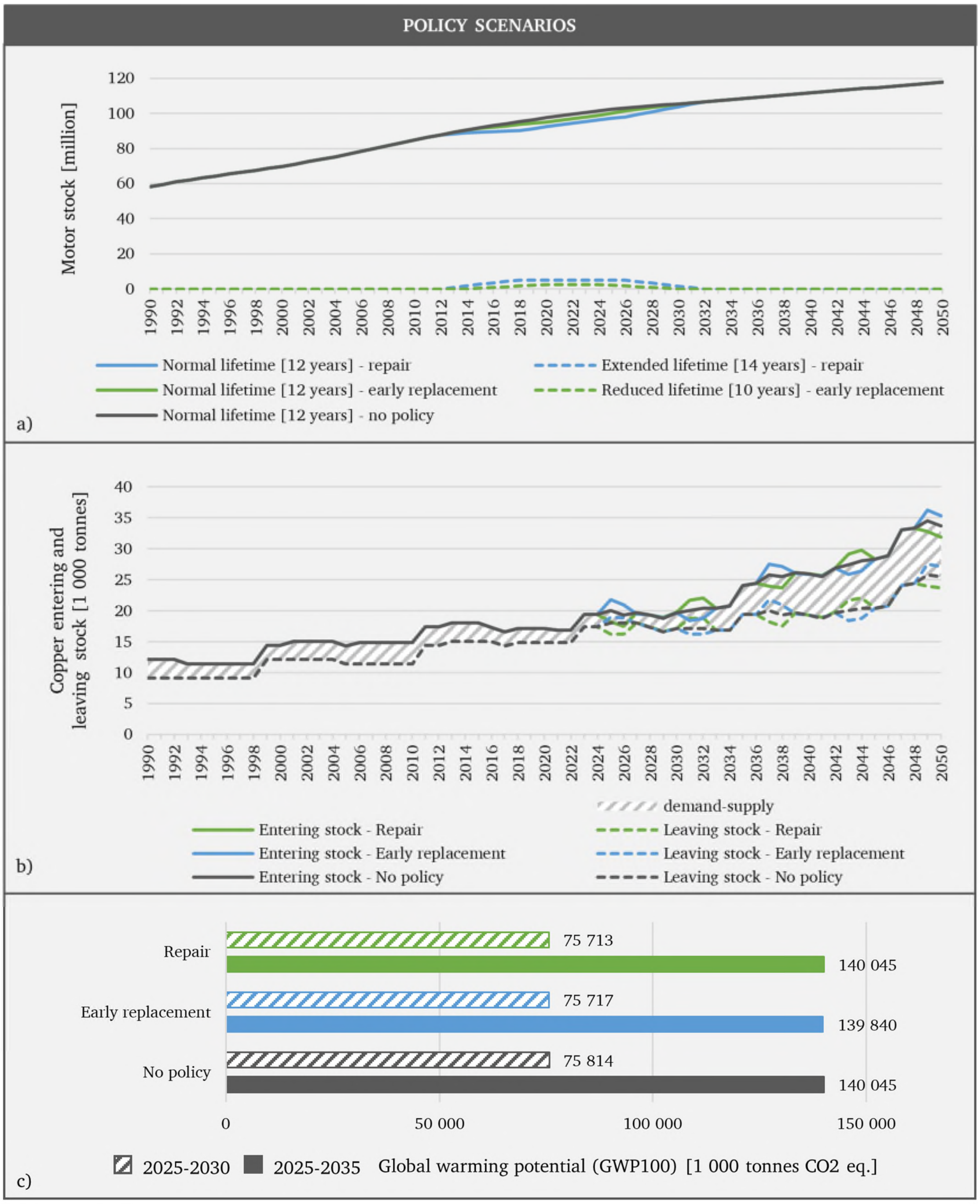


Figure 30 Exemplary case study results for activated policy measures. a) Copper entering and leaving stock 1990-205, b) All materials entering and leaving the stock 2025-2030, c) Global warming potential for the periods 2025-2030 and 2025-2040.

5.4. Discussion

We combined a stock and material flow model with a product database and an environmental assessment. Although none of the three components is new in itself, the combination gives new insights and allows to scale up the environmental assessment at product level to the systemic stock level. The potential of bringing together MFA and LCA has also been recognized by similar approaches, such as layered approaches, where the object of study is disaggregated e.g. from the product to the material level (e.g. (Pauliuk et al. 2021)) or in the study by Lausselet et al.

(2021) where "product archetypes" are defined comparable to our concept of a product database.

However, our model fills a methodological void in the domain of product policy impact assessment and in the breadth of policy options it can assess. Until now, the focus has been placed on energy consumption in the use phase, which allows for straightforward one-dimensional modelling. Circular economy policies present modelling challenges that require a renewal of existing approaches to assess multi-dimensional impacts. This is where our developed model makes an important contribution. We have proposed a way to upscale product level environmental assessment and integrated the modelling of lifetime changes in the product stock to assess the impact of novel policies.

The case study shows the variety of results that can be obtained from the model at the example of policies impacting lifetime. Product and material flows as well as environmental impacts can be analyzed over the product life cycle at the individual product level or over long time horizons at the stock level.

Similar to the results of (Boldoczki et al. 2021), who assessed the environmental impacts of the reuse of washing machines, our results indicate large differences between the impact categories for the different policy options, highlighting the need to include detailed environmental assessments when conducting studies on the impact of circular economy policies.

The case study also highlights several limitations, some of which are discussed below. In its current form, the model does not allow the integration of time series data for the environmental impact intensities in the stock model. It can be summed up as a model that integrates exogenous environmental assessment results into a static product database, which then provides the inputs for a dynamic and prospective stock and material flow model. Especially for energy-consuming products and products with a long lifetime, the development of the electricity mix over the modeling timeframe is crucial. This issue becomes less important when moving beyond energy-consuming products and including products with short lifetimes. A workaround for energy-consuming and long-lived products would be to perform the environmental assessment with zero energy consumption in the use phase and integrate energy consumption directly into the stock model using time series emissions data. It should be noted that dynamic (time-dependent) inventory data are generally not available in standard LCA datasets and may require significant additional data collection efforts while increasing uncertainty, even though methods for developing prospective inventory data have been proposed (Sacchi et al. 2022). Due to the logic of the product database, considering time series within the environmental assessment would require another dimension of product variants for different points in time, significantly increasing the computational effort involved. In a situation where we are using a static product level assessment, the quality of the data and its transparent reporting becomes critical. In addition, the case study only considered policies that change lifetime. If policies related to production/design (recycled content) or end-of-life (recycling efficiency) are considered, they need to be assessed at the individual product level in the environmental assessment. This can pose significant methodological challenges, e.g. regarding the allocation of environmental burdens or benefits.

An important challenge is related to the translation of policies into model input parameters. Here the model relies on product-specific assumptions, which can be a source of great uncertainty in ex-ante assessments, and are an important area for future research. Conversely, the results of the assessment do not equate to real-life savings. For example, the case study presented here would need to assess what actually happens to early replaced motors. The resale or export of inefficient devices could offset any savings or simply shift the burden outside the geographic system boundary. In light of the development of Ecodesign beyond energy-related products (European Commission 2022c) the question of how to consider impacts being generated outside the geographical scope of the legislation becomes important and requires further research.

Another challenge is the interpretation of results. Due to the modeling logic, oscillations of sales can occur that may distort the results for individual years or short time periods. It can be misleading to make decisions or draw conclusions based on such outliers. These fluctuations can be averaged out by considering longer periods. Therefore, interpreting the scenario results should be done over longer periods. Within the environmental results, the weighting of impact categories strongly influences the identification of hotspots and, as such, is a delicate topic that should be handled very carefully and reported transparently.

To understand the impact of different input parameters on model outputs, a sensitivity analysis can be performed in the form of what Saltelli and Annoni call "local" sensitivity analysis (Saltelli and Annoni 2010). The sensitivity analysis is not used to test the robustness of the model-based inference, but rather to investigate the influence of a single parameter. This is done by deriving variations in the model output as a function of the input parameter.

To cover another dimension of sustainability, the economic side could be considered by integrating life cycle costing into the model.

5.5. Conclusion

In light of the increased consideration of circular economy aspects in EU product policies, we developed a model for ex-ante material flow and environmental impact assessment of circular economy policies. We combined a prospective stock and material flow model with an exogenous environmental analysis via a product database in order to assess the long-term stock impact of product policies along a number of modeling dimensions. A case study on electric motors was used to demonstrate the model and show the results for policies that increase (repair) or decrease (early replacement) lifetime. The results illustrate the wide range of outcomes that can be obtained. As the theoretical example of motors shows, taking inefficient but functional products out of the stock earlier – which may seem counterintuitive from a circular economy perspective - can reduce environmental impacts, highlighting the complexity of evaluating circular economy product policies. The case study also indicates the need for further research into holistic methodologies to assess the material flows and environmental impacts of product policies, and the need for better modeling of the trade-offs between energy efficiency and environmental impacts for energy-consuming products.

5.6. Supplementary information

Supplementary information will be available online upon publication.

6. Coherence of novel policies for lithium-ion batteries for electric vehicles: A multidimensional analysis of material flows and environmental impacts

This chapter was published in June 2024 eceee Summer Study proceedings. Table 13 provides more information about the publication.

Table 13 Publication information of (Barkhausen 2024).

Title	Coherence of novel policies for lithium-ion batteries for electric vehicles: A multidimensional analysis of material flows and environmental impacts
Authors	Robin Barkhausen
Publication date	20.06.2024
Journal	eceee Summer Study proceedings
DOI	10.24406/publica-3334

6.1. Introduction

Following its commitments under the Paris Agreement, the EU has set itself the goal of achieving carbon neutrality by 2050 (European Commission 2019c). The transport sector, which accounts for 26% of final energy consumption, is of paramount importance in achieving this goal (Eurostat 2023). Within road transport, electric vehicles have emerged as a promising technology and sales continue to grow strongly (Hettesheimer et al. 2023). Lithium-ion batteries have become the dominant energy storage system for these electric vehicles, but they are not without concerns.

In the 2020 update of the EU's Critical Raw Materials List (Eynard et al. 2020a), three of the main materials used in lithium-ion batteries are classified as critical due to their economic importance and high supply risk for Europe: lithium, natural graphite and cobalt. Furthermore, the environmental impacts associated with raw material mining and battery production are high, and traditional pyrometallurgical recycling routes have long failed to efficiently recover some of the key materials, such as lithium and graphite (Chen et al. 2019).

To address these concerns, a new battery regulation was adopted in July 2023, with a strong emphasis on resource efficiency aspects (European Commission 2023a). Compared to previous battery legislation and other product regulations, it includes unprecedented requirements such as material-specific recycling efficiencies and mandatory recycled content targets (see Figure 31 for the battery life cycle and the point at which policy measures take effect) (Barkhausen et al. 2023a).

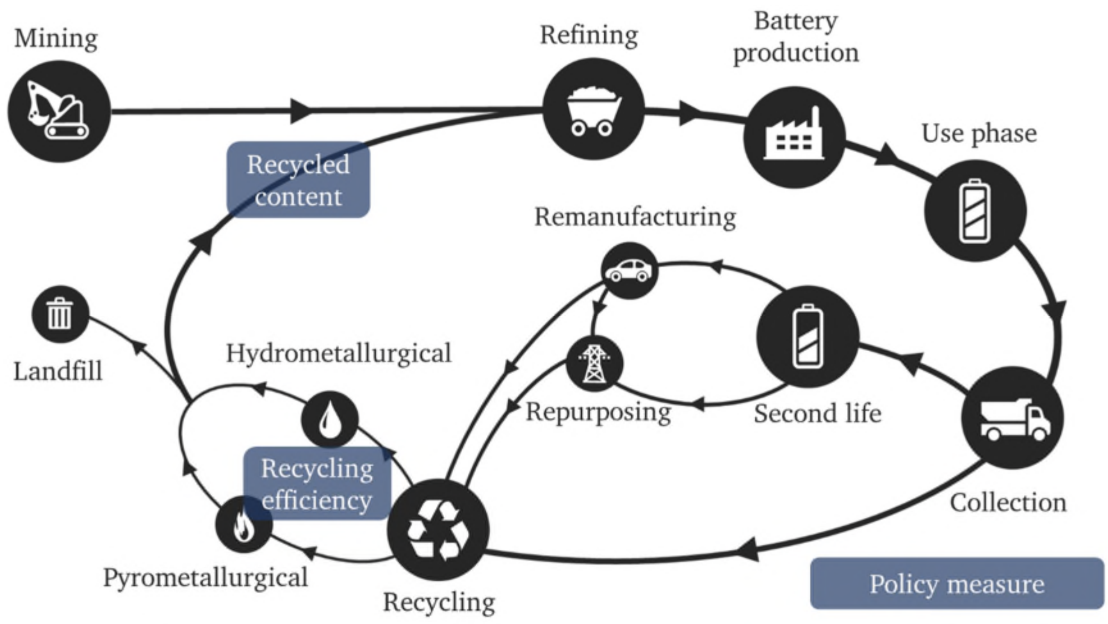


Figure 31 Life cycle of lithium-ion batteries for electric vehicles, along with two selected policy measures at the point in time when they take effect.

In the context of the new Battery Regulation, it is of interest to analyse the environmental and market impacts and trade-offs of novel policy measures in a still emerging market. How will the proposed material-specific recycling efficiencies affect the supply for recycled materials? And can the targets for recycled content be met in the coming years as the market continues to expand?

To answer these questions a MFA was combined with an environmental impact assessment to produce a scenario analysis of the EU passenger car electric vehicle market from 2011 to 2050.

The study is divided into five subchapters. Chapter 6.2 provides an overview of the scientific literature. Chapter 6.3 presents the methodology of the model used for this study. Chapter 6.4 presents and discusses the main findings. Finally, chapter 6.5 provides a brief summary and critique of the findings.

6.2. Literature review

One of the first studies to analyse the flow of electric vehicle lithium-ion batteries and compare it with end-of-life alternatives was carried out in 2014 for the US market (Richa et al. 2014). In the study, the researchers performed a MFA from 2015 to 2040 and also introduced a basic economic assessment of the materials into the discussion. To address uncertainties in the assumptions, they introduced different scenarios for factors such as electric vehicle uptake and battery lifetime.

Several studies on battery MFA followed. Sommer et al. (2015) extended the scope to all types of batteries, while Schmidt et al. (2016) focused only on lithium-ion batteries, but assessed the material flows upstream of battery production. Many studies focused on the two materials cobalt and lithium and the potential resource depletion in the face of the growing demand for electric vehicles (Pehlken et al. 2017; Olivetti et al. 2017). Thereupon, Vaalma et al. (2018) assessed the cost and resource implications of substituting lithium with sodium-ion batteries. In the same year, Ziemann et al. (2018) conducted an MFA for electric vehicle lithium-ion batteries to assess the lithium that can be recycled, using different recycling scenarios and varying the quality of recycled materials. According to their assumption, a low quality of the recycling process could degrade the lithium and make it unsuitable as a material input for lithium-ion battery manufacturing (open loop), leading to an oversupply of secondary lithium. However, if the recycled lithium is of sufficient quality to be used as a material input for manufacturing new electric vehicle lithium-ion batteries (closed loop), it could meet a significant portion of the material demand.

In 2019, Song et al. (2019) conducted an MFA of the Chinese lithium-ion batteries market from 2012 to 2025. They highlighted the policy implications of different waste management strategies, based on supply risk and economic importance of the critical lithium-ion batteries materials. Finally, an in-depth analysis of the EU market was conducted by the Joint Research Centre of the European Commission by Bobba et al. (2019). In it, the authors estimated the material flows from two different powertrains, battery electric vehicles and plug-in hybrid electric vehicles and further assessed the benefits that different battery second life quotas could have. Material flows for both cobalt and lithium were calculated for the years 2005 to 2035. In 2030, recycled cobalt was estimated to meet between 9% and 15% of demand, while the potential for recycled lithium ranged from 7% to 16%.

Increasingly, studies have aimed to add an environmental analysis to the MFA. However, such studies are fewer and often focused on individual countries (Barkhausen et al. 2023b). Huang et al. (2020) assessed the copper flows and environmental impacts associated with the increased use of electric vehicles in China. Pauliuk and Heeren (2021) assessed the environmental potentials of increased material efficiency for different product groups including passenger cars in Germany using a multi-layer model, and Nguyen-Tien et al. (2022) analysed the material flows of a future lithium-ion battery recycling industry in the UK and added a basic environmental analysis independent of the rest of the analysis. A study on the European market was carried out, again by researchers at the Joint Research Centre, to understand the environmental and material implications of increased use of traction batteries in the transportation sector (Bobba et al. 2020). Again, the environmental and material flow analyses were conducted independently. The environmental analysis statically compared different lithium-ion battery chemistries for different points in time, while the MFA provided supply and demand scenarios for the most critical battery materials. Therefore, the study produced multi-dimensional results including material flows and environmental impacts, but did not integrate the two dimensions in an integrated modelling to produce integrated results.

Due to the rapid development of the electric vehicle market, there is a need to further investigate the end-of-life material flows and associated environmental impacts along the supply chain. Such integrated modelling is of great importance for the recycling industry and policy makers, especially in the light of the new battery regulation.

