COMING FULL CIRCLE

Examining extended producer responsibility in the context of circular economy

Kieran Campbell-Johnston

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Coming full circle
Examining extended producer responsibility in the context of circular economy

De cirkel rond maken
Onderzoek naar de uitgebreide producentenverantwoordelijkheid in de context van circulariteit
(met een samenvatting in het Nederlands)

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"Unless someone like you cares a whole awful lot.
Nothing is going to get better. It's not."

Dr. Seuss, The Lorax

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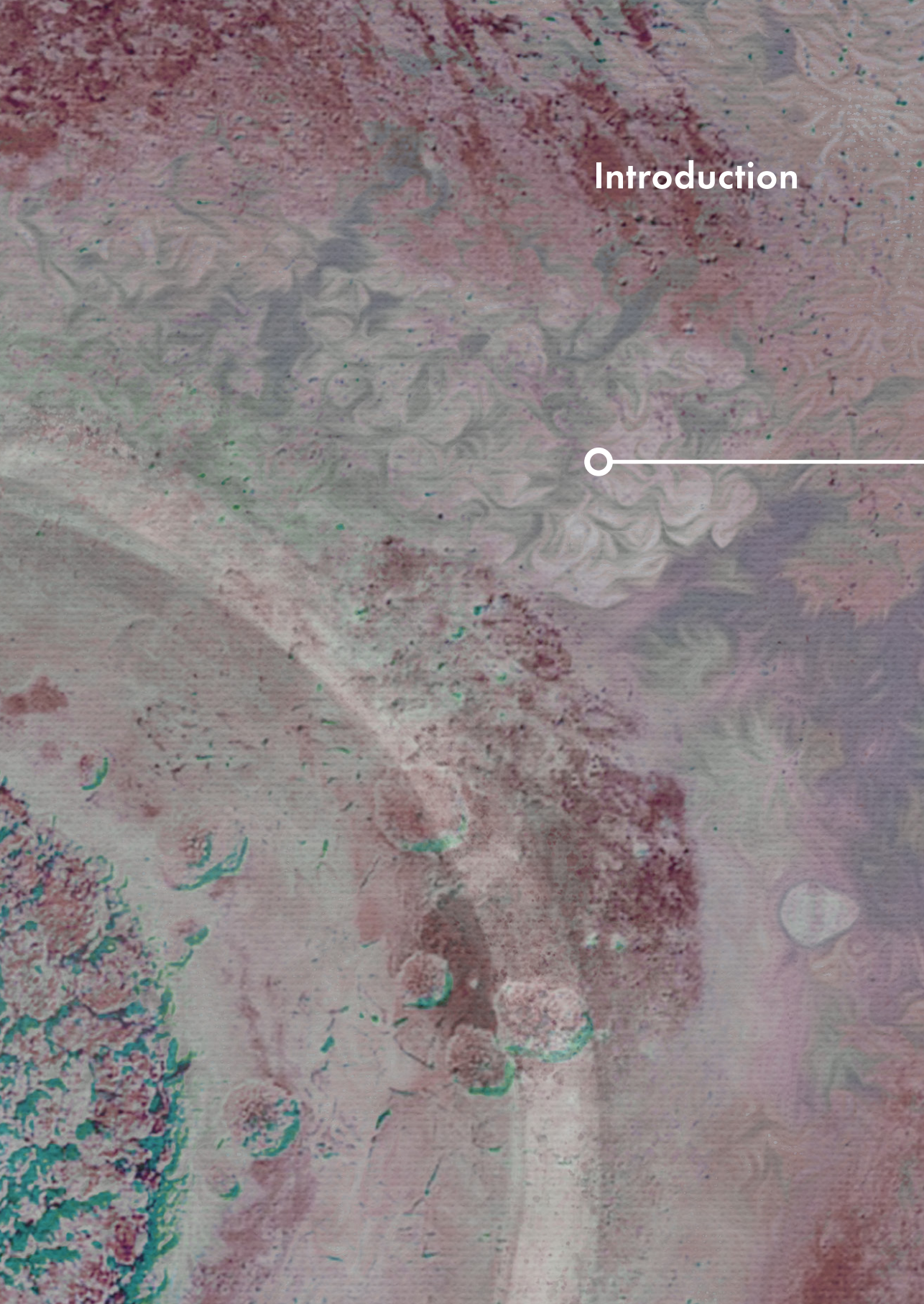

Introduction

1.1 The need for a circular economy

When ordering something online or buying something in a store, how often do we stop and think, "where has this product come from?" And, when it is broken, lost or no longer wanted, "where will this product go and what will happen to it once I discard it?". The purchasing and use of a product is only one (small) part in its lifecycle - and an even smaller part in the lifespan of many materials. Think of your mobile phone. Its various components are mined in one place, refined, designed, produced and assembled in many others. It spends two years in your possession on average, and then it is gone, replaced by another. This is one product of the billions that have come and are yet to come. Why then, and how should we think about what has come before and should come after?

The current debate on circular economy (CE) and sustainable development are driven by the realisation that the world in the 21st century is facing a plethora of interconnected, conflicting and wicked problems. The drive for industrialisation and development in the past 200 years have brought unprecedented wealth (for some) and inequality (for most), yet, at the expense of increased environmental and planetary degradation and destruction (Rockström et al., 2009; Steffen et al., 2018). The scale of humanity's influence and its current impact on the global ecological processes was plainly illustrated with the publication of the planetary boundaries framework (Rockström et al., 2009). This framework outlined the "safe operating space for humanity with respect to the Earth system and are associated with the planet's biophysical subsystems and processes". These constituted nine Earth-system processes and proposed thresholds, which, if crossed, could result in large and catastrophic environmental change (Rockström et al., 2009, p. 472). In a revision, Persson et al. (2022) indicated that of the nine boundaries proposed, five have already been crossed[1] (Climate change, Nitrogen cycle, Land system change, Novel entities[2] and Biodiversity loss), indicating the acute and immediate nature of these challenges and the urgent need of humanity to respond.

As Jackson (2009) and Steffen et al. (2007) have argued, the driving force for this change is rooted in the current practices of production and consumption that have grown exponentially since the end of the Second World War. In this period, the human-included impact on global ecosystems increased more than any other time in history. Scientists now refer to this period of growth and consumption of materials as "The Great Acceleration", in which the extraction of construction material grew by a factor of 34, ores and minerals by a factor of 27, fossil fuels by 12, and biomass by a factor of 3.6 since the 1950s (see Figure 1.1) (Steffen et al., 2007; UNEP and International Resource Panel, 2011). Several international bodies have warned, that, under business as usual, global resource extraction couple triple by 2050 (UNEP and International Resource Panel, 2011).

1 The original categorisation by Rockström et al., 2009 argued that three had been crossed: Climate change, Biodiversity loss and the Nitrogen cycle.

2 This refers to entities that are novel in a geological sense, but have large-scale and substantial impacts that could threaten Earth-system processes, e.g. plastic pollution Persson et al. (2022, p. 1510)

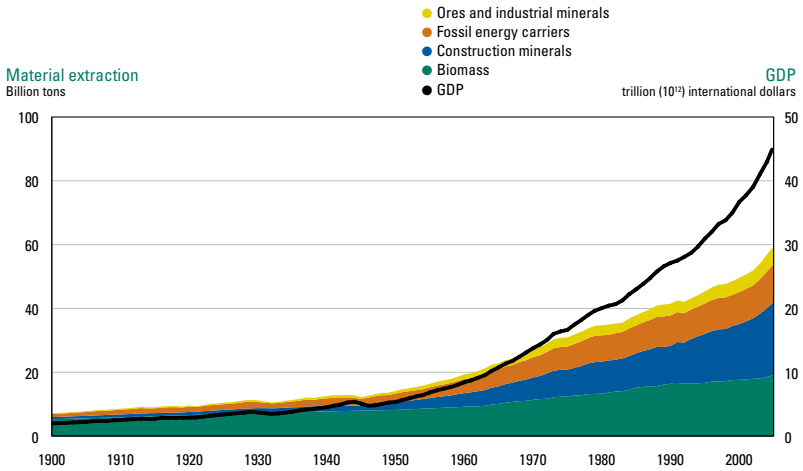

FIGURE 1.1 Global material extraction in billion tonnes, 1990 – 2005 (UNEP and International Resource Panel, 2011) .

These changes to the production and consumption of materials have raised two visible issues: overconsumption of resources and excessive waste generation. In 2018, the world generated 2100 Megatonnes (Mt) of municipal solid waste, roughly 0.74 kg of waste per capita per day. This quantity of waste roughly equates to the weight of an average grizzly bear, per person, per year going to waste. On average, 37% of all waste is disposed of in landfills, 33% is openly dumped, 19% undergoes material recovery operations such as recycling, and 11% is treated through incineration (Kaza et al., 2018). Dumping and improper waste treatment can cause serious public health and environmental issues, particularly from toxic particles from littering, burning and pollution into surface and groundwater (UNEP and ISWA, 2016).

The other side is the increased consumption of material resources. Today, around 60,000 Mt of natural resources per year are extracted, roughly the same as 41,000 Empire State Buildings. Primary resource extraction could rise to an unimaginable 140,000 Mt per year of minerals, ores and fossil and plant fuels by 2050 unless drastic action is taken (Friends of the Earth, 2009; United Nations, 2011). Why is this important? The pressure of increased resource extraction is particularly significant when looking at specific materials, such as copper, gold and antimony. Some projections indicate these materials could be completely exhausted in the next 100-150 years (Henckens et al., 2018; Henckens and Worrell, 2020; Henckens, 2021). Such pressures run contrary to notions of inter-generational equity, as included in the original definition of sustainability (Brundtland, 1987) and many international conventions (see United Nations 1992a, 1992b). These denote that the current generation is, in effect, stealing the present Earth from future ones. Therefore, future generations have a legitimate expectation of equitable access to the level of resources used today. However, the aforementioned current trends paint a picture of a world where future generations will not enjoy such material access and privileges. Such arguments have been outlined

already since the 1970s, when the Club of Rome modelled the 'Limits to Growth' due to the pressures of population growth, resource depletion, pollution and food production (Meadows et al., 1972).

Consumption and waste are inextricably tied to a country's GDP, which has generally led to calls for nations to decouple economic growth from environmental impacts and consumption (UNEP and International Resource Panel, 2011). However, as Wiedmann et al. (2015) outlined, evidence of strong decoupling and weak decoupling examples are limited. Wealthier countries in the Global North have higher material footprints than those in the Global South. This also corresponds to waste, with richer countries and regions producing far more waste than poorer ones (Kaza et al., 2018). Evidence from the last few decades shows that decoupling has not worked. We can contextualise this phenomenon within the broader sustainability discussion over *weak* and *strong* approaches. Namely, there is a tension between the assumption that economic well-being and market solutions can cover all concerns. If this assumption holds, then the relationship between the biosphere and manmade capital can be conceived through a purely economic debate (weak sustainability). If not, then (strong) sustainability implies the need to step outside of the conventional market framework to establish conditions for maintaining the biosphere and human well-being (Ayres et al., 2001).

Over the last 20 years, the concept of CE has become popularised, framed as a solution to the dual problems of excessive waste generation and resource depletion. Initially driven by organisations such as the Ellen MacArthur Foundation, CE proponents argue it is the antithesis to the current linear take-make-waste economy. Instead, a CE, so the argument goes, results in the slowing, closing and narrowing material and energy loops that retain the value of materials as long as possible. This should (theoretically) reduce resource use and waste generation (Ellen MacArthur Foundation, 2013; Geissdoerfer et al., 2017). These early movers in CE have promoted the rationale that it could maintain the current economic paradigm by decoupling the economy from resource constraints. However, CE has come to mean many things to different people, depending on the world view and interests (Bauwens et al., 2020; Calisto Friant et al., 2020).

This concept (and its assumed benefits) has captured the imagination of governments, NGOs, businesses and scholars to varying degrees. Everywhere people are talking about CE, material circularity and being circular. In particular, the European Union (EU) and China have embraced and driven two versions of the concept. In China, CE was introduced in 2002 and became formalised in the 2009 CE promotion plan. In the EU, the approach to CE was outlined in the 2015 Action plan and updated in 2020, which included updating waste management and product policy frameworks and enhancing a secondary raw materials market (European Commission 2015, 2020). China's approach is broad and encompasses pollution alongside resource and waste concerns, framed as a response to the environmental challenges caused by its rapid industrial growth. At the same time in the EU, the scope is narrower and connected to future competitiveness, environmental and

resource threats (McDowall et al., 2017). Against this backdrop, the question arises: Are the policies and practices now being promoted as 'circular' actually new, and how and in what ways are they different from what has emerged before?

1.2 Circular economy: a refurbished concept?

As a concept, the term 'circular economy' can be traced back to Pearce and Turner's work on the economics of natural resources and the environment (Pearce and Turner, 1990). Yet, the contemporary understanding of CE is muddled and contested, with scholars highlighting the multiple competing definitions (Kirchherr et al., 2017), conceptual underpinnings, origins (Blomsma and Brennan, 2017; Calisto Friant et al., 2020; Reike et al., 2018) and aims; these being economical or environmental gains or sustainable development (Korhonen et al., 2018b; Schöggl et al., 2020). This conceptual and interpretative messiness reflects the degree the concept has been embraced (and challenged see Hobson 2016). It has a plurality of understandings, resulting in a conceptually contested concept with different meanings and intentions for many societal actors (Korhonen et al., 2018b).

Nevertheless, despite this contemporary ontological plurality, the intellectual foundations of CE are both diverse and evolving. The proposition of the Earth as an entity without unlimited reserves (for pollution or extraction) was proposed by Boulding (1966) in his The Economics of the Coming Spaceship Earth. The Club of Rome report on the limits to growth, argued that without change in resource consumption would most probably lead to a sudden decline in population and industrial capacity (Meadows et al., 1972). Industrial Ecology, which examines the stocks and flows of materials and opportunities for increased efficiency (Ayres and Ayres, 1996; Graedel, 1995; Lifset and Graedel, 2002). Ecological Economics (Rees, 1992), which addresses the interdependence of human economies and natural ecosystems. The performance economy (Stahel, 2010), addresses the maintenance and exploitation of stocks. Other vital contributions to the understanding of CE come from Cradle-to-Cradle design (Braungart et al., 2007), where the design process of products is done so that they can be completely recovered via recycling at the end of their life. These authors and disciplines all, in some way, seek a remedy for the associated environmental issues caused by human-induced production and consumption practices. This thesis adopts the definition proposed by Kirchherr et al. (2017, p. 229), where the goal of CE is stressed to contribute to sustainable development "simultaneously creating environmental quality, economic prosperity and social equity, to the benefit of current and future generations".

While acknowledging the broad intra-academic discussion of CE and its intentions/purpose (a never-ending academic pursuit), what constitutes CE principles and practices is more commonly accepted. CE is primarily discussed in relation to a hierarchy of 'Rs', Reduce, Reuse, Recycle etc. These R-principles (sometimes called imperatives) outline specific value retention strategies to enact the CE vision of slowing, closing and narrowing material and

energy loops. They are often presented as a hierarchy of preference (Reike et al., 2018). Over the last 15-years, the quantities and sequential order of these specific Rs have led to much discussion and debate (Geissdoerfer et al., 2017; Kirchherr et al., 2017). Due to this messiness, Reike et al. (2018) reviewed 38 different Rs from bodies of literature, including Waste Management, Reverse Logistics and Closed-Loop Supply Chain Management, Product Design, Environmental Sciences, Cleaner Production and Industrial Ecology. From this, they synthesised a 10R hierarchy (see Table 1.1).

These 10Rs are applicable for businesses and consumers in (i) the production and use phases of a product life cycle and (ii) the product concept and design lifecycle. The implicit notion and assumption within such hierarchies are that higher R strategies, e.g. strategies aimed at behavioural change and consumption (R0, R1), are fundamentally better from a value retention/sustainability perspective. In contrast to earlier R-hierarchies and conceptualisations of CE, this framework integrates a holistic and system-wide perspective on value retention practices that are directed throughout the entire product and material lifecycle. Given the systemic nature of global production and consumption practices, this research adopts this CE framing.

TABLE 1.1 Hierarchy of R-principles based on Reike et al., (2018).

R-principle(s)	Description
R0 Refuse	For consumers to buy less. Also for producers who can refuse to use specific materials or designs.
R1 Reduce	Linked to producers, stressing the importance of concept and design cycle, e.g. less material per unit of production (dematerialisation).
R2 Resell, reuse	Second consumer of a product that hardly needs any adaptation and works as good as new.
R3 Repair	Bringing back into working order, by replacing items after minor defects. This can be done peer-to-peer or people in the vicinity.
R4 Refurbish	Referring to large multi-component product remains intact while components are replaced, resulting in an overall upgrade of the product.
R5 Remanufacture	The full structure of a multi-component product is disassembled, checked, cleaned and when necessary replaced or repaired in an industrial process.
R6 Re-purpose	Popular in industrial design and artistic communities. By reusing discarded goods or components adapted for another function, the material gets a new life.
R7 Recycling	Processing of mixed streams of post-consumer products or post-consumer waste streams, including shredding, melting and other processes to capture (nearly) pure materials. Materials do not maintain any of their product structure and can be re-applied anywhere. Primary recycling occurs B2B, whereas secondary recycling takes place post municipal collection.
R8 Recovery (energy)	Capturing energy embodied in waste, linking it to incineration in combination with producing energy.
R9 Re-mine	Capturing resources from old or existing landfills or dumpsites

These Rs denote a plethora of individual material circularity practices that have already been consciously established in companies and by countries over the previous 40 years. As Ghisellini et al. (2016) outlined, these practices manifest at the behest of different actors at

different scales: micro, meso and macro. All of which relate to the different aims, goals and views of the actors involved (Corona et al., 2019). Micro practices relating to companies or consumers have included strategies to improve circularity through design for sustainability eco- or green-design (Bakker et al., 2014; Crul et al., 2009). Others include cleaner production practices, consumer responsibly or green or circular public procurement (Klein et al., 2020). Meso level practices, e.g. eco-industrial parks, refer more to inter or intra-firm dynamics, including supply chain integration and industrial symbiosis practices, i.e. the exchange of industrial by-products between firms (Chertow and Ehrenfeld, 2012; Jacobsen, 2006; Saavedra et al., 2018; Walker et al., 2021b). Macro practices, denote those pursued by nations, regions, provinces and cities, have included eco-cities (Hofmeiser et al., 2014), national zero waste and innovative waste management plans and country-wide policy approaches to material efficiency (Allwood et al., 2011; European Commission, 2014b; Worrell et al., 2016). This indicates that CE practices that promote material circularity are diverse and manifest on multiple scales and across different values chains and actor configurations.

Therefore, as Reike et al. (2018) argue, the emergence of CE should not be seen as a new phenomenon but rather a 'refurbishment' of older concepts, practices and ideas relating to waste and consumption that date back to the 1970s. Following the example of Blomsma and Brennan (2017), they categorise CE as an evolving concept that coalesced in three evolving phases. Further refining these periods, Reike et al. (2018) classified these phases as CE 1.0, CE 2.0 and CE 3.0. Broadly, CE 1.0 (1970-1990) relates to dealing with waste, the output and end-of-life (EoL) stage, and when the 3Rs (Reduce, Reuse and Recycle) was introduced. CE 2.0 (1990-2010) relates to connecting output and input strategies and increasing material efficiency. Environmental problems were also framed as potential opportunities, and concepts such as industrial ecology took off. The emergence of parallel eco-design and waste management policies such as extended producer responsibility (EPR). CE 3.0 (2010-present) presents actions and opportunities for value retention yet, are framed in a context of societal threats due to resource depletion. Various issues that already emerged in the 1970s, such as social justice, equity, resource depletion, and grow limits, resurfaced (see Meadows et al. 1972). The re-emergence of this debate recognises that existing responses and approaches to mitigating them had not been sufficient in addressing the social and environmental sustainability issues they are framed against (CE 1.0 and 2.0) (Ellen MacArthur Foundation, 2013; UNEP and International Resource Panel, 2011). This typology and its theoretical underpinnings have been reviewed and refined (see Calisto Friant et al. 2020). This thesis adopts this broader historical and evolving conceptualisation of CE because it recognises that responses to dealing with waste and consumption have already manifested in physical practices and responses.

The extent to which CE is new or is leading to something different and transformational remains to be seen. Much of the prevailing CE discussions focus on the more technological and technical level, concerning material and energy cycles related to supply chain

management, business models and recycling practices (Schöggl et al., 2020). However, the last few years have seen the evolution of CE discussions to the social dimension (Moreau et al., 2017). This has included CE's social sustainability (Walker et al., 2021a) and the governance of CE practices (Deutz et al., 2017). These perspectives recognise the importance of understanding production and material flows and the contexts and environments in which they are embedded. This discussion on the social implications of CE has evolved to such an extent that some are calling for the focus to switch from a circular economy (focusing on material and energy flows) to a 'circular society'. The focus on a circular society includes tackling social injustices in wealth, power and knowledge (Calisto Friant et al., 2020; Genovese and Pansera, 2021; Hobson, 2016). This line of argumentation contends that a narrower CE perspective (focused on the technical and technological aspects of material and energy flows) is limited in reconciling human activities with the biosphere (weak sustainability). A broader circular society perspective allows for a pluralistic non-technological approach, linked to governance patterns and more comprehensive behavioural change, and perhaps, moving towards strong(er) sustainability outcomes.

While the emergence of CE as a 'new' paradigm or approach has attracted much praise and support, the concept, discourse and actual practices have also received many criticisms and challenges. These relate to issues of growth and decoupling, thermodynamic limits, the interrelation between CE strategies and biodiversity impacts, and the varying issues of governance and assessments (Calisto Friant et al., 2020; Korhonen et al., 2018a). Similarly, many policy objectives have received criticism for their ineffectiveness or weakness in promoting more substantive CE outcomes (Calisto Friant et al., 2021). For example, while the Netherlands has a broad CE goal of a 50% reduction in the use of primary materials by 2030 and fully circular by 2050, the progress towards it has been weak. It necessitates a more coercive and interventionalist set of policies (Hanemaaijer et al., 2021). Therefore, when considering CE as an evolutionary concept, the question arises: what policies are already established that relate to CE? How do they function, and how effective have they been?

1.3 Established circular economy policies and practices

Governments have supported earlier iterations of CE (1.0 and 2.0) through various policies and practices (see Sakai et al., 2011). Since the 1970s, governments have altered EoL practices through policy interventions to minimise pollution and reduce landfilling (Reike et al., 2018). As Sakai et al. (2011) outlined, many nations have adopted municipal solid waste policies over the previous 40 years. In Europe, the first waste framework was outlined in 1975, which set the context and direction for European waste policy by focusing on waste minimization and protecting the environment and human health (The Council of the European Communities, 1975). EoL life policy has evolved and developed in municipal solid waste streams, including packaging, glass, cardboard, paper and plastics, with recycling

rates increasing over a similar period (see Rouw and Worrell 2011). Between 2004-2017, muncipal solid waste recycling rates rose, e.g. Germany's increased from 56 to 68% and Slovenia's from 20 to 58% (European Environmental Agency, 2021).

Similarly, European countries have adopted strategies related to design for recycling and eco-design, with the former relating to the design of products to enhance their EoL recyclability and the latter connected to broader design requirements for the in-use efficiency of the product itself (Worrell and Reuter, 2014, chap. 27). Policy instruments and strategies have applied various economic and regulatory, e.g. taxes and requirements, and forms of voluntary negotiation, i.e. an agreement between government and sectors to achieve a mutually agreed goal (Worrell and Reuter, 2014, chap. 34). As Milios (2018) outlined, the EU has developed various CE-related policies and regulations related to the production, use/consumption and waste stage of a product or materials lifespan. These include eco-design requirements (production), energy and labelling (use/consumption), and extended producer responsibility for products, e.g. batteries and electronics (waste stage).

However, these policies and approaches were often formed in different departments, responding to separate (and potentially competing) demands and requirements. Thus, the degree to which they collectively can respond to the wide systemic understanding of CE and the broader global societal issues remains unclear (see Backes, 2017). In essence, *is* the function these policies were designed to fulfil *still* suitable for what is now demanded?

