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## **Towards a Circular Economy with Environmental Product Policy**

### **Considering dynamics in closing and slowing material loops for lighting products**

Richter, Jessika Luth

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# Towards a Circular Economy with Environmental Product Policy

Considering dynamics in closing and slowing material loops for lighting products

JESSIKA LUTH RICHTER | IIIIEE | LUND UNIVERSITY



# Towards a Circular Economy with Environmental Product Policy

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material loops for lighting products

Jessika Luth Richter



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<b>Abstract</b>  <p>A Circular Economy (CE) can help to achieve the sustainable development goal of responsible consumption and production. Product policies, in turn, can support CE objectives by promoting reuse of products, recycling of materials, and providing ecodesign incentives for more durable products. This thesis examined the role of product policies in meeting CE objectives in the EU, with specific focus on lighting products in the context of the Waste Electrical and Electronic Equipment (WEEE) and Ecodesign Directives. The research findings contributed to current policy questions, including how well WEEE systems have performed in closing material loops; the potential for closing loops for critical materials; and what trade-offs can occur in promoting longer lifetimes for rapidly developing products.</p> <p>A theory-based evaluation was used to assess the performance of extended producer responsibility (EPR) policies for lighting products in the Nordic countries. While the WEEE systems were generally performing well, there were issues identified, including the downcycling and loss of many recycled materials and lack of ecodesign incentives. The research also found that the requirements of the WEEE Directive were a key enabler for closing loops for rare earth elements (REE) from lighting products, but that the recycling efforts in the EU face challenges with economic feasibility and complex transactions in the value chain.</p> <p>The lifetimes of LED lighting products were examined from a consumer perspective through a life cycle cost analysis and an environmental perspective with life cycle assessment. From a consumer perspective, lifetimes much longer than the mandatory Ecodesign minimums were found to be optimal for LED products on the Swedish market. From an environmental perspective, longer lifetimes for LED lighting products can result in trade-offs between energy/climate impacts and resource depletion/toxicity impacts. However, in the context of a less carbon-intensive electricity mix, these trade-offs are minimised. The same is true if the product's energy efficiency improvements slow or mature.</p> <p>The research suggested that more specific product and material targets in the WEEE Directive could be appropriate. While the findings indicated that more stringent mandatory lifetime requirements in the Ecodesign Directive may not be appropriate for products with rapid technological developments, dynamic trade-offs should be explicitly recognised in policy mixes and accounted for in policy planning.</p>			
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# Abstract

A Circular Economy (CE) can help to achieve the sustainable development goal of responsible consumption and production. Product policies, in turn, can support CE objectives by promoting reuse of products, recycling of materials, and providing ecodesign incentives for more durable products. This thesis examined the role of product policies in meeting CE objectives in the EU, with specific focus on lighting products in the context of the Waste Electrical and Electronic Equipment (WEEE) and Ecodesign Directives. The research findings contributed to current policy questions, including how well WEEE systems have performed in closing material loops; the potential for closing loops for critical materials; and what trade-offs can occur in promoting longer lifetimes for rapidly developing products.

A theory-based evaluation was used to assess the performance of extended producer responsibility (EPR) policies for lighting products in the Nordic countries. While the WEEE systems were generally performing well, there were issues identified, including the downcycling and loss of many recycled materials and lack of ecodesign incentives. The research also found that the requirements of the WEEE Directive were a key enabler for closing loops for rare earth elements (REE) from lighting products, but that the recycling efforts in the EU face challenges with economic feasibility and complex transactions in the value chain.

The lifetimes of LED lighting products were examined from a consumer perspective through a life cycle cost analysis and an environmental perspective with life cycle assessment. From a consumer perspective, lifetimes much longer than the mandatory Ecodesign minimums were found to be optimal for LED products on the Swedish market. From an environmental perspective, longer lifetimes for LED lighting products can result in trade-offs between energy/climate impacts and resource depletion/toxicity impacts. However, in the context of a less carbon-intensive electricity mix, these trade-offs are minimised. The same is true if the product's energy efficiency improvements slow or mature.

The research suggested that more specific product and material targets in the WEEE Directive could be appropriate. While the findings indicated that more stringent mandatory lifetime requirements in the Ecodesign Directive may not be appropriate for products with rapid technological developments, dynamic trade-offs should be explicitly recognised in policy mixes and accounted for in policy planning.

# Popular Science Summary

What really happens when we dispose of our lighting products? Critical raw materials in our lighting products have high value, but can that value be retained in Europe? LED lamps have many potential benefits over traditional lighting products, including longer lifetimes; but do long lifetimes mean we miss out on potential benefits of even newer technology?

Why do these questions matter? Our present system of production and consumption exceeds and damages the planetary systems on which it depends. That means we need to rethink the way products are produced, but also how they are consumed and what happens to them after use. Transitioning towards responsible consumption and production is a United Nations Sustainable Development Goal (number 12) and the aim of a Circular Economy. A Circular Economy encourages more efficient use of resources by closing material loops with recycling, and slowing loops by keeping products longer in a cycle; for example, through increased durability or repair.

However, there are challenges. Currently, high levels of consumption contrast with low levels of reuse and recycling, especially for critical raw materials. At the same time, decreasing product lifetimes further accelerate consumption. The EU's Circular Economy Action Plan, introduced in 2015, highlighted the need for enhancing current policies targeting products. This thesis focusses on policies for lighting products, a product group that exemplifies many of the current product policy issues related to slowing and closing material loops. The aim of this research was to address gaps in knowledge about the performance of existing product policies and potential improvements in relation to the EU's Circular Economy objectives.

The thesis evaluates the performance of extended producer responsibility (EPR) policies for lighting products in the Nordic countries. EPR policies intend to make producers responsible for the environmental impacts of their products throughout their life cycles (i.e. from production to end-of-life). In the EU, the Waste Electrical and Electronic Equipment (WEEE) Directive places responsibility on producers to set up systems for end-of-life management of lighting products. The WEEE Directive also aims to incentivise ecodesign, recover valuable materials, and close materials loops with mandatory targets for collection and recycling.

The evaluation of Nordic country EPR systems for lighting products reveals that collection and recycling of lighting products have improved with the introduction

of EPR. Lighting products have been collected at a relatively high rate. The research suggest key factors for the high collection, including the convenience of the collection system and the high awareness amongst stakeholders of the risk from mercury.

However, there are still challenges with meeting some of the policy goals. The WEEE Directive's quantitative targets do not address the issues of downcycling (i.e. recovery of materials, but in low-value uses or products) or loss of valuable materials altogether in the current recycling processes. This loss of value and material is in conflict with the objectives of value retention and resource efficiency expressed in both the WEEE Directive and in the EU's Circular Economy Action Plan. There were also few indications that the WEEE Directive, as implemented in the Nordic countries (with producers sharing responsibility in collective organisations), is incentivising individual producers towards ecodesign that could make increased recycling and recovery of materials easier.

When the research for this thesis began in 2014, there was considerable concern about critical materials, which are important to the EU economy but face supply risks. There were few successful industrial-scale examples of recycling for critical raw materials in the EU – one was a company recycling rare earth elements (REEs) from lighting products. This case was analysed to understand the enabling factors and potential for recycling REE from lighting products. Interestingly, the specific requirements in the WEEE Directive to collect and remove mercury from lighting products was one of the key enabling factors for the feasibility of REE recycling.

However, REE recycling from lighting products in the EU still faced competition from supply of REE from mining and processing in China. When the high prices of REE dropped after 2012 (due to a change in Chinese export restrictions), recycling initiatives in the EU struggled. Understanding the potential for REE recycling also involved examining the dynamics of complex global value chains. To better understand these dynamics, the research described the complex transactions in the governance structure of the value chain for REE recycling from lighting products. The research identified highly complex transactions and challenges for business actors in capturing social and environmental values.

Product policies for a Circular Economy aim not only to incentivise recycling of materials, but also to slow material loops through longer product lifetimes. In the EU, the Ecodesign Directive sets minimum energy-efficiency and functionality requirements for products, including minimum lifetime requirements for lighting products. However, since the introduction of the Ecodesign Directive in 2009, LED lighting products have been rapidly developing in terms of energy-efficiency and material design, which raises a question about whether longer lifetimes would result in increased costs for consumers and higher environmental impacts by locking in less efficient technologies.

The consumer perspective to the question of longer lifetimes was analysed with an assessment of life cycle costs of LED lamps on the market in Sweden. Lifetimes around 25000 hours (much longer than the current policy-mandated minimum lifetimes) were found to be optimal. However, the optimal lifetimes of LED lamps would be lower for the consumer in the context of higher electricity prices and increasing efficiency of replacement LED lamps.

From an environmental perspective, life cycle assessment results showed longer lifetimes for LED lamps could be better in terms of energy and climate impacts but worse in terms of resource depletion and toxicity impacts. However, context matters. When the electricity mix considered is cleaner, the climate and energy-related impacts from using LED lamps lessen overall, and in relation to impacts from production of the lamps. This means that in contexts like Norway and Sweden, with relatively clean electricity mixes, longer product lifetimes (and using products longer) are preferable when considering all environmental impacts. In all contexts, trade-offs between different impacts are minimised when energy-efficiency improvements of new products slow or mature.

While further research would strengthen the findings, some preliminary implications for policy can be drawn. The research suggested that continued improvement in collection of lighting products could be motivated by a collection target set specifically for lighting products and other small electronics. Recycling targets in the WEEE Directive could specifically target critical raw materials. There is a need to explore how policies can capture the environmental and social benefits of the recycled materials, also in utilising the recycled materials in products, e.g. through voluntary green procurement criteria or mandatory ecodesign requirements for recycled content.

While the findings indicate it may be appropriate at the EU level not to implement longer lifetime requirements for products with rapid technological developments, trade-offs should be discussed. Product development, and the push of product development by minimum standards, should inform a timeline for implementing appropriate lifetime measures. For contexts with low-carbon electricity mixes, it is already preferable to promote longer lifetimes for all products.

The findings of the research have broader implications for the transition towards a Circular Economy. A holistic approach needs to be taken in considering priorities for environmental policies. At times resource efficiency aims may undermine climate mitigation aims. However, trade-offs are also dynamic and often temporary, necessitating understanding of policy targets and roadmaps from both areas in order to optimise policy synergies.

# List of Papers

- Paper I Richter, J. L., & Koppejan, R. (2016). Extended producer responsibility for lamps in Nordic countries: best practices and challenges in closing material loops. *Journal of Cleaner Production*, 123, 167–179.
- Paper II Machacek, E., Richter, J.L., Habib, K., & Klossek, P. (2015). Recycling of rare earths from fluorescent lamps: Value analysis of closing-the-loop under demand and supply uncertainties. *Resources, Conservation and Recycling*, 104, Part A, 76–93.
- Paper III Machacek, E., Richter, J. L., & Lane, R. (2017). Governance and Risk–Value Constructions in Closing Loops of Rare Earth Elements in Global Value Chains. *Resources*, 6(4), 59.
- Paper IV Richter, J. L., Van Buskirk, R., Dalhammar, C., & Bennich, P. (2019). Optimal durability in least life cycle cost methods: the case of LED lamps. *Energy Efficiency*, 12(1), 107–121.
- Paper V Richter, J.L., Dalhammar, C., Tähkämö, L. (2019). Trade-offs with longer lifetimes? The case of LED lamps considering product development and energy contexts. *Journal of Cleaner Production*, 226, 195-209.

# Contribution to Papers

- Paper I J.L.R. developed the idea and research design, collected data, conducted the analysis and wrote the paper, which received comments and input from the co-author. J.L.R. revised the article pursuant to peer reviewer comments.
- Paper II All co-authors discussed the research design. J.L.R. collected data on recycling for the modelling and supported the case study analysis for the paper. E.M. and J.L.R. took leading roles in writing and revising all sections of the paper pursuant to peer reviewer comments.
- Paper III E.M., J.L.R. and R.L. designed the analytical framework; J.L.R. collected and analysed data for the EoL lamp case; all three authors participated in several rounds of conceptual discussions; and E.M. and J.L.R. wrote the paper and revised pursuant to peer review comments.
- Paper IV J.L.R. and R.V.B. developed the idea and research design. J.L.R. conducted the literature review to frame and give data input and comments to R.V.B.'s LCC modelling. J.L.R. led the writing of the paper, with input from co-authors, and revised the article pursuant to peer reviewer comments.
- Paper V J.L.R. developed the idea and research design in discussion with co-authors. J.L.R. conducted the LCA and wrote the paper, which received comments and input from the co-authors. J.L.R. also revised the article pursuant to peer reviewer comments.

# Other Publications

Dalhammar, C., Milios, L., & Richter, J. L. (2019). Ecodesign and the Circular Economy: Conflicting policies in Europe. *Proceedings of EcoDesign 2019 International Symposium*.

Richter, J. L., & Dalhammar, C. (2019). Drivers, barriers, and stakeholder roles in electronics repair: a case study of Sweden. *3rd Product Lifetimes and the Environment (PLATE) Conference Proceedings*.

Dalhammar, C., & Richter, J. L. (2019). Interdisciplinary Research on Energy Efficiency Standards and Climate Change Mitigation: Methods, Results, and Communication. In W. Leal Filho & R. Leal-Arcas (Eds.), *University Initiatives in Climate Change Mitigation and Adaptation* (pp. 333–350). Cham: Springer International Publishing.

Palm, J., Richter, J. L., Curtis, S., Wickenberg, B., Schoonover, H. (2018). Article Incubator: building interdisciplinary academic writing skills amongst PhD students. *LTHs 10:e Pedagogiska Inspirationskonferens Proceedings*.

Svensson, S., Richter, J. L., Maitre-Ekern, E., Pihlajarinne, T., Maigret, A., Dalhammar, C. (2018). The Emerging ‘Right to Repair’ legislation in the EU and the U.S. *CARE Innovation Electronics Goes Green Conference Proceedings*.

Richter, J.L., Johnson, M., De Schepper, K., Fitzpatrick, C., Rückschloss, J., Gunter, J., Peeters, J., (2018). More Aware through Repair: educating about critical raw materials. *CARE Innovation Electronics Goes Green Conference Proceedings*.

Sonnenschein, J., Van Buskirk, R., Richter, J. L., & Dalhammar, C. (2019). Minimum energy performance standards for the 1.5 °C target: An effective complement to carbon pricing. *Energy Efficiency*, 12(2), 387–402.

Dalhammar, C., Richter, J. L., & Machacek, E. (2018). Energy efficiency regulations, market and behavioural failures, and standardization. In *Preventing Environmental Damage from Products: An Analysis of the Policy and Regulatory Framework in Europe* (pp. 176–228). Cambridge University Press.

Richter, J. L., Dalhammar, C., Tähkämö, L. (2017). Considering optimal lifetimes for LED lamps: a mixed approach and policy implications. *2nd Product Lifetimes and the Environment (PLATE) Conference Proceedings*.

Richter, J. L., Van Buskirk, R., Dalhammar, C., Bennich, P. (2017). Accounting for durability in least life cycle cost methods. *ECEEE Summer Study*.

Sonnenschein, J.; Van Buskirk, R.; Richter, J.L.; Dalhammar, C. (2017). When Prices Don't Steer – mimicking ambitious carbon pricing with energy performance standards. *ECEEE Summer Study*.

Lindhqvist, T., Richter, J. L., Rodhe, H. (2017). Global learning in local contexts: designing, maintaining, and learning from authentic tasks. In B. Kiss & N. Tojo (Eds.), *Diversity in Education: Crossing cultural, disciplinary and professional divides* (pp. 79-110). Lund University.

Richter, J. L. (2016). *Extended Producer Responsibility for Closing Material Loops: Lessons from energy-efficient lighting products*. Licentiate dissertation. Lund University.

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Plepys, A. & Richter, J. L. (2016). Public Procurement Barriers in Promoting Market Uptake of Innovative LED Lighting. *Electronics Goes Green Conference Proceedings*.

Leire, C., McCormick, K., Richter, J. L., Arnfalk, P., & Rodhe, H. (2016). Online teaching going massive: input and outcomes. *Journal of Cleaner Production*, 123, 230–233.

Karlsson, R., Kunen, J. Richter, J. L., Karlsson, T., Will, M. (2016). *D2.8 Final Report on sustainability issues*. SSLerate Project.

Mundaca, L., & Richter, J. L. (2015). Assessing “green energy economy” stimulus packages: Evidence from the U.S. programs targeting renewable energy. *Renewable and Sustainable Energy Reviews*, 42, 1174–1186.

Gielen, S, Plepys, A., Karlsson, R., Richter, J. L., Lindqvist, T., Kunen, J. (2014) *D2.1 Report and website material on sustainability issues*. SSLerate Project.

# Preface

The International Institute for Industrial Environmental Economics (IIIEE) at Lund University was the setting in which this research was conducted. The IIIEE is characterised by its interdisciplinary research and educational environment, cooperation with a variety of societal actors, and emphasis on societal relevance.

The research in this thesis was framed by several research projects indicative of this research environment. The first part of the research (Papers I-II) was conducted within the two year project (2014-2016) “Policy instruments and business models for closed material loops” financed by the Swedish Energy Agency (project number 37655-1), which focussed the case of lighting products. The original research proposal was written by Dr. Thomas Lindhqvist and Dr. Naoko Tojo, and reviewed and approved by the Swedish Energy Agency. The more detailed research design was then developed independently with guidance from academic supervisors. The research design was also influenced by Lund University’s participation in an EU coordination project, SSLerate (2014-2016). The project aimed to accelerate the uptake of high-quality Solid State Lighting (SSL) technology in Europe by supporting open innovation and bringing validated information to all relevant stakeholders. The project was part of the Seventh EU Framework Programme for Research and Technological Development.

Subsequent projects framing the research for Papers III-V included the EU Interreg project “Lighting Metropolis” (2015-2018), the Swedish Energy Agency funded project “Promoting ecodesign for advancement of innovation, competitiveness and environmental improvements: assessing the potential of the Ecodesign Directive in a dynamic setting” (2013-2017) and the FORMAS funded project “Circular Economy: capturing value in waste through extended producer responsibility policies” (2018-2020).

# Abbreviations

- CE – Circular Economy
- CFL – compact fluorescent lamp
- CRM – critical raw material
- EEE – electrical and electronic equipment
- EoL – end-of-life
- EPR – extended producer responsibility
- GDL – gas discharge lamp
- GCC – global commodity chain
- GPN – global production network
- GPP – green public procurement
- GVC – global value chain
- IPP – integrated product policy
- IPR – individual producer responsibility
- LED – light emitting diode
- LCA – life cycle assessment
- LCI – life cycle inventory
- LCC – life cycle costs
- MEPs – minimum energy performance standards
- MEErP - methodology for ecodesign of energy-related products
- MFA – material flow analysis
- PRO – producer responsibility organisation
- REE(s) – rare earth element(s)
- RoHS – restrictions on the use of hazardous substances
- WEEE – waste electrical and electronic equipment

# 1 Introduction

Sustainability research generally seeks to address the major challenges facing human society, which give rise to the need for a transition to a low carbon and resource-efficient economy. This need is evident in the many climate and resource efficiency targets in the United Nations (UN) sustainable development goals (SDGs). Achieving a more resource-efficient economy involves more responsible consumption and production – sustainable development goal (SDG) 12. Major changes are needed to our present system of consumption and production, indeed to our entire economy, that currently exceeds and damages the planetary systems on which it depends (Costanza & O’Neill, 1996). Current consumption patterns are not only associated with high greenhouse gas emissions, but also result in resource over-extraction and other forms of environmental degradation (United Nations, 2019).

However, the UN Department of Economic and Social Affairs notes that material consumption continues to expand rapidly across the globe and there remain many gaps in both knowledge and action, which slow and jeopardise progress towards SDG 12 (and the other Goals) (United Nations Economic and Social Council, 2019). As material consumption increases, the global natural ecosystem is shrinking (Korhonen, 2006). High levels of consumption are contrasted with low levels of material and product reuse and recycling (Haas, Krausmann, Wiedenhofer, & Heinz, 2015), exacerbated by decreasing product lifetimes, which spur even faster consumption rates (Bakker, Wang, Huisman, & den Hollander, 2014). The UN’s call for urgent action notes the need for “policies that improve resource efficiency, reduce waste and mainstream sustainability practices across all sectors of the economy” (United Nations Economic and Social Council, 2019, p. 18).

Environmental policies targeting production and resource efficiency are not new, but increasingly have developed to better consider consumption and take a life cycle perspective (Dalhammar, 2007). Similarly, the UN has emphasised that responsible consumption and production policies should consider the whole life cycle of products and range of stakeholders in supply chains (United Nations, 2019). The goal of responsible consumption and production, with consideration of products and their life cycles, has synergies with another emerging umbrella concept – that of the Circular Economy (CE) (Charter, 2018). Indeed, the achievement of targets for SDG 12 (and other SDGs) is argued to be supported by CE policies and practices (Schroeder, Anggraeni, & Weber, 2019).

## 1.1 From product policy to Circular Economy

The traditional focus of environmental policies concerning products tended to be on addressing environmental impact of production processes and point source pollution and were initially weak in addressing diffuse emissions throughout the product life cycle and consumption-oriented environmental impacts (Dalhammar, 2007). This changed in the EU with the institutionalisation of life cycle thinking (Heiskanen, 2002), which influenced the development of integrated product policy (IPP). With the adoption of IPP as a strategy in 2003, the EU Commission argued that “life cycle thinking needs to become second-nature to all those who come into contact with products” (Commission to the Council and the European Parliament, 2003, p. 10).

IPP reflected an ongoing transition in the role and principles of governance from the government-led command and control to negotiation with stakeholders and sharing of responsibilities and incentives (Scheer & Rubik, 2017). Early IPP discussions considered how stakeholder responsibilities and incentives should be allocated (Rubik & Scholl, 2002). Similarly, an increasing emphasis on life cycle thinking, source prevention and the need for incentives had influenced waste laws in the EU towards the waste hierarchy and integration of extended producer responsibility (EPR) (Tojo, 2004). EPR allocated responsibilities to producers (and other actors) for end-of-life (EoL) management while also intending to provide incentives for ecodesign.

Reflecting IPP, new policies were introduced and changes were made to existing policies targeting a wide range of product groups in the EU. These included electrical and electronic equipment (EEE), vehicles, packaging, chemical substances, pesticides, etc. There are different types of policies; e.g. ranging from mandatory policies such as minimum performance standards, EPR policies, restrictions on hazardous substances and labelling to voluntary policies such as procurement, process standards and eco-labelling. IPP also supported the breakthrough of “life cycle thinking” as an important concept in environmental law. The diversity of policies reflect the complexity of addressing different life cycle stages and the need to consider the unique attributes of these stages specific to the product group. In understanding the environmental impacts from different life cycle stages, tools and methods such as life cycle assessment (LCA), life cycle costing (LCC), and life cycle management (LCM) are often used (Dalhammar, 2015).

The existing product policies form the basis for developing an EU Product Policy Framework pursuant to the CE Action Plan (EU Commission, 2015). The plan commits to evaluate current EU policies and recommend adjustments of existing instruments or new actions “that could improve the design, use and recycling of products in conformity with the objectives of a circular economy” (EU Commission, 2018b, p. 1). Researchers also argue that existing product policies require extension

and further development to effectively support the transition to a CE (Lazarevic & Valve, 2017). In addition, there is a need to consider dynamic issues such as critical raw materials (CRMs) (EU Commission, 2015). As EPR and ecodesign policies are key policies in the EU's CE Action Plan (EU Commission, 2015), this means they also need to be evaluated in light of the objectives of a CE, including the need for retention of value, promoting longer product lifetimes and considering trade-offs (Charter, 2018). Indeed the need for evaluation and addressing trade-offs is well recognised in EU environmental policy and part of the EU Environment Action Programme to 2020.<sup>1</sup>

The product policy context has been shifting from a narrow focus on production and avoiding negative environmental impacts to a more complex focus on consumption and retaining value in resources. At the same time, products themselves have been developing rapidly, with many of them also becoming increasingly complex. Lighting products are an example of a rapidly developing product and a case through which multiple issues for CE policies can be explored in more depth. The evaluation of existing CE policies and exploration of issues is the broad focus of this thesis.

## 1.2 Product policies for lighting products

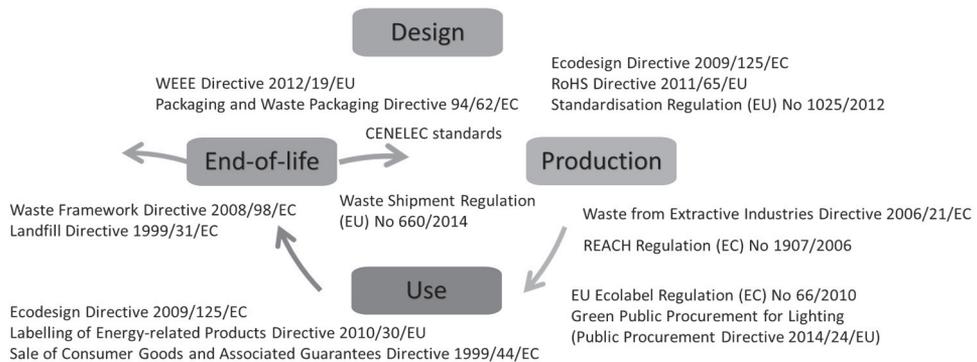
Electric lighting accounts for 15% of power consumption and 5% of global greenhouse gas emissions (United Nations Environment Programme (UNEP), 2019), switching to energy-efficient lighting products has potential for emissions reductions in developed countries and sustainable development generally. An integrated policy approach for lighting products has been promoted at both the international policy level (through UNEP's public private partnership "En.Lighten", now "United for Energy" (UNEP, 2013), and the EU policy level.

An overview of product policies relevant for lighting products and addressing different life cycle stages at the EU level is shown in Figure 1. The figure shows that there are policies addressing each major stage of the life cycle. For a general lighting product's life cycle, the highest environmental impacts stem from the use stage. This also means that there is a clear decrease in overall environmental impacts for energy-efficient lighting compared to traditional incandescent technologies (Tähkämö, Martinsons, Ravel, Grannec, & Zissis, 2014; U.S. Department of Energy, 2013; Welz, Hischer, & Hilty, 2011). At the international level, UNEP's policy programme emphasises energy efficiency in the use stage supported by

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<sup>1</sup> Decision No 1386/2013/EU of the European Parliament and of the Council of 20 November 2013 on a General Union Environment Action Programme to 2020 'Living well, within the limits of our planet'

minimum energy performance standards (MEPs) (UNEP, 2013). MEPs for lighting products have been implemented in many countries and regions including the EU (through ecodesign requirements), North America, South America, Asia and Australia (Mautone, 2017).



**Figure 1.** Overview of EU policies addressing life cycle stages relevant for lighting products. Voluntary policies are italicised. Adapted from Milios (2017).

Another environmental hotspot is the mercury contained in gas discharge lamps (i.e. fluorescent lighting products). A life cycle approach shows that energy efficient lighting products containing mercury still lead to lower mercury emissions in the life cycle compared to incandescent lighting when considering mercury from any coal electricity sources in the use stage (see e.g., Tähkämö et al., 2014; U.S. Department of Energy, 2013). This is why both the Minamata Convention and the Restriction on Hazardous Substances (RoHS) Directive<sup>2</sup> make exceptions for lighting products (but also limits on amounts of mercury per product).

While the EoL stage generally has a relatively low impact for lighting products (see Tähkämö et al., 2014; U.S. Department of Energy, 2013), there can be a high local environmental impact if large quantities of mercury-containing products are released into the environment (Wagner, 2011) or if filter technology is not used in incineration processes (Silveira & Chang, 2011). The fragility of these products also raises health concerns for handling waste gas discharge lamps (Kasser & Savi, 2013; Sander, Schilling, Wagner, & Günther, 2013). Policies for sound environmental management of lighting products can reduce mercury contamination and exposure (UNEP, 2013).

<sup>2</sup> Directive 2002/95/EC of the European Parliament and of the Council of 27 January 2003 on the restriction of the use of certain hazardous substances in electrical and electronic equipment (recast in 2011)

EPR policies have gained traction as a way to ensure sound management at EoL. EPR policies covering lighting products have been implemented in the EU, Norway, Switzerland, the U.S. (in some states, e.g. Maine and Washington), and Taiwan. New EPR legislation is also proposed or being discussed in other countries, for example India and China, where the large-scale installations of fluorescent lighting pose potential issues if not properly managed as these reach EoL (see e.g. Pandey, Hooda, & Mishra, 2012; Tan & Li, 2014).

Between 2014 and 2019, the time of this research, energy-efficient lighting products were in a period of rapid development with declining prices and increasing efficiency of LED products. Between 2011 and 2015, for example, LED household retrofit lamps declined in price by 32% per year (Gerke, Ngo, & Fisseha, 2015). Meanwhile, energy efficiency has improved drastically as well. The best available technology had an efficiency of 69 lumens per watt (lm/W) when the EU implemented ecodesign requirements for lighting products in 2009<sup>3</sup>. The global market average was 99 lm/W 10 years later, with some individual lamp efficiencies as high as 200 lm/W, and projected to increase to an average of 160 lm/W by 2030 (International Energy Agency (IEA), 2019). The IEA noted the important role of policies in ensuring continuous efficiency development of LED lighting and market uptake (IEA, 2019). At the same time, governments noted the need to ensure quality of the LED products to avoid the backlash that was seen against promotion of compact fluorescent products earlier (Sandahl, Cort, & Gordon, 2014; Sandahl, Gilbrade, Ledbetter, Steward, & Calwell, 2006).

At the EU level, the main mandatory policies ensuring energy efficiency, functionality, quality, and sound EoL management are the regulations for lighting under the Ecodesign Directive<sup>4</sup> and the waste electrical and electronic equipment (WEEE) Directive<sup>5</sup> (EU Commission, 2019a). As the WEEE and Ecodesign Directives are the policies that are the focus for this thesis, more background is provided about these policies and previous evaluation research in this area in the next section.

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<sup>3</sup> Commission Regulation (EC) No 244/2009 of 18 March 2009 implementing Directive 2005/32/EC of the European Parliament and of the Council with regard to ecodesign requirements for non-directional household lamps

<sup>4</sup> 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

<sup>5</sup> Directive 2002/95/EC of the European Parliament and of the Council of 27 January 2003 on the restriction of the use of certain hazardous substances in electrical and electronic equipment (recast in 2012)

### 1.2.1 WEEE Directive

In the EU, the WEEE Directive has implemented EPR for electronic waste, including lighting products, in EU member states and banned landfilling of WEEE covered by the legislation. The original legislation placed responsibility on producers to set up systems for collection and treatment of WEEE. Its aims aligned with the waste hierarchy to first prevent, then reuse, recycle and recover WEEE. It also sought to improve the environmental performance of operations throughout the life cycle, in particular for treatment (Article 1 of 2002 version of legislation). The original Directive set collection targets of 4kg per capita and recycling and recovery targets for particular products categories (70% recovery and 50% reuse/recycling rate for lamps and lighting equipment – belonging to category 5). Notably, gas discharge lamps (GDLs), or fluorescent lighting products, have a target rate for reuse (component, material and substance) and recycling of at least 80% by weight of the lamps (Article 7) and required removal of mercury (Annex II).

The recast of the legislation in 2012<sup>6</sup> consolidated the regulated WEEE product categories from ten to six from 2018, with lamps in category 3. From 2016, overall collection targets were changed from weight collected per capita to 45% of the average weight of EEE placed on the market in the three preceding years in the member state. From 2019, the minimum collection rate rose to 65% of the average weight of EEE placed on the market in the three preceding years, or alternatively 85% of WEEE generated in the Member state. The reuse and recycling target for all lamps is 80% and the requirement to remove mercury remains. Article 5 adds an additional obligation for distributors to provide for the free collection at retail shops of very small WEEE.<sup>7</sup> At the EU level, there is monitoring of how the WEEE Directive is performing, mainly in relation to how member states are implementing the Directive meeting the collection and recycling targets set by the Directive. The Eurostat data has indicated varying levels of performance on collection and recycling depending on member states (EU Commission, 2019b).

Improving the EoL management of WEEE through collection and recycling results in clear environmental benefits (Hischier, Wäger, & Gauglhofer, 2005). The specific achievements of the WEEE Directive are noted in evaluations (Goodship & Stevels, 2012; Huisman et al., 2008; Román, 2012), though actual performance of systems in different member states has varied (Khatriwal, Widmer, Kuehr, & Huisman, 2011; Magalini et al., 2014). Other challenges remain: in particular, incentives for waste prevention through ecodesign (Huisman, 2013; Huisman et al.,

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<sup>6</sup> Directive 2012/19/EU of the European Parliament and of the Council of 4 July 2012 on waste electrical and electronic equipment (WEEE) (recast)

<sup>7</sup> Pertains to retail shops with sales areas relating to EEE of at least 400 m<sup>2</sup> and WEEE with no external dimension more than 25 cm.

2008; Kalimo et al., 2012; Lauridsen & Jørgensen, 2010; Lindhqvist & Lifset, 2003; Sander, Tojo, & Vernon, 2007; Van Rossem, 2008; Van Rossem, Lindhqvist, & Tojo, 2006). Most producers choose to meet their WEEE obligations through collective schemes, in which a producer responsibility organisation (PRO)<sup>8</sup> arranges for collection and recycling of products together and charge fees based on types of products, but not by brand, model, or other attributes that could differentiate ecodesigned products. Therefore, there is little incentive for individual producers to design for recyclability or reuse when they do not receive benefits for doing so. While implementing individual producer responsibility (IPR) has been proposed as a solution (and even argued to be possible within a collective scheme – see Mayers, Lifset, Bodenhofer, & van Wassenhove, 2013), this is not yet a reality. There also continue to be concerns from stakeholders, in particular producers, about issues such as enforcement, competition, scavenging of valuable waste, and design incentives (Kunz, Mayers, & van Wassenhove, 2018; OECD, 2016).

While much of the general WEEE research is relevant to the case of lighting products, this product category also has unique characteristics compared to many other WEEE streams. For example, the cost for collection and recycling of waste lamps is currently relatively high compared to the price of the product, with a low or negative value of the recovered material from lamp waste. The high cost for lamps is tied to the necessary recovery of hazardous materials increasing recycling costs, but also to challenges in collecting lamps that are lightweight, small, and dispersed. While clearly it is of societal value to avoid mercury contamination, this is an externality (i.e. it is not captured in market prices). Such externalities can be difficult to quantify in economic terms (Magalini et al., 2014). Lamps represent a classic product group for EPR policy to address: there is a net cost for treatment and treatment clearly avoids environmental harm, necessitating policy to ensure sound EoL management (Huisman et al., 2008). However, with WEEE Directive targets based on the overall weight of WEEE, challenges remain for collection of small WEEE (Janz & Rotter, 2006; Melissen, 2006).

### **1.2.2 The Ecodesign Directive**

Implemented in 2009<sup>9</sup>, the Ecodesign Directive has an aim to improve the environmental performance of energy-using products. The Ecodesign Directive is a

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<sup>8</sup> Producer Responsibility Organisations (PROs) are cooperative industry associations that act on behalf of its member companies to meet their EPR obligations. Generally, PROs bear the operational responsibility of managing WEEE.

<sup>9</sup> The 2009 Directive replaced 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

framework directive, meaning that it does not set direct requirements. These are set through specific implementing regulations for each group of products in the scope of the Directive, including specific regulations for lighting products.

In effect, the Directive places responsibility on producers to comply with mandatory energy and functionality standards in order to place their products on the EU market. The standards have a goal to promote sustainable development “by increasing energy efficiency and the level of protection of the environment, while at the same time increasing the security of the energy supply.” (Article 1(2)). The preamble further notes that “[in] the interest of sustainable development, continuous improvement in the overall environmental impact of those products should be encouraged, notably by identifying the major sources of negative environmental impacts and avoiding transfer of pollution, when this improvement does not entail excessive costs.” (Preamble (3)). Environmental performance is determined with an assessment of all life cycle stages. Life cycle costs to the consumer are also assessed. In addition, the impact on manufacturers and SMEs should be considered and stakeholders consulted during the assessments (Article 15). These assessments, in turn, determine the setting of the standards (Annex 1 and 2).

Some evaluations of the Ecodesign Directive have focussed primarily on ex ante models of potential energy saving and assumptions of compliance (Siderius & Nakagami, 2013), while others have examined the effects of implementing regulations on particular product markets (e.g. the lighting product market – see Danish Energy Agency, Energy Piano, & CLASP European Programme, 2015; VITO & VHK, 2015). A comprehensive ex post evaluation of the Ecodesign Directive used a theory-based approach complemented by stakeholder surveys and interviews to identify issues (see CSES, 2012). The evaluation found that while energy-efficiency improvements in the covered product categories were observed, the role of the Ecodesign regulations was unclear. Several stakeholders interviewed and surveyed perceived the minimum energy performance standards (MEPS) as unambitious and there have been challenges in dealing with rapidly developing technologies that have differed significantly from baselines and modelling assumptions used in setting the standards (CSES, 2012). Another evaluation, also identifying issues in a stakeholder-based approach, found that stringency of the MEPS has varied by product group in terms of the energy savings potentials and implementation (see Molenbroek et al., 2014). Both evaluations highlighted the challenge of setting regulations for rapidly developing product groups. It should be noted that Ecodesign regulations are complemented by mandatory energy labelling, meaning that the effects of both policies are often difficult to distinguish in such evaluations.

Another common issue found in the solicited stakeholder perspectives in both evaluations was the missed opportunities in addressing non-energy-related impacts

or life cycle phases (CSES, 2012; Molenbroek et al., 2014). The Ecodesign regulations have included minimum functionality and quality requirements and in some cases criteria for non-energy impacts, e.g. water usage; but often non-energy impacts have been scoped out or less emphasised (CSES, 2012). Some criteria have focussed on durability aspects, including minimum product lifetimes (Maitre-Ekern & Dalhammar, 2016), but more are possible and needed to promote resource efficiency, in addition to energy efficiency (Jepsen, Spengler, & Augsberg, 2017). However, developed standards for testing are a pre-condition to implementing mandatory criteria (Bundgaard, Mosgaard, & Remmen, 2017; Dalhammar, 2016; Tecchio, McAlister, Mathieux, & Ardenete, 2017).

Under the Ecodesign Directive, there are several regulations with specific ecodesign requirements for lighting products.<sup>10</sup> In 2018, it was decided to have a single lighting product regulation from 2021 (EU Commission, 2018a). Regulations for lighting products under the Directive have set minimum energy efficiency requirements, which have resulted in the phase-out of the least efficient incandescent lamps in 2009 and halogen lamps in 2018. In addition to efficiency, the ecodesign requirements for lighting set functionality requirements related to colour rendering, colour consistency, and other aspects related to quality and durability.

Lighting products are one of the first product groups to have minimum lifetimes specified under Ecodesign Directive. Measuring lifetimes for lamps has become more difficult with the increased complexity of LED lamps. These types of lamps may simply stop working (i.e. fail catastrophically) similar to early lamp technology or the light output may fade over time (known as lumen depreciation). Lifetime was defined in Appendix II of Regulation 1194/2012:

*'lamp lifetime' means the period of operating time after which the fraction of the total number of lamps which continue to operate corresponds to the lamp survival factor of the lamp under defined conditions and switching frequency. For LED lamps, lamp lifetime means the operating time between the start of their use and the moment when only 50 % of the total number of lamps survive or when the average lumen maintenance of the batch falls below 70 %, whichever occurs first.*

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<sup>10</sup> EU Ecodesign Directive implementing regulations relevant to lighting products are: Commission Regulation (EC) No 244/2009 of 18 March 2009 implementing Directive 2005/32/EC of the European Parliament and of the Council with regard to ecodesign requirements for non-directional household lamps; Commission Regulation (EU) No 1194/2012 of 12 December 2012 implementing Directive 2009/125/EC of the European Parliament and of the Council with regard to ecodesign requirements for directional lamps, light emitting diode lamps and related equipment; Commission delegated regulation (EU) No 874/2012 of 12 July 2012 supplementing Directive 2010/30/EU of the European Parliament and of the Council with regard to energy labelling of electrical lamps and luminaires

Ecodesign requirements for lamps were based on measurements made at 6000 hours (250 days), at which the lumen maintenance has to be  $\geq 80\%$ , and the lamp survival factor  $\geq 90\%$  (Annex III, Table 5).<sup>11</sup>

Other jurisdictions have also implemented minimum lifetimes for lamps, like California with its 10000 hour lifetime based on projected lifetime tests (California Energy Commission, 2017). The question remains how to best develop considerations of lifetimes in the EU eco-design requirements to balance environmental impacts and life cycle costs to consumers, as low costs to consumers may not always align with low environmental impacts. In addition, there is a need to consider the still improving energy efficiency of LED lighting products.

### 1.3 Research Gaps

Despite the introduction of EPR policies, there is little formal research specifically about the performance of policies promoting end-of-life management and closing material loops for lighting products. The Nordic countries have generally been cited as best performing countries in evaluation of the WEEE Directive (Román, 2012; Ylä-Mella et al., 2014). With most research focused on WEEE in general, there is limited knowledge about how effectively the EPR systems in Europe are performing in regard to lighting products in particular. The few studies which have focussed on lighting products in particular have been on state level actions in the U.S. (Silveira & Chang, 2011; Wagner, 2011) or particular challenges related to mercury and collection issues for lighting products (Magalini et al., 2014; Sander et al., 2013).

There is a need to evaluate the performance of such systems in relation to the longer-term outcomes of EPR (and in light of CE objectives), i.e. improving waste management, incentivising eco-design, and closing material loops and retaining value (described more in Section 3.3.2). The CE objectives shift from a focus on waste management from avoidance of hazards to capturing value from resources (Kama, 2015). In particular, there is interest in waste lamps for recovering valuable CRMs, such as the rare earth elements (REE) in the phosphors (needed to produce warm white light). Lighting products, as a source for recycling of REE, have been given considerable attention in research of technical recycling processes (see Dupont & Binnemans, 2015; Langer, 2012; Liu et al., 2013, 2014; Tan, Li, & Zeng,

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<sup>11</sup> On 1 October 2019, updated Ecodesign regulations for lighting products were communicated by the EU Commission – see EU Commission (2019a). These will take effect 1 September 2021. They include a definition of lifetime that only refers to lumen depreciation, and referring to the L<sub>70</sub>B<sub>50</sub> standard (i.e. when the light output for 50 % of a population of lamps has gradually degraded to a value below 70 % of the initial luminous flux). The testing hours was also reduced from a 6000 testing procedure to 3600 hour endurance testing procedure (Annex V).

2014; Tunsu, Petranikova, Ekberg, & Retegan, 2016; Tunsu et al., 2015). Lighting products have also been studied in geological and industrial ecology research about material criticality in which material flow estimates of available rare earths from the product group were modelled (Guyonnet et al., 2015; Mueller, Wäger, Turner, Shaw, & Williams, 2017; Rollat, Guyonnet, Planchon, & Tuduri, 2016). While many researchers and other stakeholders call for improved collection and recycling of lighting products (Asari, Fukui, & Sakai, 2008; Binnemans et al., 2013; Hu & Cheng, 2012; Huisman et al., 2008; Lim, Kang, Ogunseitan, & Schoenung, 2013; Tähkämö et al., 2014; Tian et al., 2016; UNEP, 2013; U.S. Department of Energy, 2013; Wagner, 2011), there is little research about how policy could develop to increase opportunities for closing critical material loops. For example, research predicting the potential supply from recycling rare earth phosphors from lamps (Binnemans et al., 2013; Rollat et al., 2016), has made general assumptions about the collection and recycling systems without examination of the current state and performance of implemented policies for lighting products and barriers to realising this potential. There is also a need to identify the assumptions about the value of the CRMs in the waste material.

In addition to closing material loops, CE aims for slowing material loops through longer products lifetimes, which keep resources, including CRMs, in use longer (Kahhat, Hieronymi, & Williams, 2012; Montalvo, Peck, & Rietveld, 2016; Peck, Kandachar, & Tempelman, 2015). Ecodesign, including for durability, is theoretically incentivised by EPR policies for waste products, but this has been a weakness of such policies, as they are implemented (Huisman, 2013; Kalimo et al., 2012). Increasingly, researchers and policymakers have suggested increasing resource efficiency through standards under the Ecodesign Directive (Bundgaard et al., 2017; Dalhammar, 2014, 2016; Dalhammar, Machacek, Bundgaard, Zacho, & Remmen, 2014). One significant leverage point for this is integrating resource efficiency aspects, including lifetime aspects, in the preparatory studies and in the Methodology for Ecodesign of Energy-related Products (MEErP) (Bundgaard et al., 2017; Mudgal et al., 2013). These, in turn, underpin the formulation of requirements in the Ecodesign implementing regulations.

The use stage has dominated the impact of lighting products, yet interestingly the EU ecodesign requirements for lighting products have minimum lifetime and functionality requirements rather than solely focussing on energy efficiency. These are mainly to ensure a minimum functional quality, but they could also serve to promote longer lifetime requirements pursuant to resource efficiency objectives. Yet it is uncertain if more ambitious minimum lifetime requirements are desirable and appropriate considering that LED products continue to develop with regard to energy efficiency and different materials. To understand the implications of longer product lifetimes, the consumer perspective (applying the least life cycle cost

principle) and environmental perspective (addressed through LCAs) should be considered, aligning with the aims of the Ecodesign Directive.

Previous LCAs of lighting products have conducted sensitivity analyses concluding that longer lifetimes are always desirable (Casamayor, Su, & Ren, 2017; Casamayor, Su, & Sarshar, 2015; Tähkämö, 2013). However, these studies did not consider the implications of lighting product development; for example, that shorter product lifetimes could enable replacement with more efficient products. This suggests that there could be trade-offs in policies promoting longer lifetimes (i.e. lock-in of less efficient products). These factors and potential trade-offs have been considered in the context of other energy-using products, such as white goods (Ardente & Mathieux, 2014; Bakker et al., 2014; Bobba, Ardente, & Mathieux, 2016; Devoldere, Willems, Duflou, & Dewulf, 2008). Prior research has also noted the need to consider the dynamics of changing electricity context driven by climate policies when considering prolonging product lifetimes (in this case, through reuse of white goods – see O’Connell, Hickey, & Fitzpatrick, 2013). Cleaner electricity sources result in lower climate impacts for the use-phase, compared to the same product used in the context of electricity with a high input from fossil fuel sources. At the same time, they may have higher impacts in non-climate impact categories. Thus, there are several dynamic factors needing consideration when assessing trade-offs, these can be product-specific, and they remain largely unexplored in the case of lighting products.

## 1.4 Research aim and questions

The aim of this research is to address gaps in knowledge about the performance of existing EPR and Ecodesign policies for lighting products and the potential improvements in relation to the EU’s CE objectives. Through the case of lighting products, it aims to understand challenges in product policy for a CE and how these might be addressed to improve policies for closing and slowing materials loops.

**Research Objective 1:** Contribute to understanding of the performance and potential of policies for closing material loops for lighting products.

This part of the research aimed to evaluate EPR policies implemented pursuant to the WEEE Directive to identify best practices and remaining challenges. The research aimed to be policy relevant; therefore, the implications of the findings for future policy were also considered. This research objective entailed two research questions.

*RQ1: How have EPR systems for lighting products in the Nordic countries been environmentally effective?*

To address this RQ, the Nordic countries were taken as in-depth case studies to evaluate the implemented EPR policies and propose the factors contributing to best practice (Paper 1).

*RQ2: What influences the potential for recycling of rare earth elements from lighting products?*

To address this RQ, the case of recycling rare earth elements, in particular Europium, Yttrium, and Terbium was analysed. Simple material flow modelling was used to estimate the potential for recycling these elements from lighting products and combined with a case study of Solvay-Rhodia to identify the drivers and barriers for such recycling practices (Paper II). Additional drivers and barriers for recycling of REE from lighting products was examined through risk/value concepts integrated into a global value chain governance framework (Paper III).

**Research Objective 2:** Contribute to understanding of the performance and potential of policies for slowing material loops for lighting products.

This part of the research aimed to evaluate the existing and potential role of ecodesign requirements in promoting design strategies for slowing material loops for lighting products. The relevance and adequacy of the current requirements were examined in light of whether promotion of longer lifetimes is desirable considering the dynamics of both technology and contextual developments for LED lamps, which might give rise to potential trade-offs.

*RQ3: Are longer lifetimes desirable for LED lamps, considering possible trade-offs?*

This RQ was addressed through literature review of life cycle cost (LCC) models and life cycle assessment (LCA). LCC modelling was conducted for the specific context of the Swedish LED retrofit lamp market (Paper IV). This question for LED lamps was also tested with existing LCA data for LED lamps, considering additional factors such as product development, replacement scenarios, and electricity mixes (Paper V).

## 1.5 Scope and limitations

This research focuses on lighting products, a case which exemplifies many of the challenges in product policy. Focus on this case serves to understand product group-specific challenges. The geographical focus of the research is mainly limited to European countries, which reflects the scope of research projects within which the PhD work was based. However, relevant examples from the U.S., China and other countries were also examined through involvement in events such as the Global Efficient Lighting Forum, conferences, and through relevant cases reviewed in the literature.

The EU WEEE Directive and Ecodesign Directive are the specific policies examined for their contribution to closing and slowing material loops. It is noted that the RoHS Directive is also part of the EU's EPR policy package (Van Rossem et al., 2006). Its effect on lighting product design has been examined in prior research (see e.g. Gottberg, Morris, Pollard, Mark-Herbert, & Cook, 2006) and it was not a focus of this research explicitly. In the context of slowing loops, the research focussed specifically on the strategy of longer lifetimes for lighting products, as opposed to other strategies such as modularity and repairability.

This research takes a broad transdisciplinary approach with research questions grounded in addressing topical policy issues in the context of 2-3 year research projects. Answering the policy questions necessitated different research methods for the different questions/topics, which can be a trade-off with a more consistent and linear research design. The research approach is reflected upon in the concluding discussion (Section 5.3). The issues chosen to research are exemplified in the case of lighting products but are relevant to many products groups. Therefore, while the narrow focus on lighting products might limit some of the generalisability of the research in terms of the specific outcomes, there are still findings that pertain more generally to the selected policy issues. The generalisability of the research is revisited in the concluding discussion (Section 5).

## 1.6 Audience

As the starting point for the research questions in this thesis is the policymaking arena, a key audience are those involved in policymaking. This includes not just decision-makers, but those involved in advising and supporting the policymaking process. Much of the research was conducted with some sort of collaboration with practitioners, with the intention that it would be useful for any organisation working with CE and product policy. As a work generated from academia, the work is also intended to contribute to and continue the academic discussions around policy

evaluation, CE product policy issues, life cycle approaches in policy, and complex questions of value.

## 1.7 Thesis structure

Chapter 2 presents an overview of relevant concepts and theories underpinning the research. Chapter 3 presents a brief summary of the research design and methods. Further details of the design and methods can also be found in the appended papers. Chapter 4 summarises the main approaches and findings of the published papers. Lastly, Chapter 5 revisits the research questions, summarises the contributions of the research and discusses the implications of the conclusions for CE policies more generally, and for future research.



# 2 Theoretical and Conceptual Foundations

This thesis positions itself broadly within the field of Circular Economy and Policy Evaluation research and seeks to draw from and contribute to concepts of EPR and life cycle thinking. This chapter further introduces the key concepts underpinning the research.

## 2.1 Circular Economy

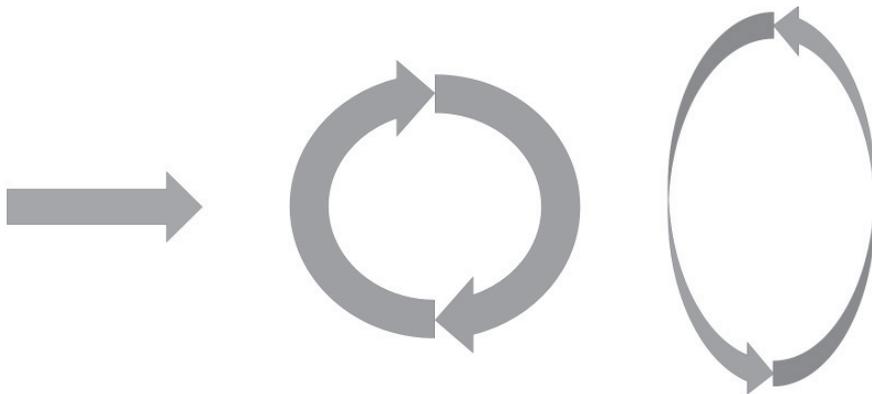
The EU Commission defines a CE as an economy “where the value of products, materials and resources is maintained in the economy for as long as possible, and the generation of waste minimised.” (EU Commission, 2015). However, while this definition presents a goal, CE as a defined concept remains contested (Korhonen, Nuur, Feldmann, & Birkie, 2018). Researchers argue that CE remains a rather ambiguous term (Korhonen, Honkasalo, & Seppälä, 2018), rooted in several disciplinary traditions (see e.g. Bruel, Kronenberg, Troussier, & Guillaume, 2019; Ghisellini, Cialani, & Ulgiati, 2016) and subject to different framings (Blomsma & Brennan, 2017; Geissdoerfer, Savaget, Bocken, & Hultink, 2017; Kirchherr, Reike, & Hekkert, 2017).

A common framing of the problem for CE is as a waste management problem (Ghisellini et al., 2016); i.e. there is a need to manage waste better so that it retains value as a resource, material loops are closed, and contamination is avoided. Another framing of the problem is as a resource problem (Geissdoerfer et al., 2017); i.e. there is a need to use resources more efficiently and restore them back into the industrial system for a productive and sustainable economy. Still another framing is as a consumption issue; i.e. there is a need to slow and limit consumption in order to use fewer resources, and thereby lessen environmental impact, for a sustainable economy (Bocken & Short, 2020; Bocken, Ritala, & Huotari, 2017). Still other framings incorporate different dimensions of sustainability including social aspects and intergenerational equity, though these tend to be under-represented in the academic literature (Kirchherr et al., 2017).

The definition of CE taken as a point of departure in this thesis is:

*a regenerative system in which resource input and waste, emission, and energy leakage are minimised by slowing, closing, and narrowing material and energy loops. This can be achieved through long-lasting design, maintenance, repair, reuse, remanufacturing, refurbishing, and recycling (Geissdoerfer et al., 2017, p. 759).*

The concepts of life cycle thinking and product life cycles are fundamental to the concept of CE; however, CE provides additional emphasis aligned with the waste hierarchy (Charter, 2018; Kirchherr et al., 2017). Therefore, while the desired shape of material flows in the system is circular, in order to lead to resource efficiency on a large scale, the flows also need to be distinguished by their speed and size (as shown in Figure 2). Bocken, Pauw, Bakker, & Grinten (2016) conceptualise strategies for the circular economy as not just closing material loops (i.e. circularity), but also narrowing and slowing materials loops (i.e. the size and speed of the flows in a loop). Slowing loops is achieved through reuse of components and products, which in turn is achieved through an extension of the utilisation period (e.g. longer life, repair, maintenance, etc.) (Cooper, 2005; McDonough & Braungart, 2010; Stahel, 1994, 2010).



**Figure 2.** Linear versus closed loop versus slow and narrow closed loop. Based on Bocken et al. (2016).

There has long been a focus on narrowing resource flows through more efficient processes as this strategy is often a win/win strategy (i.e. firms save money by reducing inefficient resource use) (e.g. cleaner production) (Khalili, Duecker, Ashton, & Chavez, 2015). While the strategy is relevant for a CE, it also works in a linear system. Within a CE system, if the speed of consumption is not addressed as well, simply narrowing loops, but increasing the rate, results in rebound that undermines the environmental aims of a CE (Bocken et al., 2016). Closing and

slowing loops are the strategies of the CE policy interventions explored in this thesis (see the conceptual framework in Section 3.1).

The concept of CE and its development is not without critique. Kirchherr et al. (2017) found that the aim to reduce (as opposed to reuse and recycle) was less emphasised in CE literature, and that less than 20% of academic literature on the topic of CE explicitly mentioned consumption (Kirchherr et al., 2017). Despite increased collection and recycling rates in the EU, there is in fact currently very little circularity or closing of loops in practice and a need for considering longer product lifetimes more seriously (Haas et al., 2015). Zink and Geyer (2017) argue that a tendency to focus only on closing material loops does not deal with rebound effects of increased consumption.

There are also limits to CE in terms of thermodynamics (i.e. there is always some level of material loss and downcycling), path dependencies, trade-offs and questions of governance which previous research argues should be a focus of continued research (Korhonen, Honkasalo, et al., 2018). The limits of business models, which still focus on profitability, to assure CE achieves its environmental aims has been noted (and the often missing consideration of markets) (Whalen & Whalen, 2018). There is a clear need for a mix of government policies to steer towards a CE and ensure environmental effectiveness (Milios, 2017).

The research in this thesis seeks to contribute to these larger questions about the effectiveness of CE policies and governance in both policy-driven recycling systems and global value chains (emphasised in Papers I-III). It also considers potential trade-offs between different sustainability objectives (emphasised in Papers IV-V).

## 2.2 Extended producer responsibility

The principle of EPR is defined as “a policy principle to promote total life cycle environmental improvements of product systems by extending the responsibilities of the manufacturer of the product to various parts of the entire life cycle of the product, and especially to take-back, recycling and final disposal of the product” (Lindhqvist, 2000, p. 154). Lindhqvist (2000) further illustrates that EPR entails different types of responsibilities: liability, physical, financial, and provision of information (i.e. informative).

The Organisation for Economic Cooperation and Development (OECD) defines EPR more narrowly as “an environmental policy approach in which a producer’s responsibility for a product is extended to the post-consumer stage of a product’s life cycle”, which is characterised by:

- 1. the shifting of responsibility (physically and/or economically; fully or partially) upstream toward the producer and away from municipalities;*
  - 2. the provision of incentives to producers to take into account environmental considerations when designing their products.*
- (OECD, 2016)

This definition illustrates the range of issues from ecodesign to waste management that can be integral to EPR policies; involvement of national states, private producers, and local municipal actors. There is a range of responsibilities that can be allocated. These responsibilities can be distributed differently amongst actors in different EPR interventions, but it is argued that placing financial responsibility on the producer is an important mechanism for creating incentives to minimise costs and environmental impacts, a core component of EPR (Kalimo et al., 2012).

Whether there is potential for ecodesign incentives and whether EPR legislation should aim for this is a matter of long-standing debate. Some researchers question whether EPR can effectively provide incentives for ecodesign (and whether it should even aim to do so, see e.g. Huisman, 2013) while others argue that such incentives for ecodesign are at the core of EPR (Kalimo et al., 2012; Tojo, 2004).

Often policies identified as EPR policies have aligned with the OECD definition, which stresses the extension of the responsibility to the post-consumer stage of the life cycle. However, if taking Lindhqvist's (2000) original broader definition of EPR, policies aimed at producer responsibility at any stage of the life cycle could be considered EPR policies. This definition implies a much more diverse range of EPR policies, depending not on the specific aims for end-of-life (EoL) management of products, but rather whether or not they focus on the producers and their responsibilities for the environmental impacts of their products. In this interpretation, promotion of ecodesign considering life cycle impacts through design standards (e.g. design without toxic chemicals at any stage of the life cycle, design for durability, energy-efficiency, and recyclability) can be seen as a complement, if not also part, of producer responsibilities.