6.3. Methods

We have developed a combined MFA and environmental assessment model of lithium-ion batteries[7] in the EU-27 passenger car electric vehicle market from 2011 to 2050 to assess impacts and prerequisites of the policy measures proposed in the new Battery Regulation (Regulation (EU) 2023/1542) (see Table 14). It should be noted that the regulation explicitly provides for the possibility to amend the targets for recycled content by 2029 if existing or projected availability does not meet the targets.

Table 14 Proposed policy measures under Regulation (EU) 2023/1542.

	Recycling target		Recycled content	
	2027	**2031**	**2031**	**2036**
Cobalt	90%	95%	16%	26%
Copper	90%	95%	-	-
Lithium	50%	80%	6%	12%
Nickel	90%	95%	6%	15%

The model uses a layered approach where a stock analysis is disaggregated to the material flow and environmental impact level through a product database. The *product stock* is defined as the cumulative products that have been sold and are in use. The *product database* contains the specifications such as material composition and lifetime of each lithium-ion battery type and serves as input for the environmental assessment. The environmental assessment is performed for one point in time and for one type of lithium-ion battery, and the results are linked back to the product database where they are crossed with the stock analysis.

The model is based on a given stock of electric vehicles as an input value (stock driven), which remains unchanged when lifetime changes are induced (e.g. by battery re-use). Stocks and sales are based on historical data from the European Alternative Fuels Observatory (EAFO 2023) and are interpolated to saturation at annual sales of seven million in 2036, one year after the zero

[7] Lithium-ion batteries are assumed to remain the dominant technology for the EU passenger electric vehicle market for the foreseeable future. Changes in the material composition of lithium-ion batteries are explained below.

emission target becomes active according to the regulation (European Commission 2023b). The interpolation uses a second-degree polynomial function starting from 1 438 009 sales in 2023. While there is uncertainty in projecting the uptake of electric vehicles, this is less critical in assessing the proposed policy measures as the impact of sales on end-of-life volumes is delayed by battery lifetime.

Sales of plug-in hybrid electric vehicles show a clear downward trend in 2023. For this reason and to simplify the modelling, only battery electric vehicles are considered.

6.3.1. Battery size and chemistry

The average battery size and chemistry of electric vehicles entering the market were calculated based on a market analysis by Takeshita et al. (2019), which includes global electric vehicle sales and projections from 2011 to 2023.

The sales figures S for each electric vehicle model (333 in total) in a given year were multiplied by the battery size B of that electric vehicle model. The sum for all models was then divided by the total number of sales to give the average battery size B_av in each year (see formula 5).

$$B_{av,BEV,2011} = \frac{\sum_{i=1}^{333} S_{i,2011} * B_i}{\sum_{i=1}^{333} S_{i,2011}} \quad (5)$$

To estimate the amount of battery constituent materials, it is necessary to know the proportion of different lithium-ion battery cathode types. To estimate this number, an analysis was performed based on historical sales data from Takeshita et al. (2019). Batteries with $LiFePO_4$ (LFP) and $LiMn_2O_4$ (LMO) cathodes were excluded, as they represent a marginal proportion of the dataset of passenger car electric vehicle sales in the control period. Therefore, the focus is lithium-ion batteries with $LiNiCoAlO_2$ (NCA) cathodes, as used by Tesla, and LiNiMnCo (NMC) cathodes, with varying weight shares, as used by most other automakers (Li et al. 2018). Future estimates take into account the trend among battery manufacturers towards higher nickel contents in the cathode to reduce dependence on cobalt. High nickel contents, used in both modern NCA and NMC cathodes, improve discharge capacity but reduce thermal stability (Li et al. 2018).

Material changes in the different lithium-ion battery chemistries are only considered in the cathode. This is considered acceptable as cathodes contain the majority by weight of the most critical materials that constitute lithium-ion batteries (cobalt, graphite, lithium, manganese, nickel and copper in decreasing order of estimated supply risk (Eynard et al. 2020b, 2020a)). For each cathode type, the average battery size is estimated based on data from Takeshita et al. (2019), resulting in a size of 24 kWh (NMC111), 45 kWh (NMC532), 63 kWh (NMC622), 66 kWh (NMC811), and 75 kWh (NCA).

In estimating the future development of cathode chemistries, the trend towards lower cobalt chemistries is expected to continue, a trend that is consistent with other studies (Thielmann et al. 2017). The sales market share of NMC811 batteries is expected to continue to grow strongly from 2023 onwards, saturating at 80% in 2030. Sales of the higher cobalt chemistries NMC111 and NMC532 are assumed to fade out in 2024 and 2025 respectively (due to replacement by newer chemistries), and NCA batteries are assumed to lose their currently high market share as automakers with different battery chemistries increase their share of the electric vehicle market.

Replacement of lithium-ion batteries by new types of electric vehicle batteries is neglected due to uncertainty about technology changes.

6.3.2. Battery lifetime

Over years of use, battery capacity fades due to cell degradation (due to growth of the solid electrolyte interface at the anode, electrolyte oxidation at the cathode, and corrosion or decomposition effects (Jana et al. 2019; Gennaro et al. 2020)). At some point, customer demand for driving range will no longer be met. The United States Advanced Battery Consortium (USABC) has defined end-of-life for electric vehicle batteries as reaching 80% of nominal battery capacity (USABC 1996). In the absence of recent and reliable data on battery lifetime, an average lifetime of 10 years is assumed, as most electric vehicle manufacturers offer a warranty of about 8-10 years for their battery pack (Hettesheimer et al. 2023, p. 18).

At the end of the first battery life, batteries are expected to be collected for appropriate end-of-life treatment. The new Battery Regulation ((EU) 2023/1542) sets the collection target for portable batteries to increase from 45% in 2023 to 73% in 2030 (Article 59), but there is no specific target for traction batteries. Electric vehicle lithium-ion batteries are assumed to have much higher collection rates compared to other types of batteries due to their large size and fixed position in the vehicle body, so they to remain in the end-of-life vehicles where they can be recovered by treatment facilities. Battery and vehicle manufacturers claim collection and recycling rates of 99%, but this does not account for informal battery losses, such as unreported exports or illegal scrapping (European Commission 2019b). While Eurostat data on collection remains incomplete, a more conservative estimate proposed by the European Commission (2019b) is followed with a fixed collection rate of 96% (while acknowledging the high uncertainty of this number due to missing information on exports).

As noted by Richa et al. (2014), there is a potential mismatch between battery and vehicle lifetime. If electric vehicle lithium-ion batteries reach end-of-life before the vehicles themselves and are replaced, this would lead to replacement batteries entering the market, creating a second inflow of batteries. Conversely, if the vehicle lifetime ends much earlier than the battery lifetime, these packs could be reused as electric vehicle batteries. Due to the lack of data on these two additional flows, they have not been considered in this analysis.

6.3.3. Material quantities

To calculate the amount of materials that can potentially be recovered from end-of-life batteries, it is necessary to know the weight percentage of each of the different lithium-ion battery chemistries. Here the analysis was carried out for the constituent materials of the battery electrodes and current collectors namely aluminium, graphite, lithium, nickel, copper, cobalt and manganese. The electrode composition was based on a material analysis of the Renault Zoe and Chevrolet Bolt, both equipped with NMC622 batteries (Takeshita et al. 2019). The changes between the different NMC chemistries were then estimated stoichiometrically, assuming constant amounts in the anode and current collector. The NCA battery composition was based on the cylindrical cells used in the Tesla Model 3 with a lithium oxide cathode with molar fractions of 92% nickel, 5% cobalt and 3% aluminium.

6.3.4. Recycling and recycled content

In the new Battery Regulation, the European Commission has announced ambitious targets for the recycling efficiency of lithium-ion batteries. While the 2006 legislation set a minimum recycling efficiency of 50% by average battery weight, the new proposed targets are specific to traction batteries and are material specific. Article 8 proposes mandatory levels of recycled content in new batteries from 2031 for cobalt (16%), lithium (6%) and nickel (6%) increasing to 26% for cobalt, 12% for lithium and 15% for nickel from 2036. In addition, material specific recycling efficiencies are introduced in 2027 for cobalt (90%), copper (90%), lithium (50%), and nickel (95%). The material specific recycling efficiencies are tightened in 2031 (to 95% for cobalt, copper and nickel and to 80% for lithium).

In the analysis, the proposed recycling levels are based on the targets proposed by the European Commission.

6.3.5. Environmental analysis

The environmental analysis was conducted following the analysis used in Task 5 of the preparatory study on ecodesign and energy labelling of batteries (Lam et al. 2019). The 2013 version of the EcoReport tool is used here.

The EcoReport tool has been developed to facilitate the analysis of environmental impacts in the context of the development of EU product legislation for energy-using or energy-related products (under the MEErP). It is an Excel-based, simplified life-cycle based tool where methodological choices have been made, such as system boundaries and included parameters (Wesnaes and Hansen 2021). In the version used, the tool includes 15 impact categories (e.g. energy consumption, greenhouse gases and persistent organic pollutants) and five life cycle stages (raw materials, manufacturing, distribution, use, end-of-life).

The environmental impact estimates for specific cathode compositions from Lam et al. (2019) were used, but the tool was adapted to the proprietary assumptions on material quantities, battery capacities and lifetimes described above. The results of the environmental impact analysis per battery type are then linked to the stock analysis to generate market environmental impacts over the study period. The environmental impacts of the raw material, manufacturing and distribution phases are considered when the batteries enter the stock, the impacts of the use phase are distributed over the lifetime and the impacts of the end-of-life phase are considered when the batteries leave the stock.

In previous impact assessment studies, environmental impacts were calculated on a per product basis only and were not linked to the EU market scenario analysis. Linking environmental impacts and material composition directly to the stock model allows a dynamic assessment of the impact of all products on the EU market over a long period of time and the impact of the policy measures.

After applying the above assumptions to the model, the results are presented and discussed in the following section.

6.4. Results and discussion

The results provide an overview of the environmental and material impacts as well as the coherence of novel policy measures under the given assumptions.

With the trend towards larger battery capacities and growing sales of electric vehicles, the amount of materials in electric vehicles in the EU-27 stock increases and does not reach saturation until the mid-40s (Figure 32). The share of the most critical material in lithium-ion batteries, cobalt (according to the EU list of critical raw materials (Eynard et al. 2020a)), decreases in the newer cathode chemistries, but the market demand increases due to the growing battery demand.

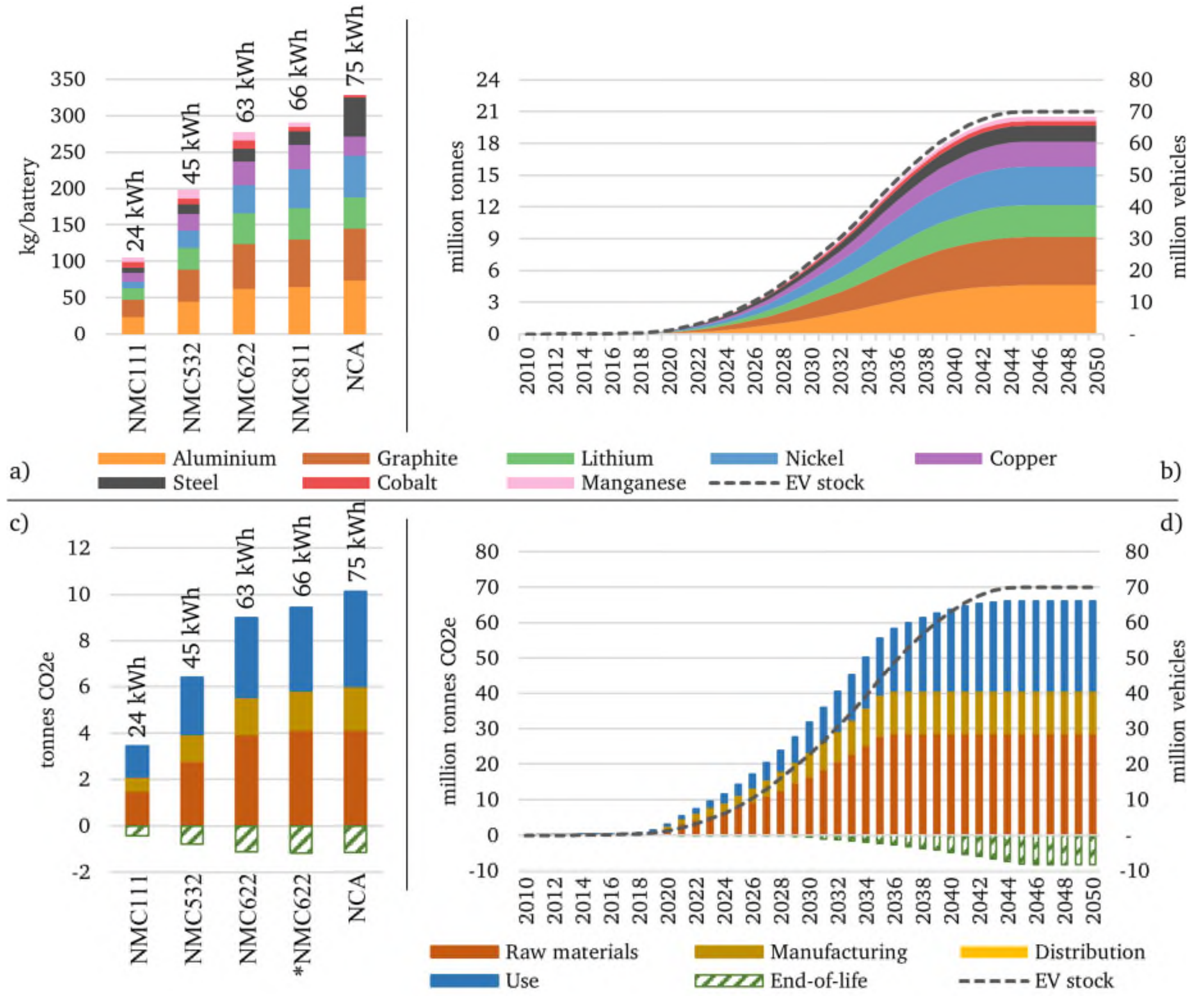


Figure 32 (a) Material composition of lithium-ion battery types. (b) Materials in electric vehicles in the EU-27 stock. (c) Greenhouse gas emissions of lithium-ion battery types over their life cycle and (d) cumulative EU-27 over modelling timeframe. * Due to missing data on the environmental impact of NMC811 cathodes, they are estimated as NMC622 cathodes with higher capacity.

In the environmental impact category of greenhouse gases, most emissions are generated in the raw material phase, which underlines the importance of an efficient recycling industry. As can be seen in Figure 32 d), the credits for avoided emissions due to recycling increase over time for the EU-27 analysis as the number of vehicles reaching their end-of-life increases.

Using the material-specific recycling efficiencies for 2031 (see Table 14) and the 96% collection rate, we can estimate the theoretical recycling potential and compare it with the demand for new batteries in that year. The results are shown in Figure 33. Under the assumptions made, the recycled content targets for all four materials can be well met in both 2031 and 2036 (Figure 33 a)).

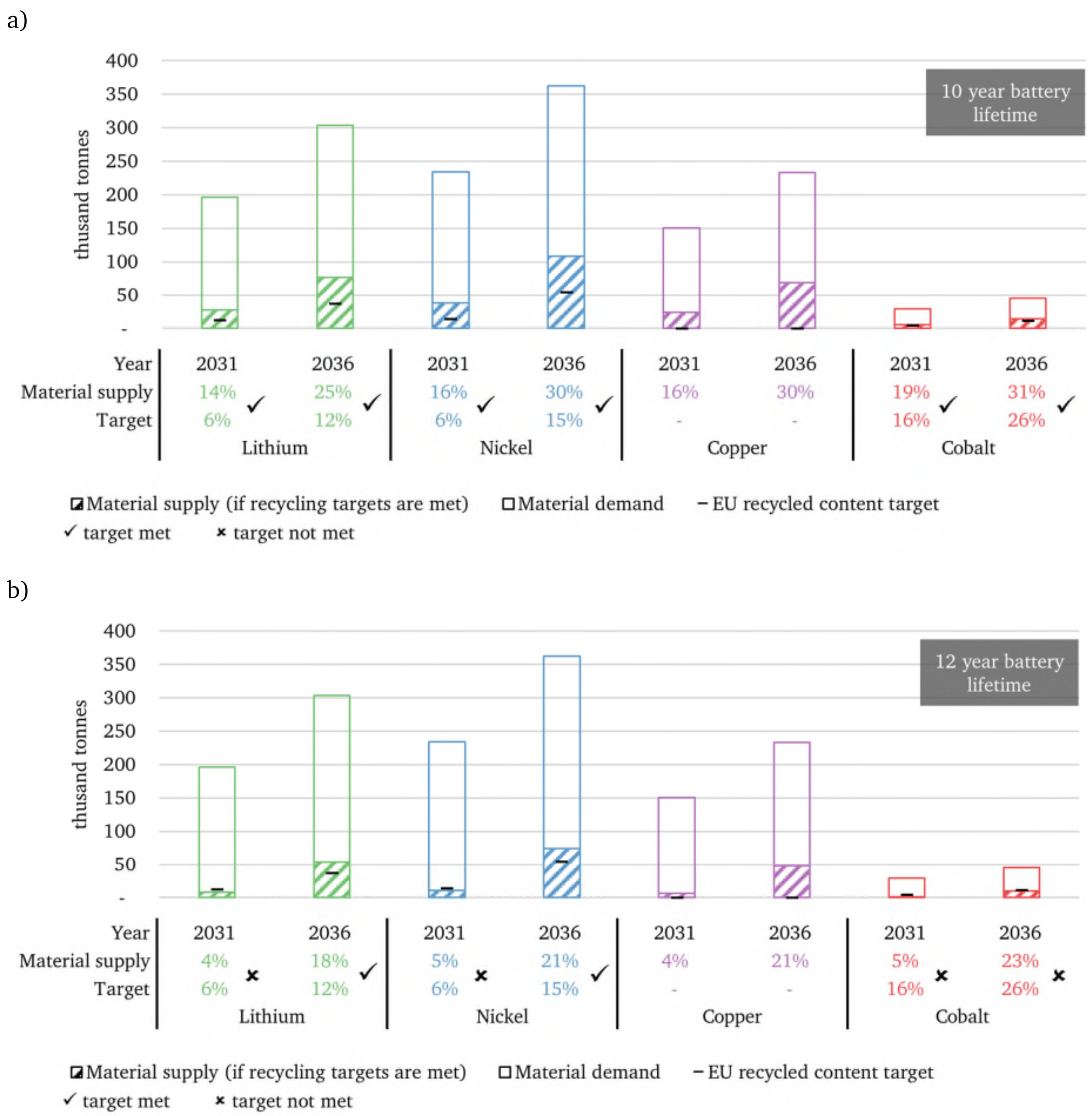


Figure 33 Calculated material supply and demand for lithium-ion battery materials in 2031 and 2036 compared to the recycled content targets of Regulation (EU) 2023/1542, calculated with lifetime of a) 10 years, b) 12 years.