Scholars, doing what scholars do, have devoted a significant amount of time to analysing the above policy approaches (cf. Milios, 2018; Savage, 2006). Yet, examining them in the context and framing of CE remains less developed and evolved. Existing research on CE policy has focused on the broad policy areas which can promote CE best. Examples include 1) policies for reuse, repair and remanufacturing; 2) green public procurement and innovation procurement; and 3) policies for improving the secondary materials market (Milios, 2018). At the same time, other authors have dedicated significant time to exploring policy-related barriers to CE in specific sectors and contexts (Campbell-Johnston et al., 2019; Chen and Ogunseitan, 2021; Govindan and Hasanagic, 2018; Van Langen and Passaro, 2021). Yet, a further and deeper examination of the CE policies is needed concerning their outcomes.

Researchers have outlined several broader issues when examining existing specific CE policies and practices, e.g. recycling practices. For instance, Worrell and Reuter (2014, chap. 30) reviewed the issue of whether recycling, a key goal of earlier and present CE-related policy strategies, is always desired. Using the case of Iron, Steel and Cement, they demonstrate how recycling processes cause the loss of material dissipation and quality degradation due to industrial processes. The increased energy demands can sometimes outweigh a higher CE R-strategy in terms of environmental impacts due to the increased processing demands on a product. A CE strategy might therefore not always be preferential from a broader sustainability perspective. Recycling has often been pursued as a goal in and of itself or under the premise of reducing landfilling and providing secondary resources.

Nevertheless, recycling is one of the (key) tools for natural resource management, not only a goal in and of itself (Worrell and Reuter, 2014, chap. 1). Gregson et al. (2015) illustrate the complexities in why recycling is pursued. Using the case of the UK, they show how the focus on turning waste into resources is challenging given issues of quality standards and are more of a moral and political response to issues of waste injustice and resource (in)security. The research above indicates that material circularity is (on its own terms) not always desirable and beneficial. There is tension between the need for circularity practices (dealing with consumption and waste), and the broader particularities and contexts in which these practices are embedded.

Acknowledging this aforementioned tension, there remains a lack of research examining existing experiences of established CE-related policies and practices (and their outcomes) from an organisational and governance perspective (Korhonen et al., 2018a). Whilst researchers have examined the limited conceptualisation and CE framing/approach of existing policies from a macro perspective, e.g. the overarching policy approach of the EU (Calisto Friant et al., 2021), or looked at the governance of specific substances (Deutz et al., 2017), a deep examination of the experiences of existing (prominent) CE policy approaches is lacking. As Gregson et al. (2015) outlined, there is a need to more deeply explore *what* has worked and what *has not* through examining and interrogating those practices which claim to contribute to CE, or, as we understand it CE 3.0 In essence, a more extensive examination and exploration of specific and individual policies that are already established is needed to understand the ways they do and do not contribute to circularity outcomes (sectors, products and R-strategies).

In examining and interrogating CE practices, this PhD follows two broad disciplinary lenses that fit within the field of sustainability science. Sustainability science seeks to understand the fundamental nature and interactions between nature and society (Kates et al., 2001). It is innately interdisciplinary, combining the knowledge from a wide range of disciplines to solve complex sustainability-related problems, e.g. fundamental issues of resource depletion, social justice, environmental governance and waste generation (Keitsch and Vermeulen, 2021; Reike et al., 2018). The first lens this research follows comes from the field of industrial ecology. Industrial ecology focuses on the flows of materials and energy in industrial and consumer activities, the effects of these flows on the environment, and the influences of economic, political, regulatory, and social factors on the flow, use and transformation of resources (Ayres et al., 2002). Therefore, the first lens focuses on the physical flows of products and materials and builds on the perspective that these flows can be optimized and directed.

Regulatory environments have generally shaped the collection and use of waste products, usually through prescriptive R-hierarchies, e.g. Reuse and Recycling. Examples include the Ladder van Lansink in the Netherlands, the original 3Rs (Reduce, Reuse and Recycling) in the US, or the Waste Hierarchy in the Waste Framework Directive (Lansink and Veld, 2010;

Waste Framework Directive 2018/851, 2018a). In exploring these regulatory environments, it is important to stress that policies that govern them do not operate in a vacuum but in an environment that contains a complex set of actor configurations and networks. These include diverse world views and perspectives related to the individual, company or institutional interests. This thesis turns to the (environmental) governance literature for the second lens to understand this complexity and diversity. Governance broadly refers to the structures, processes, rules and traditions that determine how people in societies make decisions and share power, exercise responsibly and ensure accountability (Cundill and Fabricius, 2010; Folke et al., 2005; Patterson et al., 2017). The move from government to governance can be documented from the 1970s, when increased recognition was given to the fact that the state isn't the only controlling actor. Instead, more attention needed to be played to others, including the market and civil society (Driessen et al., 2012). There are different types of governance arrangements, e.g. particular actor configurations, from the more hierarchical top-down command and control approaches to forms of private and self-governance, each with a particular actor, institutional features and policy content features. The focus on governance allows us to consider the arrangements (relating to organisational, policy and social relations) that affect the material circularity outcomes of product and material flows. These two broad yet complementary theoretical approaches form the guiding perspectives of this research, bringing together two disciplines together in a novel manner to examine CE policies.

In following these approaches, this thesis explored and examined one specific policy approach: extended producer responsibility (EPR). EPR operationalises the 'polluter pays principle' to promote cleaner production practices. Building on the notion that those who produce pollution should bear its management costs, it was introduced as a policy instrument in the early 1990s by various Northern-Western countries such as Germany, Sweden, France and the Netherlands (Ayres et al., 2002; Lindhqvist, 2000; Vermeulen and Weterings, 1997). It built on the premise of transferring or bestowing greater responsibilities on market actors for their products, in this case, responsibility for the take-back, recycling and final disposal of the product (Lindhqvist, 2000). In this sense, EPR represents a form of 'public-private governance' that emerged during the 1990s and early 2000s, where market actors were given new responsivities in pre-determined boundaries (Börner and Hegger, 2018).

The focus on EPR in the context of CE has been explored by various scholars, including examining its effectiveness to contribute to reducing packaging waste (Andreasi Bassi et al., 2020; Rubio et al., 2019) and best practices for specific products categories, e.g. lamps (Richter and Koppejan, 2016). Nevertheless, we focus on EPR as an approach to CE for two distinct reasons: 1) it is an established CE (2.0) practice, which offers a chance to explore, learn and generate theoretical insights from existing experiences (at the time of starting this PhD the studies on EPR and CE were few); and 2) this policy approach was/is used contemporary CE policy developments, such as the EU CE action plan (2015) and EU Green

New Deal (2019). The established nature and importance of EPR (in EU CE policy) means it is an ideal area to connect past experiences and current (theoretical) developments. By focusing on EPR from an evolutionary and critical perspective, this research had the opportunity to contribute directly to existing policy discussions on CE by providing insights on the lessons learnt and areas of improvement.

Therefore, to contribute to understanding the effectiveness of existing policy to contribute to CE, this PhD addresses the following main research question:

To what extent can extended producer responsibility (as a form of public-private governance) act as an effective mechanism in supporting the implementation of circular economy in the European Union?

This broad research question was approached and then analysed on two distinct levels. The first level considers the instrument's effectiveness on its own merit, i.e. the purpose and function for which it was designed (learning from past experiences). The second applies an external critical theoretical perspective to explore the policy in a new light, e.g. CE. In essence, how can we comprehend the effectiveness of the current EPR approach from the discursive claims and intentions of CE?

This central question is supported by the following sub-research questions.

1. How has EPR been implemented and organised within EU member states?
2. How do EPR strategies vary in respect to their effectiveness, limitations and outcomes?
3. What (EPR) mechanisms (and value decisions) are used to identify the trajectory and use of products and materials post-collection?
4. How can current approaches to EPR be further developed and strengthened to contribute to CE?

1.4 Research approach

This thesis is based on five articles, which form the basis of each chapter. A detailed description of each methodology is outlined in the respective chapter. This thesis examines existing approaches and experiences of CE practices and policies from theoretical and empirical perspectives.

The research process and approach was greatly inspired by the theory and practice of transdisciplinary research (TD). TD dates back to the 1960s and earlier, stressing collaborative forms of science and scientific knowledge production. The rationale is to create knowledge with society instead of for society by creating outputs that correspond to real-world needs and not just academic outcomes. In recent years, TD has become integrated with sustainability and sustainability science research, aiming to understand and

tackle wicked challenges (Keitsch and Vermeulen, 2021). Broadly, TD deals with research and problem fields in a way that it can (a) grasp the complexity of problems, (b) take into account the diversity of life-world and scientific perceptions of problems, (c) link abstract and case-specific knowledge, and (d) develop knowledge and practices that promote what is perceived to be the common good. (Pohl and Hirsch Hadorn, 2007). Consequently, TD processes and knowledge outputs differ from conventional fundamental disciplinary knowledge production. Instead, it results in what Gibbons et al. (1994) refer to as 'Mode 2' knowledge production, which is problem-orientated and interdisciplinary, in contrast to traditional theory and curiosity (Mode 1). Mode 2 knowledge creation has three main features, these being 1) problem-solving orientation; 2) communication of results going beyond the (disciplinary) scientific arena, e.g. addressing practitioners; and 3) different dynamics in its application outside specific disciplines (Gibbons et al., 1994, pp. 4–7).

In their review of TD, Vermeulen and Witjes (2021) argue there is no clear or singular means of doing TD work. Instead, they outline three "tastes" of TD, which reflect the diversity and level of actor engagement and the complexity of the problems and solutions. The three "tastes" are intra-academic transdisciplinarity, solution-driven transdisciplinarity and fairness-driven transdisciplinarity. Projects which contain few academic and external partners are labelled as "small-range transdisciplinarity". Whilst the approach and rationale for this PhD research have not strictly followed a TD approach, the aspects of a 'problem-solving orientation' and the 'dissemination of knowledge beyond disciplinary boundaries' to practitioners has been a guiding influence for both the framing and process of the research. Through working with a selection of public, civil society and market actors on the issue of CE and EPR, this research can be seen to be inspired and shaped by a solution-oriented and small-range form of TD.

The solution-orientated emphasis was a highly crucial personal aspect for my research process. I strongly support the notion of science and scientific knowledge production actively engaging in contemporary societal discussions. In this case, the ongoing and current discussion on the use and form of EPR to contribute to the EU's CE agenda. This entails a form of science and knowledge production for and with society. In this sense, the research, and I have taken a distinctly normative position with respect to both the process and output of research, i.e. that CE as a discursive concept can contribute to some of the waste, resource and sustainability challenges it is framed against. **But,** the outcomes depend on how it is conceptualised and ultimately applied and implemented. This normative position recognises the urgency of the impending societal challenges outlined above and very much seeks to become involved in the process of change and change-creation.

Owing to the inductive nature of this research, which also aimed at generating novel insights, this research followed a Grounded Theory (GT) approach (Bryman, 2012). GT refers to a general strategy for building theory from rich data derived through observation and exploration (Bryman, 2012). Data collection and analysis are interrelated, with analysis and reflection beginning as soon as the first data is collected (Strauss and Corbin, 1990). Insights

developed in the first year of research were integrated and included in later research projects and processes. This iterative manner of proceeding allows all relevant issues to be incorporated in subsequent research steps and lets the research(er) capture all potentially relevant aspects of the topic as soon as they are perceived. GT seeks to undercover the relevant conditions and determines how the actors respond to changing conditions and the consequences of their actions (Strauss and Corbin, 1990). Therefore, this apporach is one of discovery that grounds the theory in reality (Glaser and Strauss, 1967). GT represents the broad approach to the exploration of CE related practices.

This research adopted a case study approach to develop practical and theoretical insights on the experiences and approaches to CE. A case study is an intensive and detailed analysis of a single case through examining an existing or emerging phenomenon through observing its complexities and particularities within its real-life context (Stake, 1995; Yin, 2003). Using case studies also allows for a mixed-methods. However, a case or case study does not allow more universal generalisable results. Instead, it creates insights specific to that particular context. Nevertheless, they are applicable as the basis of theory development and generation (Bryman, 2012; Yin, 2003). The case study approach provides the context for examining the organisational of existing CE EPR approaches within the EU. This research used the insights from two case study countries (Italy and the Netherlands[3]), which was specified in the Cresting project research description (see research context). The focus on the national/member state level is that while EPR and waste legislation is set at the EU level, transposition and implementation is given to the member states. This research focused on several product categories within these case studies: passenger car tyres, floor covering, cars and waste electrical and electronic equipment. The choice and justification of these products are further outlined and elaborated in the specific chapters (Chapters 4, 5 and 6). Through following the notion of building theory and knowledge through grounded theory, a choice of which actors and actions needed to be made. For this, we used the governance literature and the framework on modes of governance (Driessen et al., 2012; Lange et al., 2013), that focused on actor, institutional and policy content features.

This research used a variety of methods (qualitative and quantitative), which are detailed in the specific chapter. These consisted of highly conceptual interpretative methods, e.g. critical literature review (Chapters 2 & 3), a qualitative critical policy study with interviews (Chapter 4), a mixed-methods Delphi study (Chapter 5), to those methods more closely linked with industrial ecology, e.g. thermodynamic rarity assessment (Chapter 6). The starting point and driving interest of this PhD were providing research and outcomes that were societally relevant and applicable. A definite challenge considering the demand for theoretical knowledge from academia: thus, the choice and use of various methods reflect

3 The chapters and cases presented in Chapters 4,5 & 6 are drawn from the Netherlands (4 & 5) and Italy (6). However, during the research many interviews were conducted with stakeholders from France. Many of the case study reflections and discussions for the case of France, while not presented here are being developed in a policy brief.

this intent to not only capture and theorise about phenomena, but also realise that different methods provide different insights applicable for different (societal) contexts and are equally useful, i.e. both academically and theoretically novel, whilst also relevant for contemporary CE policy discussions. The totality of interactions with stakeholders in presented in the box below (see Appendix B).

This PhD was based on

- 91 interviews
- 18 company and site visits
- 9 workshops with stakeholders
- 6 project workshops
- 6 conference presentations and discussions
- 1 secondment stay with a governmental organisation

The insights have been presented for a number of organisations including the OECD, Rijkswaterstaat and the Dutch Environmental Assessment agency (PBL)

From applying these different methods, this PhD contributes to the emergent field of CE in three distinct areas: First, an extensive theoretical contribution in the connection between CE and EPR from a Governance and Industrial Ecology perspective. Second, and based on the first, is an empirical contribution to how these concepts are concretely manifested in specific (case study) contexts. The basis of analytic generalisation as outlined in a GT approach allows insights to be developed from the cases studies that can then be explored or provide insights for other (EPR) contexts. Finally, this thesis offers a practical contribution to the emerging and evolving discussion on EPR as a tool for CE and provides concrete societal recommendations and reflections for improvement in the EU member states (see Appendix for associated policy briefs). The different chapters of this thesis address this interrelation and interdependency between theory and practice to different extent (Figure 1.2)

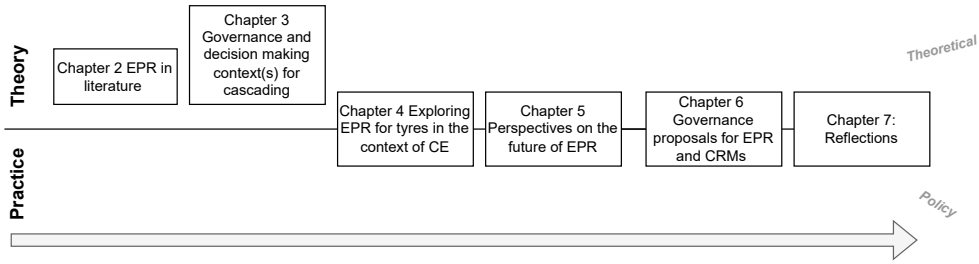

FIGURE 1.2 Chapters of this thesis and their focus on empirical and theoretical data.

Chapter 2: Extended producer responsibility in the literature

The main aim of Chapter 2 is to provide a review of the concept of EPR, including its diversity in terms of application and conceptualisation. This review also outlines several critical issues concerning the intersection between CE and EPR. This chapter provides answers to sub-research questions (s-RQs) 1 and 3. Specifically, through showing how EPR has been organised and implemented generally (and in the EU), through introducing a broad typology of the different organisation models (s-RQ 1), and what policy instruments and mechanisms are used to effect material circularity practices (s-RQ 3). This chapter was based on Vermeulen and Campbell-Johnston (2022).

Chapter 3: Governance and decision making contexts for the cascading of products and materials

This chapter also used a critical literature review, which synthesised three bodies of scientific literature: CE, cascading and up/downcycling. It proposes a framework to conceptualise the complexities of the decision-making contexts in which the allocation choices and (material circularity) outcomes for products and materials are made (or can be made) in a CE. This chapter formed the conceptual basis for critically thinking about the decision-making contexts and the mechanisms of material allocation is made throughout the research. The key aspect pertains to considering how material flows and their uses require examining the social governance context and the value decisions that (or can) inform the material circularity outcomes. This chapter provides answers to s-RQ 3, detailing the theoretical understanding of the mechanisms through which collected products and materials can be made. This chapter was based on Campbell-Johnston et al. (2020b)

Chapter 4: Exploring EPR for tyres in the context of CE

This chapter looked at a case study of tyre recycling in the Netherlands from a critical CE perspective. The main aim was to understand how the governance of EPR has been organised, and critically reflect on the key lessons and strengths of this older CE system. In addition, the research then critically highlights several weaknesses from a governance and organisational perspective that must be addressed for EPR to contribute to a more radical and transformational idea of CE, e.g. beyond collection and recycling practices. Therefore, this chapter provides answers to s-RQ 1, 2 and 3. This chapter is based on Campbell-Johnston, et al. (2020a).

Chapter 5: Perspectives on the future development of EPR

This chapter aimed to draw on the extensive experiences and knowledge of CE and EPR related practitioners. Using a Delphi study, this research approached 50 experts and asked them how EPR could be transformed to contribute to CE, in this case, the CE goals of the Netherlands. Based on the results of the Delphi, seven key recommendations were outlined for improvement, for both the Netherlands and EU. These changes were themselves clarified and validated by the experts in the study, giving weight to their suitability regarding policy changes for EPR. This chapter also reflects on the challenge of CE transitions, given the

diversity of (incumbent) actor perspectives and positions. This chapter provides answers to s-RQ 4 through detailing a series of recommendations of how EPR can be strengthened and improved at the policy level. This chapter is based on Campbell-Johnston, et al. (2021).

Chapter 6: Governance proposals for EPR considering critical raw materials

This chapter aimed to highlight a number of weaknesses within the current governance organisation of EPR, specifically the targets and conditions outlined under EU policy that relate to product and waste laws. This chapter used the example of critical raw materials, e.g. materials such as cobalt and lithium, to explore their presence, quantities and losses within an electronic recycler in Italy. Using an indicator named thermodynamic rarity, this research showed the differences between mass based (weight) and rarity targets, a key feature of EPR targets. This case study revealed the specific decision making outcomes (and their consequences) caused by the existing EPR governance structures. The research outlined a number of clear policy recommendations that include changes to both EPR and Eco-design policy and how to integrate them. In essence, this chapter outlines a conceptual understanding of how EPR as a waste tool can be more concretely connected to influencing product design outcomes. This chapter therefore provides answers to s-RQs 3 and 4 through detailing the (effectiveness/ ineffectiveness) of the existing mechanisms and value options within the EPR systems (s-RQ 3), and detailing potential improvements (s-RQ 4). This chapter was based on Campbell-Johnston et al., (forthcoming).

This PhD draws together a set of empirical and theoretical reflections (Figure 1.2). Chapter 7 ends with a reflection on the conclusions observed from the research, and reflections on the methodological approach. By synthesising and critically reflecting on the research outcomes and process, this final chapter proposes an answer to the main research question.

2

This chapter was based on Vermeulen, W.J.V., Campbell-Johnston, K. Extended Producer Responsibility. Handbook of Recycling 2nd edition forthcoming (2022)

Extended Producer Responsibility

Abstract

Extended producer responsibility is a policy approach or instrument encompassing a broad array of economic, regulatory, and informative requirements. These usually take the form of shifting the responsibility for the product after the post-consumer (waste) stage to the producer. Extended producer responsibly originated in Europe but has been applied globally over the past 30 years. This chapter reviews the origins and varying definitions of the instrument, its effectiveness, and key limitations. Finally, we discuss its role in fostering a circular economy.

2.1 Introduction

Extended Producer Responsibility (EPR) emerged as a means of operationalising the 'polluter pays principle' to promote cleaner production practices. Building on the notion that those who produce pollution should bear its management costs, it was introduced as a policy instrument in the early 1990s by various northern-western European countries such as Germany, Sweden, France and the Netherlands (Ayres et al., 2002; Lindhqvist, 2000; Vermeulen and Weterings, 1997). EPR emerged when the landfilling of post-consumer waste was high and where local government bore the costs and responsibilities of waste management. Since then, it has been applied widely, mostly in high-income countries, but also in middle- and low-income countries (OECD, 2016, 2014). In this Chapter, we discuss the various perceptions of EPR, as a normative principle and as a policy instrument. After that, we discuss the diverse forms of application and what is known about its effectiveness in practice. Next, we discuss some critical issues around applying EPR and its limitations. Finally, we discuss the role EPR can play in the pursue of a circular economy.

2.2 Defining extended producer responsibility

Extended Producer Responsibility (EPR) was initially defined in a report to the Swedish Ministry of Environment by Lindhqvist as *"an environmental protection strategy to reach an environmental objective of a decreased total environmental impact from a product, by making the manufacturer of the product responsible for the entire life-cycle of the product and especially for the take-back, recycling and final disposal of the product"* (Lindhqvist, 2000, p. iii).

Alternative definitions (see OECD, 2016) narrow the scope of EPR to the producer's responsibility only being extended to the post-consumer stage of the product lifecycle. However, in its original broader interpretation, EPR as a concept, contains two interrelated elements. Namely, that producer's 'responsibility' should extend to the post-consumer phase of a product's lifecycle, and that increased responsibility would result in product design changes to mitigate pollution and waste management costs. In this way the output side of the value chain and the input side of the value chain are linked to each other. However, even beyond this discussion of broad and narrow definitions, we see much confusion about the concept because national governments and supranational bodies apply and define EPR with even more diversity. As we will see in section 2.3, the application practice reflects this diversity even more due to different choices in the design of EPR policies.

In its original European application, EPR fitted in the shift away from command and control policies to more public-private and collaborative modes of governance (Vermeulen, 2002). Responsibility for solving problems in the commons is shifted towards market actors, with governments setting the ground rules and boundaries in the form of framework regulating,

targets and reporting requirements. The implementation of EPR may then include forms of negotiated agreements, which has been adopted in and adapted to many diverse contexts. The transfer of responsibilities includes different forms or combinations of 'responsibility'. Responsibility can take the form of:

- Liability for (proven) environmental damages caused by the product;
- Financial responsibility for covering the (partial or full) costs for collection, recycling or final disposal of the product;
- physical responsibility for product management or its effects;
- ownership of the products throughout their life span; and
- informative responsibility in providing information on the properties of the product in question (Lindhqvist, 2000; Massarutto, 2014).

In the narrow, output-oriented approach of EPR, this transfer of responsibility applies to the organisation of recycling after the user/consumer phase in the product life cycle. Organising recycling can be done through an *individual* EPR or *collective* EPR: in the first case, companies organise the take back of their own products, while in the second, *producer responsibility organisations* (PROs) are established jointly by producers to organise this. In practice, these are often combined: European regulation and member state regulations ascribe the responsibility to each producer, but they are allowed to organise this collectively, which is the dominant practice in Europe.

Thus, these PROs have a central role in EPR practices. As shown in Figure 2.1, this includes (Mayers and Butler, 2013):

- Organizing collection logistics and recycling, e.g., subcontracting waste companies by PROs to carry out day-to-day operations.
- PROs recover the organisational costs from producers either through fixed fees or variable cost-sharing between producers based on the number of products sold.
- Producers ultimately recover their take-back costs from their customers, either as part of regular product pricing or visible recycling levies charged on top.