Kalimo et al. (2015) examined the evolution of EPR as a concept first focussed primarily on the role and responsibility of producers, but which has become a system that is reliant on a much larger group of private and public actors. The authors note that the complexity of global value chains and EPR systems in practice means that allocating roles and responsibilities for optimal effectiveness and efficiency requires consideration of dynamics of fluctuating costs and value and how these should be shared between a wide group of stakeholders. However, the argument for placing substantial responsibility on the producer<sup>12</sup> remains that they have the most ability

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<sup>12</sup> "Producer" in EPR policies, such as the WEEE Directive, can refer to the manufacturer, importer, reseller, or distance seller of the products.

to implement design changes (Kalimo et al., 2012; Lindhqvist, 2000; Sander et al., 2007; Tojo, 2004; Van Rossem, 2008).

However, failure to fully implement all aspects of the EPR principle has resulted in weak ecodesign incentives in legislation based on this principle (Sander et al., 2007; Van Rossem, 2008). This has led to the suggestion that responsibility for ecodesign can be more explicitly incentivised through the mandatory design requirements in the EU Ecodesign Directive and its implementing regulations (Finnveden et al., 2013). It is argued that this ensures that at least some socioeconomic benefits (e.g. for recyclers) and environmental benefits of ecodesigned products are realised (Dalhammar, 2016). In this way, the EPR policies and the Ecodesign Directive are complementary in the development of ecodesign incentives.

## 2.3 Policy evaluation

Vedung (2009) defines policy evaluation as the “careful retrospective assessment of the merit, worth, and value of administration, output, and outcome of government interventions, which is intended to play a role in future, practical action situations” (p. 3). In addition to *ex post*, or retrospective analysis, it has been argued that *ex ante*, or prospective, evaluation is very important in the field of environmental policy (Mickwitz, 2003). The EU’s Environmental Action Programme to 2020<sup>13</sup> stresses the need for both *ex ante* and *ex post* policy evaluations. It states that improving environmental integration and policy coherence requires “carrying out *ex-ante* assessments of the environmental, social and economic impacts of policy initiatives at appropriate Union and Member State level to ensure their coherence and effectiveness” and “using *ex-post* evaluation information relating to experience with implementation of the environment *acquis* in order to improve its consistency and coherence” (pp. 195-196).

This thesis focusses mainly on the *ex post* and *ex ante* environmental effectiveness of the policies researched. It is grounded in a theory-based evaluation approach, with an emphasis on social science research and theories for deeper understanding of policies; e.g. how they are intended to work, how they are implemented and work in practice, and identifying gaps in understanding how policies can better achieve their goals.

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<sup>13</sup> Decision No 1386/2013/EU of the European Parliament and of the Council of 20 November 2013 on a General Union Environment Action Programme to 2020 ‘Living well, within the limits of our planet’

### 2.3.1 Theory-based policy evaluation

To distinguish theory-based evaluation from other evaluation approaches, Coryn et al. propose five principles for theory-based evaluations: 1) formulation of a programme theory; 2) evaluation questions formulated around a programme theory; 3) planning, design and execution of the evaluation is guided by the programme theory; 4) constructs postulated in the programme theory should be measured; 5) theory-based evaluators should identify breakdowns (of the theory), side effects, effectiveness and explanations of cause and effect relationships (Coryn, Noakes, Westine, & Schröter, 2011, p. 205). While these principles might be a starting point for defining theory-based policy evaluation, it is important to note that “theory-based evaluation is not a homogenous school of thought” (Astbury, 2011, p. 17).

A key part of theory-based evaluation is the intervention (or programme) theory, which describes how an intervention or programme is theoretically supposed to function (Coryn et al., 2011), compared to how it actually functions in reality (Bickman, 1987). Chen (1990) defined theory as “a set of interrelated assumptions, principles, and/or propositions to explain or guide social actions” (p. 40). This involves construction of an intervention theory by breaking down the policy to its inherent assumptions about actors and actions that lead to immediate, intermediate and long-term outcomes (Vedung, 2009). Vaessen & Leeuw (2010) also note that intervention, or programme, theories relate to the assumptions of the policy and stakeholders addressed by a policy and are distinct from social science theories, which are generated by social science and can serve to enrich intervention theories. It is important to note that for any programme there can be several intervention theories, depending on the perceptions of stakeholders involved in the construction (Weiss, 2000).

Evaluation scholars, such as Chen & Rossi (1992), argue that without a good understanding of why programmes work as they do, there is not enough information for programme improvement or to inform decision-making. When the theory is mapped in terms of what should happen, how the programme works can be more deeply explained and empirical checks can be made that the actions are indeed occurring. The causal mechanisms, found in the black-boxed arrows in an intervention theory, can be unpacked and examined (Astbury & Leeuw, 2010; Pawson & Tilley, 1997). Leaving causal mechanisms unexamined can result in too simplified (and even wrong) intervention theories (Weiss, 1997). Unpacking the black boxes of causal mechanisms involves using theories from social science to explain the “gaps” or arrows in the intervention theories (Vaessen & Leeuw, 2010).

Constructing an intervention theory involves mapping a sequence of events and changes that should occur in response to the policy intervention and leading to its intended outcomes (Vedung, 2009). Despite being described “chains”, representations of intervention theories can take many forms, many of them non-

linear (Coryn et al., 2011). Using intervention theory can also help to focus the scale (or multiple levels) of the policy (for example, from national to local) and identify stakeholders addressed by the evaluation (Mickwitz, 2003). Stakeholders in an intervention are active in shaping interventions through their interpretations, responses, negotiations, exchanges of information, and learning over time. Pawson et al. (2004) write that the “fact that policy is delivered through active interventions to active participants has profound implications for research method” (p.5). This makes knowledge and understanding of stakeholders’ reasoning and actions integral to understanding a programme’s outcomes (Pawson, Greenhalgh, Harvey, & Walshe, 2004).

However, there has also been criticism of theory-based evaluation approaches, most notably by Scriven (1998) who views the primary role of the evaluator as determining whether a programme works, not to explain how it works. Coryn et al. (2011) further note that just mapping out the intervention theory does not mean that it will be used in a meaningful way to drive the evaluation or even determine the results. They conclude that this can make the approach unnecessary and that some evaluations purported to be theory-based might in fact have reached the same conclusions using a different approach.

#### *2.3.1.1 Stakeholder (theory-based) evaluation*

Vedung (2009) proposes stakeholder evaluation as another form of effectiveness evaluation, in which the needs and concerns of stakeholders drive the evaluation process. The evaluation process can be driven by stakeholders or facilitated by the evaluator (Rodríguez-Campos, 2012; Vedung, 2009). A stakeholder approach to theory-based evaluation elaborates upon the basic intervention theory with the perspectives of key stakeholders in an iterative process of intervention theory construction (Pawson & Tilley, 1997). Identifying the primary stakeholders crucial to the intervention implementation highlights sources for empirical checks. Intervention theories from the perspective of different key stakeholders can be constructed to identify similarities, differences, and disagreements (Hansen & Vedung, 2010). This was the approach adopted in Paper I and described in greater depth in Chapter 3.

The advantages of stakeholder evaluations are that they reveal more in-depth knowledge of the intervention held by stakeholders and the process of including stakeholders can also increase the utilisation of the evaluation (Patton, 2008; Rodríguez-Campos, 2012). While the stakeholder evaluation approach can be vague in terms of who are the stakeholders and runs the risk of being biased in terms of stronger stakeholders or being hijacked by political ends in order to increase the utility of the evaluation (Vedung, 2009, p. 75), it is argued that the potential benefits outweigh the potential difficulties (Rodríguez-Campos, 2012).

### 2.3.1.2 *Realist evaluation*

A more specific approach to theory-based evaluation is “realist evaluation”, in which “the nature of programme theory is specified in realist terms” and with its origins in realist and critical realist movements in philosophy of science (Astbury, 2011, p. 14). The realist evaluation approach has been argued to be particularly appropriate for complex policy interventions, as they examine the context in which an intervention operates (Pawson et al., 2004). This can range from the individual capacities of key stakeholders to the wider societal structural elements supporting the intervention (Pawson et al., 2004). This focus on the context enables learning about how the context influences the policy, which in turn can help to improve policies in relation to their context and to better understand transferability of policies (Astbury, 2011).

The goal of realist evaluation is to explain, and this is done by unpacking mechanisms of why and how complex programmes work (Pawson et al., 2004). It can explain why interventions do not work; for example, Chen, Wang, & Lin (1997) found a key mechanism in an intervention theory (the residents’ reaction to the smell of garbage) was erroneous, undermining the effectiveness. Mechanisms have been defined as “underlying entities, process or structures which operate in particular contexts to generate outcomes of interest” (Astbury & Leeuw, 2010, p. 368). Mechanisms are sometimes equated with variables, but they are distinct from variables in that they try to explain the relationship between variables rather than empirically measure a mechanism (Astbury & Leeuw, 2010). Thus, mechanisms can also be hidden and unobservable, which can pose a challenge for evaluators in first identifying relevant mechanisms for testing (Astbury, 2011). Even when observed, in intervention theories they may only appear as “unexplained causal arrows” (Astbury & Leeuw, 2010, p. 367). Testing mechanisms involves testing whether the mechanism, operating in a certain context, can lead to the desired outcome (Astbury, 2011; Astbury & Leeuw, 2010).

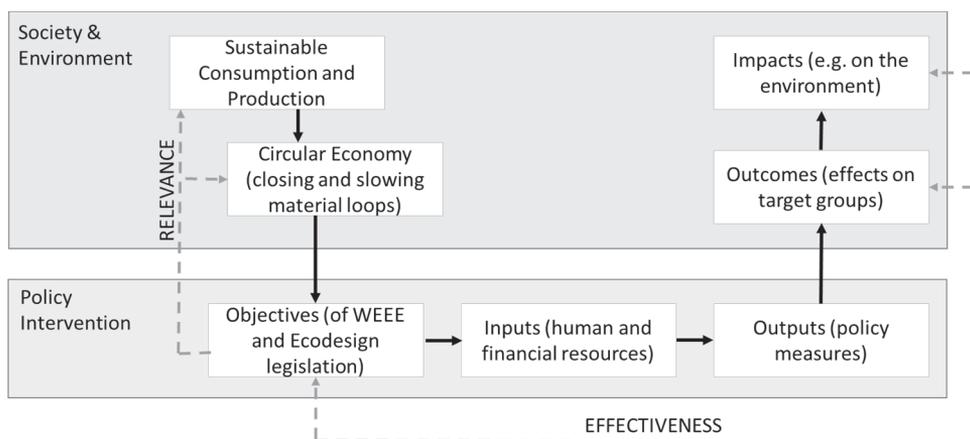
Pawson et al. (2004) point out that attempting to deal with examining complex systems necessarily means there will be limitations to evaluations that should be considered, e.g., limits on how much the reviewer can really cover, the nature and quality of the information that can be retrieved, and the extent of the recommendations that can be made. Evaluations can also only focus on one aspect, element, or chain of the intervention theory as “tailored theory-based evaluations” as opposed to comprehensive theory-based evaluations (Coryn et al., 2011, p 203).

In this research, the WEEE Directive was evaluated with a tailored theory-based evaluation that focussed on the case of lighting and specific contexts of Nordic countries. In addition, the underlying mechanism of closing material loops was further examined in the case of REE to understand the role of the WEEE policy in closing loops for such materials and enhancing its potential (Paper II). This, in turn,

led to the need to further understand the mechanisms for closing material loops for REE, which entailed use of current social science theories of risk and value constructions (further elaborated in Chapter 3 and Paper III).

### 2.3.2 Environmental Effectiveness

The focus of the research in this thesis was to contribute towards understanding the environmental effectiveness of current policies related to the CE and ways to enhance their environmental effectiveness. Effectiveness evaluations focus on the substantive results of an intervention and can focus on aspects such as goals, impacts, system components, client concerns, or stakeholder concerns (Vedung, 2009). Figure 3 shows the link between the objective and results (interpreted as either outcomes or impacts depending on the focus of the evaluation) that are the basis of effectiveness evaluations. Relevance is also shown in Figure 3, though it is not explicitly addressed in the papers in this research. It is, however, an implicit part of the aim of this thesis to discuss the findings of the papers in relation to the broader societal (i.e. CE) aims of the policies.



**Figure 3.** Effectiveness evaluation relating outcomes and/or impacts to objectives. Adapted from Guedes Vaz, Martin, Wilkinson, & Newcombe (2001).

There are different ways of addressing effectiveness in evaluations. The following sections describe the approaches used in this thesis while specifically how they are applied in analysis is further elaborated in Chapter 3.

#### 2.3.2.1 Goals, results and side effects

In a goal-attainment evaluation approach, the goals of a programme are translated into measuring objectives and the extent to which these have been realised in practice is evaluated. The strengths of the goal-attainment model have been argued to be: 1) a democratic argument in that the goals have most often been generated by

a government body representing the interests of citizens and citizens have an interest in whether programmes really achieve the goals; 2) a research argument in that evaluations are normative but the programme goals can be interpreted from the legislation or policy process documents, thus giving some level of objectivity in deciding by what measure to evaluate the programme; 3) the method is relatively straight-forward to apply (Vedung, 2009, pp 41-43). However, goal-attainment models can be criticised if goals are vague, causation between the programme goals and the results is not established, or unintended effects of the policy are ignored (Vedung, 2009).

Besides goal-attainment, environmental policies can be evaluated for their results or effects (irrespective of the goals), both intended and unintended. This could include examining side effects, both anticipated and unanticipated (Vedung, 2009), as well as beneficial and detrimental (Mickwitz, 2003). Intervention theories can be a tool for examining different causal relationships (Mickwitz, 2003), which in turn strengthen the case for causation between a programme and its effects. However, it should be noted that establishing attribution and isolating programme effects remains challenging in practice (Ferraro, 2009; Sanderson, 2002). Still, an important part of ex ante evaluation is to understand and anticipate the likely effects (both direct and side) as much as possible (Mickwitz, 2003). The EU's Environment Action Programme to 2020 also emphasises the need for "addressing potential trade-offs in all policies in order to maximise synergies and avoid, reduce and, if possible, remedy unintended negative effects on the environment" (p. 196).

In interviewing stakeholders, Paper I took a broader look at the effects of the policy. Paper II looked at one of the effects in more detail. In contrast, Papers III and IV took an ex ante approach to understanding the trade-offs and potential consumer cost and environmental impacts of lifetime requirements for lamps.

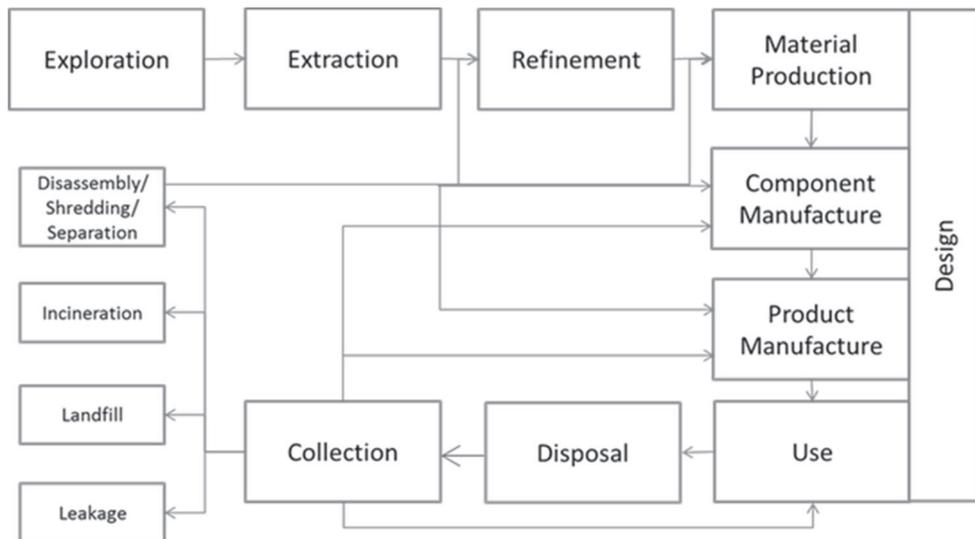
## 2.4 Life cycle concepts

Life cycle thinking helps conceptualise environmental problems as system-level issues by considering all of a product's environmental aspects from cradle to grave or cradle to cradle. Life cycle thinking is important when considering unintended consequences of improvements leading to shifting environmental impacts from one life cycle phase to another (Mont & Bleischwitz, 2007).

Life cycle thinking supports environmental policymaking, either as a qualitative or quantitative approach (Dalhammar, 2007). Application of life cycle thinking can range from conceptual high-level institutionalisation to the use of comprehensive tools like the LCA method that assesses the environmental impacts across all life cycle stages of a product or process. The result of the institutionalisation of life cycle

thinking was an increased focus on products, as sources of environmental impacts, and market actors for influencing and reducing environmental impacts (Heiskanen, 2002). Since then, comprehensive sustainability assessments have advocated for a triangulated life cycle approach incorporating environmental LCAs with life cycle costing (LCC – i.e. assessing costs over the life cycle) and social life cycle assessments (SLCAs – including social criteria across the life cycle), providing different perspectives on the various impacts of a product life cycle (Kloepffer, 2008; Mazijn, 2016).

A simplified life cycle of a product is shown in Figure 4. However, even in this simplified form the figure demonstrates that there is significant complexity in conceptualising life cycles. For example, the EoL management of a product involves choices and potential interactions with other life cycle phases; i.e. there are choices in whether and how EoL products are collected and put back into the life cycle as used products, components, or materials or whether they are burned for energy recovery or landfilled.



**Figure 4.** Simplified product life cycle

Turning raw materials into products involves numerous processing steps, or segments. Understanding the flows of materials from extraction to product to waste underlies several life cycle approaches. Different approaches focus on different aspects of the life cycle. Environmental impacts can be mapped and quantified with LCA. Costs can be quantified for the whole chain or by different actor perspectives, as in an LCC. Stakeholders and value can be examined through the lens of business management (e.g. supply chain management) or economic geography (e.g. as value-

adding activities and actor networks). Materials, activities, information and impacts throughout the product life cycle can be managed with life cycle management.

The research in this thesis utilised quantitative material flow analysis (Paper II), LCC (Paper IV) and LCA (Paper V). Particular socio-economic aspects are analysed qualitatively in the policy evaluation of the WEEE system (Paper I) and through economic geographical evaluations of the lighting product value chain, with a specific focus on REE (Paper II and III).

#### **2.4.1 Material flow analysis (MFA)**

MFA is a “systematic assessment of the flows and stocks of materials within a system defined in space and time.” (Brunner & Rechberger, 2016, p. 3). Such analysis makes stocks and flows visible, which in turn can highlight which part of the system is inefficient and where it can be optimised. MFA is used as a tool to support decision-making in the fields of resource, waste and environmental management to more efficiently manage resources. However, it should be noted that material flows in anthropogenic systems must also consider socioeconomic and other issues in considering responsible management and in order to fully interpret MFA results, i.e. understand *why* materials flow (Brunner & Rechberger, 2016). While MFA underpins the modelling of secondary and primary REE in Paper II, this is combined with qualitative examination of the value chain and policy context to understand how the flows could be optimised.

#### **2.4.2 Global value chains and production networks**

The supply chain management approach emerged out of business administration, economics, and information technology and is focused on understanding and managing firms across supply chains. This approach can consider microeconomic issues like competition, strategy, supply sources, transportation and logistics, and the role of government, multiple actors and sustainability (Bush, Oosterveer, Bailey, & Mol, 2015). Economic considerations are involved along the life cycle of a product and these are conceptualised as the value chain.

Related approaches are macro-scale, including the global commodity chain (GCC) approach that examines the empirics of a producer or buyer-driven commodity chain but is less engaged in questions of why certain governance structures occur (see e.g. Gereffi, Humphrey, & Sturgeon, 2005). The global value chain (GVC) approach has developed from the GCC approach, but considers value-added activities, transaction costs, complexity, and exchange of information (Bush et al., 2015). GVC not only examines linkages between segments of a value chain but also the role of institutions, including governments and social actors, and how power is distributed

among actors in the chain (Sturgeon, Van Biesebroeck, & Gereffi, 2008). Similarly, a global production network (GPN) approach examines “the socially and territorially embedded nature of production and consumption” (Bush et al., 2015, p. 3) and similarly explores the role of a wide range of actors and distinct contexts. Often the GCC/GVC/GPN approaches can be seen as a broad family of related approaches, more alike than different (Fridell, 2018).

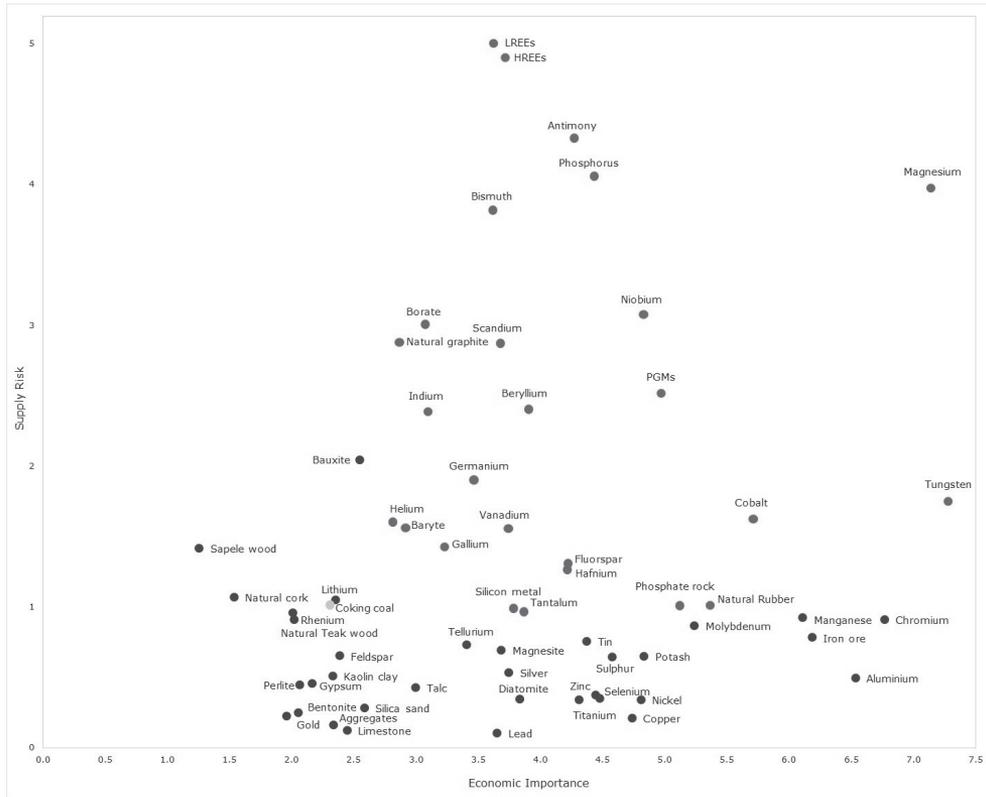
Both GVC and GPN approaches can be seen as adding a layer of real-life complexity to the GCC framework. These approaches consider how the chain is governed, contextual factors, value-adding activities, interaction amongst various stakeholders and the dynamics of networks to examine not only *how and where* materials and products flow, but also *why*. While governments and policy are key aspects of GVCs and GPNs, they are not often explicitly examined in research (Neilson, Pritchard, & Yeung, 2014).

Value in the economic geography field literature, and specifically in GVC literature, can be conceptualised as an “in-the-making rather than an intrinsic property of things” (Lepawsky & Billah, 2011, p. 126). Understanding transactions where data and information is exchanged is important to understanding the resulting prices and value in the market place (i.e. *why* materials and products flow). Additionally, in such transaction there is a social construction of risk, which relates to the “object of value” (see Boholm & Corvellec, 2011). Objects of value can have a wider range of values than just economic (Allwood, Ashby, Gutowski, & Worrell, 2011) and these values can be perceived differently by different stakeholders (Bocken et al., 2015; Bocken, Short, Rana, & Evans, 2013).

Risks in supply chains for raw materials, i.e. CRMs, has also been highlighted as part of the EU Circular Economy Action Plan (EU Commission, 2015). At the EU level, such risks to supply disruption and the importance of raw materials to end uses important for the EU economy are assessed by an expert stakeholder group and thus can be considered stakeholder-derived constructions of risk and value.<sup>14</sup> Based on the EU expert panel assessments, a list of critical raw materials (CRMs) is derived (the EU list of 27 CRMs are shown in red in Figure 5). The EU Commission’s report on Critical Raw Materials for the European Union (EU Commission, 2014), considers REE as having the highest supply risk among various CRMs. REE received increasing attention after 2010 with rising prices and concern about supply restrictions from China, where over 90% of production takes place (Binnemans et al., 2013).

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<sup>14</sup> It should be noted that this process is not without politicisation and bias towards technical and natural science-based solutions, see Machacek (2017).



**Figure 5.** Materials assessed for criticality by the EU Commission. Source: European Commission's Communication "On the 2017 list of Critical Raw Materials for the EU", COM(2017)490.

Similar processes of assessment of criticality are carried out by other governments (e.g. the U.S. maintains a list of CRMs), and by companies (e.g. Fairphone and Apple have made profiles for many of the materials in their phones to assess criticality and other risks – see e.g. Apple, 2018, p. 21; Fairphone, 2017). Lighting producers similarly recognised the criticality of REE used in phosphors to create soft white light and employed several strategies including developing recycling techniques (Otto & Wojtalewicz-Kasprzak, 2012), reducing the amount of rare earths in the design (Osram, 2017), and developing strategic partnerships to better secure supply (Fürst, 2011). However, unlike governments, manufacturers may have a choice to deal with critical materials by strategically moving production processes (i.e. to China for lighting producers).

The beginning of this research in 2014 coincided with high interest in critical material strategies and recycling processes. Many studies in the last decade worked to characterise the flow of these materials into products and it was known that electronics were a significant application. Only 7% of REE are estimated to be used

in lighting (in the phosphors), but these phosphors represent 32% of the economic value in the REE market (Binnemans et al., 2013; Schüller et al., 2011). Despite this value, there is very little recycling of these materials (less than 1%) (Graedel et al., 2011). Though there is demonstrated technical feasibility to recycling REE, it was noted that the small amounts of REE and dispersed nature of the products were significant barriers to closing the loop for these materials (Binnemans & Jones, 2014; Binnemans et al., 2013).

### **2.4.3 LCA and LCC**

Life cycle approaches are intended to support decision-making in industry, governmental, or non-governmental organisations, including public policy (Seidel, 2016). Life cycle assessment (LCA) can play a key role in the legislative process both ex ante in providing information about the potential environmental outcomes and ex post as a comparative tool to measure environmental effectiveness (Reed, 2012). Both the WEEE Directive and Ecodesign Directive take a life cycle approach. The WEEE Directive has a focus on the EoL stage but also an explicit aim to influence upstream product designs. The Ecodesign Directive explicitly uses LCA and LCC methods as part of its MEERp analysis to determine prioritisation of impacts and where the requirements should be set (Siderius & Nakagami, 2013).

LCA is a quantitative tool for demonstrating how and where in the product life cycle environmental impacts occur. Environmental impacts are typically characterised as climate impacts, toxicity impacts, resource depletion impacts, etc. It is important to note that while LCA is a scientific approach, modelling of complex product systems necessitates making assumptions and setting limitations when conducting studies. Thus, being transparent about the goal, scope and methods is part of the standards advanced by the International Organisation for Standardisation (ISO) (14040 series). Traditionally LCAs have been focussed primarily on quantitative assessment of environmental impacts, but more recently social LCA methods have been developed which consider socio-economic impacts/indicators for diverse stakeholders including workers, local communities, value chain actors and wider society (Benoît et al., 2010; Sureau, Mazijn, Garrido, & Achten, 2018).

Economic impacts of a product life cycle can be assessed using a life cycle cost (LCC) analysis. This is a quantitative assessment for considering total costs compiled from all the life cycle stages. The exact method for calculating LCC is influenced by what perspective is taken (i.e. costs for whom?). LCC can be considered from a business perspective (Kloepffer, 2008) or consumer perspective (e.g. the costs to consumers for ownership of products from purchase to disposal). LCC of energy-using products from the consumer perspective underpins ecodesign requirements in the EU (Annex II).



# 3 Research Design and Methodology

This chapter introduces the broad foundations of the research approach (Section 3.1) and how concepts from the previous chapter have been applied in the analysis (Section 3.2). The data collection methods are described in more detail (Section 3.3), as are the methods that increased the validity and transparency of the research (Section 3.4).

## 3.1 Research approach

Policy research is the overarching research approach for this thesis. Policy research is defined “as the process of conducting research on, or analysis of, a fundamental social problem in order to provide policymakers with pragmatic, action-oriented recommendations for alleviating the problem” (Majchrzak, 1984, p. 3).

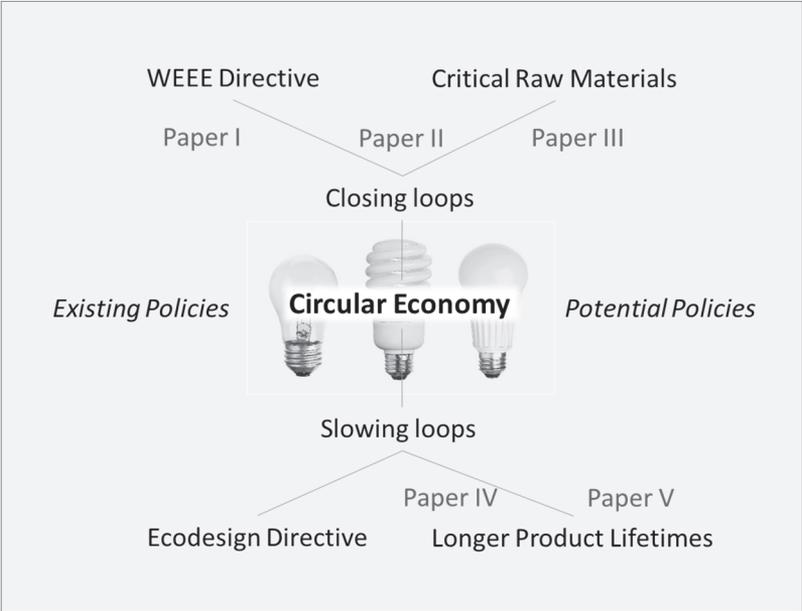


Figure 6. An overview of the conceptual framework for the papers in this thesis.

Policy research is “multidimensional in focus; uses an empirico-inductive research orientation; incorporates the future in addition to the past; responds to study users; and explicitly incorporates values” (Majchrzak, 1984, p. 8). In this research, the focus was on closing and slowing material loops with policy. The research examined current and potential development of CE policies (WEEE Directive and Ecodesign Directive), considering key issues such as CRMs and product lifetimes. Figure 6 shows an overview of the papers in relation to the conceptual framework.

As shown in Table 1 papers in this research are very diverse in terms of policy interventions considered and contexts, reflecting the different types of questions posed by policy and objectives of the different research projects within which the research was taking place. It is argued that this is the nature of transdisciplinary research, which is further elaborated in Section 3.1.2. The different research questions necessitated different approaches for analysis and data collection and these are further detailed in Sections 3.2 and 3.3.

**Table 1.** Overview of paper and research approach

<b>Paper #</b>	<b>I</b>	<b>II</b>	<b>III</b>	<b>IV</b>	<b>V</b>
<b>CE Focus</b>	Closing loops – recycling products	Closing loops – recycling CRMs	Closing loops – recycling CRMs	Slowing loops – longer lifetimes	Slowing loops - longer lifetimes
<b>Context</b>	Nordic countries WEEE Directive	EU / GVC / Solvay- Rhodia case	EU / GVC	Sweden ecodesign requirements	Sweden, Norway, EU ecodesign Requirements
<b>Approach</b>	Qualitative	Qual/Quantitative	Qualitative	Quantitative	Quantitative
<b>Main data source</b>	Interviews, literature	Literature, market data, interviews	Interviews, literature	Online market data, literature	Ecoinvent database, literature
<b>Main method(s) for data analysis</b>	Stakeholder theory-based evaluation Case study	MFA Modelling Content Analysis GPN framework Case study	Content analysis GVC framework Risk/value framework	LCC modelling	LCA modelling

### 3.1.1 Scientific research positioning

This research is framed by critical realism, an epistemological framework that encompasses a wide range of ontologies, which has been argued as a necessity for interdisciplinary research and when researching complex sustainability issues (Bhaskar, Frank, Hoyer, Naess, & Parker, 2010). The starting ontological approach for critical realism is that the world is “structured, differentiated, stratified and changing” (Danermark, Ekstrom, Jakobsen, & Karlsson, 2001, p. 5). The critical realist approach is arguably rooted in the positivist view of a world that is independent of human consciousness. However, it also acknowledges in part some aspect of a subjectivist view of a world socially constructed, i.e. our view of the world is influenced by our own perspectives and limitations in understanding its true

nature. Thus, critical realism views science as a continuous endeavour to improve our understanding of a changing, multi-level world. Danermark et al. (2005) describe the particular emphasis critical realism puts on researching the causal mechanisms of events and reasoning to gain better understanding. The authors argue that this often necessitates the use of mixed methods. Indeed, a critical realist is agnostic towards choice of methods, as they should suit the question being examined (Sovacool, Axsen, & Sorrell, 2018). This is evident in the diverse choice of methods in this thesis research, which were driven by the questions examined.

In addition to the epistemological positioning, the research is largely framed by a sustainable development approach. Sovacool & Hess (2017) argue that sustainable development is a large-encompassing theory for “multiple normative criteria” which aims to balance two main goals: satisfying the needs of the present and satisfying the needs of the future. In this sense, the research and the researcher take a normative stance overall in seeking to advance sustainable development. The researcher always includes questions about harm to the environment with questions about benefits to society, and studies in sustainable development are necessarily interdisciplinary in nature and often constrained by the specific contexts and factors considered (Sovacool & Hess, 2017).

### **3.1.2 Inter- and transdisciplinarity**

The approach taken in this research is reflective of the interdisciplinary background of the researcher, but also the complex sustainability issues at the core of the research problem. An interdisciplinary or transdisciplinary approach is argued as an appropriate approach for sustainability issues as they tend to be complex, containing different causes and outcomes on multiple levels and transcending narrow disciplinary worldviews (Bhaskar et al., 2010; Høyer & Naess, 2008; Klein, 2017; Stock & Burton, 2011). Both interdisciplinary and transdisciplinary research put emphasis on problem solving, with a focus on societal or “real world” problems (Klein, 2017; Lang et al., 2012; Stock & Burton, 2011). The main difference is the level of integration between disciplinary perspectives and cooperation amongst different actors (Klein, 2017; Lattuca, 2001; Stock & Burton, 2011). A commonly highlighted characteristic of transdisciplinary research is that it works towards “co-producing solution-oriented and transferable knowledge through collaborative research” and “(re)integrating and applying the produced knowledge in both scientific and societal practice” (Lang et al., 2012).

Environmental product policy, as a sub-topic of sustainability, is a topic that is difficult to grasp without examining complexity and considering different disciplinary perspectives. In addition, the field of policy studies is argued to be transdisciplinary in that the inquiry is often problem-oriented, rather than methodology driven. Vogel, Cherney, & Lowham (2017) explain that “[t]his may

necessitate drawing on expertise beyond the analyst's personal knowledge and methodological training in order to improve outcomes. As a framework and approach, the problem orientation breaks analysts out of their disciplinary and/or preconceived notions of the problem under inquiry, in effect asking analysts to be transdisciplinary in their approach to understanding and addressing problems" (pp 5-6). This is true for this research, which was guided in the first instance by the inquiry and resulted in a diversity of the approaches utilised.

A transdisciplinary approach also includes the involvement of non-academic partners, which could take place throughout the research process, including in the communication and discussion of results (Sakao & Brambila-Macias, 2018; Stock & Burton, 2011). Discussions and collaboration with stakeholders outside academia during the research process can increase the reflexivity of the research (Popa, Guillermin, & Dedeurwaerdere, 2015). A possible pitfall of transdisciplinary research is an "epistemic drift" that can result from the influence of non-academic stakeholders on the research itself (Tranfield & Starkey, 1998).

Conducting such diverse research involved developing a variety of research skills across a range of disciplinary fields, including economic geography, engineering, business management, economics, and law, as well as working with experts in these fields. The collaboration developed with other academics mainly was driven by an interdisciplinary research question requiring diverse knowledge and skills of a research team (e.g. Paper II-III). Collaboration with non-academics mainly was driven by policy questions from non-academic actors requiring skills from academia to apply to concrete policy questions (e.g. Papers 1, IV, and V).

The researcher engaged with stakeholders at different points in the research, including interviews, participation in dialogues, workshops, and communication of results with many of the key stakeholders. Some of these were specifically organised as part the projects within which the research took place while others, e.g. the stakeholder consultation on green public procurement for street lighting and Nordic Council of Ministers workshop on development of a CRM strategy for the Nordic region, were independent of the research projects.

### **3.1.3 Case-based approach**

The overall case selection of lighting products for this thesis was based on the initial project scope and the fact that household lighting products are representative of issues that arise in other complex product groups. Lighting products are rapidly developing (like many consumer electronics, renewable energy technologies, and electric cars, etc.). Lighting products are small and dispersed in use, which presents challenges for collection and recycling (like batteries, mobile phones etc.). Lighting products contain critical raw materials (again, like many other electronics,

renewable energy technologies and electric cars). Consequently, although the unit of analysis differed for each paper, it was situated within the broader case of lighting products. For example, Paper II dives deep into the particular case of a company recycling REE from lighting products, whereas Paper III broadens this case slightly to include the other actors in the value chain of REE recycling from lighting products (and compares to other REE recycling cases). Flyvbjerg (2006) argues that a strength of case studies is that they provide access to richly detailed and “close to reality” knowledge (p. 223). The similarity of lighting products to other electronic product groups allows for some generalisation; however, like any single case, there are limits to the generalisability. This is revisited in Chapter 5.

## 3.2 Analytical approach

The concepts introduced in Chapter 2 of this thesis formed the framework for analysis of the research and are further elaborated in this section.

### 3.2.1 Case studies

Case study methodology was used as an explicit approach for analysis in some of the papers. It has been argued that case study approaches can support policy evaluations (Yin, 2009, p. 19), and this was the purpose of using it in Paper I. An initial review identified the Nordic countries as high-performing countries in regard to WEEE collection and recycling (Román, 2012; Ylä-Mella et al., 2014), which then became the case selection for Paper I. This case selection could be argued as indicative of a “most similar” case selection (George & Bennett, 2005) as these countries share similarities in many respects, but with key differences in the WEEE and lighting EPR systems that made for an interesting comparative study. The more similar the cases being compared are in the EPR system aspects, the more likely it is to be able to see any differences in effectiveness that could be attributed to other factors independent of the EPR systems.

However, the selection of “best performers” as cases is also indicative of a heuristic case study approach (i.e. focussing on outlier cases). In this way, the case selection was used to identify variables that contributed to effectiveness. The cases were used inductively to explore causal mechanisms and build explanatory theory about best performers (George & Bennett, 2005).

The heuristic approach for case selection was also implicit in the selection of Solvay-Rhodia and rare earth recycling in Paper II with the intention of exploring some of the key dynamics of the value chain for REE and their implications for closing loops for these materials. Solvay-Rhodia was the only case of REE recycling

on an industrial scale in Europe. Thus, understanding this unique case was key to understanding how to upscale the practice of REE recycling.

The Solvay-Rhodia case was revisited and expanded in Paper III to examine the transaction between recycling suppliers, a REE processor like Solvay-Rhodia (since then another pilot processor had emerged as well) and the market for recycled REE. In that approach, the case was compared to two other transactions along the REE value chain to hypothesise about the influence of governance structures and construction of risk and value on the decision to recycle.

### 3.2.2 Policy evaluation

Paper I evaluated the effectiveness of the WEEE Directive as transposed in the Nordic countries, with a focus on lighting products. Tojo (2004) outlined the broad goal of EPR policies, such as the WEEE Directive, to be the “total life cycle improvements of product systems”, which in turn is realised through 1) design for the environment, 2) improved waste management practice and 3) closing material loops. Goal attainment evaluation of these three goals was the overarching approach taken in this research, combined with a stakeholder evaluation to test and expand elements of Tojo’s original intervention theory analysis (shown in Figure 7) and develop a more generalisable explanatory theory of the mechanisms for goal attainment in EPR policies.

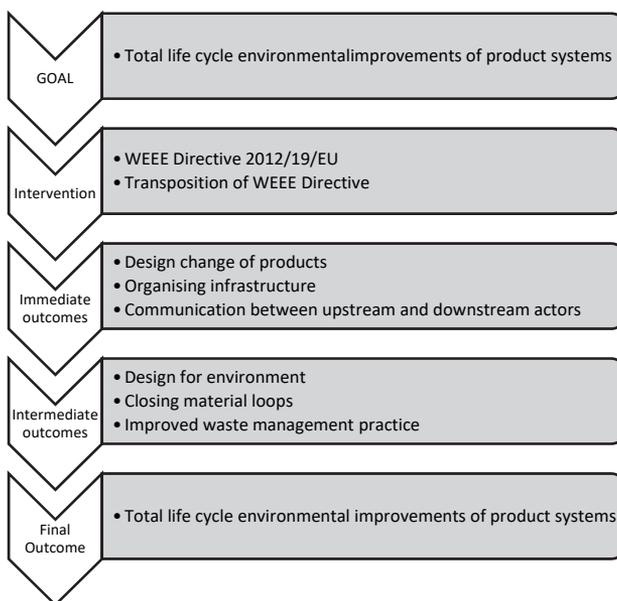


Figure 7. Simplified implementation chain for WEEE Directive based on Tojo, 2004.

Figure 8 shows a more elaborated intervention theory considering stakeholders. It shows how the intervention hinges upon several mechanisms that are important for realising the outcomes. Some of these mechanisms are more strongly linked between the WEEE Directive and the outcome, while others may be influenced by other policies and factors in addition to the WEEE Directive. In the intervention theory, there is a link between the shorter-term outcomes of collecting and recycling materials and the longer-term outcome of efficient use of valuable and critical materials. In a less developed intervention theory this might be expressed as an arrow with an underlying assumption that if materials are collected and treated in a recycling facility, they will then be used for primary production, which lead to improved resource efficiency.

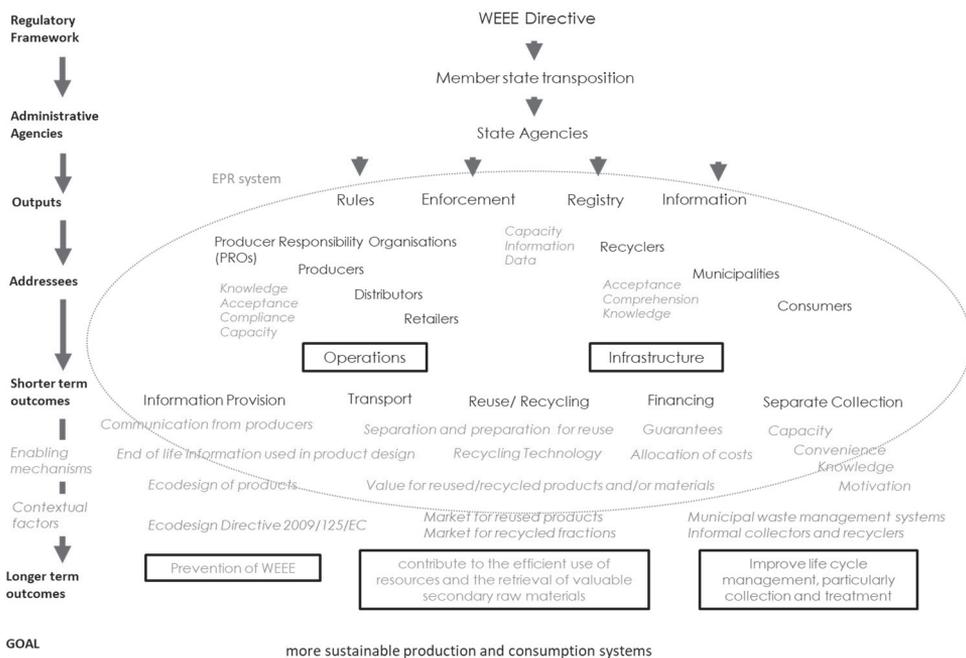


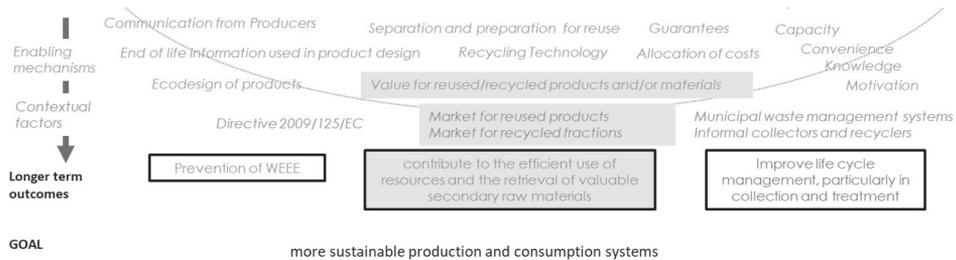
Figure 8. More elaborated intervention theory for the WEEE Directive (note: arrows removed for visual simplicity).

However, there are many more factors for this to happen in reality, including that technology is available, that the materials recovered have recognised value, and that there is a market for these materials. In addition, while EPR legislation like the WEEE Directive in principle promotes the waste hierarchy, it is still unclear how reuse is incentivised over recycling, so this can be contested (Kalimo et al., 2012).

Paper I of this research was scoped to consider more specifically the goals of closing material loops and retrieval of valuable materials for lighting products – i.e.

evaluation of specific goals, mechanisms, and issues. It also sought to elaborate on other factors that were perceived to contribute to the success of the intervention.

The issues identified regarding recycling of REE were subsequently investigated further in Papers II and III while referring back to the policy implications for the WEEE Directive and potential complementary policies. The focus of these papers, derived from the original intervention theory in Paper I, is highlighted in Figure 9.



**Figure 9.** Underlying mechanisms and context within the intervention theory that is examined in Papers II and III.

As Figure 9 demonstrates, the highlighted focus of the papers relates to the context, mechanism, and outcome conjectures described by realist evaluation scholars (Pawson, 2013; Pawson & Tilley, 1997). The simple intervention theory for the WEEE Directive is that requiring the recycling of materials will lead to efficient use of materials and retrieval of valuable secondary raw materials. There are several assumptions to make this link, particularly assumptions that:

- the recycling processes are able to retrieve the raw materials
- value is retained in the materials in the process
- there is an existing market in order to sell the materials with value
- the sale of secondary materials offsets the use of primary materials to achieve resource efficiency

There is also a question of how value is recognised by market (and other) actors. The value analysis in these papers is further elaborated in Section 3.2.3.

Papers IV and V focus on the economic and environmental impacts of the EU ecodesign requirements related to minimum lifetimes in order to examine the implications of longer lifetime requirements discussed by policymakers. Mirroring the life cycle approaches used to support ex ante policy assessment of the EU ecodesign requirements, the research for these papers use LCA and LCC and are further elaborated in Section 3.2.4.

### **3.2.3 Value analysis**

Neoclassical economics contends that price, determined by the interaction of supply and demand in markets, determines value. Paper II also took into consideration the demand for REE in lighting products, the potential of secondary supply of REE and the potential primary supply of REE. The model assumed a closed loop system in which REE recycled from lighting products are cycled back into production of lighting products, in line with an earlier study estimating potential of secondary REE supply for lighting products (Binnemans et al., 2013). However, we assumed additional complexity in considering the potential supply from primary REE sources (which could potentially compete with the secondary supply) and based estimates of recycling on empirically grounded scenarios.

While the model analysis of value built on simple concepts of supply and demand from economics, the case study of Solvay was used to further examine the value propositions for recycling REE for the company, its customers, and other stakeholder values. This incorporated a stakeholder theory of value (championed by Freeman (1994)), which underpins many of the considerations of value for circular business models in the business management field (see e.g. Bocken et al., 2015; Bocken et al., 2013; Uusitalo & Antikainen, 2018). Wider considerations of value were coded as part of the analysis of the Solvay case study in Paper II and other factors of value, such as risk and externalities are discussed.

The value of secondary REE (as part of the closing material loops for REE from lighting products) was revisited as the primary focus in Paper III. Here value was considered in the context of a global value chain (GVC). Recognition of value and negotiation of price results in transactions that make materials and products flow in a value chain. Examining factors of supply and demand on a macro-scale, or within transactions on a micro-scale can give insight into the construction of value (both economic and non-economics value) for REE. Examinations of transactions can also take into account the exchange of information and data (Gereffi et al., 2005), which in turn are underpinned by the relationship between risk and value (Boholm & Corvellec, 2011). Paper III merged a GVC framework with considerations of risk and value to analyse the value of closing the loop for REE.

### **3.2.4 Life cycle approaches**

Life Cycle Cost (LCC) calculations depend on which costs for which stakeholders are considered. The approach underpinning regulations pursuant to the EU Ecodesign Directive consider LCC from the perspective of the consumer (Annex II of the Directive, see also Siderius & Nakagami, 2013).



comparative, to the environmental impacts of product y) (Fauzi, Lavoie, Sorelli, Heidari, & Amor, 2019). It is suggested that attributional LCAs consider “the flows in the environment within a chosen temporal window” while consequential “considers how the flows may change in response to decisions” (Ekvall et al., 2016, p. 254).

An attributional approach was adopted for the LCA in Paper V for several reasons. All the previous LCA research had taken an attributional approach in using the assessment to identify hotspots in the LED lamp life cycle and for comparison of lighting products. The LCA considered products sold on the market 2012 and in 2017, a retrospective and bounded temporal window. Lastly, the aim of the LCA in Paper V was to assess trade-offs, but also demonstrate how treating lifetime as a simple factor in sensitivity analysis could lead to misleading results. This is visible when reconstructing earlier LCAs with a different approach to considerations of lifetime.

In considering the variable of product lifetimes in LCAs, a few studies had considered the dynamics of product improvements for other products. Optimal durability for refrigerators and televisions (Bakker et al., 2014), vacuum cleaners (Bobba, Ardente, & Mathieux, 2016), dishwashers (Ardente & Peiró, 2015), washing machines (Ardente & Mathieux, 2014), ovens and fridge-freezers (Boulos et al., 2015) had been considered running sensitivity analysis with the lifetime and efficiency variables or by constructing scenarios with key variables. The methods for building scenarios for lighting product improvements built particularly on method in Boulos (2015). Additional factors, e.g. lumen depreciation, were brought up by peer reviewers of Paper V and incorporated into the sensitivity analysis.

### 3.3 Data collection

Papers I and III of this research were primarily qualitative, while Papers IV and V were primarily quantitative and Paper II utilised a mixed approach. Where possible, the studies involved triangulation of data (and sources of data), including literature and documents, interviews, and observation.

#### 3.3.1 Literature review

Several literature reviews were conducted during the course of this research. The initial exploratory literature review examined the current state of knowledge about EPR systems, and specific studies about the EoL management for lighting products. Subsequent literature reviews targeted the selected case topics, most notably the issue of critical material (e.g. REE) use and recycling, reuse and recycling of

lighting products and specific literature about LED lamp lifetimes and modelling of product lifetimes in life cycle approaches.

Grey literature from companies and relevant organisations and stakeholder submission to policy processes were reviewed to understand different stakeholder perspectives on issues. EU technical reports were used to guide LCA and LCC approaches to ensure the methods adopted by this research and issues explored would be policy-relevant.

### **3.3.2 Interviews**

For Paper I, obligated stakeholders identified by the intervention theory analysis were contacted and interviewed when possible (a list of interviews can be found in the appendix of Paper I). These included the administrative/government authorities, Producer Responsibility Organisations (PROs), national and municipal waste organisations, recyclers, and retailers for each case. Snowballing (i.e. asking the interviewees what other stakeholders might be relevant to the study to interview) was used to identify additional interviewees after the initial round of interviewees. This approach identified some effects and side effects not anticipated by the initial intervention theory.

The interview format was semi-structured: after some initial clarifying questions, the majority of the questions were open ended “how” and “what” questions (Justesen & Mik-Meyer, 2012). Where possible, the interviews were conducted in person. This enabled some observations of aspects of EPR systems in practice (particularly in Denmark and Sweden, to a lesser extent in Norway, Germany, and Switzerland) – see Section 3.3.3.

In each case, stakeholders were interviewed with a core protocol developed for their stakeholder group (see examples in the appendix of Paper I). This protocol was sent to the stakeholders in advance; both so interviewees had time to think about the answers and to aid with the fact that the interviews were requested in English, which, for most respondents, was not their first language. The purpose of the interviews was to elicit knowledge that was not readily apparent from literature or to corroborate with other data sources, so making the protocol required first conducting a literature review and review of available quantitative data. In this respect, background knowledge was already obtained before the interview and the interviewing approach was “active”, involving the interview subject in “making meaning” (Holstein & Gubrium, 1995, p. 4).

For Paper II and III, PROs (3), Recyclers (1) and REE processors (3) involved in recycling of lighting products and REE from lighting products were interviewed or re-interviewed (some has already been interviewed for Paper I). These interviews were in the form of semi-structured skype interviews while earlier interviewees

responded to the specific additional questions via email (more information about the interviewees are found in Papers II and III).

The interviews for Papers I-III were recorded and notes were transcribed into a table relating to the analytical themes of each study (see Section 3.2). For Paper I, responses relating to particular issues and factors of best practice were coded inductively, through a process of initial coding followed by focussed coding (see Charmaz, 2006). The factors are presented in the results of Paper I (see Section 4.1).

In addition to formal interviews, stakeholders were also contacted by email to obtain specific information about technological processes, to clarify, or to update information from academic literature or project reports (e.g. updated information about recycling processes or REE processing).

### 3.3.3 Observation

Where possible, observation was used to aid understanding of the life cycle of lighting products. Some aspects of the life cycle were understandably easier to observe than others, and the main activities that could be observed directly were the testing of lighting products, collection points, and the recycling of lighting products (see Figure 11 for examples).



**Figure 11.** Clockwise from the top left: 1) Testing of lamps at Chinese National Lighting Test Centre; 2) Lamp collection in apartments in Sweden; 3) Lamp recycling facility in Hovmantorp, Sweden; 4) Lamp collection at municipal collection point, Sweden. Source: author.

In addition, aspects of policy processes were observed. This, of course, depended on what policies were currently in discussion. During the research, the main policy consultations that were open for participation were the EU's revisions of the green public procurement criteria for street lighting and discussions on resource efficiency requirements in the Ecodesign Directive implementing regulations. There was also a stakeholder workshop, with policymakers in attendance, discussing a critical raw materials strategy for the Nordic countries.

### **3.3.4 Quantitative data and modelling**

Quantitative data was collected from publicly available statistics from Eurostat, existing survey data, data from national authorities, and reports published by entities such as PROs and municipal waste organisations. Collecting from multiple sources revealed disparities in data, which then required follow up and checks to determine validity.

Data for Paper II was collected from the sources mentioned above and academic literature where available. However, market (i.e. metal prices) and grey data (e.g. lighting market characterisation and information about prospective REE mines) was used for construction of the MFA model for the supply and demand of REE. Where possible, these sources were triangulated or justification for the choice of data used was provided. Sensitivity analysis and the full model data was also provided in the appendix to the paper.

Market data for Paper IV was provided by Big2Great<sup>15</sup>, which was web-crawled from Swedish market websites in December 2016. A sample was drawn from this data dependent on whether the data fit the case scope (i.e. household “A” or “klot” LED lamps) and for which there was data on lifetimes, efficiency, and lumen output. The model and data was provided in an open online repository.

Life cycle inventory data for Paper V primarily relied on the Ecoinvent (V3.3) database; The life cycle inventory (LCI) was constructed with the bill of materials from an LED lamp from 2012 and three LED lamps from 2017 (data from Scholand & Dillon, 2012 and Dillon, Ross, & Dzombak, 2019). This data was then matched with Ecoinvent data with the SimaPro software. The model and data was provided in an open online repository.

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<sup>15</sup> Big2Great were introduced to the researcher by the Swedish Energy Agency as part of the Ecodesign research project.

### 3.4 Validity and transparency

It is important in research to consider validity in the research design, which broadly speaking refers to how reliable and accurate the results and interpretation are (internal validity), how these relate to the evidence (construct validity) and how generalisations are made (external validity - if indeed, that is the aim of the research) (Gibbert, Ruigrok, & Wicki, 2008; Sovacool et al., 2018). In this research, internal validity was considered by triangulating data sources and methods where possible, and choosing robust analytical methods, as described earlier. This was the case for the case studies, which were comparative or part of a mixed methods approach in Papers I-III. The quantitative approaches, in Papers IV-V, also included sensitivity analyses. Limitations to generalisability were noted and the research noted where it intended to be inductive and the need for further research/testing. Methods were described in detail in each paper and interview protocols appended for increased replicability and transparency, along with full datasets as supplementary data. In addition to the peer-reviewed publications, all published studies were made available in open-source channels.



# 4 Key Findings and Analysis

This chapter provides a brief overview of the papers in this thesis, including the approach and main findings.

## 4.1 Paper I

Paper I evaluated the environmental effectiveness of EPR schemes for collection and recycling of gas discharge lamps in the Nordic countries, previously identified as examples of best practice for WEEE systems (Ylä-Mella et al., 2014). It built upon previous research for EPR policies applying a theory-based policy evaluation approach (Manomaivibool, 2011; Tojo, 2004), but supplemented this with a stakeholder evaluation approach to inductively suggest the factors for success in the Nordic countries and the remaining challenges. The conceptual focus of this paper was primarily on closing material loops through collection and recycling of the lamps and their materials and ecodesign for recycling strategies.

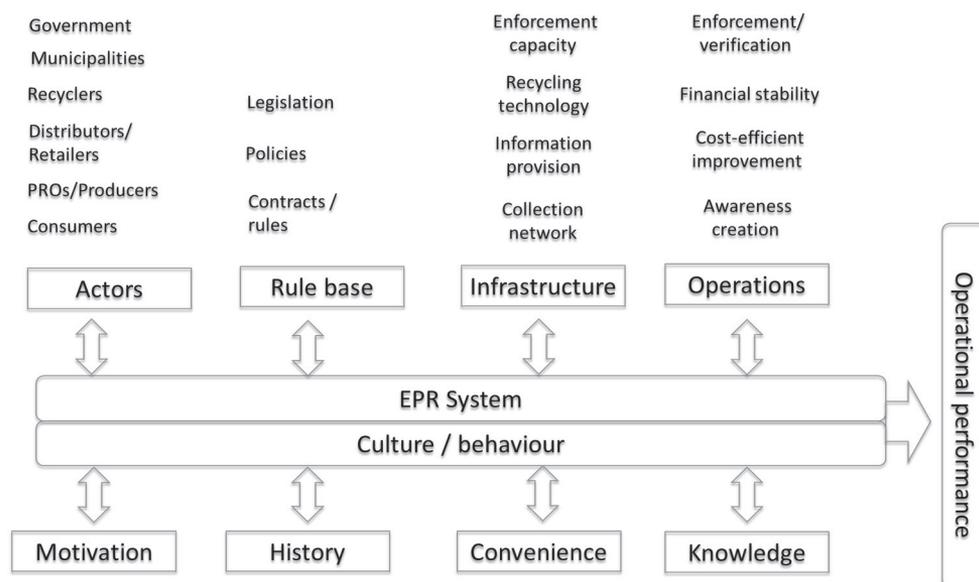
Key findings of the paper included evidence of the contribution of EPR policies towards improving performance of waste management for lamps in the Nordic countries. While counterfactual scenarios are difficult to construct, there was evidence from government documents prior to introduction of EPR policies describing the small-scale voluntary collection and recycling, mainly of business lighting product waste, concluding that this performance was inadequate (see Kemikalieinspektionen, 1998). In addition, stakeholders interviewed observed the improvement of collection and recycling of lamps in response to EPR legislation, confirming that without EPR legislation there would not be a good system for lamp collection and recycling due to the costs of such operations. The stakeholder interviews and literature both reiterated the importance of EPR for collection and recycling of this product group to avoid environmental harm and for resource recovery.