If we now change the battery lifetime assumption, the results change significantly (Figure 33 b)). An increase in battery lifetime from 10 to just 12 years keeps materials in stock for longer and, due to the strong market growth, such a scenario would reduce the supply of materials from recycling to such an extent that the recycled content targets for lithium, nickel and cobalt could not be met in 2031 (considering closed loop recycling). In 2036, the targets for lithium and nickel could be met, but the target for cobalt could still not be met.

This is a critical finding, highlighting the importance of the underlying assumptions in the modelling and the difficulty of proposing policy targets. Now, for electric vehicle batteries, it is not only the longer lifetime that is important, but also the possibility of re-use. Once batteries fall below the performance requirements for use in electric vehicles (when they reach 80% of their original capacity, batteries are classified by the USABC as having reached their end-of-life (USABC 1996)), they may still retain sufficient energy performance for other applications where weight and volume are less critical factors, such as stationary storage systems (Chen et al. 2019). This repurposing involves disassembling the battery to the module or cell level, testing the components, replacing defective cells, and reconfiguring the battery system for its new purpose (Fan et al. 2020). For example, the batteries may function as a stationary energy storage system with moderate utilisation, such as a system that provides frequency regulation to the grid or buffers intermittent renewable energy, providing energy to the grid when demand is highest. While the use of batteries in electric vehicles is highly dependent on individual driving behaviour (mileage covered, charging patterns, etc.), stationary storage systems are expected to have a more stable load profile and benefit from regular maintenance. Some researchers suggest that batteries could last another 10 years in a second life application (Neubauer et al. 2015).

The second life battery market is still in its infancy, making it difficult to predict its adoption. Critical factors such as safety or lack of regulation and clear responsibilities are still being investigated. However, Regulation (EU) 2023/1542 specifically mentions the promotion of battery re-use, so it may well be that the time it takes for the average electric vehicle battery to become available for recycling will be delayed, making it difficult to meet ambitious targets for recycled content as electric vehicle sales continue to grow.

Even if longer lifetimes only delay the number of end-of-life batteries by a few years, the impact on business decisions and the development of the recycling industry can be massive. Recycling facilities in Europe will need to handle the volumes of end-of-life batteries, and ramping up recycling capacity can be a time-consuming process. Similarly, domestic battery producers will need a stable supply of raw materials, which can be secured through long-term contracts with raw material suppliers. If battery producers want or are even required to integrate the more sustainable option of using recycled materials, the variations in end-of-life batteries for different lifetimes make it difficult to predict the amounts of recycled materials that will be available (assuming closed loop recycling). The more sustainable option of using recycled materials may therefore become less attractive if supply risks are high due to unknown end-of-life battery streams.

The generalisability of the overall results is subject to certain limitations. For instance, while the data on electric vehicle sales was very recent, the data on cathode chemistries was from 2019 and did not reflect the latest developments. Similarly, the environmental impact assessment was based on old data and should be updated in future assessments. Missing data the lifetime of electric vehicle batteries reduces the reliability of the lifetime assumptions. Finally, the collection rate does not take into account the unreported export of end-of-life vehicles, which could significantly change the availability of end-of-life batteries.

Regardless of the uncertainties, the volumes of electric vehicle batteries entering the waste stream are expected to increase dramatically in the coming years, requiring the development of an end-of-life industry that can handle the volumes of batteries.

6.5. Conclusion

We have developed a combined MFA and environmental assessment model of the EU-27 passenger car battery electric vehicle market from 2011 to 2050 to assess the material and environmental impacts and the coherence of novel policy measures proposed in the new Battery Regulation (EU) 2023/1542.

Batteries are a product group where a high proportion of the environmental impacts can be attributed to the raw material phase, making an effective collection and recycling industry important from an environmental perspective. Recycling is also important from an economic perspective due to the high import dependency and supply risk associated with key cathode materials such as cobalt, lithium and nickel. With the expected continued growth of electric vehicle sales, the amount of materials that can potentially be recovered from end-of-life vehicle batteries will increase, but with a time lag due to the lifetime of the batteries. In particular for cobalt, where the mass fraction per battery is decreasing, recovery from end-of-life batteries, where the mass fraction was higher, could become an important material input stream for domestic battery production.

The new Battery Regulation (EU) 2023/1542 proposes recycling and recycled content targets for key materials. The analysis showed that, under assumption of a ten year battery lifetime, the recycled content targets could be met with recycled battery materials, thus enabling a closed loop material cycle. However, the results are highly sensitive to battery lifetime. Given the re-use of batteries, which can be considered beneficial from an environmental perspective, there may be a shortage of materials needed to meet the recycled content targets, which could lead to market distortions.

The analysis showed that there are trade-offs between different policy measures such as recycling efficiency, recycled content and re-use even if they would be beneficial in isolation. In a fast-growing market such as batteries, uncertainty about relevant factors such as lifetime introduces a large degree of uncertainty into the results.

Policy makers should be aware of such modelling uncertainties and consider trade-offs between policy measures in the light of the market saturation of the product group under consideration.

7. Synthesis

This final chapter provides a summary and discussion of the overall findings and their interdependencies. It begins with a summary of the main findings, highlighting the overarching storyline and the interdependencies between the chapters (7.1). The findings are then critically discussed in terms of strengths, limitations and needs for further research (7.2). Finally, a conclusion is provided (7.3).

7.1. Key results and interdependencies

This dissertation is located at the intersection of the three domains (or spheres) circular economy, legislation, and (energy-related) products, with the aim of improving the understanding of circular economy product legislation, including its impact assessment.

Figure 34 provides an overview of the research questions, interlinkages and selected key result figures.

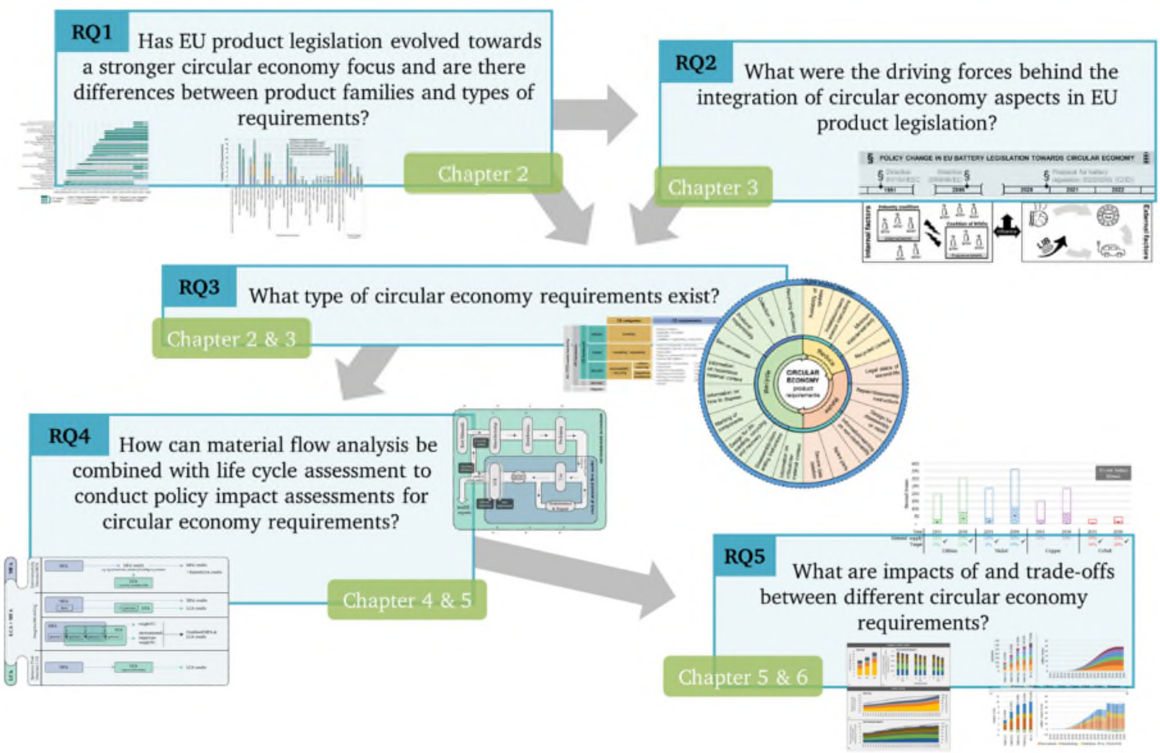


Figure 34 Overview, interdependencies and selected key result figures, linked to the research questions (RQ) and chapters of this dissertation.

The limits to human growth on a finite planet have been pointed out since the 1960s of the last century. Since then, science has advanced and concepts such as the planetary boundaries give us a better understanding of the severity of the situation than ever before in human history. Circular economy is seen by many as one of the strategies to contribute to sustainable development.

And indeed, the starting point for this dissertation was the apparent increasing importance of circular economy for EU policy-making, as evidenced by events such as the publication of the first and second Circular Economy Action Plan (European Commission 2015, 2020e), the explicit reference to the circular economy or resource efficiency in relevant EU legislation (e.g.

the Ecodesign Directive and battery legislation) and a growing societal interest, as indicated by the popularity of Google searches for the term "circular economy" in Figure 3.

By establishing the second Circular Economy Action Plan as one of the core pillars of the European Green Deal, the EU has clearly linked circular economy to the goal of achieving climate neutrality. Moreover, environmental impacts beyond greenhouse gases are greatly increased by our consumption, and energy-related products in particular contribute significantly to the fact that we, as Europeans, far exceed our per capita share for not crossing global planetary boundaries (Sala et al. 2019).

Given the growing relevance of circular economy in the EU's strategic goals, the question remains how this translates into concrete policies. The focus of interest in this dissertation has been on energy-related products. Energy-related products are responsible for large environmental impacts along their supply chains and consequently have been regulated in the EU for many years, but the focus of legislation has been mainly on reducing energy consumption during the use phase (Barkhausen et al. 2022). With the expected continued decarbonisation of the electricity mix, environmental impacts beyond the use phase are becoming increasingly important.

In fact, as the analysis of all legislative documents adopted under the Ecodesign Directive (for the 27 energy-related product groups regulated as of 2022) in chapter 2 showed, there is a clear increase in circular economy related requirements. This increase is most evident in the 2019 generation of ecodesign implementing measures, where a large number of requirements related to repair were implemented for the white goods product family.

Another important result of chapter 2 is the improved categorisation. At the beginning of this dissertation, the conceptual ambiguity of the term "circular economy" was pointed out. To reduce the ambiguity, a taxonomy of circular economy requirements was started, which was complemented by the requirements identified in the analysis of the EU battery legislation in chapter 3 (see Figure 17).

The dive into policy science in chapter 3 also rounded off the understanding of the underlying drivers for the increased consideration of circular economy requirements using the example of EU battery legislation. A hypothesis was derived that, based on a complete overhaul of the battery market with a shift to traction batteries with high EU import dependency and the automotive sector at risk, circular economy emerged as a common denominator aligning economic, environmental and social interests.

The knowledge of the increase in circular economy requirements, together with the taxonomy of requirements, provided the basis for asking how the environmental impact of such circular economy requirements could be estimated.

Already in chapter 1 of this dissertation, the scoping of impact assessment methods indicated that the combination of MFA and LCA could be a promising solution. To further investigate on this preliminary finding, in chapter 4 a systematic literature review was conducted on the existing combinations of MFA and LCA and their applicability to impact assessments with a focus on the environmental dimension.

The review showed not only many similarities between the two distinct research fields of MFA and LCA, the increasing number of publications and the wide variation in the type of modelling, but also that integrated modelling offers promising characteristics for assessing the long-term and multi-dimensional environmental impacts of circular economy policies. The combination allows to scale-up the in-depth environmental assessment of LCA by considering the interactions of material flows over long time periods and large spatial scopes of MFA.

Based on this finding, and on the identified gap in modelling tools (see subchapter 7.2), a model has been developed for the ex-ante assessment of the material flows and environmental impacts of a range of circular economy product policies. The model, presented in chapter 5, works as an integrator, filling the missing link in upscaling exogenously calculated product-level environmental impacts to the stock level to provide insights into market-wide policy impacts. The model uses a layered approach where material and environmental impacts are derived from product flows via a product database that defines physical properties for product variants.

To demonstrate its utility, the model has been applied in two case studies. Chapter 5 includes a case study on electric motors, a product group responsible for around 70% of global industrial energy consumption in 2019 (Almeida et al. 2023). Chapter 6 presents a case study on traction batteries, a product group essential for the decarbonisation of road transport.

For electric motors, chapter 5 provided scenarios to compare early replacement of inefficient motors and motor repair with a base case of no intervention to highlight the strengths and weaknesses of the model. Calculating the impact of policies inducing lifetime changes in a stock-driven logic requires methodological enhancements which were detailed in 5.2.2. The case study on batteries in chapter 6 tested the coherence of the novel policy measures proposed in the new Battery Regulation (EU) 2023/1542, demonstrating the importance of detailed impact assessment and the sensitivity and uncertainty of the results depending on the underlying assumptions such as average product lifetime, especially in a fast growing market such as traction batteries. A focus of the battery case study was on the analysis of the material flows, analysing the trade-offs between the policy measures of recycling efficiency, recycled content and re-use.

Overall, the two case studies showed that the results are very product group specific and some of the findings can be counterintuitive from a circular economy perspective. As the theoretical analysis shows, longer product lifetimes in the case of electric motors may not always be beneficial under the assumptions made (see chapter 5). Market maturity also has a potentially large impact. The analysis of traction batteries showed that under the assumptions of continued market uptake and long battery lifetime (which could well be achieved under the assumptions of battery repurposing) the recycled content targets may not be achievable with a closed-loop material stream of recycled traction batteries (see Figure 33).

7.2. Discussion and limitations

The area of circular economy product policy is a multifaceted and highly dynamic field that affects and interests several stakeholder groups such as the general public, industrial actors, policy makers and researchers.

The analysis of the Ecodesign Directive showed that many researchers are aware of the dynamics and challenges of integrating circular economy aspects into product policy (Barkhausen et al. 2022). EU legislation directly or indirectly affects us all and concerns many, as was clearly demonstrated by the high engagement of a wide range of public and private stakeholders in the policy-making process of the battery legislation (Barkhausen et al. 2023a).

The modelling challenges associated with moving from a focus on energy consumption in the use phase to all environmental impact categories throughout all life cycle stages have not gone unnoticed by policy makers. The Joint Research Centre has been commissioned by the European Commission to develop a new methodology for integrating circular economy aspects into policy impact assessment, updating the MEErP and the EcoReport tool (European Commission 2021c). The process is still ongoing - but shows the high relevance of the issue. It is known that the new EcoReport tool aligns the environmental impact categories with the 16 impact categories of the Product Environmental Footprint, improves the consideration of lifetime (e.g. the impact of repair or reuse on environmental impacts) and the allocation of impacts through the use of the Circular Footprint Formula. A dedicated tool to upscale the product level results to reflect the potential policy impacts on the EU market is not provided, but merely rough indications on how such analysis should be conducted. Those indications are not able to grasp the complexity of moving from one dimensional modelling with a focus on energy-consumption in the use phase to the multidimensional modelling required when considering all life cycle stages, various impact categories and different policy requirements. To this date it is not known if and how a new approach to scenario analysis, ergo up-scaling of the product specific results to the EU stock, will be developed. In light of the development of ecodesign to consider non-energy-related products, multidimensional modelling is urgently needed.