The introduction of collective EPR may be very complex: many thousands of producers and waste collectors need to agree on systematic methods of working together to ensure waste is collected where and when required (Figure 2.1). When established, PROs manage waste services and administer multiple payments to finance these operations. Collection points are organised, when relevant in collaboration with municipal authorities; collection points in shops and other public places are also created. Possible free-riders, companies not participating, but benefiting, need to be identified and forced to join. This also includes online sellers within and outside the country (Hilton et al., 2019; Kalimo et al., 2015). How exactly all this is organised and how the roles of public and private actors are defined varies greatly between countries and between waste categories.

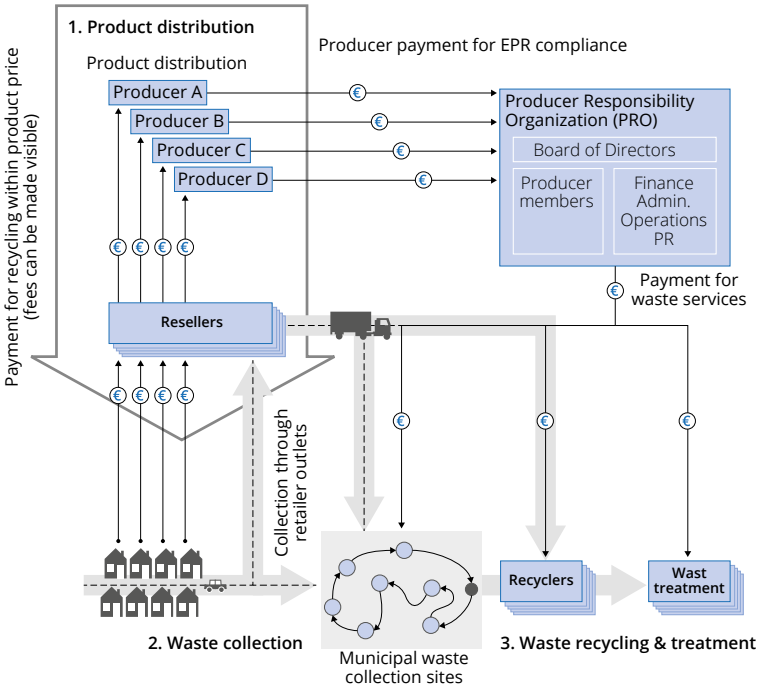

FIGURE 2.1 The role of producer responsibility organisations. From: Mayers and Butler, 2013.

EPR is implemented through administrative, economic, and informative instruments. The composition of these instruments determines the precise form of the EPR. Hickle describes four models with variations in the distribution of roles between producers and public authorities (Hickle, 2014a, p. 63), while the OECD distinguishes four slightly different models (OECD, 2016):

- One single PRO with commercial and/or municipal collection and processing services.
- Multiple PROs with the clearinghouse and commercial and/or municipal collection and processing services.
- Governance structure for tradable credits system.
- Government-run EPR system.

In the broader definition, as described by the OECD, other recycling-oriented instruments are labelled as EPR. Apart from product take-back requirements, done through collection and recycling targets as described above, the OECD includes the following in its EPR definition:

- economic and market-based instruments, e.g., deposit-funds, advanced disposal fees or (virgin) material taxes, landfill taxes.
- regulation and performance standards, e.g., minimum recycled content in products; landfill bans and

- information-based instruments, such as reporting requirements or product labelling into the (OECD, 2016).

This list emphasises that a government's toolkit for supporting recycling contains more options than the shifting of financial, organisational and managerial responsibility to producers through *individual* EPR or *collective* EPR. EPR schemes are, in many countries, one of the instruments in a policy mix. Aside from deposit refunds, recycled content requirements and labelling, other approaches may also include input side-oriented elements, like eco-design regulations and corporate sustainability programs and reporting requirements. In this chapter, we focus on EPR practices in which *financial and physical responsibility* for collection and recycling of post-consumer waste is established.

2.3 Current Practice of EPR

Over the last 30 years, EPR related policies have been steadily adopted by national governments, with the number increasing significantly from the 2000s (Kaffine and O'Reilly, 2013). Globally, most EPR related policies cover product categories such as electronics, packaging, tyres and batteries, with formal take-back requirements as the most common policy approach (OECD, 2016). EPR for electrical waste has by far received the most attention both from policy and scholarly research related to concerns over health, environment and resource depletion (Corsini et al., 2017; Ongondo et al., 2011).

2.3.1 EPR within the European Union

The European Union (EU) has used EPR as a key approach to waste management. The EU has mandated EPR schemes for Packaging (94/62/EC; 2018/852), End of Life Vehicles (ELV) (2000/53/EC), Waste Electrical and Electronic Equipment (WEEE) (2002/96/EC; 2012/19/EU), Batteries (2006/66/EC) and, most recently single-use plastic products, e.g., food containers (EU2019/904). These directives broadly require forms of financial and physical responsibility, where producers must finance and organise the post-consumer collection and recovery. They include a broad definition of 'producer': one that also contains importers and distributes, not only original equipment manufacturers. They also include collection and recovery targets, for example a minimum of 85% re-use and recovery of the weight of collected end-of-life vehicles, compared to what is placed on the market in the End of Life Vehicles (ELV) (2000/53/EC).

EU member states have generally followed two organisational approaches to EPR: the clearinghouse model and delegated governance schemes or the single PRO model. In the former, producers in the broader sense register with the government-managed clearinghouse and declare the number of products they placed onto the market. Producers are assigned a collection responsibility equivalent to their market share (Khetriwal et al., 2011). Countries such as Germany, Spain, France, and Italy have adopted this model for

the WEEE, corresponding to the second OECD model described above. Conversely, in a delegated governance scheme, implementation is charged to PROs, which assume the collection and treatment responsibilities on behalf of all the producers. This corresponds with the first OECD model described above. This mode of organisation is prevalent in countries such as the Netherlands, Sweden, Belgium and Switzerland. In delegated governance schemes, non-profit organisations are usually set up by producers or trade-associations. A final element in the organisation of EPR is the producer responsibility providers. These are specialised organisations that can assume collection and treatment responsibilities on behalf of producers or PROs (Khetriwal et al., 2011). Beyond the mandated schemes, member states have deployed EPR for other products, including tyres, used oils, textiles, graphic paper, medicines, mobile homes etc. France is the member state that has used the instrument most frequently, with over 20 schemes (European Commission, 2014c).

2.3.2 Global application

Globally, EPR is predominantly applied in high-income countries. The four most often addressed product categories are electronics, tyres, vehicles and packaging (OECD, 2016), with 90% of the EPR schemes being applied in Europe, North America, Australia and New Zealand. The uptake in Asia and Africa is far more limited, although examples are available, like in India (Turaga et al., 2019), Malaysia, Thailand and Vietnam (Akenji et al., 2011) and Nigeria (Odeyingbo et al., 2019)[4].

In North America, EPR is applied in both the USA and Canada. Canada's Ministry of Environment first outlined working principles on EPR in 2007, with a national plan outlined in 2009. This included broad policy objectives and targets on the number of operational EPR programmes and product categories. Although non-binding, most provinces adopted a similar organisational approach, except Québec, which pursued a programme that included progressively increasing collection and recovery targets, coupled with financial penalties for producers or PRO's. Following a national framework, key principal differences emerged regarding the scope of products covered, targets, physical and financial responsibility (Leclerc and Badami, 2020).

Conversely, the USA has followed a much less coordinated but state-driven approach to EPR. Consequently, the adoption of EPR is more fragmented, with states such as Vermont, Maine and California being early movers and frontrunners (Nash and Bosso, 2013). EPR laws have been introduced in several states for rechargeable batteries, mercury thermostats, auto-switches, paints, and electronics. Although these are not uniform in their requirements, they represent successful applications, including performance incentives and goals (Nash and Bosso, 2013). Since 2010, US states have begun examining the Canadian EPR framework to learn from their successes (Hickle, 2013).

4 See also the members of the WEEE forum

In Asia, EPR for WEEE is also organised in different ways. For example, Taiwan's government initiated EPR 1998. The design of EPR gave immediate emphasis to modulating the fees paid by producers based both on the collection and recovery costs but also the environmental design of the product. In 2012, producers could reduce their EPR costs by up to 30% if products were certified with eco-labels, including environmental, energy and water-saving performance (Cheng et al., 2019). To tackle the environmental impacts of WEEE, Japan introduced and enforced the home appliance recycling law from 2001, focusing on washing machines, refrigerators, TVs and air-conditioning units. This law was amended and broadened in 2008, with additional measures focusing on containers and packaging materials, food waste and end-of-life vehicles (Ogushi and Kandlikar, 2007). A key difference in this system's organisation is the disposal fee, which is paid by consumers themselves at the point of disposal and is not incorporated within the product's price (Ongondo et al., 2011).

In China, the massive demand for electronics and the growth of electronic waste led to calls for EPR (Yu et al., 2008). EPR is a core component of the "Circular Economy Promotion Law", outlined in 2008 (Veenstra et al., 2010). These were developed as a policy approach in 2012, covering five product categories, televisions, washing machines, refrigerators, air conditioners and computers, adding another nine by 2014 (Zhang et al., 2020).

South Korea's first deposit refund systems, being unsuccessful, was replaced by an EPR system that mandated physical and financial responsibility in the early 2000s (Kim and Mori, 2015; Manomaivibool and Hong, 2014).

Additional studies have focused on applying EPR for WEEE, particularly backyard recycling practices from WEEE shipped (il)legally from, for example the EU, to developing countries (Nnorom and Osibanjo, 2008). Other global studies have examined the use of EPR for the management of waste tyres. For example, research from Ecuador emphasising social sustainability goals included in EPR goals, such as building resilience for vulnerable communities through civil engineering projects (Cecchin et al., 2019). Additionally, research on waste tyres in Brazil emphasised the challenges of transferring a policy to the context of "insufficient institutional capacity" resulting in limited waste preventative measured being introduced (Milanez and Bührs, 2009).

2.4 Effectiveness

Statements about the effectiveness of EPR need to consider the vast diversity of its application around the world (Cahill et al., 2011; Khetriwal et al., 2011; Tasaki et al., 2019), and the double ambition of the instrument: organising the recycling at the end of the value chain and stimulation the redesign of products toward more sustainable and circular versions. Many studies have analysed cases of EPR for specific waste streams in particular geographies. For this section, we use some of the meta-evaluative reviews published in the last decade, to show the main observations about factors affecting the success of EPR,

summarized in Table 2.1.

TABLE 2.1 Strengths and weaknesses of EPR, as mentioned in academic literature.

	Strengths	Weaknesses
Organising recycling	• EPR schemes do divert waste streams from landfilling or incineration to forms of material recycling, which leads to environmental benefits [3, 4, 7, 8, 10, 11] • National (or in Europe EU-) targets are met in frontrunning countries [8, 11] • It uses industry's managerial capacity to organise recycling markets [2, 7]	• Targets and standards are not harmonized and weakly enforced and are not met everywhere [3] • Lack of harmonized definitions [10] • Responsibility for recycling beyond the targeted collection rates is not taken (non-separated waste, littering, orphans) [5] • Recycling process choices need to be based on better assessments [4, 5, 8, 11] • EPR promotes material recycling over re-use and other R-options [1, 8] • Exports of waste to low-income countries prevail [3, 5]
Efficiency	• Low operation costs (2% to 0,1% of product price), but data incomplete [2, 8, 9, 11] • Higher volume of materials collected in collective EPRs enable more efficient recycling technology [2] • In practice, both competing and single national PROs exist in different countries; views on which is most efficient are contested [3]	• Voluntary PROs face freeriding [4, 6] • The level of costs of recycling allocated to producers differs strongly between countries [2] • Data collection and sharing is weak due to cost avoidance [1, 4, 5, 10] • In case recycling is profitable, recycling processers compete with collective systems, cherry-picking the easy gains [3,4]
Stimulating eco-design	• Being responsible for the end-of-life is assumed to stimulate redesign of products by producers [11, 12]	• Low impact on eco-design [3, 4, 8, 10] • Weak incentives on eco-design, fee systems ignore eco-design efforts [1, 2, 3, 5, 8] • The lack of harmonized legislation hinders impacts on product design [3]

Legend: 1 = (Lifset et al., 2013); 2 = (Massarutto, 2014); 3 = (Kunz et al., 2018); 4 = (Atasu, 2019), 5 = (Kalimo et al., 2015), 6 = (Hermann et al., 2020), 7 = (Shan and Yang, 2020); 8 = (Campbell-Johnston et al., 2021); 9 = (Favot et al., 2018); 10 = (Pouikli, 2020); 11= (Vermeulen et al., 2021); 12 = (Lindhqvist, 2000)

A simple answer to the question of effectiveness is not available. All EPR schemes do achieve some level of diversion of waste into recycling, thus limiting landfilling and its environmental impacts (OECD, 2016). Assessing and comparing EPR systems between product categories and countries is difficult, owing to the differences in definitions, reporting and monitoring requirements and data quality (Ongondo et al., 2011).

2.4.1 Organising recycling

Cases of full or almost full collection of post-consumer waste do exist, for example, for cars and tyres (OECD, 2016). EPR for tyres has resulted in material recycling rates of up to 80-95% in various European countries (Winternitz et al., 2019). With its high end-of-life material value, worldwide end-of-life vehicle policies in high-income countries, mostly applying EPR, generally achieve 95% or more recovery results (Sakai et al., 2014).

EPR schemes under the EU Directives are meant to apply standardised reporting and monitoring. Despite inevitable data quality issues, we can make some observations as to the effectiveness of these Directives. Table 2.2 shows EPR schemes for glass, packaging and WEEE for some EU countries, with examples of the highest and the lowest performers.

TABLE 2.2 EPR regulated waste streams and performance in some EU countries (2017). Data from: Eurostat, 2021.

Country	*Glass packaging (recycling %)*	*Plastic packaging (recycling %)*	*WEEE (collection % 3-year average)*
Austria	84.1	33.4	62.4
Belgium	100	45.5	49.4
Germany	84.4	49.7	45.1
Greece	36	41.4	42.4
Hungary	34.2	32	60.6
Poland	63	34.7	45.4
Portugal	49	34.9	53.9
Romania	63	47.6	-
Slovenia	98.5	60.4	39.1
Sweden	93	48.4	56.3
EU28	75.9	41.7	47.6

EPR schemes have proven to contribute to high recycling rates. For example, in Germany and Sweden, both early movers in implementing EPR systems, saw recycling rates for glass packaging above 80% of the quantities collected. EU recycling targets for glass were set at 60% of weight in 2008 and rising to 70% by 2025, which many countries have already met.

Conversely, recycling targets for plastic packaging were set at 22.5% of weight in 2008 and then rising to 55% by 2025. Whilst the countries illustrated in Table 2.2 have all exceeded the 2008 targets, some way is yet to go for Hungary, Poland, and Portugal to meet those in 2025.

EU targets for WEEE dictate that from 2016 a 45% collection rate of products sold in the previous three years will apply, climbing to 65% in 2019 or 85% of WEEE generated in that year. From 2017, it is clear that, whilst EPR has led to increased WEEE collection, few countries have met the proposed target. Table 2.2 shows that the best performing EU countries achieved about 60% collection in 2017, while at the lower side 40% was achieved.

WEEE recycling in the US is organised at state level, with various approaches (Hickle, 2014b; OECD, 2016). The State of Minnesota applies a clearinghouse model, with a financial tax and reward system, aiming for 80% recycling of electronics, which claims to be successful, but performance scores like in Table 2.2 are not available (Alev et al., 2019).

ERP has also been applied for batteries around the world, collection targets being more moderate, aiming at 40-50% collection in the mid-2010s in the EU and Northern America, and 50-80% of that to be recycled (Turner and Nugent, 2016). Japan achieved 50-77% recycling, varying per type of battery (OECD, 2016).

An important element in assessing the success of EPR is the quality of the recycling processes applied. In many cases, lower-level forms of recycling are dominant, including incineration with energy recovery. Considerations of cost-effectiveness may, when the decision making is delegated to market actors, result in cheaper forms of recycling, like with car tyres, being processed to fillings for artificial sports fields (Campbell-Johnston et al., 2020a (Chapter 4)). A comparison of EPRs for batteries in Europe and North America showed that, in many cases, no requirement on the best technologies to be applied is included in the regulations (Turner and Nugent, 2016). Indeed, post-EPR processing operations have led to downcycling, e.g. materials of lower quality or used in a lower value application. For example, Ortego et al. (2018) showed that the recycling of cars under the EU's EPR scheme, in its current form caused downcycling and loss of geologically scarce materials. Whilst in the US, Fishbein (2000) showed that for carpets, the quality of the feedstock and choice of recycler within an EPR scheme affect the degree of downcycling. Currently proposals for more detailed value retention and recycling targets are discussed in Europe. At the same time a better connection to eco-design regulation is promoted.

Markets for recycled materials are often poorly developed, and higher prices of recycled materials and quality concerns are barriers to replacing primary raw material with recycled materials. EPR schemes do not play an active role in improving the functioning of markets for secondary materials. It is left to the recycling companies and the original producers to make their choices on economic grounds. Currently, few data is available on the rate of application of recycled materials in EPR-regulated product categories, such as electronic equipment, batteries, plastic packaging, cars and tyres.

Information about the effectiveness of its application in lower- and middle-income countries is hardly available. With less established waste management systems in place, organizing producer responsibility poses bigger challenges than in the high-income countries (Ferronato and Torretta, 2019). Some countries have introduced it, but it often remain only a concept on paper, or with voluntary uptake, like in Malaysia (Agamuthu and Victor, 2011).

2.4.2 Economic efficiency

It is argued that delegating responsibility for end-of-life treatment and recycling is efficient (Dubois and Eyckmans, 2015; Winternitz et al., 2019), though limited data on this is available. Collective systems avoid setting up multiple infrastructures and create economies of scale (Monier et al., 2014; Tojo, 2006). In Italy, the costs for organizing recycling of electronic equipment are estimated at 0.4% of the sales prices (Favot et al., 2018), while in the Netherlands, the costs of various EPR schemes are estimated between 2% and 0.2% of the product sales prices (Vermeulen et al., 2021). However, such estimates are very dependent on how EPR is organised, and which costs are included in the financial responsibility, making it also hard to compare the efficiency of individual and collective setups of EPR (Kunz et al., 2018).

2.4.3 Stimulating Eco-design

EPR has been introduced with the ambition to stimulate the redesign of the entire life cycle of products (Van Rossem et al., 2006). However, there is a clear consensus in the scientific community that the application of EPR has so far hardly stimulated producers to widely apply eco-design or Design for Sustainability (Gottberg et al., 2006; Huisman, 2013; Kautto, 2006; Kemna, 2011; Kunz et al., 2018; Mayers, 2007; Subramanian et al., 2009; Tojo, 2006).

Researchers analyzing motives of producers for applying eco-design have found no direct connection between eco-design practices or improved environmental product performance and the participation in the EPR scheme. Design incentives in EPR are not explicit, while only few of the 'producers' addressed in EPR schemes are manufacturers designing original equipment (OEM) and most are thus not in a position to apply eco-design (Kalimo et al., 2015). Meanwhile, eco-design and Design for Sustainability approaches address more sustainability aspects than energy use, resource use or preparations for recycling (Braungart et al., 2007; Crul et al., 2009; Deutz et al., 2013; Ramani et al., 2010), and some approaches are especially designed to address aspects of circularity (Mudgal et al., 2013). While eco-design approaches have to some extent been adopted in industry, the main drivers for this are related to strategic positioning and market strategies connected to certification and reporting, while research on eco-design implementation does not show evidence of the influence of EPR schemes (Albino et al., 2009; Gottberg et al., 2006; Richter and Koppejan, 2016; Rossi et al., 2016).

One of the ways to strengthen the eco-design incentives in EPR schemes, is by introducing fee modulation related to the sustainability performance of products (Vermeulen et al., 2021). Examples of this can be found in Canada (Leclerc and Badami, 2020), France, Italy and Germany, while the EU is moving towards further application of such fee modulations (Hogg et al., 2020).

2.4.4 General remarks

We noted in section 2.2.2, that EPR was introduced as a form of self-regulation with delegated responsibilities. Such delegation has enabled producers to set up systems in product-, culture- and country -specific ways, adapting to the specifics points of distribution and collection. These are very different for cars, tyres, packaging, or batteries. In the same way assessing its success depends on data available about sales volumes, average use times and stockpiling behaviour of consumers, as with batteries or old electronic equipment, as well as second-hand sales practices, which may include cross border movement, e.g., cars. Such product-specific contextualizing has advantages, but also results in a widely diverse diffusion of EPR practices in time and across the world.

Copying models from specific policy cultures (like in Northwest Europe), may not always work well. Consequently, we see a wide practice of adapting EPR models to national policy cultures and contexts. This also includes the different ways of combining it with other policy

measures, e.g. landfill bans and landfill taxation (Niza et al., 2014). Some have suggested solving these issues of confusion and competing or even counterproductive diversity by developing global EPR systems. These have been proposed for plastics (Raubenheimer and Urho, 2020), WEEE (Ongondo et al., 2011), or for specific strategic materials, e.g. platinum, that are often lost in product orientated EPRs (Wilts et al., 2011).

Such global harmonisation, however desired it might be, will most likely not be achieved in the short run. But with successes as described in organizing take-back and recycling of a variety of product categories, the challenge is to take lessons from the successful practices and apply them widely, while also looking for opportunities to strengthen where weaknesses are spotted. Here we see a role for the OECD and UNEP.

2.5 The future of EPR and circular economy?

EPR will continue to be a crucial policy instrument in promoting recycling. At places where it was implemented successfully, it efficiently organised infrastructures for take-back and recycling.

In recent years many countries around the world have transformed traditional waste management policies into circular economy policies. Crucial in this context are:

1) a stronger emphasis on extension of the lifetime of products in use;
2) applying far more options of value retention than the recycling of mixed streams of post-consumer waste; and
3) a stronger emphasis on producers developing new circular business models (Bocken et al., 2016).

The old 3R's waste hierarchy (like Refuse, Reduce, Recycle) has been replaced by the more detailed 10R's hierarchy, distinguishing short loops (R0 refuse, R1 Reduce, R2 Resell/reuse), middle-long loops (R3 Repair, R4 Refurbish, R5 Remanufacture, R6 Re-purpose) and long loops (R7 Recycle materials, R8 Recover energy, R9 Re-mine) (Reike et al., 2018).

EPR systems so far have organised R7 and R8 at low costs and have delegated responsibility to (end-)producers, importers and retailers selling new products. The new circular policy framing requires recognition of the roles of other (circular) value chain actors. Extending life-time through repairing and reselling is often in the hands of social enterprises which need to be included in EPR organisations (Bahers and Kim, 2018). Making EPR a better fit in the pursue of circular economy requires a more explicit link to rewarding eco-design improvements of products, rewarding the application of recycled materials, and strengthening the market for recycled materials. These may very well need the application of a broader and more comprehensive set of policy instruments, such as taxes on virgin materials and the promotion of circular business models.

However, these adjustments still assume continued and unlimited consumption. The R0 Refuse refers to overconsumption and issues that go beyond the interest of producers. An example of this is 'planned obsolescence' in products, which benefits both producers and retailers, and puts pressure on the consumer to continue consuming. Instruments such as EPR fail to address issues of waste prevention and decreased consumption (Bartl, 2014). EPR systems will not likely be able to address these questions of consumption. It will need to be complemented with other instruments, promoting a more structural transformation of our economies (Calisto Friant et al., 2020).

3

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The circular economy and cascading: towards a framework

Abstract

The principle of cascading, the sequential and consecutive use of resources, is a potential method to create added value in circular economy (CE) practices. Despite conceptual similarities, no research to date has explored how cascading has been operationalised and how to integrate it with CE R-imperatives (Reduce, Reuse etc.) to facilitate implementation practices. CE practices emphasise value creation and retention, yet, there has been little reflexive examination of explicit and intrinsic value considerations; namely, how allocation choices, i.e. the decision-making process, for resource utilization are made. This chapter aims to (1) examine how cascading has been operationalised (empirically and theoretically) to understand its normative underpinnings and value considerations; and (2) integrate cascading with the CE practices in a manner that accounts for the complexities of material allocation choices. Through a literature review of 64 articles from three bodies of literature (CE, cascading and up/downcycling), plus additional material on sustainable development, we show the cascading concept is a suitable framework to direct material uses and provides an overarching concept to integrate with CE R-imperatives. From this, we propose a new theoretical framework that considers the socio-organisational necessities for a CE-cascading system, specifically by deconstructing the allocation choices and exchanges of product material combinations between actor groups. This considers a dual perspective of the physical aspects of materials and the social context in which material allocation is made. The framework transcends individual value chain actor configurations to propose an overarching steering/governance framework, based on the triple-P of sustainability (People, Planet, Prosperity), to examine and direct CE-cascading exchanges, between and above individual users/firms.