While prior research had found evidence of ecodesign changes in anticipation of the WEEE Directive legislation (Tojo, 2004), it has been argued that the full potential of design incentives has not been transposed in the member-state legislation resulting in less incentives in practice (Van Rossem et al., 2006). The findings of

this study confirmed that ecodesign incentives remain weak in the way EPR is currently implemented for WEEE.

The collection and recycling performance of the Nordic countries was generally good relative to other countries in the EU, but this picture changed for Norway depending on how this was measured, e.g. by weight collected per capita or as a percentage of put on market in the previous three years.

The study also suggested factors contributing to high collection and recycling rates in the Nordic countries (shown in Figure 12). These provide a descriptive framework that can be tested and further refined in studies of other EPR systems in future research.



**Figure 12.** Factors for best practice in EPR systems. Adapted from Richter & Koppejan (2016).

There were several barriers to closing materials loops for lighting products in practice. The low quality of the glass fraction, possible contamination with mercury, and high transport distances/costs to lamp production facilities were found to be barriers for the use of recycled glass in many applications. In most of the Nordic countries this fraction was mainly used as landfill cover, which is considered recycled, though arguably the use is quite close to “backfilling”<sup>16</sup>. In addition,

<sup>16</sup> Defined by Commission Decision 2011/753/EU,<sup>16</sup> “backfilling” is not considered recycling by the Article 3 of Directive 2008/98/EC (also referred to in Article 3 of Directive 2012/19/EU).

plastics from lamps were mainly incinerated, leaving only the small metal fraction reliably recycled from lighting products.

Critical materials recovered in the mercury phosphor layer were technically recoverable for recycling, but whether this was the case in reality was subject to contextual factors (explored further in Papers II and III). Interestingly, the mercury was required to be landfilled as hazardous waste, in accordance with laws minimising toxicity and hazardous waste and recycling the mercury was not an option. This highlights the tensions between closing material loops and “cleaning” such loops to eliminate toxic and hazardous materials. Lastly, the LED lamp technologies and designs were also rapidly evolving, making anticipation of what the future waste stream would look like and when lighting product would reach end-of-life (EoL) more complicated (issues further explored in Papers IV and V).

## 4.2 Paper II

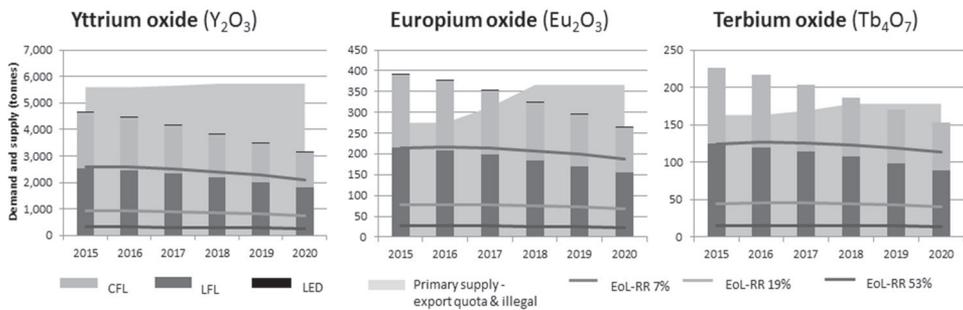
In 2014, there was much interest worldwide in the area of critical material strategies and recycling processes for CRMs. While lighting products are a relatively small end use for CRMs like REEs, only accounting for 7% of REEs in its end use for phosphors, this at the same time represents 32% of the value for the REE market (Binnemans et al., 2013; Schüler et al., 2011). Studies found that there was very little recycling of these materials (less than 1%). However, there was demonstrated technical feasibility of recycling processes on a lab and some even emerging at the industrial scale (Binnemans & Jones, 2014; Binnemans et al., 2013). Still, the small amount of REE per product and dispersion of lighting products posed significant barriers to recovery of the REE, which in turn affected the potential for secondary supply.

Paper II aimed to address the gap in knowledge of the potential to supply the REE market with secondary sources from recycling. It further developed recycling rate estimates (first presented in Binnemans et al., 2013) by considering the actual reported recycling rates for lamps based on existing policy measures (building upon the research in Paper I), and the potential from further policy development. This was then multiplied by the efficiency of the lamp recycling processes and the efficiency of the REE recycling process (these last two rates had not been considered separately in previous studies).

Considering the data about actual collection rates for gas discharge lamps, it became clear that a 40% collection rate assumed in the pessimistic scenario by Binnemans et al. (2013) was in fact still optimistic. This rate was only achieved by countries (and some U.S. states) with mandatory EPR legislation in place (and even within the EU, there was not uniform performance at this level). Linking the collection and

recycling rates to current policies and practices enabled the modelling in Paper II to give different insights into the influence of, and implications for, policies on the recycling of REE and potential contribution of secondary REE to offset primary supply in the market. For example, the Paper highlighted the important role EPR policy had already played in incentivising collection and creating resource recovery opportunities like the REE case examined. The WEEE Directive has been the impetus for setting up collection and recycling infrastructures in the EU.

The modelling of the secondary supply of REE in comparison to the primary supply revealed the realities of these two supplies in competition and that, for the most part, primary could meet most of the projected demand (see Figure 13). This dynamic had not been addressed explicitly by prior research modelling potential secondary supply of rare earths from recycled lamp phosphors.

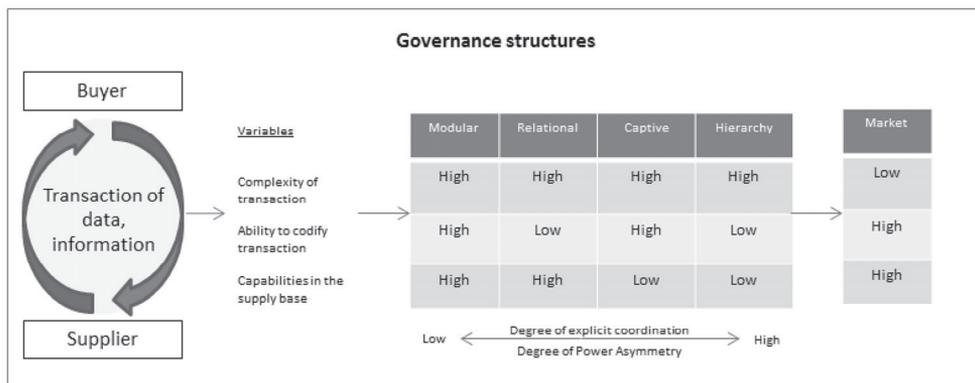


**Figure 13.** Potential secondary supply distribution for Y, Eu and Tb based on our three end-of-life recycling rates (EoL-RR) as compared to demand per lamp type (bars) and 3 year delay base case primary supply forecast (grey shading) from 2015 to 2020. NOTE: different y-axis scales. We use rare earth oxide (REO) as unit to enable a comparison of demand, and primary and potential secondary supply estimates, however, the use of REO in lamp phosphors requires their prior purification to metal. Source: Machacek, Richter, Habib, & Klossek (2015).

The paper also discussed challenges such as the lower volumes of secondary supply and the fact that there are negative externalities not considered in the market price of primary REE supply and positive externalities not considered in the market price of secondary REE supply. The modelling, while simplifying reality, helped to identify important challenges for secondary supply. These challenges were confirmed by the case study looking more in depth at Solvay-Rhodia's experience in recycling REE phosphors. The paper confirmed the importance of policies in driving a transition to CE, but that global market conditions still presented barriers that need to be addressed by these policies.

### 4.3 Paper III

Continuing from the groundwork laid in Paper II, this paper examined the role of risk and value in transactions affecting recycling and closure of material loops for REE. This research focussed on empirical case studies along the global value chain (GVC) – the chemical separation facility, pre-consumer magnet recycling and post-consumer phosphor powder recycling from waste lamps. The novel analytical framework combined conceptual elements of governance derived from a GVC framework (Gereffi et al., 2005) with risk-value constructions from a relational theory of risk (Boholm & Corvellec, 2011). Both of these elements arise in transactions of data and information between interacting individuals or entities, in Figure 14 simplified as the buyer and the supplier. The transaction occurs prior to an agreement to exchange a material, product or service. Thus, rather than a transaction based on an established price, the focus in this paper was on the transaction of data and information that occurs prior to (and shaping) a price, and considered in light of both GVC governance and risk and value constructions.



**Figure 14.** Analytical framework linking transactions before the price formation and governance structure of the GVC. Source: Machacek, Richter & Lane (2017).

The cases explored different aspects of governance, risk and value along the global REE value chain. The case of the REE phosphor recycling from EoL lamps (Solvay-Rhodia and other processors) was ultimately a failure. We argued that the extensive need for information (and current lack thereof) in post-consumer recycling gave rise to a hierarchy governance structure that impedes the closure of REE loops. This suggests that alternative governance structures, in which the involvement of the actors to the transaction is more balanced, should be explored to close material loops in this segment.

The paper concluded that the government could play a pivotal role in closing material loops when the risk-value construction by industrial actors does not recognise, or is at odds with, the societal values such as public and environmental health. Policy recommendations included specific measures such as the elaboration of standards for secondary material processing and general recommendations for governments to consider how they could better capture societal and environmental values and manage public risks through policies.

## 4.4 Paper IV

In this paper the case of lighting products, one of the first product groups to have mandatory minimum durability requirements, was examined to investigate the question of optimal durability (in this case, considering lifetime), with a focus on the optimal (i.e. least) life cycle costs (LCC) for the consumer. The LCC depends on factors such as the purchase price and running costs. This paper defined LCC as:

$$LCC=P_A+PWF\cdot P_E\cdot UEC$$

where  $P_A$  is the appliance price, PWF is the present worth factor,  $P_E$  is the price of electricity, and UEC is the annual unit energy use.

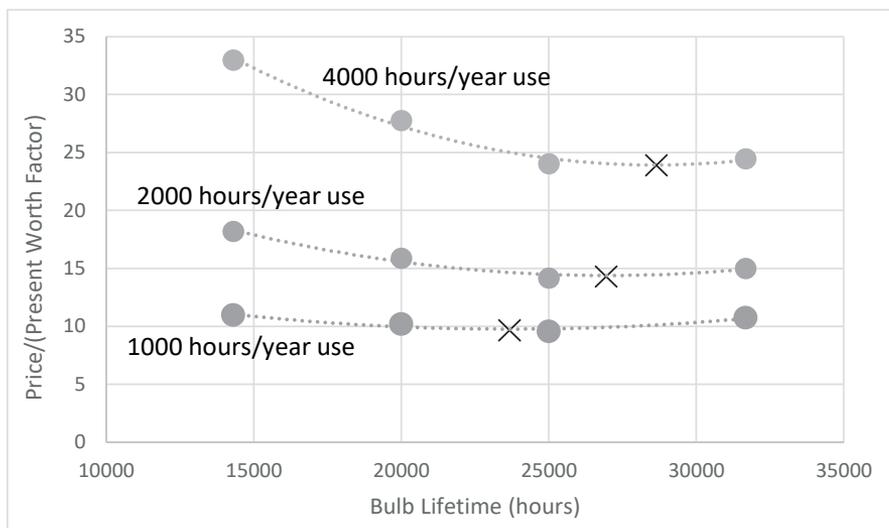
Dividing by the PWF (which takes into account the influence of inflation and discount rates) gives the annualised LCC (i.e. the measure of the costs of the lamps that occur every year):

$$LCC/PWF=P_A/PWF+P_E\cdot UEC$$

The focus of the analysis is then on the change in  $P_A/PWF$  with respect to the lifetime in hours. A regression analysis was used to calculate price regression coefficients for four lifetime categories:  $\leq 15000$  hours, 20000 hours, 25000 hours, and  $\geq 30000$  hours. Then  $P_A/PWF$  was calculated as a function of lifetime and considering different scenarios for intensity of use. Figure 15 shows the results of the analysis.

The analysis indicated that from a consumer LCC perspective, considering the real-time market in Sweden, between 20000 and 30000 hours was the optimal lifetime of LEDs, assuming a 6% discount rate. A low discount rate implied even higher optimal lifetimes. The implications of this result were considered for the policy with its current minimum requirement of 6000 hours. The paper explored the possibilities of more stringent mandatory requirements as part of the Ecodesign requirements versus warranty, labelling and more voluntary approaches, as alternative policy measures to promote longer lifespans. While longer lifetime could be advantageous

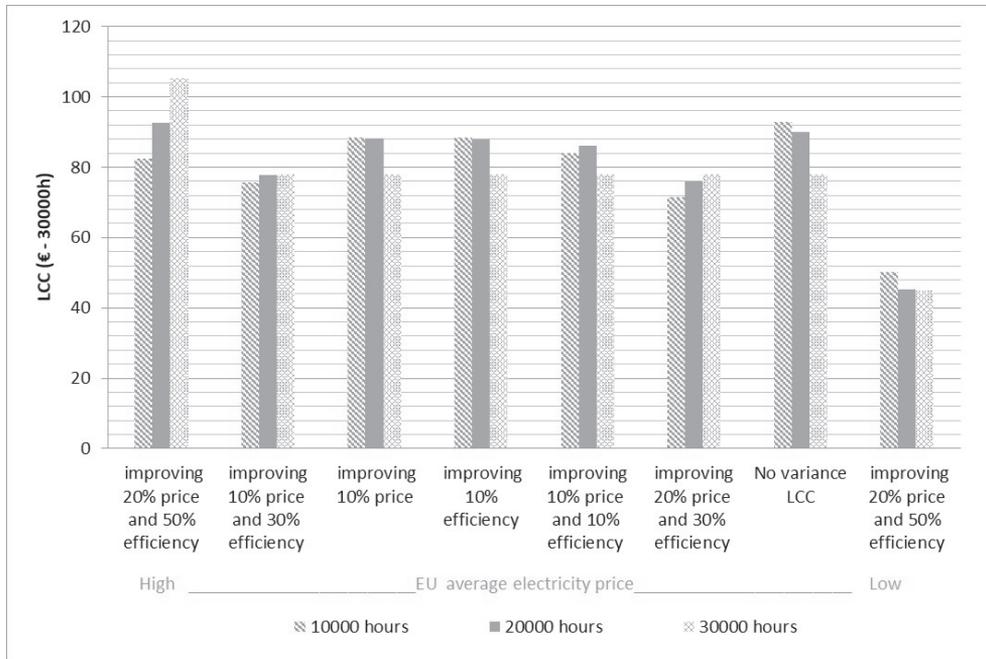
from the LCC perspective, setting mandatory standards must also consider the environmental impacts. This question was the focus then of Paper V.



**Figure 15.** Model approximating optimum lifetimes (marked with x) for different scenarios of use, assuming 6% discount rate. Source: Richter, Van Buskirk, Dalhammar, & Bennich (2019).

In addition to finding that longer product lifetimes could be desirable from an LCC perspective, the paper also demonstrated an approach for real-time calculation and monitoring of lifetimes in the market. Since real-time product market surveillance is increasingly used by national agencies (e.g. the Swedish Energy Agency), this paper demonstrated a valuable method to contribute to this monitoring and for informing policy developments.

However, the case needed a sensitivity analysis to understand the factors that could make it more generalisable. A conference paper following this paper (examining a ~800 lumen subset of the same dataset) followed up the findings with a scenario-based sensitivity analysis, summarised in Figure 16. The main findings of the analysis were that longer lifetimes were preferable when 1) products matured in either purchase price or efficiency, or 2) in the context of low energy prices (even when technology and purchase prices are decreasing). In the context of high energy prices, moderate improvements in price and efficiency may make shorter lifetimes (i.e. earlier replacement) desirable from an LCC perspective. The results showed that optimal lifetime is highly dependent on consumer choices amongst wide variance in product prices, efficiencies and lifetimes. (Richter, Dalhammar, & Tähkämö, 2017).



**Figure 16.** LCC scenarios considering improvements in purchase price and energy efficiency, in the context of low EU electricity prices (price in Bulgaria - scenario 1), average EU electricity prices (scenarios 2-7), and high electricity prices (price in Denmark - scenario 8). Adapted from Richter, Dalhammar, & Tåhkämö (2017).

Paper IV also considered if there would be trade-offs and/or differences between an LCC and an LCA approach. However, it was difficult to draw concrete conclusions on this question since LCAs specific to LED lamps reviewed had considered longer lifetimes only in a static approach (i.e. the dynamics of product development were not considered). A dynamic LCA approach considering LED products was needed to further address this question.

## 4.5 Paper V

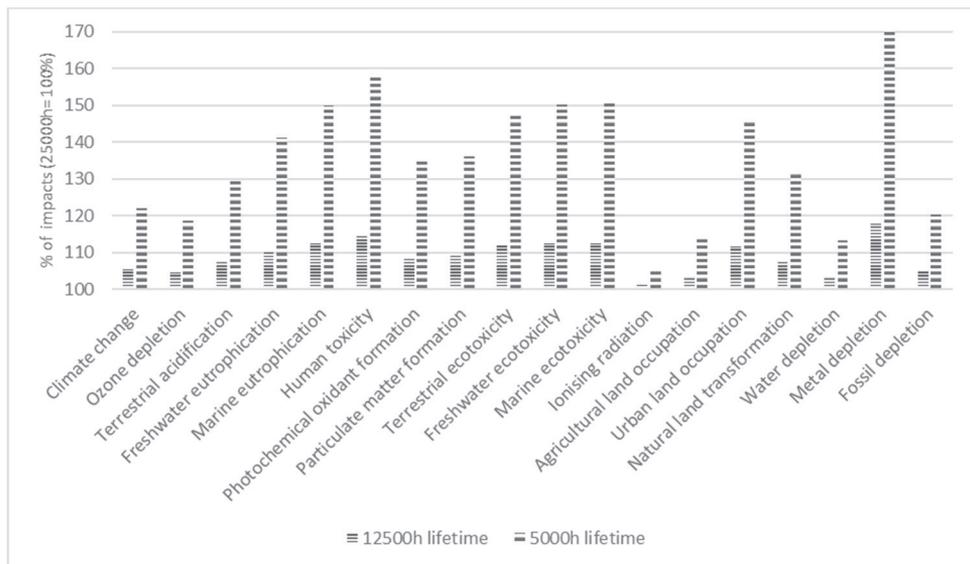
This paper considered the question of whether longer lifetimes for LED products were desirable from an environmental, i.e. LCA, perspective and if there may be potential trade-offs between different types of environmental impacts and life cycle stages. The LCA considered dynamic factors such as rapid product developments and changing energy mixes, and their implications for the ‘optimal durability’ level from an environmental perspective. To do this, several scenarios were constructed – see Table 2.

Table 2. Overview of scenarios in Paper V.

	Product assumptions	Electricity mix assumptions
Static Scenario	2012 product replaced at 12500h or 5000h by identical product	EU electricity mix
Improved product scenario: EU electricity mix	2012 product replaced at 5000h by 1 of 3 potential 2017 products	EU electricity mix
Improved product scenario: decarbonised electricity mix	2012 product replaced at 5000h by 1 of 3 potential 2017 products	Norway and Sweden electricity mixes

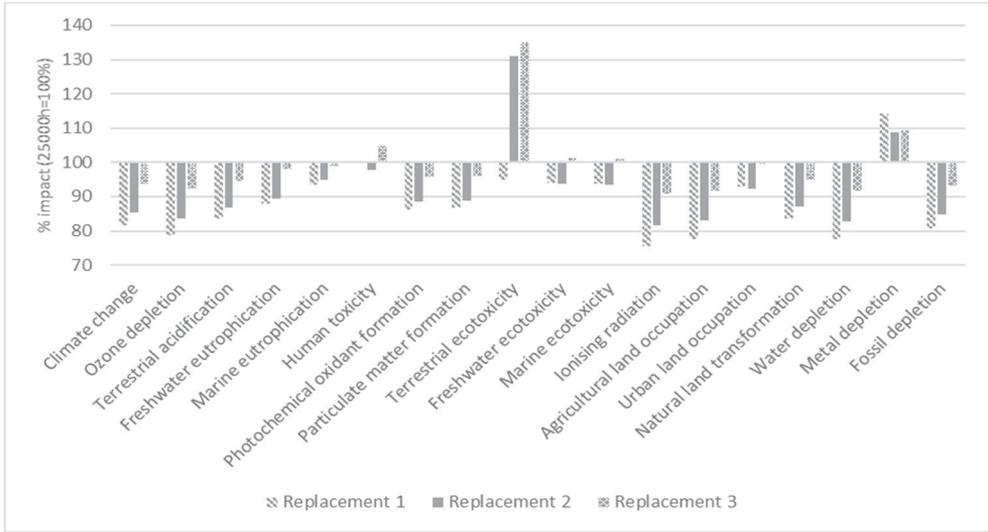
Source: Richter, Tähkämö, & Dalhammar (2019).

The results indicated that including the dynamic factors did indeed have a significant effect compared to the static methods used in past LED lamp LCAs (Figure 17).



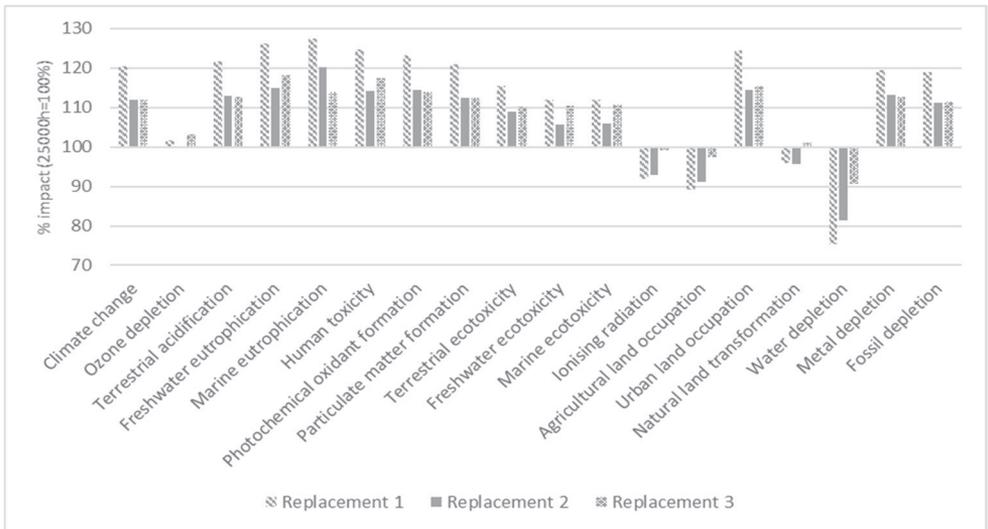
**Figure 17.** Static analysis comparison of environmental impacts of identical 2012 LED lamps, varying the lifetime (12500h, 5000h) compared to the 25000h base case LED lamp (100 % on y axis – dotted line) in the context of EU average electricity. Source: Richter et al. (2019).

Considering dynamic factors within the same electricity mix yielded a much more complex picture that resulted in trade-offs between different environmental impacts depending on which replacement lamp was chosen (Figure 18).



**Figure 18.** Comparison of environmental impacts of 3 replacement options (original lamp replaced after 5000 h of use) relative to no replacement (i.e. base case – dotted line) in the context of EU average electricity mix. Replacement 1 represented the most efficient replacement and replacement 3 the least efficient. Replacement 1 has a lifetime of 10950 hours, while all other product lifetimes are 25000 hours. Source: Richter et al. (2019).

Not only product development, but also the energy context and development of electricity mixes, can have a large influence on whether longer lifetimes lead to lower environmental impacts, as Figure 19 reveals.



**Figure 19.** Comparison of environmental impacts of 3 replacement options (original lamp replaced after 5000 h of use) relative to no replacement (i.e. base case – dotted line) in the context of Norwegian average electricity mix. Source: Richter et al. (2019).

The findings of the LCA study indicated that promoting durability in the context of product groups experiencing rapid technological changes and an electricity mix with fossil fuels is likely to result in trade-offs between energy and material/toxicity-related environmental impacts. The approach showed that it is important to consider a broad range of impacts (i.e. not just climate impacts) in order to fully assess these trade-offs. Assumptions about what product is used as a replacement also matters to the results of the LCA. The case of LED lamps demonstrated that in addition to energy efficiency, material design, such as decreased use of aluminium for heat sinks, lower weight of metals and other materials, or small electronic components, can influence trade-offs, particularly for toxicity-related impact categories.

This research confirmed the importance of electricity mix for environmental impacts. A less carbon-intensive electricity mix minimises the trade-offs between environmental impacts in the case of improving product efficiencies. It is then important that developments leading towards decarbonisation of the electricity mix are considered in determining the overall impact of longer product lifetimes as it was shown to both minimise the overall impacts of the LED lamps and minimise the trade-offs.



# 5 Concluding Discussion

This chapter revisits the aims and research questions. It presents the contributions of the thesis in light of ongoing research and policy developments. It reflects on the research approach, and on the need and direction for future research.

## 5.1 Performance and potential for closing and slowing material loops

This research evaluated how EPR systems for lighting products in the Nordic countries have been effective (RQ1). The Nordic countries have relatively high performance levels for collection and recycling of lighting products according to the statistics, but the statistics proved unreliable for drawing concrete conclusions for several reasons. For example, only examining statistics from Eurostat would lead to the conclusion that Estonia and Latvia's EPR systems for lamps out-perform the Nordic countries. While the numbers were confirmed by the reporting PROs, stakeholders interviewed in the countries had different explanations, including that a lot of the waste was historical. At the same time, stakeholders interviewed in Norway had several explanations for statistics indicating low performance for collection of gas discharge lamps (including that they were wrong). Minor errors (on subcategory specifications) were found in the Swedish statistics for gas discharge lamps. Relying only on statistics could lead to over-simplified (and even wrong) conclusions (see Richter (2016) for further discussion of WEEE statistics).

The theory-based evaluation method utilised in this research revealed a more nuanced and rich picture of the performance of the EPR systems for lamps in the Nordic countries. Among the factors noted by stakeholders as contributing to high collection was a culture of recycling behaviour in the Nordic countries. In literature, recycling behaviour has been explained in many ways, most commonly referring to attitudinal/motivational aspects, situational contextual factors, capabilities and awareness, and habits and routines (see e.g. Darby & Obara, 2005; Meneses, 2015; Saphores, Ogunseitán, & Shapiro, 2012; Sidique, Lupi, & Joshi, 2010; Tonglet, Phillips, & Bates, 2004; Vesely, Klöckner, & Dohnal, 2016; Wang et al., 2011). Melissen's review (2006), with a focus on recycling behaviour of small electronics,

provided a more simplified synthesis of important factors for this behaviour using the Poiesz triad model (1999) of motivation, opportunity (i.e. having access to recycling), and capacity (i.e. the knowledge and awareness about how to recycle). These broad factors are both explicit and implicit in the recycling “culture” findings from the research for Paper I; and illustrate that “culture” is inter-dependent with other factors in an EPR system. The introduction of EPR policy and setting up of convenient collection systems can develop a culture of recycling, but some existing elements of culture (e.g., how people generally respond to new policies) can also be influential in EPR system development.

The fact that lighting products were found to be collected at a relatively high rate did not appear to be motivated by the recycling target, which is based on overall WEEE collection and based on weight. This suggested that the high awareness amongst stakeholders of the risk from mercury played a large role in motivating the collection of the stream. However, with the transition to mercury-free LED technologies, this motivation could weaken. Further improvement in collection could be motivated by an individual collection target for lamps. In a report for the EU Commission on individual targets for WEEE, Magalini et al. (2014) found the feasibility of implementing such individual targets in Sweden to be “high” (also in Denmark, while “medium” for Finland). For already high-performing countries, an individual target at the same level as the general target (as France has done) may not motivate continuous improvement, as the target is already achievable. In this case, framing individual targets to reflect an ideal target and taxing the difference can provide more continuous incentives while also acknowledging the challenges (e.g. similar to Norway’s tax on beverage packaging, which reduces from 25% and does not apply above 95% of collection and recycling – see Infinitum, 2014).

The research in Paper I found evidence of improvements to the collection and recycling of lighting products with the introduction of EPR. This is not surprising considering waste lighting products are a classic case for EPR – i.e. a product with little value at end-of-life (EoL) and with a potentially significant negative environmental impact from the mercury contained within the product. As this waste stream changes in response to the consumer adoption of LED lamps, however, this could change. Some of the latest research, for example, has indicated a positive scrap value for LED lamp waste (Dzombak, Antonopoulos, & Dillon, 2019). However, this research showed that retrieving the potential scrap value of LED lamps is still challenging.

High collection rates do not necessarily lead to high recycling rates. The recycling is complicated by many factors, including changing product designs that incorporate different (and increasingly complex) materials. In the course of this research the materials used in LED lamps changed rapidly towards an increase in plastics and then towards glass, with significant implications for recyclers in meeting recycling

targets (e.g. the plastic-heavy designs were a risk for recyclers and PROs in particular).

Incentives for design for recyclability are still lacking in the system for lighting products, as prior research indicated is the case for EPR for WEEE more generally (Sander et al., 2007; Van Rossem, 2008). The collective PRO and market share system in practice means that, in most member states, fluorescent lighting waste is financed also by LED waste product fees, despite the lack of mercury in these products. The argument for this is that both types of lamps are in reality collected and processed together and thus incur the same costs. Systems based on current market shares also generally mean that old technologies are financed by the new technologies. Though there are some promising technological developments in separating the LEDs from fluorescent lighting in the waste stream (see e.g. Illuminate Project, 2016), it is currently still difficult to do this at source. Doing so in the future would allow for LED lamps to be treated separately and incur costs that are more accurate (though, of course, the cost of such separation must also be considered).

Individual producer responsibility (IPR) could further enhance incentives for better design, but there currently seems to be little traction in this area. In 2014, there were processes that had been developed for individual brand recycling of fluorescent tubes (mainly B2B lamps), and in the context of high REE prices there was development of processes to separate fluorescent lighting by type of REE phosphor used (Binnemans et al., 2013). However, manufacturing of lighting products has continued to shift outside of the EU (Ornelis, J. personal communication 17 November 2016); and this, combined with low material prices generally (even the spike in REE prices was temporary), makes closing loops within an IPR less attractive without government intervention. Some researchers have proposed there are changes needed to the transposition of EPR legislation in general to ensure IPR remains at the member state level while other have proposed ways to enhance individual incentives within collective schemes (see e.g. Mayers et al., 2013; van Rossem et al., 2006).

“Eco-modulation” of financial contributions paid by producers could also differentiate product design, where ecodesigned products are charged lower EPR fees than standard products (or conversely, products with designs especially detrimental are surcharged). The 2018 amendment to the EU Waste Framework Directive<sup>17</sup> has included obligatory provisions for eco-modulation where EPR is collectively fulfilled (Article 8a(4b)). France has such differentiated fees for LED lighting products (LED producers pay 20% less than the standard fee for other

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<sup>17</sup> Directive (EU) 2018/851 of the European Parliament and of the Council of 30 May 2018 amending Directive 2008/98/EC on waste

lighting products) and some PROs differentiate between lighting products based on mercury content (Damgaard, Skotte, Drescher, & Holten, 2015). However, there remain several challenges to this approach. Currently, LED lighting products are considered to be the best example of ecodesign (i.e. France does not distinguish between LED products with its modulated fees, only LED from other lighting products), but this approach does not reward designs incorporating modularity or the ability to recover all the materials in the product. Indeed, producers have argued that LED products are actually less recyclable than fluorescent lighting products with the current recycling processes (EucoLight & LightingEurope, 2018).

The revised EU Waste Framework Directive promotes a variety of ecodesign strategies that should be incentivised, including durability and recyclability, and advocates a life cycle approach (Article 8a(4b)). This could help in determining which strategies and product groups to prioritise, and trade-offs between different strategies. Papers IV and V highlighted the need for the life cycle approaches to consider the dynamic factors such as product development and changing contexts (e.g. electricity mixes).

However, even when a life cycle approach is taken to setting eco-modulation fees to better reflect design differences, there is still the challenge that such fees may have little effect on ecodesign in practice. For the case of lighting products, product development towards LED lighting products is occurring (and likely to continue) independently of any EPR incentives (EucoLight & LightingEurope, 2018). Even if differentiated fees are introduced, the low cost of EPR is unlikely to be a large enough incentive for either producers or consumers, even in the case of lighting where the EPR cost is relatively high compared to the purchase price (Damgaard et al., 2015; Gottberg et al., 2006). This highlights the need to consider direct mandatory Ecodesign requirements to incentivise producers. The relatively low cost of EPR compliance in relation to other costs faced by producers may also change in the future, particularly if recycling is subject to more qualitative requirements, which could potentially raise the costs of treatment and compliance.

There is a need to consider how EPR legislation like the WEEE Directive, recycling standards and ecodesign requirements are aligned (which is acknowledged in the WEEE Directive). Products designed for reuse, disassembly, and recycling need to end up in processes that can take advantage of such designs (i.e. aim for the highest level of recycling quality rather than shredding to deal with a low quality mix). Without alignment, actions in one policy addressing one life cycle stage, but intended to have an impact on another, are undermined. Qualitative requirements for reuse, collection and recycling need to be developed that ensure any requirements for reparability and recyclability are not undermined by systems designed for the lowest common denominator.

Quantitative targets on recycling by weight can ensure that recycling takes place, but they do not address the issues with downcycling and loss of valuable materials in the current recycling processes. Other researchers have noted that such design for recycling aspects for WEEE generally are not addressed by the WEEE Directive (Balkenende & Bakker, 2015). The metal and glass fractions in lighting products can be downcycled and critical raw materials (CRMs) lost while still meeting the current recycling targets. This loss of value and material is in conflict with the objectives of value retention and resource efficiency expressed in both the WEEE Directive and the CE Action Plan.

The research more closely examined the current state and potential for closing loops for CRMs for the case of REE from lighting products (RQ2). The research in Paper II highlighted the important role EPR policy has already played in incentivising collection and creating resource recovery opportunities for CRMs. The requirements of the WEEE Directive to collect and recycle lighting products were a key enabler for recycling of REE from lamps. The unique characteristics of the lamps and the special requirements for them in the WEEE Directive (and in transposition into the member state legislation) resulted in lamps being processed separately from the rest of WEEE with a process targeted for mercury removal, which was the same fraction as the REE phosphors. The unique characteristics of lamps isolated this waste stream from the general trend of increasingly mixed WEEE being shredded together for recovery of only common metals.

Paper III began to explore the question of complex value by examining the risk/value construction for actors along the secondary REE value chain, finding that the complex transactions need to be taken into account when considering policies in this area. The paper argued that recycling of REE has environmental and social value (similar to other findings see e.g. Ali, 2014; Yang et al., 2019), and concludes that governments have a role to provide stronger incentives for recycling to ensure externalities and non-economic values are captured. This could involve investing in building capacity and lowering transaction costs for recycling (i.e. subsidies or providing grants, which has already been done to some extent by Life+ funding). Alternatively, governments could seek to internalise the environmental and social value of recycling REE by setting more qualitative EoL treatment requirements and more detailed (i.e. on the material level) targets that can increase both the volume and quality of recycled materials. Such requirements could give more certainty and stability to critical material recovery initiatives. Researchers have argued that such requirements for recycling can incentivise more innovative recovery technologies (Kalimo et al., 2012).

The research revealed that even if recycled materials are supplied, there may still be issues with demand in the market to ensure the materials are actually sold and utilised in as high value applications as possible. Some of these issues may be

addressed through ensuring the quantity and quality of supply and having contracts with large companies (e.g., Apple is now sourcing post-industrial recycled REE for its iPhones – see Nellis, 2019). However, in the case of REE recycling from lighting products, even if technical processes can produce high quality materials and economies of scale can be improved, the low price of primary materials on the market means the business case for recycling is simply lacking (Qiu & Suh, 2019). Where recycling processes have been demonstrated, market pull policies may still be needed to ensure there is a demand for the recycled material. For instance, criteria related to recycled content of specific materials, such as REE, could be voluntary as part of Green Public Procurement or Ecolabelling schemes. It could also be part of mandatory requirements under the Ecodesign Directive (though supply chain certifications would likely be necessary – see Dalhammar, 2016).

In considering possible policies for slowing materials loops for lighting products, the research first examined whether longer lifetimes are desirable for LED lamps (RQ3) to understand the contexts and conditions such policies should take into account. The LCC from the consumer perspective considering the Swedish market in Paper IV, revealed that longer lifetimes than the current minimum lifetimes in the policy are beneficial. Longer lifetimes are preferable for lighting products in other contexts when products matured in either purchase price or efficiency or in the context of low energy prices (even when technology and purchase prices are decreasing).

Trade-offs between environmental impacts in considering longer lifetimes to LED lamps (RQ3) confirmed findings from studies of other product groups with improving energy efficiency in terms of trade-offs between energy-related and material/toxicity-related impacts (i.e. Ardenne & Mathieux, 2014; Bobba et al., 2016; Boulos et al., 2015). The findings also confirmed the importance of considering electricity mixes, as found by O’Connell et al. (2013) in their study of the Irish context.

The specific policy implications of the findings are that it may be appropriate currently at the EU level not to push tougher lifetime requirements if considering environmental impacts (though some impacts do lessen with longer lifetimes, as illustrated by Paper V) so trade-offs should be discussed and prioritisations made explicit). This seems in line with the current development of the Ecodesign lighting standards, which will implement a shorter “endurance” testing method involving continuous switching cycles for a total of 3600 hours (effectively decreasing the minimum lifetime requirement, but involving a more rigorous testing method - see EU Commission, 2019c). Product development (and where it levels off, or matures) should be projected so the timeline for implementing additional measures can be made in advance. It should also be noted that product development towards

increased energy efficiency, and other known desirable attributes, can (and should) be pushed by minimum standards and pulled by energy labels and ecolabels.

However, considering the member state level, the policy implications are different. Member states with cleaner electricity mixes should conduct an analysis specific to their region. The research in Paper V revealed that there were fewer trade-offs in the Norwegian and Swedish contexts, which indicated that policies promoting longer lifetimes could be appropriate in these countries already now. A prospective assessment with sensitivity analysis of the key factors identified could give specific insights for policies promoting durability of lighting products in these contexts. In setting such policies, voluntary policies like green public procurement (GPP) may be the most appropriate in the first instance, as mandatory minimum lifetime requirements on the member state level may pose additional burdens on producers. That said, Paper IV identified differentiation on policies, such as minimum guarantees, in the Nordic countries could be used to incentivise longer lifetimes in practice.

Beyond technical specifications of lifetimes, lifetimes in practice and consumer heterogeneity should also be considered by such policies. Lifetimes for most products, such as lighting products, also depend on patterns of use and user perceptions of obsolescence (Cooper, 2005). There is already evidence of functioning LED lighting products in the waste streams, indicating that consumers are discarding lighting products before the end of their functional lifetime (dos Santos, da Silveira, Colling, Moraes, & Brehm, 2020; Rahman, Kim, Lerondel, Bouzidi, & Clerget, 2019). Such studies suggest that it is not enough to only examine policies targeting ecodesign, but consumer behaviour needs to be addressed to ensure longer lifetime in practice.

The findings of the research have broader implications for the transition towards a CE. They indicate that a holistic approach needs to be taken in considering priorities for environmental policies; at times resource efficiency aims may undermine climate mitigation aims. However, trade-offs are also dynamic and often temporary, necessitating understanding of policy targets and roadmaps from both areas in order to optimise the policy synergies.

## 5.2 Contributions of this research

This research made several contributions in terms of method development. Paper II developed a more comprehensive consideration of recycling rates in prospective modelling of resource availability for REE (i.e. consideration of collection, technical recycling recovery rates in multiple steps of recycling). Paper III experimented with merging GVC and value risk analytical frameworks for a novel

analytical framework of value chain governance. Paper IV developed (with co-authors) a simple method for determining optimal lifetimes as part of real-time market surveillance. Paper V built upon dynamic LCA research for product lifetimes to test additional parameters (e.g. electricity contexts, product development and lumen depreciation).

Empirically, a particular contribution of this research has been to address the need for increased collection and recycling of EoL lighting products with analysis of the actual performance of current policies, identification of factors in best practices and to discuss the challenges (Paper I). The empirical evidence provided in Papers I and II were extensively used in an advisory report and workshop on considering how to enhance recycling of CRMs from EoL products in the Nordic region (Punkkinen, Mroueh, Wahlström, Youhanan, & Stenmarck, 2017). Papers II and III provided empirical evidence of the secondary material REE value chain that could be relevant for understanding transactions and challenges for secondary REE in general and lessons for recycling of other CRMs. Papers IV and V presented empirical cases of LCC and LCA data and analysis for lighting products that can be compared to other product groups. These cases were presented during an Ecodesign policy workshop in Brussels in January 2018. The methodologies were also noted by the EU Commission for discussion of updated MEERP methodology for setting Ecodesign standards.

The research contributed explanatory factors for the success of EPR in Nordic countries that can be part of a theoretical testing in future research. The research began to unpack the black box of “value” in considering recycling and other CE strategies, which is often implicit in assumptions underlying the intervention theories of policies incentivising such strategies. It should be noted, however, that the research was exploratory in nature in using a novel pairing of governance and risk/value construction concepts in one framework. It was a first attempt at really opening the black box of construction of value in the case of REE and recycling, both of which warrant further research.

### 5.3 Reflection on research approach

The focus on the case of lighting products in this research allowed for a deeper understanding of the product and exploration of a variety of CE issues related to this product group (which reflected the different aims of the projects framing this research). A focus on particular products was appropriate considering product policies often differentiate issues depending on product groups (e.g. recycling targets are specific to product groups for the WEEE Directive and ecodesign requirements are specific to a product group for Ecodesign requirements). Of

course, focus on a particular product group was a trade-off compared to examining issues related to electronics or products more broadly. That noted, the issues chosen to focus on in the papers were broader issues relevant for other product groups as well. The issues in the case of lighting are relevant especially for other complex energy-using products, where the historical focus has often been on their energy use and climate impacts, while resource use and efficiency is increasingly becoming an important issue. While some findings are likely specific to the case (e.g. the particular actors involved in the REE recycling for lamps) many of the challenges are generalisable. The focus on identifying explanatory factors in the research also made the findings more applicable for testing with other cases and contexts.

The approach of this research was transdisciplinary, starting with the policy questions posed in the various research projects. This approach necessitated a large variety of methods and concepts used in this research. Learning new methods and applying new frameworks took time, an open mind, and significant effort. It also required flexibility in filling in gaps in what was needed to answer the research question. Sometimes there was a member of the research team with the time and skills to take the lead on an aspect of the research (e.g. I mainly supported the actual modelling in Papers II and worked together with an expert on mathematical modelling for Paper IV). Other times I had to learn a new method to fill the gap and was supported by another team member (e.g. I had to learn LCA for Paper V, with the ability to check with an expert co-author when I had questions related to the design of the study). I found in my own research experience in this thesis that the ability to flexibly respond to policy questions requires collaboration and teamwork, which has been argued as a key principle of inter- and transdisciplinary research (Borrego & Newswander, 2010).

However, it must be acknowledged that this approach comes with a trade-off of depth in terms of deeply developing my skills in just one of the approaches used (e.g. developing the policy evaluation to encapsulate other criteria or policies or following up with more in-depth LCA questions). As environmental issues are complex, a systemic perspective is valuable for understanding the interrelations. Applying different approaches has increased my understanding of how to better utilise the findings of other studies using these methods, and their own limitations and assumptions (i.e. black boxes). Lastly, the transdisciplinary approach enabled exposure to a wide variety of researchers and practitioners working in this area that has been valuable to better understanding the reality of environmental product policy work.

In terms of specific approaches utilised in this research, each had its own strengths and weaknesses. Descriptive studies of the performance of the WEEE Directive in case countries have been performed by previous research without a policy evaluation approach (see e.g. Ylä-Mella et al., 2014). The theory-based stakeholder

policy evaluation approach allowed for a rich evaluation of the policy and identification of stakeholders and issues not readily apparent from examination only of the statistics. The triangulation of data collection from documents, interviews, and observation was also important to critically assess the impacts of the EPR systems. However, though attempts were made to establish causation for some of the policy impacts, this was difficult to establish. This has been noted as a challenge in policy evaluation generally.

The mixed approach for Paper II added an important empirical element to the previous models of the potential for supply of REE from recycling. The case study made issues like competition with primary supply and the complexity of GVC transactions more salient. The case study complemented the simplicity of the model with the complexity of a real life example. This complexity was further explored with the GVC framework in Paper III. The original intention of the study was to examine value, but with the initial literature, it became apparent just how complex the concept of value can be and the different disciplinary and interdisciplinary approaches that could be used. In the end, the choice of the risk and value approach was a practical way to scope the analysis to a relevant issue, but it excluded many other approaches to analysing value. The research in Paper III was the most exploratory and novel, but also the most difficult to replicate because of its more constructionist approach. Its findings are likely the least generalisable of the studies because of the narrow focus on certain actors within segments of a particular supply chain. However, the framework developed could be used more generally and further tested in other studies of transactions in GVCs.

Papers IV and V utilised life cycle modelling approaches. All models are only as good as their data and should make clear their assumptions. The modelling choices were constrained by the available data and the need to learn the modelling. Having better expertise in modelling and access to data might have led to more sophisticated prospective LCA modelling. For example, prospective modelling of newer filament LED lamps was considered, but was deemed impractical for the first LCA study due to the need to build competence in LCA first and the lack of current data for these lamps. Manufacturer-specific data on the LED filaments would have been desirable to make the model more accurate, but this was difficult to obtain within the timeframe of the research (and is often a barrier for LCA researchers in general). These improvements would be suitable for future research on the environmental impacts of lighting products.

## 5.4 Future research

Evaluation to understand effectiveness of policies towards a CE should be a subject of ongoing research. This research focussed on closing and slowing materials loops independently. As policies that have initially been more focussed on climate and energy aspects (e.g., the Ecodesign Directive) incorporate more resource efficiency and CE-related aspects, considering how they can further work in synergy with policies addressing the EoL of products should be further explored. There were several suggestions in this thesis of how the WEEE Directive and Ecodesign Directive could be more synergistic, but future work could explore the role of a wider range of product policies, including Green Public Procurement and Ecolabelling, and non-product policies that are needed in a CE like consumer, chemical, and tax policies.

There is still more research about the WEEE systems in the Nordic countries to be done as well. While they are perceived as leading countries in terms of recycling, less is known about the state of repair of electrical and electronic equipment in the region. A study by the Swedish PRO, El-Kretsen, showed that 41% of WEEE they sampled from their cage collection was repairable, but it was deemed uneconomical to pursue (El-Kretsen, 2015). Indeed, as solutions become technically feasible, there is a need to consider how costs are covered to capture environmental and social benefits. The WEEE Directive allows access to WEEE by reuse centres, but access to WEEE for reuse remains a point of contention between different actors (Richter & Dalhammar, 2019). This reflects still unresolved questions about rights and responsibilities and how costs and benefits are shared between actors.

There is much more work to be done around trade-offs in policies as policies themselves become increasingly complex in their objectives (Dalhammar, Milios, & Richter, 2019). Further development of the Ecodesign requirements will need to consider trade-offs between strategies such as design for recycling, reuse, repairability, and durability. The WEEE Directive aims to promote reuse, but only has targets for reuse combined with those for recycling; meaning reuse does not have to be prioritised in practice. Efforts to further promote reuse, and potential conflicts between recycling and reuse strategies, are only beginning to be explored (Richter & Dalhammar, 2019). Similarly the Waste Framework Directive suggests a number of resource efficiency strategies including durability, modularity, design for disassembly, but how to put these into practice in policies like the Ecodesign Directive is an ongoing discussion (see e.g. Dalhammar, 2016; Tecchio et al., 2017).

Though CE strategies are often presented as win-win strategies for businesses, it is increasingly recognised that businesses alone cannot, and most being focussed primarily on profit will not, capture societal and environmental value (see e.g. Whalen & Whalen, 2018). The research in this thesis suggested that many CE

strategies will need policies and will involve increased costs in order to achieve the longer term CE benefits. Parsimony in policymaking may be at odds with the complexity and urgent need for more ambitious policy solutions. There needs to be consideration of how costs (and benefits) of the transition to a CE are shared.

More ambitious policies towards a CE transition require examination of stakeholder values. Traditional environmental policy theory has promoted the role of government to address externalities (Baumol & Oates 1988). However, increasingly there is recognition of the need of government to be a strong actor in transitions (Mazzucato, 2018; Raworth, 2017). The transition to a CE also involves a transition of the role of government in capturing and creating circular value, but future research needs to consider possible ways policies and policy processes can do this.

# 6 References

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# Paper I

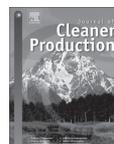






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## Extended producer responsibility for lamps in Nordic countries: best practices and challenges in closing material loops



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### ABSTRACT

Extended Producer Responsibility (EPR) schemes are adopted not only to promote collection and recycling of waste products but also to close material loops and incentivise ecodesign. These outcomes are also part of creating a more circular economy. Evaluations of best practices can inform how to further optimise systems towards more ambitious collection, recycling and recovery of both hazardous and critical materials. Gas discharge lamps in particular are a key product category in this regard, considering both the presence of mercury and of rare earth materials in this waste stream. Nordic countries in particular are known for advanced collection and recycling systems and this article compares the EPR systems for gas discharge lamps. The EPR systems for lamps are evaluated using theory-based evaluation approaches to analyse both the performance of lamp EPR systems and challenges perceived by key stakeholders. The cases were constructed based on primary and secondary literature, statistical data, and interviews with stakeholders. The findings indicate that the collection and recycling performance is generally still high for gas discharge lamps in the Nordic countries, despite some differences in approach and structure of the EPR systems, but there remain opportunities for further improvement. In terms of EPR goals, there is evidence of improved waste management of these products as a result of the systems; however, there also remain significant challenges, particularly in terms of ecodesign incentives. The key factors for best practice are discussed, including aspects of the rule base, infrastructure, and operations. The particular characteristics of this waste category, including the rapidly changing technology, also pose challenges for EPR systems in the future.

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### 1. Introduction

Energy efficient lighting is an important part of addressing climate change and transitioning towards a green economy with electricity for lighting accounting for approximately 15% of global power consumption and 5% of worldwide greenhouse gas (GHG) emissions (UNEP, 2012). Energy efficient gas discharge lamps (also known as fluorescent or mercury lamps), and now increasingly LEDs, have been gradually replacing traditional incandescent lamps for the last few decades and this trend has accelerated recently due to the tightening of energy efficiency regulations in most regions of the world (see e.g. UNEP, 2014). In Europe for example, EU Commission Regulation EC No 244/, 2009 and EU Commission

Regulation EC No 245/, 2009 introduced stricter energy efficiency requirements for lighting products and a similar approach has been adopted through energy efficiency regulations in the U.S. (UNEP, 2014). Lighting represents a key area for achieving the European Union (EU) goal to increase energy efficiency by 20% by 2020 and replacement of inefficient lighting by 2020 is expected to enable energy savings to power 11 million households a year (EU Commission, 2013). The 2009 regulations initiated a phase-out of incandescent lamps (EU Commission, 2014a) and resulted in an increase in gas discharge lamps in the EU general lighting market (accounting for an estimated 43% of units sold in 2011 and 2012 (McKinsey and Company, 2012)). A further increase of both gas discharge lamps and LEDs is expected with the phase out of halogen lamps (originally scheduled for 2016, but now delayed to 2018).

However, in transitioning to energy efficient lighting, an integrated policy approach must also consider end-of-life management of energy efficient lamps (UNEP, 2012). The WEEE Directive (EU 2002/96/EC and recast 2012/19/EU) has implemented Extended

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Producer Responsibility (EPR) for such waste in EU member states and banned landfilling of WEEE covered by the legislation. Gas discharge lamps are covered under category 5 of the WEEE Directive. As a product group, they have special characteristics that make them particularly challenging for collection and recycling. They contain mercury that can be detrimental when released into the environment in large enough quantities (Wagner, 2011) or result in high mercury emissions when incinerated without adequate filter technology (Silveira and Chang, 2011). The fragility of lamps makes safe collection and transportation more complex to ensure the health of handlers (Kasser and Savi, 2013; Sander et al., 2013). Avoiding this environmental harm from waste gas discharge lamps is a compelling reason for “collecting as much as possible and in a safe way (avoidance of breaking) and to treat them properly” (Huisman et al., 2008, p. 281). However, collection and recycling of gas discharge lamps represents relatively high cost compared to the value of the product (Philips Lighting, 2012) and the low or negative value of the recovered material from lamp waste (G. Lundholm, personal communication, 13 August 2014). While clearly it is of societal value to avoid mercury contamination, this is a positive externality and moreover, it is a benefit difficult to quantify in economic terms.<sup>1</sup> As such, legislation, targets and other drivers are integral to incentivising end-of-life management (Huisman et al., 2008; G. Lundholm, personal communication, 13 August 2014). The high cost for lamps is tied to necessary recovery of hazardous materials increasing recycling costs, but also to challenges in collecting lamps. Lamps are lightweight, which means they are a small part of total WEEE and that filling trucks for optimal transportation can be an issue. Lamps are also dispersed in high quantities, geographically and between consumers and businesses. This necessitates the need for an extensive capillary network for collection.

The collection and recycling of gas discharge lamps can also create opportunities to recycle valuable materials. Waste gas discharge lamps contain rare earth elements (REE) in the phosphor layer, which is necessary for producing white light. Nearly all global supply of europium, 85.2% of terbium and 76.7% of yttrium is used for phosphors, and the majority of these are used for lighting applications (Moss et al., 2013; Tan et al., 2014). Despite only using 7% of global REE by volume, due to the high level of purity needed for lighting applications, phosphors represent 32% of the value for rare earth applications (Binnemans et al., 2013; Schüller et al., 2011; U.S. Department of Energy, 2011). The EU Commission's report on Critical Raw Materials for the European Union (EU Commission, 2014b), considers the REE group as having the highest supply risk and REE have received increasing attention in the last few years with rising prices and concern about supply restrictions from China, where over 90% of production takes place (Binnemans et al., 2013; Bloomberg News, 2015). The presence of REE in only small amounts in waste products represents a challenge for recycling, but increased recycling has the potential to address supply risks (Binnemans et al., 2013; Rademaker et al., 2013; Sprecher et al., 2014). However, currently less than 1% of REE is recycled and examples of closing this material loop are rare (Binnemans et al., 2013) but the experience in recycling REE from gas discharge lamps is promising (Dupont and Binnemans, 2015).

EPR systems for lamps have been in place in the EU under the WEEE Directive, but legislation has been present even longer in some countries, like Norway, Sweden, and Austria. Academic literature has evaluated various aspects of WEEE systems in the EU,

including the challenges for collecting small WEEE (Huisman et al., 2008; Khetriwal et al., 2011; Melissen, 2006) However, there has been not been a comprehensive evaluation of the best practices and challenges for end-of-life management of gas discharge lamps specifically, despite this product stream having been acknowledged to be of particular relevance both for recovery of critical materials and for avoidance of mercury contamination. The literature that has addressed this waste stream has tended to focus on the set up of EPR systems for lamps in the EU in general (Wagner, 2011, 2013; Wagner et al., 2013) or has emphasised recycling over collection aspects (Silveira and Chang, 2011). Very little is known about how EPR systems for lamps compare or differ from the structure and performance of the overall WEEE systems.

The research presented in this paper evaluates EPR systems for lamps in the Nordic countries of Denmark, Finland, Norway and Sweden.<sup>2</sup> The Nordic countries have been recognised for best practices in the area of end-of-life management of WEEE (Román, 2012; Ylä-Mella et al., 2014a,b) and as such also provide good cases for a deeper analysis of EPR for lamps in particular. Such analysis can provide further insight into how to address the unique challenges for this waste stream and the factors that potentially contribute to better attainment of EPR goals and a more circular economy for this key product category. EPR includes goals to conserve source materials by promoting better waste management, ecodesign, and closing material loops and such goals are also an integral part of a circular economy (EU Commission, 2014c). This article presents analyses of EPR systems for lamps in Nordic countries in relation to EPR goals and discusses the factors that contribute to well-functioning systems as well as challenges still to be addressed in further optimising such systems.

Section 2 describes the methodology used in this policy evaluation and comparative case study methodology. Section 3 presents the findings of the comparative case study and evaluation of the performance of the Nordic EPR systems in relation to the EPR outcomes. Section 4 discusses these findings and presents factors identified as influential to the success of the systems as well as remaining challenges.

## 2. Methodology

The research approach used embedded multiple cases in which multi-level perspectives were explored simultaneously (e.g. gas discharge lamps, country perspectives, key stakeholder groups, etc.) (Yin, 2003). Comparative analysis of multiple cases particularly suits research evaluating multiple holistic systems and allows comparison of factors influencing performance (Druckman, 2005). The framework for the initial comparison of the EPR systems for lamps was based on important elements of such systems identified by Murphy et al. (2012). Nordic countries are the focus cases in evaluating EPR systems for lamps because they have been described for their best practices in performance for WEEE in general, but they have not been examined in regard to gas discharge lamps. High performing systems can be studied to identify the common elements that could be the key to their effectiveness. It can also reveal context-specific or organisational differences that have or have not influenced effectiveness, as well as challenges perceived about the different systems from corresponding stakeholder groups in each system.

<sup>1</sup> Some studies, for example, Hylander and Goodsite (2006) have tried and estimated a cost of USD 2500 to 1.1 million per kg Hg isolated from the biosphere depending on local factors quantity, nature of pollution, media, geography, technology used etc.

<sup>2</sup> Iceland has been excluded in this research as its context as well as the implementation and experience thus far with WEEE systems has been quite different than other Nordic countries so far. It is expected to further develop and resemble other Nordic country systems in the future (Baxter et al., 2014).

Policy evaluation, using multiple methods of inquiry to generate policy-relevant information that can be utilised to resolve policy problems (Dunn, 1981), framed this research. In terms of focus criteria, the WEEE legislation in regard to gas discharge lamps in the Nordic countries is evaluated primarily for its environmental effectiveness, a common criterion evaluating the policy in relation to its goals (Mickwitz, 2003; Vedung, 2008). While there is data related to collection and recycling rates, more comprehensive information about EPR systems for energy efficient waste lamps is still lacking. Moreover, the goals of the WEEE Directive and the legislation transposed in the member states refer to WEEE collection overall, with few product level specifications. A separate target for gas discharge lamps within the Directive is being investigated until August 2015 (Article 7.6). In such cases where the data or explicit goals may be lacking, the use of intervention theories can support the evaluation of the policy (Kautto and Similä, 2005; Manomaivibool, 2008).