This is where the model developed in this dissertation makes its unique contribution. It extends the traditional stock model approach by adding material flows, allowing lifetime changes to be considered, and by upscaling exogenously calculated environmental impacts to the stock level. This makes it possible to calculate impacts over the entire stock and over the modelled time horizon, and to compare novel policy requirements such as reuse or recycled content. As shown in the case studies, the model can compare early replacement with repair, or set or verify recycled content targets. End-of-life policies calculated in an environmental assessment can be scaled up to the stock level to show trade-offs and support policy makers decision making. The model comes at a crucial time when new modelling approaches are needed.

Understandably, there are limitations to the model and to the research conducted in this dissertation in general. Therefore, at this point, some important limitations of this dissertation will be discussed, divided into aspects related to scope (product group, geographical and temporal) and methodologies.

The product scope was set on energy-related products, which have a high overall environmental impact and where the transition from energy efficiency to circular economy related requirements creates fundamental changes that pose new challenges and opportunities. The focus on energy-related products allowed a wide variety of existing legislation to be analysed in the light of its historical development, namely the regulations developed under the Ecodesign Directive and the three generations of battery legislation. However, the EU has announced its aim to extend the reign of ecodesign from energy-related to also non-energy-related products such as furniture and clothing (European Commission 2022c). This will bring new opportunities and challenges. While for many energy-related products the use phase remains the dominant hotspot in terms of environmental impacts (even though for already regulated product groups improvement potential might be small), this no longer applies for non-energy-related products, highlighting new modelling challenges such as normalisation and weighting of impact categories (see discussion on single score below). In addition, the taxonomy of circular economy requirements will need to be expanded as the product scope expands. In this dissertation, the focus of the impact assessment was on the assessment of environmental impacts and material flows. Future analysis could extend the approach to include, for example, detailed economic modelling.

In terms of geographical scope, the dissertation focused on the EU, where the transition to a more circular economy has been clearly stated as one of the strategic objectives (European Commission 2015, 2020e). While limiting the geographical scope allowed for a more detailed analysis, it is of academic interest to look at countries outside the EU as well. As part of the second Circular Economy Action Plan, the EU launched the Global Alliance of Circular Economy in 2021 and circular economy, as a core building block for sustainable development, plays an important role in many of the UN Sustainable Development Goals (such as Goal 12 on responsible consumption and production). Many countries outside the EU have also adopted circular economy related strategies or legislation. China, for example, was an early adopter of legislation on environmental pollution and cleaner production, adopting a specific law to promote the circular economy as early as 2008 (Standing Committee of the National People's Congress 2008). Compared to China and the EU, the US appears to be late in developing a holistic policy framework for the circular economy. The term circular economy was only defined in 2020 in the Save our Seas 2.0 Act to address plastic pollution (U.S.C. 2020) and a circular economy strategy series is still under development. As this is anecdotal evidence, it is interesting to look more closely at countries beyond the EU to analyse, understand and compare global developments.

In terms of the point of time in the policy-making process that could be defined as the temporal scope, this dissertation has focused on ex-ante impact assessment. In addition to analysing the potential impact of policies before they are adopted, the ex-post evaluation of policies that have already been implemented is and will remain very relevant. We are at a crucial juncture where the magnitude, methods and measures of impact assessment are being overhauled. Policy makers in a democracy have to navigate between stakeholder support, own values and societal balance. In this difficult endeavour, the role of researchers is to provide decision support (e.g., methodologies and analyses) that is as unbiased as possible and to be transparent about the limitations of the work. Therefore, the methodology for evaluating past policies also needs to be revised due to the discussed changes in the policy process to evaluate and learn from the mistakes of previous ex-ante assessments.

The remainder of the discussion is a closer look at the underlying methodologies.

A wide range of different methods were used, including qualitative, quantitative and case study methods, each of which has its strengths and weaknesses, some of which are highlighted below.

The content analysis of legislative texts is limited by its qualitative nature, so exact figures should be treated with caution. Nevertheless, the overall trend was evident and, together with the keyword analysis, provides a sufficiently reliable understanding. Less reliable are the findings in the area of political science, where the Advocacy Coalition Framework was applied. A limitation here was the small sample size of respondents, which reduces the generalisability of the findings. In addition, it was difficult to find information about the policy-making process in the early 2000s, when information flows were less digitised. Hypotheses were derived based on the blind analysis of written stakeholder positions, but further analysis would be required to derive more reliable findings. Overall, it could also be hypothesised that the field of political science is an area where findings are inevitably less salient compared to more technical analysis.

In any case, the findings provide a valuable complement to put the more technical aspects of this dissertation into perspective.

The systematic literature review followed the PRISMA scheme, which provides a clear and transparent way of conducting the analysis and reporting the results. Despite this structured approach, it cannot be ruled out that important publications may have been overlooked. The most relevant scientific databases were searched, but it is not unlikely that not all authors explicitly referred to the terms MFA and LCA, which were used as key search terms. Studies combining MFA with EIA without performing a full LCA may not have used the keyword LCA and therefore may have been missed by our systematic search. Conversely, the results showed that many studies use the term LCA very liberally, even when it is not obvious that the required steps of the ISO standard have been followed. The range of what is considered an LCA varies widely between studies. It is therefore important to emphasise that in our classification of combined LCA and MFA approaches, LCA refers to both ISO standard analyses and simplified (based on life cycle thinking) environmental assessments. It can only be emphasised that a clear understanding of what constitutes LCA should be a prerequisite for labelling one's own analysis as LCA. As our analysis has shown, the fields of LCA and MFA are increasingly merging, making it even more important to be consistent in nomenclature and classifications to avoid misunderstandings.

In particular, further research could explore the combination of LCA and MFA with additional methodologies such as input-output analysis, which could provide elegant solutions for top-down assessments of material flows and environmental impacts across economic sectors. Looking beyond mere environmental impacts, life-cycle sustainability assessment comes into consideration. In particular, when evaluating circular economy policies, social and economic impacts could be considered and incorporated into further analysis.

The results of the systematic literature review, although it may have missed some publications, provided a solid basis for deriving a methodology for assessing the impacts of circular economy policies. As Box (1976) noted, all models are flawed and require realism and humility when interpreting results. However, models can still be useful. Attempting to represent a hypothetical future can support decision making and help make science-based decisions, but there is never certainty, and sometimes going into too much detail can overcomplicate regulations that end up doing more harm than good.

For the model developed in this dissertation, some important considerations come to mind, such as the processing speed of the modelling. Currently, the environmental analysis is performed for a single point in time, which leads to time-consuming database traversal when many product variants are being assessed, and makes it difficult to account for temporal changes in environmental impacts (e.g. changes in the electricity mix over time). In addition, the computational effort is high due to the multi-dimensionality of product and material flows with many environmental impact categories for different life stages over long modelling periods. A shift of the modelling logic to a more powerful software environment seems inevitable to efficiently model and compare a large number of scenario variations.

Considering that the modelling is intended to improve decision making, the normalisation and weighting of the impact categories comes into focus. As the hotspot of environmental impact moves away from energy use in the use phase, a balance must be struck between different impact categories - impact categories that differ greatly in the robustness of their quantification.

Which should we prioritise? Ultimately, this is a political decision. More research should be done on a single environmental impact score to see if the advantages could outweigh the disadvantages and to assess the impact its use would have on the policy recommendation.

In addition, the environmental benefits of circular economy may be delayed until end of the products life cycle and therefore less appealing to policy makers compared to the direct credits of an energy efficiency policy. On top of that, energy efficiency policies often reduce costs for consumers in the use phase due to lower energy consumption, offsetting higher purchasing costs. In contrast, circular economy policies may lead to higher overall costs for the consumer, depending, for example, on the availability and price for secondary materials. Here, the concept of societal costs (taking into account externalities) could play an important role in motivating the adoption of circular economy legislation.

7.3. Concluding remarks

This dissertation project aimed to improve the understanding of the intersection between energy-related products, legislation and the circular economy.

It was shown that EU product legislation is evolving towards a stronger focus on circular economy, with differences between product families and circular economy categories. A hypothesis was derived that circular economy could act as a common denominator to align economic and environmental interests. A taxonomy of circular economy requirements was proposed based on the analysis of the Ecodesign Directive and battery legislation, and it was shown that combined modelling of MFA and LCA has suitable characteristics to catch the complexity and model the material flow and environmental impacts of such requirements. To fill a methodological void in product policy impact assessment, a model was developed and applied to case studies on industrial electric motors and traction batteries, showing the range of policy requirements that can be assessed and the multidimensionality of the results.

Further analysis can extend the scope beyond energy-related products, beyond the EU and beyond ex-post evaluations. In addition, certain aspects of the methods and modelling can be further improved, such as the integration of the dynamic evolution of environmental impacts and the weighting of impacts.

This analysis takes place in a dynamic time and place, and it will be exciting to follow the ongoing policy and scientific developments in the light of its findings. This brings the dissertation full circle.

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Abbreviations

ACF	Advocacy coalition framework
EU	Europe Union
ICT	Information and communication technologies
ISO	International organization for standardization
LCA	Life cycle assessment
LFP	Lithium iron phosphate ($LiFePO_4$)
LMO	Lithium ion manganese oxide ($LiMn_2O_4$)
MEErP	Methodology for ecodesign of energy-related products
MFA	Material flow analysis
NCA	Lithium nickel cobalt aluminium oxide ($LiNiCoAlO_2$)
NGO	Non-governmental organisation
NMA	Lithium nickel manganese cobalt (LiNiMnCo)
RQ	Research question

55 European organisations (2018): Open letter on package of Ecodesign and Energy Labelling regulations to President European Commission Jean-Claude Juncker. Brussels, 9/26/2018. Open Letter.

Almeida, Anibal T. de; Ferreira, Fernando J. T. E.; Fong, João (2023): Perspectives on Electric Motor Market Transformation for a Net Zero Carbon Economy. In *Energies* 16 (3), p. 1248. DOI: 10.3390/en16031248.

Almeida, Aníbal T. de; Ferreira, Fernando J.T.E.; Fong, Joao; Fonseca, Paula (2008): EUP Lot 11 Motors. ISR-University of Coimbra. Available online at https://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=&cad=rja&uact=8&ved=2ahUKEwiI3O_Bt9mBAxU77LsIHZ6hCLIQFnoECBQQAQ&url=https%3A%2F%2Fcircabc.europa.eu%2Fsd%2Fd%2F62415be2-3d5a-4b3f-b29a-d1760f4dc11a%2FLot11&usg=AOvVaw1wX5Tve7aa7XnhGYpWjo-r&opi=89978449.

Anandh, Gurunathan; PrasannaVenkatesan, Shanmugam; Goh, Mark; Mathiyazhagan, Kaliyan (2021): Reuse assessment of WEEE: Systematic review of emerging themes and research directions. In *Journal of environmental management* 287, p. 112335. DOI: 10.1016/j.jenvman.2021.112335.

Ardente, Fulvio; Mathieux, Fabrice (2012): Integration of resource efficiency and waste management criteria in European product policies - Second phase. Report n° 2 Application of the project's methods to three product groups. Edited by Joint Research Centre of the European Commission.

Ardolino, Filomena; Colaleo, Giuseppina; Arena, Umberto (2020): The cleaner option for energy production from a municipal solid biowaste. In *Journal of Cleaner Production* 266, p. 121908. DOI: 10.1016/j.jclepro.2020.121908.

Arvidsson, Rickard; Tillman, Anne-Marie; Sandén, Björn A.; Janssen, Matty; Nordelöf, Anders; Kushnir, Duncan; Molander, Sverker (2018): Environmental Assessment of Emerging Technologies: Recommendations for Prospective LCA. In *Journal of Industrial Ecology* 22 (6), pp. 1286–1294. DOI: 10.1111/jiec.12690.

Aryapratama, Rio; Pauliuk, Stefan (2022): Life cycle carbon emissions of different land conversion and woody biomass utilization scenarios in Indonesia. In *The Science of the total environment* 805, p. 150226. DOI: 10.1016/j.scitotenv.2021.150226.

B. Müller, Daniel (2006): Stock dynamics for forecasting material flows—Case study for housing in The Netherlands. In *Ecological Economics* 59 (1), pp. 142–156. DOI: 10.1016/j.ecolecon.2005.09.025.

Baars, Joris; Domenech, Teresa; Bleischwitz, Raimund; Melin, Hans Eric; Heidrich, Oliver (2020): Circular economy strategies for electric vehicle batteries reduce reliance on raw materials. In *Nature sustainability*. DOI: 10.1038/s41893-020-00607-0.

Baars, Joris; Rajaeifar, Mohammad Ali; Heidrich, Oliver (2022): Quo vadis MFA? Integrated material flow analysis to support material efficiency. In *Journal of Industrial Ecology*, Article jiec.13288. DOI: 10.1111/jiec.13288.

Barkhausen, Robin (2024): Coherence of Novel Policies for Lithium-Ion Batteries for Electric Vehicles: A Multidimensional Analysis of Material Flows and Environmental Impacts. In *eceee 2024 Summer Study proceedings*. DOI: 10.24406/publica-3334.

Barkhausen, Robin; Durand, Antoine; Fick, Katharina (2022): Review and Analysis of Ecodesign Directive Implementing Measures: Product Regulations Shifting from Energy Efficiency towards a Circular Economy. In *Sustainability* 14 (16), p. 10318. DOI: 10.3390/su141610318.

Barkhausen, Robin; Durand, Antoine; Fong, Yan Yi; Zeller, Vanessa (2024): Modeling stock, material and environmental impacts of circular economy product policies. Trade-offs between early replacement and repair of electric motors. In *Resources, Conservation and Recycling* 205, p. 107600. DOI: 10.1016/j.resconrec.2024.107600.

Barkhausen, Robin; Fick, Katharina; Durand, Antoine; Rohde, Clemens (2023a): Analysing policy change towards the circular economy at the example of EU battery legislation. In *Renewable and Sustainable Energy Reviews* 186, p. 113665. DOI: 10.1016/j.rser.2023.113665.

Barkhausen, Robin; Rostek, Leon; Miao, Zoe Chunyu; Zeller, Vanessa (2023b): Combinations of material flow analysis and life cycle assessment and their applicability to assess circular economy requirements in EU product regulations. A systematic literature review. In *Journal of Cleaner Production* 407, p. 137017. DOI: 10.1016/j.jclepro.2023.137017.

Bertoldi, Paolo (Ed.) (2021): Energy efficiency in motor systems. Proceedings of the 11th International Conference EEMODS'19. Cham: Springer (Springer Proceedings in Energy).

Bertram, M.; Ramkumar, S.; Rechberger, H.; Rombach, G.; Bayliss, C.; Martchek, K. J. et al. (2017): A regionally-linked, dynamic material flow modelling tool for rolled, extruded and cast aluminium products. In *Resources, Conservation and Recycling* 125, pp. 48–69. DOI: 10.1016/j.resconrec.2017.05.014.

BIO Intelligence Service (2003): Impact Assessment on Selected Policy Options for Revision of the Battery Directive. Final Report. With assistance of Eric Labouze. Edited by Directorate General Environment, European Commission.

Birat, Jean-Pierre (2020): MFA vs. LCA, particularly as environment management methods in industry: an opinion. In *Matériaux & Techniques* 108 (5-6), p. 503. DOI: 10.1051/mattech/2021004.

BMU (1996): Press release: Das Kreislaufwirtschaftsgesetz tritt in Kraft. Dr. Angela Merkel: Neues Abfallrecht ein Beitrag für eine nachhaltige Umweltpolitik. Available online at https://www.bmuv.de/pressemitteilung/das-kreislaufwirtschaftsgesetz-tritt-in-kraft, checked on 1/27/2022.

Bobba, Silvia; Ardente, Fulvio; Mathieux, Fabrice (2016): Environmental and economic assessment of durability of energy-using products: Method and application to a case-study vacuum cleaner. In *Journal of Cleaner Production* 137, pp. 762–776. DOI: 10.1016/j.jclepro.2016.07.093.

Bobba, Silvia; Bianco, Isabella; Eynard, Umberto; Carrara, Samuel; Mathieux, Fabrice; Blengini, Gian Andrea (2020): Bridging Tools to Better Understand Environmental Performances and Raw Materials Supply of Traction Batteries in the Future EU Fleet. In *Energies* 13 (10), p. 2513. DOI: 10.3390/en13102513.

Bobba, Silvia; Mathieux, Fabrice; Blengini, Gian Andrea (2019): How will second-use of batteries affect stocks and flows in the EU? A model for traction Li-ion batteries. In *Resources, Conservation and Recycling* 145, pp. 279–291. DOI: 10.1016/j.resconrec.2019.02.022.

Boldoczki, Sandra; Thorenz, Andrea; Tuma, Axel (2021): Does increased circularity lead to environmental sustainability?: The case of washing machine reuse in Germany. In *Journal of Industrial Ecology* 25 (4), pp. 864–876. DOI: 10.1111/jiec.13104.

Bonsu, Nana O. (2020): Towards a circular and low-carbon economy: Insights from the transitioning to electric vehicles and net zero economy. In *Journal of Cleaner Production* 256, p. 120659. DOI: 10.1016/j.jclepro.2020.120659.