3.1 Introduction

In response to numerous interrelated socio-environmental challenges, the circular economy (CE) – while not completely new – is being embraced as a means of realising sustainable development (Geissdoerfer et al., 2017). The concept has found its champions in governments, policymakers, scholars and businesses who call for a departure from the current linear-like economy, i.e. an economy where resources are extracted, processed and wasted, to a closed-loop system, which prioritises value retention and regenerative design (Blomsma and Brennan, 2017; Ellen MacArthur Foundation, 2013).

CE draws its influence from various disciplinary backgrounds, including industrial ecology (IE) (cf. Blomsma and Brennan, 2017), which has provided many theoretical and methodological tools used (Saavedra et al., 2018). The underlying purpose of adopting CE practices is (presumed to) ultimately reduce virgin material consumption, eliminate waste and decouple growth from material use (Ghisellini et al., 2016; Murray et al., 2017). Notwithstanding the longer lineage of CE (Blomsma and Brennan, 2017), or its multiple definitions (Kirchherr et al., 2017), the implementation of CE is being pursued through utilising the so-called R-imperatives or strategies. The number and sequence these R-imperatives is inconsistent and has evolved. A older framing presented the 3Rs (Reduce, Reuse and Recycle), whilst a recent synthesis outlined 10R-value retention options that can be initiated by consumers and businesses throughout the entire value chain of a product (Reike et al., 2018).

In recent CE publications, the principle of cascading is mentioned as a method of retaining the 'added value' of materials as long as possible (Bezama, 2016; Mair and Stern, 2017; Gontard et al., 2018; Lüdeke-Freund et al., 2018). Cascading is understood as the sequential use of resources for different purposes, usually (or ideally) through multiple material (re) use phases before energy extraction/recovery operations. It is most established within IE (Olsson et al., 2018; Teuber et al., 2016). Like CE, cascading is concerned with resource efficiency through promoting consecutive resource circulation; however, the concept is often conflated with recycling or downcycling (Blomsma and Brennan, 2017). Proponents of cascading contend it contributes to higher natural resource efficiency over the entire material lifecycle; from resource extraction, product consumption to disposal (Sirkin and ten Houten, 1994). In practice, cascading is predominantly studied within the lumber industry, concerning exchanges between lumber, paper and energy companies (cf. Korhonen and Niutanen, 2003; Sathre and Gustavsson, 2006; Mehr et al., 2018; Jarre et al., 2019). Research on cascading in the context of CE has focused on its possibility to use waste by-products (Venkata Mohan et al., 2016; Egelyng et al., 2018; Zabaniotou and Kamaterou, 2019), secondary textile use (Fischer and Pascucci, 2017) and wood (Bais-Moleman et al., 2018; Husgafvel et al., 2018). Of these authors mentioned, most only tacitly reference cascading without thoroughly detailing how the cascading principle is operationalised in order to facilitate CE decision-making processes and activities.

Two key articles have begun examining the interconnection between cascading and CE. Olsson et al. (2018) provide a historical review and critical examination of 'imposed' material hierarchies for cascading wood use over energy recovery. They argue that prescribing static hierarchies creates the high risk of unwarranted consequences; instead, cascading processes should emerge bottom-up, with cascading "treated as a guiding principle or tool – not an end in itself" (Olsson et al., 2018, p. 8). Moreover, Mair and Stern (2017, p. 291) reviewed the conceptual interlinkages between cascading and CE. They concluded that the former "perfectly" fits into the latter, but the lack of integration between them is likely due to divergent research communities. Thus, Mair and Stern (2017) recommend actively integrating cascading with CE, potentially as a communication tool to describe specific CE processes. Nevertheless, despite these reviews, there is a lack of knowledge that illustrates the practical interlinkages between cascading and CE practices. No author, to our knowledge, has integrated cascading with the CE R-imperatives. Furthermore, little is known about how cascading can be operationalised for practitioners and connected to value creation and retention in a CE.

This suggests the need to connect CE and cascading in a manner that can analyse and facilitate CE decision-making and implementation processes. CE has a long-established theoretical lineage (cf. Ghisellini et al., 2016; Saavedra et al., 2018). In light of this, there is a precedent to thoroughly examine the existing knowledge base to provide greater insight and support for the present and future CE-cascading developments. Therefore, this chapter aims to review the existing literature on CE and cascading. The purpose is to examine how cascading has been operationalised in a theoretical and empirical sense to understand its normative underpinnings and value considerations including higher use options (up/downcycling). Based on this review, we propose a new framework that integrates cascading with the CE R-imperatives whilst accounting for the social complexities of decision-making processes.

This chapter is structured as follows. Section 3.2 describes the methodology. Section 3.3 provides an overview of CE practices and cascading, including historical origins, frameworks and empirical case studies, illustrating how it has been operationalised (empirically and theoretically). Section 3.4 deconstructs the value assumptions within CE and cascading processes, reflecting on the terms upcycling and downcycling to understand how these practices determine the innate value of material exchanges. Section 3.5, for the first time, integrates cascading with CE R-imperatives in a new framework. Section 3.6 discusses and concludes illustrating the applicability for the proposed framework for CE decision-makers.

3.2 Methodological approach

To explore the gaps outlined in Section 3.1, this chapter conducted a critical review of three bodies of literature: CE, cascading and up/downcycling. There are various types of literature

reviews, each with their own attributes and limitations (see Grant and Booth, 2009). A critical review goes beyond a mere topic description and is useful to identify significant items in a field, synthesise knowledge and derive new theories or models (Grant and Booth, 2009). This review was comprised of two layers of analysis and four distinct steps (labelled a, b, c and d), each step developing insights for the subsequent analysis (see Figure 3.1). Layer one consists of an overview and description of the key concepts: CE, cascading (step a and b, Section 3.3) and up/downcycling (step c, Section 3.4). Layer two consists of a critical interpretation and synthesis of these bodies of literature, to integrate CE and cascading (step d, Section 3.5). For this final step we also incorporated insights from the sustainable development literature. All of the selected literature for this study are listed in the supplementary materials of Campbell-Johnston et al. (2020b).

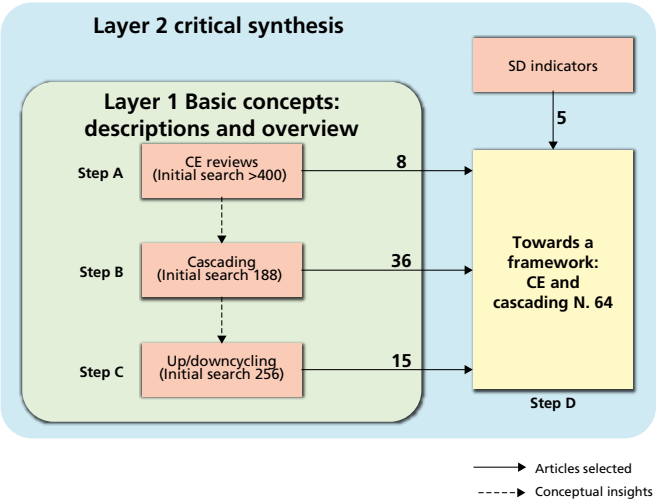

FIGURE 3.1 Research process of chapter 3.

3.2.1 Layer one: descriptive overview

The first step (a) consisted of an overview of recent CE literature. Many in-depth reviews have been written on this exact topic, outlining its history, contestations and theoretical diversity. As a short-cut (and to avoid repetition of those studies), we searched for key reviews since 2015, which address conceptual diversity and operationalisation of the CE. Of the 400 potential articles in Scopus, most are not in-depth reviews. Thus, we chose eight key articles (supplementary materials in Campbell-Johnston et al. (2020b)), with high citation counts and in-depth conceptualisation of the concept of CE, implementation practices and operationalisation. These articles were analysed to provide a brief description of the operationalisation of CE R-imperatives (Section 3.1). For these reviews, we adopted the 10R framing of CE as outlined by Reike et al. (2018).

Following this overview, we initiated (step b) string searches (see supplementary materials of Campbell-Johnston et al. (2020b)) for articles on cascading and its connection to CE. We set our timeframe from the 1990s, as that is the commonly accepted date for when the term CE first appeared (Blomsma and Brennan, 2017). Little has been written on the connection between these two bodies of literature (Figure 3.2), giving further credence for our study. We selected 30 of 188 articles, by scanning abstracts and keywords for their relation to historical overview, conceptualisation, links to CE, operationalisation and detailed case studies. Four articles were added by searching through the references of selected articles. Two additional articles (Mantau, 2012; Vis et al., 2016) were recommended during the review process, which we added.

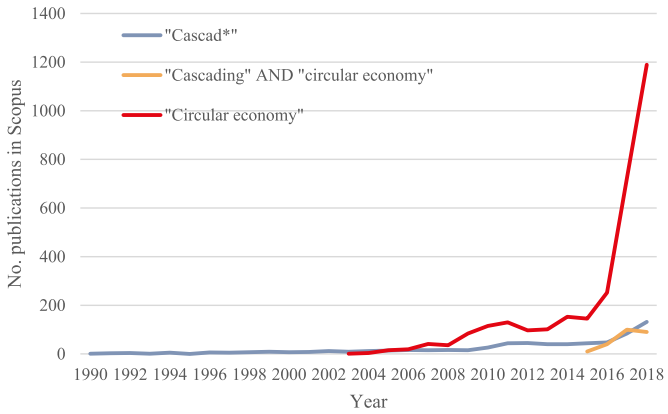

FIGURE 3.2 Total scientific publications in Scopus that cite cascading and circular economy, 1990 – 2018.[5]

To analyse the literature concerned only with cascading we constructed an overview of the history and development of the concept (Section 3.3.2), including the environmental and material benefits and complexities of implementing it (Section 3.3.2.1) and its connection to CE (Section 3.3.2.2). Next, we described the operational and theoretical frameworks for implementing cascading that already exist (Section 3.3.3). A key question of this research concerned the empirical operationalisation of cascading systems. Cases found during this search were coded according to key attributes observed in the CE literature. This included coding which products/materials were cascaded, the number of cascades in the process, the operational scale and exchanges between the actors. The operational scale of cascading activities concerns either the micro, meso and macro (see Ghisellini et al., 2016). Exchanges were divided to describe the specific sectors involved, whether they (material and energy) took place between different sectors or within the same sector, and the type of exchanges, e.g. business-to-business (B2B), business-to-consumer (B2C) or consumer-to-consumer

5 "Cascading AND circular economy" numbers (orange) are multiplied by a factor of 10 for readability. For relative numbers see supplementary materials of Campbell-Johnston et al. (2020b).

(C2C), an important feature as outlined by Reike et al., (2018). We further classified these cascading chains according to the CE value retentions options based on the framework of Reike et al., (2018) (see Table 3.1). We excluded examples that deal solely with energy and water cascading (eco-industrial parks), instead, focusing on studies that included product and material cascading.

A key issue that derived from this initial analysis was the contextual application and allocation, i.e. the decision-making process regarding materials and the perceived value of it. The terms up/downcycling were closely tied to this process. Thus, we initiated (step c) more string searches for these two terms (supplementary materials of Campbell-Johnston et al. (2020b)) and selected 15 of 256 potential articles. The selection criteria consisted of choosing examples where the above terms were used to describe a specific material use and justification for doing so. Articles were analysed for the context in which they were used and justification, in terms of value, for doing so (see Section 3.4). In all topics, we deemed we had reached saturation when we encountered similar perspectives within the text.

3.2.2 Layer two: critical synthesis

The second layer of analysis consisted of a critical interpretation of all three bodies of literature (CE, cascading and up/downcycling), to ultimately integrate CE and cascading into one framework (Section 3.5). For this step, we reengaged with the broader literature on sustainability (step d) to provide insights for broader social and contextual ambitions; concerns that emerged during the initial analysis, which are not generally explored in CE (Kirchherr et al. 2017). In total, 64 articles were reviewed and analysed for this study. This analysis is based on a critical interpretation of the literature, to outline research issues and develop new theoretical insights. The emphasis is on a conceptual contribution. Whilst this is an interpretivist approach, the outcome is the starting point for future exploration. We acknowledge the subjectivity on the choice over which articles were included and which were not. However, the sparse number of articles on this subject and inter-author agreement on the selection gives us confidence in the thoroughness of our sample.

3.3 Circular economy and cascading

3.3.1 Circular economy operationalisation

CE encompasses multiple production and consumption strategies, which crucially can operate in varying forms and at different scales. Different approaches are pursued at the micro (company or consumer level), meso (eco-industrial parks) and macro (nations, regions, provinces and cities) (Ghisellini et al., 2016, p. 12; Murray et al., 2017)

CE practices are described as following the R-imperatives, sometimes referred to as R-hierarchies or strategies (Reike et al., 2018). There are various numbers and sequences of Rs, which normally relate to product value retention options. For our analysis, we adopt a synthesis of 69Rs into a 10R typology outlined by (Reike et al., 2018) (Table 3.1). Such Rs represent value retention strategies that occur B2B (business-to-business), B2C (business-to-consumer) or C2C (consumer-to-consumer). Yet, whilst such 'R-hierarchies' are commonly discussed, this does not mandate a prescriptive set of 'R-interventions' within a material or product lifecycle, merely a set of value retention options that can be initiated to derive added/additional value. Indeed, indefinite and perpetual recyclability is not thermodynamically feasible (Korhonen et al., 2018a), nor, in the case of material recycling, always environmentally and economically beneficial when a cut-off point for these benefits is reached (Ghisellini et al., 2016). Thus, some broader considerations and trade-offs complicate emerging CE approaches that must be considered when implementing R-strategies.

TABLE 3.1 R0 → R9 Hierarchy of CE options (Reike, Vermeulen and Witjes, 2018)

R-imperative	Description
R0 Refuse	For consumers to buy less. Also for producers who can refuse to use specific materials or designs.
R1 Reduce	Linked to producers, stressing the importance of concept and design cycle, e.g. less material per unit of production (dematerialisation).
R2 Resell, reuse	Second consumer of a product that hardly needs any adaptation and works as good as new.
R3 Repair	Bringing back into working order, by replacing items after minor defects. This can be done peer-to-peer or people in the vicinity.
R4 Refurbish	Referring to large multi-component product remains intact while components are re-placed, resulting in an overall upgrade of the product.
R5 Remanufacture	The full structure of a multi-component product is disassembled, checked, cleaned and when necessary replaced or repaired in an industrial process.
R6 Re-purpose	Popular in industrial design and artistic communities. By reusing discarded goods or components adapted for another function, the material gets a new life.
R7 Recycling	Processing of mixed streams of post-consumer products or post-consumer waste streams, including shredding, melting and other processes to capture (nearly) pure materials. Materials do not maintain any of their product structure and can be re-applied anywhere. Primary recycling occurs B2B, whereas secondary recycling takes place post municipal collection.
R8 Recovery (energy)	Capturing energy embodied in waste, linking it to incineration in combination with producing energy.
R9 Re-mine	Capturing resources from old or existing landfills or dumpsites

3.3.2 Cascading historical overview

According to Sirkin and ten Houten (1994, p. 215), a cascade chain can be described using the analogy of a "river flowing over a sequence of plateaus", where water falls from one level to the next, dissipating energy and matter into other forms until it reaches equilibrium at

the lowest level. This depiction idealises the theoretical vision for any potential resource exploitation – at a specific time – to the point of equilibrium and represents the most seminal and detailed elaboration of the cascading principle (Sirkin and ten Houten, 1994). In recent years, authors have assigned the origin of material cascading primarily to the biomass domain (Kalverkamp et al., 2017). However, cascade chains have a historical association with developing interconnected food, energy and nutrient chains and IE (Olsson et al., 2018).

In practical terms, cascading is seen in IE applications, most noticeably in eco-industrial parks. In such arrangements, formerly separate industries (re)organise and become engaged in multiple interplays of resource and by-product exchanges. Such industrial symbiosis arrangements between firms emerge either in a prescriptive planned or spontaneous arrangements (Ghisellini et al., 2016). The most famous (and referenced) example of the latter (spontaneous) is the eco-industrial park in Kalundborg, Denmark. Here, various separate industries, e.g. power plant, an oil refinery, a biotech and pharmaceutical company, a producer of plasterboard, and a soil remediation company engage in B2B cascading of water/steam and residual energy (Chertow and Ehrenfeld, 2012; Jacobsen, 2006).

IE research engages with the organised recycling of low-quality materials, often discarded consumer items, which is known as cascade recycling (Graedel and Allenby, 2003). Proponents in the 1990s, conceptualised 'hierarchies' of material use of products post-use, specifying 'higher' uses of secondary materials was desirable. From a policy perspective, this was evident in the Lansink Ladder in the Netherlands, which promotes recycling over incineration and landfill (Lansink and Veld, 2010). From an organisational perspective, IE has proposed 'preferable' material recovery options, e.g. in tyres, which includes retreading, engineering applications, granulation and energy recovery options (Ayres and Ayres, 1996). Cascading strategies have similarly been connected to regional self-sufficiency, where material 'throughput' relies on replacing imported non-renewables by cascading 'roundput' flows that relies on regional wastes and renewables (Niutanen and Korhonen, 2003). Much of this early work focused on technological feasibility, overlooking the importance of the complexities of societal organisations, which can complicate the implementation and success of IE (Vermeulen, 2006); although such contextual complexities, such as existing regulatory frameworks, have subsequently received attention (Deutz et al., 2017).

3.3.2.1 Cascading and policy: benefits and complexities

Research on cascading wood has illustrated the material and environmental benefits that can result from replacing fossil fuels whilst conserving forest stock (Mantau, 2012; Suter et al., 2017). A study of the forest industry in Switzerland showed the need for a systems perspective to weigh the substitution and cascading effects. Following a supply chain perspective, Bais-Moleman et al. (2018) compared two cascading scenarios of wood use demonstrating the potential GHG emissions could be reduced by 42% and 52%. Mehr et al. (2018) modelled 200-year horizon of wood cascading compared to immediate incineration of wood, concluding there is high climate mitigation potential. Similarly, Garcia and Hora

(2017) discuss the German Renewable Energy Act, which promotes the cascading of untreated or only mechanically treated wood; they argue that peak availability, competing market demands, collection logistics and the location of recycling facilities are crucial parameters that must be considered to promote non-fuel uses. Although there are material benefits from cascading, uncertainty exists over the number of cascade steps (reuse, recycling etc.) and their environmental impacts; which are affected by the subsequent application and alternative material substitution (Höglmeier et al., 2017). Whilst the studies mentioned have modelled the potential benefits, less research has examined the social context for facilitating it; namely, the policy implications and key mechanisms that direct decision-making for a cascading process. The focus of the cascading studies mentioned here, reflect the priorities of European Union towards carbon mitigation.

Cascading is often associated with consecutive utilization bio-materials. Olsson et al. (2018) provide a comprehensive overview of policies for cascading wood. From the late 1990s, cascading was connected to improving the material efficiency of wood consumption and recycling practices (see Lafleur and Fraanje, 1997). Early cascading frameworks stressed the need to develop cross-sectoral policy structures to alleviate the competition risks between different end-users (Haberl and Geissler, 2000; Olsson et al., 2018). This issue arose as a consequence of increased demand for bioenergy in the European Union in the 2000s, where cascading remerged as a model to reconcile these competing demands and contribute to mitigating climate change (see Brunet-Navarro et al., 2018). Keegan et al. (2013) built on this on-going energy vs. materials debate in the context of biomass, arguing for supply chain logistics that facilitate reuse, integrated sectoral decision-making and a policy framework geared towards the production of bioenergy. Nevertheless, critical issues concerning cascading and bio-materials include the quality of materials, various market barriers (e.g. competition with upstream materials), and policy issues between different sectors have remained (Vis et al., 2016). This indicates the strategic and competing considerations ingrained within cascading decision-making processes, particularly competing sectoral and policy demands.

Indeed, as Olsson et al. (2018) argued, policies which impose prescriptive material hierarchies to achieve greater levels of cascading are challenging to implement and can cause competing demands from actors if a certain process is prescribed as more economically valuable than another. The interdependencies between actors in a cascading process and the potentially unequal benefit sharing provide additional complications (Vis et al., 2016). This raises a question of what the underlying purpose is for pursuing a cascading approach, with examples being extracting the maximum value, increasing the circulation time of materials, or mitigating environmental burdens (Olsson et al., 2018). Whether the economic and energy aspects of cascading outweigh the material circulation benefits remains an open issue (Vis et al., 2016). Whilst Olsson et al. (2018) touched on the social challenges of using cascading, they do not establish exactly what the conditions or innate decision-making contexts are in order to successfully implement and realise a cascading system.

Thus, Olsson et al. (2018) suggest that such systems should emerge organically, instead of imposed through "politically determined hierarchies". Yet, there is limited research into the specific contexts in which such successful outcomes have emerged.

3.3.2.2 Circular economy and cascading

Recently, interests in cascading bio-based materials have also been interrelated with the emerging discussion on CE. A popular imagine of CE, as proposed by UK consultancy the Ellen MacArthur Foundation (2013), shows a technical and biological cycle of materials, with cascading presented as a key concept in the latter cycle. Subsequently, questions have been posed over what the bio-economy can learn from cascading successes (Jarre et al., 2019; Mair and Stern, 2017). The notion of a circular bio-economy has been ascribed to the cascading and valorisation of bio-based wastes in bio-refinery processes (Venkata Mohan et al., 2016; Zabaniotou and Kamaterou, 2019). Additional studies have connected these concepts in the context of utilizing co-waste streams from agriculture, fisheries and poultry (Egelyng et al., 2018) and secondary wood streams (Husgafvel et al., 2018).

Examples of CE and cascading discussed in the literature includes product-service systems in the Dutch textile industry (Fischer and Pascucci, 2017), secondary construction and demolition streams (Husgafvel et al., 2018), end-of-life product management (Kalverkamp et al, 2017) and cascading as a CE action for new businesses models (Lüdeke-Freund et al., 2018). Most noticeably, Mair and Stern (2017) reviewed the conceptual interlinkages between CE and cascading; calling for both concepts to be combined, as both are concerned with extending the use of products/materials. However, they do not explicitly show how these concepts can be practically combined. Thus, we propose using cascading as a more fundamental concept that goes beyond biological nutrient cycling.

3.3.3 Cascading frameworks

In their original cascading framework, Sirkin and ten Houten (1994, p. 215-16) presented it as a design tool "meant to be applicable, in general, for the utilization of all resources". Resource cascading is determined through the interaction between two sets of theoretical entities: (1) a dimensional model concerning resource economy and (2) principles that modify them. Both these entities together guide a cascading approach. The dimensional model of resource economy contains four sup-elements: Resource quality, defined as an expression of the capacity to perform various tasks, or denoting its potential functionality, i.e. embedded energy, or structural organisation and chemical composition; Utilization time (Figure 3.3), which is the timespan and together with resource quality stipulate the higher the material quality, the better its potential to perform demanding tasks; Salvageability, the resource quantities of a material that can be recirculated into secondary closed-loop cascades or alternative chains; and Consumption rate, how the present consumption rate will affect future stocks.

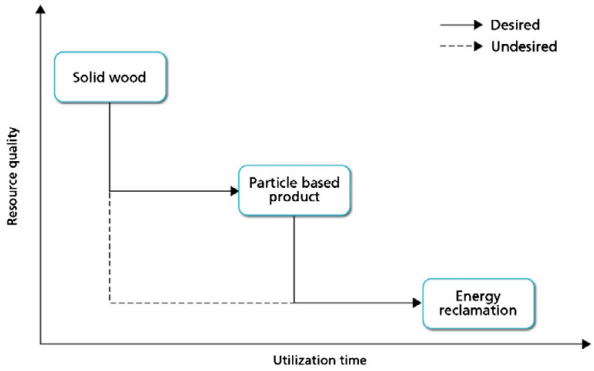

FIGURE 3.3 Basic cascading example based on Sirkin and ten Houten (1994) (own creation).