The main policy interventions governing the end-of-life management of gas discharge lamps in the Nordic countries are based on the principle of EPR, defined as “a policy principle to promote total life cycle environmental improvements of product systems by extending the responsibilities of the manufacturer of the product to various parts of the entire life cycle of the product, and especially to take-back, recycling and final disposal of the product” (Lindhqvist, 2000, p. 154). Moreover, Lindhqvist (2000) argues that EPR entails different types of responsibilities: liability, physical, financial, and provision of information (i.e. informative) responsibilities. Policy mixes can vary in how these responsibilities are realised and distributed amongst actors but there are specific goals and outcomes of EPR that should be common to all EPR programmes. These have been outlined by Tojo (2004) and are shown below in relation to the WEEE Directive 2012/19/EU. While the WEEE Directive is the main focus of this article, it is also acknowledged that the Restriction on Hazardous Substances in EEE (RoHS) Directive is part of the EU’s EPR policy package (van Rossem et al., 2006a). The RoHS Directive’s influence on design for lamps is also discussed in Section 3.2.1. The EU Ecodesign directive also has an indirect effect on EPR policies (OECD, 2014).

Theory based (also known as program theory/theory-driven) evaluation includes reconstruction of the intervention (program) theory to model how a policy is supposed to function (Bickman, 1987). Using an intervention theory as a basis for environmental evaluations focusses the evaluation in terms of scale and stakeholders (Mickwitz, 2003). Hansen and Vedung (2010) propose that an intervention theory consists of three elements: a situation theory concerning the context of the intervention; a causal theory concerning the implementation and outputs that lead to certain impacts of the intervention; and a normative theory concerning the envisioned outcomes of the intervention. This study includes these elements with the context, implementation and outcomes of the intervention all examined.

In addition, theory based evaluations are grounded in a stakeholder approach (Hansen and Vedung, 2010), but it is a recognised challenge that there can exist competing program theories (Dahler-Larsen, 2001). When dealing with more complex program evaluations, Hansen and Vedung (2010) suggest a “theory-based stakeholder evaluation” that elaborates upon a “raw” intervention theory with the perspectives of key stakeholders. Identifying key stakeholders stems from the intervention theory and from this the primary stakeholders crucial to its implementation and likely to have in-depth knowledge of the intervention are selected. The intervention theories from the perspective of these key stakeholders can then be reconstructed to identify similarities, differences, and disagreements (Hansen and Vedung, 2010) or the distinction between the “espoused theory” and the “theory-in-use”

(Friedman, 2001). The latter distinction is included in this paper while stakeholder perspectives of success factors and continuing challenges for EPR systems are discussed.

Both the evaluation and cases used data collected from publicly available statistics from Eurostat, national authorities, and producer responsibility and municipal waste organisation reports. This data was supplemented and triangulated with peer-reviewed and grey literature as well as semi-structured interviews with key stakeholders and additional email correspondence (based on interview protocols). For each country case, similar stakeholders were interviewed with identical protocols. When possible, interviews were recorded and in person, though they were also conducted by telephone. Extensive notes were taken and when necessary, clarified again with the interviewed stakeholder via email correspondence. Lighting producers themselves were not interviewed as earlier research has examined EPR from the perspective of lamp and lighting sector producers (see Gottberg et al., 2006). The focus of this study is instead on stakeholders downstream from producers involved in the practical implementation of the EPR systems for lamps. These stakeholders included managers of producer responsibility organisations (PROs) in each country dealing with lamp collection, lamp recyclers responsible for recycling lamps in Nordic countries, and managers of WEEE issues in national waste management associations representing municipalities and municipal waste management companies in each country. In addition, a few specific Nordic retailers and municipal waste management companies with initiatives for lamp collection were also interviewed. A list of organisations and representatives interviewed is included in the appendix as well as sample interview protocols. Where specific information from an interview is presented, the interviewed person is identified, but where there was general consensus amongst a group of interviewed stakeholders, the group is identified.

### 3. Findings and analysis

It has been demonstrated and generally accepted that end-of-life management of WEEE is environmentally beneficial and benefits can be better realised through increased collection and recycling rates (Hischier et al., 2005; Khatriwal et al., 2011). In the first version of the WEEE directive collection rates differed widely between member states, with ten countries failing to meet the 4 kg per capita target in 2010 but most exceeding and the Nordic countries well exceeding the target (EU Commission, 2013). Ylä-Mella et al. (2014a) and Román (2012) describe the performance of WEEE systems in the Nordic countries as exemplary, citing their high collection rates in Nordic countries (ranging from 8 kg/capita/year in Finland to over 20 kg/capita/year in Norway) despite low population densities and high transport distances, especially in the northern parts of Norway, Sweden, and Finland. Such per capita collection rates rank Nordic countries all in the top five performing countries in Europe. Aside from system architecture, Ylä-Mella et al. (2014a) attribute the success of the Nordic WEEE systems in part to high awareness of environmental issues among Nordic citizens and further argue that one of the strengths of the WEEE recovery systems in Nordic countries is the strong civic support of environmental protection and willingness to use the WEEE systems in place.

While this measure of performance has been consistent with historic WEEE Directive targets measuring performance in terms of kilograms per capita, the WEEE recast brings new targets which measure collection rates in comparison to product put on market in the previous three years. In the recast the target is 45% of the sales of products in the three preceding years with an increase to 65% by 2019 (or 85% of generated WEEE). This has implications for Nordic countries where there is a high level of EEE products put on the market, reflecting both the challenging climate conditions and high

living standards that make EEE and information technology an important part of everyday life in Nordic societies (Ylä-Mella et al., 2014a). Despite this, according to Eurostat statistics, Denmark, Norway and Sweden remain in the top five performing countries and are already poised to meet the 45% collection target of previous three years EEE put on market, which is in place from 2016 to 2019. Sweden is already meeting the 65% target that will be in place from 2019. However, Finland, having collected only 36% in 2012 compared to the previous 3 years EEE put on market, still has improvements to make to meet this target. However, it has also been suggested that Finland's lower figures have more to do with collection reporting rather than actual collection being low (see Baxter et al., 2014). Another important change with the recast of the WEEE Directive has been the increased responsibility for retailers and this is examined in further detail in relation to the specific cases.

### 3.1. Comparing Nordic country cases

In our analysis, we compare the systems for gas discharge product group specifically, though of course the overall WEEE design has a large influence on how this waste category is collected. As described earlier, EPR consists of financial, informative, and physical responsibility for waste products and these responsibilities can be allocated differently in different systems. Table 1 below outlines the basic components and context of the WEEE systems for lamps in the Nordic countries.

**Table 1**  
Comparison of EPR for lamp systems in Nordic countries.

		Denmark	Finland	Norway	Sweden
Context	Population 2013 (mil)	5.6	5.4	5.1	9.6
	Area (km <sup>2</sup> )	43,094	338,424	385,178	449,964
WEEE/lamp legislation beginning		2005 <sup>a</sup>	2004	1998	2001/2000
System architecture	Legislated responsibility (italics responsibility in practice)	Lamp scope legislation	Filament bulbs excluded <sup>b</sup>	Filament bulbs excluded <sup>b</sup>	All lamps covered
	Physical responsibility	Producer/municipality <sup>c</sup>	Producer/municipality/retailer	Producer/municipality/retailer	Producer/Municipality/Retailer
	Informative responsibility	Producer/municipality	Municipality	Producer/recycler/municipality/retailer	Producer/municipality
	Financial responsibility	Producer/municipality	Producer Municipality (part)	Producer Municipality (part)	Producer Municipality (part)
	Retailer take-back	Voluntary	1:1; 0:1 ( $\geq 1000$ m <sup>2</sup> grocery stores/200 m <sup>2</sup> EEE)	All selling EEE	1:1; 0:1 ( $\geq 400$ m <sup>2</sup> EEE sales space)
	Recycling stations	Yes	Yes	Yes	Yes
Kerbside collection		Limited (2 municipalities)	Mobile collection a few times/year	Mobile collection a few times/year	1.5 million households
Permanent collection sites 2013		398	526 <sup>d</sup>	~2700	~2600
Main PROs dealing with lamps		LWF (lamp specific)	FLIP (lamp specific); Elker Oy	RENAS, Elretur	El Kretsen
Collection and recycling	Avg. tonnes Put on Market (POM) 2009–2011	1670	1926	3018	3203
	Collected tonnes 2012	706	850	890	2165
	% Recycled of collected 2012	93%	90.1%	92.7%	100%
	2012 collected/avg. 2009–2011 POM	42%	44%	29%	68%
	kg per Capita collection 2012	0.126	0.157	0.177	0.227

<sup>a</sup> Since 1998 the Danish Environmental Protection Act included a section about ecodesign for producers.

<sup>b</sup> Inevitably some filament bulbs are collected and recycled with gas discharge and LED lamps.

<sup>c</sup> Municipalities have responsibility for collection from households only, while producers are responsible only for collection from municipal collection to recycling (not from households directly).

<sup>d</sup> Does not include retailer collection locations which were implemented in 2013.

Sources: Dansk Producent Ansvar, 2015; Elker Oy, 2014; El Kretsen, 2014; Elretur, 2014; Eurostat, 2014; RENAS, 2014; "Danish WEEE legislation", 2014; "Swedish WEEE legislation", 2014; "Norwegian WEEE legislation", 2015; "Finnish WEEE Legislation", 2014; personal communication with the following organisations: Dansk Affaldsforening; Avfall Sverige; Avfall Norge; JLY Finland (see interview information in Appendix A).

### 3.1.1. System architecture

With the exception of Denmark, each Nordic country has transposed the WEEE Directive with the financial responsibility for collection, transportation, and treatment being the responsibility of producers. In Denmark, municipalities are currently financially responsible for collection of WEEE from households and cover this cost by fees charged to households. Physical responsibility has been extended to retailers in the recast of the legislation in Finland and Sweden and was already part of the responsibility in Norway prior to the recast. In practice, municipalities in all Nordic countries are responsible for most of the household collection of WEEE, including gas discharge lamps. Municipal waste organisations and municipal stakeholders interviewed in these countries reported that financial compensation for municipal collection of WEEE did not cover the full costs of the services provided by the municipalities. The financial compensation in Sweden is negotiated as a contractual arrangement every few years between municipalities and the main producer responsibility organisation, El Kretsen. In Norway and Finland, contracts are negotiated between individual municipalities and individual PROs. As such, the individual arrangements often reflect the negotiating power of the municipality (i.e. in larger urban areas there are often other waste service providers who can compete with the municipalities and thus these municipalities often receive less compensation for their services than rural municipalities). In Denmark, though municipal waste organisations have requested financial compensation for collecting WEEE, they

have so far been unsuccessful in this endeavour and do not foresee any changes in the near future due to a recent agreement between the government and industry regarding ecodesign (N. Remtoft, personal communication, 15 December 2014).

In all Nordic countries, producers are solely responsible for transport and treatment of the waste lamps collected by municipalities and retailers, though the exact details of the financial and physical responsibility for transport of lamps from retailers in Sweden remains to be seen with this aspect remaining vague in the recast legislation. Annex V of the WEEE Directive specifies a target of 80% of collected gas discharge lamps to be recycled and Annex VII specifies that treatment should include removal of mercury.

The duty to provide information to consumers about the WEEE system for lamps is distributed differently in the Nordic countries, with different emphasis on the roles of PROs, municipalities, and retailers. PROs interviewed generally felt that adequate information was being provided while municipal organisations were more likely to acknowledge that this was an area that could still be improved. While consumer knowledge about WEEE in general was perceived as high, there were different perceptions about consumer awareness of disposal requirements and environmental impact of waste discharge lamps in particular. In Sweden, lamps were specifically targeted in information campaigns by the main PRO (El Kretsen) and the national waste management association (Avfall Sverige). In Denmark, the provision of this information was seen to be more the responsibility of the lamp PRO, and it did run awareness campaigns every few years. In Norway, the national waste management association (Avfall Norge) began an awareness campaign for small WEEE, including lamps in 2014. In Finland there have not been lamp-specific campaigns, and better information provision, particularly from retailers with new responsibilities under the recast, was seen as an area for improvement.

The organisation of PROs also differs between the Nordic countries. Lamp-specific PROs, like those found in Denmark and Finland, were initiated by the lamp producers who were aware that they were putting a product that contained a hazardous substance on the market and who wanted to ensure the hazards were managed properly at the end-of-life phase for these products and thus not jeopardise market acceptance of these products. Larger umbrella PROs run the risk of having decisions dominated by other waste streams and not ensuring the interests of lamp producers (J. Bielefeldt, personal communication, 26 August 2014). Examining the boards of larger PROs in Norway, it is the case that there is no representation by lighting producers or organisations on the boards of two largest PROs handling lamps and luminaries (see RENAS, 2014; Elretur, 2014).

The competing nature of PROs in Norway has resulted in general issues with collection of WEEE with incidences of PROs refusing to collect from municipalities once they had reached their targets, requiring intervention from authorities. This situation has improved, but the lack of a clearinghouse structure in Norway remains a perceived challenge (E. Halaas, personal communication, 9 December 2014). Lamp-specific PROs and national waste management associations reported more cooperation than competition amongst the several PROs in Finland and Denmark and perceived this as strengths of the systems.

In Sweden, a representative of the lighting industry is a present on the board of the largest PRO, El Kretsen, though the lighting association is only one of over twenty owning industry associations (El Kretsen, 2014). Environmental management of waste gas discharge lamps has also been given priority in Sweden the past few years by Swedish Environment Minister Lena Ek, who has pushed for increased collection of this waste stream from 2011 when meeting with El Kretsen and the national waste management association, Avfall Sverige, about improvements to lamp collection

(Pehrson and Balksjö, 2011, 2012; Von Schultz, 2013). This led to a pledge to increase lamp collection by 2 million pieces in 2013 and an information campaign focussed on lamps from households (Avfall Sverige, 2013). In response to this pressure for increased collection of lamps as well as other small WEEE, El Kretsen also initiated a project to make collection of lamps even more convenient with in-store “Collectors” (“Samlaren” in Swedish). The Collectors are closed cabinets positioned most often next to reverse vending machines for beverage packaging in grocery stores. The pilot program with them in Gothenburg, Sweden, was deemed a success. At 14–20 SEK/kg (1.5–2.1 Euro/kg) the Collectors were found to be more expensive than other forms of collection but became more cost effective with time as consumers became more aware of this option and collection increased (El Kretsen and Sörab, 2011). Collectors are currently being deployed first in major cities and increasingly in municipalities throughout southern Sweden where over 60 Collectors have been placed in grocery stores in 2014 and early 2015. The initiative is being led by municipal waste companies and is partially financed by producer compensation to municipalities for collection of WEEE (A. Persson, personal communication, 9 September 2014).

### 3.1.2. Collection and recycling performance

The general WEEE system architectures in the Nordic countries are described as best examples and perform well in relation to the WEEE Directive goals (Román, 2012; Ylä-Mella et al., 2014a). The general architecture also encourages high performance in the category of gas discharge lamps with the Nordic countries among the top five in Europe in 2012 (Fig. 2) when measuring collection in terms of kilograms per capita.

However, when considering the collection rate compared to the amount of gas discharge lamps put on market, a different situation is found. Nordic countries performed better than the overall EU average of 37% in 2012 (see Table 1), with the exception of Norway. It should be noted that statistics for this product category are highly variable for countries with small amounts of gas discharge lamps recorded (for example, Eastern European countries). When countries with larger lighting markets are compared (Fig. 3), Sweden,

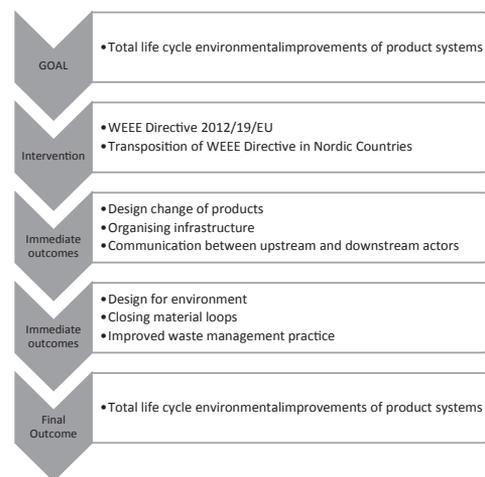
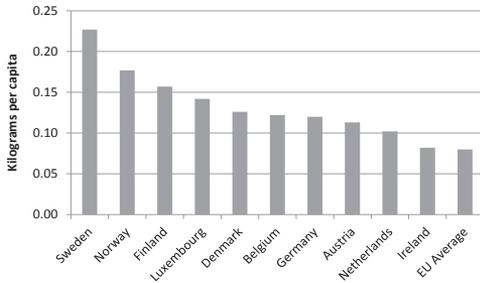


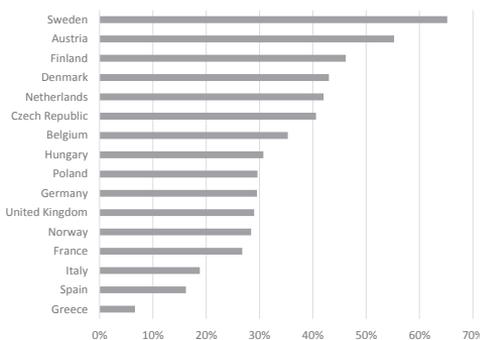
Fig. 1. Simplified intervention theory for EPR programmes and specifically the WEEE Directive, based on Tojo (2004).



**Fig. 2.** Top 10 performing European countries, kg per capita collection of gas discharge lamps. Source: Eurostat, 2014.

Denmark and Finland show consistent collection rates that compare well with other countries and again indicate advantages to the WEEE systems in these countries. The same cannot be said for Norway, for which statistics indicate a consistently lower performance than the EU average. In terms of recycling, all four countries have high treatment rates for the collected lamps, exceeding the minimum 80% recycling in the WEEE Directive (see Table 1). Additionally, all Nordic countries comply with the requirement to remove the mercury in recycling process for gas discharge lamps.

In the absence of specific information about a possible target for the collection of lamps under the WEEE Directive, it is difficult to gauge how Nordic countries will perform if one is introduced after the review in 2015. In relative terms to other countries though, it can be anticipated that Nordic countries are well-positioned to meet such a target, though Norway may need to improve if the target takes into account put on market data for collection rather than weight per capita. However, regardless of any specific targets, increasing the collection and recycling of gas discharge lamps results in environmental benefits that should make continuous improvement of collection and recycling a goal.



**Fig. 3.** Collection % for countries with market over 1000 tonnes based on 2010–2012 average collection % of gas discharge lamps (GDLs) compared to put on market 2007–2011 based on Eurostat (2014). Note: Netherlands data estimated based on 2012 (tonnes) Eurostat (2014) put on market data and Huisman et al. (2012) estimates of per capita lamps put on market 2010. Collection % from 3 years (2010–2012) were averaged to account for higher variability when looking at this product category. Note that GDL data in practice often contains LEDs and other light sources and can be deemed an estimate only.

### 3.2. EPR outcomes for energy efficient lamps in Nordic countries

In general, EPR interventions should produce three intermediate outcomes that lead to the policy goal of total life cycle environmental improvements of product systems (Fig. 1 and Tojo, 2004): 1) design for environment, 2) closing material loops and 3) improved waste management practice. The performance of the Nordic EPR systems for lamps is considered in light of these outcomes.

#### 3.2.1. Design for the environment

Interestingly, gas discharge lamps are one of the only EEE product categories whose lifespan has increased in recent years (Bakker et al., 2014). Additionally, levels of mercury in gas discharge lamps have also decreased and LED technology now becoming more competitive can eliminate mercury altogether in new energy efficient lamps. These developments have significant implications for the end-of-life impact of energy efficient lighting products. In some cases, such developments are likely also to have been motivated by other EPR-related legislation, for example the Restriction on Hazardous Substances (RoHS) Directive which limits mercury content. In other cases factors beyond EPR are likely also influential, for example, the Ecodesign Directive phasing out less efficient light sources, competitive technology development, company culture, etc.

Earlier research by Gottberg et al. (2006) explored the impact of EPR legislation in the lighting sector, including several Swedish producers, and found little evidence of ecodesign in response to the financial responsibility of EPR. Despite initial concerns by lighting producers about the costs of EPR legislation being higher than relative to the product price (Philips Lighting, 2012), lighting products are also characterised by inelastic demand that has allowed producers to more easily pass on compliance costs to consumers. The cost of EPR compliance depends at which point this cost is being considered. EPR compliance costs have been found in some cases to be a small percentage in relation to total product costs and in others quite high. Despite the wide range, Gottberg et al. (2006) argued that the cost of EPR was a small economic driver for ecodesign changes in relation to other product requirements. In all Nordic systems, undifferentiated fees (fixed in Sweden, but by market share in the other countries) are faced by all producers and this also gives little financial incentive or comparative advantage for improving products. For example, there is no differentiation among the producer responsibility organisations in the fees charged for LEDs in comparison to gas discharge lamps, despite the presence of mercury only in the latter. One challenge to doing this is the reality that LEDs and gas discharge lamps in Nordic countries are collected, transported, and treated together so they incur the same costs, though it is unclear whether LEDs, if separated, could be recycled in a more cost efficient process. LEDs do not contain mercury, but do contain some hazardous materials such as lead (see Lim et al., 2013). Another concern with differentiation expressed by PROs interviewed is that if LEDs were differentiated that treatment for gas discharge lamps would be left underfunded.

In their research Gottberg et al. (2006) consider EPR mainly as an economic instrument and only the financial responsibility as a motivation for product design improvements. However, EPR is also about information flows between consumers, recyclers, and producers. Interviewed producer responsibility organisations and recyclers for lamps reported different levels of communication with producers about the end-of-life attributes of their products. In cases where the recycler or producer organisation had information to provide in this regard, it was reported that the contacts with the producers were generally not in the design department, which was often located in another country. Such anecdotal evidence indicates a possible prerequisite for design change may be missing; namely,

communication between upstream and downstream participants may not be taking place in a way that facilitates relevant information from downstream reaching those working with producers who have an influence over design decisions. However, even if this information does reach product designers, its usefulness may be limited due to the (increasingly) long life of lighting products. Indeed, other drivers including market competition and company culture were found to also be able to explain design improvements in the lighting sector and causation to EPR legislation alone could not be established (Gottberg et al., 2006). This is not surprising given the challenges for design incentives for lamps and these are further discussed in Section 4.2.

### 3.2.2. Closing material loops

In theory, almost all the material from gas discharge lamps can be recycled and some components even re-used, for example the glass tubes if using an end-cut method (Nordic Recycling, 2014) or phosphor coating if reused by the same type of lamp and manufacturer (Binnemans et al., 2013). Table 2 illustrates the possible end uses or disposal options for fraction from gas discharge recycling processes; however the actual end use of fractions is highly context specific.

In practice, materials from the recycling process in Nordic countries are not used again in the production of new lamps. Currently, most waste lamps in Nordic countries are shredded together in a wet process (as opposed to the end cut method, for example) (Nordic Recycling, 2014). In Finland, collected lamps are recycled at one location in Finland (Ekokem, 2014). PROs in Norway and Sweden (and at the time of writing, also Denmark) send waste lamps to be recycled in one location in central Sweden. While this arrangement helps to increase economies of scale in treatment, the recycler faces challenges in returning glass and other materials long distances to lamp manufacturers and this is part of the reason these materials are not recycled in a closed loop.

It is also difficult to transport the glass fractions long distances to glass recyclers in Sweden and Europe as the cost for the transportation will decrease profit. For this reason, much of the glass is currently used as construction material in landfill cover; though higher level alternative uses are being actively sought (G. Lundholm, personal communication, 26 October 2014). The lamp PRO (LWF) in Denmark had been sending crushed lamps for recycling in Germany where more fractions could be used for new lamps, but the recycler has since closed, forcing it to use the same recycler as PROs in Norway and Sweden (but in a new tender process at the time of writing). In Finland the glass fraction is delivered to a nearby glass recycler who can use it to produce foam glass, as well as glass powder (Uusioaines Oy, 2014).

Other fractions, such as the metal, are easily sold and used by local metal recyclers. The small fraction of plastics is generally incinerated in the Nordic countries. In many EU countries the mercury containing phosphor layer is landfilled or stored in salt mines rather than recycled (Solvay, 2014). Solvay Rhodia in France

began the first commercial scale recycling of lamp phosphors, separating rare earth oxides for use in new phosphor powders in 2011 (Walter, 2011). It buys fractions from recyclers based on the amount of rare earth material and deducting for the amount of mercury, glass, and other impurities. The recycling process used for Swedish, Danish, and Norwegian lamps produces a phosphor fraction of high enough quality that it can be sold for this recycling. Though not at a large profit, this further recycling also avoids the cost of hazardous landfill. This is made possible both by the recycling process and the scale of the centralised treatment. By contrast the Finnish recyclers have studied the use of phosphor but it is currently produced in such small quantities, and in a less useful form, that it does not make sense to recycle the phosphors (J. Koskinen, 29 January 2015, personal communication).

### 3.2.3. Improved waste management practice

The collection and recycling of gas discharge lamps represents a significant improvement in waste management practice compared to a situation where there is no legislation or policy for collection and recycling. Even before EPR legislation, the mercury present in gas discharge lamps did make them a concern in countries like Sweden. Voluntary programs for collection and recycling were set up in Sweden, mainly for business end-users (who were the majority of the users in the early stages of the technology). Between 1993 and 1998 the collection rates for gas discharge lamps in Sweden was roughly estimated between 10 and 25% and this was perceived as inadequate in light of the risks of mercury emissions associated with the waste products (Kemikalieinspektionen, 1998). OECD countries with some waste legislation or voluntary programs, but lacking mandatory EPR legislation, also have very low collection and recycling rates of lamps. For example, it is estimated that 95% of fluorescent lamps in Australia are landfilled (Lighting Council Australia, 2014), while Canada, Japan, and Mexico are estimated to collect and recycle less than 10% of waste lamps (EU Commission, 2008). The United States has some, mainly state level, legislation for management of waste lamps, focussed on end user (primarily business) responsibility. However, enforcement is low and the collection and recycling rate is estimated around 23% (Silveira and Chang, 2011).

EPR systems in Nordic countries continue to evolve, with Finland and Sweden using the recast to include new retailer take back options for consumers. Increasing the collection of small WEEE in particular requires increasing attention to factors which influence recycling behaviour, for example motivation, convenience and capacity and the available recycling infrastructure can influence all three of these (Melissen, 2006; Wagner, 2013). Using more retailers to take back waste lamps regardless of purchase (prior, retailers were required to take back a product if an equivalent product was purchased) is a way to further increase the number of convenient return options for household consumers. Such retailer take-back has been successful at the municipal level in the U.S. (where other recycling options for households are not

**Table 2**  
Fractions and end uses from waste gas discharge lamps.

Fractions	Possible part (compact fluorescent – fluorescent tube)	End use/disposal
Aluminium/other metals	18–30%	Reused or recycled
Mix of plastic and metal	20%	Recycling; energy recovery; landfill
Glass	45–80%	Reused for fluorescent tubes; lamp glass; glazing; glass wool insulation; fusion agent with black copper foundry; abrasive sand for cleaning, under layer for asphalt; sand replacement; silicon substitute, landfill cover
Rare earth powder, also containing mercury and small glass particles	2–3%	Separated and reused as mercury or phosphors in new lamps, separated and recycled after rare earth processing; powder and Hg landfilled as hazardous waste

Sources: Nordic Recycling, 2014; WEEE Forum, 2011.

provided), achieving recycling rates of over 36% from near 0% previously (Wagner et al., 2013). However, because of the existence of established and better known recycling centres in municipalities in Nordic countries, the impact of retailer take back is anticipated by some stakeholders to have a small, but still positive, impact on collection of lamps. In Denmark and Sweden there was also evidence of municipalities collecting waste lamps through kerbside collection for detached households through plastic bags or boxes attached to the top of kerbside recycling bins. While this type of kerbside collection is relatively new and effectiveness has yet to be fully assessed, the initiatives represent attempts to further optimise collection of this waste stream. Another form of kerbside collection, collection small bins in apartment complexes, has been more established in these countries, as is mobile collection from households a few times year.

There were mixed views on whether more market oversight was necessary or whether enforcement was adequate in all countries. In the Nordic countries market enforcement is undertaken by typically small authorities (in terms of resources devoted to enforcement of WEEE legislation) and takes the form primarily of guidance about rules and response in the cases of complaints. Interviewed stakeholders perceived that high levels of cooperation amongst PROs and municipalities were part of why general WEEE systems performed well in the Nordic countries. While there were some concerns about free-riders in the systems, this was not perceived to be a major inhibitor of the function of the system, but rather an area where the system could still be optimised, but requiring greater resources than currently available.

#### 4. Best practices and remaining challenges

##### 4.1. Factors in best practice

In contrast to other waste streams, lamps are small, meaning they can be easily disposed of in residual waste, and represent a net cost to collect and recycle, meaning there is no natural economic incentive in absence of legislation (Huisman et al., 2008). Mandatory EPR legislation for lamps is it appears key for higher collection and recycling of this product group. However, the fact that collection and recycling rates in the EU member states and even amongst the Nordic countries also vary indicates that having the legislation, or a rule base, itself is not enough for excellent collection and recycling rates. From the analysis of the Nordic systems, we identified several common factors that contribute to excellence in operational performance (Fig. 4).

Building on a robust and transparent rule base, the system infrastructure is also essential. Enforcement of the rules needs to be adequate to allow focus on continuous improvement rather than

incentivising a focus on lowest costs by avoiding compliance. As is seen in the Nordic cases, the strength and resources devoted to the authorities can be fewer in a situation with high compliance and cooperation. Such voluntary action on the part of actors is key, particularly in areas where the rule base is vague. For example, sound financial management is stipulated by the WEEE Directive (Article 12) but how producers and PROs incorporate end-of-life costs is still open to interpretation (Article 12.6 invites the Commission to report “on the possibility of developing criteria to incorporate the real end-of-life costs into the financing of WEEE by producers...”). With the requirement for a financial guarantee waived in most Nordic countries with the participation in a collective scheme (i.e. a PRO with a sufficient number of members to guarantee financing), the financial stability of the collection system rests upon the financial management of these PROs. Whether the arrangements are adequate remains to be seen and tested with more experience. The recycling technology used in the Nordic countries ensures significant mercury emissions are avoided. In addition, despite being small markets on their own, the high level of collection and recycling of these lamps in Nordic countries, the recycling technology to produce powder fractions, and the development of Solvay Rhodia’s capacity to utilise these powders, has made recycling of rare earths from waste lamps a reality. In view of the criticality of rare earths (Koninklijke Philips Electronics N.V., 2011; Moss et al., 2013), this development in closing the rare earth loop from lamps is a significant contribution to a more circular economy in the EU.

Information provision ensures that key actors in the EPR system architecture know their role. It is also the basis for continually improving the system. In Nordic countries, a variety of actors engage with information provision to consumers through a variety of media. While the high collection rates could be indicative of the effectiveness of information campaigns, this is unclear in the case of Norway. The high visibility of waste lamps in the media due to the attention of the Minister for the Environment in Sweden may have been just as effective as the subsequent information campaign from the PROs and waste management organisations. The actual level of awareness and responding behaviour of households in the Nordic countries remains an area for further study.

In terms of the collection system in place in the Nordic countries, it can be seen that there has been a concerted effort to provide multiple means of taking back products and this continues to evolve with retailer-takeback and kerbside collection. Such options further increase the convenience of services offered to households, which in turn are particularly key aspects for optimising collection systems for small WEEE like lamps (Melissen, 2006; Wagner, 2013).

##### 4.2. Remaining challenges

The experience with EPR systems in the Nordic countries reveals well-performing systems, however, with the exception of Sweden, not as dominant as for WEEE in general. The general collection of lamps compared to some other categories of WEEE is consistent with the challenges identified with lamp and small WEEE collection in general. Small WEEE is more easily disposed into other waste streams, and there is some evidence of this still happening, particularly in the general glass recycling and residual waste (see e.g. El Kretsen and Sorab, 2011; Elretur, 2012; Pehrson and Balcksjö, 2012). However, the small documented amounts in these streams indicate that knowledge is still missing about how consumers deal with lamps at the end-of-life (for example, they are also small enough to be stored and not ending up in any waste stream for several years). This was noted as a continuing challenge by interviewed stakeholders in all four countries.

Obtaining accurate and useful data for measuring and comparing collection rates remains a significant challenge.

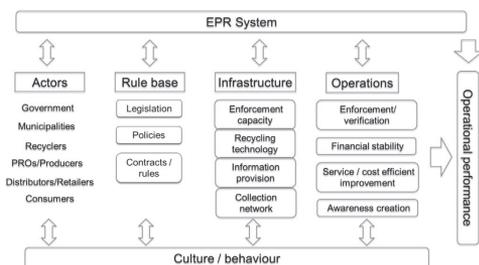


Fig. 4. Common factors contributing to excellence in operation performance of an EPR system.

Producers are required in some countries to report based on amounts (C. Andersson, personal communication, 13 May 2015), which are then converted into kilograms for reporting at the EU level, which in turn leaves room for error and inconsistency. This is particularly the case regarding put on market data, which also utilise the combined nomenclature (CN) codes used for trading and customs. For lighting products these codes are quite general (Wang et al., 2012) and do not align with WEEE product categories. It does not help that lighting technology is also changing at a rapid pace, faster than codes which explains why LEDs can be classified under different CN codes and with which the distinction between lamps and luminaires becomes less obvious (LightingEurope, 2014). With this complexity comes the risk that put on market data can be multiplied though double-counting or codes used erroneously. Additionally, lag times resulting from consumers delaying disposal of waste lamp products could affect the collection data. Also, as lifetimes of lamp products have extended, the three year average from put on market may not be the most relevant measure of collection effectiveness. It has been proposed that at least 6 years is a more accurate measure of the historic collection rate (European Lighting Companies' Federation, 2003). Even if this change was made, it would be a few more years before there is adequate data to measure this robustly (Sander et al., 2013).

Despite the reasons for making collection and recycling of gas lamps a priority, there is still the risk that this product category receives less emphasis in the overall WEEE system with targets still based on the overall weight of collected WEEE. There is some evidence from Denmark and Finland that the presence of lamp-specific PRO may ensure that lamps are adequately emphasised. However, the case of Sweden demonstrates that the emphasis on this product category can also be made by other stakeholders (in that case, the Minister for the Environment) and in fact this may be even more effective in motivating collection. The effectiveness of recent education campaigns in Norway to raise awareness of small WEEE collection, in which lamps are given special emphasis, still has to be gauged, but thus far having neither lamp specific PROs nor a particular emphasis on collection of lamps from other influential stakeholders may help explain the significant difference in performance between this category compared to WEEE collection overall in that country. Interviewed stakeholders also indicated that there was still room for raising the level of consumer awareness about gas discharge lamps to include not only disposal options, but the benefits of recycling these products for the environment and closing valuable material loops.

Further optimisation of materials in closing the loop and improving design requires communication between (the right) upstream and downstream actors. The problems with EPR systems incentivising design change are not unique to lamps, but an overall acknowledged challenge for WEEE systems in general (Huisman, 2013; Kalimo et al., 2012; Lifset et al., 2013; van Rossem et al., 2006b). However, there are challenges also unique to lamps due to the increasingly prolonged life of lighting products. Unlike many other categories of WEEE products in which turnover of products becomes shorter and shorter, new energy saving lamp products have an average lifespan of 8500 h for a CFL and 25,000 h for LEDs (U.S. Department of Energy, 2012), which can correspond from a few years to several decades depending on actual use.<sup>3</sup> The lighting industry has used an

average of six years (European Lighting Companies' Federation, 2003), but even this means communicating information to upstream producers as information from actual recycling is often too late to be relevant for the current design of lighting products. Product designers then must be incentivised to design with end-of-life management in mind without empirical knowledge of that management. The challenge of providing such incentives is compounded by the fact that consumers of lighting products do not necessarily respond to environmental design and reward such efforts. Despite new standards and more efficient lighting options available, the least expensive and least environmentally beneficial lighting products continue to dominate the market in Europe (Bennich et al., 2014). In light of these challenges, it may well be that EPR, while part of the means to communicate and incentivise consideration of end-of-life management at the design stage, is not sufficient to overcome the other influences on design. These barriers may need to be addressed through more direct tools to influence ecodesign.

The development of new technology such as LEDs and more integrated products in lighting is increasing in its pace and market penetration (McKinsey and Company, 2012). Such technologies bring a new set of challenges for WEEE system for lamps. It is unknown whether the smaller amounts of rare earth material (in addition to other critical materials like Gallium and Indium) will have the same potential for recycling as the gas discharge lamps. The longer lifetimes of these products may also result in less waste material overall to be collected and recovered. The best ways to deal with hazardous materials as LEDs become the dominant lamp type in the waste streams remains a question as to the best recycling techniques for integrated LED products. The long life of these products and the rapid development of the products may mean that they are disposed before their end-of-life, in which case opportunities for reuse of some components may become possible. Prevention of waste and product design for recycling, one of the key aims of EPR is still a challenge for lamps, and consideration of the new technology will be key to further advancing a circular economy.

## 5. Conclusion

Collection and recycling of gas discharge lamps should be a priority in a circular economy, in consideration of both the avoided environmental harm of mercury emissions and the potential for recycling of valuable materials. Nordic countries perform well in the collection and recycling of gas discharge lamps compared to other EU countries, and this performance can be attributed to robust system architectures, as a result of the rule base but also other factors. There is evidence that the systems continue to improve in terms of convenience and in closing material loops, with the recycling of rare earths from lamp phosphors a notable development. However, challenges remain to further optimise the systems, particularly in terms of meeting EPR goals for better design and in light of rapidly changing technology.

## Acknowledgements

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<sup>3</sup> 8500 is an average but it should be noted that the use varies significantly. In a professional situation the product would typically be used more intensely than home use, but the users also typically purchase different specifications of lamps (i.e. 6000 h CFLs versus a 15,000 h LFLs). So this could result in a majority of fluorescent lamps being disposed around 6000 h, but with a long tail extending decades.

## Appendix A. List of interviewed stakeholders

	Name	Organisation, position	Stakeholder group	Interview date
Denmark	Jan Bielefeldt	Lyskildebranchens WEEE Forening (LWF), Administrative Director	Producer Responsibility Organisation (lamps)	In person interview – 26 August 2014
	Jonas Engberg	Ikea, Sustainability Manager Denmark	Retailer	In person interview – 26 August 2014
	Hardy Mikkelsen	Reno Djurs, Environmental Manager	Municipal Waste Organisation	Phone interview – 4 December 2015
	Lotte Wammen	Forsyning Helsingør, Waste planner	Municipal Waste Organisation	In person interview – 15 December 2014
	Rahbek			
	Niels Remtoft	Dansk Affaldsforening, Special Consultant	National Waste Management Association	In person interview – 15 December 2014
Finland	Senja Forsman	SOK Grocery Chain Management, Compliance Manager	Retailer	Phone interview – 4 December 2014
	Timo Hämäläinen	Finnish Solid Waste Association, Development Manager	National Waste Management Association	Phone interview – 19 December 2014
	Jorma Koskinen	Ekokem, Sales Group Manager	Recycler	Email correspondence – 29 January 2015
	Jesse Mether	Rautakesko Ltd, Sustainability Manager	Retailer	Email correspondence – 19 December 2014
	Perrti Raunamaa	FLIP, Administrative Director	Producer Responsibility Organisation (lamps)	Phone interview – 8 December 2014
Norway	Tuomas Räsänen	Elker Oy, Chief Operations Officer	Producer Responsibility Organisation	Email correspondence – 22 January 2015
	Ellen Halaas	Avfall Norge, Adviser for framework and law collection, sorting and recycling	National Waste Management Association	Phone interview – 9 December 2014
	Guro Kjorsvik Husby	El Retur, Information Officer	Producer Responsibility Organisation	Email correspondence – 24 November 2014 and 8 January 2015
Sweden	Bjørn Thon	RENAS, Administrative Director	Producer Responsibility Organisation	Phone interview – 30 January 2015
	Carina Andersson	IKEA of Sweden, Product Laws & Standard specialist – Producer Responsibility	Producer	Email correspondence 13 and 15 May 2015
	Jessica Christiansen	Avfall Sverige, Education Manager/Controller	National Waste Management Association	In person interview – 16 December 2014
	Jonas Carlehed	Technical Advisor WEEE	Association	
	Lars Eklund	IKEA, Sustainability Manager Sweden	Retailer	Phone interview – 30 January 2015
		Natursvårdsverket (Swedish EPA), Advisor	Government authority	Phone interview – 2 December 2014
	Göran Lundholm	Environmental Enforcement		
		Nordic Recycling, General Manager	Recycler	In person interview – 13 August 2014; phone interview – 27 October 2014
Additional correspondence	Dolores Öhman	Hässelholm Miljö, Head of Waste Collection and Customer Service	Municipal Waste Organisation	In person interview – 3 September 2014
	Anders Persson	SYSAV, CEO	Municipal Waste Organisation	In person interview – 9 September 2014
	Mårten Sundin	El-Kretsen AB, Marketing Manager	Producer Responsibility Organisation	In person interview – 5 December 2014
	Hans Standar	Svensk GlasÅtervinning AB, CEO	Glass recycler	Phone interview – 4 December 2014
	Joseph Tapper	Elektronikåtervinning i Sverige, CEO	Producer Responsibility Organisation	In person interview – 5 December 2015
		SERTY (Finland), ERP (Denmark)	Producer responsibility organisations	Email correspondence

## Appendix B. Sample interview protocol for producer responsibility organisations

- In other countries there are different situations regarding a separate PRO for lamps. What are the advantages and disadvantages having a PRO focussed solely on lamps? What else distinguishes [organisation] from other PROs operating in [country]?
- How does the general WEEE system affect the take back of lamps? Would you characterise the system as competitive or cooperative for collection between the PROs?
- What do you find to be the particular challenges to take back of lighting products? For example, collection, transport and recycling for lamps have been described as very expensive compared to other WEEE categories but the costs are different in each country context. What are the main cost factors and how is [organisation] working to make the system as cost efficient as possible?
- There are statistics from Eurostat regarding recycling in [country]. The collection rates vary depending on how you count, for example historically versus same year as well as how you divide product categories. How does [organisation] measure collection and recycling effectiveness for lamps and are there challenges to collecting good information (e.g. from producers)?
- How does your organisation communicate with other stakeholders like producers, producer responsibility organisations and government authorities – is there a specific forum for this?
- Is there any information or communication with producers regarding the end-of-life/recyclability of products? How do the producers respond?
- Do you have information about how recycled fractions from collected and treated products are used? Is there interest/action on using these fractions in particular ways (e.g. in lighting products)?
- Do you differentiate fees in any way depending on the product? Is there likely to be any differentiation between CFL and LEDs in the near future?
- How are producers active in the system through your PRO?
- The EU is considering a separate target for gas discharge lamps. What is your organisation's view about this?
- In the media in some countries, it has been highlighted that there are still lamps ending up in incineration and glass recycling. Is it an issue in [country]?
- Transporting hazardous waste such as lamps could pose risks from mercury for waste handlers. Is handling mercury-containing waste products or broken lamps an issue in [country]?

13. There is the website and some material from [organisation], are there any other ways [organisation] is working with education and information to raise awareness about WEEE recycling?
14. Are there strengths or weaknesses you perceive to the [country] WEEE system compared to other Nordic countries?
15. Nordic countries are often cited as the best practitioners of WEEE recycling - what do you think are the main factors in success?
16. Improving collection and recycling is a continuous challenge, what do you think are the main areas that still need significant improvement? Is there more that can be done with critical materials recovery for instance?

### Appendix C. Sample interview protocol for national waste management associations

1. What are the main issues in producer responsibility for WEEE where your organisation is involved on the member's behalf?
2. How does your organisation communicate with other stakeholders like producers, producer responsibility organisations and government authorities – is there a specific forum for this?
3. Are there any issues with working with the relationship between municipalities and PROs in [country]? Is it a contract or other agreement on how the responsibility is allocated and managed for collection points and collection?
4. Would you characterise the system as competitive or cooperative for collection between the PROs?
5. Transporting hazardous waste such as lamps could pose risks from mercury for waste handlers. Is handling mercury-containing waste products or broken lamps an issue in [country]?
6. From [organisation] reports there are still some lamps found in residual waste. Are these and other small electronic waste perceived as a particular problem?
7. How are municipalities and/or your organisation working with increasing collection of lamps and other small WEEE? Are there any pilot projects or innovative examples to further optimise the WEEE system in this respect?
8. There is the website and some material from [organisation], is there more [organisation] is doing to educate about hazardous waste like gas discharge lamps?
9. The EU is considering a separate target for gas discharge lamps. What is your organisation's view about this?
10. Are there strengths or weaknesses you perceive to the [country] WEEE system compared to other Nordic countries?
11. Nordic countries are often cited as the best practitioners of WEEE recycling - what do you think are the main factors in success?
12. Improving collection and recycling is a continuous challenge, what do you think are the main areas that still need significant improvement?

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# Paper II







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Full length article

## Recycling of rare earths from fluorescent lamps: Value analysis of closing-the-loop under demand and supply uncertainties

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## ABSTRACT

Rare earth element (REE) recycling remains low at 1%, despite significant uncertainties related to future supply and demand and EU 2020 energy efficiency objectives. We use a global production network framework of REE flows from mine to REE phosphors in energy-efficient lamps to illustrate the potential of closed-loop recycling for secondary supply under different scenarios of primary supply and forecasted demand for LEDs, CFLs and LFLs. We find that different End-of-Life Recycling Rate scenarios for REE secondary supply range between meeting forecasted REE demand and filling primary supply gaps, and competing with primary supply. Our argument centres on diversifying REE sourcing with recycling and the choice between primary and secondary supply. We stress that secondary REE phosphor supply requires further policy support for lamp collection and a discussion of the value of REE phosphor recycling which underlies its economic feasibility.

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## 1. Introduction

With an increase in energy efficiency of 20% to be achieved by 2020 within the European Union (EU), lighting presents a core area of interest. Replacement of inefficient bulbs by 2020 is expected to enable energy savings to power 11 million households a year. In 2009, regulations pursuant to the EU Eco-design Directive introduced stricter energy efficiency requirements for lighting products, which induced a phase-out of incandescent lamps (EU Commission, 2009, 2014a). By 2016 it is expected that a majority of these lamps will be phased out, with similar legislations implemented in other nations including Australia, BRIC countries, Japan, South Africa, and the United States (UNEP, 2014).

The lifetime of incandescent lamps is about four times shorter and their efficiency significantly less than compact fluorescent lamps (CFLs), with 15 lumens of visible light per watt of electricity consumed (lm/W) versus 63 lm/W (Wilburn, 2012). A linear rather

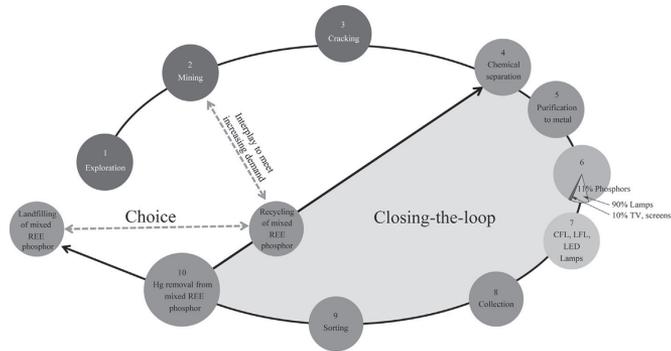
than bulb shape characterizes linear fluorescent lamps (LFLs). Fluorescent lamps emit light when voltage is applied to the mercury gas within the glass body, which produces UV light that is transformed to white light by the phosphor powder coating of the lamp (Lim et al., 2013). Light emitting diodes (LEDs) have a lifetime approximately three to six times that of CFLs (Wilburn, 2012). LEDs emit light when electric current passes through a semiconductor chip and they are distinct to fluorescent lights in that they contain minor proportions of phosphor powder and no mercury.

While the market share of LEDs is projected to accelerate, the transition from fluorescent lights will take time partly due to the upfront costs of LEDs in comparison to CFLs and LFLs. McKinsey & Company (2012) expect CFLs and LFLs to remain with a share in the lighting technology distribution until 2020, yet their significance is anticipated to decrease faster jointly with market demand for REE in fluorescent lamps, as envisioned by Solvay and General Electric and illustrated in Fig. 2 (Cohen, 2014). Of central concern to the lighting industry are phosphor powders in these lamps, which contain rare earth elements (REE) used for their luminescent properties and key to producing white light (Binnemans et al., 2013a).

Since the early 1990s, China has gradually emerged as the largest consumer and producer of REE. The country hosts the majority of global mining and processing of these elements and has enacted numerous policies including quotas for mining and export (latter

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**Fig. 1.** Global production network of REE phosphor-based, energy-efficient lamps. Source: adapted from Ereon (2014) and Simoni (2013) with % indication of REE phosphor share of total REE market (derived from Castilloux, 2014a) subdivided into estimated 90% of phosphors used in energy-efficient lamps, and 10% for TVs and screens (Balachandran, 2014).

replaced by export licences in January 2015, see Bloomberg News, 2015) and a two-tier pricing system, under which REE cost less in China than in the rest of the world (ROW), introduced by export duties and trading rights, which significantly increases the price of exported REE products (WTO, 2014). Concerns about decreases in REE availability outside China intensified with the price increase of export-directed REE products by up to +600% in 2011 (Massari and Ruberti, 2013). Lawsuits against the REE export policies by China were filed at the WTO (2012) by the EU, Japan and the U.S. and in response to the WTO (2015) Dispute Settlement Body, China removed the application of export duties and export quotas to REEs, and the restriction on trading rights of enterprises exporting REEs. It remains uncertain how subsequent new Chinese industrial policy measures, including new export licences and the ad-valorem tax, will affect the market over the long-term. Strategies to target these concerns address the diversification of REE supply outside China, including re-opening old mines or establishing new mines, and include discussions about whether government intervention would be justified in recognizing the need for integrated value chains (Machacek and Fold, 2014; Tukker, 2014; Zachmann, 2010). Simultaneously, efforts in design to reduce and substitute REE in product components and recycling have surged, aiming to prevent future supply risks.

This study contributes to the discourse on REE recycling with a value analysis of recycled heavy REE europium (Eu), terbium (Tb) and yttrium (Y) from phosphor powders of fluorescent lamps as source of supply at times of EU and U.S. REE criticality classification (EU Commission, 2014b; Richter and Koppejan, 2015; U.S. Department of Energy, 2011). Today, at most 1% of all REE used in different applications are recycled (Binnemans, 2014; Binnemans and Jones, 2014). The role of REE recycling has been explored and critically reviewed in general (Guyonnet et al., 2015; Moss et al., 2013; Schüler et al., 2011; U.S. Department of Energy, 2011) and from the viewpoint of specific REE, laboratory experimentation and product groups (Bandara et al., 2014; Binnemans et al., 2013a; Dupont and Binnemans, 2015; Eduafo et al., 2015; Habib et al., 2014; Kim et al., 2015; Rademaker et al., 2013; Sprecher et al., 2014; Tunsu et al., 2015). While several studies have concluded that recycling of REEs is worthwhile and requires a broader strategy to enable REE processing capacities, including tracing the REE from mine to end of life (EoL) waste (Rademaker et al., 2013; Sprecher et al., 2014), none have provided an in-depth analysis of commercial scale recycling and what is needed to upscale recycling. To this end, this study provides an empirical analysis, using a case study of REE

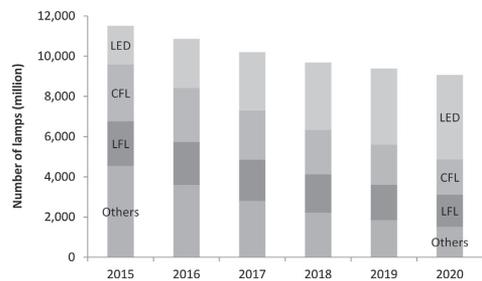
phosphor recycling on a commercial scale and an ex-ante analysis of the market from 2015 to 2020 to assess and discuss the potential for recycling of REE from energy-efficient lamp phosphors. We discuss what factors, including regulatory instruments and rethinking value propositions, are necessary to realize such potential.

**2. Methodology**

Our conceptual approach involves a qualitatively informed global production network framework to depict value adding, or processing steps from REE-containing ore to REE content in phosphor powders as used in energy-efficient lamps. This is the framework from which we then research the potential for secondary supply and closing the loop for REE in lamp phosphors through a mixed methods approach involving both a case study and modelling. Our case study provides an ex-post analysis of the experience of commercial REE recycling of REE phosphor containing lamps. This and our forecasts of supply and demand of Y, Eu and Tb then underpin the ex-ante analysis of the potential for development of secondary supply of REE phosphors from 2015 to 2020.

**2.1. Global production network of rare earths and phosphors**

Five steps, depicted in Fig. 1, precede the production of REE phosphors. Investor interest in favourable returns on investment finances prospecting and exploration of REE which enables data



**Fig. 2.** Forecasted development of the total global lighting market and lamp type shares. Source: Adapted from McKinsey & Company (2012).

collection for sequential reporting required for the decision on the granting of an exploitation licence. Mined REE-containing ore is beneficiated by crushing and grinding, mineral separation, adjusted to the REE-mineral type, REE-grade and the mineral assemblage.

Next a cracking process leaches the REE from the REE-minerals resulting in a concentrate of mixed REE solution. A chemical separation into individual REE follows. Most recent estimates partially produced from primary data suggest that REE use in phosphors accounted for 11% of total REE market demand in 2013 and for 19% of REE market demand value in the same year (derived from Castilloux, 2014a). Usually, phosphor manufacturers buy a concentrated REE product (oxides or compounds, see Lynas Corporation, 2014) for direct use in producing various patented phosphor powder compositions (Wilburn, 2012). The 11% phosphors are then used by various phosphor using applications (Castilloux, 2014a), with an estimated 90% for phosphors in energy-efficient lamps, and 10% for TVs and screens (Balachandran, 2014).

REE-based phosphor powders use varying amounts of REE, resulting in a wide variety of powder compositions (Ronda et al., 1998), but primarily phosphor powders contain some proportion of Y, Eu and Tb to generate red, green and blue phosphors (Balachandran, 2014). Almost all global supply of Eu, about 85% of Tb and close to 77% of Y are used for phosphors (Moss et al., 2013; Tan et al., 2014). The high purchase cost of phosphors can be attributed to the high (99.999%) purity requirements (Binnemans et al., 2013a) on the REE used and the lower abundance of these heavy REE, relative to lighter REE, in REE-bearing minerals as explained by the Oddo-Harkins rule (Chakhmouradian and Wall, 2012).

The balancing problem (Binnemans et al., 2013b; Falconnet, 1985) adds to this the challenge of selling all REE mined (if stockpiling is disregarded), as demand does not match the natural distributional occurrence of REE. At the time of writing, supply of light REE (e.g. lanthanum and cerium) is not met by equally high demand while some heavier REE (e.g. dysprosium and europium) are in higher demand than supply (Binnemans et al., 2013b). In addition, REE phosphors are both essential and hardly substitutable in the functioning of fluorescent lamps.

In this article we first examine the relationship between global primary supply and secondary supply of lamp phosphor REE through an empirical case, following Guyonnet et al. (2015, pp.1) who emphasize that ‘any global (systemic) analysis of mineral raw material supply should consider both types of sources’. We also model the dynamics in the global production network of REE linked to demand and supply of Y, Eu and Tb for phosphor powders in fluorescent lamps. The assumptions underlying our forecasts are presented below and uncertainties are addressed in Section 4.

## 2.2. Demand forecast

REE content varies in CFLs, LFLs and LEDs, see Table 1. The data related to the elemental composition of phosphors contained in LFL, CFL and LED has been derived from Castilloux (2014b) for phosphor (g), and Wu et al. (2014) for REE composition in standard tricolour phosphor. The estimated phosphor composition for all these three lamp types is shown in Table 1.

**Table 1**  
Approximate REO content (g/unit) of various energy-efficient light types.

Range of content	Y <sub>2</sub> O <sub>3</sub> (g) 46.9–51.2%	Eu <sub>2</sub> O <sub>3</sub> (g) 3.9–4.4%	Tb <sub>4</sub> O <sub>7</sub> (g) 2.2–2.6%
LFL	1.0975–1.1981	0.0913–0.103	0.0515–0.06084
CFL	0.7035–0.768	0.0585–0.066	0.033–0.039
LED	0.0047–0.0051	0.0004–0.0004	0–0

Sources: Castilloux (2014b); Wu et al. (2014).

In our model we use McKinsey & Company (2012) data on general lighting applications (which encompasses lighting in residential applications and six professional applications, namely office, industrial, shop, hospitality, outdoor and architectural) on the number of lamp types, both new installations and replacements, from 2015 to 2020 for all world regions (Europe, North America, Asia incl. China, Latin America, Middle East & Africa). The number of lamps is multiplied with the averaged total REO (g) as per lamp type in Table 1 to estimate the final demand of Y<sub>2</sub>O<sub>3</sub>, Eu<sub>2</sub>O<sub>3</sub> and Tb<sub>4</sub>O<sub>7</sub> for these three energy-efficient lamp types.

## 2.3. Secondary supply forecast

To enhance our understanding of the future demand of phosphors for lighting purposes and the potential role of secondary supply originating from recycling these waste lighting applications, we model demand for Y, Eu and Tb in energy-efficient lamps from 2015 until 2020. The McKinsey & Company (2012) data estimates a range of lifetimes for the different lamp technologies and we use this combined with U.S. Department of Energy (2012) data to estimate the lifetime of the different lamp technologies in whole years to anticipate availability of lamp waste in the model. We use a lifetime of three years for CFLs and LFLs and of eleven years for LEDs, yet noting that lifetimes are highly sensitive to how the lamps are used (i.e. switch cycles, length of use per day, and other factors). We also conducted a sensitivity analysis for lifetimes as part of the later discussion, provided in Table B.4.

Secondary supply of phosphor from lamps is then estimated with end-of-life recycling rates (EoL-RR), also known as recovery rates (Graedel et al., 2011) which consider a collection rate for lamps as well as an estimate of total recycling of the REE from waste lamps. In the model we use collection rate scenarios of 15–40–70%. The 15% collection rate scenario assumes, in line with status quo and global trends, collection rates in Europe of nearly 40% (Eurostat, 2014) and lower collection rates on the state and sub-state level in the U.S., Canada, and Australia (FluoroCycle, 2014; Silveira and Chang, 2011) as well as more environmentally sound management of waste lamps in developing countries (see e.g. UNEP, 2012). The 15% collection rate also reflects a slow uptake of policies and a lack of collection to date in key regions, for example in China (Tan et al., 2014).

The 40% collection scenario assumes that legislation on extended producer responsibility (EPR) and other supportive legislation will be applied globally in major regions, such as the U.S., China and India. Essentially it reflects the average EoL fluorescent lamp collection rate to-date observed among EU countries, with large disparities between countries but an overall 40% average (Eurostat, 2014). This scenario expects the continuous implementation of legislation related to EoL lamp management on U.S. state and sub-state level (Corvin, 2015; Silveira and Chang, 2011), fruition of plans and pilot projects in India (Pandey et al., 2012), and expansion of China's existing EPR legislation to include lamps (Tan and Li, 2014).