Boulding, Kenneth E. (1966): The Economics of the Coming Spaceship Earth. In *Environmental Quality in a Growing Economy*. Available online at http://arachnid.biosci.utexas.edu/courses/THOC/Readings/Boulding_SpaceshipEarth.pdf.

Box, George E. P. (1976): Science and Statistics. In *Journal of the American Statistical Association* 71 (356), pp. 791–799. DOI: 10.1080/01621459.1976.10480949.

Bracquené, Ellen; Brussalaers, Jan; Dams, Yoko; Peeters, Jef; Schepper, Karel de; Duflou, Joost; Dewulf, Wim (2018): Repairability criteria for energy related products. Study in the BeNeLux context to evaluate the options to extend the product life time.

Brignon, Jean-Marc (2021): Costs and benefits of recycling PVC contaminated with the legacy hazardous plasticizer DEHP. In *Waste management & research : the journal of the International Solid Wastes and Public Cleansing Association, ISWA* 39 (9), pp. 1185–1192. DOI: 10.1177/0734242X211006755.

Brunner, Paul H.; Rechberger, Helmut (2017): Handbook of Material Flow Analysis: For Environmental, Resource, and Waste Engineers, Second Edition. Edited by Brunner: CRC Press.

Bundgaard, Anja Marie; Mosgaard, Mette Alberg; Remmen, Arne (2017): From energy efficiency towards resource efficiency within the Ecodesign Directive. In *Journal of Cleaner Production* 144, pp. 358–374. DOI: 10.1016/j.jclepro.2016.12.144.

Camana, Daniela; Manzardo, Alessandro; Toniolo, Sara; Gallo, Federico; Scipioni, Antonio (2021): Assessing environmental sustainability of local waste management policies in Italy from a circular economy perspective. An overview of existing tools. In *Sustainable Production and Consumption* 27, pp. 613–629. DOI: 10.1016/j.spc.2021.01.029.

Castellani, Valentina; Sanyé-Mengual, Esther; Sala, Serenella (2021): Environmental impacts of household goods in Europe: a process-based life cycle assessment model to assess consumption footprint. In *Int J Life Cycle Assess* 26 (10), pp. 2040–2055. DOI: 10.1007/s11367-021-01987-x.

Charpentier Poncelet, Alexandre; Helbig, Christoph; Loubet, Philippe; Beylot, Antoine; Muller, Stéphanie; Villeneuve, Jacques et al. (2021): Life cycle impact assessment methods for estimating the impacts of dissipative flows of metals. In *Journal of Industrial Ecology* 25 (5), pp. 1177–1193. DOI: 10.1111/jiec.13136.

Chen, Mengyuan; Ma, Xiaotu; Chen, Bin; Arsenault, Renata; Karlson, Peter; Simon, Nakia; Wang, Yan (2019): Recycling End-of-Life Electric Vehicle Lithium-Ion Batteries. In *Joule* 3 (11), pp. 2622–2646. DOI: 10.1016/j.joule.2019.09.014.

Chu, Jianwen; Zhou, Ya; Cai, Yanpeng; Wang, Xuan; Li, Chunhui; Liu, Qiang (2022): Life-cycle greenhouse gas emissions and the associated carbon-peak strategies for PS, PVC, and ABS plastics in China. In *Resources, Conservation and Recycling* 182, p. 106295. DOI: 10.1016/j.resconrec.2022.106295.

Ciacci, L.; Fishman, T.; Elshkaki, A.; Graedel, T. E.; Vassura, I.; Passarini, F. (2020): Exploring future copper demand, recycling and associated greenhouse gas emissions in the EU-28. In *Global Environmental Change* 63, p. 102093. DOI: 10.1016/j.gloenvcha.2020.102093.

Cordella, Mauro; Alfieri, Felice; Sanfelix, Javier (2019): Analysis and development of a scoring system for repair and upgrade of products: final report: Publications Office.

Corona, Blanca; Shen, Li; Reike, Denise; Rosales Carreón, Jesús; Worrell, Ernst (2019): Towards sustainable development through the circular economy—A review and critical assessment on current circularity metrics. In *Resources, Conservation and Recycling* 151, p. 104498. DOI: 10.1016/j.resconrec.2019.104498.

Council of the European Communities (1991): Council Directive of 18 March 1991 on batteries and accumulators containing certain dangerous substances. 91/157/EEC. In *Official Journal of the European Communities*. Available online at https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:31991L0157&from=EN.

CRediT (2023): CRediT. Available online at https://credit.niso.org/, updated on 10/24/2023, checked on 10/24/2023.

Cucurachi, Stefano; van der Giesen, Coen; Guinée, Jeroen (2018): Ex-ante LCA of Emerging Technologies. In *Procedia CIRP* 69, pp. 463–468. DOI: 10.1016/j.procir.2017.11.005.

Cullen, Jonathan M.; Allwood, Julian M.; Bambach, Margarita D. (2012): Mapping the global flow of steel: from steelmaking to end-use goods. In *Environmental science & technology* 46 (24), pp. 13048–13055. DOI: 10.1021/es302433p.

D'Amico, Bernardino; Pomponi, Francesco; Hart, Jim (2021): Global potential for material substitution in building construction: The case of cross laminated timber. In *Journal of Cleaner Production* 279, p. 123487. DOI: 10.1016/j.jclepro.2020.123487.

Dalhammar, Carl (2014): Promoting Energy and Resource Efficiency through the Ecodesign Directive. Edited by Scandinavian Studies in Law. Available online at https://www.scandinavianlaw.se/pdf/59-4.pdf.

Dalhammar, Carl (2016): Industry attitudes towards ecodesign standards for improved resource efficiency. In *Journal of Cleaner Production* 123, pp. 155–166. DOI: 10.1016/j.jclepro.2015.12.035.

Dalhammar, Carl; Machacek, Erika; Bundgaard, Anja Marie; Overgaard Zacho, Kristina; Remmen, Arne (2014): Adressing resource efficiency through the Ecodesign Directive. Edited by Nordic Council of Ministers.

D'Amato, Alessio; Mazzanti, Massimiliano; Nicolli, Francesco (2021): Green technologies and environmental policies for sustainable development: Testing direct and indirect impacts. In *Journal of Cleaner Production* 309, p. 127060. DOI: 10.1016/j.jclepro.2021.127060.

Di Dong; Tukker, Arnold; Steubing, Bernhard; van Oers, Lauran; Rechberger, Helmut; Alonso Aguilar-Hernandez, Glenn et al. (2022): Assessing China's potential for reducing primary copper demand and associated environmental impacts in the context of energy transition and "Zero waste" policies. In *Waste management (New York, N.Y.)* 144, pp. 454–467. DOI: 10.1016/j.wasman.2022.04.006.

Di Maio, Francesco; Rem, Peter Carlo; Baldé, Kees; Polder, Michael (2017): Measuring resource efficiency and circular economy: A market value approach. In *Resources, Conservation and Recycling* 122, pp. 163–171. DOI: 10.1016/j.resconrec.2017.02.009.

EAFO (2023): Vehicles and fleet | European Alternative Fuels Observatory. Available online at https://alternative-fuels-observatory.ec.europa.eu/transport-mode/road/european-union-eu27/vehicles-and-fleet, updated on 12/18/2023, checked on 1/16/2024.

EC-JRC (2010): International Reference Life Cycle Data System (ILCD) Handbook - General guide for Life Cycle Assessment - Detailed guidance. Luxembourg: Publications Office (EUR (Luxembourg), 24708).

Economidou, M.; Todeschi, V.; Bertoldi, P.; D'Agostino, D.; Zangheri, P.; Castellazzi, L. (2020): Review of 50 years of EU energy efficiency policies for buildings. In *Energy and Buildings* 225, p. 110322. DOI: 10.1016/j.enbuild.2020.110322.

Egenhofer, Christian; Drabik, Eleanor; Alessi, Monica; Rizos, Vasileios (2018): Stakeholders' Views on the Ecodesign Directive. An assessment of the successes and shortcomings. Edited by CEPS Energy Climate House.

Elia, Valerio; Gnoni, Maria Grazia; Tornese, Fabiana (2017): Measuring circular economy strategies through index methods: A critical analysis. In *Journal of Cleaner Production* 142, pp. 2741–2751. DOI: 10.1016/j.jclepro.2016.10.196.

Elijošiutė, Erika; Balciukevičiūtė, Jolita; Denafas, Gintaras (2012): Life Cycle Assessment of Compact Fluorescent and Incandescent Lamps: Comparative Analysis. In *EREM* 61 (3). DOI: 10.5755/j01.erem.61.3.2425.

IEC Guide 109:2012, 2012: Environmental aspects – Inclusion in electrotechnical product standards. Available online at https://webstore.iec.ch/publication/7520.

ISO 14040:2006, 2006: Environmental management - Life cycle assessment - Principles and framework. Available online at https://www.iso.org/standard/37456.html.

ISO 14044:2006, 2006: Environmental management - Life cycle assessment - Requirements and guidelines. Available online at https://www.iso.org/standard/38498.html.

European Commission (1978): Council Directive of 13 February 1978 on the performance of heat generators for space heating and the production of hot water in new or existing non-industrial buildings and on the insulation of heat and domestic hot-water distribution in new non-industrial buildings. 78/170/EEC.

European Commission (1979a): Council Directive 79/530/EEC of 14 May 1979 on the indication by labelling of the energy consumption of household appliances. 79/530/EEC. Available online at https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A31979L0530.

European Commission (1979b): Council Directive 79/531/EEC of 14 May 1979 applying to electric ovens Directive 79/530/EEC on the indication by labelling of the energy consumption of household appliances. 79/531/EEC. Available online at https://eur-lex.europa.eu/legal-content/EN/ALL/?uri=CELEX%3A31979L0531.

European Commission (1992a): Council Directive 92/42/EEC of 21 May 1992 on efficiency requirements for new hot-water boilers fired with liquid or gaseous fuels. 92/42/EEC. Available online at https://eur-lex.europa.eu/legal-content/EN/ALL/?uri=celex%3A31992L0042.

European Commission (1992b): Council Directive 92/75/EEC of 22 September 1992 on the indication by labelling and standard product information of the consumption of energy and other resources by household appliances. 92/75/EEC. Available online at https://eur-lex.europa.eu/legal-content/EN/ALL/?uri=CELEX%3A31992L0075.

European Commission (1996): Directive 96/57/EC of the European Parliament and of the Council of 3 September 1996 on energy efficiency requirements for household electric refrigerators, freezers and combinations thereof. 96/57/EC. Available online at https://eur-lex.europa.eu/legal-content/en/ALL/?uri=CELEX%3A31996L0057.

European Commission (2000): Directive 2000/55/EC of the European Parliament and of the Council of 18 September 2000 on energy efficiency requirements for ballasts for fluorescent lighting. 2000/55/EC. Available online at https://eur-lex.europa.eu/legal-content/EN/ALL/?uri=CELEX%3A32000L0055.

European Commission (2003): Consultation Document on the Battery Directive Revision. Brussels. Available online at https://www.batteri.dk/hotel/batteri/webpages.nsf/ba9f50795226e2afc12565b5005aa2de/9c6a5a8fd40b6c7ec1256d17003ba3d7/$FILE/consultation%20doc%20on%20Bat%20Dir%20rev.pdf.

European Commission (2005): Directive 2005/32/EC of the European Parliament and of the Council of 6 July 2005 establishing a framework for the setting of ecodesign requirements for energy-using products and amending Council Directive 92/42/EEC and Directives 96/57/EC and 2000/55/EC of the European Parliament and of the Council. 2005/32/EC. Available online at https://eur-lex.europa.eu/legal-content/en/ALL/?uri=CELEX%3A32005L0032.

European Commission (2006): Directive 2006/66/EC of the European Parliament and of the Council of 6 September 2006 on batteries and accumulators and waste batteries and accumulators and repealing Directive 91/157/EEC. Battery Directive. Source: https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:02006L0066-20131230&rid=1. Available online at https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:02006L0066-20131230&rid=1, checked on 6/30/2020.

European Commission (2008): Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives. 2008/98/EC. Available online at https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A02008L0098-20180705.

European Commission (2009): Directive 2009/125/EC of the European Parliament and of the Council of 21 October 2009 establishing a framework for the setting of ecodesign requirements for energy-related products. 2009/125/EC. Available online at https://eur-lex.europa.eu/legal-content/EN/ALL/?uri=celex%3A32009L0125.

European Commission (2010): Directive 2010/30/EU of the European Parliament and of the Council of 19 May 2010 on the indication by labelling and standard product information of the consumption of energy and other resources by energy-related products. 2010/30/EU. Available online at https://eur-lex.europa.eu/legal-content/EN/ALL/?uri=CELEX%3A32010L0030.

European Commission (2011): Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions - Roadmap to a Resource Efficient Europe. COM(2011) 571, checked on https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:52011DC0571&from=EN.

European Commission (2012a): Directive 2012/19/EU of the European Parliament and of the Council of 4 July 2012 on waste electrical and electronic equipment (WEEE) (recast). 2012/19/EU. Available online at https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:02012L0019-20180704.

European Commission (2012b): Single Market for Green Products - Environment - European Commission. Available online at https://ec.europa.eu/environment/eussd/smgp/index.htm, updated on 7/5/2022, checked on 9/21/2022.

European Commission (2015): Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions - Closing the loop - An EU action plan for the Circular Economy. COM(2015) 614. Available online at https://eur-lex.europa.eu/resource.html?uri=cellar:8a8ef5e8-99a0-11e5-b3b7-01aa75ed71a1.0012.02/DOC_1&format=PDF.

European Commission (2016): Communication from the Commission - Ecodesign Working Plan 2016-2019. COM(2016) 773. Available online at https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:52016DC0773&from=EN.

European Commission (2017): Regulation (EU) 2017/1369 of the European Parliament and of the Council of 4 July 2017 setting a framework for energy labelling and repealing Directive 2010/30/EU. Available online at https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32017R1369&from=EN.

European Commission (2018): Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. Europe on the Move. Sustainable Mobility for Europe: safe, connected, and clean. COM(2018) 293 final. Available online at https://eur-lex.europa.eu/resource.html?uri=cellar%3A0e8b694e-59b5-11e8-ab41-01aa75ed71a1.0003.02/DOC_1&format=PDF.

European Commission (2019a): Commission Regulation (EU) 2019/1781 of 1 October 2019 laying down ecodesign requirements for electric motors and variable speed drives pursuant to Directive 2009/125/EC of the European Parliament and of the Council, amending Regulation (EC) No 641/2009 with regard to ecodesign requirements for glandless standalone circulators and glandless circulators integrated in products and repealing Commission Regulation (EC) No 640/2009. (EU) 2019/1781. In *Official Journal of the European Union*. Available online at https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex%3A32019R1781.

European Commission (2019b): Report from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions.

on the implementation and the impact on the environment and the functioning of the internal market of Directive 2006/66/EC of the European Parliament and of the Council of 6 September 2006 on batteries and accumulators and waste batteries and accumulators and repealing Directive 91/157/EEC. Brussels.

European Commission (2019c): The European Green Deal. Communication from the Commission to the European Parliament, the European Council, the Council, the European Economic and Social Committee and the Committee of the Regions. Available online at https://eur-lex.europa.eu/resource.html?uri=cellar:b828d165-1c22-11ea-8c1f-01aa75ed71a1.0002.02/DOC_1&format=PDF, checked on 3/3/2020.

European Commission (2020a): Batteries - modernising EU rules. Feedback and statistics: Inception impact assessment. Available online at https://ec.europa.eu/info/law/better-regulation/have-your-say/initiatives/12399-Batteries-modernising-EU-rules/feedback_en?p_id=7925105, updated on 2/8/2023, checked on 2/15/2023.

European Commission (2020b): European Battery Alliance - Internal Market, Industry, Entrepreneurship and SMEs - European Commission. Available online at https://ec.europa.eu/growth/industry/policy/european-battery-alliance_en, updated on 5/8/2018, checked on 6/10/2020.

European Commission (2020c): Proposal for a Regulation of the European Parliament and of the Council concerning batteries and waste batteries, repealing Directive 2006/66/EC and amending Regulation (EU) No 2019/1020. Battery Directive. Source: https://ec.europa.eu/environment/waste/batteries/pdf/Proposal_for_a_Regulation_on_batteries_and_waste_batteries.pdf. Available online at https://ec.europa.eu/environment/waste/batteries/pdf/Proposal_for_a_Regulation_on_batteries_and_waste_batteries.pdf, checked on 1/10/2021.

European Commission (2020d): Rules of Procedure of the Regulatory Scrutiny Board. Ref. Ares(2020)2060246. Available online at https://ec.europa.eu/info/sites/default/files/rsb_rules_of_procedures_-_revision_2020_en.pdf, checked on 2/3/2022.

European Commission (Ed.) (2020e): Communication from the Commision to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions on a new Circular Economy Action Plan for a cleaner and more competitive Europe (COM(2020) 98 final). Available online at https://eur-lex.europa.eu/resource.html?uri=cellar:9903b325-6388-11ea-b735-01aa75ed71a1.0017.02/DOC_1&format=PDF.