Sirkin and ten Houten (1994) modify the above resource economy dimensions with the following four principles:

- **Appropriate fit**: qualities of the utilised resource to match the scope and demand level of the task to be performed. Low-quality tasks should not be done with high-quality materials, i.e. primary plastics should not be used for a task that recycled plastics can perform, e.g. shampoo bottles;
- **Augmentation**: increasing the resource utilization time by counteracting decline (repair) and developing systems to extend lifespan;
- **Consecutive relinking**: determining the optimal and highest value pathway for materials, including into alternative value chains;
- **Balancing resource metabolism**: establishing a balance between the rate of resource consumption and the rate of resource extraction. This dimension seeks to incorporate the importance of inter-generational thinking within material uses and product cycles.

This framework has been adopted or modified in subsequent research. Kim et al., (1997) examined the allocation burdens in the life cycle of a cascading recycling system. They assert that a cascading recycling framework should consider quality degradation (e.g. appropriate fit) and environmental pressures in assigning materials, and proposed a method that accounted for material quality in each lifecycle system (e.g. consecutive relinking). Lafleur and Fraanje (1997) outline a six-step methodology to achieve more sustainable use of primary wood, arguing that cascading is an essential step for sustainability. This involves an input-output analysis of primary wood, reducing the (end) use of wood-derived products, determining the appropriate fit (by applying resources to highest quality products), cascading, increasing efficiency processes and finally evaluating the process (Lafleur and Fraanje, 1997). Mellor et al. (2002) developed an acceptance criterion for extended producer responsibility organisations using waste polymers, to determine the potential utility of said waste in different applications.

All the above frameworks provide little indication of how the decision-making process should be carried out and the context of where cascading operations materialise, i.e. the socio-governmental context (cf. Vis et al., 2016). A cascading process includes multiple use phases to (ideally) maximise the highest value of the product or material. Yet, this requires multiple sets of actors in the value chain, which raises the question of *how* these processes should be governed to assure that the appropriate fit and subsequent use considers the other aspects of cascading, e.g. balancing resource metabolism, without compromising on other indicators, e.g. energy use. Sirkin and Houten (1994) describe this as a problem of product design, requiring both a resource management policy aimed at sustainability and incentives for designing for cascading that promote resource quality, not market value. Instead, we propose that a cascading process must be understood from a dual perspective. First, through the physical dynamics of sequential material use, and second, through the social dynamics of the individual actors embedded within the broader societal system (Vermeulen and Witjes, 2016). This social dimension highlights a significant challenge for instigating a cascading process, particularly concerning the decision-making, e.g. regulatory and market context, and the mechanisms that determine the appropriate fit of materials.

3.3.4 Cascading case studies

Recent articles call for learning from cascading (Jarre et al., 2019) and integrating it with CE (Mair and Stern, 2017). However, a comprehensive CE-cascading framework to examine and facilitate decision-making is lacking, particularly one that considers 'consecutive relinking', i.e. determining the highest value pathway for a material. Therefore, to integrate cascading and CE, understanding how cascading has been empirically operationalised is important (Table 3.2).

The results from Table 3.2 show a limited diversity of cases. However, the concept has been explored in various sectors such as textiles, automotive and food processing; still, this analysis confirms previous claims that wood receives the highest attention in conceptual and empirical attention (cf. Mair and Stern, 2017). Thus, cascading not been universally integrated within product/material decision-making processes, from either product design to macro policy approaches across multiple material streams. Except for tyres, all of the cases reviewed focus on material streams, not specific products, indicating the limited focus on detailing product or component cascading.

Cascading operations have primarily taken place on the macro scale, with the exchanges of materials overwhelmingly occurring B2B. This raises questions about geographical proximity and economic conditions on site that allow such exchanges (cf. Vis et al., 2016). The maximum number of direct cascade chain links presented is four (Korhonen, 2001), with a maximum of seven single links (Zabaniotou and Kamaterou, 2019). The majority of these exchanges occur through primary recycling (R7), i.e. B2B exchanges of by-products, which is advantageous over secondary recycling, i.e. mixed collections through municipalities

(Stahel, 2010). Moreover, all R-imperatives are apparent in cascading processes, except R0 (Refuse), R1(Reduce), R6 (Repurpose) and R9 (Re-mine). Whilst not evident within the existing literature, R1 and R6 are still appropriate strategies, in the design phase of product (Reduce) and the potential usage of material (Repurpose). Imperatives R0 (Refuse) and R9 (Re-mine) do not seem applicable once a cascading strategy has been adopted.

There is a noticeable temporal disparity between the number of chain links presented in the studies, with earlier (wood) cases presenting multiple sequential uses (Lafleur and Fraanje, 1997; Korhonen, 2001; Sathre and Gustavsson, 2006; Dodoo et al., 2014) and subsequent cases detailing multiple additional single uses of materials (De Besi and McCormick, 2015; Teuber et al., 2016; Fischer and Pascucci, 2017; Egelyng et al., 2018; Gontard et al., 2018; Husgafvel et al., 2018; Echeverria et al., 2019; Zabaniotou and Kamaterou, 2019). This disparity might reflect the longer practice of cascading wood compared to the more recent emphasis given to waste valorisation within the CE.

The above cases are highly specific, detailed and often technical processes, and highlight the specific opportunities for valorising and utilizing waste. This can involve the cascading of specific products in multiple iterations of use, e.g. through Repair and Reuse activities (Kalverkamp et al., 2017). Yet, understanding cascading processes (in the broadest sense) we propose to think beyond specific materials, to the more encompassing concept of Product/Material Combination (PMC). However, what is lacking is an understanding of how the valorisation of the product or material was undertaken at each stage or chain within the cascading process. Thus, understanding how the highest value pathway is determined within CE and cascading processes is important (consecutive relinking, Section 3.3.3), which necessitates exploring how value is perceived, conceptualised and ascribed.

3.4 Up/downcycling and value considerations

CE activities can preserve or derive added value from materials and products while cascading involves the sequential use of resources. Both processes involve context-dependent valorisation of a PMC and decision of its subsequent application. This section explores the terms 'upcycling' and 'downcycling', which are used to describe the valorisation of a specific material process, to illustrate how the allocation, i.e. the processes of assigning their use, is categorised.

3.4.1 Definitions and examples

Downcycling is recycling "something in such a way that the resulting product is of lower value than the original item" (Ortego et al., 2018a, p. 25). Here, downcycling concerns value or purpose lost in comparison to the original item, which indicates a loss of material/product functionality due to quality. Downcycling is usually attributed to describe a product's

TABLE 3.2 Examples of the cascading principle in the scientific literature

Cascaded material(s)	Scale	Exchanges		
		Sectors	**Within/between**	**B2B, B2C or C2C**
Wood	Meso	Forestry, paper mill, pulp mill and energy generation	Between	B2B
Recovered wood	Macro	Forestry, particleboard, building sector, energy generation	Between	B2B
Wood	Unspecified	Forestry, wood processing, energy generation.	Between	Unspecified
Wood	Macro	Forestry, processing industry, consumer use, post-consumer uses	Between	B2B
Wood (recovered)	Macro	Sawing, planning and impregnation, mountable parquet manufacturing, the joinery industry, wood package manufacturing, the furniture industry and the recycling industry	Between	B2B & C2C
Wood polymer composites	Unspecified	Wood sector	Within (wood)	B2B
Textiles	Micro	Textile Retail, Assembly, Manufacturing, Extraction.	Within	B2B & B2C
Textiles	Macro	Textile and building	Between	Unspecified
Foods	Macro	Processing industries (meat/ fish/vegetables) to innovations (chemical processing and consumer goods)	Between	B2B & B2C
Biomass	Unspecified	Unspecified	Unspecified:	Unspecified
Agricultural residues	Macro	Agricultural, anaerobic digestion and biochemical	Between	B2B
Coffee grounds	Micro	Waste collectors and biorefinery	Between	B2B
Tyres	Macro	Automotive and recycling	Between	B2B & B2C

material properties, their level of degradation, or, in the case of metals, if they have become impure, which leads to a loss of economic value (Koffler and Florin, 2013; Stotz et al., 2017; Worrell and Reuter, 2014). For example, Stotz et al. (2017) discuss the process of aluminium recycling, arguing that downcycling occurs when the cycled materials lose their original purity. Material functionality can also be understood in terms of the quality of a material to perform or not perform tasks relative to that of virgin materials; such as the use of recycled polymers in low economic applications due to degradation (La Mantia, 2004). Also, Di Maria et al., (2018) argue that the use of construction and demolition waste in backfilling in many European countries is a low-grade low-value application. Thus, downcycling is commonly ascribed to demarcate the lower physical properties of a material.

Cascading chain links	Value retention options	Citation
Four	Recycling (R7) and Recovery (R8)	(Korhonen, 2001)
Two	Recycling (R7) and Recovery (R8)	(Sathre and Gustavsson, 2006)
Three	Recycling (R7) and Recovery (R8)	(Dodoo, et al., 2014)
Three	Recycling (R7) and Reuse (R2)	(Lafleur and Fraanje, 1997)
Five additional individual chains	Recycling (R7) and Reuse (R2)	(Husgafvel et al., 2018)
Six additional individual chains	Recycling (R7)	(Teuber et al., 2016)
Unspecified	Reuse (R2), Refurbish (R4), Remanufacturing (R5) and Recycling (R7)	(Fischer and Pascucci, 2017)
One additional chain	Recycling (R7)	(Echeverria et al., 2019)
One	Recycling (R7)	(Egelyng et al., 2018)
One additional chain	Reuse (R2), Recycling (R2) and Recovery (R8)	(De Besi and McCormick, 2015)
One additional individual chain	Recycling (R7)	(Gontard et al., 2018)
Seven additional individual chains	Recycling (R7)	(Zabaniotou and Kamaterou, 2019)
Four	Reuse (R2), Repair (R3), Recycling (R7) and Recovery (R8)	(Kalverkamp et al., 2017)

Upcycling involves the conversion of waste material(s) into a more valuable product(s). "It can be purely artistic, scientific, or anything simply useful" (Pol, 2010, p. 4753). Some studies claim upcycling results in products with higher quality and performance than the original, using refurbishing and remanufacturing strategies (Stahel, 2010). However, the literature is inconsistent with what constitutes upcycling. Consequently, we use two descriptive thematic classifications that traverse materials, products and sectors (Table 3.3). Value-added upcycling involves turning wastes into new products, i.e. creating new value (monetary or environmental) from nothing. Extracting higher-value describes how a 'higher-value' use could be obtained by changing the specific material trajectory, thus creating increased value in either monetary terms or environmental performance, e.g. salvaging rare materials.

TABLE 3.3 Varying examples of the use of the terms up/downcycling.

Characterisation	Example (description)	Citation
Value-added	Harvesting silicon from waste sludge as input for high-performance lithium batteries.	(Bao et al., 2015)
	Using agricultural wastes and by-products in anaerobic digestion to produce high-value bioproducts and bioenergy. These wastes are recommended over using arable land to cultivate bioenergy stocks.	(Gontard et al., 2018)
	Using discarded geomembranes applied in fracking, which can be repurposed into pellets used for railroad tires, structural beams and non-containment products.	(Stark et al., 2013)
	Using offcuts from the aerospace industry in the US, previously landfilled, to go into applications including prosthetic feet, skateboards and constructional material.	(Nilakantan and Nutt, 2015)
	Strategies to turn waste plastic into carbon nanotubes post-consumer use.	(Zhuo and Levendis, 2014)
	Turning fish, meat, fruit and vegetable co-streams from the Norwegian food industry as inputs for activities including using second-grade vegetables for smoothies and potato peels for biodegradable plastics in the vegetable (potato) processing industries.	(Egelyng et al., 2018)
Extracting higher value	Using waste paper as the basis for textile fibres, which is justified given the global demand for fabrics and current low-value use of cycled paper.	(Ma et al., 2016)
	Using recycled aggregates as inputs for concrete, instead of backfilling within roads.	(Vandecasteele et al., 2013)
	Substituting natural aggregates, e.g. sand, with crushed and sieved concrete demolition waste.	(Weimann et al., 2003)
	Using collected recycling glass into glass-ceramic lightweight aggregates. These are then employed in construction processes where they have a higher economic value.	(Velis et al., 2014)

3.4.2 Value considerations for allocation choices

The cases in Table 3.3 illustrate the literature is inconsistent on classifying and assigning the up/downcycling characterisation. Extrapolating from this, there is an implicit issue of how to determine a material trajectory (i.e. consecutive relinking) of a material at a specific stage in its lifecycle. No uniform normative criteria exist for governing the appropriate trajectory and determining the appropriate fit, with the literature suggesting these are all materially, geographically and temporally contextual. Implicit in these discussions is the issue of how 'value' should be determined for materials with cascading potential which can determine its trajectory and continued allocated use. Olsson et al. (2018) describe this as a tension between market value and inherent value. Whilst Sirkin and ten Houten (1994, p.221) argue that resource quality refers to its inherent and intrinsic qualities, i.e. "qualities that cannot be altered by the landscape of human interest" (e.g. energy or enthalpy contents) or human

interests, e.g. private economic interests, socio-cultural importance or global environmental significance. Therefore, value contains a physical and interpretive element. Thus, value determination - how 'value' is determined and by whom - is a fundamental issue at the interception between theoretical conceptualisation and practical/fundamental application of the cascading principle. There is a subjective element to how value is ascribed, in addition to market context, which can connect with its inherent or interpreted properties and physical attributes, with this process a factor in the subsequent allocation outcome.

The inevitable loss of material quality has commonly seen materials sequentially utilized in lower grade downcycling applications. Yet, this can partly reflect how a process is labelled, with the term upcycling also being ascribed to degraded materials. This categorisation interconnects with the valorisation process, e.g. the innate perception of that material or product at a specific point in time. We observe several broader mechanisms through which valorisation occurs that more explicitly consider the innate characteristics of a material. These include material quality (Stotz et al., 2017), natural capital replacement, i.e. replacing virgin material in production processes (Di Maria et al., 2018), or thermodynamic rarity, i.e. the exergy cost (kJ) required to extract and process the given material from cradle to gate, and the hypothetical exergy cost required if the given mineral must be restored to its initial conditions of composition in the original mine (Ortego et al., 2018a).

While the debate over the fundamental purpose of the benefits of becoming circular differs between contexts (Section 3.1), there are underlying concerns with waste generation, resource supply and the reduction of virgin material consumption (Murray et al., 2017; European Commission, 2018). Therefore, the implications for integrating cascading with CE requires a reflection on what the appropriate value consideration is, and the underlying purpose of the process, i.e. what are the innate value considerations that drive CE preferences, and whether the allocation choice, e.g. the chosen R-imperative is also preferable over others from an integrated sustainability perspective (see Section 3.5).

3.5 Proposing a framework interlinking CE practices and cascading

The above review outlined the concept of cascading, as a framework that promotes the consecutive and sequential use of materials. This consecutive use contains a dual element: (1) the physical properties and subsequent uses of the product/material and (2) the social context in which decision-making processes occur, i.e. the application context, which contains actors involved in material exchanges, the regulatory and market context in which they operate and eventual value considerations of the material. The application context is neglected in the literature on cascading.

Common visualisations of a cascading process (e.g. Figure 3.3) depict it as a simple sequential process (cf. Sirkin and ten Houten, 1994; Olsson et al., 2018). These imply an automatic transfer of the PMC from one use to the next (focusing solely on original PMC quality),

without reflecting the social complexities (or energy considerations market conditions) that decide what (if any) the subsequent use is. Like cascading, the concept of CE is engaged with measures to close material cycles; yet there is also a contextual question of how allocations choices are directed. We argue CE practices can be integrated within a cascading framework using the 10Rs (Reike et al., 2018). Cascading is a preferable overarching framework as it gives a systems perspective, e.g. intergenerational perspectives and resource constraints contextualised with the individual allocation actions of a PMC, as opposed to specific CE R-imperatives that describe exchanges between specific actors/users.

Figure 3.4 depicts the value chain actor configuration for any PMC, from initial mining, material production, to eventual retailing, consumer use, collections and processing. We use the 10R framework of Reike et al., (2018) to describe the initial PMC flow and potential value retention options within the same actor configuration. Figure 3.4 also includes useful design inputs, e.g. design for recycling. In a previous paper, we outlined two distinct lifecycles: product production and use lifecycle and product concept and design lifecycle (Vermeulen et al., 2018a). Different R-imperatives apply to different actors in the lifecycle. For example, concerning the product production and use, with R0→6 relating to the product and R7→8 relating to the material. Refuse (R0), Reduce (R1), Resell (R2), Repair (R3) and Recycling (R7) are applicable for consumers including product-service system business models. Whilst Resell (R2), Repair (R3), Refurbish (R4), Remanufacture (R5), Recycling (R7), Recover (R8) and Re-mine (R9) for producers, businesses and retailers.

A first exploration of integrating the CE R-imperatives and cascading is shown in Figure 3.5. The figure demonstrates the cascading process for any PMC, from the first combination to the final use phase (n), including energy recovery options (R8) in each phase. We exclude the CE imperative Re-mine (R9) as that occurs post-user, although, the framework applies to subsequent application of the re-mined materials. Once in use, keeping a PMC in its highest quality form is the first value retention consideration (requiring longer-lasting products), which can be pursued through counteracting decline (Augmentation, Section 3.3.2), e.g. through repair (R3), refurbishing (R4) and remanufacturing activities (R5). Or, when a new product is made available which has lower environmental impacts than the one in use (cf. van Nes and Cramer, 2006).

R-imperatives 0→7 are applicable within this original 'closed-loop' value chain. The crucial question concerns the subsequent and sequential material allocation of the PMC, e.g. when the PMC leaves its original user or actor configuration and moves to a new one. We show this by separating different value chain actor configurations, using the CE R-imperatives 2→7 to describe the potential exchanges of the PMC between different users and actors. For example, if the PMC moves to a new user/actor configuration the specific value consideration will result in a particular R-imperative being adopted, e.g. Repair (R3) or Recycling (R7). In this way, we recognise that the subsequent PMC use(s) are - at their essence - a social, geographical and temporally contextual phenomenon; moving between actor configurations (as outlined above) through either B2B, B2C or C2C exchanges.

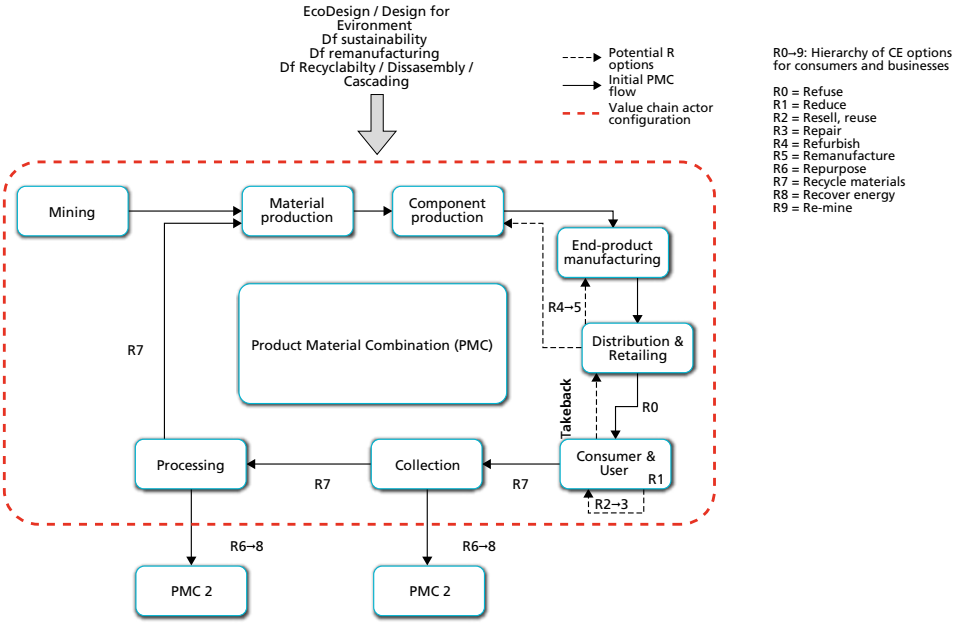

FIGURE 3.4 Actor configuration of a product or material value chain and potential R-strategies.

Central in this depiction is the moment when a PMC moves between users or actor configurations. Here, we suggest, resides the innate value considerations of specific decision-making contexts. This exchange between actors represents the moment which can steer or decide where the subsequent PMC is used or value retention considerations adopted (up/downcycling). Instead of an automatic sequential use, we contend there are likely multiple options or actor configurations and preferences which can be chosen from, all of which relate to the temporal, geographical, economic and innate value choices over the particular PMC. For example, an upcycling process could include turning used plastic bottles into high-quality jumpers, e.g. Patagonia, instead of conventional recycling into pellets. Alternatively, in Northern European waste management systems, a PMC is discarded by consumers, processed by recyclers and finally sold/used in a subsequent value chain.

Whilst the recovery of materials in Europe has been steadily increasing, research has suggested these systems fail to recover critically scarce rare earth metals (Ortego et al., 2018a). There is a question of how such processes are governed and what the current mechanisms and decision-making processes are that occur in these contexts, which facilitate (or not) a CE-cascading process, i.e. an R-imperative with the best sustainability outcomes. In different contexts, for example developing countries, we would perceive different governance contexts and value perceptions, which would alter what the appropriate fit of the PMC is. Such context is lost in the strictly technical debate. We further explore and detail the potential value considerations and decision-making context below (Figure 3.6).

For this, we reviewed a key selection of sustainability reviews (see supplementary materials of Campbell-Johnston et al. (2020b)) and selected the integrated perspective as proposed by Vermeulen (2018).

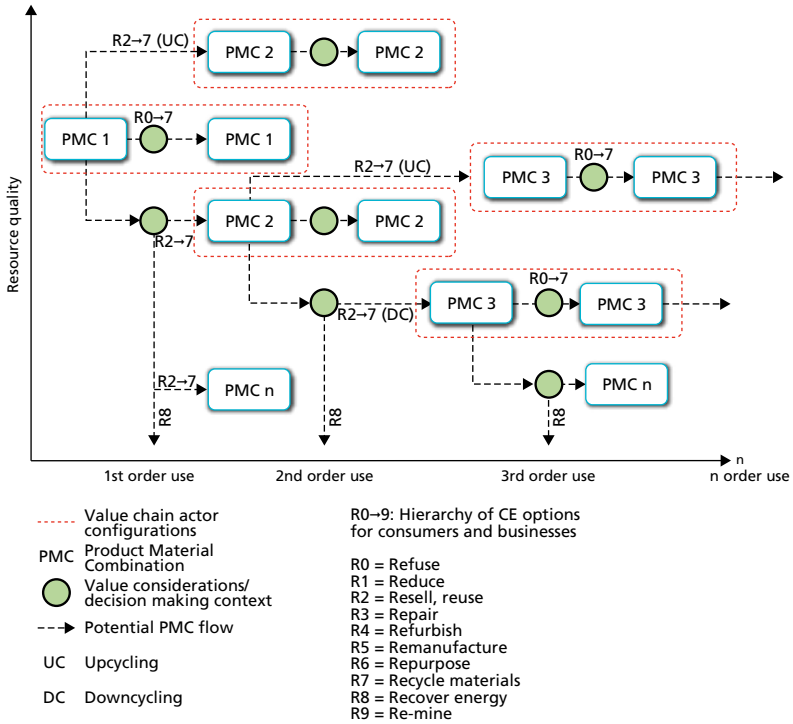

FIGURE 3.5 Product or material cascading in a circular economy.

Such value considerations integrate the original cascading principles as outlined by Sirkin and ten Houten (1994) (Section 3.3.3) with a synthesis of sustainability indicators. We develop three dimensions and broader considerations that should guide every step of the CE-cascading decision-making and valorisation processes. These three dimensions and their explanations are:

- **Monetary value**, guided by market forces, which in principle covers labour costs and energy requirements and the cascading principle of consecutive relinking;
- **Quality and functionality** of the material in question. Understood as it's physical or interpreted properties (see Section 3.3.3) resulting in the salvageability, i.e. the quantities of a PMC that can be circulated into subsequent uses. This is guided by the user requirements (e.g. appropriate fit), which also interconnect with market forces; and
- **The steering framework**, or governance approach, guided by the triple-P (People, Planet, Prosperity), including the cascading principles augmentation and balancing resource metabolism.