Lastly, the 70% collection rate reflects the EU top-end observed in a few countries (Sweden for example—see Eurostat, 2014 and Table B.5; and Taiwan (Environmental Protection Administration, 2012)) and thus the high end of anticipated global collection. This rate represents a scenario with implemented legislation and well-designed systems in place in major regions around the world.

The efficiency of the recycling process also needs considering to estimate secondary supply. Binnemans et al. (2013a) and Tan et al. (2014) assume an overall recycling process efficiency rate of 80%, and we use this assumption with an amendment. We add a key step to the 80% assumption, namely the *recycling of REE phosphors from the waste lamp powders* between collection and recovery of REE. EoL fluorescent lamps are collected and treated to prevent

**Table 2**  
Components of end-of-life-recycling-rates under three different scenarios.

Scenarios	Collection rate of lamps (%)	REE phosphor recycling rate (%)	Recycling process efficiency rate (%)	EoL-RR (%)
Low ambition—top down calculation of EoL-RR of global REE phosphor capacity eqv. to Solvay's capacity (450t) compared to global total capacity (11,150 t)	15	55	80	7
Medium ambition—top down calculation equivalent to Solvay's capacity (450 t) in Europe, North American and China (1,350 t) compared to global total capacity (11,150 t)	40	59	80	19
High ambition—bottom up calculation from best-case Sweden scenario (70% collection as per Sweden and Taiwan; 95% REE phosphor recycling rate in Sweden)	70	95	80	53

Sources: EoL-RR concept as per Graedel et al. (2011) and adapted with REE phosphor recycling rate conceptualized by authors. Overall REE recycling process efficiency rate are adopted from Binnemans et al. (2013a) and Tan et al. (2014). Note: The REE phosphor recycling rate is calculated on the basis of informed estimates of the 'collection rate of lamps' and the 'recycling process efficiency rate'.

mercury contamination as there are few other drivers for lamp collection in the first place. For this reason, the EU WEEE Directive explicitly requires removal of mercury for these types of lamps in the recycling process and therefore EU recycling rates for collected lamps are in general over 90% (Eurostat, 2014). *While removal of mercury from lamps involves isolation of the phosphor powder layer where the majority of mercury is present, it does not always involve the further recovery of REE from this powder and this fraction is often landfilled as hazardous waste in the EU* (Walter, 2011; Interviewee C, 2015). Thus, this step leaves a significant gap in the potential for recycling to achieve higher EoL-RR.

In the low-ambition 15% global lamp collection scenario we assume that overall, only 7% REE phosphors are recovered of the lamps collected and recycled. The medium scenario assumes 40% global lamp collection and a tripling of REE recycling capacity worldwide with an EoL-RR of 19%. In the most ambitious scenario, high collection rates like those seen in Sweden are coupled with the recycling process used in that country in which nearly all waste phosphor powders are sent for further recovery of REE at Solvay for a final EoL-RR of 53% (assumptions for each scenario are summarized in Table 2).

#### 2.4. Supply forecast

To calculate the volumes of Eu, Tb and Y available to energy-efficient lamps in 2015–2020, we estimate total primary supply volumes. We use current estimates that 100% of Eu, 85% of Tb and 77% of Y are used for phosphors (Moss et al., 2013; Tan et al., 2014) of which 90% would be available to the production of LEDs, CFLs, and LFLs and the remaining 10% for TVs and screens (Balachandran, 2014).

We assume that total primary supply volumes consist of Chinese rare earth oxide (REO) production, current rest of world (ROW) production and forecasted REO production in the ROW. To forecast future ROW production volumes, we consider a set of ROW developed REE projects with publicly available data including planned production volumes, REO distribution, and costs (see Appendix A). Company-reported dates for starting production provided us with a sequence for their potential market entry, which we modelled with three scenarios that considered delays of one, three and five years in the start of production.

We assume that these projects enter the market when REE prices make the anticipated production start economically feasible. To find these prices we used a price model represented by a REE industry cost curve which is based on the indirectly proportional relation of the supply quantities and prices of individual REO (see Appendix A, more details available from Klossek et al., n.d.). The price model is based on REO prices from March 2015 (Metal-Pages) and current supply volumes of individual REO.

Two approaches guided our calculation of current supply volumes of individual REO for the price model (1) based on the total mining quota in China, (2) based on the Chinese export quota and illegal supply. Approach (1) departs from the phase-out of Chinese export quota by May 1st, 2015, and new export licensing. We assume that the total REO mining quota for 2015 in China could be a proxy for the potential maximum supply volume which could come from China. To calculate total REO supply volumes (for each REO individually) we added the expected REO mining quota in 2015 to current ROW supply volumes.

The second approach uses 2015 REO mining quota in China as a proxy for maximum REO production volumes in China in 2015. As the quota for the 1st half of 2015 increased by 11% (Argusmedia, 2015; Shen, 2015) compared to the quota for the 1st half of 2014, we assume that the total quota in 2015 will be 11% higher than the total quota in 2014. For 2015 we assume the same distribution of REO in the mining quota as in 2014 (Chen, 2014). We estimate current ROW production at 14%, assuming China's share of global production has not changed significantly from the 86% share in 2012 (Tse, 2013 in Wübbeke, 2013). To calculate the volumes of individual oxides we used the same REO distribution as in the total mining quota (which in our view represents an average distribution in a typical hard rock REE deposit).

In the second approach of Chinese export quota and illegal supply, we acknowledge that Chinese REE export quotas have been phased out and replaced with an export licensing system. We expect that in such a situation a part of the illegal export volumes would be sold via official channels; however the total export volumes (consisting of official and illegal volumes) would not change significantly. To estimate the export volumes in 2015 (to be a proxy for the supply volumes coming from China at FOB prices<sup>1</sup>) we considered the REE export quota in 2014 as a proxy for the maximum official export volumes. To find total export volumes we added illegal supply volumes to the export quota (assuming a 40% rate of illegal supply in total export volumes as estimated by Argusmedia, 2015).

The results of these two different approaches to calculate current supply volumes differed slightly. In our view, the Chinese export quota and illegal supply approach is more realistic as it represents maximum REO volumes for export to be sold at FOB prices and considers the domestic REO demand of China, while the first approach assumes that the total REO production of China could be exported which is unrealistic. Our results are therefore showing the

<sup>1</sup> Free On Board (FOB) implies that the seller fulfils her/his obligation to deliver when the goods have passed over the ship's rail at the named port of shipment. This means that the buyer has to bear all costs and risks of loss of or damage to the goods from that point. The FOB term requires the seller to clear the goods for export (WCS International, 2013).

second approach, while the first approach is illustrated in Fig. A.1. Results of these calculations (tonnes of REO) were converted to tonnes of rare earth metals to be compared to the results of the demand and secondary supply analysis.

### 2.5. Case study

Known as European key player in REE chemical separation, Solvay-Rhodia, hereafter 'Solvay', operates across a bandwidth of industrial sectors including energy, automotive and electronics. It is, jointly with Japanese Shin-Etsu, among the only outside of China capable of chemically separating REE into both light and heavy individual REE products to purities of acceptance for customers on a commercial scale (Interviewee B, 2012; Shin-Etsu Chemical, 2014). Solvay runs REE chemical separation facilities in China and in La Rochelle, France. It is the first large supplier to the lighting market with a commercialized recycling process of La, Ce, Eu, Tb, and Y (Osram and Philips also ran pilot scale recycling projects) (Binnemans et al., 2013a; Moss et al., 2013; Otto and Wojtalewicz-Kasprzak, 2012).

Our case study relies on both literature review and semi-structured interviews with key actors. To enhance reliability of our empirical data, we interviewed representatives of firms involved at different stages of the recycling network, specifically collectors, recyclers and the REE chemical separator and refiner, Solvay, in line with methods as proposed by Kvale and Brinkmann (2009). Our data collection followed an iterative process. Phases of empirical data collection followed desk research for cross-checking available public data and to triangulate industry data with data from regulatory institutions, including from the European Commission, and from scholarly recycling experts. The empirical data served for identifying factors key to recycling REE, such as related to logistics and material components which add to the economic feasibility of REE phosphor recycling. This data also supported our ex-ante modelling.

## 3. Results

### 3.1. Closing-the-loop with REE phosphor recycling: The case of Solvay

EU funding through LIFE+ of 50% of the project (equivalent to about EUR 1.1 million for 24 months from June 2012) (EU Commission, 2011) supported Solvay to commercially recycle waste lamp phosphors following four years of prior research and development and industrialization (Solvay, 2014). As Fig. 3 shows, Solvay receives phosphor powder from recyclers and first removes the mercury, glass and other components to physically liberate the rare earth concentrate, which is then sent to the chemical separation plant. There, the halophosphates are removed and the REE phosphors are cracked resulting in a REE concentrate that can be fed, as in a primary process, into a solvent extraction process for individual REE chemical separation. High-tech knowledge of technical staff and internally developed, sophisticated software is applied to manage this solvent extraction process (Leveque, 2014). The REEs are then reformulated into new phosphor precursors for new energy-efficient lamps (Solvay, 2014).

Solvay has developed a flow sheet for the recovery of REEs from a mixture of halophosphate and REE phosphors. According to the patent for the process, the final yield of REE is at about 80% (Braconnier and Rollat, 2010). The objective has been to demonstrate the industrial processing of 3,000 t of lamp waste/year, which corresponds to the forecasted European waste production for 2020 (Golev et al., 2014; Solvay, 2014) and results in 90% waste stream valorisation corresponding to 10–20% of REO (rare earth oxides),

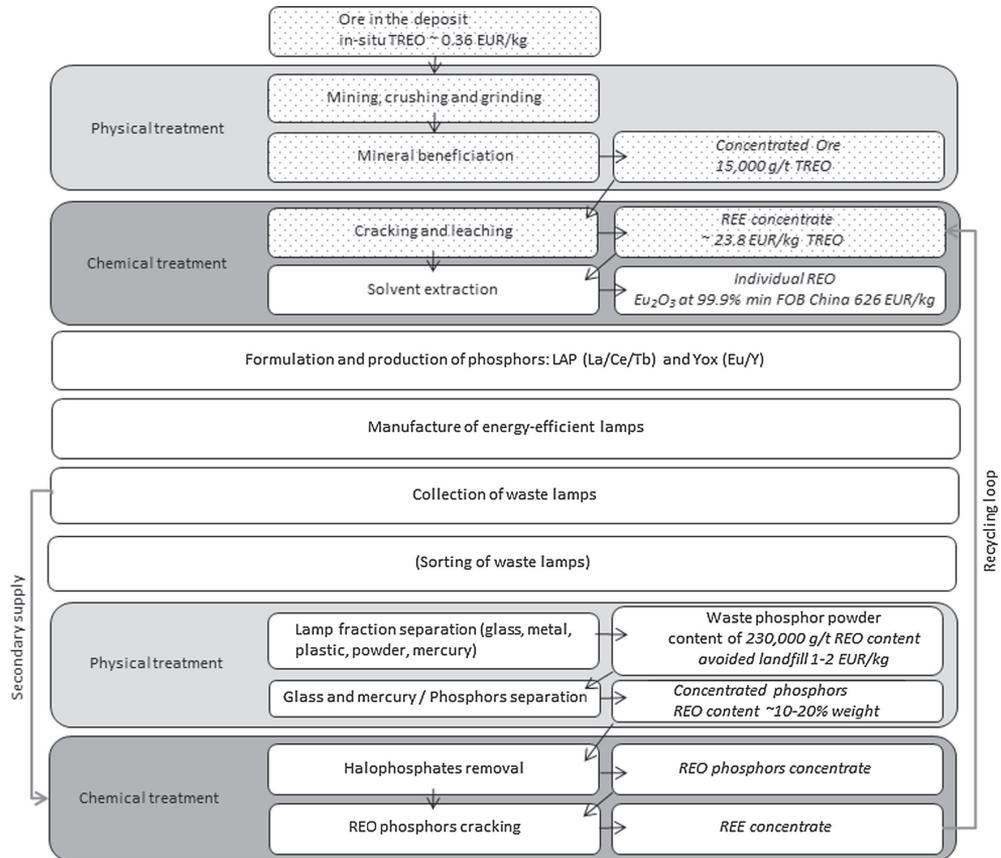
glass (by-product) and phosphate (by-product) at Solvay (2014). The REEs lanthanum, cerium, europium and gadolinium, terbium, and yttrium are being recovered (Rollat, 2012). Technical complexities of the recycling process can be better understood in context: Lamp phosphor powder mixtures are the essence of the light characteristics of a fluorescent lamp and different mixtures are used in the powder manufacture (Koninklijke Philips Electronics N.V., 2011) which form the competitive base of lamp manufacturers and they are protected by patents (Li, 2012). Currently Solvay recycles several hundred tonnes of rare earth phosphors each year, primarily from Europe.

While the recycling of phosphor powders is less economical than other internal projects (Walter, 2011), we conclude that it strengthens Solvay's core competence. Depending on global market pricing of REOs, the firm's resources are liberated by sourcing separated REOs externally rather than conducting the chemical separation in-house and thus, the firm's resources are freed-up to pursue high technical sophistication in REE formulation to customer specification. Phosphor recycling reflects an adaptation of Solvay's REE subdivision to overall REE industry dynamics and a diversification strategy in raw material sourcing. Specifically, increasing challenges encountered by Solvay in accessing REE-bearing ore in China (Interviewee A, 2012, 2013) and general REE price volatility initiates quarterly decision-making on whether to chemically separate REEs internally or to purchase them and solely focus on formulation (Walter, 2011). Golev et al. (2014) stress that the annual objective of 3,000 t of waste lamp waste recycling would secure 'Solvay's need for critical rare earths to manufacture new lamp phosphors (...)' In-house solvent extraction is the precursor to formulation and production of phosphors, constituting a key process to unique compositions of the phosphor powder. Thus, operating the solvent extraction processes might provide avenues to run process test routes and potentially explore new patents.

To understand the drivers for Solvay's commercialization project it is important to put it into context of the overall REE market. As described in the introduction, particularly between 2009 and 2011 concerns about the supply and price of REE arose as a result of numerous issues including Chinese restrictions on REE exports. This was a driver for increased attention in the EU for possible sources of supplies outside of China, as well as, potential secondary supplies. Lamps were a viable source of waste phosphor powder in the EU for a couple of reasons. First, existing legislation (the WEEE Directive) already mandates the collection of this waste stream. Secondly, the recycling process of this waste stream typically involved isolating mercury in the phosphor powder, so this powder was already an available end fraction of the recycling process (Récylym, 2014). Moreover, the costs of collection and recycling in EU countries is borne by the lamp producers and in some cases by municipalities (such as of collection in Denmark). REE chemical separator/refiner Solvay only needed to pay for the fractions from recyclers and the processing from that point onwards. Both researchers and practitioners argue that without legislation, collection and thus recycling of energy-efficient lamps are unlikely to take place (Huisman et al., 2008; Interviewee D, 2014; Richter & Koppejan, 2015). The role of legislation and market drivers for secondary supply are further discussed later, and we contextualize recycling scenarios within a future market context in our model.

### 3.2. Future potential for closing loops: Our model

Our EoL-RR scenarios illustrate how closing-the-loop at different REE phosphor powder recovery ratios can contribute to secondary supply of Y, Eu and Tb for new phosphor production to be used in lamps, TVs or screens. We contextualize these EoL-RR scenarios in a comparison with forecasted demand, as per our modelled scenario that stipulates the uptake of different lighting



**Fig. 3.** Primary processing steps from mine to the manufacture of energy-efficient phosphor lamps, secondary supply specific recycling processes of phosphor powders and closing-the-loop with a second run through chemical separation (Tan et al., 2014; Metal-pages, 2015; TMR, 2015; Rollat, 2012). Foregone primary REE processing steps are dotted. Note: Possible variations in the ore grade can impact the ore and REE values in further processing steps. This figure exemplifies a classic REE carbonatite which might provide conservative values when compared to not yet-commercially exploited REE-bearing minerals.

technologies until 2020, and with primary supply as per our supply forecast model.

Our results demonstrate that a global EoL-RR of 53% as per our best case REE phosphor powder recycling ratio modelled in line with to-date Swedish and Taiwanese lamp collection and REE phosphor recycling efficiency could provide secondary supply of Y, Eu and Tb equivalent to our modelled demand forecast of these three REE for LFLs, and an increasing share of CFL demand for Y, Eu and Tb until 2020.

Our EoL-RR of 19% is based on a tripling of REE phosphor recycling capacity for major markets of China and North America in line with our European Solvay case. Such an EoL-RR rate could contribute with secondary supply of Y, Eu and Tb of close to 50% of demand for these REE to be used in phosphors in CFLs.

The 7% EoL-RR corresponds to the estimated current global secondary supply of Y, Eu and Tb. In this scenario, secondary supply of Y, Eu and Tb contributes less than a third of 2020 demand of Y, Eu and Tb and hardly contributes to the demand by the CFL or LFL.

In 2015, the 7% EoL-RR of the three REE phosphors can fill the demand gap with about 7% and can account for up to 9% in 2020. The bandwidth of the 19% EoL-RR to meet demand is at 20% in 2015 and forecasted to more than a quarter of the demand (27%) in 2020. In contrast, and most significant, the 53% EoL-RR enables a secondary supply of the three REE phosphors of more than half of the demand by phosphor-based lamps in 2015 and three quarters of demand by these lamps in 2020 and thus competes directly with primary supply. This 53% EoL-RR illustrates choices about recycling in policy and business decisions which affect future recycling options. It also highlights that these choices require awareness on preferences—whether REE are to be sourced from a host rock or from recycling and why, see Fig. 1.

#### 4. Discussion

The case study and our model demonstrate the potential of secondary supply from waste lamp phosphor recycling to meet

some of the forecasted demand, but the question remains about what factors impede and promote closed-loop recycling. The Solvay case demonstrated that market mechanisms as well as legislative drivers are key to making secondary supply viable. Accessibility of adequate quantities of REE phosphor waste lamps, marketability of the recycled REE phosphors, as well as ability to derive adequate value (the right price at the right time) for these products have been argued as key bottlenecks to realize closed-loop systems (Guide and Van Wassenhove, 2009). In this section we first discuss these bottlenecks in the context of market mechanisms and uncertainties inherent in forecasting the future of the REE market. We discuss the factors that enabled REE phosphor recycling so far and what drivers are necessary for REE phosphor recycling to play a substantial role in meeting future REE phosphor supply.

#### 4.1. Uncertainties of demand

REE use in phosphors is dependent on technological and socio-economic developments which impact the market uptake of lighting technologies. The minor REE content in LEDs is noteworthy for demand projections as is the potential redundancy of Tb in a market dominated by LEDs (U.S. Department of Energy, 2011; Wilburn, 2012). While the development of LED technology has progressed faster than anticipated (Danish Energy Agency, Energy Piano & CLASP European Programme, 2015), there are still concerns about the technology being ready to replace all lighting applications (and this was the reason underpinning the recent delay of Stage 6 EcoDesign requirements for lamps in the EU, see Ala-Kurikka, 2015). Also, phosphor powder substitutes might be found which would strongly influence the price customers are willing to pay for products and alternative ROW supplies of REE (Zachmann, 2010). Such a scenario would affect the attractiveness of developing the secondary supply in absence of other drivers.

Recycling can contribute to remedying the balancing problem, described earlier, which affects both primary supply and demand, as argued by other scholars (Binnemans et al., 2013b; Falconnet, 1985). In our supply and demand forecasts we have only considered the phosphors used for lighting and the technological development within this field of application. Yet other applications such as TVs and background lighting screens in tablets, phones and others also currently demand phosphors based on REE (Balachandran, 2014) and there may be growth in this demand by these or future applications (Castilloux, 2014b). Such growth could create new markets for the secondary supply from recycled lamp phosphors. In addition, it is uncertain which technologies will dominate the future lighting market, a factor which will influence the significance of REE lamp phosphor recycling further: For instance, remote phosphor screw-based or tube LED lamps (T8) will demand more REE than regular white LED lamps and LFL tubes (T8) (Castilloux, 2014b).

#### 4.2. Uncertainties for recycled REE phosphor demand

Binnemans and Jones (2014) outlined three possible recycling routes: (1) direct re-use of the recycled lamp phosphors, (2) recycling of the various phosphors by physiochemical separation methods, and (3) chemical attack of the phosphors to recover their REE content. Options 1 and 2 are linked to a reuse of the powder by the same manufacturer, while option 3 allows for the use by a different party (Binnemans and Jones, 2014). The first two options would likely require a take-back system by the manufacturer, or the implementation of “closed-loop supply chain management” (Guide and Van Wassenhove, 2009). To date, closed-loop supply is not unknown, though more typical for industrial goods like machinery, tools, and process catalysts (Graedel et al., 2011). With the first two options, uncertainties as to the quality of the powder would need to be considered. The powder deteriorates over the lifetime of a lamp

due to exposure to UV radiation and mercury. In addition, recycling processes will affect the quality of the phosphor powder such as particle size and thus, the recycled phosphor powder will expectedly be inferior to the original product. Our article addresses the third recycling route which reflects the Solvay approach in which the recycled powder is chemically attacked.

Demand for recycled phosphors depends on price, which involves the cost of recycling lamps (which can be relatively high compared to the price of the product) (Philips Lighting, 2012). Key factors include efficient design of the scheme, but also transport distances and end use for the recycled glass, the main fraction by weight of the recycling process (Interviewee D, 2014; WEEE Forum, 2010). Notably, depending on the country, costs for recycling and collection of lamps in Europe can be the responsibility of producers and are not necessarily borne by lamp phosphor recyclers, e.g. Solvay, which only pays the recycler for the separated waste phosphor powder (Interviewee D, 2014). Currently, several externalities are not part of the price of both primary and secondary phosphors. These are discussed later in relation to value.

#### 4.3. Uncertainties of primary supply

The primary supply of REE is uncertain with China possibly further consolidating the industry and using integrated production steps subsidized by the two-tier pricing mechanism for import substitution, expected to be upheld under the licensing regime. Further, alternative ROW supply might need to meet growing Chinese REE demand. High-tech skill requirements at each of the processing steps are obstacles for alternative suppliers and add to alternative REE supply risk (Hatch, 2011). Project feasibility, which is evaluated from the reconnaissance exploration to the mine development stage, is tied to several factors as put forward by Klossek and van den Boogaart, (2015): land title and location of the deposit; experience of regulatory authorities with the deposit type and commodity; REE grade and REE distribution representing estimated values from geological studies; potential for significant volume production (t); presence of radioactive elements combined with environmental legislation; REE mineralogy; opportunities for project financing; business relationships with separation facilities; availability of expertise, technology, and equipment; strategic alliances and off-take agreements with end-users; cost competitiveness; obtaining mining permits and social licences to mine; as well as estimated values of REE and market conditions.

REE deposits are characterized by the different mineralogy of various REE-containing ores, which may require new, tailor-made processing routes to be developed (Jordens et al., 2013), as well as additional financial and human resources, and time in testing their feasibility which could result in significant project delays, which is why we model with a one, three and five year delay. Our modelled three year base scenario is shown in Fig. 4, produced from data presented in Table A.3. A project start delay by one year modelled on the export quota and illegal supply approach, affects primary supply insofar as that primary supply of Eu first meets and exceeds demand by 20 t in 2019 while in a five year delay Eu demand would first be met and exceeded by 50 t in 2020. The one year delay resembles the results of a three year delay regarding Eu primary supply. In none of the delay scenarios, Tb primary supply meets demand and in all delay scenarios, Y primary supply first meets and exceeds demand in 2020. Regulations regarding radioactivity applying to certain deposits, limited access to ROW chemical separation know-how and the capital and operational cost requirements to establish a new separation facility, also represent major bottlenecks for alternative value chains of junior REE exploration projects in the ROW (Golev et al., 2014; Machacek and Fold, 2014). As some projects could become infeasible, future primary REE supply could be lower

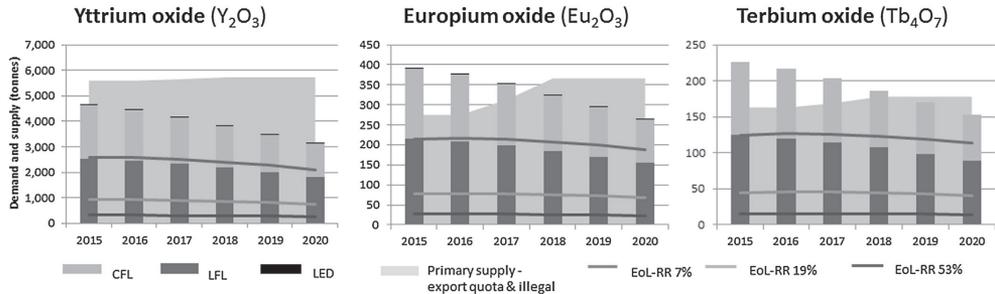


Fig. 4. Potential secondary supply distribution for Y, Eu and Tb based on our three EoL-RR as compared to demand per lamp type (bars) and 3 year delay base case primary supply forecast (grey shading, data in Table A.3) from 2015 to 2020. Please note the different y-axis scales. We use REO as unit to enable a comparison of demand, and primary and potential secondary supply estimates, however, the use of REO in lamp phosphors requires their prior purification to metal, as depicted in Fig. 1, step 5.

or postponed while the commercialization of new physical and chemical separation technologies might increase supply.

#### 4.4. Uncertainties of secondary supply

Our model demonstrates the potential of increased REE powder recycling for REE phosphor powder demand of CFLs/LFLs over the time period in which the general lighting market shifts towards LEDs (and beyond). Even in 2020, the CFL/LFL technologies are expected to account for between 26% (McKinsey & Company, 2012) and a third of the lighting market (Hykawy, 2014). Anticipations on the pace of LED uptake differ: For instance, General Electric anticipates that LEDs will reach a 70% market share in 2020 (see Cohen, 2014; Hykawy, 2014), while Wilburn (2012) emphasizes the role of fluorescent lighting for general lighting in the short to medium term. In our model we tried to find a balance, in line with McKinsey & Company (2012) which anticipates a LED technology market share of 62% in 2020. While LEDs require significantly less REE and a different individual REE mix, such as reduced or no Tb, this lamp technology continues to require small quantities of Y and Eu. Continued heavy REE demand by phosphors used in general lighting (CFL, LFL, LED) and for background lighting (TVs-plasma, LCD, and X-ray intensifying screens) is anticipated (Balachandran, 2014).

Secondary supply is affected by the availability of waste lamps. We have mentioned the uncertainty about actual lifetimes of energy-efficient lamps because this is a function of actual use. If the lamp lifetimes are longer than our modelled assumptions, our sensitivity analysis showed that this would slightly increase the amount of waste lamps available until 2020 (and most likely beyond).

#### 4.5. Promoting secondary supply

As mentioned, our low ambition scenario for recycling is an estimate of the status quo. Achieving higher recycling rates depends on a number of factors, beginning with collection. The case study demonstrated the large role of legislation in making the opportunity for further recovery of REE viable through mandatory collection of lamps (in absence of economic drivers for collection). Our model illustrates that similar timely legislative measures targeting fluorescent lamp collection, could impact end-of-life recycling rates (EoL-RR). The second scenario illustrates a case with more stringent EPR legislation.

The case of Solvay illustrates that such legislation can make REE recycling viable, but it is important to consider the other factors that incentivized Solvay to invest in commercial recycling: First

and foremost, the availability of a separation unit at Solvay was a main driver for the decision towards REE recycling. In addition to an available supply of waste lamps, high REE prices and significant supply risk at the time as well as significant EU interest resulted in financial support. The decision to invest was also part of a business strategy that considered the value of the secondary supply differently (this perception of value is discussed in the subsequent section). In absence of price, supply risk, or other value drivers there may be a need for further legislation to drive not only the collection of waste lamps, but also the further recycling of REE. Such legislation, like a business strategy, can be driven by a different perspective of value and alignment with the goals of a circular economy agenda. Ideally, combining either economic or legislative drivers to promote both collection and recycling of REE would further advance the potential contribution of the secondary supply closer to the case we already observe in Sweden, with a high level of lamp collection coupled with subsequent recycling of REE e.g. through Solvay in France.

Timing is significant in the elaboration and implementation of legislative measures that require REE recycling from fluorescent lamp phosphors. At the time of writing, the majority of EoL REE phosphor powder, even from collected EoL CFL/LFL lamps, is still landfilled. While there is evidence of socio-economic value of REE recycling that could already drive recyclers to further process REE, see Balcan (2015) and Ondrey (2014), some recyclers continue to see the small amount of powder as a barrier to act and prefer the small cost of landfilling over a possible change in their operations required to send the powder for further recycling (Interviewee C, 2015). This is problematic and unfortunate from a resource conservation perspective, as the REEs contained in these EoL phosphor powders (Wu et al., 2014) are already enriched, and since they stem from resource-intensive concentration processes – physical and chemical beneficiation that involves high energy, water and chemical use – of the mined REE-containing mineral.

#### 4.6. Rethinking the value of recycling phosphors

Aside from the challenges in accessibility and marketability of recycled lamp phosphor powders, which can be addressed, the feasibility of recycling is tied to its economics. We have already discussed that the overall cost factors for recycling REE from phosphors entail both costs for collection and the actual recovery. We now look closer at the overall value of a secondary supply of REE phosphors. Consideration of the overall value of recycling would depart from juxtaposing the processes of a secondary supply loop of recycling lamp phosphor powders with those of primary extraction of REE-containing ore. The latter comprises,

as depicted in Fig. 3, mining, mineral beneficiation, and cracking and leaching. These processes involve significant costs for energy and solvents, and operating expenditures comprising future costs for mine rehabilitation, effluent, radioactive material and waste handling. When compared with recycling, even if it involves a second chemical attack of the phosphor powders, the mining and processing costs up to chemical separation associated with the primary supply will not need to be borne.

Researchers have found higher concentrations of REE in the waste lighting products (i.e. anthropogenic deposits, see Mueller et al., 2015); some pointing to more than 15 times higher concentration of Eu, Tb and Y in waste phosphor mixtures as compared to natural concentrations in REE-bearing hard rock minerals (Tan et al., 2014). We would like to stimulate a critical reflection on how it can be possible that recycling of phosphors is not economically feasible despite their high concentration in waste lamps (Langer, 2012; Walter, 2011) and when “only a dozen natural minerals have high enough quantities to be worth the cost of extraction” (Meyer and Bras, 2011). Using a specific process applied to foreign and Canadian EOL phosphor powders with a 98–99% recovery for REE, it has been indicated that REEs could be extracted for as low as USD 6 per kg of mercury phosphor dust (Cardarelli, 2014 in Chemical Engineering, 2014). This cost stands in contrast to the basket value of REE contained, which, averaged on the projects we identified for this study, would amount to about USD 28 per kg (see Table C.1).

With this in mind the focus in *assessing the feasibility of phosphor recycling from waste lamps may need to turn to additional, different value dimensions, beyond the conventional exchange value of the phosphor powder* (from both the primary and secondary processing routes) to include for instance resilience as a factor in business sustainability. Such value propositions could drive business opportunities for closed-loop recycling.

The Solvay business case of closing-the-loop with REE recycling, illustrated how the core competence of the firm is reiterated in a strategy that addresses two objectives: augmenting resilience against supply criticality and further increasing competitiveness. This case has been enabled by EU legislation that has attached societal value to recycling by means of committing producers to collect and recycle waste products, limiting the landfilling of hazardous waste, and promoting closed-loop opportunities. The direct value potential of recycling Eu, Tb and Y used in phosphors of fluorescent lamps is manifested in the addition of a new material stream through phosphor recycling that makes use of existing production capital in a situation of concern and uncertainty over material access and REE supply. As Guyonnet et al. (2015) argue, complementarity between the primary and secondary sources to meet supply requirements is of particular importance in the case of REE for which requirements are increasing. The firm opens up opportunities for value creation as operating its chemical separation plant might facilitate product and process improvements. Use of secondary supply for phosphors is also attractive for its domestic or regional availability as opposed to a dependence on a few key players, primarily China, and the uncertain development of new REE deposits. Secondary supply can augment certainty about short and medium term supply.

Developing the domestic secondary supply of REE can have wider societal benefits while supporting regional and national goals towards more circular economies. The collection and recycling of lamps has a high societal value in the avoided mercury contamination, which is difficult to quantify in economic terms (though some studies, for example, Hylander and Goodsite (2006) have tried and estimated a cost of USD 2,500 to 1.1 million per kg Hg isolated from the biosphere depending on local factors quantity, nature of pollution, media, geography, technology used etc.) At the same time, collection and recycling of energy-efficient lamps represents a cost in terms of overall material recycling (for example,

costs for collection and recycling systems of lamp waste in EU have varied between EUR 0.15 per kg and EUR 2 per kg according to the WEEE Forum, 2010). Reconciling different costs and benefits of avoided pollution by collection and recycling is why legislation is often needed to drive this part of the process (Li et al., 2015).

Recycling of phosphors as indicated above, has societal value in the form of forgone costs of protecting human and environmental health and safety as primary REE processing involves the handling of radioactive elements that have been related to higher health risks, for example to cancer (Lim et al., 2013; Weng et al., 2013). It should be noted that the exposure to radioactive material is also dependent on the geology of the mined deposit as well as the method of mining utilised (Ali, 2014).

Closing-the-loop further saves environmental costs associated with the generation and treatment of 63,000 m<sup>3</sup> waste gas, 200 m<sup>3</sup> acidic water and 1.4 t of radioactive waste (all per tonne of REO) (Navarro and Zhao, 2014; Weng et al., 2013). Processes such as in-situ leaching can also result in water contamination and erosion resulting in landslides that potentially endanger lives (Yang et al., 2013). Moreover, the extraction process is very energy intensive, so that REE production is associated with higher greenhouse gas emissions than many other mined metals (Weng et al., 2013). Both the health and environmental effects can persist long after mining operations have ceased (Yang et al., 2013). In this light, regulation to limit these negative effects is needed, yet, only reliable checks will ensure regulatory effects, and thus, governance within the country in which REE minerals are mined, is key.

Ali (2014) stipulated that recycling can avoid many of the negative environmental and health externalities described and that these should be considered in valuing the secondary supply. In addition, there are also potential positive externalities in developing a secondary supply of REE. For example, overall it is estimated that various recycling activities yield potential for the creation of 580,000 new jobs and for R&D and innovation, thus contributing to EU 2020 targets and to sustaining competitiveness (Meyer and Bras, 2011). Finally, the less tangible value potential from investing in recycling lies in its long-term orientation towards adapting the current economic system. This valuation is built on ideas derived from conceptualizing economics of practice (Bourdieu, 1985) and include broader societal value (Foster, 2006) such as from recycling, reducing and reusing, as part of an economic model constructed on waste prevention—and over the long term, a reduction in resource extraction. Legislative targets will be necessary to drive this transition and to emphasize recycling, as is the case within the EU.

## 5. Conclusion

We have demonstrated that secondary supply has the potential to contribute to supply of phosphors for lamps (and other products). Secondary supply has considerable advantages over primary supply, of which one of the most notable is that it bypasses the extraction phase and many of the environmental impacts and costs involved in this stage. Secondary supply can constitute a source of supply of REE independent of Chinese quotas or licences and as such contribute to supply security of REE phosphors, at least in the short term. Lastly, establishing secondary supplies for recycling is in line with many policy goals in countries that advance closing loops of critical materials for a circular economy.

We have also demonstrated that establishing and encouraging secondary supply requires driving factors. We have pointed to the role of legislation in establishing the collection systems for energy-efficient lamps in Europe and enabling commercial recycling. Our model indicates the rationale for such legislative measures in other regions to increase global recycling rates. Also within Europe energy-efficient lamp collection and recovery of REE from lamp phosphor powders can be improved. The latter step is currently not

required by legislation. In absence of legislative drivers, we have discussed the need for rethinking the value in recycling phosphors. Lastly, our article demonstrates the timely essence for drivers to enable REE recovery potential and to close loops of critical materials for a circular economy.

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### Appendix A.

Input data for primary supply Tables A.1–A.3  
Fig. A.1

**Table A.1**  
Input data for the primary supply analysis (1/2).

Company (REE deposit) <sup>1</sup>	Deposit location	Eu <sub>2</sub> O <sub>3</sub> (t/yr) <sup>2</sup>	Tb <sub>4</sub> O <sub>7</sub> (t/yr) <sup>2</sup>	Y <sub>2</sub> O <sub>3</sub> (t/yr) <sup>2</sup>	Basket Price, USD/t REO	Planned production start <sup>3</sup>	Life of Mine (LOM)	Planned capacity, TREO (tpa)	Total CAPEX, USD	CAPEX Annuity USD/t	Annual OPEX, USD/t	Total annual unit costs, USD/t REO
Lynas (Mount Weld CLD)	Australia	117	20	167	26,780	2015	20	20,000 <sup>4</sup>	612,991,200	3,791	14,636	18,427
Avalon (Nechalacho, av. <sup>5</sup> )	Canada	44	39	780	36,196	2017	20	10,000	1,068,835,426	12,555	22,536	35,090
Tasman (Norra Kärr)	Sweden	19	34	1,842	43,152	2018	20	5,119	378,000,000	8,674	39,690	48,364
Frontier (Zandkopsdrift)	South Africa	118	34	824	28,416	2015	20	20,000	935,057,016	5,492	12,360	17,851
Quest (Strange Lake, av. <sup>6</sup> )	Canada	13	60	2,934	38,656	2020	30	10,424	1,631,000,000	16,598	34,248	50,846
RES (Bear Lodge)	U.S.A.	56	11	112	28,641	2017	45	8,500	453,000,000	8,082	16,995	25,077
Matamec (Kipawa)	Canada	14	20	824	39,522	2016	15.2	3,653	360,502,449	17,492	26,057	43,549
Arafura (Nolans)	Australia	77	17	270	28,144	2019	23	20,000	1,084,209,280	6,103	15,670	21,773
Product purity		99.9%	99%	99.999%								

Sources: Reports available on the websites of the companies.

<sup>1</sup> Molycorp is not included in the analysis due to unavailable data on production costs to perform the profitability check.

<sup>2</sup> Calculation of the supply quantities of considered REO, based on the projects' REE production capacities and relative distribution of individual elements in the selected deposits, latter data stems from TMR (2015).

<sup>3</sup> Year when planned capacity is started or expected to be reached.

<sup>4</sup> Lynas is currently producing 3008 t of REO per year (in 2014) and targeting 11,000 t in 2015.

<sup>5</sup> Averaged Nechalacho Basal and Upper values.

<sup>6</sup> Averaged Strange Lake enriched and Strange Lake Granite values.

**Table A.2**  
Input data for the primary supply analysis (2/2).

Market volumes	La <sub>2</sub> O <sub>3</sub>	CeO <sub>2</sub>	Pr <sub>6</sub> O <sub>11</sub>	Nd <sub>2</sub> O <sub>3</sub>	Sm <sub>2</sub> O <sub>3</sub>	Eu <sub>2</sub> O <sub>3</sub>	Gd <sub>2</sub> O <sub>3</sub>	Tb <sub>4</sub> O <sub>7</sub>	Dy <sub>2</sub> O <sub>3</sub>	Y <sub>2</sub> O <sub>3</sub>	Total
% In mining quota 2014	27.9%	42.7%	4.7%	15.4%	1.6%	0.2%	1.2%	0.1%	0.7%	4.6%	
Volumes (t) <sup>1</sup>	29,179	44,602	4899	16,096	1667	236	1297	140	717	4798	<b>104,545</b>
Mining quota 2014 (t)	<b>104,545</b>										
Mining quota 2015 (expected <sup>2</sup> ), (t)	<b>116,550</b>										
Volumes China 2015 (86%), (t) <sup>3</sup>	32,530	49,724	5462	17,944	1858	<b>263</b>	1446	<b>156</b>	799	<b>5349</b>	<b>116,550</b>
Volumes ROW (14%), (t) <sup>3</sup>	5,296	8,095	889	2,921	303	<b>43</b>	235	<b>25</b>	130	<b>871</b>	<b>18,973</b>
Total current market volumes 1, (t) <sup>4</sup>	<b>37,825</b>	<b>57,818</b>	<b>6351</b>	<b>20,865</b>	<b>2161</b>	<b>306</b>	<b>1681</b>	<b>181</b>	<b>929</b>	<b>6220</b>	<b>135,523</b>
Approx. volumes export official, (t) (assumingly 60%)	8,544	13,060	1434	4,713	488	69	380	41	210	1,405	<b>30,611</b>
Smuggling rate (40%), (t) <sup>5</sup>	5,696	8,706	956	3,142	325	46	253	27	140	937	20,407
Total export volumes (100% equiv. to 86% of world mkt) <sup>6</sup>	14,239	21,766	2391	7,855	814	115	633	68	350	2341	51,018
Volumes ROW (14%), (t)	5,296	8,095	889	2,921	303	43	235	25	130	871	18,973
Total current market volumes 2, (t) <sup>7</sup>	<b>19,535</b>	<b>29,860</b>	<b>3280</b>	<b>10,776</b>	<b>1116</b>	<b>158</b>	<b>868</b>	<b>94</b>	<b>480</b>	<b>3212</b>	<b>69,992</b>

<sup>1</sup> Chen (2014).

<sup>2</sup> Assuming an increase of 11% similar to the increase in 1st batch production quota from 2014 to 2015 (Argus Media).

<sup>3</sup> Assuming similar distribution of REO as in the mining quota (see line 3).

<sup>4</sup> To be used in price model (available from Klossek) as current market volumes in the total Chinese REE mining quota approach.

<sup>5</sup> 40% of Chinese REEs being sold are illegally sourced (Argusmedia, 2015).

<sup>6</sup> Assumed current market volumes for the price model—assuming that overall exported volumes stay the same (as per TMR estimate), a part of illegal REO will be sold via official channels.

<sup>7</sup> To be used in price model (available from Klossek) as current market volumes in the export quota and illegal supply approach

**Table A.3**  
Base case, supply—mining quota and export quota & illegal, 3 year delay scenario.

	2015	2016	2017	2018	2019	2020
ROW current (t) (see Table A.2—line 7)						
Eu <sub>2</sub> O <sub>3</sub>	43	43	43	43	43	43
Tb <sub>4</sub> O <sub>7</sub>	25	25	25	25	25	25
Y <sub>2</sub> O <sub>3</sub>	871	871	871	871	871	871
China (t) (see Table A.2—line 6)						
Eu <sub>2</sub> O <sub>3</sub>	263	263	263	263	263	263
Tb <sub>4</sub> O <sub>7</sub>	156	156	156	156	156	156
Y <sub>2</sub> O <sub>3</sub>	5,349	5,349	5,349	5,349	5,349	5,349
(1) Mining quota approach						
Supply volumes (ROW) based on model (t)						
Eu <sub>2</sub> O <sub>3</sub>			42	219	219	219
Tb <sub>4</sub> O <sub>7</sub>			7	51	51	51
Y <sub>2</sub> O <sub>3</sub>			61	968	968	968
Projects on the market			Lynas <sup>1</sup> —11,000 t	Lynas—22,000 Frontier	Lynas Frontier Matamec <sup>2</sup>	Lynas Frontier RES <sup>2</sup> Avalon <sup>2</sup>
Total world production (t) (supply volumes + ROW current + China)						
Eu <sub>2</sub> O <sub>3</sub>	306	306	348	525	525	525
Tb <sub>4</sub> O <sub>7</sub>	181	181	188	232	232	232
Y <sub>2</sub> O <sub>3</sub>	6,220	6,220	6,281	7,188	7,188	7,188
Allocated to LEDs, CFLs and LFLs (90% <sup>3</sup> ) (t)						
Eu <sub>2</sub> O <sub>3</sub>	275	275	313	472	472	472
Tb <sub>4</sub> O <sub>7</sub>	163	163	169	209	209	209
Y <sub>2</sub> O <sub>3</sub>	5,598	5,598	5,653	6,469	6,469	6,469
(2) Export quota & illegal supply approach						
Supply volumes (ROW) based on model (t)						
Eu <sub>2</sub> O <sub>3</sub>			42	101	101	101
Tb <sub>4</sub> O <sub>7</sub>			7	17	17	17
Y <sub>2</sub> O <sub>3</sub>			61	144	144	144
Projects on the market			Lynas <sup>1</sup> —11,000 t	Lynas—22,000 Frontier <sup>2</sup>	Lynas Matamec <sup>2</sup>	Lynas RES <sup>2</sup> Avalon <sup>2</sup>
Total world production (t)						
Eu <sub>2</sub> O <sub>3</sub>	306	306	348	407	407	407
Tb <sub>4</sub> O <sub>7</sub>	181	181	188	198	198	198
Y <sub>2</sub> O <sub>3</sub>	6,220	6,220	6,281	6,364	6,364	6,364
Allocated to LEDs, CFLs and LFLs (90% <sup>3</sup> ) (t)						
Eu <sub>2</sub> O <sub>3</sub>	275	275	313	366	366	366
Tb <sub>4</sub> O <sub>7</sub>	163	163	169	178	178	178
Y <sub>2</sub> O <sub>3</sub>	5,598	5,598	5,653	5,728	5,728	5,728

<sup>1</sup> This project is not profitable with current production rate but producing (planning production rate increase).

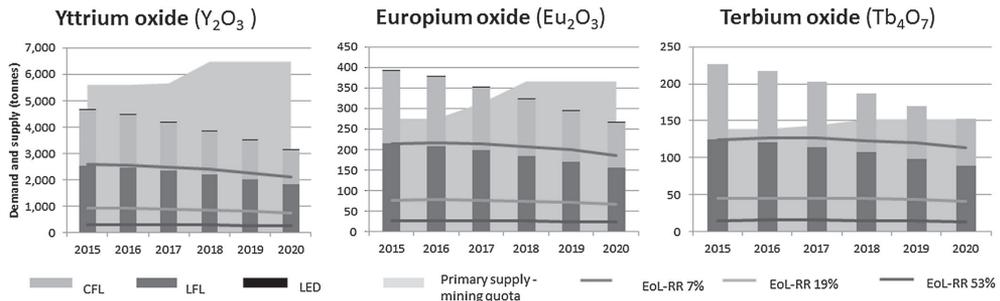
<sup>2</sup> This project does not enter the market (or exit) due to the expected (or actual) economic unfeasibility resulting from the price decrease.

<sup>3</sup> According to Balachandran (2014), 90% of phosphors are used in energy efficient lamps.

The results presented in this table are illustrated in Fig. 4 for approach (2) and in Fig. A.1 for approach (1).

The figure compares secondary supply to demand and primary supply. Potential secondary supply distribution for Y<sub>2</sub>O<sub>3</sub>, Eu<sub>2</sub>O<sub>3</sub> and Tb<sub>4</sub>O<sub>7</sub> based on our three EoL-RR as compared to demand (bars) and 3 year delay base case primary supply forecast (grey

shading) from 2015 to 2020. This figure is based on approach 1 of the primary supply forecast which uses the total REE mining quota in China. Please note different y-axis scales. Source: authors.



**Fig. A.1.** Base case, total Chinese REE mining quota approach, 3 year delay scenario.

Input data for primary supply.

Prices (USD/kg)	La <sub>2</sub> O <sub>3</sub>	CeO <sub>2</sub>	Pr <sub>6</sub> O <sub>11</sub>	Nd <sub>2</sub> O <sub>3</sub>	Sm <sub>2</sub> O <sub>3</sub>	Eu <sub>2</sub> O <sub>3</sub>	Gd <sub>2</sub> O <sub>3</sub>	Tb <sub>4</sub> O <sub>7</sub>	Dy <sub>2</sub> O <sub>3</sub>	Y <sub>2</sub> O <sub>3</sub>
Base case (3 yr delay of production start) for export quota & illegal										
2015	9.000	4.200	108.000	63.000	5.000	600.000	49.000	815.000	390.000	12.500
2016	9.000	4.200	108.000	63.000	5.000	600.000	49.000	815.000	390.000	12.500
2017	8.199	3.726	95.937	55.533	4.256	473.153	44.532	756.914	374.416	12.268
2018	7.304	3.225	83.154	47.744	3.533	366.507	39.567	689.296	354.896	11.962
2019	7.304	3.225	83.154	47.744	3.533	366.507	39.567	689.296	354.896	11.962
2020	7.304	3.225	83.154	47.744	3.533	366.507	39.567	689.296	354.896	11.962
Base case (3 yr delay of production start) for mining quota										
2015	9.000	4.200	108.000	63.000	5.000	600.000	49.000	815.000	390.000	12.500
2016	9.000	4.200	108.000	63.000	5.000	600.000	49.000	815.000	390.000	12.500
2017	8.568	3.941	101.415	58.909	4.586	527.030	46.586	783.930	381.793	12.379
2018	7.177	3.210	83.090	47.828	3.510	349.917	37.925	635.969	319.959	10.816
2019	7.177	3.210	83.090	47.828	3.510	349.917	37.925	635.969	319.959	10.816
2020	7.177	3.210	83.090	47.828	3.510	349.917	37.925	635.969	319.959	10.816

Best case, supply—mining quota and export quota &amp; illegal, 1 yr delay.

	2,015	2,016	2,017	2,018	2,019	2,020
ROW current (t) (see Table A.2—line 7)						
Eu <sub>2</sub> O <sub>3</sub>	43	43	43	43	43	43
Tb <sub>4</sub> O <sub>7</sub>	25	25	25	25	25	25
Y <sub>2</sub> O <sub>3</sub>	871	871	871	871	871	871
China (t) (see Table A.2—line 6)						
Eu <sub>2</sub> O <sub>3</sub>	263	263	263	263	263	263
Tb <sub>4</sub> O <sub>7</sub>	156	156	156	156	156	156
Y <sub>2</sub> O <sub>3</sub>	5,349	5,349	5,349	5,349	5,349	5,349
(1) Mining quota approach						
Supply volumes (ROW) based on model (t)						
Eu <sub>2</sub> O <sub>3</sub>	42	219	219	219	219	219
Tb <sub>4</sub> O <sub>7</sub>	7	51	51	51	51	51
Y <sub>2</sub> O <sub>3</sub>	61	968	968	968	968	968
Projects on the market	Lynas <sup>1</sup> —11,000 t	Lynas—22,000 Frontier	Lynas Frontier Matamec <sup>2</sup>	Lynas Frontier RES <sup>2</sup> Avalon <sup>2</sup>	Lynas Frontier Tasman <sup>2</sup>	Lynas Frontier Arafura <sup>2</sup>
Total world production (t) (supply volumes + ROW current + China)						
Eu <sub>2</sub> O <sub>3</sub>	348	525	525	525	525	525
Tb <sub>4</sub> O <sub>7</sub>	188	232	232	232	232	232
Y <sub>2</sub> O <sub>3</sub>	6,281	7,188	7,188	7,188	7,188	7,188
Allocated to LEDs, CFLs and LFLs (90% <sup>3</sup> ) (t)						
Eu <sub>2</sub> O <sub>3</sub>	313	472	472	472	472	472
Tb <sub>4</sub> O <sub>7</sub>	169	209	209	209	209	209
Y <sub>2</sub> O <sub>3</sub>	5,653	6,469	6,469	6,469	6,469	6,469
(2) Export quota & illegal supply approach						
Supply volumes (ROW) based on model (t)						
Eu <sub>2</sub> O <sub>3</sub>	42	101	101	101	101	101
Tb <sub>4</sub> O <sub>7</sub>	7	17	17	17	17	17
Y <sub>2</sub> O <sub>3</sub>	61	144	144	144	144	144
Projects on the market	Lynas <sup>1</sup> —11,000 t	Lynas—22,000 Frontier <sup>2</sup>	Lynas Matamec <sup>2</sup>	Lynas RES <sup>2</sup> Avalon <sup>2</sup>	Lynas Tasman <sup>2</sup>	Lynas Arafura <sup>2</sup>
Total world production (t)						
Eu <sub>2</sub> O <sub>3</sub>	348	407	407	407	407	407
Tb <sub>4</sub> O <sub>7</sub>	188	198	198	198	198	198
Y <sub>2</sub> O <sub>3</sub>	6,281	6,364	6,364	6,364	6,364	6,364
Allocated to LEDs, CFLs and LFLs (90% <sup>3</sup> ) (t)						
Eu <sub>2</sub> O <sub>3</sub>	313	366	366	366	366	366
Tb <sub>4</sub> O <sub>7</sub>	169	178	178	178	178	178
Y <sub>2</sub> O <sub>3</sub>	5,653	5,728	5,728	5,728	5,728	5,728

This table is presented for evidence of the 1 year delay scenario for approach (1) and (2).

<sup>1</sup> This project is not profitable with current production rate but producing (planning production rate increase)<sup>2</sup> This project does not enter the market (or exit) due to the expected (or actual) economic unfeasibility resulting from the price decrease.<sup>3</sup> According to Balachandran (2014), 90% of phosphors are used in energy efficient lamps.

Worst case, supply–mining quota and export quota &amp; illegal, 5 yr delay.

	2,015	2,016	2,017	2,018	2,019	2,020
ROW current (t) (see Table A.2—line 7)						
Eu <sub>2</sub> O <sub>3</sub>	43	43	43	43	43	43
Tb <sub>4</sub> O <sub>7</sub>	25	25	25	25	25	25
Y <sub>2</sub> O <sub>3</sub>	871	871	871	871	871	871
China (t) (see Table A.2—line 6)						
Eu <sub>2</sub> O <sub>3</sub>	263	263	263	263	263	263
Tb <sub>4</sub> O <sub>7</sub>	156	156	156	156	156	156
Y <sub>2</sub> O <sub>3</sub>	5,349	5,349	5,349	5,349	5,349	5,349
(1) Mining quota approach						
Supply volumes (ROW) based on model (t)						
Eu <sub>2</sub> O <sub>3</sub>					42	219
Tb <sub>4</sub> O <sub>7</sub>					7	51
Y <sub>2</sub> O <sub>3</sub>					61	968
Projects on the market					Lynas <sup>1</sup> –11,000 t	Lynas–22,000 tFrontier
Total world production (t) (Supply volumes + ROW current + China)						
Eu <sub>2</sub> O <sub>3</sub>	306	306	306	306	348	525
Tb <sub>4</sub> O <sub>7</sub>	181	181	181	181	188	232
Y <sub>2</sub> O <sub>3</sub>	6,220	6,220	6,220	6,220	6,281	7,188
Allocated to LEDs, CFLs and LFLs (90% <sup>3</sup> ) (t)						
Eu <sub>2</sub> O <sub>3</sub>	275	275	275	275	313	472
Tb <sub>4</sub> O <sub>7</sub>	163	163	163	163	169	209
Y <sub>2</sub> O <sub>3</sub>	5,598	5,598	5,598	5,598	5,653	6,469
(2) Export quota & illegal supply approach						
Supply volumes (ROW) based on model (t)						
Eu <sub>2</sub> O <sub>3</sub>					42	101
Tb <sub>4</sub> O <sub>7</sub>					7	17
Y <sub>2</sub> O <sub>3</sub>					61	144
Projects on the market					Lynas <sup>1</sup> –11,000 t	Lynas–22,000Frontier <sup>2</sup>
Total world production (t)						
Eu <sub>2</sub> O <sub>3</sub>	306	306	306	306	348	407
Tb <sub>4</sub> O <sub>7</sub>	181	181	181	181	188	198
Y <sub>2</sub> O <sub>3</sub>	6,220	6,220	6,220	6,220	6,281	6,364
Allocated to LEDs, CFLs and LFLs (90% <sup>3</sup> ) (t)						
Eu <sub>2</sub> O <sub>3</sub>	275	275	275	275	313	366
Tb <sub>4</sub> O <sub>7</sub>	163	163	163	163	169	178
Y <sub>2</sub> O <sub>3</sub>	5,598	5,598	5,598	5,598	5,653	5,728

This table is presented for evidence of the 5 year delay scenario.

<sup>1</sup> This project is not profitable with current production rate but producing (planning production rate increase).<sup>2</sup> This project does not enter the market (or exit) due to the expected (or actual) economic unfeasibility resulting from the price decrease.<sup>3</sup> According to Balachandran (2014), 90% of phosphors are used in energy efficient lamps.

## Appendix B. Input data for demand and secondary supply

### Tables B.1–B.5

EU member states show some of the highest rates in the world along with other countries with compulsory collection legislation like Taiwan, which collects and recycles over 75% of lamps (EPA Taiwan, 2014), however Table B.1 demonstrates further potential for improvement in collection rates. Outside the EU there is

less available data, but it is estimated that 95% of fluorescent lamps in Australia are landfilled (FluoroCycle, 2014), while Canada, Japan, Mexico, and South Africa all recycle less than 10% (EU Commission, 2014a,b) The United States has some, mainly state level, laws for management of waste lamps, requiring recycling by business users; however, enforcement is low and the recycling rate in these states is estimated around 23% (Silveira and Chang, 2011).

**Table B.1**  
Demand input data (1/2)—Lamp market and  $Y_2O_3$ ,  $Eu_2O_3$  and  $Tb_4O_7$  content in lamps.

New installations (million) <sup>1</sup>						
	2015	2016	2017	2018	2019	2020
Others	1073	935	831	749	698	610
LFL	659	630	594	562	521	489
CFL	704	653	616	564	509	436
LED	1365	1780	2163	2508	2818	3125
Lamp replacement (million)						
	2015	2016	2017	2018	2019	2020
Others	3469	2654	1975	1467	1150	903
LFL	1560	1512	1447	1346	1229	1106
CFL	2121	2039	1842	1644	1491	1322
LED	560	658	733	848	977	1078
Total lamps on market (million)						
	2015	2016	2017	2018	2019	2020
Others	4542	3589	2806	2216	1848	1513
LFL	2219	2142	2041	1908	1750	1595
CFL	2825	2692	2458	2208	2000	1758
LED	1925	2438	2896	3356	3795	4203
Rare earth in lamps <sup>2</sup>						
	Phosphor (g)	TREO (g)				
LFL	2.34	1.665				
CFL	1.5	1.069				
LED	0.01	0.006				
Composition as per standard tricolor phosphor (%) <sup>3</sup>						
	$Y_2O_3$ (Range 46.9–51.2)	$Eu_2O_3$ (3.9–4.4)	$Tb_4O_7$ (2.2–2.6)			
LFL	Averaged to 49.05	4.15	2.4			
CFL	49.05	4.15	2.4			
LED	49.05	4.15	0			
REO content (calculated from phosphor(g) × averaged REO content (%) above)						
	$Y_2O_3$ (g)	$Eu_2O_3$ (g)	$Tb_4O_7$ (g)	Total RE (g)		
LFL	1.14777	0.09711	0.05616	1.301		
CFL	0.73575	0.06225	0.036	0.834		
LED	0.004905	0.000415	0	0.005		

<sup>1</sup> McKinsey & Company (2012). The data summarizes general lighting applications (residential, office, industrial, shop, hospitality, outdoor and architectural, yet excluding automotive and backlighting) for all world regions.

<sup>2</sup> Castelloux (2014b).

<sup>3</sup> Averaged from Wu et al. (2014), Table 3.