European Commission (2021a): Batteries - modernising EU rules. Feedback and statistics: Proposal for a regulation. Available online at https://ec.europa.eu/info/law/better-regulation/have-your-say/initiatives/12399-Batteries-modernising-EU-rules/feedback_en?p_id=16675210, updated on 2/8/2023, checked on 2/15/2023.

European Commission (2021b): Better Regulation Toolbox. November 2021 edition. Available online at https://ec.europa.eu/info/sites/default/files/br_toolbox-nov_2021_en_0.pdf, checked on 1/20/2022.

European Commission (2021c): Product groups documents | Product Bureau. MEErP revision. Available online at https://susproc.jrc.ec.europa.eu/product-bureau/product-groups/521/documents, updated on 1/20/2022, checked on 1/20/2022.

European Commission (2021d): Ecodesign preparatory Study for Batteries. Available online at https://ecodesignbatteries.eu/planning.html, updated on 10/26/2021, checked on 12/14/2022.

European Commission (2022a): Communication from the Commission Ecodesign and Energy Labelling Working Plan 2022-2024 2022/C 182/01. 2022/C 182/01.

European Commission (2022b): EU agrees new law on more sustainable and circular batteries. 09.12.2022. Available online at https://ec.europa.eu/commission/presscorner/detail/en/ip_22_7588.

European Commission (2022c): Proposal for a regulation of the European Parliament and of the Council establishing a framework for setting ecodesign requirements for sustainable products and repealing Directive 2009/125/EC. 2022/0095 (COD). Available online at https://ec.europa.eu/environment/publications/proposal-ecodesign-sustainable-products-regulation_en, checked on 3/31/2022.

European Commission (2023a): Regulation (EU) 2023/1542 of the European Parliament and of the Council of 12 July 2023 concerning batteries and waste batteries, amending Directive 2008/98/EC and Regulation (EU) 2019/1020 and repealing Directive 2006/66/EC. (EU) 2023/1542. Available online at https://eur-lex.europa.eu/eli/reg/2023/1542/oj.

European Commission (2023b): Regulation (EU) 2023/851 of the European Parliament and of the Council of 19 April 2023 amending Regulation (EU) 2019/631 as regards strengthening the CO2 emission performance standards for new passenger cars and new light commercial vehicles in line with the Union's increased climate ambition. (EU) 2023/851. Available online at https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32023R0851, checked on 1/24/2024.

European Environmental Bureau (Ed.) (2021): Open Letter - Battery Regulation (NGOs). 40 human rights and environmental NGOs warn against unnecessary and harmful delays to new sustainable battery rules. Available online at https://www.transportenvironment.org/wp-content/uploads/2021/12/2021_12_open_letter_battery_regulation_NGOs_ministers.pdf, checked on 1/23/2023.

Eurostat (2021): Generation of waste by waste category, hazardousness and NACE Rev. 2 activity [env_wasgen]. Available online at http://appsso.eurostat.ec.europa.eu/nui/show.do?lang=en&dataset=env_wasgen, updated on 10/15/2021, checked on 10/15/2021.

Eurostat (2023): Energy statistics - an overview. Final energy consumption by sector, EU, 2021. Available online at https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Energy_statistics_-_an_overview#Final_energy_consumption, updated on 9/27/2023, checked on 1/23/2024.

Eynard, Umberto; Georgitzikis, Konstantinos; Wittmer, Dominic; EL Latunussa, Cynthia; Torres de Matos, Cristina; Mancini, Lucia et al. (2020a): Study on the EU's list of Critical Raw Materials. Critical Raw Materials Factsheets. Edited by European Commission.

Eynard, Umberto; Georgitzikis, Konstantinos; Wittmer, Dominic; EL Latunussa, Cynthia; Torres de Matos, Cristina; Mancini, Lucia et al. (2020b): Study on the EU's list of Critical Raw Materials. Non-Critical Raw Materials Factsheets. Edited by European Commission.

Fan, Ersha; Li, Li; Wang, Zhenpo; Lin, Jiao; Huang, Yongxin; Yao, Ying et al. (2020): Sustainable Recycling Technology for Li-Ion Batteries and Beyond: Challenges and Future Prospects. In *Chemical reviews* 120 (14), pp. 7020–7063. DOI: 10.1021/acs.chemrev.9b00535.

Farjana, Shahjadi Hisan; Mahmud, M. A. Parvez; Huda, Nazmul (2021): Life cycle assessment for sustainable mining. Amsterdam, Cambridge, MA: Elsevier. Available online at https://www.sciencedirect.com/science/book/9780323854511.

Fick, Katharina (2022): Policy Change Around Batteries : Understanding The Change Towards Resource Efficiency In The EU Batteries Policy With The Advocacy Coalition Framework. Available online at http://essay.utwente.nl/93640/.

Finkbeiner, Matthias (2014): Product environmental footprint—breakthrough or breakdown for policy implementation of life cycle assessment? In *Int J Life Cycle Assess* 19 (2), pp. 266–271. DOI: 10.1007/s11367-013-0678-x.

Finkbeiner, Matthias (2021): Commentary: System Expansion and Substitution in LCA: A Lost Opportunity of ISO 14044 Amendment 2. In *Front. Sustain.* 2, Article 729267. DOI: 10.3389/frsus.2021.729267.

Gama Caldas, Miguel; Spiliotopoulos, Christoforos; Alfieri, Felice; Eynard, Umberto; Zampori, Luca; Blengini, Gian Andrea et al. (2021): Draft Review of the MEErP - Methodology for Ecodesign of Energy-related Products. EC-JRC. Available online at https://susproc.jrc.ec.europa.eu/product-bureau//sites/default/files/2021-09/MEErP_revision_draft_report_Task_1-2_24-06-2021.pdf, checked on 9/29/2022.

Gao, Chengkang; Gao, Chengbo; Song, Kaihui; Fang, Kejing (2020): Pathways towards regional circular economy evaluated using material flow analysis and system dynamics. In *Resources, Conservation and Recycling* 154, p. 104527. DOI: 10.1016/j.resconrec.2019.104527.

Gennaro, Michele de; Paffumi, Elena; Martini, Giorgio; Giallonardo, Andrew; Pedroso, Samuel; Loiselle-Lapointe, Aaron (2020): A case study to predict the capacity fade of the battery of electrified vehicles in real-world use conditions. In *Case Studies on Transport Policy* 8 (2), pp. 517–534. DOI: 10.1016/j.cstp.2019.11.005.

Georgescu-Roegen, Nicholas (1971): The Entropy Law and the Economic Process. s.l.: Harvard University Press.

Giordono, Leanne S. (2020): Advocacy Coalitions in Low Salience Policy Subsystems: Struggles Under a Smooth Surface. In *Policy Studies Journal* 48 (4), pp. 1135–1167. DOI: 10.1111/psj.12330.

Giosuè, Chiara; Marchese, Daniele; Cavalletti, Matteo; Isidori, Robertino; Conti, Massimo; Orcioni, Simone et al. (2021): An Exploratory Study of the Policies and Legislative Perspectives on the End-of-Life of Lithium-Ion Batteries from the Perspective of Producer Obligation. In *Sustainability* 13 (20), p. 11154. DOI: 10.3390/su132011154.

Glöser-Chahoud, Simon; Pfaff, Matthias; Schultmann, Frank (2021): The link between product service lifetime and GHG emissions: A comparative study for different consumer products. In *Journal of Industrial Ecology* 25 (2), pp. 465–478. DOI: 10.1111/jiec.13123.

Google trends (2023): Google Trends NGram Viewer: Popularity of searches for "circular economy", "Kreislaufwirtschaft" and "Économie circulaire". Available online at https://trends.google.com/trends/explore?date=all&q=circular%20economy,Kreislaufwirtschaft,%C3%89conomie%20circulaire, updated on 12/10/2023, checked on 12/10/2023.

Göswein, Verena; Silvestre, José Dinis; Sousa Monteiro, Cláudia; Habert, Guillaume; Freire, Fausto; Pittau, Francesco (2021): Influence of material choice, renovation rate, and electricity grid to achieve a Paris Agreement-compatible building stock: A Portuguese case study. In *Building and Environment* 195, p. 107773. DOI: 10.1016/j.buildenv.2021.107773.

Graedel, Thomas E. (2019): Material Flow Analysis from Origin to Evolution. In *Environmental science & technology* 53 (21), pp. 12188–12196. DOI: 10.1021/acs.est.9b03413.

Gupta, Vimal Kumar; Kaushal, Rajendra Kumar; Shukla, Sheo Prasad (2018): Multi-stakeholder policy modeling for collection and recycling of spent portable battery waste. In *Waste management & research : the journal of the International Solid Wastes and Public Cleansing Association, ISWA* 36 (7), pp. 577–593. DOI: 10.1177/0734242X18773544.

Haes, Helias Udo; Heijungs, Reinout; Huppes, Gjalt; Voet, Ester; Hettelingh, Jean-Paul (2000): Full Mode and Attribution Mode in Environmental Analysis. In *Journal of Industrial Ecology* 4 (1), pp. 45–56. DOI: 10.1162/108819800569285.

Häge, Frank M. (2013): Coalition Building and Consensus in the Council of the European Union. In *British Journal of Political Science* 43 (3), pp. 481–504. DOI: 10.1017/S0007123412000439.

Harris, Steve; Martin, Michael; Diener, Derek (2021): Circularity for circularity's sake? Scoping review of assessment methods for environmental performance in the circular economy. In *Sustainable Production and Consumption* 26, pp. 172–186. DOI: 10.1016/j.spc.2020.09.018.

Harrison, Robert; Jehle, Christoph (2007): European Commission DG TREN - Preparatory studies for Eco-design Requirements of EuPs. Simple Digital TV Converters (Simple Set Top Boxes) - Final Report.

Hartley, Kris; van Santen, Ralf; Kirchherr, Julian (2020): Policies for transitioning towards a circular economy: Expectations from the European Union (EU). In *Resources, Conservation and Recycling* 155, p. 104634. DOI: 10.1016/j.resconrec.2019.104634.

Heeren, Niko; Hellweg, Stefanie (2019): Tracking Construction Material over Space and Time: Prospective and Geo-referenced Modeling of Building Stocks and Construction Material Flows. In *Journal of Industrial Ecology* 23 (1), pp. 253–267. DOI: 10.1111/jiec.12739.

Heijungs, Reinout; Allacker, Karen; Benetto, Enrico; Brandão, Miguel; Guinée, Jeroen; Schaubroeck, Simon et al. (2021): System Expansion and Substitution in LCA: A Lost Opportunity of ISO 14044 Amendment 2. In *Front. Sustain.* 2, Article 692055. DOI: 10.3389/frsus.2021.692055.

Hettesheimer, Tim; Neef, Christoph; Rosellón Inclán, Inés; Link, Steffen; Schmaltz, Thomas; Schuckert, Felix et al. (2023): Lithium-Ion Battery Roadmap - Industrialization Perspectives toward 2030. With assistance of Fraunhofer-Gesellschaft.

Hettesheimer, Tim; Thielmann, Axel; Neef, Christoph; Durand, Antoine (2019): Preparatory study on Ecodesign and Energy Labelling of batteries. Task 4 - Technologies - For Ecodesign and Energy Labelling. Edited by European Commission, checked on 5/20/2020.

Hischier, R.; Wäger, P.; Gauglhofer, J. (2005): Does WEEE recycling make sense from an environmental perspective? In *Environmental Impact Assessment Review* 25 (5), pp. 525–539. DOI: 10.1016/j.eiar.2005.04.003.

Huang, Chu-Long; Xu, Ming; Cui, Shenghui; Li, Zirong; Fang, Hongda; Wang, Peng (2020): Copper-induced ripple effects by the expanding electric vehicle fleet: A crisis or an opportunity. In *Resources, Conservation and Recycling* 161, p. 104861. DOI: 10.1016/j.resconrec.2020.104861.

Islam, Md Tasbirul; Huda, Nazmul (2019): Material flow analysis (MFA) as a strategic tool in E-waste management: Applications, trends and future directions. In *Journal of environmental management* 244, pp. 344–361. DOI: 10.1016/j.jenvman.2019.05.062.

Ismail, Haikal; Hanafiah, Marlia M. (2021): Evaluation of e-waste management systems in Malaysia using life cycle assessment and material flow analysis. In *Journal of Cleaner Production* 308, p. 127358. DOI: 10.1016/j.jclepro.2021.127358.

ISO 14044:2006, 2006: ISO 14044:2006. Available online at https://www.iso.org/standard/38498.html.

ISO 9000:2000, 2000: ISO 9000:2000 Quality Management Systems-Fundamentals and Vocabulary.

Istrate, Ioan-Robert; Galvez-Martos, Jose-Luis; Dufour, Javier (2021): The impact of incineration phase-out on municipal solid waste landfilling and life cycle environmental performance: Case study of Madrid, Spain. In *The Science of the total environment* 755 (Pt 1), p. 142537. DOI: 10.1016/j.scitotenv.2020.142537.

Jacobi, Nikolai; Haas, Willi; Wiedenhofer, Dominik; Mayer, Andreas (2018): Providing an economy-wide monitoring framework for the circular economy in Austria: Status quo and challenges. In *Resources, Conservation and Recycling* 137, pp. 156–166. DOI: 10.1016/j.resconrec.2018.05.022.

Jana, Aniruddha; Shaver, Gregory M.; García, R. Edwin (2019): Physical, on the fly, capacity degradation prediction of LiNiMnCoO2-graphite cells. In *Journal of Power Sources* 422, pp. 185–195. DOI: 10.1016/j.jpowsour.2019.02.073.

Jenkins-Smith, Hank C.; Nohrstedt, Daniel; Weible, Christopher; Sabatier, Paul A. (2014): Theories of the policy process (3rd ed.). The advocacy coalition framework: Foundations, evolution, and ongoing research (pp. 183-223): Westview Press.

Kayo, Chihiro; Dente, Sébastien M.R.; Aoki-Suzuki, Chika; Tanaka, Daisuke; Murakami, Shinsuke; Hashimoto, Seiji (2019): Environmental Impact Assessment of Wood Use in Japan through 2050 Using Material Flow Analysis and Life Cycle Assessment. In *Journal of Industrial Ecology* 23 (3), pp. 635–648. DOI: 10.1111/jiec.12766.

Kemna, René; Azaïs, Nelly; van Elburg, Martijn; van der Voort, Maaike; Li, William (2011): MEErP 2011. Methodology for Ecodesign of Energy-related Products. Edited by European Commission. Available online at https://op.europa.eu/en/publication-detail/-/publication/be880e05-7528-415d-b592-e9f29e787635, checked on 1/20/2022.

Kemna, René; van Elburg, Martijn; Li, William; van Holsteijn, Rob (2005): MEEUP. Methodology Report. Edited by European Commission. Available online at https://www.vhk.nl/research/reports.htm.

Kerdlap, Piya; Low, Jonathan Sze Choong; Tan, Daren Zong Loong; Yeo, Zhiquan; Ramakrishna, Seeram (2022): UM3-LCE3-ISN: a methodology for multi-level life cycle environmental and economic evaluation of industrial symbiosis networks. In *Int J Life Cycle Assess*. DOI: 10.1007/s11367-022-02024-1.

Kirchherr, Julian; Piscicelli, Laura; Bour, Ruben; Kostense-Smit, Erica; Muller, Jennifer; Huibrechtse-Truijens, Anne; Hekkert, Marko (2018): Barriers to the Circular Economy: Evidence From the European Union (EU). In *Ecological Economics* 150, pp. 264–272. DOI: 10.1016/j.ecolecon.2018.04.028.

Kirchherr, Julian; Reike, Denise; Hekkert, Marko (2017): Conceptualizing the circular economy: An analysis of 114 definitions. In *Resources, conservation, and recycling* 127, pp. 221–232. DOI: 10.1016/j.resconrec.2017.09.005.

Kishita, Yusuke; Matsumoto, Mitsutaka; Inoue, Masato; Fukushige, Shinichi (Eds.) (2021): EcoDesign and Sustainability I. Products, Services, and Business Models. 1st ed. 2021. Singapore: Springer Singapore; Imprint Springer (Springer eBook Collection). Available online at http://swbplus.bsz-bw.de/bsz1741573521cov.htm.

Klöpffer, Walter (2014): Background and Future Prospects in Life Cycle Assessment. Dordrecht: Springer Netherlands.

Köhler, Susanne; Pizzol, Massimo (2019): Life Cycle Assessment of Bitcoin Mining. In *Environmental science & technology* 53 (23), pp. 13598–13606. DOI: 10.1021/acs.est.9b05687.

Lacovidou, Eleni; Millward-Hopkins, Joel; Busch, Jonathan; Purnell, Philip; Velis, Costas A.; Hahladakis, John N. et al. (2017): A pathway to circular economy: Developing a conceptual framework for complex value assessment of resources recovered from waste. In *Journal of Cleaner Production* 168, pp. 1279–1288. DOI: 10.1016/j.jclepro.2017.09.002.

Lam, Wai Chung; Peeters, Karolien; van Tichelen, Paul (2019): Preparatory Study on Ecodesign and Energy Labelling of Batteries under FWC ENER/C3/2015-619-Lot 1. Task 5 Environment & economics. Available online at https://ecodesignbatteries.eu/sites/ecodesignbatteries.eu/files/attachments/ED%20Battery%20study%20Task5_v3_20190823.pdf, checked on 1/26/2024.