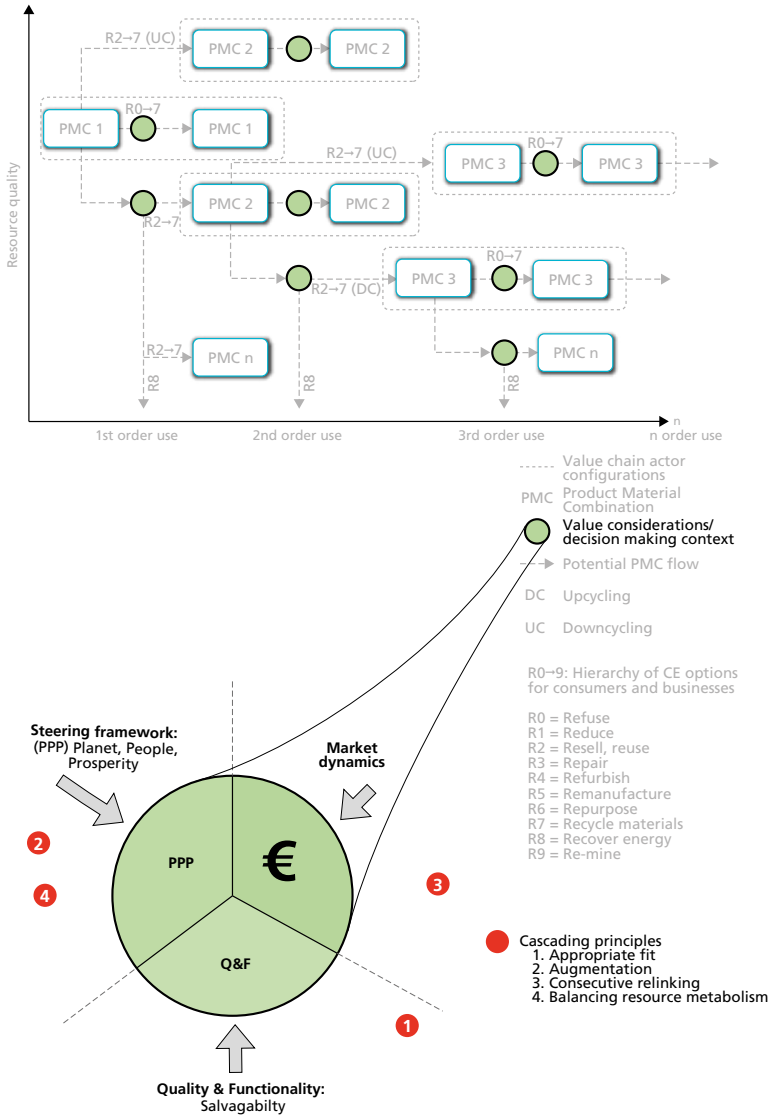

FIGURE 3.6 Value considerations/decision-making context of cascading in a circular economy.

We contend to use cascading, integrated within CE, as a socially contextual concept used to contribute to the broader sustainable development agenda. Two of the above dimensions are easily understood: market dynamics and quality and functionality (see Section 3.3.3). The third, the steering framework, represents the overarching governance approach to promote cascading between and above all actor configurations and is (likely) organised at the macro scale (see Section 3.3.4). This can potentially modify the other dimensions to reach a certain outcome. There are numerous comprehensive works on sustainability that can be used to outline integrated ambitions (cf. Giddings et al., 2002; Gupta and Vegelin, 2016; Mebratu, 1998; Parris and Kates, 2003; Vermeulen, 2019). A recent synthesis of sustainable

development indicators proposes the triple-P (Planet, People, Prosperity), displaced the original triple bottom line (Planet, People Profit), to provide a means of reaching both planetary and human well-being (Vermeulen, 2018). Planet refers to the ecological threats, e.g. resource depletion, land use degradation, climate change etc. related to production and consumption activities. People refer to direct threats to individuals linked to those systems. Prosperity refers to well-functioning social systems, e.g. value-chain actors and socio-economic institutions (Vermeulen, 2018; Vermeulen and Witjes, 2016). This framing accounts for intergenerational justice (time) and displaced impacts (place), and should represent the steering framework for organising a cascading system. This framework goes beyond the immediate fixation of CE, e.g. waste, climate change and resource security to a more holistic conceptualisation with applicable assessment indicators. This synthesis provides the analytical basis for the proposed cascading steering framework which can direct individual exchanges between users and actor configurations.

Using such a framing allows for goal orientation relating to the direct actions, long-term outcomes and the desired end state of a CE and cascading process. This expands conventional CE narratives beyond resource fixation towards more holistic goals, e.g. worker well-being, community livelihoods, etc. This can be realised by specifically steering PMCs towards a specific user or actor group that score well on these goals. Whilst this is currently discussed in more abstract terms, this allows for the development of problem framing, policy development, implementation and evaluation of both macro (nation-state) and micro (a company or firm) involved in existing (or potential) cascading processes. Thus, this provides the basis for practitioners to engage with a more holistic description of higher-value upcycling options regarding the specific allocation choice of a PMC.

3.6 Discussion and conclusion

This chapter reviewed the principle of cascading, to understand its theoretical and empirical underpinning and to integrate it with CE. Cascading is a useful tool to examine and direct product and material exchanges and transfers; through matching a particular product or material to its highest-value use at a specific point in time. We connect the framework of Sirkin and ten Houten (1994) and CE, as operationalised through the 10Rs of Reike et al. (2018), to describe the exchanges or transfers of PMC's between individual value chain actors (Figure 3.4) and between different actor configurations (Figure 3.5). This process contains a dual process (1) the physical properties of the PMC and (2) the social context, the how and where, in which decision decision-making and value (economic or inherent) are ascribed and the preferential allocation realised, and which has so far which is underexplored in the cascading literature.

We argue this decision-making context is one moment to examine and direct the subsequent allocation. The sequential allocation of a PMC represents an exchange or

transfer of a specific PMC combination (within a specific actor configuration) to another group. This requires decision-making to transcend individual PMC or value chain actors, relating to the coordination and decision-making procedures. However, such value chains are likely isolated, meaning this exchange is in the hands of the sending and receiving actors based on the PMC quality, market value and economic conditions. The CE literature of up/downcycling provides an insight into the varied value considerations over material applications which relates to the innate or socially contextual value perception of a PMC. These have primarily been interconnected with the innate physical properties and market demand. Yet, in the cascading literature, this assumes an automatic allocation of such PMC's to the highest functional use. Our analysis indicates the necessity of transcending specific PMC's, or sets of actor configurations, to an overarching level, i.e. the governance structure, which shapes the decision-making context. We build on this further, proposing three dimensions that capture current and potential CE-cascading mechanisms and decision-making contexts. Guiding the steering framework for specific cascading systems (micro to macro) we propose the triple-P (People, Planet, Prosperity) as an analytical basis to analyse existing R-imperative exchanges. This expands CE-cascading systems beyond resource supply and waste generation, to evaluate the social and environmental processes in which they are embedded.

This framework raises further questions about how current allocation choices are steered. In particular, a question emerges of what the existing institutional design for decision-making between and/or above actor configurations is, e.g. extended producer responsibility organisations. Furthermore, the question of governance raises questions not, as yet, detailed in either the cascading or CE literature. Namely, the coordination of, knowledge generation for, and decision-making procedures currently existing in cascading(-like) processes. For example, what the institutional design is for existing allocation choices for materials and what is further needed for materials to reach their highest-value (monetary or otherwise) use? Future research will test the validity of the above framework by examining existing CE-cascading like systems, their spatial, geographical and sectoral contexts, and the value considerations and the decision-making that facilitates these systems.

4

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How circular is your tyre: experiences with extended producer responsibility from a circular economy perspective

Abstract

The circular economy (CE) emphasises closing material loops to retain material value. The current practice of tyre recycling in the Netherlands, through a system of extended producer responsibility (EPR), appears an overwhelming success, with claims of 100% recovery. Yet, there is limited critical understanding regarding the system's circularity, considering alternative value retention options and resource recovery outcomes. This chapter analyses this Dutch tyre EPR system and reflects on how it can be improved from a systemic CE perspective. It uses a qualitative case study approach, including interviews and a review of policy, legal and EPR reporting documents. This chapter assesses the governance of this sector and reflects on the existing system, including its circularity and value retention outcomes. Our analysis reveals seven central issues concerning how the EPR system currently functions, resulting in limited circularity and sustainability outcomes, despite high material recovery levels. To address these issues we recommend the continuous improvement of recovery and sustainability targets beyond a single product life cycle, a more transparent and inclusive governance system, as well as a greater focus on sufficiency strategies, e.g. design for durability and a broader transformation of transport models. This chapter adds a practical understanding of the capacity of EPR to contribute to CE.

4.1 Introduction

National, regional and local governments have recently begun to present the concept of circular economy (CE) as a new pathway to sustainability and economic prosperity. The championing of this inconsistent and contested concept (cf. Korhonen et al., 2018b) comes amid increasing concerns over resource depletion, waste generation and the potential overshoot of planetary boundaries induced by human activities on the biosphere (Henckens et al., 2014; Rockström et al., 2009a). CE is broadly argued to meet these emerging challenges through slowing, closing and narrowing resource loops, i.e. maximising the functional utility of materials and energy (Geissdoerfer et al., 2017; Stahel, 2010). CE theoretically builds upon and goes beyond earlier measures of waste valorisation and cleaner production initiatives to an integrated systems perspective addressing both production and consumption practices (Ellen MacArthur Foundation, 2013; Vermeulen et al., 2018a).

The European Commission (EC) frames CE in conjunction with economic opportunities stating that "[CE] will boost the [European Union] EU's competitiveness by protecting businesses against scarcity of resources and volatile prices, helping to create new business opportunities and innovative, more efficient ways of producing and consuming" (EC, 2013, p. 1). National governments have similarly outlined specific strategies, including the Netherlands, France and Italy; with the Netherlands setting an initial target of 50% less primary material use by 2030 (Ministry of Infrastructure and Environment and Ministry of Economic Affairs, 2016b). Whilst the environmental and economic concerns underpinning CE might be perceived as new, the means through which they are being addressed are manifesting through more conventional or longer-standing organisational practices, including increased recycling targets, waste legislation and extended producer responsibility (EPR) commitments (Milios, 2018).

Scholars have devoted much time to analysing new business models and strategies related to CE (cf. Bocken et al., 2016; Lüdeke-Freund et al., 2018). Yet, there is also a need to reflect and examine these older CE initiatives and practices to understand their suitability and capacity to facilitate and address the emerging societal concerns evidenced within the existing CE debate.

One such system is EPR, which has been collectively and voluntarily adopted in many EU member states for different products, including passenger car tyres (EC, 2014). EU member states are free to choose how to organise the collections and treatment of tyres, which are reported to the European Tyre and Rubber Manufacturers Association (ETRMA); most member states have adopted EPR systems, which have successfully recovered high quantities of used tyres (recovery rates for end-of-life (EoL) tyres in Europe are above 90% since 2007) (ETRMA, 2017; ETRMA, 2018). However, despite such high levels of recovery, there is little direct substitution (closed-loop), i.e. new tyres have a low content of recycled rubber (EC, 2017). Indeed, up to 50% of collected tyres are burned – usually, for energy

recovery (Scott, 2015) – a problem further compounded as natural rubber is a designated critical raw material (EC, 2017). Whilst the technological feasibility of such direct material substitution through devulcanization is being debated and explored (Myhre et al., 2012), there is a broader question about the organisation and performance outcomes of EPR as an older CE system to meet emerging societal challenges.

Previous research on EPR and tyre recycling in the EU have examined the various treatment options (Torretta et al., 2015) and progress across member states, including the steady departure from landfilling. Alternatively, Winternitz et al. (2019a) examined the EPR systems of three European countries, reflecting on their varying policy approaches, successes and potential limitations. Their findings demonstrated that an EPR system does not necessarily guarantee that waste tyres are disposed of in the most environmentally beneficial manner. Similarly, Lonca et al. (2018) examined the trade-offs of increased material circularity of tyres, contracted against other sustainability indicators, e.g. human and ecosystem health. Their research found that increased material circularity is beneficial from a resource perspective, but not necessarily from other environmental perspectives (Lonca et al., 2018). Such research adds to the complexity of organising disposal systems in a dynamic way that accounts for potentially conflicting issues within EoL processes.

Building on these examples, this chapter aims to critically examine the organisation and performance of an existing EPR system, to reflect on its strengths and suitability to deal with the broader needs within the contemporary CE debate. Based on this, we examine the question "how effectively do current ERP systems function from the current ambitions of CE?" We use EPR for tyres in the Netherlands as a case study to explore this question. This chapter, therefore, adds a practical understanding of the contribution of EPR to CE and provides insights for new and existing EPRs globally.

This chapter is structured as follows. First, a literature review of CE, EPR and tyre treatment practices is presented to further contextualize the analysis (Section 4.2[6]). Next, the research methods are presented (Section 4.3). This is followed by a description of the structure and outcomes of the EPR system for tyres in the Netherlands (Section 4.4). Our analysis (Section 4.5) builds on these results, showing the limitations and challenges for EPR systems to lead to a sustainable CE transition before concluding (Section 4.6).

4.2 Literature Review

4.2.1 Circular Economy: Origins, History and Implementation

While the CE concept itself dates back to 1990 (Pearce and Turner, 1990), the idea builds on a long history of literature on resource limits and ecological transformations such as

6 Readers guide: to avoid repetition on what is CE and EPR readers are advised to skip section 4.2.1 and 4.2.2.

the "Limits to Growth" (Meadows et al., 1972), the "Tragedy of the Commons" (Hardin, 1968), the "Economics of the Coming Spaceship Earth" (Boulding, 1966), "Small is Beautiful" (Schumacher, 1973) and "The Closing Circle" (Commoner, 1971).

More recently the CE has drawn its theoretical underpinnings from Industrial Ecology (IE) (Aryes, 1989; Saavedra et al., 2018), cradle-to-cradle (McDonough and Braungart, 2002) and performance economy (Stahel, 2010). The concept of CE is muddled and convoluted but is broadly based on the premise of retaining the functional use of products and materials within the economic sphere as long as possible. It is being advocated, in particular, by private sector consultancies, e.g. the Ellen MacArthur Foundation (UK) and Circle Economy (NL). Estimates suggest the cumulative outcome of earlier CE-policies has resulted in the (re) cycling of as little as 6% of global materials, 12% within the EU27, leading to an increased focus on increasing the value retention of material throughput (Haas et al., 2015).

The CE is also discussed as an evolutionary concept (cf. Blomsma and Brennan, 2017; Reike et al., 2018). Of particular importance for our analysis are the three phases of the CE concept proposed by Reike et al., (2018). First, CE 1.0 (1970 to 1990), is characterised by early waste management practices focused on waste output as an environmental pollution problem to be dealt with through EoL policies. This is when waste treatment and incineration plants started to be developed and operated, especially in the Global North.

The second phase CE 2.0 (1990 to 2010), saw the development of many "win-win" strategies, which make use of waste outputs as valuable resource inputs such as IE (Frosch and Gallopoulos, 1989), Cleaner Production (Fresner, 1998), Industrial Symbiosis (Chertow, 2000), Product-Service System (PSS) (Goedkoop et al., 1999), and EPR (Davis and Wilt, 1994). This is when the concept of CE was first coined by Pearce and Turner (1990) and when associated ideas appeared, such as "biomimicry" (Benyus, 1998), "cradle to cradle" (McDonough and Braungart, 2002), and "performance economy" (Stahel, 2010). This period also saw the widespread implementation of integrated waste management and recycling systems in the Global North, including EPR systems, which mandated new responsibilities for private sector actors (Reike et al., 2018).

The third phase of CE 3.0 (from 2010), when discussions of the concept of CE became more widespread and began to be framed against encroaching societal threats, including planetary limits (Rockström et al., 2009), resource depletion, biodiversity loss, excessive waste generation etc. (Reike et al., 2018). This has led to a more integrated and holistic understanding of material use, which aims to slow, reduce, narrow and close resource cycles in a systemic manner through changes of consumption and production structures and patterns (Reike et al., 2018). However, this is also a period where varying visions of CE are conceived, which are either transformative or reformist depending on their position regarding the capacity for capitalism to overcome resource limits and decouple ecological degradation from economic growth (see Reike et. al., 2018; Friant et. al., 2019).

The implementation of CE-related activities and policies occur in a variety of geographic contexts and scales. CE practices thus range from national programmes, e.g. China's 2009 CE 'Promotion Law' or international policies, e.g. the EU's 2015 CE 'Action Plan' (Ghisellini et al., 2016), to business models and individual company strategies (see Lüdeke-Freund et al., 2018). Scholars have sought to define CE activities through the potential value retention options that can be initiated throughout a product or material lifecycle, commonly described as the R-hierarchy. These range from 3Rs (Reduce, Reuse and Recycle) to iterations from four to ten. A recent review of 69 such R-imperatives outlined a synthesis of 10 comprehensive value retention options, which we adopt as our conceptual framing (Reike et al., 2018) (Table 1.1). Whilst the narrative and framing around CE articulates its "newness", much of the EU policy approach follows or seeks to build upon older CE practices (EC, 2013; cf. Gregson et al., 2015; WFD 2018/851, 2018).

4.2.2 Extended Producer Responsibility

One such older CE practice is EPR, which is defined as "an environmental protection strategy to reach an environmental objective of a decreased total environmental impact from a product, by making the manufacturer of the product responsible for the entire life-cycle of the product and especially for the take-back, recycling and final disposal of the product" (Lindhqvist, 2000, p. 37). Crucially, the concept implies integrating responsibility in the whole product life cycle, where the physical and monetary waste managerial responsibilities (usually assigned to authorities and consumers) are transferred to the product producers.

EPR emerged in the 1990s, building on the experiences of waste managers, recyclers and a policy approach concerned with promoting cleaner production initiatives (Lindhqvist, 2000). Such developments illustrated the more proactive role private sector actors played in these earlier CE systems, giving them greater responsibility for the stewardship of their products. Such 'public-private' configurations represented new steering programmes practised by governments, as opposed to the conventional waste management policy of earlier years (CE 1.0) (Reike et al., 2018; Vermeulen and Weterings, 1997).

EPR builds on the "polluter pays principle", incentivising producers to prevent waste generation, whilst (supposedly) encouraging eco-design and supporting the appropriate EoL processes, e.g. promoting recycling and reusing activities (Deutz, 2009; Ferrão et al., 2008). However, previous studies show EPR activities are overtly focused on EoL activities, negating an integrated lifecycle perspective that pursues continuous improvement and higher environmental performances through, for example, material choices and design for disassembly options (Vermeulen and Weterings, 1997). The EU has mandated responsibility of EoL disposal of vehicles, batteries and accumulators, waste electrical and electronic goods to producers, whilst most member states have additionally implemented a producer responsibility organisation to process used tyres (Deutz, 2009; EC, 2014; ETRMA, 2015). Member states must ensure their EPR schemes have an appropriate collection and accessible schemes.

Alternatively, EPR has also been adopted in various countries in the Global South as a product management tool for EoL tyres (Banguera et al., 2018; Zarei et al., 2018). However, recent studies have illustrated the challenges of adopting EPR in these countries. Such challenges include the limited knowledge of effective practices in Botswana (Mmereki et al., 2019), incentivizing and integrating necessary actors in operations in Colombia (Park et al., 2018), and directly transposing a European policy tool to Brazil (Milanez and Bührs, 2009). Conversely, Cecchin et al.'s (2019) study in Ecuador highlighted the potential of integrating social economy goals with conventional EoL practices associated with EPR.

EoL processing in EPR systems can be organised in various ways. Spicer and Johnson (2004a) outline three approaches to implementation: (1) 'Original Equipment Manufacturer' takeback, where the original producer takes direct responsibility for collecting and processing; (2) 'Pooled Takeback', where responsibly is shared between a consortium of producers, known as the producer responsibly organisation (PRO), usually organised by a product category code, e.g. tyres; and (3) 'Product Responsibility Providers' (PRP), where a private third-party is contracted by the PRO and assumes EoL responsibly for the product on their behalf. This (theoretically) results in dual benefits for manufacturers and the general public, including, eliminating the financial risk associated with complex EoL processing activities (recycling, incineration, disassembly, remanufacturing, refurbishing etc.). Governments are responsible for rewarding and motivating good behaviour. Key regulatory aspects of an effective EPR system includes formulating long-term objectives, fostering continuous improvements and updating targets, e.g. future scenarios, whilst encouraging frontrunners and compelling laggards (Vermeulen and Weterings, 1997). Public benefits include distributed local demanufacturing facilities and immediate economic feedback to product design, driving improvements (Spicer and Johnson, 2004). Challenges for local demanufacturers include knowledge of the original product blueprints, which producers can be unwilling to transfer, and finding suitable markets for recyclable materials. Earlier studies argued that this collective responsibility will weaken the eco-design drive of individual companies (Castell et al., 2008). Next, we document the characteristics and treatment options for tyres.

4.2.3 Composition and Treatment Options for Tyres

Rubbers are thermosetting materials, which makes material recovery challenging because of the vulcanization process during manufacturing (see Adhikari et al., 2000; Medina et al., 2018). Pneumatic tyres are a combination of synthetic and natural rubber, carbon black, elastomer compounds, steel chords, textiles fibres in addition to several other inorganic and organic compounds (Torretta et al., 2015). Natural and synthetic compounds act as sealants while fibre and steel chords give structure and carry tension (Feraldi et al., 2013).

There are several principal treatment practices for EoL tyres (see Table 4.1). First, product reuse (R2), which involves the direct sale of a tyre whose tread is still deep enough for safe use (the minimum tread depth is 1.6 mm in the EU). Second, retreading (R5), which involves replacing the outer tread of a tyre, when its general condition is insufficient. Repurposing

(R6) is the reuse of a tyre for alternative uses, for which it was not originally designed, such as protection of racing tracks, materials for artwork, swings etc. Grinding (R7), involves the crushing and granulation of tyre to extract rubber and other components, such as steel and textile fibres (Aiello et al., 2009; Landi et al., 2018a, 2018b). Grinding produces rubber that is of relatively low quality, meaning only a small percentage (1-5%) can be used in new tyres. Devulcanization (R7) is a technological process where the rubber is chemically recycled to obtain higher quality rubber that can be used in higher percentage in new tyres (up to 30%) (Myhre et al., 2012). However, this technology is not yet commercially viable and has not been deployed on a large scale (Saiwari et al., 2019). Pyrolysis (R8) is the uses high temperatures (without oxygen) and chemical additives, for the recovery of energy, carbon black, activated carbon, oil and steel from EoL tyres; if well managed the process can have relatively low emissions (Myhre et al., 2012; Myhre and MacKillop, 2002; Sienkiewicz et al., 2012). Finally, incineration (R8) involves the burning of tyres with oxygen for the recovery of energy (often for cement kilns and other industrial furnaces); this process is less complex than pyrolysis but creates a significant amount of greenhouse gases and other air pollutants (Myhre and MacKillop, 2002).

TABLE 4.1 R-hierarchy for tyre treatment.

R	Treatment Options
R0	Refuse via reducing vehicle ownership and using alternative modes of transport;
R1	Reduce via life extension
R2	Resell/Reuse discarded tyres which are safe and functional
R5	Remanufacture by retreading functionally sound discarded tyres
R6	Repurpose without or using less physical or chemical treatment
R7	Recycling via processes including devulcanization and grinding.
R8	Recovery of energy via pyrolysis or incineration

Whilst the notion of the 'R-hierarchy' might presuppose a prescriptive and preferable set of recovery operations, these only relate to the product or material attributes and do not account for contextual and broader systems factors, e.g. energy recovery; this might mean a lower R-strategy, could be preferable under some contexts and conditions. Deciding on the most effective treatment option can usually be ascertained through conducting a life cycle assessment (LCA). Various studies have explored this exact question in different national contexts (cf. Corti and Lombardi, 2004; Clauzade et al., 2010; Li et al., 2010; Fiksel et al., 2011; Feraldi et al., 2013; Ortíz-Rodríguez et al., 2017). There is a broad consensus that energy recovery as fuel can only capture up to 40% of the embedded energy within tyres (Amari et al., 1999). However, these assessments differ in terms of the geography and scope, are non-standardised, hard to compare and, overall, they show conflicting and inconsistent outcomes. This points to the need for more standardised impartial regional (Social)LCAs, attributional and consequential, with local data, that can inform specific EPR systems as to the most preferable recovery and treatment option.