**Table B.2**  
Demand input data (2/2)— $Y_2O_3$ ,  $Eu_2O_3$  and  $Tb_4O_7$  demand per LFL, CFL and LED.

$Y_2O_3$ (t)						
	2015	2016	2017	2018	2019	2020
LFL	2546.90	2458.52	2342.60	2189.95	2008.60	1830.69
CFL	2078.49	1980.64	1808.47	1624.54	1471.50	1293.45
LED	9.44	11.96	14.20	16.46	18.61	20.62
Total (rounded)	4635	4451	4165	3831	3499	3145
$Eu_2O_3$ (t)						
	2015	2016	2017	2018	2019	2020
LFL	215.49	208.01	198.20	185.29	169.94	154.89
CFL	175.86	167.58	153.01	137.45	124.5	109.44
LED	0.79	1.01	1.20	1.39	1.57	1.74
Total (rounded)	392	377	352	324	296	266
$Tb_4O_7$ (t)						
	2015	2016	2017	2018	2019	2020
LFL	124.62	120.29	114.62	107.15	98.28	89.58
CFL	101.7	96.91	88.49	79.49	72	63.29
LED	0	0	0	0	0	0
Total (rounded)	226	217	203	187	170	153
Total demand (t)						
	2015	2016	2017	2018	2019	2020
LFL	2887.01	2786.83	2655.42	2482.38	2276.82	2075.16
CFL	2356.05	2245.13	2049.97	1841.47	1668.00	1466.17
LED	10.24	12.97	15.41	17.85	20.19	22.36
Total (rounded)	5253	5045	4721	4342	3965	3564

**Table B.3**  
Secondary supply—Availability of Y<sub>2</sub>O<sub>3</sub>, Eu<sub>2</sub>O<sub>3</sub> and Tb<sub>4</sub>O<sub>7</sub> as per different EoL-RR.

Y <sub>2</sub> O <sub>3</sub> (t)						
	2015	2016	2017	2018	2019	2020
LFL-EoL	2688.08	2667.42	2616.92	2546.90	2458.52	2342.60
CFL-EoL	2166.05	2161.63	2078.49	1980.64	1808.47	1624.54
LED-EoL						
EoL-RR 7%	320.37	318.72	309.90	298.82	281.62	261.83
EoL-RR 19%	931.99	927.18	901.52	869.29	819.26	761.69
EoL-RR 53%	2582.39	2569.06	2497.96	2408.65	2270.04	2110.52
Eu <sub>2</sub> O <sub>3</sub> (t)						
	2015	2016	2017	2018	2019	2020
LFL-EoL	227.43	225.68	221.41	215.49	208.01	198.20
CFL-EoL	176.54	183.26	182.89	175.86	167.58	153.01
LED-EoL						
EoL-RR 7%	26.66	26.99	26.68	25.83	24.79	23.18
EoL-RR 19%	77.56	78.52	77.63	75.14	72.11	67.43
EoL-RR 53%	214.91	217.56	215.09	208.19	199.81	186.84
Tb <sub>4</sub> O <sub>7</sub> (t)						
	2015	2016	2017	2018	2019	2020
LFL-EoL	130.80	131.98	131.53	130.52	128.04	124.62
CFL-EoL	102.10	105.98	105.77	101.70	96.91	88.49
LED-EoL						
EoL-RR 7%	15.37	15.71	15.66	15.33	14.85	14.07
EoL-RR 19%	44.72	45.69	45.56	44.59	43.19	40.92
EoL-RR 53%	123.90	126.59	126.24	123.54	119.68	113.37

**Table B.4**  
Sensitivity analysis (SA) for lifetimes of lamps.

	Lifetime original (yrs)	SA (yrs)				
CFL	3	5				
LFL	3	5				
LED	11	15				
Y <sub>2</sub> O <sub>3</sub> (t)						
	2015	2016	2017	2018	2019	2020
LFL-EoL	2673.16	2697.26	2688.08	2667.42	2616.92	2546.90
CFL-EoL	1815.83	1953.42	2086.59	2166.05	2161.63	2078.49
LED-EoL	–	–	–	–	–	–
EoL-RR 7%	296.27	306.94	315.13	319.01	315.38	305.28
EoL-RR 19%	861.89	892.93	916.74	928.03	917.48	888.08
EoL-RR 53%	2388.14	2474.16	2540.12	2571.40	2542.19	2460.71
Eu <sub>2</sub> O <sub>3</sub> (t)						
	2015	2016	2017	2018	2019	2020
LFL-EoL	226.17	228.21	227.43	225.68	221.41	215.49
CFL-EoL	153.63	165.27	176.54	183.26	182.89	175.86
LED-EoL	–	–	–	–	–	–
EoL-RR 7%	25.07	25.97	26.66	26.99	26.68	25.83
EoL-RR 19%	72.92	75.55	77.56	78.52	77.63	75.14
EoL-RR 53%	202.05	209.33	214.91	217.56	215.09	208.19
Tb <sub>4</sub> O <sub>7</sub> (t)						
	2015	2016	2017	2018	2019	2020
LFL-EoL	130.80	131.98	131.53	130.52	128.04	124.62
CFL-EoL	88.85	95.58	102.10	105.98	105.77	101.7
LED-EoL	–	–	–	–	–	–
EoL-RR 7%	14.50	15.02	15.42	15.61	15.43	14.94
EoL-RR 19%	42.17	43.69	44.86	45.41	44.89	43.45
EoL-RR 53%	116.85	121.06	124.29	125.82	124.39	120.40

**Table B.5**  
Put on market, collection, and recycling of fluorescent lamps in selected EU countries.

	Put on Market (t) avg 2007–2009	Waste collected (t) 2010	% of put on market collected 2010	% of put on market recycled 2010
Belgium	3,100	1247	40.2	37.5
Denmark	1,606	694	43.2	41.1
France	13,070	3839	29.4	27
Germany	28,204	11,092	39.3	34.4
Greece	1,757	124	7.1	6.6
Sweden	3,141	1973	62.8	62.3

Source: Eurostat (2014).

## Appendix C.

Table C.1

Table C.1  
Cost estimates of different REE products for comparison of REE phosphor from primary and secondary (EoL) supply.

Input	Ore ~0.42 USD/kg (in-situ TREC) <sup>1</sup>	Mine & beneficiation	Cracking & leaching (impurity removal)	REE concentrate (~27.6 USD/kg (in-situ TREC) <sup>1</sup> )	Collected EoL lamps	Material from EoL lamps	EoL lamp recycling	Phosphor powders	EoL REE phosphor concentrate	EoL REE phosphor concentrate
Process				Chemical separation	Sorting			Remaining glass & mercury/phosphor separation	Halophosphates removal	
Output	By gravitational beneficiation concentrated ore (15,000 g/t REO <sup>6</sup> or 0.2–15% grade in deposit)	REE concentrate	REE concentrate	Phosphor powders: Y <sub>2</sub> O <sub>3</sub> (99.999%) 14USD/kg <sup>2</sup> Eu <sub>2</sub> O <sub>3</sub> (99.9%) 25USD/kg Tb <sub>2</sub> O <sub>3</sub> 620USD/kg	Sorted EoL lamps	Sorted glass/metal/plastic	Phosphor powder	Concentrated phosphors (230,000 g/t REO <sup>6</sup> or 10–20 wt% REO <sup>7</sup> ) or 15% (CMI, 2014) or 30% tricolour phosphor powder content in recycled phosphor <sup>8</sup>	REE phosphor concentrate	REE phosphor concentrate
Cost of treatment					btw. 0.15 and 2 EUR/kg for collection & recycling (WEEE forum)	Class = net	Selling the powder to Solvay is slightly less the cost of landfilling with Solvay paying transport costs <sup>5</sup>	Price reductions for mercury and glass content in phosphor powder. Separating Hg out is very expensive <sup>6</sup>		REE extraction from mercury phosphor ~6 USD/kg <sup>3</sup>
(Avoided) environ-mental cost	63,000 m <sup>3</sup> waste gas; 200 m <sup>3</sup> acidic water; 1.4 t of radioactive waste (all per t of REO) <sup>4</sup> Note: 'Impurity removal' refers to the removal of radioactive elements (if present in the REE-bearing mineral host rock)					Metals = very small positive return Plastics = small cost for incinerating	~1–2 USD/kg (or SEK 7–15/kg) to landfill phosphor powder <sup>5</sup>	~2500–1.1 million USD/kg Hg isolated from biosphere <sup>6</sup>		

Notes: Current hydrometallurgical processes applied recover 15 wt. pct. of rare-earth metals as oxides contained in phosphor dust (CMI Aug 2014 presentation).

<sup>1</sup> Ad glass: not clean enough for high-end uses—generally NOT used in new lamps.

<sup>2</sup> Ad metals: must be pre-treated and separated by magnets and flotation.

<sup>3</sup> As of TMR database, average of data from TMR for 7 junior REE projects (as per Table A.1., excluding Lynas and China), whereby the listing of two projects by one company has been summed up and averaged.

<sup>4</sup> Prices accessible from price charts to non-subscribed users of metal-pages, Oct 2014.

<sup>5</sup> Cardarelli (2014) in Chemical Engineering (2014) who originally states in the context of 98–99% recovery that 'REEs can be extracted for as low as USD 7/kg of mercury phosphor dust', presumably since it is a Canadian based firm, the price is indicated in CAD.

<sup>6</sup> Najarro and Zhao (2014), Weng et al. (2013).

<sup>7</sup> interview D (2014).

<sup>8</sup> MLR (2002); Noble, 2013 in Tan et al. (2014)

<sup>9</sup> Otto and Wojtalowicz-Kasprzak (2012).

<sup>10</sup> Khetriwal et al. (2011).

<sup>11</sup> Hylander and Goodsite (2006).

The recycling process is cheaper the better and more homogenous the powder quality is. The sorting and collection can influence the efficiency and cost so that enabling a sourcing of the same quality and type of lamp phosphors could make the REE powder recycling process more efficient and cheaper, as pointed to in Binnemans and Jones (2014) with the three recycling routes, yet requires either smaller loops from companies taking back their own product or better separation of lamp types.

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# Paper III







Article

# Governance and Risk–Value Constructions in Closing Loops of Rare Earth Elements in Global Value Chains

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**Abstract:** This article addresses a research gap on the challenges—specifically risk and value—connected to realizing the potential for closing loops for rare earth elements (REE). We develop an analytical framework from conceptual elements of the global value chain (GVC) framework and the relational theory of risk to examine several empirical REE industry cases for loop closure. The aim of the paper is to identify how risk–value relationships are constructed by different actors as governance structures form in transactions prior to price setting and how these have impacts on the closure of REE loops. Often, REE loops are not closed, and we find that constructions of the risk–value relationship by industrial actors and by government agencies are unstable as they pursue different motivations, consequently hindering REE loop closure in GVCs. In light of this, we propose that governments mediate against the construction of risk–value relationships by facilitating information on the characteristics of end-of-life materials that qualify these for re-entry into loops.

**Keywords:** rare earth elements; recycling; risk; value; governance; global value chain; transaction

## 1. Introduction

The closure of material loops has come to be central to circular economy debates, but conventional literature hardly discusses the role of risk, especially where low recycling rates indicate the absence of loop closure. More often than not, lack of loop closure is explained in terms of uneconomic processes reflected in price signals, i.e., where prices of an output material per volume of a recycling process are framed as determinants of feasibility as they are compared to primary processing routes. This focus on output-price, however, ignores the dynamics that occur prior to, and in shaping the formation of prices; in particular, the construction of risk and value in the context of transactions between the different actors that participate in the processing segments. Importantly, focus on output rather than process hinders understanding of the supporting mechanisms needed for transiting to a circular economy in which loop closure is one constituent. This paper contributes an analysis of transaction processes in three case studies in which risk and value is constructed in a way that affects loop closure as a first step towards addressing this wider research gap.

At least since the late 1970s, the ideas of circular economy (CE) have been gaining momentum amid concerns about sustainability of material and mineral-dependent lifestyles [1–3]. Pearce and Turner [4] first conceptualized the notion of a CE in ecological economics. Various schools of thought have engaged with it since, including cradle-to-cradle [5], systems thinking [6] and closed-loop approaches to production processes that are integral to industrial ecology in which industrial waste serves as input to another industry [7]. While the understanding of the CE concept has evolved to

incorporate different features and concepts from the above schools see [8], most share the idea of closed loops [9].

Recently key institutions have advocated for a transition to a CE, which maintains the value of resources, emphasizing durability and circularity, in juxtaposition to the linearity of “take-make-dispose” models [10–14]. The annex to the European Union (EU) Action Plan for the CE for instance, outlines actions for “closing the loop” of product lifecycles through greater recycling and re-use which focuses on recycling target increases for municipal and packaging waste, landfill reduction targets and bans, improvements in definitions and calculations of recycling methods, as well as a focus on industrial symbiosis and economic incentives for producing greener products [15]. In the EU, the CE has also been linked to addressing raw material criticality, and as such critical materials are specifically targeted in the EU’s CE Action Plan in Section 5.3 [15].

The European Commission [16] defines raw material criticality as a material that: (1) faces high risk with regard to access (i.e., high supply or environmental risks); and (2) is of high economic importance; such that there is a risk of interruption of supply that could significantly affect the economy. Both the U.S. [17] and EU assess criticality regularly and since 2010 have compiled lists of critical materials which both include rare earth elements (REE), which includes 14 of the 15 lanthanide elements (promethium is not assessed), yttrium and scandium [18]. A subsequent EU assessment [19] further grouped heavy (HREE), light (LREE) and scandium. The EU also launched a European Rare Earths Competency Network (ERECON) to examine how the supply chain for rare earths can be strengthened [20]. One of the working groups within ERECON was specifically focused on EU REE resource efficiency and recycling, highlighting the link with CE strategies that could potentially mitigate such risks through eco-design and closing material loops for critical materials.

An understanding of the materiality of our economies is essential to closing the loop of materials and needs to extend beyond a physical-material focus. The Multi-Stakeholder Platform for a Secure Supply of Refractory Metals in Europe [21] argues that a “valorization of the resources” is required through “coordination and networking between researchers, entrepreneurs and public authorities”. However, CE research still largely emphasizes physical flows [22]. A lack of analysis of social and institutional factors constitutes a barrier to further development of the CE [23]. Reck and Grædel [24] stated that social behavior poses one of the limitations for closing material cycles. Without the social dimension, the “*how*” and “*why*” of materials flows remain unanswered. These questions need exploring to understand how metal recycling rates can be improved [25].

Studies based on methodologies centering on physical flows cannot explain why material loops are frequently not closed even when there are demonstrated stocks and technological feasibility is proven at lab-scale. What is required is an understanding of the interaction between individual actors in the market [26]. This involves opening up the “black box” of the firm to study “circulation processes” [27]. Barriers and enabling factors for facilitating such flows can then be identified along with the socio-institutional change required to transit to a CE [28]. Risk–value constructions may play a significant role in circulation processes, even more so if these concern the reintroduction of material into processing loops [29]. Lepawsky and Billah [30] make an appeal to the value chain and network scholars to rethink how the capture and creation of value is theorized, proposing that “waste” and “value” be thought relationally.

This paper aims to progress interdisciplinary understanding for scholars, researchers and policy-makers on two aspects. Firstly, it offers an insight into how industrial actors conceive of a risk, an object at risk and form a relationship of risk between the former two at specific segments of a chain/production network. Secondly, it explores how this constructed risk emerges amid governance structures that form in transactions, revealing some of the power dynamics at play. To achieve this, the paper brings the governance structures of the global value chain (GVC) conceptual framework [31] into conversation with the relational theory of risk [32]. This serves to showcase how a focus on transactions in the GVC can bring about useful insights for policy and it reaffirms the social construction

of risk. In so doing, the paper aims to create an understanding of how and why governance structures and risk communication decisively influence whether or not REE-material loops are closed.

The REE have manifold uses in applications spanning civil, industrial and military use, as components or dopants due to their specific chemical and physical properties. Close to ten different industrial sectors have been delineated that rely on REE input [33]. Among the most widely cited REE uses are permanent magnets, as well as applications that draw on the fluorescent properties of REE, such as (background-) lighting, including in housing but also in electronic equipment that relies on screens, such as computers, smart phones and tablets.

Empirically we examine the relevant GVC segments for three REE recycling case studies. The first case study is that of a proposed chemical separation facility. The second is magnets and the third phosphor powder from EoL lamps (which contain mostly HREE including terbium, europium, yttrium but can also include LREE such as cerium, to a lesser extent). Both magnets and phosphors from lamps are specified as REE priority sectors in ERECON [20]. Our conceptual lens is applied to each in order to explain how industrial actors construct risk at these segments and why loops are not closed despite available, lab-scale tested recycling technology and support from publicly-funded projects. As Balomenos [34] outlined, the EU spent close to 90 million EUR on REE projects over the past five years leading up to 2017. Binnemans et al. [35] cite the improvement of REE recycling as “an absolute necessity” for reasons of their supply risk, economic importance, and the “balance problem” and Binnemans [36] recommends legislative adaptation of recycling directives to account for minor metals. In light of a demonstrated potential for recovery of REE in anthropogenic deposits [37–39], the reason for low REE recycling rates is framed as “a lack of incentives” [37,40–42]. Questions of risk and value have recently begun to be explored in relation to steel and REE respectively [43–47]. However, this is the first paper to apply the relational theory of risk to empirical evidence from the REE industry and to systematically analyze the construction of risk–value relationships in the context of forming governance structures as transactions take place in the GVC of REE.

The paper is structured into six sections. In Section 2, we describe the conceptual elements that underpin the development of our analytical framework. Section 3 describes the methodology and in Section 4 we analyze three empirical cases from the REE-industry. In Section 5, we offer a discussion and we conclude in Section 6.

## 2. Linkages, Boundaries and Risk: Exploring the How and Why of Material Flows

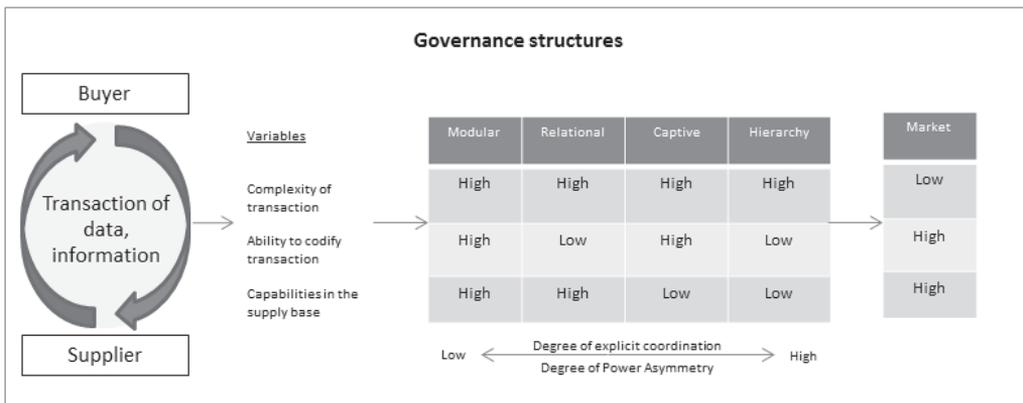
From a (bio-)physical perspective, the transformation of geogenic into anthropogenic resources involves numerous segments at which processing occurs, including the transformation of mined mineral-containing rocks to beneficiated minerals, to separated elements, components and their assemblies, to final products, their collecting, sorting and reintroducing into material transforming processes. At the minimum, some of these segments represent the baseline for mapping material stocks, processes and flows [48] such as, in its broadest sense, in input–output/material flow analyses (MFA) [49,50] but also in studies of a global scale [24]. However, flows of resources and materials do not just occur. They result from decisions made in multidimensional interaction. This interaction may involve different actors, namely individuals, usually associated with a firm, which activities are tied to the segments in which transformations of resources into materials and reprocessing take place.

Global Value Chain (GVC) analysis explores and analyses the interaction of firms, specifically their transactional characteristics, at particular processing steps or so-called segments. It is a parallel school of thought to Global Production Networks (GPN), which also originates in studies of commodity chains and world systems theory [51]. Questions such as how and why materials flow in a particular way and who is affected advantageously or disadvantageously by these flows are central to GVC analysis. The GVC framework therefore enables us to identify where loopholes exist for material loop closure. In addition to mapping the input–output structure of particular GVCs, a focus is on delineating the geographical location of a segment of transformation. This is a central feature, as it reveals the geography of material flows, as well as market shares in specific segments of the GVC of

particular countries and their firms. Lepawsky [52] demonstrates the limitations of statistical analysis of Comtrade data for explaining e-waste trade networks, specifically for understanding, “the purposes for which such trade occurs or the end to which the commodity so traded is put (e.g., final disposal, reuse, recycling or recovery)”, and emphasizes the importance of empirical studies.

While the mapping of activities provides a macro-perspective of particular GVCs, the scholarly and policy discourses could benefit from connecting this perspective to the meso- and micro-level analyses to understand how different geographical outcomes arise. This is where the conceptual framework of GVC can assist: GVC analysis places emphasis on governance. It is a central conceptual and analytical element of GVC analysis, and, put differently, rests on an identification of forms of coordination and control among firms. Specifically, decisions on material input and output at GVC segments are informed by governance structures and affect manufacturing process and End-of-Life (EoL) product handling, including recycling. Governance structures are key to gaining an understanding of how material loops can be closed.

The GVC framework enables an analysis of how these forms of coordination and control come into place [31,53]. It provides three variables for a given transaction between a buyer and a supplier to examine how a transaction takes effect: (i) the complexity of the transaction; (ii) the ability to codify transactions; and (iii) capabilities in the supply-base (see Figure 1). These GVC variables are allocated a high or low value to derive five governance structures, as depicted in Figure 1. Market and hierarchy structures are at the extreme ends, where price determines a transaction at the former and the acquisition of one firm by another defines the latter. Essentially these structures define the limits of transactions i.e., whether these are effectuated within firm boundaries, among particular firms, or, in principle, accessible for all potentially interested firms in a national or global environment. The latter also gives rise to the regulatory framework, the fourth analytical dimension of the GVC framework. In between these extreme forms of coordination and control are network forms of governance—modular, relational and captive structures—in which the buyer–supplier interaction is seemingly less skewed towards one actor of the transaction [31].



**Figure 1.** Governance structures in the global value chain (GVC) framework. Source: modified after [31] (p. 87).

Recycling activities have become the subject of scholarly focus as these appear to be underpinned by dynamics different to those known from conventional linear models. Crang et al. [54] challenge the accuracy of the described governance forms for different supply–demand dynamics in recycling activities, i.e., as supply arises independent of, and not in response to demand when products reach their EoL and are reintroduced into material cycles through reuse or recycling. The authors demonstrate the prevalence of brokered forms of governance in recycling networks, the “co-ordination from the middle by brokers”, tied to the “heterogeneous materiality” of used goods [54]. Lepawsky and Billah [30] showed that the trade in EoL electronics lacks the formal systems of control that standardize

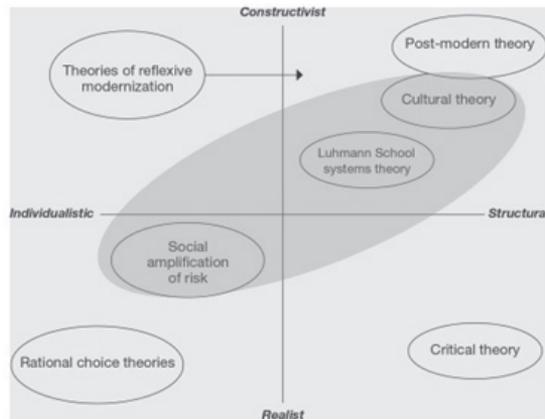
the commodity in terms of quality or that adjudicate disputes in cases of unsatisfactory exchanges. They emphasize the significance of “personal attention” by Bangladeshi rubbish electronic importers to their shipments. The prevalence of brokers [54] and the “personal attention” to shipments of waste electronics [30] point to a form of coordination and control with similarities to the relational governance structure. The observations of these authors also imply that some risk–value construction linked to the quality of the materials in a particular transaction is considered by brokers to be worth managing.

The presence of product or process standards can alleviate this risk, as empirically demonstrated and manifested in the market and modular governance structures of the GVC framework. As Humphrey and Schmitz [55] (p. 23) pointed out, “*the main reason for specifying process parameters along the chain is risk*”. As firms engage in non-price competition, these performance risks augment [55]. These performance risks include continuity and consistency of supply, and the conformity of a product to a standard [55]. Standards have the potential to determine the transactional characteristics, particularly in market and modular governance forms of the GVC framework where there is a specific point of transaction, in other words, a clear handover point between the buyer and the supplier. In the networked governance forms, a transaction continues to center on the provision of information as well as price. Gregson et al. [10] emphasize the importance of quality outputs from recycling processes and the challenge in meeting quality standards for recyclates. In a study of the recycling of steel from demolished buildings, Santos and Lane [29] suggest that the lack of standards for EoL construction materials, which would “express” their material quality to suppliers and clients, is a significant barrier to their reintroduction into material processing. Their point echoes that of Crang et al. [54] who further emphasize that standards and the related classification of goods and materials as hazardous waste, affect the movement of recycled goods by mandating particular forms of processing. Santos and Lane [29] (p. 46) conclude that the building construction regime is characterized by “a particular set of practices” which excludes reused steel components. Transactional characteristics, such as those discussed on certain industries, in which risk and value considerations are constructed, may shed light on particular practices and help to explain the lack of material loop closure.

In this paper, we therefore combine the governance structures arising in transactions with the relational theory of risk [32] and apply this to our REE case studies. The aim of this marriage of two theoretical frameworks into one is to support the specific objective of this paper, namely to inform on the pre-price forming dynamics in transactions between a buyer and a supplier of material. The transaction is key to both the governance structures as well as to the risk construction, where the transaction is that of information and data.

The relational theory of risk builds on the work of Hilgarnter [56] who argued for a shift in focus from asking “*What is risk?*” to “*How do people understand something as a risk?*”, and on similar arguments developed other scholars of risk [57–59]. This constructivist perspective of risk, which focuses on how risk is constructed by the various actors, takes risk assessment as an inherently normative evaluation [32,57]. Importantly, the description of a risk object—which may take the form of a physical, cultural or social artifact—necessarily involves ascribing it “some value” [32] (p. 177). Corvellec [60] also argues that value is derived from organizational practice which coincides with the theoretical underpinnings of the GVC framework.

In the risk taxonomy [61] (Figure 2), we situate the relational theory of risk [32] predominantly in the upper right corner of the systems and cultural theory. Systems theory, most prominently argued by Luhmann [62], works with risk as a social construct, alongside the importance of system boundaries and the focus on the “communication between systems” [61]. Cultural theory also works with risk as a social construct and places analytical emphasis on cultural patterns, and therefore extends beyond the focus of this paper. The relational theory of risk [32] also seems to point to a link with the social amplification of risk, namely that of causal relations and the integration of different perspectives of risk, allowing a systematic analysis of empirical findings.



**Figure 2.** Review of sociological approaches to risk. Source: [61] (p. 40); oval highlight added to situate the relational theory of risk [32].

The analytical focus of the relational theory of risk is the communication in which risk is semantically created [32] (p. 186). This is also emphasized by Howes [63] who noted that “risks are partially socially constructed by discourse”. It places the actor center-stage to explain the dynamic of how risk is constructed by drawing on a “tripartite deconstruction of risk elements” namely the semantic networks that contain “objects at risk”, “risk objects” and “relationships of risk”, and their evolution over time and in space [32]. These three elements are further explored hereafter.

The risk objects are characterized by a fluid, dangerous identity. Risk is introduced into the social space when an object is designated as risky from which it swiftly becomes independent while remaining tied to social practices and representations [32]. As societies evolve, so does understanding and formulation of values as well as of dangers with the result that the definition of risk objects also changes.

The objects at risk are endowed with a value at stake. Here the reference to value bypasses moral judgment of good or bad, moving to “something that is held to be of worth” and might be nature, life, principles or a state of affairs. This is in stark contrast to presenting value solely as a monetary unit. Boholm and Corvellec [32] (p. 180) pinpoint that “objects at risk are constituted around traits such as value, loss, vulnerability, and need for protection”. Thus, designating an object at risk is equivalent to assigning value. Allwood et al. [1] acknowledge the involvement of a wider range of values in their work on material efficiency. Bocken et al. [64] also consider the value proposition between and amongst different stakeholders, including network actors (e.g., firms, suppliers, etc.), customers, society and the environment. This is useful in demonstrating how values can be allocated or traced amongst different stakeholders.

At times, agreement in society is reached about what is valued, and what objects are perceived to be at risk, and how, taking a normative turn, these should be protected. It is here where government, including with its legislative arm, plays a significant role [32] (p. 180). Indeed, there are many established environmental setting targets for businesses to deal with the environmental impacts of their products; for example, eco-design policies (see e.g., [65]), top-runner programs (see e.g., [66]) and extended producer responsibility policies (see [67]). This is reiterated by Porter and Kramer [68] who recommend that governments learn how to regulate in ways that enable, what they define as “shared value”. Notably, Porter and Kramer [68], while presenting some interesting criticism of neoclassical theory, specifically that of Milton Friedman, continue to work with a growth paradigm in which economic and social progress, are separate phenomena, a clear friction with the theoretical underpinning of this paper where these are inseparable. In this paper, individuals are viewed as engaging in transactions, independent of whether they represent a firm, a customer, a government

or are associated with another organizational form. When Gereffi and Korzeniewicz [69] first conceptualized commodity chain analysis, they were partially inspired by M.E. Porter [70], specifically his “value chain” as opposed to “value added” notion which allowed an exploration of linkages among economic activities. Gereffi and Korzeniewicz [69] merged these elements with some from sociology to add explanatory potential to the framework for the different socio-economic outcomes of what is now known as global value chains in the field of economic geography.) They specifically point to regulatory measures that work with attainable, yet ambitious targets rather than prescribing a particular mode of reaching these. However, critics of Porter and Kramer point out that reaching consensus in practice might be more difficult, not least because of the complexity of value chains and how systemic problems are perceived by organizations [71].

Both popular and scholarly narratives can galvanize societal agreement on the object of value and at risk, as well as the approaches towards protection. This is evident in the shifting narratives that frame materials as either useful resources or waste [72]. Another example is the classification of resources into either primary or secondary materials versus a singular category that embraces both. Moreau et al. [23] refer to geogenic and anthropogenic resources while Mueller et al. [44] do not emphasize origins of resources but turn to the issue of “accessibility” of resources. Similarly, the term “stocks” in industrial ecology bridges the natural and the social when, for example, stocks of metals are examined. Here, the term “urban mines” enables a comparison of anthropogenic stocks to geological occurrences. Interestingly, it is here where the different ontologies of the social- and material-/natural sciences come to the forefront. These examples pinpoint scholarly efforts to create narratives to bridge rather distinct worldviews between schools of thought, i.e., whether a particular geogenic occurrence exists without human interference, and is therefore “natural”, or whether it exists solely because of human action, as we “socially construct” it by conceptualizing, describing, and classifying.

Striving towards a co-existence of conceptual understandings seems to offer most positive outcomes, especially as any discourse will inevitably transport particular values of any school of thought participating in it. In other words, when interdisciplinary discourses turn to depicting value including in distorted conceptualizations of the GVC, more often than not, the default understanding will be that value is reflected as the price of a material (and other value, i.e., environment remains vaguely associated).

Relationships of risk are observer established, see Figure 3: When an observer constructs “a link” between a risk object and an object at risk, whereby the former is understood as potentially threatening the value of the latter, then a relationship of risk is present [32]. Critical to the conceptualization of the relationship of risk in the relational theory of risk is understanding that the relationship is a construct. It needs to be made and crafted, and this process occurs by “semantic association between objects” [73].

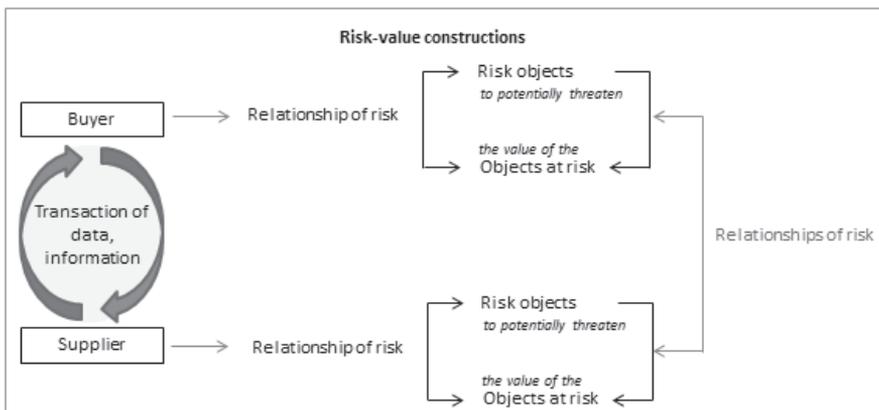
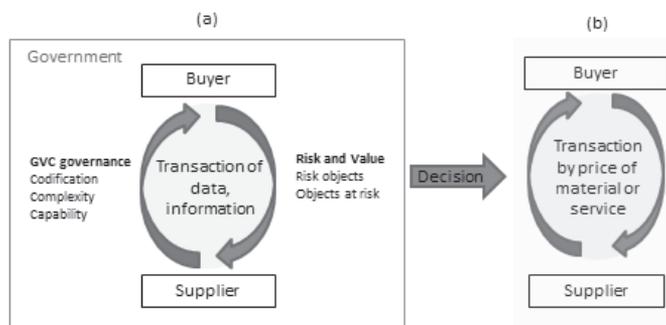


Figure 3. Risk–value constructions. Source: adapted from [32].

Examples of these constructions are models, laboratory tests, narratives or probabilities [32]. The parameters of risk relationships are contingency, causality and action and decisions to act. Exploring these parameters in more depth leads firstly to the “What if?” question which is central to the relationship of risk, and contingent in as far as risk describes a potentiality of occurrence rather than a certainty. Secondly, it is evident that the relationship of risk needs to establish the causality between the risk object and object at risk. Thirdly, action and decisions to act are key to the relationship of risk. As Boholm and Corvellec [32] (p. 181) put it, “*Risk is conditioned by a modern will to know that remains welded to a will to decide and act under conditions of uncertainty*”. It is the assembly and reciprocity in the form of a causal-contingent relationship that allows risk to be established. The continuous reframing and redefinition shapes relationships of risk, as well as the coexistence of various relationships of risk that reflect diverse views, cultures and knowledges embedded in society. Thus, as Boholm and Corvellec [32] (p. 182) summarize “*What is a risk object for some can be an object at risk for others*”.

### 3. Methodology and Data

Our analytical framework is built from the conceptual elements of governance derived from the GVC framework, and from the risk–value constructions of the relational theory of risk [31,32]. We conceptualize both of these elements as arising in transactions of data and information between interacting individuals or entities, here simplified as the buyer and the supplier. The transaction occurs prior to a (contractual or informal) agreement to exchange a material, product or service, see Figure 4a,b, with the former (Figure 4a) illustrating the analytical and empirical focus of this paper. Thus, rather than a transaction based on an established price, the focus in this paper is on the transaction of data and information that occur prior to and shape a price, in light of GVC governance and risk and value constructions.



**Figure 4.** Analytical framework based on stylized linkages of transactions: (a) of data and information preceding price formation; and (b) of price of a material or service. Source: adapted from conceptual elements of [31,32]. Note: Figure 4a shows the GVC variables upon which governance structures are determined. The construction of risk through the relational theory of risk is included as a significant dimension of the transaction (and based on Figure 3). From the exchange of data and information under risk and GVC governance in Figure 4a, a decision is made which, presuming successful risk communication, results in the construction of a price (Figure 4b), upon which an exchange of material takes place.

This transaction simplifies one interaction at a particular segment in which a processing activity occurs that requires an input and an output. As the buyer and supplier exchange data, information is built up on the extent to which the material or service desired can be codified for a “handover”, this process is represented in the GVC governance variable “codification of transaction”. The exchange of data also provides insights into the “complexity of the transaction”, the second GVC governance variable. As the parties exchange data, information builds up that enables them to gather an overview of the respective “capabilities” of the transactional partner, the third GVC governance variable.

The exchange of data between the partners is suggestive of their perception of its value, and thus, in line with the relational theory of risk, they are simultaneously constructing relationships of risk, by identifying risk objects and objects at risk. These constructed risk relationships at the buyer and supplier interaction might converge or diverge from each other. In conjunction with governance structures, they determine whether material loops, such as of REE, are closed in practice. When a positive decision is made, a risk relationship was constructed by either partner that matches that of the other providing a foundation of a stable risk relationship that allows a transaction of a material or service to “materialize”, as shown in Figure 4b. This conceptualization speaks to that of Lepawsky and Billah [30] (p. 126) who convincingly argue for recognizing, “value as in-the-making rather than an intrinsic property of things”.

Our methodology follows the analytical framework elaborated above and is inspired by the relational theory of risk which emphasizes the importance of the “lived-in world” as opposed to the “intangible world of concepts” that define risk [32]. Thus, the relational theory of risk focuses on examples of “communities of practice” [74] or of “organizational contexts” to “concretize the study of risk objects, objects at risk, and relationships of risk” [75,76]. In so doing, it underlines the importance of empirical case studies for creating an understanding of risk.

On three empirical cases we demonstrate how the construction of risk affected the implementation of business plan conceptualizations and lab-scale tests of new technologies and why risk in REE-loops needs to be targeted with governmental response for transparent material characterization. We delineate the governance forms that appear to determine the particular transaction at the segments which are subject to our observations of construction of risk. We then describe the situated view of each concerned actor on the “risk object” and “object at risk” to explore the nature of the actors’ observed “relationship of risk”, i.e., whether it is considered “stable” or “unstable” and with which effect for loop closure. This narrative is guided by our analytical framework.

Our data consist of empirical material, specifically transcripts of interviews conducted between February and June 2017, in addition to empirical evidence gathered since 2012, and literature reviews that include websites of start-up firms in the REE industry and of EC-funded REE-focused research projects on which industry developments are discussed. Some of the semi-informal interviews with industry representatives arose from simple requests for information and clarification in the course of the preparation of a research proposal that aimed at closing REE-loops in practice. This is an important aspect of the data collection, as this approach enabled data collection that, retrospectively, proved highly useful in shedding light on the *daily practices of businesses* and, importantly, also on the *narratives defining their daily practices*, in addition to the organizational context of a particular firm.

The analytical variables of the global value chain (GVC) framework—complexity of a transaction, ability to codify a transaction, capability of the supplier—support our assessment of the transactional characteristics, i.e., the governance structures between a buyer and supplier at the segments of our empirical cases. We assess whether a given transaction is characterized by a “low” or “high” complexity, whether the information transferred is easily codifiable or not, i.e., when product or process standards are present, or procedures established, and whether low or high capabilities to execute the transactional requirements are observed at the supplier (and buyer).

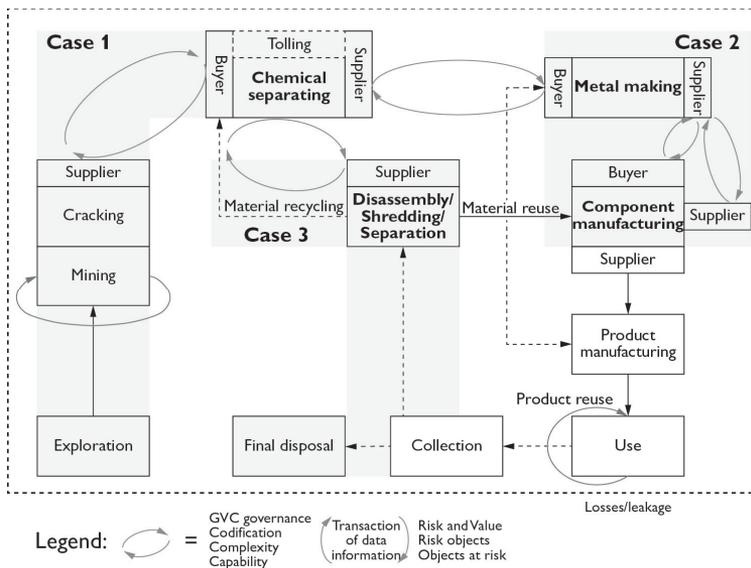
We then draw on the three elements—risk object, object at risk, and relationships of risk—that constitute the relational theory of risk, as described earlier, to create an understanding for policy-makers and scholars alike of the risk communication at selected REE-value chain segments, as well as of governance structures, and how these affect the possibility of closing REE loops. To substantiate this approach with pragmatic entry points for action, we follow the proposal of Boholm and Corvellec [32] (p. 187), that the key to successful risk communication is establishing “*a common understanding of what constitutes a threat, a value, a contingency, and a causal relationship*”. This methodological approach provides us with the means to depict the sequence of narratives that impact significantly on, for example, the translation of proven technologies for REE recycling, of which many have arisen over the last few years at sub-commercial scale, to testing these on a commercial-scale for market-readiness.

### 4. Findings

China, as dominant REE-supplier, was identified by both government and industrial actors as the risk object in the aftermath of REE-price peaks of 2011, and the continuous accessibility of stable-priced separated REE products as the object at risk. A stable relationship of risk was constructed between governments in the EU and in the US, and a common object of risk agreed upon that was evident in narratives of “supply risk”. In response, many publicly funded projects have developed technologies for REE-separation and recycling on a lab-scale (see i.e., EC-funded FP7 or H2020 projects such as [77–84]) and the potential for recycling was discussed (e.g., [37,85–89]). Surprisingly little progress occurred from lab-scale tests of the technologies to commercial implementation.

With continuity of China as dominant REE-producer and user, supply risk (notably from registered, documented sources of production) of the REE remains unchanged. However, the REE prices have changed: Since the 2011 REE price peaks they have returned to or even dropped below pre-peak levels [78] (p. 135). It is proposed that prices are the reason for struggles of REE firms in Europe [36]. Is it possible that price developments alone are the single reason for why the publicly funded and developed technologies are not transiting into commercial life and are not pursued to close loops of REE? This explanation appeared to be too simplistic given that risk–value constructions occur prior to price formation. As the purpose of initiating many of the projects was to mitigate risk, and this risk arguably remains, closer examination is warranted.

The case studies of specific GVCs of REE reveal insights into numerous complexities: The REE value chain is global and multi-layered with numerous actors interacting at each segment, inevitably bringing a myriad of data and information together in any given transaction prior to reaching agreement for the actual exchange of a material or service, or both, based on price and, in some cases, information accompanying this exchange. Figure 5 delineates a stylized schema of the global REE value chain segments in which the three empirical cases to be discussed in the following sections are highlighted.



**Figure 5.** Stylized schematic of the empirical cases addressed of the GVC of REE. Note: This figure illustrates, clock-wise starting left, the GVC segments (in blue-edged squares) from exploration of underground REE-mineral occurrences through various processing steps of REE-bearing minerals to REE-metals, components, final products with REE-components, up to the segments attached to closing material loops. The segments that are discussed in our case studies are highlighted in blue.

#### 4.1. Case 1: Construction of Risk Relationships for a REE-Tolling Station

Chemical separation is a key segment in the global REE value chain, both in processing of the rocks and of EoL material for closing a loop. Its outputs are individual separated REE, such as oxides of praseodymium, neodymium, dysprosium or europium. Commercialized technological processes for chemical separation are tied to high capital- and operating expenditures (CAPEX, OPEX) and require cross-cutting knowledge of mineralogy, geology, chemistry and metallurgy. The purchase of both batteries and annex equipment for the liquid-liquid extraction (i.e., solvent extraction (SX)), and induced equipment (e.g., for specific effluent treatment) constitute major CAPEX [90]. OPEX stems from the use of energy and solvents (including losses), handling and storage of radioactive material, effluent treatment and taxes for discharge streams.

From 2012 to early 2016, when REE-industry participants anticipated another imminent rise in REE prices, the discussions on mitigating a potential REE-supply risk centered on the idea of establishing particular organizational structure, referred to as a tolling station. This station had the aim of minimizing the CAPEX a single firm would need investing for a plant infrastructure. It was conceptualized as a centralized facility operated by a consortium of mining companies and end-users, see [91] (p. 59). The tolling station would provide chemical separation services by processing a mixed REE solution (salts/oxides/chlorides/nitrates), the output of a flotation process, from numerous suppliers of different REE-containing ore into individual REE, complying with quality requirements of potential buyers.

In addition to the government regulator, four actors come together with interests in the processing segment of chemical separation (see Figure 5 for the processing segments and notes on the GVC of REE): the exploration firm (which seeks to sell its developed deposit to a mining firm), the mining firm (that delivers the input of REE-minerals and conducts in many cases also the cracking of the REE-mineral into a mixed REE solution), the chemical separator (which separates the REE-minerals into individual REE products), and the customer (often a metal maker, which uses the individual separated REE product).

In the transaction between the mining firm selling the REE-containing rock and the chemical separator buying it the ability to codify the transaction is high as it is limited to the knowledge gained from assaying the mineral cores of the drilling programs along with other information from the bankable feasibility study, both of which can easily be exchanged. There is a degree of uncertainty as to the exact composition of each mineral concentrate. However, the complexity of this transaction is low as no additional information needs to accompany the handover of the mineral concentrate. With a view to the capabilities at the supplier end, they are high when it comes to producing a mineral concentrate from an ore that has already been commercially processed in the past. These characteristics of the transaction suggest market governance determined by price, summarized in Table 1.

However, if the REE-bearing ore has not yet been commercially processed, as would be the case for a tolling station that buys from several REE-bearing deposits that have not been mined previously, the characteristics of the transaction change to one in which the complexity of the transaction is high, as coordination needs arise between the various suppliers of the ore to the operator of the central tolling station. The ability to codify the transaction would be low, as information in addition to price must be exchanged. The capability of the supplier would also be low as it is no longer the individual capability of one supplier but the aggregated capability of the suppliers that must be accounted for. This suggests a hierarchy governance form in which the actual integration of chemical separation with the mining firms is most feasible. This decision-making process for or against entering into such a transaction on either side (supplying mining firm and buying chemical processor and operator of a potential tolling station) gives rise to the construction of risk-value relationships, described hereafter.

**Table 1.** Case study 1: Transactional characteristics between the mining and chemical separation segments.

Transactional Partners	Ability to Codify the Transaction	Complexity of the Transaction	Capability of the Supplier	Governance Form
Transaction between the mining firm and the chemical separator for a commercially processed REE-ore	High	Low	High	Market
Transaction between mining firms and the tolling station operator for lab-tested REE-ore	Low	High	Low	Hierarchy

As exploration firms eagerly worked towards bankable feasibility studies for their respective REE-bearing mineral deposits, their focus was on attracting customers to demonstrate the feasibility of their business plan to mining firms. The latter would then be willing to purchase their developed mineral deposit, as mining is rarely an activity exploration firms pursue, see [92]. Thus, their business plan development centered on establishing integrated chains from exploration and mining to chemical separation (see [33,93]), when they realized that customers had an interest in separated, individual REE products. REE-mineral mining was therefore to be combined with further processing of the minerals.

The exploration firm with rights to explore deposits, e.g., in Australia, Canada and Greenland, perceived the inexistence of independent *chemical separation facilities* outside of China as a *risk object*, and the resulting dependence on Chinese suppliers for *individual REE oxides*, the output of chemical separation plants, as the *object at risk* ([91], p. 59). The mining firm agreed in principle with this conceptualization of elements in the construction of a risk relationship, while acknowledging that its business portfolio and expertise commonly remains limited to mining and physically benefiting the minerals. Therefore, the mining firm relies on the chemical separator and customer (metal maker) to confirm the proposed risk relationship of the exploration firm.

For the chemical separator in the EU, the *risk object* is *access to REE-ores* under tight regulation in China, and the *object at risk* is its business activity of *separating REE-minerals* into individual REE products. The action that was taken in response to this risk relationship was the establishment of plants in China, where major REE-demand originates. This took the form of, e.g., joint ventures with local firms (see [33]).

The tolling station concept would provide input from different REE-mineral deposits, and here, the chemical separator would see the *risk object* as the *properties of the minerals* fed into the separation process, with the *object at risk* being the *cost structure* of its operation, and thus, of its final product, the individual REE elements. This argument rests on the significance of the correct choice of the solvent for a cost-effective separation, followed by the selected technology and the number and fixed sequence of REE to be individually separated.

The risk relationship portrayed by the exploration firm is unlikely to be confirmed by the chemical separator who has already mitigated against the risk relationship with plants in China which also represents a major growth market. Further, the separator appears to perceive the risk relationship for the tolling station as unstable, framing the risk relationship differently to the exploration firm, in the context of REE-industry dynamics in which China continues to play a dominant role and where undocumented production accounts significantly distorts of REE market prices and affects the willingness to invest.

The customer of the individual REE products in the EU, i.e., a metal producer, perceives the *risk object* as the *limited supply channels* outside China from which individual REE products can be sourced and the *object at risk* as its *continuous supply of high-quality REE products at stable prices*.

In response to this defined risk relationship, the user has taken steps to relocate manufacturing activities to China, in addition to engaging in in-process recycling of REE-materials with its customer, i.e., a magnet manufacturer (see next empirical case). Thus, while the customer shares a common risk object and object at risk with the junior exploration firm, the customer appears to have taken measures to address this risk relationship.

With regards to the proposed tolling station, the customer sees the *risk object* as the *adaptation of the separation process* required to accommodate REE-mineral types, and the *object at risk* as its standards of *high purity* for the individual REE products. Lab scale tests of the adaptation of the chemical separation process, or new technologies tested at lab scale are unlikely to provide sufficient assurance that the risk is sufficiently addressed.

While the definition of risk object and object at risk by the exploration firm would have in principle approval from the various actors, the differing conceptualizations of the elements of risk by the same actors and their mitigating actions, a common risk object cannot be defined. Thus, the communication of risk is unsuccessful. This may explain why the push for a tolling station has stalled.

#### 4.2. Case 2: Pre-Consumer REE-Magnet Recycling

At the magnet manufacturing segment three actors come together: The magnet manufacturer who purchases a REE-magnet alloy, the metal and alloy producer, and the customer who purchases the REE-magnets. Between the former two the governance form that arises is modular in that the specifications for the metal required are easily codified, including by standards that incorporate material performance qualities. Nonetheless, the complexity of the transaction is high since many performance criteria need to be met. In the context of a very capable supplier, the metal producer, the complexity of the transaction can be handled without problems, as the metal producer possesses the knowledge that enables the codified transaction [94,95].

In contrast, when it comes to closing material loops, the ability to codify the transaction of scrap magnet metal, a byproduct of shaping the magnets into the form desired by the customer, is low. The complexity of the transaction between the magnet manufacturer and the metal maker is high, as the highest possible level of detailed information must be exchanged between the two actors in the transaction. This includes for instance confirmation that only sintered magnet material is being returned, as bonded magnet material includes epoxy that poses a contamination risk for the material streams of the metal producer. Further information on the type of magnet alloy, i.e., the composition of the alloy including REE content, is useful in the exchange. The capability of the supplier is high in as far as the magnet manufacturer is a competent partner in the transaction who understands how the magnet alloy should be handled and what type of information facilitates a successful transaction that will deliver new REE-containing metal alloys in return. These characteristics of the transaction suggest a relational governance form in which coordination between the transactional partners is required although they are still relatively independent from each other. However, there has been a clear change in the type of governance structure from the first, conventional linear transaction under modular governance, to that of closing the loop with pre-consumer recycling and relational governance due to increasing coordination needs. The characteristics are summarized in Table 2.

**Table 2.** Case study 2: Transactional characteristics between the metal making and magnet manufacturing segments.

Transactional Partners	Ability to Codify the Transaction	Complexity of the Transaction	Capability of the Supplier	Governance Form
Transaction between the metal maker and the magnet manufacturer	High	High	High	Modular
Transaction between the magnet manufacturer and the metal maker	Low	High	High	Relational

For the magnet manufacturer, the risk object is the availability of a high-purity REE-metal *alloy* and the *object at risk* is the *accessibility and stability of the price of this alloy over time*. For the supplier of the metal-alloy, the risk object is as described in the previous section, the *limited supply channels* outside China from which individual REE products can be sourced, and the *object at risk* is its *continuous supply of high-quality REE products at stable prices* to the buyer.

To mitigate the established and agreed risk relationship by the magnet manufacturer and alloy producer, the latter communicates the risk to the magnet manufacturer who loses material in the manufacturing process from out-of-spec magnets or from shaping the magnet. The REE-magnet manufacturer provides REE-material for reprocessing to the REE metal producer. This REE-material is solid sintered material and re-melts well and cleanly. The REE-metal producer reprocesses it in batches of material belonging to a particular REE-magnet manufacturer and, while it may be of different compositions, a first-stage melt is conducted to understand the composition (and adjust it accordingly, if needed) and then blend it with 70% of virgin material to produce a new metal alloy for the magnet manufacturer.

The REE-magnet purchasing customer frames its risk object as the REE-alloy used in, and the manufacturing process itself, of the magnet, and the object at risk as the accessibility, price and performance according to magnetic standards of the magnet purchased.

#### 4.3. Case 3: REE Recycling of End-of-Life Lamps

While REE are found in many End-of-Life (EoL) electronics, commercial recycling of REE from this source has so far only been technically and economically feasible for a select number of product groups, including fluorescent lamps. However, before EoL lamps can be processed for recovery of REE, lamps must first be collected. Collecting and environmentally sound recycling of lamps is a net cost for recyclers, making it unlikely that this will happen beyond small-scale voluntary initiatives without legislation [46]. Extended producer responsibility (EPR) schemes in EU countries is mandated by Waste Electrical and Electronic Equipment (WEEE) legislation, which requires collection and recycling infrastructure for EoL lamps (fluorescent and LEDs) and specifies at least 80% of collected mercury lamps must be recycled and mercury removed. EPR schemes involve multiple actors including national authorities, local municipalities, producers, retailers, local waste management companies, specialized recyclers, and consumers who engage in multiple transactions enabling the physical, financial, and informational flows that underpin EPR schemes (see [47] for an overview of actors and transactions in EPR systems for lamps in the Nordic countries).

While collection of EoL products is a necessary precondition to recycling of REE from these products, the focus in this case is on the decision to recycle REE from lamps, not on the collection decisions. The main actors influential in this decision are lamp recyclers who process the initial EoL lamp waste, chemical separators of REE, and producers of products using REE, who are the customers buying the recycled REE. Due to the net costs involved in lamp recycling, recyclers operate in mandatory and voluntary schemes with a focus on sound environmental management of the mercury in the lamps and in keeping recycling costs low. Mandatory WEEE legislation in the EU (as well as voluntary standards for mercury containing lamp recycling) require special processes for removal of the mercury, most of which is generally contained in the phosphor powder fraction along with the majority of REE. Recycling processes also aim to recover glass, metal, and plastic fractions though it can be challenging to find markets for recycled fractions (other than metal) [46].

The treatment of EoL phosphors for recovery of REE typically involves two main steps (as well as several specific technical process steps): (1) removal of mercury, glass and other impurities from the powder, yielding a REE-rich mixture; and (2) separation of REE mixture into individual REO. Both steps can be performed by the same firm (e.g., Solvay-Rhodia operated a commercial process until the end of 2017, with Step 1 in their Saint-Fons plant and then sent the mixture to their La Rochelle plant) or by two different firms (e.g., there are several pilot projects now performing Step 1 and looking for customers for the REE mixture as is or Step 2 chemical separators). While there can be markets for REE mixtures from Step 1, these have lower market value than individual REOs, however chemical separators able to perform individual REE separation are limited and there are no longer options for heavy REE found in lighting phosphors available in the EU with the closing of the Solvay-Rhodia operation [96].

The recycled phosphor powder fraction comprises 2–3% of the volume by weight of the total recovered material from the lamp recycling process. If disposing of the phosphor powders, lamp recyclers face costs depending on the mercury content of the phosphor powder and the specific hazardous waste requirements for landfilling or permanent storage (e.g., in salt mines in Germany). These costs are driven by disposal costs in the jurisdiction and can be easily quantified and anticipated, and can be characterized as a market governance form. By contrast, the processing the phosphor powder depends on changing this business practice, finding a chemical separator, and negotiating prices with Step 1 chemical processors (also different lamp recycling processes and input waste yield different phosphor powder mixes, some of which may not be compatible with processes for REE recovery [97]). The lamp phosphor waste represents a new source of REE with its own characteristics requiring refining processes (Step 1) to be specifically designed. At the same time, the supply is dependent on collection of the EoL lamps and is also influenced by product technology change. Thus, there are several challenges to codifying, such as uncertainties about the capabilities of the supplier and high complexity, indicative of a hierarchical governance form.

The ability of chemical processors to operate, in turn, depends on customers and market values for the REE mixtures and REOs, both of which have been dynamic and unpredictable in recent years. While large established chemical separators have the capability to perform Steps 1 and 2 of the chemical separation for lamp phosphors, some smaller operators only perform Step 1 and attempt to sell the refined REE mixture or carbonates to end customers or Step 2 refiners. Even so, the transaction can be codified and the supplier is capable, though the transaction is still complex, indicative of a modular governance form, see Table 3.

**Table 3.** Case study 3: Transactional characteristics between the lamp recycling and the chemical separation, and the landfill/permanent storage segments.

Transactional Partners	Ability to Codify the Transaction	Complexity of the Transaction	Capability of the Supplier	Governance Form
Transaction between the recycler and landfill/permanent storage operators for EoL lamp phosphors	High	Low	High	Market
Transaction between the recycler and the chemical separator for EoL lamp phosphors	Low	High	Low	Hierarchy
Transaction between the chemical separator and customers	Med-High	High	Med-High	Modular

A key actor in the decision of whether to recover REE from phosphor powders is the lamp recycler who first manages the treatment of the waste after collection. However, lamp recyclers are contracted by producers who are fulfilling EPR obligations, municipalities, or actors behind voluntary initiatives. The value for these actors is to soundly manage the waste, particularly the mercury which is often pursuant to mandatory obligations and treatment requirements. Recycling of fluorescent lamps is not economically viable based on material value of the recycled materials alone, so the environmental and health benefits of treating the mercury drive voluntary initiatives as well as mandatory EPR legislation [40]. The risk object for the government in recycling lamps is the mercury in energy efficient lamps, while the public health and the environment comprise the main object at risk. Lamp products utilize mercury in the design in order to dramatically increase the energy efficiency of the product, in comparison to incandescent lamps, and result in lower overall emissions of mercury when considering the entire lifecycle of the product (due to decreased energy needed, which in turn have associated mercury emissions if there is any coal in the mix of energy used to produce or use the product). To manage the risk of mercury, the WEEE directive specifies that mercury must be removed in the recycling process, and, since 2011, there has been an export ban and disposal obligation for mercury.

Though closing material loops is an explicitly stated aim of EPR legislation in the EU, the legislation does not require the recovery or use of the REE material and thus the decision to send material for further recycling depends on the motivation of the recycler to send the phosphor powder on to a chemical separator. However, to keep the cost of treatment low (to retain contracts for the recycling), the recycler is also incentivized to do this at the least cost while still complying with the legislation as another risk object for the recycler is the cost of treatment in order to preserve competitiveness. The recycler compares the cost of disposing of the waste lamp phosphor powder with sending the phosphor powder to a chemical separator. Some recyclers also investigated refining the phosphors themselves but had little capacity and found there was no business case for small batches, thus necessitating a transaction with a larger chemical separator. The recycler's decision is also final for the fate of the REE content as a common method of disposal of mercury waste, including waste phosphors, is as mercury sulfide in permanent storage, e.g., in salt mines in Germany or in controlled landfills, depending on mercury content and legislation. Once waste phosphors are stored in this manner their potential as a source of REE is lost [98].