Laner, David; Zoboli, Ottavia; Rechberger, Helmut (2017): Statistical entropy analysis to evaluate resource efficiency: Phosphorus use in Austria. In *Ecological Indicators* 83, pp. 232–242. DOI: 10.1016/j.ecolind.2017.07.060.

Lausselet, Carine; Urrego, Johana Paola Forero; Resch, Eirik; Brattebø, Helge (2021): Temporal analysis of the material flows and embodied greenhouse gas emissions of a neighborhood building stock. In *Journal of Industrial Ecology* 25 (2), pp. 419–434. DOI: 10.1111/jiec.13049.

Li, Matthew; Lu, Jun; Chen, Zhongwei; Amine, Khalil (2018): 30 Years of Lithium-Ion Batteries. In *Advanced materials (Deerfield Beach, Fla.)*, e1800561. DOI: 10.1002/adma.201800561.

Liu, Manzhi; Chen, Xuhui; Zhang, Mengya; Lv, Xueqing; Wang, Haihao; Chen, Zhizhi et al. (2020a): End-of-life passenger vehicles recycling decision system in China based on dynamic material flow analysis and life cycle assessment. In *Waste management (New York, N.Y.)* 117, pp. 81–92. DOI: 10.1016/j.wasman.2020.08.002.

Liu, Manzhi; Tan, Shuai; Zhang, Mengya; He, Gang; Chen, Zhizhi; Fu, Zhiwei; Luan, Changjin (2020b): Waste paper recycling decision system based on material flow analysis and life cycle assessment: A case study of waste paper recycling from China. In *Journal of environmental management* 255, p. 109859. DOI: 10.1016/j.jenvman.2019.109859.

Lueddeckens, Stefan; Saling, Peter; Guenther, Edeltraud (2020): Temporal issues in life cycle assessment—a systematic review. In *Int J Life Cycle Assess* 25 (8), pp. 1385–1401. DOI: 10.1007/s11367-020-01757-1.

Malabi Eberhardt, Leonora Charlotte; van Stijn, Anne; Kristensen Stranddorf, Liv; Birkved, Morten; Birgisdottir, Harpa (2021): Environmental Design Guidelines for Circular Building Components: The Case of the Circular Building Structure. In *Sustainability* 13 (10), p. 5621. DOI: 10.3390/su13105621.

Malinauskaite, J.; Anguilano, L.; Rivera, X. Schmidt (2021): Circular waste management of electric vehicle batteries: Legal and technical perspectives from the EU and the UK post Brexit. In *International Journal of Thermofluids* 10, p. 100078. DOI: 10.1016/j.ijft.2021.100078.

ÖNorm S 2096, 1/1/2005: Material flow analysis.

Mathieux, Fabrice; Ardente, Fulvio; Bobba, Silvia (2020): Ten years of scientific support for integrating circular economy requirements in the EU ecodesign directive: Overview and lessons learnt. In *Procedia CIRP* 90, pp. 137–142. DOI: 10.1016/j.procir.2020.02.121.

Meadows, Donella H.; Meadows, Dennis L.; Randers, Jorgen; Behrens, William W. (1972): The Limits to Growth. A report for the Club of Rome's project on the predicament of manking. New York: Universe Books.

Mehr, Jonas; Vadenbo, Carl; Steubing, Bernhard; Hellweg, Stefanie (2018): Environmentally optimal wood use in Switzerland—Investigating the relevance of material cascades. In *Resources, Conservation and Recycling* 131, pp. 181–191. DOI: 10.1016/j.resconrec.2017.12.026.

Melin, Hans Eric; Rajaeifar, Mohammad Ali; Ku, Anthony Y.; Kendall, Alissa; Harper, Gavin; Heidrich, Oliver (2021): Global implications of the EU battery regulation. In *Science (New York, N.Y.)* 373 (6553), pp. 384–387. DOI: 10.1126/science.abh1416.

Merli, Roberto; Preziosi, Michele; Acampora, Alessia (2018): How do scholars approach the circular economy? A systematic literature review. In *Journal of Cleaner Production* 178, pp. 703–722. DOI: 10.1016/j.jclepro.2017.12.112.

Milford, Rachel L.; Pauliuk, Stefan; Allwood, Julian M.; Müller, Daniel B. (2013): The roles of energy and material efficiency in meeting steel industry CO2 targets. In *Environmental science & technology* 47 (7), pp. 3455–3462. DOI: 10.1021/es3031424.

Milios, Leonidas (2018): Advancing to a Circular Economy: three essential ingredients for a comprehensive policy mix. In *Sustainability science* 13 (3), pp. 861–878. DOI: 10.1007/s11625-017-0502-9.

Millward-Hopkins, Joel; Busch, Jonathan; Purnell, Phil; Zwirner, Oliver; Velis, Costas A.; Brown, Andrew et al. (2018): Fully integrated modelling for sustainability assessment of resource recovery from waste. In *The Science of the total environment* 612, pp. 613–624. DOI: 10.1016/j.scitotenv.2017.08.211.

Ministère de la Transition écologique (2021): Indice de réparabilité. Available online at https://www.ecologie.gouv.fr/indice-reparabilite, updated on 1/12/2022, checked on 1/12/2022.

Morimoto, Shinichirou; Cheng, Yanfei; Mizukoshi, Norio; Tahara, Kiyotaka (2020): Methodological Study of Evaluating Future Lightweight Vehicle Scenarios and CO2 Reduction Based on Life Cycle Assessment. In *Sustainability* 12 (14), p. 5713. DOI: 10.3390/su12145713.

Mudgal, Shailendra; Tinetti, Benoît; Prado Trigo, Alvaro de; Schischke, Karsten; Proske, Marina; Geibler, Justus von; Teubler, Jens (2013): Material-efficiency Ecodesign Report and Module to the Methodology for the Ecodesign of Energy-related Products (MEErP). Part 1: Material Efficiency for Ecodesign. Final report to the European Commission - DG Enterprise and Industry. Prepared for: European Commission - DG Enterprise and Industry. With assistance of Fraunhofer IZM. Edited by BIO Intelligence Service. Available online at https://single-market-economy.ec.europa.eu/industry/sustainability/sustainable-product-policy-ecodesign_en.

Neubauer, Jeremy S.; Wood, Eric; Pesaran, Ahmad (2015): A Second Life for Electric Vehicle Batteries: Answering Questions on Battery Degradation and Value. In *SAE Int. J. Mater. Manf.* 8 (2), pp. 544–553. DOI: 10.4271/2015-01-1306.

Nguyen-Tien, Viet; Dai, Qiang; Harper, Gavin D.J.; Anderson, Paul A.; Elliott, Robert J.R. (2022): Optimising the geospatial configuration of a future lithium ion battery recycling industry in the transition to electric vehicles and a circular economy. In *Applied Energy* 321, p. 119230. DOI: 10.1016/j.apenergy.2022.119230.

Nohrstedt, Daniel (2005): External shocks and policy change: Three Mile Island and Swedish nuclear energy policy. In *Journal of European Public Policy* 12 (6), pp. 1041–1059. DOI: 10.1080/13501760500270729.

Northvolt; Talga Group; Verkor; Vulcan Energy; Eramet; Sekeleton Technologies; ACC, InnoEnergy (Eds.) (2021): Open Letter - Battery Regulation (industry). European companies warn against unnecessary and counterproductive delays to new EU battery regulation. Available online at https://www.transportenvironment.org/wp-content/uploads/2021/12/2021_-batteries_reg_open_letter_companies.pdf, checked on 1/23/2023.

Olalo, Kristin Faye; Nakatani, Jun; Fujita, Tsuyoshi (2022): Optimal Process Network for Integrated Solid Waste Management in Davao City, Philippines. In *Sustainability* 14 (4), p. 2419. DOI: 10.3390/su14042419.

Olivetti, Elsa A.; Ceder, Gerbrand; Gaustad, Gabrielle G.; Fu, Xinkai (2017): Lithium-Ion Battery Supply Chain Considerations: Analysis of Potential Bottlenecks in Critical Metals. In *Joule* 1 (2), pp. 229–243. DOI: 10.1016/j.joule.2017.08.019.

Padilla-Rivera, Alejandro; Russo-Garrido, Sara; Merveille, Nicolas (2020): Addressing the Social Aspects of a Circular Economy: A Systematic Literature Review. In *Sustainability* 12 (19), p. 7912. DOI: 10.3390/su12197912.

Parchomenko, Alexej; Nelen, Dirk; Gillabel, Jeroen; Vrancken, Karl C.; Rechberger, Helmut (2020): Evaluation of the resource effectiveness of circular economy strategies through multilevel Statistical Entropy Analysis. In *Resources, Conservation and Recycling* 161, p. 104925. DOI: 10.1016/j.resconrec.2020.104925.

Pauliuk, Stefan; Heeren, Niko (2020): ODYM—An open software framework for studying dynamic material systems: Principles, implementation, and data structures. In *Journal of Industrial Ecology* 24 (3), pp. 446–458. DOI: 10.1111/jiec.12952.

Pauliuk, Stefan; Heeren, Niko (2021): Material efficiency and its contribution to climate change mitigation in Germany: A deep decarbonization scenario analysis until 2060. In *Journal of Industrial Ecology* 25 (2), pp. 479–493. DOI: 10.1111/jiec.13091.

Pauliuk, Stefan; Heeren, Niko; Berrill, Peter; Fishman, Tomer; Nistad, Andrea; Tu, Qingshi et al. (2021): Global scenarios of resource and emission savings from material efficiency in residential buildings and cars. In *Nature communications* 12 (1), p. 5097. DOI: 10.1038/s41467-021-25300-4.

Pauliuk, Stefan; Hertwich, Edgar G. (2016): Prospective Models of Society's Future Metabolism: What Industrial Ecology Has to Contribute. In Roland Clift, Angela Druckman (Eds.): Taking Stock of Industrial Ecology. Cham: Springer International Publishing, pp. 21–43.

Pauliuk, Stefan; Sjöstrand, Karin; Müller, Daniel B. (2013): Transforming the Norwegian Dwelling Stock to Reach the 2 Degrees Celsius Climate Target. In *Journal of Industrial Ecology* 17 (4), pp. 542–554. DOI: 10.1111/j.1530-9290.2012.00571.x.

Pearce, David W.; Turner, Robert Kerry (1990): Economics of natural resources and the environment. Baltimore: The Johns Hopkins Univ. Press.

Pedersen, Emil; Remmen, Arne (2022): Challenges with product environmental footprint: a systematic review. In *Int J Life Cycle Assess* 27 (2), pp. 342–352. DOI: 10.1007/s11367-022-02022-3.

Pehlken, Alexandra; Albach, Sabine; Vogt, Thomas (2017): Is there a resource constraint related to lithium ion batteries in cars? In *Int J Life Cycle Assess* 22 (1), pp. 40–53. DOI: 10.1007/s11367-015-0925-4.

Perminova, Tatiana; Sirina, Natalia; Laratte, Bertrand; Baranovskaya, Natalia; Rikhvanov, Leonid (2016): Methods for land use impact assessment: A review. In *Environmental Impact Assessment Review* 60, pp. 64–74. DOI: 10.1016/j.eiar.2016.02.002.

Pierce, Jonathan J.; Peterson, Holly L.; Hicks, Katherine C. (2020): Policy Change: An Advocacy Coalition Framework Perspective. In *Policy Studies Journal* 48 (1), pp. 64–86. DOI: 10.1111/psj.12223.

Pierce, Jonathan J.; Peterson, Holly L.; Jones, Michael D.; Garrard, Samantha P.; Vu, Theresa (2017): There and Back Again: A Tale of the Advocacy Coalition Framework. In *Policy Studies Journal* 45 (S1), S13-S46. DOI: 10.1111/psj.12197.

Plevin, Richard J.; Delucchi, Mark A.; Creutzig, Felix (2014): Using Attributional Life Cycle Assessment to Estimate Climate-Change Mitigation Benefits Misleads Policy Makers. In *Journal of Industrial Ecology* 18 (1), pp. 73–83. DOI: 10.1111/jiec.12074.

Polverini, Davide (2021): Regulating the circular economy within the ecodesign directive: Progress so far, methodological challenges and outlook. In *Sustainable Production and Consumption* 27, pp. 1113–1123. DOI: 10.1016/j.spc.2021.02.023.

Polverini, Davide; Miretti, Ugo (2019): An approach for the techno-economic assessment of circular economy requirements under the Ecodesign Directive. In *Resources, conservation, and recycling* 150, p. 104425. DOI: 10.1016/j.resconrec.2019.104425.

Potting, José; Hekkert, Marko; Worrell, Ernst; Hanemaaijer, Aldert (2017): Circular Economy: Measuring innovation in the product chain. Policy Report. Available online at https://www.researchgate.net/publication/319314335_Circular_Economy_Measuring_innovation_in_the_product_chain.

Rachidi, Ntebatše R.; Nwaila, Glen T.; Zhang, Steven E.; Bourdeau, Julie E.; Ghorbani, Yousef (2021): Assessing cobalt supply sustainability through production forecasting and implications for green energy policies. In *Resources Policy* 74, p. 102423. DOI: 10.1016/j.resourpol.2021.102423.

Reale, F.; Castellani, V.; Hischier, R.; Corrado, S.; Sala, S. (2019): Consumer Footprint. Basket of Products indicators on household appliances. Edited by Publications Office of the European Union. European Commission. Available online at https://publications.jrc.ec.europa.eu/repository/handle/JRC116704.

Resch, Eirik; Andresen, Inger; Cherubini, Francesco; Brattebø, Helge (2021): Estimating dynamic climate change effects of material use in buildings—Timing, uncertainty, and emission sources. In *Building and Environment* 187, p. 107399. DOI: 10.1016/j.buildenv.2020.107399.

Richa, Kirti; Babbitt, Callie W.; Gaustad, Gabrielle; Wang, Xue (2014): A future perspective on lithium-ion battery waste flows from electric vehicles. In *Resources, Conservation and Recycling* 83, pp. 63–76. DOI: 10.1016/j.resconrec.2013.11.008.

Rockström, Johan; Steffen, Will; Noone, Kevin; Persson, Åsa; Chapin, F. Stuart, III; Lambin, Eric et al. (2009): Planetary Boundaries: Exploring the Safe Operating Space for Humanity. In *E&S* 14 (2). DOI: 10.5751/ES-03180-140232.

Rostek, Leon; Tercero Espinoza, Luis A.; Goldmann, Daniel; Loibl, Antonia (2022): A dynamic material flow analysis of the global anthropogenic zinc cycle: Providing a quantitative basis for circularity discussions. In *Resources, Conservation and Recycling* 180, p. 106154. DOI: 10.1016/j.resconrec.2022.106154.

Ryter, John; Fu, Xinkai; Bhuwalka, Karan; Roth, Richard; Olivetti, Elsa (2022): Assessing recycling, displacement, and environmental impacts using an economics-informed material system model. In *Journal of Industrial Ecology* 26 (3), pp. 1010–1024. DOI: 10.1111/jiec.13239.

Ryter, John; Fu, Xinkai; Bhuwalka, Karan; Roth, Richard; Olivetti, Elsa A. (2021): Emission impacts of China's solid waste import ban and COVID-19 in the copper supply chain. In *Nature communications* 12 (1), p. 3753. DOI: 10.1038/s41467-021-23874-7.

Sabatier, Paul A. (1988): An advocacy coalition framework of policy change and the role of policy-oriented learning therein. In *Policy Sciences* 21 (2-3), pp. 129–168. DOI: 10.1007/BF00136406.

Sabatier, Paul A.; Weible, Christopher M. (Eds.) (2015): Theories of the policy process. Third edition. Boulder, Colorado and Boston, Massachusetts: Westview Press, a member of the Persus Books Group and Credo Reference.

Sacchi, R.; Terlouw, T.; Siala, K.; Dirnaichner, A.; Bauer, C.; Cox, B. et al. (2022): PRospective EnvironMental Impact asSEment (premise): A streamlined approach to producing databases for prospective life cycle assessment using integrated assessment models. In *Renewable and Sustainable Energy Reviews* 160, p. 112311. DOI: 10.1016/j.rser.2022.112311.

Sadeleer, Irmeline de; Brattebø, Helge; Callewaert, Pieter (2020): Waste prevention, energy recovery or recycling - Directions for household food waste management in light of circular economy policy. In *Resources, Conservation and Recycling* 160, p. 104908. DOI: 10.1016/j.resconrec.2020.104908.

Sadri, Habib; Pourbagheri, Peyman; Yitmen, Ibrahim (2022): Towards the implications of Boverket's climate declaration act for sustainability indices in the Swedish construction industry. In *Building and Environment* 207, p. 108446. DOI: 10.1016/j.buildenv.2021.108446.

Saidani, Michael; Yannou, Bernard; Leroy, Yann; Cluzel, Francois; Kim, Harrison (2021): Multi-tool methodology to evaluate action levers to close the loop on critical materials – Application to precious metals used in catalytic converters. In *Sustainable Production and Consumption* 26, pp. 999–1010. DOI: 10.1016/j.spc.2021.01.010.