New CE business models of the 'performance economy' such as Product-Service Systems (PSS), that promote the leasing of products, services or performance instead of direct consumer ownership could facilitate high-value retention options (Camilleri, 2018; Kjaer et al., 2019; Stahel, 2010). Indeed, firms that maintain the ownership of their tyres are incentivised to design long-lasting (R1), reusable (R2), recyclable (R7) and retreadable (R6) tyres. However, this is not always the case, and strong regulation and careful management of possible rebound effects are needed to ensure that PSS' lead to positive environmental outcomes (Demyttenaere et al., 2016; Hobson and Lynch, 2016; Junnila et al., 2018).

4.3 Materials and Methods

To evaluate the organisation and performance of an EPR scheme, this research adopted a case study research design, following procedural insights as outlined by Yin (2003). Case studies are defined as an in-depth description of a bounded system and are useful to examine phenomena in their contextual settings; they are particularly adept to understanding contemporary events (Yin, 2003, p. 5). Case studies are suited for qualitative methods, including those used in the study: interviews, literature review, policy and document analysis (Bryman, 2012).

This research uses the case study of EPR of tyres in the Netherlands, a system which has been in operation (to some degree) since 1995. This case selection was justified through two core reasons: (1) the Netherlands has, since 2005, had a high collection rate (≥ 100%) (ETRMA, 2015); and (2) the Netherlands has a substantially higher level of material reuse (e.g. direct reuse and recycling) than the European average, which is roughly 50% recycling and 50% energy recovery (Scott, 2015). This second point corresponds to the intentions of moving up the waste hierarchy, the underlying principle for all EU recycling activity (EC, 2008). On this basis, the Netherlands represents a successful European EPR example and therefore the case for this research (cf. EC, 2014).

A limitation of a case study approach of a single EPR system is that it cannot lead to generalizable recommendations, even though the analysis provides useful practical insights for other cases. Nonetheless, the analysis of a single case can be used to generate preliminary observations and questions that can form the basis to evaluate future case-studies or comparative research. Indeed, considering the specific history, geopolitical situation, socio-economic conditions and governance mechanisms in the Netherlands. The main lessons from this research cannot be generalized to other contexts, especially in the Global South, where conditions differ greatly. Moreover, all waste streams are unique due to their complex composition, legalities, processing techniques, hazardous nature etc. Therefore, the results and recommendations from this research are most relevant to our specific case study. Nevertheless, some of the lessons might apply to other socio-economic contexts and material streams, when supplemented by additional research on those other sectors and conditions.

Data collection was undertaken in two phases. First, we reviewed the available literature on CE, EPR and tyres (Section 4.2). This set our theoretical framing and perspectives for critically evaluating the EPR system (Section 4.4). The core data is comprised of policy and legal documents on EPR in the Netherlands since its inception in 1995 to 2017. This was supplemented with the EPR performance data, which (from 2005) has been reported annually to the government. Fieldwork was conducted between January to May 2019 which included nine in-depth unstructured interviews, lasting between 30 and 90 minutes, with government officials, industry and EPR representatives for tyres in the Netherlands. Interviewees either worked for the PRO, were members (producers, importers, distributors or EoL processors of tyres) or government officials involved in the monitoring the performance of the EPR system. Fieldwork also included two site visits to tyre manufacturing and recycling facilities based in the Netherlands. Interviews were used to explain and elaborate on insights gained from the literature and documents analysis. A complete list of the interviewees, data and their sources are in the supplementary materials of Campbell-Johnston et al. (2020a).

Next, we analysed the data. First, we reviewed the policy documents and performance data and, in conjunction with interviews, constructed an overview of the EPR system in the Netherlands (Section 4.4); this included history, an overview of the policy structure, actors, targets and key roles. Furthermore, we coded the performance of the EPR data using the 10R framework of Reike et al. (2018) to categorise the treatment outcomes. Second, we undertook a critical evaluation and reflection, using insights from the interviews and the literature to reflect on the strengths, weaknesses and issues about organisation and performance; including aspects of continuous improvement, policy scope and value retention outcomes (see Section 4.5).

4.4 Case Study Description

4.4.1 Regulatory and Legal Overview

The introduction of EPR in the Netherlands originates to the 1988 'Note on Prevention and Recycling of Waste', in which context the government introduced the concept of EPR in 1990 to enable a series of participatory policy projects designing the recycling strategies for 29 waste streams (Vermeulen et al., 1997; Vermeulen and Weterings, 1997).

Consequently, for the tyres waste stream the Dutch government introduced the Besluit Beheer Personenwagenbanden (Management of Passenger Car Tyre Decree) in 1995. Broad responsibilities were attributed to producers and importers to organise the collection and treatment of EoL tyres. In this EPR system, garages and tyre service companies collected old car tyres (mostly after replacing them for new ones) and charged the customer a fee for this collection and purchase of new ones. Garages and tyre service companies then passed the

used car tyres to collection and processing companies along with the collection fee, to sort and adequately process used car tyres. A provisional collection target in the Decree was set at 60% product reuse (direct reuse is defined here as any recovery activity from R2 to R8, see Table 4.1), which included a minimum 20% material reuse (R2 to R7) and maximum 20% energy recovery (R8).

However, this system was open to exploitation, primarily through collectors taking the consumer fee and not passing the tyres onto processors. The consequential stockpiling resulted in municipalities and provinces financing the collection and treatment of illegally dumped EoL tyres (RecyBEM B.V., 2017, see supplementary material of Campbell-Johnston et al., (2020a)).

Following several meetings between sectoral representatives and Ministry of Housing, Spatial Planning and the Environment in 2000, resulted in the 2003 Besluit Beheer Autobanden (Car Tyre Management Decree). Producers were responsible for organising EoL collection and treatment, either individually or collectively. Key provisions of this act included (i) a focus on car tyres, caravans and trailers; (ii) a broad definition of 'producer', to include all producers, distributors and importers, who are responsible for organising the collection and treatment; and (iii) an old-for-new or 1-for-1 regulation, where the final user of the tyre, must be allowed to return the old tyres at no cost when purchasing a new one. All producers are required to pay a disposal fee, for every product brought onto the Dutch market. The treatment targets were not adjusted from the 1995 Decree, setting material reuse (R2 to R8) at 20% of the total weight of collected materials[7]. Moreover, producers and importers were required to report their performance to the government each year. This report must include (a) the number of car tyres that were made available to a party for the first time in that calendar year; (b) the number of used tyres collected in that calendar year; and (c) the percentage of used tyres processed.

Besides the 2003 Decree, the treatment for tyres has been regulated by EC Directive 1999/31/EC, which prohibits rubber tyres going to landfill, and the Dutch Landelijk Afvalbeheerplans (LAPs) (National Waste Management plans) of 2003 (LAP 1), 2009 (LAP 2) and 2017 (LAP 3).

The first National Waste Plan of 2003 establishes the goal for 50% of the total weight of used rubber tyres to be reused as material (R2 to R8). However, the 20% goal of the Car Tyre Management Decree of 2003, has precedence over any objective of the LAPs. LAP 2 continued with the same objectives as the previous one but in its 2014 modification, it adds a "minimum standard" of at least "material recycling" (R7) for all tyres that can be recycled for less than €175 per tonne. For tyres that are not suitable for recycling or that cannot be recycled for less than €175 per tonne, energy recovery is considered the "minimum standard", and is thus allowed. In 2017, LAP 3 further increases the "minimum standard" for energy recovery to tyres that cannot be recycled for less than €205 per tonne.

7 Material reuse in the Decree is defined as: reuse of materials for the same purpose for which they were designed or for other purposes (R2, R5, R6, and R7), including energy recovery (R8).

The "minimum standard" is based on the 'Ladder van Lansink' (a motion accepted in the Dutch Parliament in the 1980s), which recommends reuse, recycling, energy recovery and landfilling as the appropriate sequence of treatment options (Lansink and Veld, 2010). A 2014 modification to LAP2 further expanded the collection responsibilities from passenger cars and light commercial vehicles to also include motor tyres, trucks, buses, agricultural vehicle tyres etc. Tyres from bicycles and scooters are excluded.

In 2018, the EU outlined a CE package, which amended the framework directive on waste (Directive (EU) 2018/851 of the European Parliament and of the Council of 30 May 2018 amending Directive 2008/98/EC on waste). The renewed waste directive creates new requirements for EPR systems, including having effective data collection processes, transparent operations (including on the selection procedure for waste management operators), and dialogue and collaboration with civil society organisations including social economy actors. The Directive also encourages (meaning it is not mandatory) member states to establish eco-design requirements that ensure products are easily recyclable, reusable, repairable and technically durable, contain recycled materials, and have reduced environmental impacts throughout their entire lifecycle.

These requirements were set to ensure that EPR contributes to a CE transition and operate according to the EU waste hierarchy, as established in article 4 of Directive 2008/98/EC. However, these new requirements have not been transposed into Dutch law yet as the Member States have until the 5th of July 2020 to do so, whilst EPR systems have until the 5th of January 2023 to update their structure and operations. Whether this results in substantial changes in the Dutch EPR scheme remains to be seen. However, it provides an opportunity to revisit the governance and circularity of the EPR system for tyres.

4.4.2 Extender Producer Responsibility: Structure and Implementation

In response to the 1995 Decree tyre importers, distributors and producers founded the 'Vereniging Band en Milieu' (Association BEM), to implement their obligation under this Decree. This body is formerly responsible for communications with the government. To manage the updated system established by the Car Tyre Management Decree of 2003, the tyre producers and importers founded two other organizations. First, the Stichting Fonds Band en Milieu (Foundation Funds for Tyre and Environment, hereafter known as the Foundation) which is responsible for the financial management of the waste management system, and the collection and management of recycling fees. The Foundation functions to keep individual members financial contributions and market share confidential (Winternitz et al., 2019). The Foundation then established RecyBEM B.V., a private company, which is the collective implementation organization of the Association BEM. RecyBEM B.V. is thus contracted by the Foundation to manage the collection, processing and reporting of the EPR system (see Figure 4.1). From 2013, RecyBEM B.V. began setting voluntary processing targets, starting from 70% material and product reuse (R2, R5, R6 and R7) in 2013 to 90% in

2015. The system is thus structured as a third-party takeback where RecyBEM B.V. is the PRP (see Section 4.2.2).

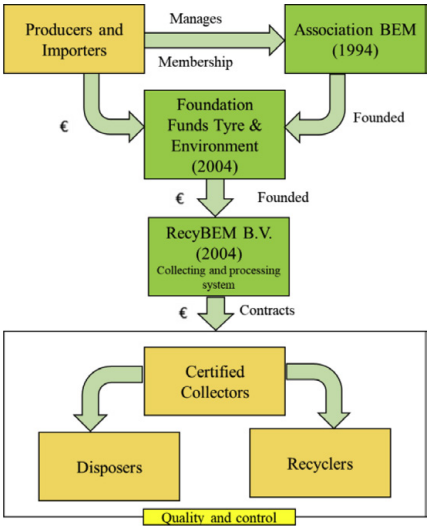

FIGURE 4.1 Organization of the Dutch EPR (Source: RecyBEM B.V., 2019, edited).

To finance the system, all producers and importers of car tyres, caravans and trailers, must pay a waste management contribution fee to the Foundation for every tyre they put on the Dutch market. Between 2004 and 2015 only producers that were members of the Association contributed the waste management fee. In response from protests from the Foundation over free-riders not contributing fees, a 2015 government "general binding statement" (see supplementary material of Campbell-Johnston et al., (2020a)) allowed the PRP to oblige all producers, distributors and importers (both from retail and internet sales) to pay the waste management contribution fee to the Foundation or to establish another EPR system. Non-members can face legal action from the PRP for not contributing.

RecyBEM B.V. is the main operator of the waste management activities, the costs of which are covered by a contribution fee paid to it by the Foundation (see Figure 4.2). It uses the fee to contract and pay third party collectors, which are in charge of bringing the tyres to processors, who recover the value from tyres based on the market conditions, RecyBEM B.V. criteria and state targets and regulations. To ensure the quality of the recycling operations, collectors can only operate with recyclers, disposers and processors that have been certificated by RecyBEM B.V., which includes quality management system, as of 2018 following ISO 9001: 2015 standard (RecyBEM B.V., 2019, see supplementary material of Campbell-Johnston et al., (2020a)).

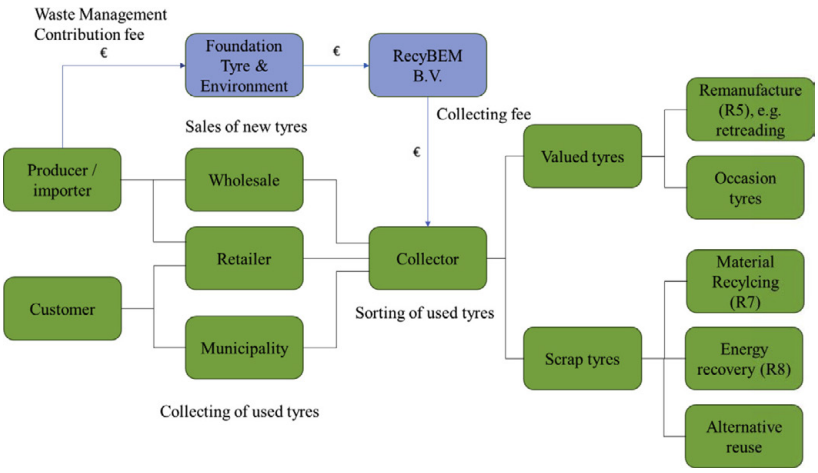

FIGURE 4.2 Financial mechanism of the Car Tyre Management Decree, source: RecyBEM B.V., 2019, (edited).

In 2004, the waste management contribution fee, paid by importers and producers per tyre sold, was set at € 2,00 and by 2017 this had been reduced to € 1,30. This fee is internalised in the consumer price of a new tyre. Collectors (garages) are paid from this fee, which in 2004 was € 1,25 per collected tyre and in 2017 had been reduced to € 1,05 (see Figure 4.3). The difference between the collecting and the recovery fee is used by the PRP to cover administrative costs and unexpected expenses. Every year, the waste management contribution fee and the collecting fee is revised and updated based on a market study conducted by an independent third-party consultancy: Fact Management Consultants. The system operates with a pay-as-you-go structure where each year, a maximum waste management contribution fee is charged and, at the end of the year, a definitive waste management contribution fee is calculated based on the actual sale and recovery outcomes of the year and any surpluses and/or shortfalls are thus settled.

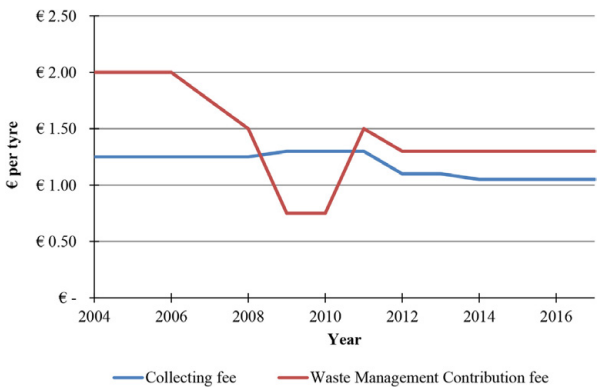

FIGURE 4.3 Collecting and waste management contribution fee 2017 (own work, source: annual reports see supplementary material of Campbell-Johnston et al., (2020a)).

4.4.3 Performance

The membership of the Association BEM has been rising continuously (Figure 4.4), representing over 90% of producers by 2015. The notable rise from 2015 is a consequence of the "general binding statement" of 2015, giving the PRO the power to compel non-compliant actors to pay into their system.

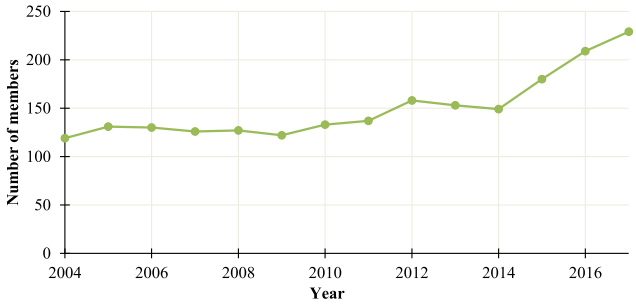

FIGURE 4.4 Association BEM Members between 2004 and 2017 (own work, source: annual reports see supplementary material of Campbell-Johnston et al., (2020a)).

Figure 4.5 shows the high collection rates of the Dutch EPR system. The higher volume of sold tyres in 2010 and 2011 are explained by the particularly cold winters of those years, and correspondingly higher sales of winter tyres. The higher collection rates of 2016 and 2017 is explained by the implementation of the "general binding statement" of 2015, which led to new members joining the scheme.

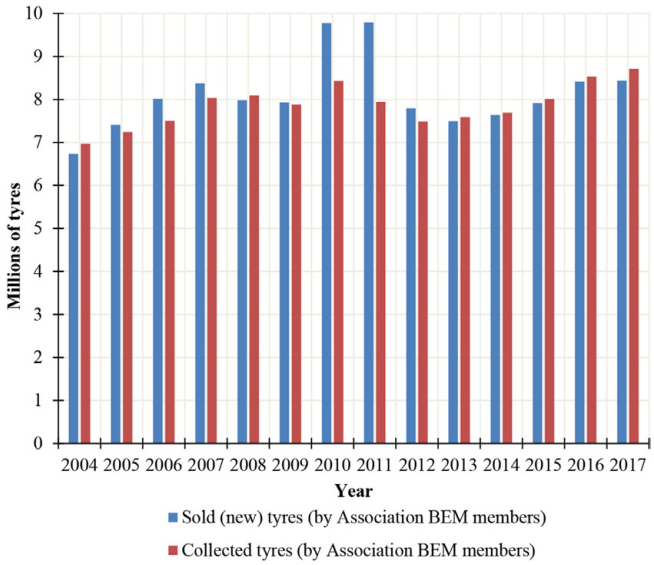

FIGURE 4.5 Sold tyres vs. collected tyres between 2004 and 2017 2017 (own work, source: annual reports see supplementary material of Campbell-Johnston et al., (2020a)).

Figure 4.6 presents the destination of used rubber tyres managed by the PRP between 2005 and 2017. The red dotted line represents the 50% material and product reuse (i.e. R2, R5, R6 and R7) target established by the first National Waste Plan (2003). The red line indicates the 20% reuse as materials (i.e. R2, R5, R6, R7 and R8) target of the Car Tyre Management Decree of 2003. The dotted black line represents RecyBEM B.V.'s voluntary material and product reuse targets (i.e. R2, R5, R6 and R7): 70% by 2013, 80% by 2014 and 90% by 2015. The solid black line represents RecyBEM B.V.'s voluntary material reuse target (R7): 25% by 2013, 35% by 2014 and 50% by 2015.

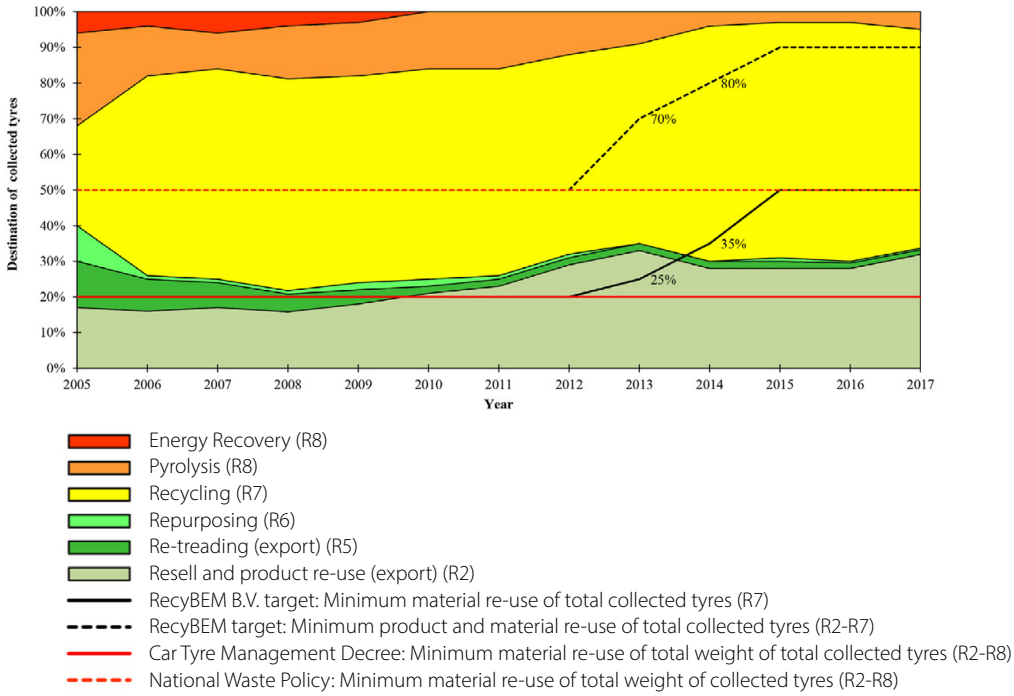

FIGURE 4.6 Destination of collected used rubber tyres by RecyBEM B.V. between 2005 and 2017 (own work, source: annual reports see supplementary material of Campbell-Johnston et al., (2020a))

Figure 4.6 and Table 4.2 show that the Dutch PRO has continuously met the targets in the National Waste Plan and the Car Tyre Management Decrees, as well as voluntary targets (see supplementary material of Campbell-Johnston et al., (2020a)). Moreover, the interviews from the public and private sector confirmed that the minimum standard for incineration was also met, meaning no tyres that can be recycled for less than € 175 (2014-2016) or € 205 (2017 onwards) were sent for energy recovery. Therefore, no fines have been given to the organization for violating the rules.

The explicit nature of the recovery outcomes was further investigated and clarified during the interviews (see supplementary material interviewees of Campbell-Johnston et al., (2020a)). This allowed a better understanding of the implications and complexities of each

recovery option. In the case of "product reuse" (R2), representing over 30% of EoL tyres in 2017, interviewees commented that many tyres are sold to countries in Eastern Europe, although the actual destinations are known only to the PRO[8]. Dutch consumers tend to change their tyre before the minimum recommended tread depth in the EU of 1.6 mm, due to the obliged annual car inspection (EC, 2019), so many discarded tyres still have a high use value. However, the future EoL and safe recovery of those tyres are no longer guaranteed once they are exported, if they go to destinations without the capacity to process them.

TABLE 4.2 Recycling targets and results for Dutch tyre recycling 2005 – 2017

	RecyBEM B.V. target: Minimum material re-use of total collected tyres (R7)	**RecyBEM target: Minimum product and material re-use of total collected tyres (R2-R7)**	**Car Tyre Management Decree: Minimum material re-use of the total weight of total collected tyres (R2-R8)**	**National Waste Policy: Minimum material re-use of the total weight of collected tyres (R2-R8)**
Target	20% (2005-2012) 25% (2013) 35% (2014) 50% (2015-2017)	50% (2005-2012) 70% (2013) 80% (2014) 90% (2015-2017)	20% (2005-2017)	50% (2005-2017)
Result	2005-2012: 54% average 2013: 56% 2014: 66% 2015-2017: 64,8% average	2005-2012: 82% average 2013: 91% 2014: 96% 2015-2017: 96% average	2005-2017: 100% average	2005-2017: 100% average

Regarding retreading operations (R5), very few tyres are suitable for retreading due to quality imperatives, hence very few EoL tyres can take this recovery route. Moreover, the Netherlands does not have any retreading plant, so tyres must be exported for this purpose and, once again, their EoL and safe recovery is not guaranteed in the importing country.

Repurposing (R6) represents a very small fraction of EoL tyres and concerns punctual and limited uses such as cart-track protections, and bumpers on quays and waterways.

Finally, recycling (R7), the most common recovery operation for EoL tyres, is carried out through granulation, which is used in a multiplicity of lower value outcomes, such as insulation materials, engineering applications (mainly for road construction), filling for artificial sports fields etc. Due to energy efficiency, safety and quality imperatives, new tyres currently contain about one to five per cent of granulated rubber from EoL tyres.