While some Producer Responsibility Organizations (PROs) who contract the recyclers also indicated that additional value around closing material loops could add to the recycler's competitiveness, not all PROs interviewed identified this value or expectation in their recyclers. Thus the decision for recyclers to recycle or dispose of waste lamp phosphors can be best framed as dependent on the cost of disposal in controlled landfill or permanent hazardous waste storage (depending on the mercury content of the powder and specific rules in the jurisdiction) compared to the cost of sending this powder to a chemical separation process.

Transactions between lamp recyclers and chemical separators capable of further treating lamp waste phosphors are in turn dependent on the salability of REE recovered from the chemical separation processes (i.e., the object of risk). The risk object is the unpredictable REE market, which in the case of lamp phosphors, reflects not only the uncertainties about supply in the context of price fluctuations in response to the dominant production share of China, its control measures and a significant undocumented/illegal market, but also large uncertainties about demand for rare earths as the lighting market shifts from fluorescent to LED lighting technology. The effect of this technology shift is twofold: (1) it decreases the future supply of REE available for recycling from EoL lamp products as LEDs have substantially smaller amounts of REE; and (2) it decreases demand for some phosphor REE, such as Europium (Eu), due to the fact that phosphors currently dominate the demand for this type of REE. In essence, the loop itself is shrinking unless other sectors increase the demand for the REE used in lamp phosphors.

While this could be viewed positively in that recycled Eu could then more easily satisfy the more limited demand of Eu for lighting producers [38], this is further complicated by the fact that Eu mining is also driven by demand for other rare earths found in the same deposits (i.e., as a by-product, reflecting the balance problem described by [35]). As industry faces supply risks, recycling represents one mitigation strategy, but also represents complex transactions between multiple actors and can represent an increased cost. Thus, the value of recycling REE is compared to the value of primary mining and substitution, as these are other strategies for industry to manage such risks. In addition, more focus is needed on the losses of other elements in a REE recycling process [38].

However, mitigation of the described risks through recycling from anthropogenic sources can also provide environmental benefits through avoiding primary mining [99–104]. It was clear from Solvay-Rhodia's communication of its commercial lamp phosphor REE recycling process that the value of recycling REE from phosphors in the EU was beyond pure economic considerations. A respondent from Solvay was quoted characterizing the value for the company for its sustainability and corporate social responsibility agenda, stating that "This project is driven by our sustainable development approach" [105]. Validation of the process entailed a €2 million investment in a two-year project during 2012–2014 (after investment in development of the process itself), half of this coming from EU Life+ funding (a financial instrument supporting environmental projects).

The assistance from the EU in terms of Life+ funding again reflects the perception that Chinese dominance of the REE market was perceived as a risk, not only by industry actors such as Solvay-Rhodia, but also by the government actors at the EU level. The recycling of REE in the EU was perceived as socioeconomic value including by enabling a domestic/EU source of REE, 30–40 direct new jobs in the EU, and capacity building in urban mining in the EU [106]. The project report further declared that recycling REE from lamp phosphors would *“increase the independence of Europe as regards to REE. It will also help conserve natural resources and reduce the use of environmentally damaging processes in their transformation. This will ensure Europe has access to a sustainable provision of these elements without the risk of shortage that could have dramatic social and economic effects.”* (p. 11).

However, the closure of the Solvay-Rhodia process, which cited poor economics due to decreased REE prices and decreased demand for the REE in the lighting market [107], and the continued struggle of pilot technology-ready recycling processes to find markets for their products reflects that the risks and values of REE recycling from waste lamp phosphors are perceived differently by industry in comparison to governments. It is clear in this case that industrial actors, while they may be aware of the environmental and societal values that recycling could bring (as evidenced by the framing of the process by Solvay-Rhodia), in reality have risk–value constructions that do not reflect environmental and societal values beyond the framing of waste as a hazard to be managed. Even then, this risk–value is most often underpinned by legislation. Addressing such risks and capturing value to society and the environment in closing material loops then suggests a role for governments as well as business.

## 5. Discussion

Our comparison of the three case studies highlights some of the key factors that influence loop closure. Firstly, we observed that industrial actors are more prone to realize the value of closing REE loops when they operate at adjoining segments, as they already have a transaction established, such as on the case of REE-metal pre-consumer recycling. This is opposed to post-consumer recycling where the relevant transactions span longer communicational distances between a myriad of actors, such as from a consumer and its EoL product, e.g., a REE-phosphor containing lamp, to collecting it and risk–value constructions by all the actors that affect whether the material is permanently stored/landfilled or sent for reprocessing of its material content.

### 5.1. Governance Structures

We observed distinct patterns around the influence of governance structures. It appeared that hierarchy governance is not conducive to closing REE loops, especially when the alternative route to REE-product accessibility is through a market governance structure. This was the case both for the conceptualized centralized facility, the tolling station, and for the EoL REE-phosphor containing powder recycling through chemical separation. The extensive need for information in post-consumer recycling gives rise to a hierarchy governance structure that impedes the closure of REE loops. When, diverging and unstable risk–value constructions emerge among actors involved at the particular segments in combination with hierarchical governance structures in transactions of data and information, disincentives result. Alternatives to hierarchy are required in which the involvement of the actors to the transaction is more balanced.

In contrast, where the supplier–buyer relationship appears modular in the conventional transaction between the metal producing supplier and the metal alloy buying magnet manufacturer, a relational governance structure formed at the pre-consumer recycling stage as the buyer of the metal alloy for magnet enters into a transaction of data and information with the supplier of the alloy. This case of the metal-magnet transaction showed how value was perceived and a stable risk relationship constructed by the relevant actors at these segments so that pre-consumer recycling could take effect. Here, supply of scrap material exists and demand is constructed by making a compelling case of a risk relationship upon which buyer and supplier agree.

Please refer to Table 4 for a summary of the mapped relationships of risk discussed in this paper.

Table 4. Mapping of relationships of risk.

Actions	Concerned REE-Segment(s)	Specific Actors	Relationships of Risk * (Stable/Unstable) Actor of a GVC Segment	Risk Objects Potentially Threatens	Value of Objects at Risk	Success/Failure * in Risk Communication				
							Industry and Government	Stable	China as Dominant REE Producer	Accessibility of REE outside China
Building a tolling station	Chemical separation	Chemical separator	Stable	Selecting the feed-in minerals (properties) Deciding on the solvent for the chemical separation Choice of technology	Saleability of output: Cost competitive production of refined separated REE oxides of industrial off-taker's interest	Failure				
							REE-metal manufacturer	Stable	Price fluctuations in REE alloy	Loyal customer to buy REE-magnet alloy
							REE-magnet manufacturer	Stable	Access to REE-magnet alloy for magnet making	Continuous reasonable-cost purchase of REE-magnet alloy
Pre-consumer recycling of processing residues	Metal-Alloy-Magnet	REE-metal producer/REE-magnet manufacturer	Stable	Composition of metal fed from processing waste into recycling project	Purity of REE-alloy made	Success				
							Government	Stable	Hazardous content (mercury) of EoL lamps	Population's and environmental health
							Chemical separator	Unstable	Access to low price geogenic REE	Saleability of recycled REE-oxide under fluctuating REE-prices from China
Post-consumer recycling of EoL products	EoL fluorescent lamps/LEDs-chemical separation	EoL lamp recycler	Unstable	Cost of environmental management of mercury/REE phosphor powder	Competitiveness on price	Failure				
							REE customer	Stable	Access for REE for products	Continuous reasonable-cost purchase of REE for products

Source: Authors. Note: \* These are point-in-time evaluations of a dynamic situation, and therefore open to change based on targeted action.

From that case, new roles emerge that challenge the conceptualization of GVC governance structures [31]: The metal buying magnet manufacturer becomes a magnet scrap metal supplier to the metal maker. The latter, however, does not turn into a buyer, but simply a service providing supplier. This speaks to the different dynamics in anthropogenic material flows [53], namely that supply does not follow demand but exists where the scrap material emerges from processing. An interaction between the buyer and supplier is needed to initiate a transaction of data and information to rethink their existing supply–buy relationship for closing the loop by pre-consumer recycling. This is an explicit call for the scholarly and research community to conceptualize in a systematic way the patterns observed in both existing and new empirically evidenced supplier–buyer–service provider relationships. This paper thus points to the limits of the way GVC governance structures are currently conceived of in the literature as framed around one transactional stage rather than two. The first set of transactions involves data and information which provide the foundation for the second transaction of materials or services based on price and accompanying information if required.

### *5.2. Risk–Value Constructions*

The risk–value construction of private sector actors (i.e., the industrial actors) diverges significantly from that of public sector actors (i.e., government agencies) as each has different agendas and motivations. Private actors are concerned with the interests of equity holding individuals of a private firm or of shareholding investors of a stock market listed firm operating at the local–regional–global scale. Public actors are motivated to safeguard the interests of nation-wide economic development while ensuring the protection of environmental and human health for the well-being of citizens who are embedded in a global economy. It should not be assumed that either actor will protect the other’s values unless a stable risk–value construction is formed between them.

### *5.3. Role of Government*

This motivational discrepancy arising from diverging risk–value constructions that seemingly impede the closure of loops supports the argument of Hagelüken [26] that in times of low raw material prices, including of the REE, it is the role of the government to engage with strong leadership and bring measures into place that frame the risk–value construction so that the closure of loops is incentivized. This has been observed in initiatives for other forms of materials recycling. In a study of urban stormwater recycling initiatives, Lane et al. [108] highlighted this need for top down approaches to risk allocation while pointing to significance of clarity around the definition of risk and allocation of risk management responsibilities. Porter and Kramer [68] have pointed to the need for a constructive policy design by government that enables industry to be innovative and find solutions to reaching legislative targets including for recycling. Further accounting for some of the complexities highlighted by the risk–value constructions along GVCs, we argue that the role of government is to “bridge” the risk–value conceptualizations among actors at the pre- and, specifically, post-consumer recycling segments by means of facilitating the flow of information. This should incentivize a transition from a hierarchy governance structure to a relational or modular governance structure in which information is more easily available and codifiable. Possible approaches could include the elaboration of international standards, imprinting barcodes on components that indicate their materials and, as in the EoL lamp case, through legislation. On the latter, Binnemans [36] noted the necessity of fine-tuning regulations to delineate the importance of the minor metals including of REEs, where currently weight percentages cast a shadow over these.

With a view to standards, work on the elaboration of international product and process standards for REE is already in its early stages under the ISO/Technical Committee 298 [109] since late 2016. The drafting of standards is thematically divided into rare earth terms and definitions (minerals, oxides and other compounds in part 1, and rare earth metals and their alloys, in part 2), as well as into rare earth elements recycling (communication formats for providing recycling information on rare earth elements in by-products and industrial wastes; measurement method of REE in by-products

and industrial wastes; method for the exchange of information of REE in by-products and industrial wastes). This paper speaks to the communication formats and methods for the exchange of information that facilitate recycling.

The bar-coding option is likely to be accompanied with numerous policy-regulatory challenges, in particular with a view to the protection of the intellectual property rights of firms. Rather than prescribing a particular way of bar-coding, policy-discussions may need to center on how the information is to accompany the material in the best possible way. While we note the challenges attached to this option, we encourage industry and policy-makers to jointly discuss and find suitable approaches.

## 6. Conclusions

A transition to a Circular Economy involves identifying and addressing barriers to loop closure, particularly for critical materials such as REE. In this paper, we argue that it is essential to gain an understanding of the transactional dynamics of data and information between a buyer and a supplier in which governance forms arise and risk–value constructions are made that precede pricing and material or service transactions. This supports explanations of how (segments of) rather complex material loops are currently closed or how they might be. Through a focus on REE, we draw attention to how key critical minerals are framed in these discourses, which are complex due to their geological occurrence, processing specifics and industrial uses.

We bring the relational theory of risk into conversation with the governance structures of the GVC framework to assess existing governance structures and explore how risk–value relationships are constructed by the various actors that have interests at specific GVC segments, with implications for why REE loops are closed or not. We observed different governance structures for closing material loops at the pre- and post-consumer recycling stages, with some more likely to enable loop closure than others. Such findings give relevant background information for policymakers and researchers further investigating policy measures to support closing loops.

By drawing on the relational theory of risk, which understands value and risk as intrinsically linked, we take a constructivist angle. The risk–value construction depends on what type of value is being considered and who is assessing it, i.e., the perspective of the actor. A broader notion of value that includes environmental and social values as well as economic ones highlights the difficulty in weighing and assessing value objectively. A clear starting point for policymakers pursuing circular economy aims of closing material loops is to identify what values are perceived, and by whom. From the delineated governance forms of the empirical cases, we argue that the government needs to play a pivotal role in closing material loops when the risk–value construction of industrial actors is at odds with the societal values such as public and environmental health which governments are obliged to protect. The role of government, arguably, is to put measures into place that augment transparency of material qualities at a given segment to facilitate data and information availability for transactions, and, thus, foster the formation of closed loops. Along these lines, we recommended specific measures such as the elaboration of standards to qualify materials for re-entry into material processing, which could then be communicated through bar-coding of materials. These measures could be accompanied by appropriate regulatory amendments.

Finally, we encourage more empirical scholarship that systematically maps transactions and reveals governance forms in which risk–value constructions occur that affect loop closure, a significant element of the circular economy. While we have suggested roles for government and policy approaches to address specific issues, further research with a focus on these issues specifically is still needed. With this paper, we hope to initiate a lively, interdisciplinary discourse on this subject and invite scholars with cross-cutting research interests to participate.

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Paper IV





# Optimal durability in least life cycle cost methods: the case of LED lamps

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**Abstract** In the European Union (EU), mandatory durability ecodesign requirements have recently been set for some products, including lighting products; further development of durability standards is also expected in the future. Durability standards can bring environmental and consumer benefits, but the question remains about what optimal durability is. In this paper, the product lifetime aspect of durability is considered, and optimal lifetimes in relation to least life cycle cost (LCC) for the consumer are analysed. The paper focusses the analysis on a case of LED lamps available in an online market in December 2016 and models optimal lifetimes from an LCC perspective. The statistical error of the regression does not allow for calculation of the optima with precision, but the calculation indicates optimal

lifetime is close to 25,000 hours. The influence of smaller discount rates and more intensive use of the product are also modelled, which indicate that durability is desirable in intense-use scenarios in particular. The usefulness of the method is discussed and the findings are compared to previous literature and studies examining durability and increased lifetimes for products, including those using an alternative approach of life cycle assessment (LCA). The initial results of this LCC method indicate that longer lifetimes than those currently required by legal standards in the EU could be appropriate for LED lamps. As such, the advantages and disadvantages of different policy instruments to stimulate increased durability are also discussed. The paper concludes with suggestions for potential future research and further policy development.

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## Introduction

One of the substantial policy developments related to the circular economy is the interest for incentivising more durable products (European Commission 2016). Durability refers to the “ability of a product to perform its function at the anticipated performance level over a given period (number of cycles/uses/

hours), under the expected conditions of use and under foreseeable actions” (Boulos et al. 2015, p. 4). This interest has been manifested in several policies and initiatives already, including national schemes to promote product repairs. Public procurers in some countries have started to purchase remanufactured furniture and remanufactured IT products, and there is a general interest in promoting product durability in public procurement (Montalvo et al. 2016). France has banned planned obsolescence and set up incentives for manufacturers to provide spare parts (Maitre-Ekern and Dalhammar 2016). Iceland and Norway have extended the limitation period for legal guarantees of products from 2 to 5 years and strengthening lifespan legal guarantees is being investigated across the EU (Tonner and Malcolm 2017). It has been argued that durability information should be also be included in the mandatory EU energy labelling scheme (Burrows 2016; RREUSE 2015; ENDS 2016). This is contested, however, and an alternative approach is to make use of voluntary labelling to promote information about product durability (European Parliament 2017). Mandatory ecodesign durability requirements have recently been set for vacuum cleaners and lighting products through EU regulations<sup>1</sup> under the EU Ecodesign Directive,<sup>2</sup> and it is expected that more product groups will follow in the future. In general, the various initiatives reflect growing momentum and debate about how resource efficiency should be addressed through policy interventions. While there are a few different policy options to address durability, there is one central question for policy development moving forward: what durability is desirable for different products?

In this paper, the case of lighting products, one of the first product groups to have mandatory minimum durability requirements, is examined to investigate the question of optimal durability, with a focus on the lifetime

aspect. The EU Ecodesign regulations on lighting products<sup>3</sup> have set functionality requirements relating to non-directional and directional lamps. Most of the requirements refer to dimensions that influence the lifetime of the lamps. Lifetime, as used in a declaration by a manufacturer, is defined in Appendix II in the Regulation 1194/2012 and is a combination of remaining luminous flux and survival factor:

‘lamp lifetime’ means the period of operating time after which the fraction of the total number of lamps which continue to operate corresponds to the lamp survival factor of the lamp under defined conditions and switching frequency. For LED lamps, lamp lifetime means the operating time between the start of their use and the moment when only 50 % of the total number of lamps survive or when the average lumen maintenance of the batch falls below 70 %, whichever occurs first.

EU ecodesign requirements for LED lamps relate to measurements made at 6000 hours (250 days), at which the remaining luminous flux has to be  $\geq 80\%$ , and the lamp survival factor  $\geq 90\%$ , both based on statistical averages. The requirements do not go beyond the 6000-h measurement for practical reasons of time and capacity for such tests and because the dynamic nature of the LED market (as with many electronic products) can create challenges for testing and market surveillance (VITO and VHK 2015a, b). Shorter testing times would be preferred, but this can be a trade-off with reliable testing methods for durability (Narendran et al. 2016). This being said, there are also positive developments in accelerated testing methods that may help to address these issues (Narendran et al. 2016; Narendran, personal communication, 3 March 2017) and some jurisdictions like California are working with a combination of lumen maintenance and “time to failure” tests to set requirements for minimum rated lifetimes of

<sup>1</sup> Regulation 666/2013/EU of 8 July 2013 Implementing Directive 2009/125/EC of the European Parliament and of the Council with regard to Ecodesign Requirements for Vacuum Cleaners, [2013] OJ L192/24 and Regulation 1194/2012 of 12 Dec 2012 Implementing Directive 2009/12/EC of the European Parliament and of the Council with regard to Ecodesign Requirements for Directional Lamps, Light Emitting Diode Lamps and Related Equipment, [2012] OJ L342/1

<sup>2</sup> Directive 2009/125/EC of the European Parliament and of the Council 21 October 2009 establishing a framework for the setting of ecodesign requirements for energy-related products, OJ 2009 L 298/10.

<sup>3</sup> Commission Regulation (EU) No 1194/2012 of 12 December 2012 implementing Directive 2009/125/EC of the European Parliament and of the Council with regard to ecodesign requirements for directional lamps, light emitting diode lamps and related equipment; and Commission Regulation (EC) No 244/2009 of 18 March 2009 implementing Directive 2005/32/EC of the European Parliament and of the Council with regard to ecodesign requirements for non-directional household lamps

10,000–25,000 hours, depending on the LED lamp type (California Energy Commission 2016).<sup>4</sup>

Currently, several manufacturers are promoting the long life of LED lamps as a valuable attribute to consumers, with many now sold claiming lifetimes exceeding 50,000 hours (Hixon 2012); however, there is also speculation that lifetimes for LED lamps may be decreasing in the future if business models for longer life products are not viable (MacKinnon 2016). While prolonging product lifetimes and durability is argued to have environmental benefits (Casamayor et al. 2015; Dzombak et al. 2017; Hendrickson et al. 2010), it is important to also consider any trade-offs in terms of costs for consumers and environmental impacts. Before additional policies should be considered concerning lifetimes for lighting products, there needs to be further exploration of what are optimal lifetimes for these products. One approach to determining if longer lifetimes are desirable is a life cycle cost (LCC) approach (i.e. calculating the costs for a consumer over the lifetime of the product - see methodology).

The aim of this paper is to present a practical method for determining optimal lifetime from an LCC approach and discuss the findings in context of potential policy inventions for promoting durability. The LCC methodology for analysing optimal lifetimes for LED lamps is first described, followed by the results of the analysis. The results from the LCC analysis are discussed in relation to previous LCC studies on other products and also in relation to life cycle assessment (LCA) studies examining lifetimes for LED lamps. Lastly, relevant policies for addressing product lifetimes for lighting products are discussed and recommendations made for future research and policy development.

### Accounting for durability in LCC methods

Previous research has utilised LCC methods to determine when durability is optimal by constructing cases of conventional versus durable product options (also focussing on the lifetime aspect). In their study of refrigerators and ovens, Boulos et al. (2015) found

that generally the more durable products yielded a lower LCC compared to the standard product scenario, primarily due to the avoided cost of the replacement product. Other comparative LCC studies for LED street lighting showed that even with increased efficacy and falling prices of lighting products, delaying purchase of replacements could still be advantageous from an LCC perspective; this is attributed to the large role of the purchase price in the LCC (see Ochs et al. 2014; Tähkämö et al. 2016). Another LCC study of 800-lm household LED lamps by Richter et al. (2017) constructed multiple scenarios with variables of increasing efficiency of LED technology, decreasing purchase price and high or low electricity prices. The LCC for the 10,000-, 20,000- and 30,000-h lamps were compared. The study confirmed the significance of the initial purchase price and found shorter lifetimes were preferred when there were significant improvements in both efficacy (i.e. at least 30% higher) and purchase prices (at least 10% lower) or moderate improvements in the context of high energy cost.<sup>5</sup>

The previous LCC approaches with comparative cases illustrated how the different factors influence LCC; however, the results are constrained to the assumptions made in the individual cases as well as assumptions about future choices by consumers in replacing products. For example, scenarios of improved efficacy and price assume consumers will take advantage of these factors when buying replacement products. Lastly, scenario-based LCC answers the question of under which conditions longer lifetimes may be preferable, but do not necessarily give a more specific indication of optimal lifetimes and for an overall consumer market for the product.

In contrast to previous research on optimal lifetimes, the main objective of this study was not to develop scenarios for LCC, but rather to track the role of durability (focussing on lifetime) based on a snapshot of a current LED market. Web crawling techniques for tracking attributes in a market have been proposed as a way to generate data to effectively calculate and track LCC for product markets (Van Buskirk 2015; Bennich et al. 2017). Similarly, the research presented in this paper analysed web crawled market data, using an LCC methodology

<sup>4</sup> The test procedures are found in 10 C.F.R. 430.23(ee) (Appendix BB to Subpart B of Part 430) <https://www.gpo.gov/fdsys/pkg/CFR-2017-title10-vol3/pdf/CFR-2017-title10-vol3-part430-subpartB-appBB.pdf>

<sup>5</sup> The study used 0.3€ in Denmark as the high energy cost point.

to then determine optimal lifetimes for LED lamps in a market.

### LCC methodology

In the preparatory studies for the lighting product ecodesign standards (VITO and VHK 2015a), LCC for base cases were calculated as:

$$LCC = PP + PWF \times OE + EoL \quad (S1)$$

where  $LCC$  is life cycle costs,  $PP$  is the purchase price,  $OE$  is the operating expense,  $PWF$  is present worth factor, which is a factor of the product life and the discount rate and  $EoL$  are the end of life costs.

Similar to the EU Methodology for Ecodesign of Energy-related Products,<sup>6</sup> this paper defines LCC as:

$$LCC = P_A + PWF \cdot P_E \cdot UEC \quad (S2)$$

where  $P_A$  is the appliance price,  $PWF$  is the present worth factor,  $P_E$  is the price of electricity and  $UEC$  is the annual unit energy use. End of life costs are excluded from this analysis as they constitute a very small portion of the LCC for LED lighting products<sup>7</sup> and these costs are likely incorporated in the purchase price for EU countries where the WEEE Directive applies.

The LCC has a dependence on durability because of the relationship between lifetimes ( $L$ ) and the present worth factor ( $PWF$ ), in which the durability of a product determines the lifetime. The relationship between  $PWF$  and lifetime is provided by the following equation:

$$PWF = \frac{1 - (1 + i)^{-L}}{i} \quad (S3)$$

Where  $i$  is the interest or discount rate and  $L$  is the product lifetime. If the model is optimised to minimise LCC (applying an LCC optimisation regression method from Van Buskirk et al. 2014<sup>8</sup>), both  $PWF$  and  $UEC$  are optimised. Under  $UEC$  optimisation,

$UEC$  decreases with increasing  $PWF$ , which in turn increases with lifetime.

Dividing by the  $PWF$  (which takes into account the influence of inflation and discount rates) gives the annualised LCC:

$$\frac{LCC}{PWF} = \frac{P_A}{PWF} = +P_E \cdot UEC \quad (S4)$$

Annualised LCC measures the costs of the lamps that may occur every year (taking into account that these are not regular). The focus is on the change in  $P_A/PWF$  with respect to the lifetime in hours. To do this, the LED models in the data were binned into four categories:  $\leq 15,000$ ,  $20,000$ ,  $25,000$  and  $\geq 30,000$  hours and the price regression coefficients for each bin were calculated for a selected subset of LED lamps.

The regression results were then used to calculate  $P_A/PWF$  as a function of lifetime.  $PWF$  is also dependent on the intensity of operation, so  $PWF$ s for three different consumer use scenarios, based on hours of operation per year—1000, 2000 and 4000—were considered. Then,  $P_A/PWF$  was calculated for each of the cases, yielding three curves for the 1000, 2000 and 4000 hours per year as well as the minima (i.e. optimal cost points) for the 1000, 2000 and 4000 hours/year use scenarios, respectively. While the effect of energy use ( $UEC$ ) is not modelled, the implications of the model in relation optimising LCC with respect to  $UEC$  are discussed. The method described above assesses the optimum lifetime for the entire market studied, accounting for the product attributes of total lumen output, lumens/watt efficacy and colour temperature.

### Data

The data used for the regression analysis were 344 LED products on the online market in Sweden and Denmark in December 2016, focussing on the most common category of LED retrofit lamps for households (“klot” in Swedish or “A” lamps) with E27, E14 and B22 bases. To construct the dataset, web crawling was used, which is a technique for extracting information from websites, transforming unstructured data on the web into a structured dataset (i.e. Excel sheet with features including brand (masked), price, lumen output, power, colour rendering index, temperature, among others) (see Van Buskirk and Richter 2017). Such methods have been applied to price monitoring and calculation of learning

<sup>6</sup> This methodology can be found at <https://ec.europa.eu/docsroom/documents/10024/attachments/1/translations/en/renditions/pdf>.

<sup>7</sup> For example, approximately €0.04 per LED is the end of life cost charged to producers in the Danish EPR system based indicative fees charged by a Lighting Producer Responsibility Organization; see [www.lwf.nu](http://www.lwf.nu).

<sup>8</sup> LCC optimization method is only briefly presented here; for a full explanation, please refer to supplementary data (“Supporting Information”) which can be accessed online in Van Buskirk et al. (2014).

**Table 1** Data characteristics for LED lamps in each lifetime category (Van Buskirk and Richter 2017)

Lifetime	≤ 15,000 h ( <i>n</i> = 130)	20,000 h ( <i>n</i> = 45)	25,000 h ( <i>n</i> = 139)	≥ 30,000 ( <i>n</i> = 30)
Price (€ based on 1SEK = €0.105)	AVG, 13 Range, 3–100.7	AVG, 15.7 Range, 0.95–68.1	AVG, 14.25 Range, 2–75.6	AVG, 15.2 Range, 2–41
Luminous flux (lm)	AVG, 475 Range, 8–1800	AVG, 489 Range, 110–2200	AVG, 573 Range, 136–1522	AVG, 455 Range, 82–1500
Efficiency (lm/W)	AVG, 83 Range, 16–128	AVG, 72 Range, 37–100	AVG, 79 Range, 46–125	AVG, 68 Range, 27–120
Correlated colour temperature (CCT) (K)	AVG, 2700 Range, 1900–6500	AVG, 2850 Range, 1800–6500	AVG, 2700 Range, 2100–6500	AVG, 3000 Range, 2700–6000

curves for LED household lamps (see Gerke et al. 2015), as well as monitoring of general attributes of a given product market over time (Bennich et al. 2017). Similar to the method used by Gerke et al. (2015), the dataset was cleaned to consider household lamps for which there was data for sale price, luminous flux, wattage and correlated colour temperature (CCT).

The products in the dataset were binned into four lifetime groups for analysis. Other characteristics of the LED products in the dataset are shown in Table 1.

The dataset showed a correlation between price and luminous flux, a weak correlation with CCT, but no significant correlation with efficiency or lifetime.<sup>9</sup> Gerke et al.'s (2015) study of LED lamps also found that brand names play a role in the price of LED lamps. A lack of relationship between price and efficiency has been highlighted as problematic in using LCC to set MEPS (see Siderius 2013) and the relationship between the lifetime and LCC is further discussed later in this paper.

### Optimal lifetimes for LED lamps

The results of the modelling for optimal lifetimes in the three use scenarios are shown in Fig. 1. This modelling focusses on the optimisation of the PWF in optimisation of LCC and shows lifetime related to the price/present worth factor. The “x” marks the minimum of the curves, or the lowest value for  $P_A/PWF$ , which then corresponds to the optimal lifetime for each scenario of yearly use. Assuming other factors of the LCC are also optimised (e.g. energy use), these lifetimes would in turn yield the

<sup>9</sup> The Pearson's correlation coefficient analysis was conducted with the dataset in SPSS considering a 0.05 significance level.

optimised or least LCC. The statistical error of the regressions does not allow for calculation of the optima with precision, but the calculation is illustrative that optimal lifetime for this range of LED lamps is close to 25,000 hours, with slightly longer lifetimes optimal the more intensely they are used. For comparison, the average lifetime for the data modelled in the sample is approximately 21,500 hours.

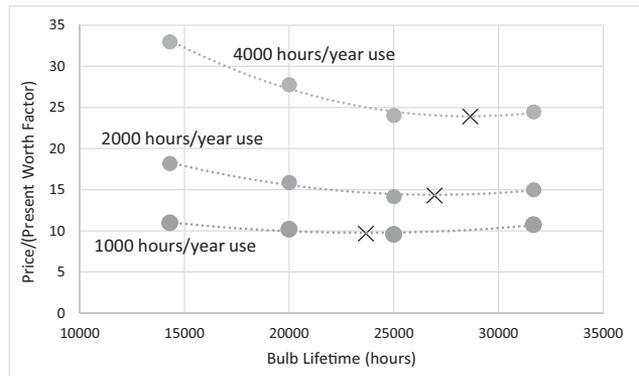
In this analysis, a discount rate of 6% was used in the calculation. Different assumptions about the interest or discount rates shift the  $P_A/PWF$ , favouring slightly longer lifetimes with a smaller discount rate and shorter lifetimes with a high discount rate (as shown in Fig. 2), in relation to the base case with 6% (as shown in Fig. 1). The effect of the discount rate is on the  $P_A/PWF$  ratio. A lower discount rate leads to a lower LCC/PWF while a higher discount rate leads to a higher  $P_A/PWF$ . Furthermore, the less intense use, the higher the impact of the discount rate on the shift of the optimal LCC point towards shorter lifetimes. It is also noted that for low intensities of use, the annualised price is roughly constant for different lamp lifetimes. Lamp lifetime has the largest impact on optimal LCC in the higher intensity use scenarios.

### Discussion

#### LCC approach

Our findings are in line with previous LCC product studies of appliances and street lighting that found generally more durable products yield a lower LCC compared to a standard product scenario (Boulos et al. 2015; Ochs et al. 2014; Tähkämö et al. 2016). However, the model does not capture the opportunity costs of longer

**Fig. 1** Model approximating optimum lifetimes (marked with x) for different scenarios of use



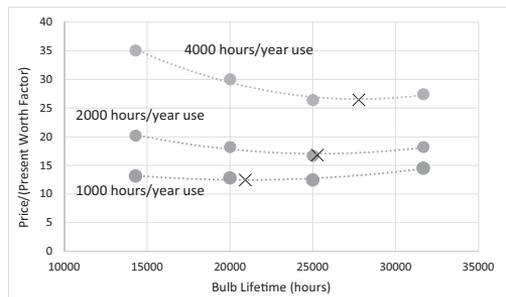
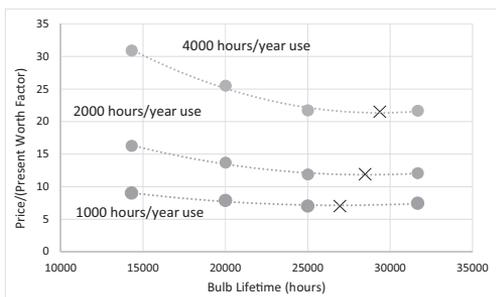
lifetimes (e.g. savings from more efficient replacement products). As the Richter et al. (2017) scenario-based approach demonstrated, however, these costs may only matter in certain scenarios in which there are significant improvements with price and efficiencies. Significant price improvements with household LED lamps may be unlikely after 2025, according to U.S. Department of Energy predictions (Navigant 2016). Efficacy improvements may still be possible, but it is also clear from the dataset in this study that there are a range of efficacies available to consumers, and if this parameter is not influential in their purchasing decisions (Rodemeier et al. 2017), consumers may not be buying products with the least LCC. This also suggests that optimal lifetime needs to be considered in the context of its relationship to other parameters in the LCC equation.

It should also be considered that the lifetimes in the modelling for LED lamps are the rated (i.e. stated) lifetimes supplied by the producers in selling the products on the market. In reality, actual lifetimes may differ

(Casamayor et al. 2015). However, there is little information yet on whether actual lifetimes for LED lamps differ greatly from the rated lifetimes and at present the rated lifetime is also the only information for consumers to incorporate this dimension in life cycle costing.

In theory, optimum LCC policies should move markets to an optimum where there are specific relationships between price, energy use, cost of energy and PWF (and by implication lifetimes and durability). Returning to the original equation for LCC (S2), it shows that in an optimised LCC, there is a direct synergy between smaller energy use and increased lifetime under LCC optimisation through the relationship between PWF (with lifetime implicit) and UEC (annual unit energy use).

If minimum standards on durability increase product lifetimes relative to an unregulated market, the increase in product lifetime increases PWF. In calculating optimum LCC with respect to energy use, higher values of PWF imply lower values of UEC at LCC optimum. In



**Fig. 2** Model with discount rates of 3% (left) and 9% (right). The “x” corresponds to the minimum of the quadratic trend line fit for each curve

other words, solving market imperfections for this parameter can increase lifetime, which in turn leads to increased product efficiency for LCC-optimised MEPS. This implies that durability standards can indirectly have an effect on climate change mitigation by allowing for LCC-optimised efficiency standards to become more stringent. This also implies a benefit to optimising lifetimes for both consumers and society.

Tracking optimal lifetimes with the method introduced in this paper can be easily implemented as a regular part of monitoring and analysis of product markets. As the case of LED lamps has demonstrated, this method is useful for monitoring how the optimal lifetime in a market compares to the average lifetime in the market. With real-time monitoring, the market average lifetime and optimal lifetime can also be tracked over time to show trends and changes. The case of LED lamps in the Swedish online marketplace demonstrated that the optimal lifetime may be higher than the average, suggesting a role for policies to push or pull the market in towards longer lifetimes and optimal LCC.

#### LCA approach

Longer product lifetimes have potential environmental benefits as well as consumer benefits. To consider this, the LCC approach can be complemented with an LCA approach, which can identify environmental impacts associated with durability. Studies considering optimal product lifetimes from an LCA perspective (looking at full range of impacts, or in some cases only energy demand) have demonstrated that longer product lifetimes can be preferred for some product groups, particularly when the environmental impacts in the extraction, production and waste phases are the most significant; this generally applies for ICT products (Bakker et al. 2012; Cooper and Gutowski 2015). For these products, extension of lifetime may be positive even if the technology is becoming more energy efficient (Bakker et al. 2014; EU Commission 2015; Prakash et al. 2015; VHK 2014). However, for energy-using products for which the majority of life cycle impacts occur in the use phase, studies have indicated that increased durability may not be preferred to replacement with more efficient products (Boulos et al. 2015; Cooper and Gutowski 2015; Gutowski et al. 2011).

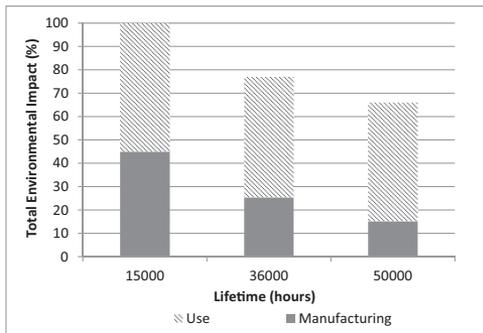
Tähkämö et al. (2013) examined the role of lifetime in influencing the overall environmental impact for the case of an LED downlight luminaire. The authors found

that the average environmental impact of a luminaire with 50,000 hours useful life was 34% lower (with a range of 2–70% among different impact categories) and 36,000 hours useful life was 23% lower (1–47%) compared to 15,000 hours useful life. The difference in impacts varied depending on what impacts were being considered, with the largest differences evident in the waste categories (both hazardous and non-hazardous) and the smallest in the primary energy.<sup>10</sup> A more recent LCA also confirmed the findings of greater overall environmental impacts associated with shorter lifespans for LED lamps (see Casamayor et al. 2017).

A review of several LCAs of lamps, including LED lamps, found that the energy consumption in the use phase generally dominates the total life cycle environmental impacts (Tähkämö and Dillon 2017). However, certain factors can influence the distributions, including lifetimes. Tähkämö et al. (2013) also found that the shorter the LED lifetime, the larger the share of manufacturing in the total life cycle impacts (due to the need for manufacturing additional replacement lamps), as shown in Fig. 3. The results of the Tähkämö et al. (2013) study were confirmed in a more recent comparative LCA for LED lighting products, which also considered an even shorter scenario of 1000 hours lifetime (compared to 15,000 and 40,000 hours lifetimes) (Casamayor et al. 2017). Not only did the assumption of shorter lifetimes result in significantly higher impacts of both LED products considered, but it also resulted in the main environmental impacts coming from the manufacturing, rather than the use phase.

The relative importance of the manufacturing versus use phase also varies depending on the assumptions about the energy mix during the use phase. An energy mix composed of higher renewable energy sources changes the dynamic of the impact, with increased renewable energy resulting in a decreased impact of the use phase and increasing the relative impact of the manufacturing stage, relative to the overall life cycle impact (Tähkämö 2013). The implication of this is that longer life lighting products might be even more important in the context of decarbonised energy mixes, as the increased relative impact from manufacturing implies using the product longer rather than shorter would be

<sup>10</sup> It should be noted that the differences are far less in considering energy impacts than considering other impacts related to waste, water pollution, resource efficiency, etc.



**Fig. 3** Overall life cycle environmental impacts by life cycle phase and lifetime considered. Source: own, based on relative impact data from Tähkämö et al. (2013) (Note that transport and end of life were shown to have almost no effect on the overall environmental impacts, so these are omitted.)

desirable to allocate the manufacturing impacts over a longer functional lifetime.

However, these prior LCAs have considered lifetimes with the assumption of identical replacements products (i.e. products with a lifetime of 15,000 hours required three replacements identical to the first LED product to meet the same function as the LED product with a lifetime of 50,000 hours). In reality, consumers can replace shorter life products with newer, improved products. Scholand and Dillon's LCA study (2012) for the U.S. Department of Energy projected the efficacy for LED lamps would improve from 65 to 134 lm/W from 2012 to 2017 and the 2017 LED lamp, which resulted in 50% less overall environmental impacts compared to the 2012 LED lamp. At the same time, there are also material developments to consider, for example, decreased use of aluminium for heat sinks, which can also decrease environmental impacts (e.g. Scholand and Dillon 2012). The study did not, however, consider then whether replacing the 2012 lamp before its lifetime of 25,000 hours would result in less environmental impact than continuing to use the 2012 lamp until the end of its lifetime.

Thus far, LCAs for LED lamps have not considered the question of optimal lifetimes taking into account improving LED lamps as replacements. Initial exploratory research using a scenario-based LCA approach indicated that there can be trade-offs between energy-related and resource-related impacts (Richter et al. 2017). Such trade-offs would disappear as the technology matures (and the scenario becomes more akin to the

identical replacements considered by earlier LCA studies of LED lamps). It is also therefore relevant to consider the projections for development of LED technology in assumptions about replacement scenarios (e.g. the maximum LED package efficacy is projected to increase up to 250 lm/W by 2025; see U.S. Department of Energy 2013). Continued research developing the scenario-based LCA approach would be a complementary approach for determining optimal LED product lifetime to better understand the environmental benefits and trade-offs that may result from longer lifetimes. This would, in turn, inform the optimal timing of policies promoting longer lifetimes from an environmental perspective.

### Policy options for longer life LED products

When it comes to LED lamps, the controversies surrounding the banning of traditional incandescent lamps and the mistrust of lighting regulations (cf. Sachs 2012) mean that it is paramount to set quality standards for new lighting technologies. Therefore, it is appropriate that there are minimum durability/lifetime standards currently set as a means to guarantee product quality and increase consumer confidence in LED lamps, which is important for uptake of LED lamps (Sandahl et al. 2014). The current mandatory standards for durability in the EU ecodesign regulations are shown in Table 2.

However, in comparison to the 6000-h minimum, the analysis of a current LED market from an optimised LCC perspective suggest the optimal lifetime for household LED lamps is around 25,000 hours. Consumers may also expect longer minimum lifetimes for LED lighting products, since most of the LED lamps on the

**Table 2** Ecodesign requirements for LED lamps related to durability and quality

Lamp survival factor at 6000 hours	≥ 90%
Lumen maintenance at 6000 hours	≥ 80
Number of switching cycles before failure	≥ 15,000 if rated lamp life ≥ 30,000 hours, otherwise ≥ half the rated lamp life expressed in hours
Premature failure rate (maximum number of failure products in %)	≤ 5% at 1000 hours
Colour rendering requirements for various applications	≥ 80

market analysed claim lifetimes of at least 10,000 and up to 50,000 hours. Thus, minimum functionality requirements on lifetimes are lower than the optimal lifetimes and likely also lower than consumer expectations. Moreover, the transition of the lighting market towards LED lamps has meant a rapid improvement in durability of lighting products, with an increasing number of models in the market lasting longer periods and with good quality lighting output (Bennich et al. 2015).

If increased longer lifetimes are desirable, as the findings from modelling LCC in the market suggest, one way is to strengthen the minimum requirements in the ecodesign regulation. However, mandatory standards are not the only policy option and can have drawbacks; therefore, two other options are also considered: mandatory labelling and (mandatory or voluntary) customer warranties. These approaches each have their merits and limitations, which are discussed and summarised at the end of this section.

#### More ambitious mandatory ecodesign requirements

Generally, mandatory durability standards have benefits compared to the other policy options such as warranties and labelling. Firstly, it allows policymakers to make the appropriate trade-offs between different functions (e.g. energy use, technological developments and durability), based not only on optimal LCC but also technology assessments and LCAs. Secondly, the high complexity of establishing ‘durability’ for lighting, and the problems for consumers to understand information about durability, implies that mandatory requirements can be a good idea cf. to labelling and warranties.

The increasing importance of resource efficiency is likely to raise the relevance of more ambitious durability standards in the near future (not only for Circular Economy objectives, but also for climate policy objectives to address embodied emissions; see Scott et al. 2017). For example, long lifetimes can enable design where it is possible to repair, reuse and upgrade components or complete lighting solutions (Dzombak et al. 2017; Hendrickson et al. 2010). In turn, longer lifetimes may make efforts to design with modularity and standardisation more viable. These are currently challenging, but being discussed (see Gossart and Ozaygen 2016). While the LCC market analysis indicates a role for more ambitious standards, additional research is needed to examine optimal durability from

an LCA perspective, where issues such as resource use and production phase impacts are part of the analysis.

In addition, practical methods for lifetime testing are required to implement and enforce any mandatory standards. In order to enforce such standards, there would need to be practical testing procedures (this applies also for labelling). Currently, standard testing methods consider the lifetime of the LED components rather than the whole system and often focus on lumen depreciation over catastrophic failure (i.e. complete non-functioning) though both are of concern (Narendran et al. 2016). Practical methods that can reliably predict the important sources of failure are a necessary first step in setting minimum standards. Such methods that stress test important parameters (e.g. switch cycles, change in temperature) and consider all important components in the lighting system (not only the LED but also e.g. drivers, solder between the LED and PCB, etc.) (Narendran et al. 2016). While these are promising developments in accelerated testing procedures (Narendran et al. 2016; Narendran, personal communication 3 March 2017), there may still be issues with how to establish test methods in legislation and the practical enforcement by member states.

Some jurisdictions like California are making requirements based on minimum *rated* lifetimes (and interestingly requiring longer lifetime minimums of 25,000 for higher intensity of use applications—in line with findings in this study) (California Energy Commission 2016). The IEA 4E SSL Annex also has voluntary performance standards with minimum rated lifetime requirements over 15,000 hours (in addition to 6000-h lumen maintenance and survival factor tests and endurance tests for switch cycles; see IEA 4E SSL Annex 2016). While development of an acceptable accelerated test is preferable, the currently available combination of LM80 measurements and TM21 extrapolation to assess the lifetime could be used in the interim as testing methods continue to be refined with new research and available data.

#### Mandatory labelling

Lifetime information is already required on lamp packaging, but not for specification in a label (i.e. the energy label). There is growing momentum in the EU to include durability requirements in mandatory energy labels, and this is an option that allows consumers to differentiate products not only in relation to energy efficiency but

also durability. In the EU debate, there has been proposals that most products should be labelled with an ‘average expected product lifetime’, calculated through standardised methodologies, to allow better consumer decision-making (RREUSE 2015). Already today, energy labelling in the EU includes some non-energy-related information. One example is the label for vacuum cleaners, as it is a multi-dimensional label, where mandatory information includes energy rating, annual energy use, emission (dust in exhaust air), noise level, pick-up performance for carpets and pick-up performance for hard floors.

However, there is some general concern regarding the design of energy labelling and how consumers interpret the energy efficiency information (Molenbroek et al. 2014; Waechter et al. 2015) that implies it can be difficult to also include information on expected lifetime. The first question is whether the producer should account for minimum lifetime, or expected lifetime of the product, and how the choice of parameter can be communicated in an easy-to-understand fashion to consumers. Further, as discussed previously, lifetime entails many dimensions in the case of lighting. It is not realistic to expect consumers to understand all of them, nor to have information about all of them on the product (i.e. expected lifetime in terms of acceptable luminous flux, expected lifetime for acceptable colour rendering, etc.). One potential way forward is that the labelling regulation stipulates a minimum for all these categories and that the expected lifetime indicated by the producer implies that all these dimensions are fulfilled to satisfactory level during the indicated lifetime. For most LED applications, it is primarily lumen output that matters, so lumen depreciation could be a potential first category to include in labelling.

Generally speaking, the issue of whether and how consumers react to labelling is quite complex (see e.g. Waechter et al. 2015; Dalhammar et al. 2018). For example, there are indications that this partly depends on the product group, as consumers are more likely to consider energy labelling for some product purchases than others. Research on consumer behaviour with LED products has also shown energy efficiency does not motivate many consumers (Rodemeier et al. 2017), so it is unclear how consumers will act upon durability information for lighting products. Further, consumers have an easier time understanding some information provided in energy labels than others. Most notably, consumers understand the information provided on

what energy class an appliance belongs to (in Europe this is presented through letters, with ‘A’ being the best-performing category), but often do not understand other types of information provided through the labelling such as information on expected annual energy use (Waechter et al. 2015). Furthermore, there are indications that the European practice of updating standards through adding additional plus signs to the letters (e.g. ‘A+’ and ‘A++’) is confusing (Dalhammar et al. 2018).

The main advantage of using labelling to communicate lifetime is that it allows consumers to choose products according to preferences and provides for competition in the market. The main disadvantage is that there may be incentives to cheat for producers as there are challenges related to market monitoring and product testing. Further, the wide range of products and applications may imply that it is hard to put a meaningful number for the expected lifetime in all cases, as LEDs are often integrated into various systems (Next Generation Lighting Industry Alliance 2014).

#### Warranties and guarantees

Another possible option for ensuring the lifetime of LED lamps is extended guarantees or warranties. A warranty is a term of a contract, breach of which gives rise to a claim for damages, but (usually) not the repudiation of the whole contract. Such warranties can be pursued either through *mandated* warranty periods, or through *voluntary* warranties. As a baseline, consumers in most jurisdictions have a legally mandated warranty for a certain period of time, often ranging from 1 to 3 years. Both in the EU and the USA, there are different rules in different jurisdictions related to warranties for consumers. Some jurisdictions such as Iceland and Norway also provide consumer rights for non-conforming products for a longer period of 5 years when the products are meant to last for a considerably longer time (Tonner and Malcolm 2017). It should be noted that it is not only the general warranty that is of importance; in some jurisdictions, producers’ claims about lifetime could lead to a consumer claim if the product falls short of its indicated lifetime, as this can constitute a breach of satisfactory quality (Stone 2015).

It is not only the length of the warranty per se that is of importance, but also other factors, most notably when the burden of proof for showing that a product defect was present at the time of purchase is transferred from seller to buyer, as this can be difficult to prove. In most

EU countries, this burden of proof is moved from the seller to the buyer after 6 months. The EU NGO RREUSE has proposed that products can be more durable and repairable if the burden of proof lies with the seller/manufacture for 2 years instead of 6 months, and that this can be enforced through higher “Mean Time Between Failure (MTBF)” requirements for critical sub-assemblies such as those with electromechanical parts/components (RREUSE 2015).

EU law on consumer protection is a mix of acts that aim at minimum harmonisation and acts that aim at total harmonisation. The main benefits of minimum harmonisation are that it secures minimum rights for the consumer while allowing Member States to strengthen consumer protection. The main drawback is that practices in EU Member States differ, which forces producers to adopt different business practices throughout the EU (Maňko 2015).

Whether warranties actually provide incentives for durability depends on the circumstances. When it comes to LED lamps, the rather limited cost of the product and its longevity may mean that consumers do not pursue a warranty claim, e.g. because the reward is limited compared to the effort. And, consumers may be suspicious towards warranty claims from firms that may be on the market only temporarily (Price and Dawar 2002). Industry associations seem to view the use of warranties, reliability claims, etc., as good source of information for customers (Next Generation Lighting Industry Alliance 2014), but in reality, this mainly applies to professional users as private consumers cannot be expected to understand this information and assess its validity.

Generally, for most products groups, there are indications that EU companies prefer ecodesign requirements setting mandated minimum lifetime in hours, to mandated extended warranties in years (Dalhammar 2016). The reasons are likely that (1) guaranteeing lifetime in hours rather than years protects the producers from intensive product use by consumers and (2) mandated long warranty times undermine the lucrative business of selling longer warranties to consumers (Dalhammar 2016). Also for LED lamps, providing warranties in hours (in use) rather than years appears most suitable (Next Generation Lighting Industry Alliance 2014).

For professional users, there is the option for producers to voluntarily offer extended warranties that include both replacements of faulty products and other services such as maintenance. The buyers can then

choose a contract that suits their risk preferences and the technical installation. It is doubtful if a mandated warranty should be legislated for B2B relations, as the LED lamps can be used for many different purposes. Regarding mandatory warranties for consumers, it is also doubtful if LED guarantees going beyond what is provided through general consumer protection legislation should be implemented, although such warranties could further improve consumer confidence in LED products.

#### Summary of options for increased durability

Table 3 gives a summary of the advantages and disadvantages of different policy approaches. While this study and other LCC studies suggest a role for policy to promote longer lifetimes to achieve optimal durability and optimal LCC, there can also be arguments against such policies. The regulation stipulating functionality requirements stated that their aim is “to ensure consumer satisfaction with energy-saving lamps, in particular LEDs...”<sup>11</sup> It has been argued that domestic consumers are not usually interested in very durable products; whereas, professional buyers can make use of warranties when they want durable LED lamps (cf. Next Generation Lighting Industry Alliance 2014). This would imply that more policy drivers for inducing increased durability for lighting products are not necessary or desirable. While research in the USA has found that consumers do value durability as an attribute for lighting products, with stated willingness to pay more between 0.52 and 0.66 USD for every 1000-h increase in lifetime (Min et al. 2014), the purchase price of LED lamps has decreased dramatically in recent years, which call into question again the perceived value of longer lifetimes for consumers.

Another argument is that manufacturers are already selling LED lamps highlighting long-life LED products to consumers who value this, and this could in itself push the market towards increased durability without policy. At the same time, there is speculation about planned obsolescence for LED lighting products (MacKinnon 2016). The U.S. Department of Energy market analysis of LED lamps shows that there can be

<sup>11</sup> Commission Regulation (EU) No 1194/2012 of 12 December 2012 implementing Directive 2009/125/EC of the European Parliament and of the Council with regard to ecodesign requirements for directional lamps, light emitting diode lamps and related equipment, at (15)

**Table 3** Summary of options for increased durability requirements

Policy choice	Advantages	Disadvantages
Mandatory requirements	<p>Allows policymakers to make the appropriate trade-offs between different functions (e.g. energy use, technological developments and durability)</p> <p>The complexity of establishing ‘durability’ for lighting, and the problems of consumers to understand information about durability, implies that mandatory requirements can be a good idea cf. to labelling and warranties</p>	<p>By setting durability standards that goes further than a mere ‘baseline’, policymakers may interfere with decisions that are best taken by designers, based on customer needs and user patterns</p> <p>May be better to let customers use labelling to differentiate product lifetime according to their preferences</p>
Mandatory labelling	<p>Allows consumers to choose products according to preferences and provides for competition in the market</p> <p>Less intrusive for producers than mandatory lifetime requirements</p>	<p>Difficult for consumers to understand/ interpret the information</p> <p>Risk of cheating</p> <p>The broad range of LED products and applications can lead to quite varying definitions of lifetimes</p>
Voluntary extended warranties	Useful in B2B applications where buyers can interpret technical information and enter into relevant contracts that are suitable for the purpose where the LED products are used	Less useful for private buyers as the information is complex and the limited price of many LED products may mean that buyers are not very interested
Mandatory extended warranties	Could be useful for consumers and increase confidence in LED products	Not so useful in B2B relations

a range of design choices for LED lamps and they can be designed with or without trade-offs between different parameters, including energy efficiency and lifetimes (U.S. Department of Energy, Solid-State Lighting Program 2016). In light of environmental policies often having both energy and resource efficiency aims (the latter of increasing importance in the context of circular economy goals), such trade-offs should be avoided and optimising both energy and resource efficiency encouraged. Thus, policy addressing lifetimes may be relevant to ensure environmental benefits from longer lifetimes are realised in practice.

### Conclusions and recommendations

This paper has demonstrated how modelling the relationship between LCC and PWF can approximate optimal lifetimes for the product market being considered. The optimum lifetimes for the LED lamp market considered was indicated by the analysis to be higher (approximately 25,000 hours) than the market average for lifetime (21,500), suggesting there is likely a role for durability policies to move the market closer to its LCC optimum. The analysis also indicated that longer lifetimes are important when smaller discount rates and more intensive use

of a product are factors, suggesting LED lamps typical in intense-use applications of LED lamps should be the initial policy focus. There was also found to be a relationship in optimised LCC between longer lifetimes and lower energy use. The method presented in this paper can be useful for determining and monitoring optimal durability as part of tracking attributes and LCC in a product market. Further research can investigate optimal lifetimes for other products to compare to the case of LED lamp products presented in this paper.

The findings in this case motivate further investigation into the feasibility of setting more stringent lifetime requirements for LED lamps. It is recommended that the LCC approach adopted in this study is complemented by an LCA approach that also determines the environmental impacts of lifetimes and replacement scenarios for LED lamps, considering the context of continued development of LED technology and markets, to determine the appropriate timing for promoting durability from an environmental perspective. The paper discussed promoting durability through different types of policies, which have different advantages and drawbacks. Increasing stringency of lifetime requirements for the case of LED lamps also requires implementation of accelerated testing methods to ensure such standards can be practically enforced. Overall, it is recommended

that lifetime continues to be addressed first and foremost by minimum performance standards, but there is also a role for development of better labelling and warranties for these products in terms of durability.

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#### Compliance with ethical standards

**Conflict of interest** The authors declare that they have no conflict of interest.

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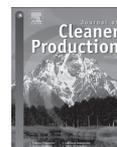
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Paper V







## Trade-offs with longer lifetimes? The case of LED lamps considering product development and energy contexts

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### ABSTRACT

Longer product lifetimes are promoted by the EU's Circular Economy Action Plan, but incentivising longer lifetimes could also result in trade-offs between different environmental impacts for some product categories. LED lamps are still experiencing improvements in efficacy and material design, which raises questions about whether longer lifetimes are desirable from an overall environmental perspective. Applying a comprehensive life cycle assessment using actual product cases from 2012 to 2017, the research builds on previous product lifetime studies and lighting product research to determine the scenarios in which longer lifetimes are desirable from an overall environmental perspective. The factors explored in the scenarios included improving products in terms of efficiency and dematerialisation as well as decarbonised electricity contexts. The results indicate that product replacement with improved products resulted in environmental benefits compared to keeping longer life products in use, but there are some trade-offs between environmental impacts. However, these trade-offs are minimised in the context of decarbonised electricity mixes and will further decrease as LED lamp technology matures and product development slows. The policy implications of the findings are also discussed.

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## 1. Introduction

In the transition to a Circular Economy, there is a need for more efficient use of resources and reconsideration of how products are designed. Promoting longer product lifetimes is a key component of Circular Economy policies for both the EU (EU Commission, 2015; Montalvo et al., 2016) and the member state level (Montalvo et al., 2016). At the same time, research notes that trends in lifetimes are getting shorter for some products; for example consumer electronics (Bakker, Wang, Huisman, & den Hollander, 2014; Prakash et al., 2016). This, in turn, has implications for resource efficiency and waste produced from higher volumes of product consumption (Rivera and Lallmahomed, 2016). Countries like France have responded with legislation targeting planned obsolescence specifically and there is increasing interest in further incorporating durability standards into the EU Ecodesign Directive and associated regulations (Maitre-Ekern & Dalhammar, 2016).

Lighting products are one of the first product categories for which there are durability standards in the Ecodesign Directive

(2009/125/EC). The requirements of regulations 244/2009 and 1194/2012 mostly focus on different dimensions of lifetime and set a minimum lifetime of 6000 h (Richter et al., 2019). Lifetimes of lamps may vary depending on environmental conditions and user behaviour (e.g., intensity of use, switching, etc.). Rated lifetimes, as used in declarations by manufacturers, are a combination of lumen depreciation and survival factor:

'lamp lifetime' means the period of operating time after which the fraction of the total number of lamps which continue to operate corresponds to the lamp survival factor of the lamp under defined conditions and switching frequency. For LED lamps, lamp lifetime means the operating time between the start of their use and the moment when only 50% of the total number of lamps survive or when the average lumen maintenance of the batch falls below 70%, whichever occurs first.<sup>1</sup>

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<sup>1</sup> See Annex II in Commission Regulation (EU) No 1194/2012 of 12 December 2012 implementing Directive 2009/125/EC of the European Parliament and of the Council with regard to ecodesign requirements for directional lamps, light emitting diode lamps and related equipment. <http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A32012R1194>.