Sakai, Shin-ichi; Yano, Junya; Hirai, Yasuhiro; Asari, Misuzu; Yanagawa, Ritsuki; Matsuda, Takeshi et al. (2017): Waste prevention for sustainable resource and waste management. In *J Mater Cycles Waste Manag* 19 (4), pp. 1295–1313. DOI: 10.1007/s10163-017-0586-4.

Sala, Serenella; Benini, Lorenzo; Beylot, Antoine; Castellani, Valentina; Cerutti, Alessandro; Corrado, Sara et al. (2019): Consumption and consumer footprint. Methodology and results : indicators and assessment of the environmental impact of European consumption. European Commission. Luxembourg (JRC technical reports, JRC113607).

Saltelli, Andrea; Annoni, Paola (2010): How to avoid a perfunctory sensitivity analysis. In *Environmental Modelling & Software* 25 (12), pp. 1508–1517. DOI: 10.1016/j.envsoft.2010.04.012.

Sandberg, Nina Holck; Brattebø, Helge (2012): Analysis of energy and carbon flows in the future Norwegian dwelling stock. In *Building Research & Information* 40 (2), pp. 123–139. DOI: 10.1080/09613218.2012.655071.

Schaubroeck, Thomas; Schrijvers, Dieuwertje; Schaubroeck, Simon; Moretti, Christian; Zamagni, Alessandra; Pelletier, Nathan et al. (2022): Definition of Product System and Solving Multifunctionality in ISO 14040–14044: Inconsistencies and Proposed Amendments—Toward a More Open and General LCA Framework. In *Front. Sustain.* 3, Article 778100. DOI: 10.3389/frsus.2022.778100.

Schischke, Karsten; Clemm, Christian; Berwald, Anton; Proske, Marina; Dimitrova, Gergana; Reinhold, Julia et al. (2021): Ecodesign preparatory study on mobile phones, smartphones and tablets: final report: Publications Office. Available online at https://data.europa.eu/doi/10.2873/175802.

Schlegel, Moritz-C.; Akkerman, Floris (Eds.) (2019): One step back, two steps forward - resource efficiency requirements within ecodesign. ECEE Summer Study Proceedings.

Schmidt, Tobias; Buchert, Matthias; Schebek, Liselotte (2016): Investigation of the primary production routes of nickel and cobalt products used for Li-ion batteries. In *Resources, Conservation and Recycling* 112, pp. 107–122. DOI: 10.1016/j.resconrec.2016.04.017.

Schrijvers, Dieuwertje L.; Loubet, Philippe; Sonnemann, Guido (2016): Critical review of guidelines against a systematic framework with regard to consistency on allocation procedures for recycling in LCA. In *Int J Life Cycle Assess* 21 (7), pp. 994–1008. DOI: 10.1007/s11367-016-1069-x.

Sevigné-Itoiz, Eva; Gasol, Carles M.; Rieradevall, Joan; Gabarrell, Xavier (2015): Contribution of plastic waste recovery to greenhouse gas (GHG) savings in Spain. In *Waste management (New York, N.Y.)* 46, pp. 557–567. DOI: 10.1016/j.wasman.2015.08.007.

Shinde, Rhythima; Froemelt, Andreas; Kim, Aleksandra; Hellweg, Stefanie (2022): A novel machine-learning approach for evaluating rebounds-associated environmental footprint of households and application to cooperative housing. In *Journal of environmental management* 304, p. 114205. DOI: 10.1016/j.jenvman.2021.114205.

Sommer, P.; Rotter, V. S.; Ueberschaar, M. (2015): Battery related cobalt and REE flows in WEEE treatment. In *Waste management (New York, N.Y.)* 45, pp. 298–305. DOI: 10.1016/j.wasman.2015.05.009.

Song, Jiali; Yan, Wenyi; Cao, Hongbin; Song, Qingbin; Ding, He; Lv, Zheng et al. (2019): Material flow analysis on critical raw materials of lithium-ion batteries in China. In *Journal of Cleaner Production* 215, pp. 570–581. DOI: 10.1016/j.jclepro.2019.01.081.

Spiliotopoulos, C.; Alfieri, F.; La Placa, M. G.; Bracquene, E.; Laps, E.; van Moeseke, T. et al. (2022): Product Reparability Scoring System: Specific Application to Smartphones and State Tablets. Available online at https://publications.jrc.ec.europa.eu/repository/bitstream/JRC128672/JRC128672_01.pdf, checked on 7/13/2022.

Stahl, H.; Mehlhart, G.; Gsell, M.; Sutter, J.; Dolega, P.; Baron, Y. et al. (2021): Assessment of options to improve particular aspects of the EU regulatory framework on batteries: final report. Edited by Publications Office of the European Union. European Commission. Directorate General for Environment.; Oeko Institut.; Ramboll.; Umweltbundesamt. Available online at https://data.europa.eu/doi/10.2779/432234.

Stamminger, Rainer; Bues, Alexander; Alfieri, Felice; Cordella, Mauro (2020): Durability of washing machines under real life conditions: Definition and application of a testing procedure. In *Journal of Cleaner Production* 261, p. 121222. DOI: 10.1016/j.jclepro.2020.121222.

Standing Committee of the National People's Congress (2008): Circular Economy Promotion Law of the People's Republic of China. CLI.1.107971(EN). Available online at http://www.lawinfochina.com/display.aspx?id=7025&lib=law, checked on 1/1/2022.

Steffen, Will; Richardson, Katherine; Rockström, Johan; Cornell, Sarah E.; Fetzer, Ingo; Bennett, Elena M. et al. (2015): Sustainability. Planetary boundaries: guiding human development on a changing planet. In *Science (New York, N.Y.)* 347 (6223), p. 1259855. DOI: 10.1126/science.1259855.

Stephan, André; Crawford, Robert H.; Bunster, Victor; Warren-Myers, Georgia; Moosavi, Sareh (2022): Towards a multiscale framework for modeling and improving the life cycle environmental performance of built stocks. In *Journal of Industrial Ecology* 26 (4), pp. 1195–1217. DOI: 10.1111/jiec.13254.

Steubing, Bernhard; Mutel, Christopher; Suter, Florian; Hellweg, Stefanie (2016): Streamlining scenario analysis and optimization of key choices in value chains using a modular LCA approach. In *Int J Life Cycle Assess* 21 (4), pp. 510–522. DOI: 10.1007/s11367-015-1015-3.

Suh, Sangwon; Yang, Yi (2014): On the uncanny capabilities of consequential LCA. In *Int J Life Cycle Assess* 19 (6), pp. 1179–1184. DOI: 10.1007/s11367-014-0739-9.

Takeshita et al. (2019): LIB-equipped Vehicle Market Bulletin (19Q4) and Long-term LIB Market Forecast. Edited by B3 Corporation.

Talens Peiró, Laura; Polverini, Davide; Ardente, Fulvio; Mathieux, Fabrice (2020): Advances towards circular economy policies in the EU: The new Ecodesign regulation of enterprise servers. In *Resources, conservation, and recycling* 154, p. 104426. DOI: 10.1016/j.resconrec.2019.104426.

Tecchio, Paolo; Ardente, Fulvio; Mathieux, Fabrice (2019): Understanding lifetimes and failure modes of defective washing machines and dishwashers. In *Journal of Cleaner Production* 215, pp. 1112–1122. DOI: 10.1016/j.jclepro.2019.01.044.

Technische Universität Wien (2012): STAN. Institute for Water Quality, Resource and Waste Management. Vienna. Available online at https://www.stan2web.net, checked on 8/17/2022.

Thakker, Vyom; Bakshi, Bhavik R. (2021): Toward sustainable circular economies: A computational framework for assessment and design. In *Journal of Cleaner Production* 295, p. 126353. DOI: 10.1016/j.jclepro.2021.126353.

Thiébaud, Esther; Hilty, Lorenz M.; Schluep, Mathias; Faulstich, Martin (2017): Use, Storage, and Disposal of Electronic Equipment in Switzerland. In *Environmental science & technology* 51 (8), pp. 4494–4502. DOI: 10.1021/acs.est.6b06336.

Thielmann, Axel; Neef, Christoph; Hettesheimer, Tim; Döscher, Henning; Wietschel, Martin; Tübke, Jens (2017): Energiespeicher-Roadmap (Update 2017). Hochenergie-Batterien 2030+ und Perspektiven zukünftiger Batterietechnologien. Edited by Fraunhofer Institute for Systems and Innovation Research ISI. Karlsruhe. Available online at https://www.isi.fraunhofer.de/content/dam/isi/dokumente/cct/lib/Energiespeicher-Roadmap-Dezember-2017.pdf.

Tillman, Anne-Marie; Ekvall, Tomas; Baumann, Henrikke; Rydberg, Tomas (1994): Choice of system boundaries in life cycle assessment. In *Journal of Cleaner Production* 2 (1), pp. 21–29. DOI: 10.1016/0959-6526(94)90021-3.

Tiruta-Barna, Ligia; Pigné, Yoann; Navarrete Gutiérrez, Tomás; Benetto, Enrico (2016): Framework and computational tool for the consideration of time dependency in Life Cycle Inventory: proof of concept. In *Journal of Cleaner Production* 116, pp. 198–206. DOI: 10.1016/J.JCLEPRO.2015.12.049.

Turner, James Morton; Nugent, Leah M. (2016): Charging up Battery Recycling Policies: Extended Producer Responsibility for Single-Use Batteries in the European Union, Canada, and the United States. In *Journal of Industrial Ecology* 20 (5), pp. 1148–1158. DOI: 10.1111/jiec.12351.

U.S.C. (2020): Save our Seas 2.0 Act. 116-224. Available online at https://www.congress.gov/116/plaws/publ224/PLAW-116publ224.pdf.

UN Environment (Ed.) (2017): Accelerating the Global Adoption of Energy-efficient Electric Motors and Motor Systems. Available online at https://www.unep.org/resources/report/accelerating-global-adoption-energy-efficient-electric-motors-and-motor-systems.

USABC (1996): Electric Vehicle Battery Test Procedures Manual. Revision 2. Edited by United States Advanced Battery Consortium (USABC). Available online at https://www.uscar.org/guest/publications.php, checked on 7/6/2020.

USGS (2022): Mineral commodity summaries 2022. Edited by U.S. Department of the Interior, U.S. Geological Survey. Available online at https://pubs.usgs.gov/periodicals/mcs2022/mcs2022.pdf.

Vaalma, Christoph; Buchholz, Daniel; Weil, Marcel; Passerini, Stefano (2018): A cost and resource analysis of sodium-ion batteries. In *Nat Rev Mater* 3 (4). DOI: 10.1038/natrevmats.2018.13.

van Notten, Philip (2005): Writing on the Wall. Scenario Development in Times of Discontinuity (38).

van Stijn, A.; Eberhardt, L.C.M.; Wouterszoon Jansen, B.; Meijer, A. (2022): Environmental design guidelines for circular building components based on LCA and MFA: Lessons from the circular kitchen and renovation façade. In *Journal of Cleaner Production* 357, p. 131375. DOI: 10.1016/j.jclepro.2022.131375.

van Tichelen, Paul; Mulder, Grietus; Durand, Antoine (2019): Preparatory Study on Ecodesign and Energy Labelling of rechargeable electrochemical batteries with internal storage under FWC ENER/C3/2015-619- Lot 1. Task 7 Report Policy Scenario Analysis. Edited by European Commission. Directorate General for Internal Market, Industry, Entrepreneurship and SMEs.

Velásquez-Rodríguez, Oscar Fabián; Løvik, Amund N.; Moreno-Mantilla, Carlos Eduardo (2021): Evaluation of the environmental impact of end-of-life refrigerators in Colombia by material flow analysis. In *Journal of Cleaner Production* 314, p. 127884. DOI: 10.1016/j.jclepro.2021.127884.

Venkatesh, G.; Hammervold, Johanne; Brattebø, Helge (2009): Combined MFA-LCA for Analysis of Wastewater Pipeline Networks. In *Journal of Industrial Ecology* 13 (4), pp. 532–550. DOI: 10.1111/j.1530-9290.2009.00143.x.

Wäger, P. A.; Hischier, R.; Eugster, M. (2011): Environmental impacts of the Swiss collection and recovery systems for Waste Electrical and Electronic Equipment (WEEE): a follow-up. In *The Science of the total environment* 409 (10), pp. 1746–1756. DOI: 10.1016/j.scitotenv.2011.01.050.

WCED (1991): Our common future. 13. impr. Oxford: Univ. Press (Oxford paperbacks).

Wender, Ben A.; Foley, Rider W.; Prado-Lopez, Valentina; Ravikumar, Dwarakanath; Eisenberg, Daniel A.; Hottle, Troy A. et al. (2014): Illustrating anticipatory life cycle assessment for emerging photovoltaic technologies. In *Environmental science & technology* 48 (18), pp. 10531–10538. DOI: 10.1021/es5016923.

Wesnaes, Marianne; Hansen, Peter Skov (2021): EcoReport Tool and Aspects of Circular Economy: Analysis of the possibilities for implementing aspects of Circular Economy in EcoReport Tool. Edited by SDU Livscykluscenter.

Wiedenhofer, Dominik; Fishman, Tomer; Lauk, Christian; Haas, Willi; Krausmann, Fridolin (2019): Integrating Material Stock Dynamics Into Economy-Wide Material Flow Accounting: Concepts, Modelling, and Global Application for 1900–2050. In *Ecological Economics* 156, pp. 121–133. DOI: 10.1016/j.ecolecon.2018.09.010.

Wierda, Leo; Li, William; Kemna, René (2021): Ecodesign Impact Accounting Annual Report 2020. Overview and Status Report. Publications Office of the European Union. Available online at https://op.europa.eu/en/publication-detail/-/publication/568cac02-5191-11ec-91ac-01aa75ed71a1/language-en.

Wiesmeth, Hans (2021): Implementing the Circular Economy for Sustainable Development. Amsterdam, Netherlands: Elsevier.

Wiprächtiger, Maja; Haupt, Melanie; Heeren, Niko; Waser, Eliane; Hellweg, Stefanie (2020): A framework for sustainable and circular system design: Development and application on thermal insulation materials. In *Resources, Conservation and Recycling* 154, p. 104631. DOI: 10.1016/j.resconrec.2019.104631.

Wiprächtiger, Maja; Rapp, Martina; Hellweg, Stefanie; Shinde, Rhythima; Haupt, Melanie (2022): Turning trash into treasure: An approach to the environmental assessment of waste prevention and its application to clothing and furniture in Switzerland. In *Journal of Industrial Ecology* 26 (4), pp. 1389–1405. DOI: 10.1111/jiec.13275.

Withanage, Sohani Vihanga; Habib, Komal (2021): Life Cycle Assessment and Material Flow Analysis: Two Under-Utilized Tools for Informing E-Waste Management. In *Sustainability* 13 (14), p. 7939. DOI: 10.3390/su13147939.

Yang, Xining; Hu, Mingming; Tukker, Arnold; Zhang, Chunbo; Huo, Tengfei; Steubing, Bernhard (2022a): A bottom-up dynamic building stock model for residential energy transition: A case study for the Netherlands. In *Applied Energy* 306, p. 118060. DOI: 10.1016/j.apenergy.2021.118060.

Yang, Xining; Hu, Mingming; Zhang, Chunbo; Steubing, Bernhard (2022b): Key strategies for decarbonizing the residential building stock: Results from a spatiotemporal model for Leiden, the Netherlands. In *Resources, Conservation and Recycling* 184, p. 106388. DOI: 10.1016/j.resconrec.2022.106388.

Yokota, Kazuhiko; Matsuno, Yasunari; Yamashita, Masaru; Adachi, Yoshihiro (2003): Integration of Life Cycle assessment and population balance model for assessing environmental impacts of product population in a social scale case studies for the global warming potential of air conditioners in Japan. In *Int J Life Cycle Assess* 8 (3), pp. 129–136. DOI: 10.1007/BF02978457.

Zeller, V.; Lavigne, C.; D'Ans, P.; Towa, E.; Achten, W. M. J. (2020): Assessing the environmental performance for more local and more circular biowaste management options at city-region level. In *The Science of the total environment* 745, p. 140690. DOI: 10.1016/j.scitotenv.2020.140690.

Zhang, Fan; Ding, Yuze; Deng, Xiangzheng; Liu, Litao (2022): Prediction of China's copper material flows under carbon emissions projections for the shared socioeconomic pathways.

In *Resources, Conservation and Recycling* 182, p. 106340. DOI: 10.1016/j.resconrec.2022.106340.

Zhang, Ning; Wang, Huiyue; Gallagher, John; Song, Qingbin; Tam, Vivian W.Y.; Duan, Huabo (2020): A dynamic analysis of the global warming potential associated with air conditioning at a city scale: an empirical study in Shenzhen, China. In *Environmental Impact Assessment Review* 81, p. 106354. DOI: 10.1016/j.eiar.2019.106354.

Ziemann, Saskia; Müller, Daniel B.; Schebek, Liselotte; Weil, Marcel (2018): Modeling the potential impact of lithium recycling from EV batteries on lithium demand: A dynamic MFA approach. In *Resources, Conservation and Recycling* 133, pp. 76–85. DOI: 10.1016/j.resconrec.2018.01.031.