Several manufacturers have promoted the long life of LED lamps, with some lamps introduced to the market claiming lifetimes exceeding 50000 h (Hixon, 2012). However, more recent trends in household LED lighting indicate that the costs for manufacturing LED lamps have decreased dramatically, which puts less pressure on manufacturers to make longer lifetime claims (Katona et al., 2016). There are also some actors who believe that manufacturers might intentionally produce LED lamps with shorter lifetime to increase sales (MacKinnon, 2016).

Research has examined optimal lifetimes for LED lamps from a life cycle cost (LCC) perspective, where the main parameters include the upfront purchase price of the product as well as the costs of use during its lifetime (e.g. electricity costs). The study found that the optimal lifetimes for lamps on the market in Sweden in 2016 was approximately 25000 h (Richter et al., 2019), higher than the current 6000 h minimum lifetime in EU legislation. The findings indicate that from an LCC (economic) perspective, a much longer lifetime than the legal minimum of 6000 h would be optimal. Longer product lifetimes have also been motivated by studies citing potential economic, social and environmental benefits (Montalvo et al., 2016).

However, long lifetimes are not always associated with lower environmental impacts for all products; in fact, there can be trade-offs between different environmental impacts in promoting longer lifetimes. An example is electrical and electronic equipment with improving energy efficiency. Shorter lifetimes are often preferable for these products since there are environmental benefits derived from the efficiency improvements in replacing the old products which outweigh the environmental benefits of longer lifetimes (Bakker et al., 2014; Boulos et al., 2015; Cooper and Gutowski, 2015). Thus, Circular Economy policies promoting longer lifetimes for such products could result in trade-offs that could undermine the environmental benefits.

Life cycle assessment (LCA) is a method to assess the environmental impacts of products and can be used to explore the question of optimal lifetimes. In their study exploring longer product lifetimes, Bakker et al. (2014) apply a “fast track” LCA of the optimal durability for refrigerators and televisions. While the study gives an indication of optimal lifetimes, the authors also mention the “fast track” LCA as a limitation and suggest a more comprehensive LCA would be interesting for future research. Previous research has also addressed whether extended lifetimes for vacuum cleaners (Bobba et al., 2016) and washing machines (Ardenne and Mathieux, 2014) result in reduced environmental impacts. The studies constructed baseline (i.e. product “A” replaced with new product “B” versus  *durable* product “A” replaced at a later time with new product “B”) and used environmental assessments based on LCA to test varying assumptions about lifetime extension and energy efficiency improvements. The studies note that the focus was not to present a comprehensive LCA of the product, but rather provide an indication of whether durability made sense for the product cases considered. To simplify the method, the studies restricted the detailed analysis of environmental impacts to a few impact categories: global warming potential (GWP), abiotic depletion, and human toxicity (Bobba et al., 2016) or terrestrial ecotoxicity (Ardenne and Mathieux, 2014). The studies found that some extension of the lifetime could reduce the GWP even if the replacement product was more energy efficient while the other impact categories showed lower impacts with lifetime extension. Ardenne and Mathieux (2014) also noted the challenges in making assumptions about product development (particularly when product “A” is still in an early development stage) and recommended conducting sensitivity analyses of key parameters; for example, the energy efficiency of replacement products. Bobba et al. (2016) conduct such a

sensitivity analysis considering decreased energy consumption of the replacement vacuum cleaner (i.e. product “B”). The study found that the replacement resulted in less global warming potential impact if the product replacement was 25% more efficient, but did not result in less impact in the other categories examined (abiotic depletion and human toxicity).

Boulos et al. (2015) applied a combined LCC and LCA approach considering durable versus energy efficient models of fridge-freezers and ovens. The study used the International Reference Life Cycle Data System (ILCD) 2011 method (Wolf et al., 2012) to characterise 15 environmental impact categories and identify trade-offs between the impacts. They found replacing an oven or refrigerator with a 10% more energy efficient new model was preferable to the durable model for most environmental impacts considered, with the exception of impacts stemming from the production or end-of-life phase (e.g. ozone depletion, human toxicity, freshwater ecotoxicity, and mineral, fossil and renewable resource depletion for the refrigerator), which were always less with the durable model. While the research identified the trade-offs between different kinds of impacts, there was no further investigation of the relative significance of these different impacts. The research demonstrated the importance of the assumptions about product development (particularly energy efficiency improvements) when considering the role of lifetimes, as these have large implications for the outcomes. Further, it was evident that the actual trade-offs between impacts are product specific.

Previous LCA research for lighting products has found that longer lifetimes decrease overall environmental impacts from LED lighting products (Casamayor et al., 2017; Casamayor et al., 2015; Tähkämö, 2013), and there has been some research promoting design for longevity for lighting products (Casamayor et al., 2015; Dzombak et al., 2017; Hendrickson et al., 2010). However, previous LCA research did not consider the improving efficacy of LED lighting products, which has been substantial (see e.g. Bennich et al., 2015; Gerke et al., 2015), when considering lifetimes. Some research has indicated that there are potential trade-offs between different lifecycle phases (Nissen et al., 2012). The question remains whether longer lifetimes for LED lighting products produce less environmental impact in the long term, and what trade-offs there may be in promoting longer lifetimes for such products.

The aim of this research was to build on previous LCA-based durability studies and lighting product research to explore possible trade-offs with promoting longer lifetimes for LED lamps and determine the contexts in which longer lifetimes are desirable from an environmental perspective. The research applied LCA methodology similar to previous research (Ardenne and Mathieux, 2014; Bobba et al., 2016; Boulos et al., 2015; Richter et al., 2017) considering lifetimes in relation to other dynamic factors, including improved efficacy of the LED products and design changes (including dematerialisation). While previous research has focussed on prospective assumptions about product development, this research considered a retrospective case with more specific data from products available in 2012 and 2017 to construct scenarios. 2012–2017 was a period of rapid development of LED lamps, allowing for more empirical consideration of a situation explored hypothetically in other studies. In addition to scenarios considering product development factors, the scenarios in this research also considered different electricity mixes, as this parameter can have a strong influence on the results of LCAs for lighting products (Franz and Wenzl, 2017; Tähkämö, 2013; Welz et al., 2011). This implies that whether longer lifetimes for LED lamps are preferable from an overall environmental perspective is also specific to the electricity context considered. The electricity mix is also important considering the fact that electricity mixes are expected to change, albeit slowly, in response to climate and energy

policies in the near future.

The article first describes the methodology of the study and details of the scenarios are presented, followed by the results of the different modelled scenarios. Possible approaches for how to handle trade-offs are further explored through alternative characterisation and normalisation methods. Lastly, the remaining challenges of the research approach and implications of the results for Circular Economy policies promoting longer product lifetimes are discussed.

## 2. Methodology

The LCA method in this study followed the ISO 14040 (ISO, 2006a, b) and 14044 standards (ISO, 2006a, b), and analysis was conducted using SimaPro (V.8.5.0) software with the Ecoinvent (V3.3) database; these have been used in several studies about LCA of LED lamps and LED lighting products (Boulos et al., 2015; Casamayor et al., 2017; Tähkämö et al., 2013). The life cycle inventory (LCI) included the processes and materials for the LED lamps (the full LCI can be found in the Appendix and is summarised in section 2.4). The inventory was constructed with the bill of materials (BOM) from an LED lamp from 2012 and three LED lamps from 2017 (data from Scholand and Dillon, 2012 and Dillon et al., 2019). The BOM was then matched with Ecoinvent data with the SimaPro software.

### 2.1. Goal and scope of LCA

The goal of the LCA is to explore factors of product development and electricity mix in relation to the LED lamp product lifetimes to assess in which cases longer product lifetimes result in lower overall environmental impacts. The results of the study can be used to inform policies such as lifetime standards for lighting products. This research considers four approximately 800 lumen retrofit LED lamps (A-19 shape with E-27 base), building on previous LCAs of such products from 2012 (Scholand and Dillon, 2012) and three lamps from 2017 (Dillon et al., 2019).

### 2.2. Functional unit

The choice of an appropriate functional unit is fundamental to LCA. Lumen-hours, the functional unit used by the two studies on which this research is based, is one of the most common functional units for lighting products, incorporating important functional parameters of luminous flux and operating hours (Tähkämö and Dillon, 2017). Casamayor et al. (2017) noted that quality parameters such as correlated colour temperature (CCT) and colour rendering index (CRI) can also influence energy efficiency (e.g. high CCTs tend to be more slightly more efficient and higher CRIs less

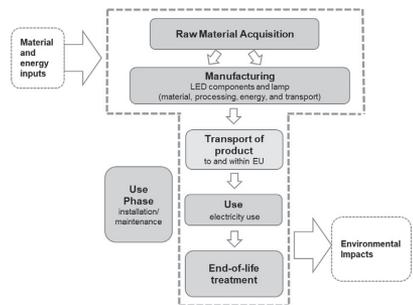


Fig. 1. System boundaries.

efficient). However, in this study the lamps compared are for the same 60 W equivalent retrofit household application and the CCT, CRI and conditions of use (e.g. room temperature and intensity of use) are assumed to be the same.

The choice of lumen-hours as a functional unit is appropriate for enabling comparisons between lamps (this unit is easily used for comparison between lighting products) as well as with previous LED lamp LCA studies upon which this study builds (Dillon et al., 2019; Scholand and Dillon, 2012). Thus, the functional unit used in this research was 20.3 million lumen hours (Mlmh) – equivalent to the function of the base case product (2012 LED lamp from Scholand and Dillon, 2012).

### 2.3. System boundaries

The product system considered in all scenarios was cradle to grave, i.e., raw materials acquisition, manufacturing, transport, use, and end-of-life. Fig. 1 shows the main life cycle stages considered.

### 2.4. Life cycle inventory

Table 1 indicates important attributes and materials for the LED lamps considered in the 2012 LCA by Scholand and Dillon (2012) and the updated study by Dillon et al. (2019). Only a summarised inventory of major material groups is presented here. The inventory data used as the long life base case in all scenarios was obtained from the 2012 United States (U.S.) Department of Energy's (DOE) comprehensive LCA of an 800lm 12.5W E-27 LED lamp in 2012 with a lifetime of 25000 h, CCT of 2700 K and CRI of 80 (Scholand and Dillon, 2012). The updated comparison of the 2017 LED lamps to the 2012 LED lamp concluded that the performance of LED lamps in

Table 1  
Overview of case products.

Product (source)	2012 LED lamp (Scholand and Dillon, 2012)	2017 LED lamp replacement 1 (Dillon et al., 2019)	2017 LED lamp replacement 2 (Dillon et al., 2019)	2017 LED lamp replacement 3 (Dillon et al., 2019)
Luminous flux (lm)	812	800	800	815
Power (W)	12.5	8.5	9.5	11
Lifetime (h)	25000	10950	25000	25000
Aluminium (g)	68.20	11.03	20.69	–
Other metals (g)	10.65	1.92	2.19	1.90
Electronics (g)	80.25	13.14	21.72	17.38
Plastic (g)	11.10	19.26	35.71	27.55
LEDs (pieces)	12	11	20	8
Product mass (g)	176.00	45.66	81.96	47.09
Packaging (g)	37.00	17.99	30.68	26.72

**Table 2**  
Overview of use stage for static scenario.

Lamp in scenario	2012 LED lamp	2012 LED lamp	2012 LED lamp
Lifetime (h)	25000	12500	5000
Number of products needed for 20.3Mlmh	1	2	5
Electricity use for 20.3Mlmh (kWh)	312.5 kWh	312.5 kWh	312.5 kWh

**Table 3**  
Overview of use stage for improved lamp replacement scenario.

Lamps in scenarios	2012 LED lamp no replacement	2012 LED lamp 2017 replacement 1	2012 LED lamp, 2017 replacement 2	2012 LED lamp 2017 replacement 3
Additional number of products needed for 20.3Mlmh	0	1.83	0.8	0.8
Electricity use for 20.3Mlmh (kWh)	312.5	232.5	252.5	282.5

terms of environmental impacts had improved from 2012 (Dillon et al., 2019). The study confirmed the dominance of the use phase, making greater efficacy of the newer lamps advantageous even if the lifetime was shorter. While improved compared to 2012, it is clear that the luminous efficacy (lm/W) still varied between the lamps, as did the range of materials. Aluminium heatsinks were noted as large contributors to environmental impacts from the manufacturing phase (Dillon et al., 2019; Scholand and Dillon, 2012). As can be seen from Table 1, the aluminium content decreased in some newer products and was even designed out in product 3. The total lamp weight has also reduced from 2012 to 2017, indicating improving material efficiency as well.

Four changes were made to the overall inventory from the original research cited: 1) the electricity mix in the use stage was changed to the European context (from a U.S. mix in the original model); 2) as wafer sizes used in LED manufacturing have increased in size (and therefore efficiency) in recent years (Dillon and Ross, 2015; Roos, 2017), the yield of the LED die component for the 2017 lamps compared to the 2012 lamps was increased from 2438 to 3500 (using the projections in the Scholand & Dillon LCA study (2012)); 3) the 11g plastic phosphor host was modelled as polycarbonate plastic rather than rare earth mix indicated in the original inventory, though the 1g phosphor coating itself is still modelled as per the original inventory (see Scholand and Dillon, 2012 p. 35 p. 35)<sup>2</sup>; 4) an end-of-life waste treatment scenario in the European context was developed considering a 30% collection and recycling rate, which aligns with the European average recycling rate for gas discharge lamps (which are in most current cases recycled together with LED lamps, so assumed to be indicative - see Richter and Koppejan, 2016). Recycling of 50% of the aluminium (from the lamps collected as well as some discarded into other waste streams), 30% of glass was assumed treated as part of the collected lamp waste stream, while plastic was assumed to be incinerated based on waste material routes from literature and practice (e.g. Richter and Koppejan, 2016, Nordic Recycling, 2016). More detailed inventories can be found in the Appendix and the dataset in supplementary materials.

## 2.5. Life cycle impact assessment (LCIA) method

The environmental impacts were assessed using the life cycle

<sup>2</sup> The reason for changing the material from REE to plastic is that we could not find any evidence in literature about REE content in LEDs that the amount would be higher than 1g and certainly not as high as 11g (c.f. Andre, Soderman, & Tillman, 2016; Buchert et al., 2012; Franz and Wenzl, 2017; Machacek et al., 2015; Rollat et al., 2016; Tunsu et al., 2015). Also, as the description of the material in the original inventory is plastic host, we believe this was an error in the original inventory.

impact assessment (LCIA) method ReCiPe (see Huijbregts et al., 2016). The midpoint level was used to give a more detailed indication of what impact categories are affected by the assumptions of the scenarios midpoint. The midpoint level assesses environmental impacts categorised into 18 different impact categories (as opposed to the ReCiPe endpoint level which further aggregates impacts into 3 categories and even a single score). The ReCiPe method harmonises the CML (Centrum Milieukunde Leiden) methodology and Eco-indicator 99 methodology and is one of the most recently updated impact assessment methods. The hierarchist perspective is based on a consensus model (between the shorter term individualist perspective and the longer term egalitarian perspective (see Goedkoop et al., 2009) and can be considered the default approach (PRé Sustainability, 2018).

## 2.6. Scenarios

Three scenario sets for general lighting service household LED lamps were considered in this LCA.

### 2.6.1. Static scenario

The approach in this scenario set considers shorter and longer lifetimes as a sensitivity analysis. It is the same approach used in previous LED lamp LCA studies (Casamayor et al., 2017; Tähkämö et al., 2013), in which the lifetime is varied with the assumption that the product is replaced by an identical product to fulfil the functional unit (i.e., 2 products are needed to fulfil the functional unit if the lifetime is changed to 12500 h and 5 products if 5000 h). All other variables are held constant (i.e. the inventories in Table 1 and the end-of-life assumptions). The reference flow and electricity use for this scenario set is shown in Table 2.

### 2.6.2. Improved product scenario: EU electricity context

This scenario set used the same 2012 DOE LED lamp data as the static scenario, but assumes that the original 2012 lamp is replaced at 5000 h by an improved lamp (considering both efficacy and material design from real 2017 lamps) rather than using the lamp for its full lifetime. As such, this scenario set represents replacing a lamp on the market in 2011–2012 with a lamp on the market in 2016–2017. A reference use of 1000 h per year (approximately 3 h a day) is assumed in this scenario. Three possible replacement products are compared with LCA data for three 800–815 lumen LED lamps on the market in 2017 (Table 1) and the reference flows and electricity use are shown in Table 3. Several products for comparison demonstrate real variation between energy efficiency and material design improvements. The scenarios are considered in the context of a European average electricity mix.

**Table 4**  
Composition of electricity supply mix (EU,NO, SE) (Itten et al., 2012).

Source	European Electricity Mix <sup>a</sup>	Electricity supply mix Norway	Electricity supply mix Sweden
Renewable	5.8%	0.9%	6.8%
Hydro	17.6%	96.2%	43.3%
Nuclear	26.6%	0%	38.1%
Fossil Fuels	49.6%	0.4%	2.3%
Waste	1.2%	0.1%	1.3%
Imported	0.1%	2.4%	8.1%

<sup>a</sup> Ecoinvent uses the aggregation of country electricity mixes within the specified region (Europe- RER) in 2012.

**Table 5**  
Overview of scenarios in this study.

	Product assumptions	Electricity mix assumptions
Static Scenario	2012 product replaced at 12500 h or 5000 h by identical product	EU electricity mix
Improved product scenario: EU electricity mix	2012 product replaced at 5000 h by 1 of 3 potential 2017 products	EU electricity mix
Improved product scenario: decarbonised electricity mix	2012 product replaced at 5000 h by 1 of 3 potential 2017 products	Norway and Sweden electricity mixes

**2.6.3. Improved product scenario: decarbonised electricity mix**

While the improved product scenario set considered an average EU electricity mix for the use stage, this scenario set considers the same products and replacement assumptions but in the context of decarbonised electricity. The Norwegian electricity supply mix average and the Swedish electricity supply mix average are examined (low voltage for households). The compositions of these mixes in Ecoinvent are shown in Table 4.

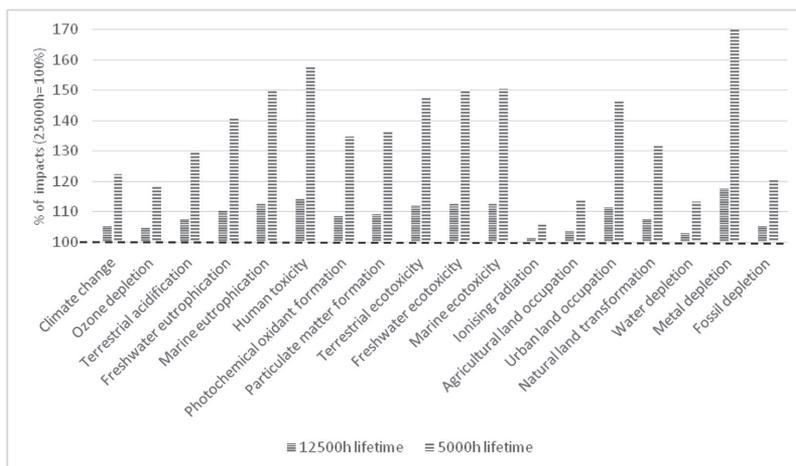
Prior LCAs conducted by Tähkämö et al. found that considering the LED product lifetime in the context of a Norwegian electricity mix, with very low fossil fuel sources, can have significantly lower overall environmental impacts (Tähkämö, 2013; Tähkämö et al., 2013; Tähkämö et al., 2014). Moreover, these previous LCAs found that in the Norwegian mix context, the manufacturing stage was responsible for most of the overall environmental impacts, rather than use stage (which normally dominates). This would then imply that durability would be desirable in this context, even with improved energy efficiency of the replacement products. However,

the Norwegian mix can also be considered a very special case, with its high share of hydroelectricity, so the Swedish context is also considered, which has a higher share of nuclear and non-hydro renewable electricity. While still an extreme case, consideration of both contexts can shed light on the influence of lifetimes and improving lamp technologies in a decarbonised context versus the average EU context.

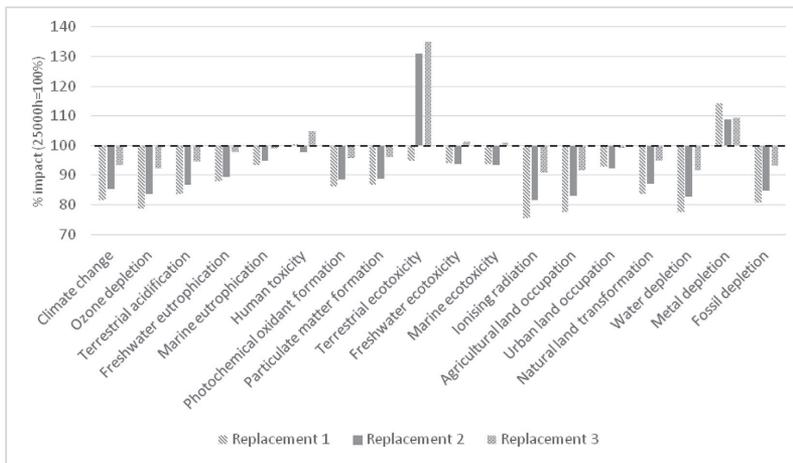
**3. Results**

**3.1. Static scenario**

Based on the scenario detailed in Table 2, Fig. 2 shows the results of comparative impacts of the DOE 2012 LED lamp (y axis – 100%) with various lifetimes, assuming that the replacement technologies for the shorter lifetime products are identical. It is clear that the longer life product has lower environmental impacts in all impact categories than a product with a fifth and half the lifetime. The



**Fig. 2.** Comparison of environmental impacts of identical 2012 LED lamps, varying the lifetime (12500 h, 5000 h) compared to the 25000 h base case LED lamp (100% on y axis – dotted line).



**Fig. 3.** Comparison of environmental impacts of 3 replacement options (original lamp replaced after 5000 h of use) relative to no replacement (i.e. base case – dotted line) in the context of EU average electricity mix. Replacement 1 represented the most efficient replacement and replacement 3 the least efficient. Replacement 1 has a lifetime of 10950 h, while all other product lifetimes are 25000 h. More specific details of products are found in Table 1.

largest differences are seen in material resource depletion and toxicity tied to the resources and processing needed for manufacturing the additional products to satisfy the 20.3 Mlmh functional unit. The fact that in many energy-related impact categories the difference is minimal also indicates the dominance of the use stage (and associated energy consumption) in driving impacts throughout the LED life cycle in this scenario.

### 3.2. Improved product scenario: EU electricity context

The picture changes when comparing longer and shorter lifetimes considering improvements in energy efficiency and material design of the replacement lamps (from the scenario in Table 3). In Fig. 2 it can be seen that the no replacement (i.e. longer lifetime) scenario has greater relative impacts in energy-related categories compared to the replacement scenarios. This makes sense given that the improved efficiency of the replacement products in the shorter life scenario result in decreased energy consumption in the use phase, but also requires manufacture and disposal of an additional product, which incur increased environmental impacts. There are also material improvements in the replacements that result in lower impacts for the shorter lifetime scenarios even in many of the toxicity categories. Compared to all of the replacement scenarios, the no replacement scenario only has relative benefits in the category of metal depletion; however, the no replacement scenario has less impact than at least one replacement lamp for each of the toxicity impacts, underscoring the importance of the assumptions about the replacements in scenarios. The most important assumption was the energy efficiency of the replacement lamp (which drives the lower impacts of replacement lamp 1 in many of the impact categories).

### 3.3. Improved product scenario: decarbonised electricity context

The results confirm that assumptions about the electricity mix can change the results of the comparison, as considered by this scenario outlined in Table 4. Fig. 4 compares the no replacement

scenario with the replacement products in context of the decarbonised Norwegian electricity mix. The results here do not indicate the same trade-offs as in the context of the EU energy mix, with the longer lifetime LED lamp having relatively less impacts compared to the more efficient replacement scenario in the majority of the environmental impact categories.

In contrast to the Norwegian mix, the use of nuclear and some fossil fuel sources of electricity in the Swedish mix (Fig. 5) results in larger magnitude in the trade-offs, as well as additional trade-offs between impact categories, compared to the Norwegian mix (Fig. 4).

### 3.4. Sensitivity analysis

Aside from the static scenario set, all the scenarios involving improving the material/energy efficiency of replacement lamps resulted in some trade-offs between impacts. It was tested whether similar trade-offs were observed if using another LCIA method. The impacts were characterised with the ILCD recommendations for LCIA in the European context. (ILCD method, see Wolf et al., 2012). While Fig. 6 indicates that using different characterisation methods can result in slightly different results (c.f. Owsianiak et al., 2014), the trade-offs remain similar to those with the ReCiPe method illustrated in Fig. 3.

Another possible method of further interpreting impacts and possible trade-offs is to use normalisation to identify the magnitude of the impacts *relative to reference information*; for example the impacts in each category relative to the per capita impacts globally in 2010. Fig. 7 shows the comparison of the no replacement and replacement scenarios in the context of the EU and Swedish electricity mix, this time with the ReCiPe global per capita normalisation applied.<sup>3</sup>

<sup>3</sup> Global normalisation is applied because the processes within the system boundary extend beyond the European context (see Pizzol et al., 2017). ILCD normalisation was also applied, with similar results that can be viewed in the supplementary materials dataset.

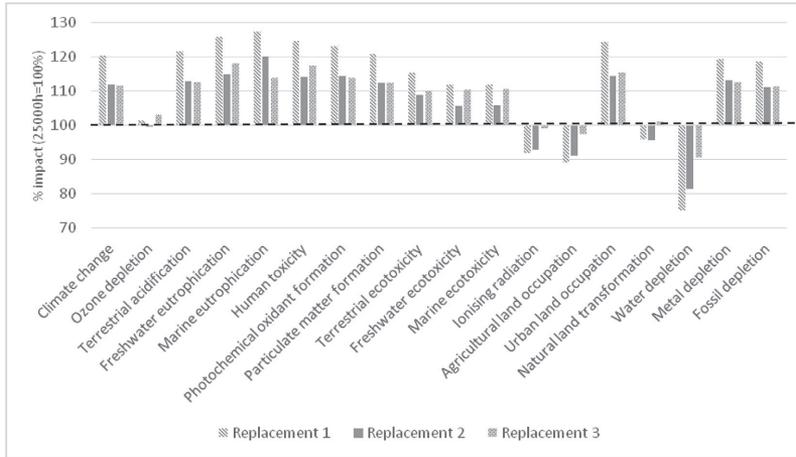


Fig. 4. Comparison of environmental impacts of 3 replacement options (original lamp replaced after 5000 h of use) relative to no replacement (i.e. base case – dotted line) in the context of Norwegian average electricity mix.

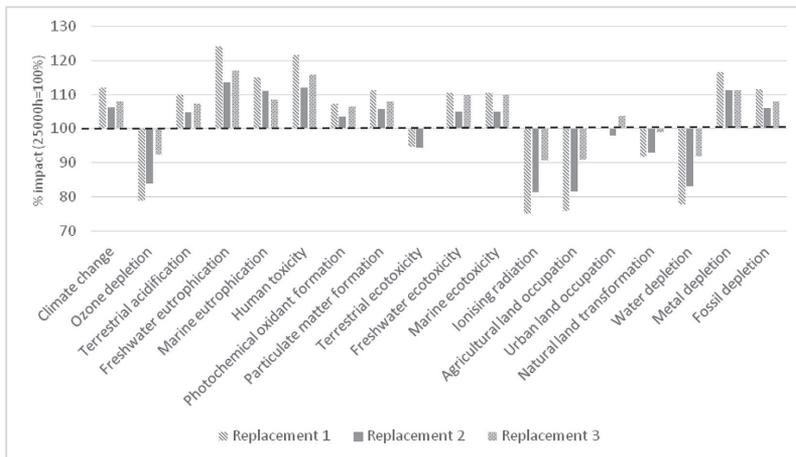


Fig. 5. Comparison of environmental impacts of 3 replacement options (original lamp replaced after 5000 h of use) relative to no replacement (i.e. base case – dotted line) in the context of Swedish average electricity mix.

It can be seen that normalisation identifies the impacts with the largest magnitude as the human and multiple ecotoxicity categories as well as the freshwater eutrophication impact categories. Within the impact categories with the highest magnitude, the no replacement scenario generally has the highest impacts in the EU average electricity context (except compared to replacement 3) while the no replacement scenario has the lowest impacts in these

categories in the Swedish electricity context.

Normalising impacts into common units, i.e. Points in ILCD,<sup>4</sup> and aggregating results can produce a single score for the environmental impact. Fig. 8 shows the single aggregated scores for the different scenarios in this study. The method also highlights that the differences between scenarios within a given context are small, further underlining the importance of the choice of electricity source during use.

### 3.4.1. Lumen depreciation

LED lamps are distinctive from other light sources when considering lifetime. Lifetime includes not just failure to produce

<sup>4</sup> The normalisation factors for ILCD that support the conversion to Points can be found on the JRC website ([http://eplca.jrc.ec.europa.eu/uploads/Table\\_ILCD\\_NFs\\_08-03-2016.xlsx](http://eplca.jrc.ec.europa.eu/uploads/Table_ILCD_NFs_08-03-2016.xlsx)) and in the underlying studies (see Sala et al., 2015).

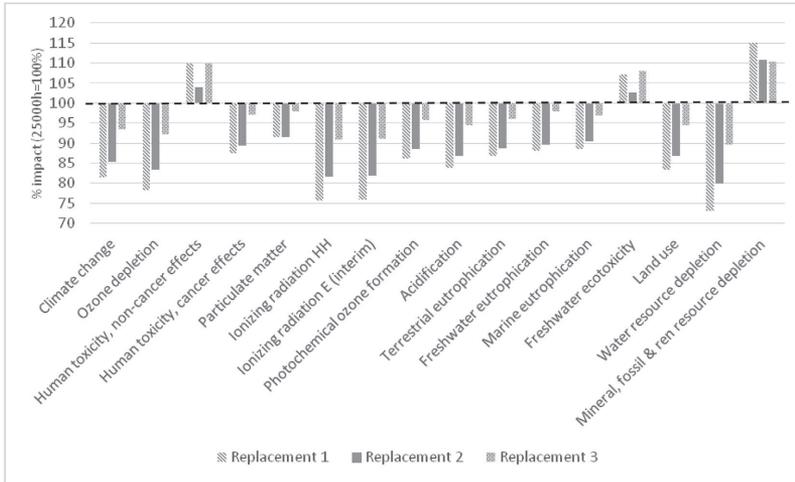


Fig. 6. Comparison of environmental impacts of 3 replacement options (original lamp replaced after 5000 h of use) relative to no replacement (i.e. base case – dotted line) in the context of EU electricity mix using ILCD.

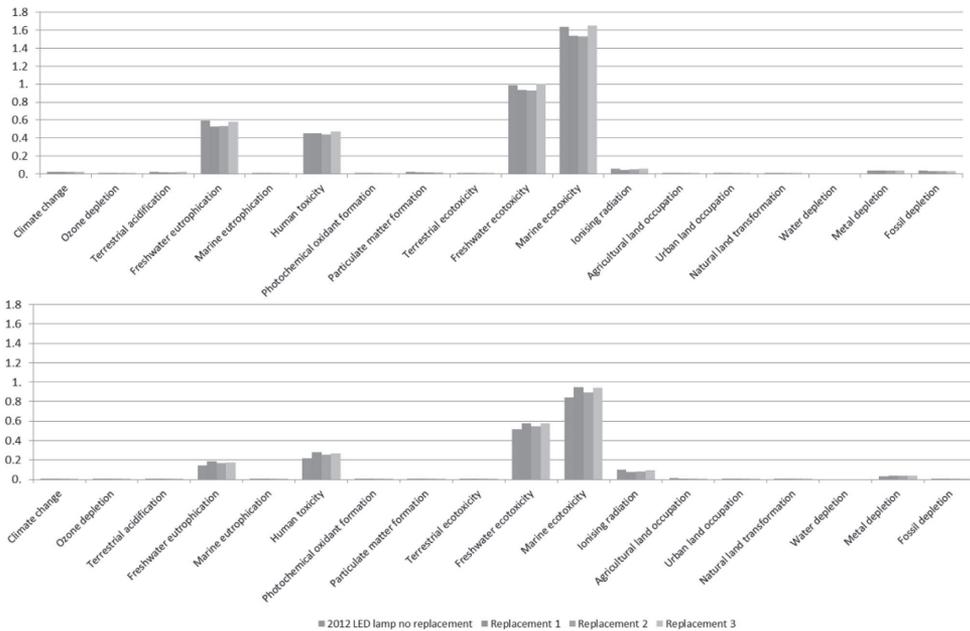


Fig. 7. Normalised environmental impacts of no replacement (i.e. base case – blue column) compared to 3 replacement options (original lamp replaced after 5000 h of use) in the context of EU (top) and Swedish electricity mix (bottom) using the ReCiPe midpoint hierarchist, normalised method (i.e. divided by the average impact in that category globally per capita in 2010, so a score of 1 is average impact in that category). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

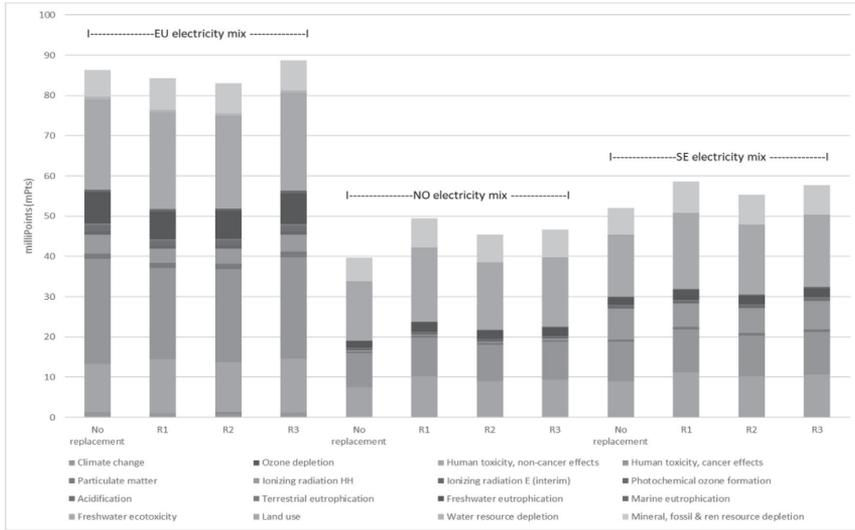


Fig. 8. Comparison of relative environmental impacts of the no replacement baseline scenario compared to 3 replacement options at 5000 h, in the context of an EU, NO, and SE electricity mix using ILCD single score method (normalising midpoint impacts by 2010 EU 27 normalisation factors to convert to Points and aggregating results).

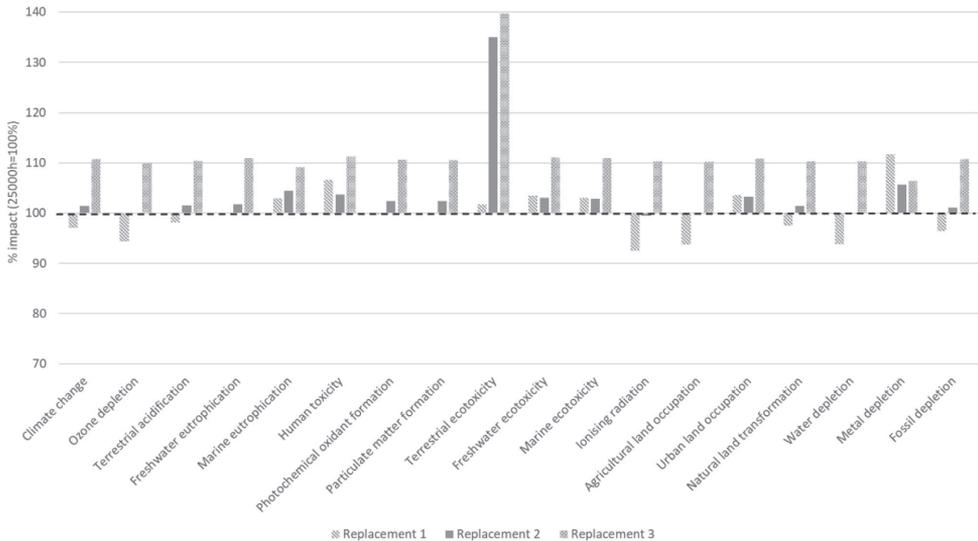
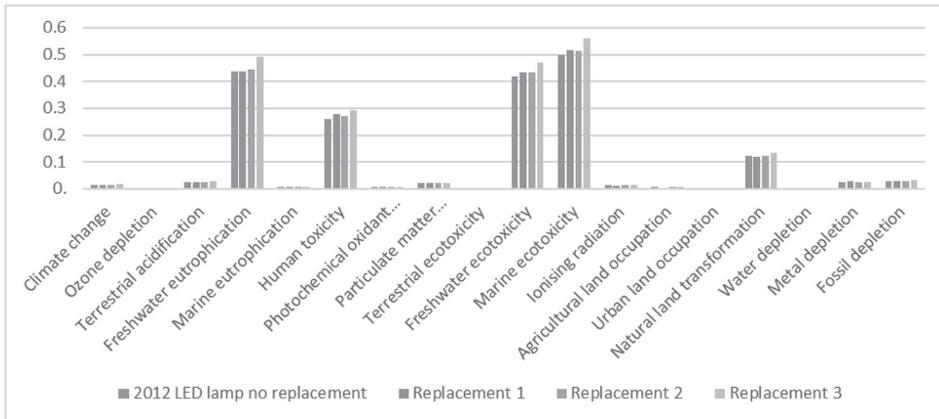


Fig. 9. Comparison of environmental impacts of 3 replacement options (original lamp replaced after 5000 h of use) relative to no replacement (i.e. base case – dotted line) in the context of EU average electricity mix and considering 30% lumen depreciation throughout the lamp lifetime (ReCiPe midpoint).



**Fig. 10.** Normalised environmental impacts of no replacement (i.e. base case – blue column) compared to 3 replacement options (original lamp replaced after 5000 h of use) in the context of EU electricity mix and considering 30% lumen depreciation throughout the lamp lifetime (using the ReCiPe midpoint hierarchist, normalised method). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

light, also deterioration of light (i.e. lumen maintenance/depreciation) and quality. Casamayor, Su & Sarshar (2017) conducted a sensitivity analysis of lumen depreciation in which the lifetimes of the luminaires considered were varied (similar to the static scenario presented in section 3.1). As the LED product lifetimes are the focus of this analysis, this kind of sensitivity analysis for lumen depreciation as it relates to lifetime has already been demonstrated.

However, lumen depreciation also affects the functional unit if considering decreasing lumen output, requiring additional product input to meet the 20.3 lmh functional unit. While there is very little specific data on lumen depreciation rates as lamps are only tested for a portion of their expected lifetime (commonly 6000 h is standard), there are methods for estimating the lumen maintenance based on the LM-80 and TM-21 standards, both of which are used for extrapolating the rate of lumen depreciation and determining the lifetime (i.e. when 70% of the original lumen output is reached) (see Royer, 2014). It can be assumed then that the LED lamps lose 30% of their lumen output during their lifetimes.

The reference flows in the improving product EU scenario can be adjusted to account for this, as shown in Fig. 9. The figure further illustrates the difference this assumption makes in comparison to the original analysis of this scenario (Fig. 3). While replacement product 1 still has lower impacts in many of the impact categories, the difference has been minimised due to the need for slightly more products to fulfil the same functional unit (see Fig. 10). Fig. 10 shows the normalised results for this scenario set, showing the normalised impacts of no replacement can be less than replacement 1 for the impact categories with the highest normalised impacts (i.e. toxicity and eutrophication categories).

## 4. Discussion

### 4.1. Retrospective scenario approach

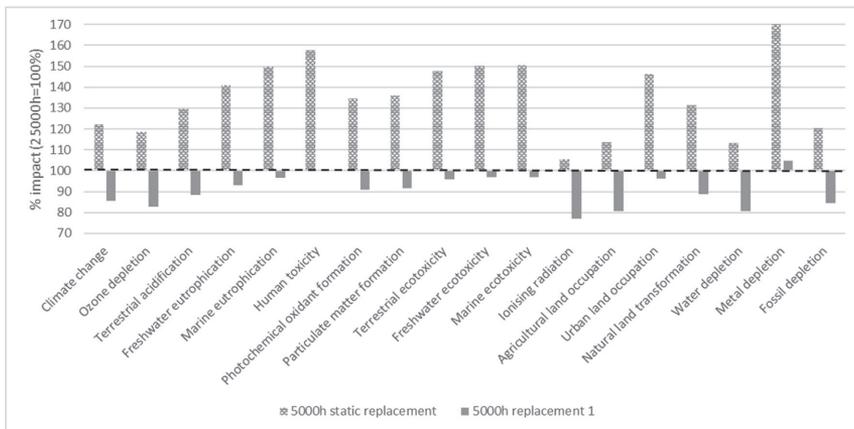
Using additional scenarios in considering product improvements for LED lamps revealed significant differences compared to the standard LCA sensitivity approach varying only the lifetimes. Fig. 9 shows the difference between considering replacement at 5000 h with the same 2012 product versus a more energy efficient

replacement. The findings from the improving product EU scenario set (section 3.2) resulted in the opposite finding compared to the static sensitivity approach (section 3.1). Fig. 11 demonstrates the difference between considering a static replacement at 5000 h (approach of previous LED LCAs) and replacement 1 at 5000 h, which accounts for real LED product development. This underscores the need to consider appropriate dynamic factors such as improving technologies when considering lifetimes in LCAs.

Past studies have attempted to arrive at absolute numbers for optimal lifetime (Bakker et al., 2014; Kim et al., 2006) based on general assumptions about product characteristics (i.e., average efficiencies of products and materials used). The retrospective case approach in this study highlighted how influential assumptions can be, as there were different outcomes for the 3 possible replacement products considered. The consumer would not necessarily choose the most energy efficient option in reality, in which case the more modest product improvements represented by replacement products 2 and 3 were not preferable to keeping the original product its full lifetime.

However, this study also found that even when the specific product characteristics are known, acquiring specific data for upstream processes can be challenging, and this challenge has been highlighted more generally for LED inventories (Franz and Wenzl, 2017). Using specific lamp data for the comparisons can also limit the generalisability of the results. As could be seen from the case of the three different 2017 products, different conclusions could be drawn depending on what products were considered. The same could be true for the 2012 product, of which there was only data for one product. That particular product had been chosen for its high efficacy (65 lm/W) compared to other LED lamps in 2012 (Scholand and Dillon, 2012), so can be considered a better than average lamp for the market at that time.

The analysis also showed sensitivity to the functional unit, for instance, by including lumen depreciation in the calculation of reference flows. While lumen-hours is a common functional unit for lamps, in reality the household user (as opposed to a professional or special use application) is unlikely to notice lumen depreciation and account for it (Next Generation Lighting Industry Alliance, 2014). This might also mean that in this application, simply hours may be a suitable functional unit.



**Fig. 11.** Comparison of relative environmental impacts of the no replacement baseline scenario (dotted line) with the static replacement (i.e. no product development considered) and more energy efficient replacement product 1.

While rated lifetimes were used in this analysis, it is known that LED lamps can also fail earlier than the rated lifetime (Narendran et al., 2016). Even if products improve in terms of energy efficiency, if a replacement fails prematurely, the replacement lamp is likely to be similar in terms of energy efficiency and other product characteristics (as less time will have elapsed between installation and replacement). Premature failure would then lead to a more static scenario (or even replacement product 2 or 3) in which replacement products do not offer enough benefits in comparison to durable products. Minimum quality and lifetime requirements then are still very relevant in the policy context to ensure products are not replaced too rapidly.

The research demonstrated that normalisation and calculation of single aggregated scores can be useful for clearly identifying the scenarios with the lowest impact. However, the usefulness of normalisation for decision-making in comparative LCAs has been questioned, as it is observed that toxicity-related impacts tend to be emphasised by normalisation methods regardless of LCIA method used (see Prado et al., 2017). The same emphasis is evident in the ReCiPe and ILCD normalisations seen in Figs. 7 and 8 of this study. Moreover, some impacts such as water depletion could not be normalised with the available data. So while normalisation appears useful for putting trade-offs into perspective, there are caveats to using it as the basis for decision-making (Prado et al., 2017).

#### 4.2. Are longer lifetimes better?

The findings of this research have confirmed the complexity of considering longer product lifetimes for improving products but indicate the factors and contexts under which longer lifetimes are preferable. While previous LCA of LED products has shown favourable results for long lifetimes (Casamayor et al., 2017; Tähkämö, 2013; Tähkämö and Dillon, 2017), the comparison was made as a sensitivity analysis considering identical product assumptions. In reality shorter lifetimes also mean that consumers can replace products with more energy-efficient and improved products can make the benefits of durability less straightforward.

Previous scenario-based research on durability for other product categories reached conclusions that more durable options were favourable in many types of many energy-using products, but not

for all impact categories if there were substantial energy efficiency improvements (Ardenne and Mathieux, 2014; Bobba et al., 2016; Irlando et al., 2017). The case of LED lamps in this study indicated that shorter life products and faster replacement cycles appear to be beneficial in terms of energy-related environmental impacts if replaced with improved products in the context of fossil fuel based energy mixes. However, shorter lifetimes resulted in higher impacts in metal depletion and toxicity categories (depending on what replacement product is considered).

The results indicate that promoting durability in the context of improving products and an electricity mix with fossil fuels is likely to result in trade-offs between energy and material/toxicity-related environmental impacts. It is important to consider a broad range of impacts in order to fully assess these trade-offs. It should also be considered that LCA does not capture all impacts or issues which may be important in assessing these trade-offs (i.e. criticality of materials – see Klinglmaier et al., 2014). The presence of trade-offs in an LCA-only approach also highlights the need to consider multiple tools and strategies for decision-making (Berlin and Iribarren, 2018).

Assumptions about what product is used as a replacement also matters to the results of the LCA. The case of LED lamps demonstrated that in addition to efficiency, material design, such as decreased use of aluminium for heat sinks, lower weight of metals and other materials, or smaller electronics, can also influence trade-offs, particularly for toxicity-related impact categories. Despite the availability of improved products on the market in 2017, the 65 lm/W efficiency of the 2012 product modelled in this case was a common efficacy for lamps in the low price sector in Europe (see Franz and Wenzl, 2017), indicating that the static scenario can also be a reality for many consumers. The better policy in this context might be to influence product choice towards improved products through eco-labelling and more ambitious minimum energy performance standards.

This research also confirmed the importance of electricity mix for environmental impacts. While earlier LCA research on LED lamps by (Tähkämö, 2013) found that the assumption of a Norwegian energy mix resulted in the relative impact of manufacturing phase to increase compared to the use phase, this study further illustrated that energy-related impacts are less significant overall

(for example the climate impacts in the improving product scenario were 168.2 kg CO<sub>2</sub> eq. for the 2012 product and 137.3 kg CO<sub>2</sub> eq. for the more energy-efficient replacement 1, while the climate impacts were 23.5 and 29.5 kg CO<sub>2</sub> eq., respectively, in the Norwegian context - see absolute impact figures in Appendix). This, in turn, minimises the trade-offs between environmental impacts in the case of improving product efficiencies. It is important that developments leading towards decarbonisation of the electricity mix are considered in determining the overall impact of longer product lifetimes as it was shown to both minimise the overall impacts of the LED lamps and minimise the trade-offs. This is relevant for policies considered on the member state level and in considering future product policies and their interaction with EU climate and energy policies promoting decarbonisation.

In considering product durability policies for lamps and other improving products, it is important to also look forward at projections of how the products will continue to develop. The context of this study was a period of rapid LED lamp development between 2012 and 2017. This development has even continued, as there are now LED lamps more than twice as efficient, using less materials (Philips Lighting, 2018), though many (but not all) have significantly shorter lifetimes than previous projections (Franz and Wenzl, 2017). Such lamps begin to approach the projected limits for efficiency improvement for LEDs (Navigant, 2016; U.S. Department of Energy, 2016). Moving towards the limits for efficiency developments means that replacement lamps will not present significant efficiency improvements, implying that as LED lamp technology matures the scenarios will increasingly resemble the static scenario set, in which early replacements or shorter lifetimes do not offer advantages from an environmental perspective (assuming no technology replaces LEDs before they mature).

## 5. Conclusion

This research has demonstrated some of the important factors to consider in whether longer lifetimes for products with improving technology are beneficial from an overall environmental perspective. The scenario-based approach indicated that considering improved efficiency, improved material design and decarbonisation of electricity supply can all influence whether longer lifetimes have lower environmental impacts for LED products. Policies to promote longer life for such products may only be appropriate in

contexts with relatively decarbonised electricity supply, where trade-offs can be clearly weighed and valued, or for mature product categories where further substantial energy efficiency improvements are unlikely. The retrospective modelling approach presented in this paper identified that there are key factors beyond energy efficiency alone that should be considered in answering questions about optimal product lifetimes and that it is important to recognise trade-offs between different environmental impacts and when these are minimised as EU policy seeks to transition to both a circular and low carbon economy.

## Author contributions

J.L.R. conceived the idea for this research, and developed the research design in collaboration with L.T. and C.D.; J.L.R. modelled the LCA in SimaPro, conducted the analysis and made all graphs and figures, with comments provided by L.T.; J.L.R. wrote the article with review and comments provided by C.D. and L.T.

## Conflicts of interest

The authors declare no conflict of interest.

## Acknowledgments

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## Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.jclepro.2019.03.331>.

## Appendix

**Table A-1**  
Lifecycle inventory for LED lamp products considered (based on Scholand and Dillon, 2012; Dillon et al., 2019).

Material	Unit	2012 LED lamp	2017 LED lamp 1	2017 LED lamp 2	2017 LED lamp 3	Ecoinvent process (market for   Alloc Def, U)
LEDs units	p	12	11	20	8	LED unit (based on Scholand and Dillon, 2012).
Remote Phosphor <sup>a</sup>	g	1	0	0	0	Rare earth concentrate, 70% REO, from bastnasite (GLO)
Plastic Phosphor host	g	11	0	0	0	Polycarbonate (GLO)
Aluminium	g	68.20	11.03	20.69	0	Aluminium, cast alloy (GLO)
Copper	g	5	0	0	0	Copper (GLO)
Nickel	g	0.003	0	0	0	Nickel, 99.5% (GLO)
Brass	g	1.650	0	0	0	Brass (RoW)
Cast iron	g	4	0	0	0	Cast iron (GLO)
Chromium	g	0.0002	0	0	0	Steel, chromium steel 18/8 (GLO)   market for   Alloc Def, U
Galvanised Steel	g	0	1.919	2.190	1.904	Zinc concentrate/Steel, low-alloyed (GLO)
Silicon	g	0	1.322	0	0	Silicon, electronics grade (GLO)
Light Plastic	g	0	12.49	25.15	25.27	Polymethyl methacrylate, sheet (GLO)   market for   Alloc Def, U
Heavy Plastic	g	0	6.772	10.56	2.277	Polycarbonate (GLO)
LED board	g	0	1.734	4.665	6.320	Printed wiring board, surface mounted, unspecified, Pb free (GLO)
Printed board	g	15	3.466	1.617	1.927	Printed wiring board, surface mounted, unspecified, Pb free (GLO)
Inductor	g	4.8	0.668	0.804	0.913	Copper concentrate (GLO)
IC Chip	g	0.158	0	0.079	0	Integrated circuit, logic type (GLO)
Capacitor SMD	g	0.377	0.023	0.050	0.115	Capacitor, for surface-mounting (GLO)
Electrolytic Capacitor	g	24.73	1.747	5.637	4.920	Capacitor, electrolyte type, < 2 cm height (GLO)
Diode	g	1.091	0.139	0.181	0.222	Diode, glass-, for surface-mounting
Resistor SMD	g	0.993	0.104	0.136	0.253	Resistor, surface-mounted (GLO)

Table A-1 (continued)

Material	Unit	2012 LED lamp	2017 LED lamp 1	2017 LED lamp 2	2017 LED lamp 3	Ecoinvent process (market for   Alloc Def, U/)
Resistor	g	0.993	0.104	0.136	0.253	Resistor, wirewound, through-hole mounting (GLO)
Transistor	g	1.387	0.085	0.608	0	Transistor, wired, big size, through-hole mounting (GLO)
Transformer	g	30.15	4.956	7.384	2.667	Transformer, low voltage use (GLO)
Resin Glue	g	4.5	0	0	0	Epoxy resin, liquid (GLO)
Solder paste	g	0.3	0.3	0.3	0.3	Flux, for wave soldering (GLO)
<b>Product Mass</b>	<b>g</b>	<b>176.0</b>	<b>45.66</b>	<b>81.96</b>	<b>47.09</b>	
Paper packaging	g	37	17.99	18.91	19.80	Corrugated board box (GLO)
Plastic packaging	g	0	0	0	0	Polymethyl methacrylate, sheet (GLO)
<b>Total Mass</b>	<b>g</b>	<b>213.00</b>	<b>63.65</b>	<b>100.9</b>	<b>66.88</b>	

<sup>a</sup> Phosphor for 2017 units is modelled as part of the LED unit.

Table A-2. Comparison of environmental impacts of improving product scenario (2012 LED lamp 25000 h versus 3 replacements at 5000 h in context of EU average electricity mix)

Impact category	Unit	2012 lamp	Replacement 1	Replacement 2	Replacement 3
Climate change	kg CO2 eq	168.2036	137.3253	143.5552	157.4445
Ozone depletion	kg CFC-11 eq	2E-05	1.57E-05	1.67E-05	1.84E-05
Terrestrial acidification	kg SO2 eq	0.794248	0.665028	0.68884	0.751399
Freshwater eutrophication	kg P eq	0.172309	0.151666	0.154243	0.168752
Marine eutrophication	kg N eq	0.061574	0.057489	0.05836	0.060963
Human toxicity	kg 1,4-DB eq	147.4093	147.9493	143.8671	154.5649
Photochemical oxidant formation	kg NMVOC	0.391296	0.337471	0.346231	0.375215
Particulate matter formation	kg PM10 eq	0.276833	0.240556	0.245906	0.266147
Terrestrial ecotoxicity	kg 1,4-DB eq	0.013443	0.012412	0.019883	0.020732
Freshwater ecotoxicity	kg 1,4-DB eq	4.26944	4.016406	3.99953	4.320185
Marine ecotoxicity	kg 1,4-DB eq	4.029693	3.779661	3.768118	4.073004
Ionising radiation	kBq U235 eq	78.19658	59.15445	63.78136	71.10953
Agricultural land occupation	m2a	22.07145	17.11072	18.30404	20.26817
Urban land occupation	m2a	1.252462	1.162318	1.156899	1.244858
Natural land transformation	m2	0.018849	0.015756	0.0164	0.017879
Water depletion	m3	2.431133	1.885543	2.015163	2.232203
Metal depletion	kg Fe eq	14.83869	17.29755	16.33812	16.45904
Fossil depletion	kg oil eq	44.51786	35.935	37.75395	41.50693

Table A-3

Comparison of environmental impacts of decarbonised product scenario set (2012 LED lamp 25000 h versus 3 replacements at 5000 h in context of NO average electricity mix)

Impact category	Unit	2012 lamp	Replacement 1	Replacement 2	Replacement 3
Climate change	kg CO2 eq	23.4966	29.5117	26.9876	26.98504
Ozone depletion	kg CFC-11 eq	3.15E-06	3.2E-06	3.15E-06	3.26E-06
Terrestrial acidification	kg SO2 eq	0.137775	0.175924	0.160644	0.160182
Freshwater eutrophication	kg P eq	0.038276	0.051806	0.046062	0.047704
Marine eutrophication	kg N eq	0.018324	0.025266	0.023419	0.02187
Human toxicity	kg 1,4-DB eq	65.13799	86.6533	77.48768	80.28738
Photochemical oxidant formation	kg NMVOC	0.082491	0.107397	0.098118	0.097458
Particulate matter formation	kg PM10 eq	0.066037	0.083503	0.076473	0.076478
Terrestrial ecotoxicity	kg 1,4-DB eq	0.005489	0.006486	0.006069	0.006154
Freshwater ecotoxicity	kg 1,4-DB eq	2.145389	2.433888	2.284923	2.40167
Marine ecotoxicity	kg 1,4-DB eq	1.986474	2.257366	2.119854	2.228591
Ionising radiation	kBq U235 eq	5.147194	4.729208	4.784018	5.099449
Agricultural land occupation	m2a	4.558123	4.062463	4.155017	4.437871
Urban land occupation	m2a	0.395683	0.523977	0.4711	0.476809
Natural land transformation	m2	0.007978	0.007657	0.007637	0.008073
Water depletion	m3	9.90344	7.452764	8.055053	8.989434
Metal depletion	kg Fe eq	12.60326	15.63205	14.67001	14.57633
Fossil depletion	kg oil eq	5.681647	7.000191	6.463242	6.487943

**Table A-4**

Comparison of environmental impacts of decarbonised product scenario set (2012 LED lamp 25000 h versus 3 replacements at 5000 h in context of SE average electricity mix)

Impact category	Unit	2012 lamp	Replacement 1	Replacement 2	Replacement 3
Climate change	kg CO <sub>2</sub> eq	30.4599	34.69969	32.61394	33.27987
Ozone depletion	kg CFC-11 eq	2E-05	1.58E-05	1.68E-05	1.85E-05
Terrestrial acidification	kg SO <sub>2</sub> eq	0.20019	0.222427	0.211076	0.216605
Freshwater eutrophication	kg P eq	0.040665	0.053585	0.047992	0.049863
Marine eutrophication	kg N eq	0.026766	0.031555	0.03024	0.029501
Human toxicity	kg 1,4-DB eq	71.59839	91.46661	82.70769	86.12759
Photochemical oxidant formation	kg NMVOC	0.137307	0.148238	0.14241	0.147012
Particulate matter formation	kg PM10 eq	0.089488	0.100975	0.095421	0.097678
Terrestrial ecotoxicity	kg 1,4-DB eq	0.011906	0.011267	0.011254	0.011956
Freshwater ecotoxicity	kg 1,4-DB eq	2.227805	2.495291	2.351515	2.476173
Marine ecotoxicity	kg 1,4-DB eq	2.082555	2.328952	2.197488	2.315449
Ionising radiation	kBq U235 eq	131.2554	98.68578	106.6795	119.1013
Agricultural land occupation	m <sup>2</sup> a	53.34824	40.4134	43.57743	48.54414
Urban land occupation	m <sup>2</sup> a	0.887749	0.89059	0.86869	0.921636
Natural land transformation	m <sup>2</sup>	0.0099	0.009089	0.00919	0.00981
Water depletion	m <sup>3</sup>	2.267833	1.763876	1.885482	2.086845
Metal depletion	kg Fe eq	13.73016	16.47164	15.58055	15.59506
Fossil depletion	kg oil eq	7.189354	8.123504	7.68147	7.85091

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What really happens when we dispose of our lighting products? Critical raw materials in our lighting products have high value, but can that value be retained in Europe? LED lamps have many potential benefits over traditional lighting products, including longer lifetimes; but do long lifetimes mean we miss out on potential benefits of even newer technology? This thesis focusses on policies for lighting products, a product group that exemplifies many of the current product policy issues related to slowing and closing material loops as part of a Circular Economy. The aim of this research was to address gaps in knowledge about the performance of existing product policies and potential improvements in relation to the EU's Circular Economy objectives.