Most interviewees reported a high level of satisfaction with the EPR system in the Netherlands. Tyre producers and distributors value the low cost of tyre recovery operations and the "hands-off" approach that this third party take back structure gives them. The PRO enjoys a great level of legitimacy due to its track record of compliance with government targets and low recovery costs. Producers and importers thus give a significant amount

8 We contacted the PRO for the data on the final destination of tyres on various occasions, but we were unable to obtain this information.

of autonomy to the organization (and PRP) and let it manage collection and recovery operation. Producers, importers, collectors and processors are not directly connected and don't collaborate, nor share information to improve tyre recycling outcomes or increase the uptake of recycled rubber in new tyres. There is little evidence that the Dutch EPR system provides an incentive for eco-design, rather it incentivizes producers and importers to outsource recovery operations at the lowest possible cost. While the PRO has financed several research and development projects on devulcanization, this is not enough to foster lifecycle thinking and a full closure of resource loops.

Despite this apparent success, there has been a recent backlash against recycled rubber and the EPR system in response to public concerns over the human and environmental health impacts of artificial sports fields made with recycled rubber granulate (Zembla, 2016). This led to a government inquiry on the topic and a series of reports were commissioned. In line with recent academic research (Bleyer and Keegan, 2018; Peterson et al., 2018), and evaluations of the European Chemicals Agency (ECHA, 2017), the Dutch government report on human health has found no evidence of cancer risks related to artificial turf fields made with recycled rubber (RIVM, 2017). However, other government reports evidenced important environmental impacts, especially for aquatic life (STOWA, 2018; Verschoor et al., 2018). This demonstrates the complexities of a circular system, which aim to narrow, slow, shrink, and close material cycles, but do so in ways that do not affect human and environmental health. This is often complicated, especially when dealing with complex recycling processes and materials containing a mixture of often unknown or toxic chemicals. This complexity poses the main obstacle to tyre management in the Netherlands.

4.5 Analysis and Future Implications

Since the initial experiments in 1995, the Dutch EPR system for passenger car tyres has reached 100% collection rate, with low energy recovery levels (5% in 2017) and zero landfilling. Interviewees viewed the system as stakeholder friendly, financially efficient, and effective at preventing the widespread illegal dumping of tyres, which occurred before the 2003 Decree. The system meets the minimum standards and targets set in the 2003 LAPs and the PROs voluntary targets. However, it also has many key obstacles, weaknesses and limitations both from the perspective of CE 2.0 and of CE 3.0. This section outlines these challenges, and proposes recommendations, which, after careful adaptation, could also provide useful insights for new and existing EPRs in the global North and South alike:

Recommendations from a CE 2.0 perspective:

Promoting higher-value recovery: Figure 4.6 and Table 4.2 demonstrate a high focus on recycling, yet the recycling of tyres currently produces low quality granulate that cannot be used in large quantities in new tyres. This focus on material recovery is thus a form of downcycling, which does not allow for the closing of resource loops. Instead, greater priority

should be given to other recovery options such as retreading, reuse and repurposing. Moreover, eco-design must be encouraged so that EoL tyres are easier to remanufacture and recycle and so that new tyres can contain higher quantities of granulated rubber without compromising on their quality. In this regard, further investment in R&D would be necessary and could be implemented by an obligation to use a percentage of the waste management contribution fee to finance it. An autonomous or government established fund can be established to manage this part of the fee to finance transformative and disruptive innovations, which can challenge incumbents. Another option is to establish a differentiated fee based on the sustainability of tyres (durability, recyclability, percentage of recycled content etc.) to incentivise eco-design and innovation in the marketplace.

Managing exports and leakages: A large percentage of EoL tyres are exported for reuse and retreading (about 33% in 2017). While these are high-value recovery options, in theory, the lack of monitoring on the destination of these tyres does not guarantee an environmentally safe recovery. It is thus key to set up mechanisms to prevent exports from happening and to have greater oversight over the export destination and final disposal of tyres. This is a critical concern since tyres can significant adverse human and environmental health impacts if they are not properly recycled (Li et al., 2010; Verschoor et al., 2018). However, controlling exports and following tyres through their multiples uses and owners is a complex process. A possible solution to this problem would be to raise consumer awareness and improve the annual car inspection process so tyres are not discarded before they reach the minimum tread depth. This would keep tyres in use for longer, improve their value for customers, and prevent them from being exported, thus reducing transport emissions and impacts oversees. The above measure would have to be combined with strong controls on the export of second-hand tyres so that tyres with a tread depth under the minimum standard are not exported for direct re-use. Moreover, enforcement of EoL tyre export controls should be reinforced so they are not exported to countries that do not meet Dutch social and environmental standards.

Recommendations from a CE 3.0 perspective:

Aiming for sufficiency to reach the highest value retention options (R0, R1): Having longer-lasting tyres is perhaps one of the most important strategies, which can lead to significant sustainability improvements, as it directly reduces overall tyre consumption (R1 - reduce). The current EPR system has so far done nothing in this regard, and tyre consumption has increased between 2004 and 2017 (see Figure 4.5). The PRO could directly work with rubber tyre manufacturers and importers to design tyre in a way that guarantees their durability. This has the added benefit of reducing the number of resources spent dealing with EoL tyre management further down the product lifecycle. Awareness campaigns among consumer can also increase the lifespan of tyres and be done through a combination of product labels and media campaigns. This R1 strategy is second in the value retention hierarchy, leading to considerable environmental benefits,

thanks to the reduced pressure on natural resources (rubber, iron, fibres etc.) and the avoided impacts from production, use and disposal of tyres.

An even more effective strategy would be to reduce tyre consumption by reducing the need for tyres in the first place (R0 – refuse). This could be achieved through effective urban and regional planning, as well as transport policies that encourage public transportation, rail, cycling and walking. However, these policies are beyond the concern of a PRO and can thus only be established by national, provincial and municipal governments. This shows the limitations of EPR systems in general, especially with the highest value retention options: R0 and R1. To implement these measures, a percentage of the waste management contribution fee can be given to a government agency or an autonomous institution responsible for reducing the overall domestic material consumption and ecological footprint through sufficiency strategies. This agency could thus develop innovative transportation solutions which work towards reducing the need for rubber tyres such as improved national rail networks, and sustainable urban planning solutions.

Collaboration and multi-stakeholder governance: The existing EPR system lacks effective connection and collaboration between tyre producers and recyclers. This inhibits product innovation concerning the application of reclaimed rubber. The EPR system for tyres in the Netherlands could hence be improved by further integrating recyclers, disposers and processors members with the BEM Association. This would reinforce collaboration across the whole value chain and ensure that the EPR system does not just incentivize low-cost recovery options.

Socially inclusive governance considerations have been disregarded by the Dutch EPR system. Various scholars have pointed out the importance of these aspects to construct a fair and fully sustainable CE (Hobson and Lynch, 2016; Kirchherr et al., 2017; Merli et al., 2018; Millar et al., 2019; Moreau et al., 2017), which tackles questions of intellectual property, technology transfer, ownership, production methods, benefit sharing and participation in decision-making processes. While the Dutch EPR does have a successful governance structure that includes all the relevant producers and importers (see Section 4.4.2), it is not particularity inclusive beyond direct industry members. This reduces the capacity for democratic oversight, transparency and accountability, leading to suboptimal outcomes in terms of recovery options and human and environmental health (see Section 4.4.3). To improve this, it is key to foster greater participation of civil society and public authorities in the governance, oversight and management of the EPR system. This can be achieved by forcing the BEM Association to include a certain percentage of civil society members, which represent the interests of citizens and the natural environment. This would force the EPR system to consider wider social and environmental concerns and improve the overall transparency and accountability of the system.

Effective monitoring and continuous improvement of the EPR system: Considering that collection targets have not been adjusted since 2003, and remain vaguely defined, it is key to update targets and explore the future direction for the sector. In fact, not only are the established recovery targets not ambitious enough but they were already met in the year they were set (see Section 4.4.3).

Setting renewed goals is particularly important as the current system promotes a standard and generally low waste management contribution fee, which has incentivised low-cost and low-quality recovery options over higher-value-retention ones. Moreover, the existing monitoring system reports only collected volumes and treatment processes. This leaves data gaps regarding how recovered materials are used and what is the final fate of exported EoL tyres, all of which can hide unsustainable practices.

The careful regulation and monitoring of the EPR system through effective government policy, civil society oversight, and continuously improving targets and incentives for higher-value retention options (especially R0-R6) is thus key. Moreover, it is necessary to overhaul the ways by which the best processing options are chosen (including the selection procedure for waste management operators) and the ways by which investments are carried out to achieve continuous improvements in new recovery options (e.g. R&D in devulcanization or pyrolysis). Better monitoring, transparency, oversight and civil society participation in these processes is key to ensure the continuous improvement of the EPR system and to promote socially and environmentally sustainable design and recovery practices.

Improving overall social and environmental outcomes beyond EoL tyres: The consequences of potentially socially and/or environmentally harmful uses of granulated rubber shows the weakness of focusing on recovery alone rather than actual sustainability outcomes. It also raises the question regarding extended value chain governance, whether producers should have continued responsibility beyond the first EoL processing of the product. Such expansions of capacities must be done only after an impartial, non-conflicting, regional LCAs aimed at maximizing circularity, social fairness and sustainability. In fact, in such complex situations, having clear research and data at hand is vital to plan the best possible recovery options with human and ecological health in mind. Furthermore, a plan to improve the sustainability outcomes of the entire tyre supply chain should be established and implemented in coordination with a more democratic and inclusive EPR structure. This can ensure that the EPR system doesn't just recycle EoL tyres but also leads to tangible improvements in terms of socio-ecological outcomes, and raw material demand. The overall aim of a CE is not just to close resource loops, but to reduce the pressure of human activities on the planet to ensure the well-being of current and future generations (Kirchherr et al., 2017; Korhonen et al., 2018a). An EPR system should thus be understood as a component of a broader policy objective, which aims to sustainably and equitably reduce a country's overall environmental footprint.

Circular business models: Circular service or leasing business models based on the performance of tyres, rather than selling large quantities of tyres could be encouraged to incentivize higher-value maintenance for producers and consumers (Stahel, 2010). Indeed, under the right conditions, PSS can lead to a sustainable CE, since industries which keep ownership of their tyres have a direct incentive to develop long-lasting and easily recyclable products (Camilleri, 2018; Kjaer et al., 2019). It could thus improve reduce, reuse, retreading and quality recycling within the Netherlands, henceforth reducing the overall consumption and export of tires whose fate remains unknown once exported. However, this necessitates careful government oversight and regulation to prevent rebound effects and ensure that PSS lead to reduced overall resource use and create positive social and environmental sustainability outcomes (Hobson and Lynch, 2016).

The identified gaps and these proposed solutions provide an opportunity for the EPR organization to transform from being an EoL tyre management entity to a true driver of circularity, playing a transformative role in addressing prominent contemporary social and environmental challenges. In this transition, the system must be more inclusive, democratic and adaptive to continuous improvements. The existing fragmented systems of isolated EoL tyre management must be integrated into a value chain governance approach and high-value maintaining targets must be envisioned together and collectively worked towards with greater transparency.

The abovementioned recommendations are in line with those of the updated EU waste directive, which calls for EPR systems to include eco-design requirements to reduce environmental impacts as well as to improve transparency, reporting, monitoring and collaboration with civil society. There is thus now a unique opportunity to overhaul the Dutch EPR system through holistic CE 3.0 strategies, leading to both improved human well-being and ecosystem functioning.

However, a possible limitation of the above recommendations is the small size of the Netherlands in the global market for tyres. Indeed, the country imports most of its tyres and can hardly force large tyre producers oversees to significantly change their design and production processes. EU-wide directives with ambitious targets for tyre recycling, retreading, repurposing, and percentage of recycled content in new tyres is necessary. Indeed, while the EU has established a new CE action plan with various new policies, it has not taken further action on tyres or rubber recycling. Further action from a holistic CE 3.0 perspective is hence needed both nationally and internationally. Another key limitation of the above recommendations is that they are directed towards the unique social, historical, political, economic and technical circumstances of the Dutch EPR system for EoL tyres. Therefore, further research is needed to validate and apply our insights and commendations to other case studies and waste streams.

4.6 Conclusion

This chapter examined and evaluated the structure, organisation, performance and potential limitations of the Dutch EPR system as a case study to explore how this older CE 2.0 systems can be adapted to fulfil the broader societal concerns embedded in the current CE 3.0 debates (i.e. concerns over resource supply, planetary limits, waste generation). It adds a practical understanding of the relationship between EPR and CE, and the former's capacity to contribute to the latter.

Despite this representing a successful example of CE 2.0 initiatives and fulfilling the obligations of the national legislation, our analysis outlined seven limitations and issues, which, we argue, can be the basis of modifying and creating an EPR that meets the needs of the existing CE 3.0 debate. Current EPR systems of CE 2.0 can achieve high recovery rates, but they do not reduce overall resource consumption and promote full circularity, in line with CE 3.0. Thus, our paper suggests strengthening the EPR system by proposing a long-term transformative perspective, which can address issues concerning transparency, inclusion, sufficiency, sustainability and continuous improvement. These lessons could be applied to different contexts and waste streams with careful research and adaptation. Moreover, we examined the internal consideration of the Dutch EPR system. As Circularity in the Netherlands is inherently is tied to a European and global circularity, any exports should be strictly controlled and regulated to ensure high-value retention and sustainability.

This research further illustrates the limits of recycling and traditional recovery operations. CE is often characterised as a tool for closing resource loops and turning wastes into resources. However, low-quality recovery options complicate this as a closed-loop for tyres cannot simply be established with current technologies. Whilst devulcanization could potentially improve recovery outcomes, it is not commercially operational on a large scale and only enables the use of up to 30% secondary rubber in new tyres; still far from a closed-loop. This shows the limits of R3-10 and the importance of sufficiency strategies, especially R0-1 to reach a CE with tangible results in terms of reduced material demand and ecological footprint. The above points are beyond the scope of this chapter and demonstrate the complexity of the CE, and the need for specific case studies to improve its governance and implementation.

Moreover, the insights and recommendations learned from this chapter are limited to the recovery of tyres in the Netherlands, and further research is needed in other contexts to develop specific and culturally adapted recommendations. In particular, transdisciplinary research with key actors and stakeholders could be an effective manner to build solutions for a sustainable, circular, and participatory overhaul of EPR systems.

Future comparative analysis of EU EPR systems is also needed to uncover how they interfere with each other in the context of the single market. A broader study could also provide further insights into structural issues and challenges for EPR systems in general and uncover other possible best practices for EPR systems from a CE perspective.

5

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Future perspectives on the role of extended producer responsibility within a circular economy: A Delphi study using the case of the Netherlands

Abstract

Extended producer responsibility (EPR) is a proposed policy approach to promoting the circular economy (CE) within the European Union. This research used a policy Delphi to explore perspectives on improving EPR policies to further contribute to the CE goals of the Netherlands. Both the potential improvement and critical reflections discussed by CE and EPR experts and practitioners from this study contribute to a more detailed understanding of the future governance of CE practices. We present various activities to improve EPR and insights from Delphi participants that emerged from the study. This chapter shows that whilst actors agree, in essence, that there is a need for modifying EPR, what the specific changes to the form are and to whom the new responsibilities apply is contested.

5.1 Introduction

The European Union (EU) has embraced the concept of circular economy (CE) as a vehicle to address various societal challenges, including overconsumption of resources, waste generation and high carbon emissions (European Commission, 2020b, 2018b). Multiple member states of the EU have initiated CE strategies and are at various levels of engagement (cf. Marino and Pariso, 2020). CE is also a core tenant of the Green New Deal, viewed as the basis for the post-Covid-19 economic recovery of the EU. Strengthening the policy instrument of extended producer responsibility (EPR) is one feature of this strategy (European Commission, 2020c).

EPR is an environmental management strategy that makes producers responsible for organising the take-back, treatment and recycling of their products' waste (Mayers, 2007). Originally conceived as a means to incentivise eco-design and sustainable product innovation (cf. Lifset and Lindhqvist, 2008; Lindhqvist, 2000). However, in practice, across EU applications, it primarily focuses on the collection and processing of post-consumer products while claiming to encourage eco-design (Atasu, 2019; Deutz, 2009; European Commission, 2008). EPR is mandatory within the EU for Waste Electrical and Electronic Equipment (WEEE) (2002/96/EC; 2012/19/EU), Batteries (2006/66/EC), End of Life Vehicles (ELV) (2000/53/EC), Packaging (94/62/EC; 2018/852), and, more recently for single-use plastic products, e.g. food containers (EU2019/904). The Waste Framework Directives (2008/98/EC) and subsequent amendment (2018/851[9]) outlined principles on the implementation and minimum requirements for the instrument. Many member states have initiated additional EPR schemes, most commonly for tyres, graphic paper, waste oils, paper and cardboard (Monier et al., 2014).

Earlier writings on CE have criticised the prioritisation of end-of-life (EoL) solutions within current regulatory frameworks (Gregson et al., 2015) or showed the complexities of governing materials in a circular manner within them (Deutz, Baxter, Gibbs, Mayes, & Gomes, 2017). However, the implementation of CE practices confronts competing discourses and visions of how CE will look and whether these practices can capture the transformational intentions with which some view CE (cf. Calisto Friant et al., 2020; Geissdoerfer et al., 2017). Moreover, existing research on the connection and contribution of EPR to CE has focused on examining best practices and challenges within existing systems (Campbell-Johnston et al., 2020a; Kunz et al., 2018; Richter and Koppejan, 2016), challenges of resource recovery (Deutz, Baxter, & Gibbs, 2020), the impact of higher recycling targets and greater source separation of EoL waste streams (Andreasi Bassi et al., 2020; Beccarello and Di Foggia, 2018), issues of waste trade – especially from the Global North to Global South – and reduced consumption (Liu et al., 2018). However, such

9 See Article 8a

research has not captured the plurality of perspectives on improving EPR nor sought a common understanding of how it can better contribute to the implementation of CE.

Herein, we examine insights and perspectives on strengthening EPR based on a Delphi study conducted with practitioners in the Netherlands. We focus on the Netherlands because (1) it was an early mover with EPR (see Vermeulen and Weterings, 1997), and thus has a long basis of practical experience to draw from, and (2) it has, since 2016, a CE strategy that explicitly outlines the objective of a 50% reduction in the use of primary raw materials (Ministry of Infrastructure and Water Management & Ministry of Economic Affairs, 2016). A Delphi study draws on expert opinion to derive insights and conclusions (see Section 5.3). Thus, we developed the following research question: how can EPR be further strengthened or transformed to contribute to the CE goals of the Netherlands? This chapter builds on Kunz et al. (2018), further outlining stakeholder perspectives on EPR in the context of CE. It additionally provides a clear set of outputs to practitioners to strengthen EPR.

This chapter is structured as follows. First, we review the existing literature on EPR, its issues and its implementation within the EU (Section 5.2[10]). Next, we outline the methodology (Section 5.3), followed by the results (Section 5.4), discussion (Section 5.5) and conclusion (Section 5.6).

5.2 Literature review

5.2.1 EPR: a brief historical overview

EPR emerged in response to the increasing complexity and volumes of waste, which were overburdening municipalities in the late 1980s (Vermeulen and Weterings, 1997). EPR was originally defined as "an environmental protection strategy to reach an environmental objective of a decreased total environmental impact from a product, by making the manufacturer of the product responsible for the entire life-cycle of the product and especially for the take-back, recycling and final disposal of the product." (Lindhqvist, 2000, p. 2). Nevertheless, subsequent definitions, whilst paying lip-service to the whole 'life-cycle' framing, have, in practice, only focused on the EoL and post-consumer phase of a product's lifecycle. This is done through shifting this responsibility away from municipalities to producers (European Parliament and Council, 2012, 2002). The limited scope of EPR is particularly evident in the EU, where the potential for EPR to enable eco-design is more "aspirational than real" due to a lack of targets and objectives (Deutz, 2009, p. 283). Since its introduction in the 1990s, EPR, as a principle of policy design or policy instrument, has become widespread globally, but in a diverse array of practices (cf. Atasu and Subramanian, 2012; OECD, 2016; Ongondo et al., 2011; Yu et al., 2008).

10 Readers guide. To avoid repetition from earlier chapters, readers are advised to skip section 5.2.1 on EPR.

The implementation of EPR has not been uniform; instead, 'responsibility' has manifested in different forms and configurations depending on the contextual and legislative decisions of the implementing country (OECD, 2016). However, as outlined by Lindhqvist (2000), formal[11] EPR systems can consist of different combinations of the following five elements: (1) liability for the proven environmental damages caused by the product. The extent of which is determined by legislation; (2) economic/financial responsibility for the EoL stage, e.g. collection, recycling or final disposal; (3) physical responsibility, producers are involved in the physical management of the products and/or their effects; (4) informative responsibility, which can include requiring producers to supply information on environmental properties and contents for the products they produce; and (5) ownership of the product, which can be retained by the producer.

Whilst producers can fulfil their EoL obligations individually, the OECD (2016) distinguishes four modes through which EPR is generally organised: (1) One single organisation made from a consortium of producers, known as a Producer Responsibility Organisation (PRO) with commercial and/or municipal collection and processing services; (2) Multiple PROs with the clearinghouse and commercial and/or municipal collection and processing services; (3) Governance structure for tradable credits system; and (4) Government-run EPR system.

Over the last 30 years, EPR policies have been steadily adopted by national governments, with most schemes covering electronics, packaging tyres and batteries (OECD, 2016). The broader definition of EPR, as outlined by the OECD (2016), includes other recycling-related policy instruments: market-based instruments, e.g. deposit funds, performance standards, e.g. minimum recycled content and information instruments, e.g. product labelling.

5.2.2 EPR in European Policy and Law

EPR is a significant component of product and environmental policy that has emerged in the EU over the last 30 years. Other key policies include the Ecodesign Directive 2009/125/EC, Energy Labelling 2017/1369, REACH Directive 1907/2006 and the Waste Shipment Regulation 1013/2006. In this context, the scope of EPR has been narrow: combining economic and physical responsibility requirements for producers, namely for the collection and recycling of waste products (Deutz, 2009). Specific characteristics are consistent across all EU EPR Directives, e.g. Batteries, ELV, WEEE and Packaging. Namely, a broad definition of producers to one that includes importers and distributors, i.e. the actor who brings the specific product onto the national market; collection, treatment and reporting targets, for example, from 2015, a reuse and recovery target for ELV was set at 95% by an average weight (Directive 2000/53/EC); and the ability of producers to respond to obligations either individually or collectively.

11 By formal EPR systems, we refer to policies or laws that specifically adopt the term extended producer responsibility. Informal EPR systems refer to systems which promote greater EoL responsibility for producers, yet are not codified as EPR. For example, voluntary agreements or covenant, see OECD (2016) and Worrell and Reuter, (2014).

EPR across the EU is also governed by 'general minimum requirements' as set out in Article 8 and 8a (inserted by Directive 2018/851) of the Waste Framework Directive 2008/98/EC. Member states have flexibility when implementing EPR practices, as long as they conform to the general minimum standards. Core specifications of the minimum standards include the definition of clear roles and responsibilities for relevant actors, from producers, waste operators, local authorities, and reuse organisations where appropriate; a reporting system for products put on the market by producers, which also includes specific collection and recycling targets; and the equal treatment of producers regardless of their size.

In responding to the aforementioned requirements, member states have generally followed two operational frameworks at the national level: a national compliance scheme or a clearing house model (Khetriwal et al., 2011; Savage, 2006). A national compliance scheme is one dominant organisation, which takes responsibility for all EoL requirements for producers (see Khetriwal et al., 2011). Conversely, in a clearing house model, producers, or PROs, report the number of products put on the market to a government-managed organisation: the clearing house. The clearing house assigns collection responsibilities based on market share (Khetriwal et al., 2011). The Netherlands complies with EU requirements for EPR, with a national compliance scheme in place for each established product category, such as cars, batteries and WEEE. EPR is also used as an approach to organise EoL tyres (Campbell-Johnston et al., 2020a) and is the basis of a sectoral agreement on flat glass (Ministry of Infrastructure and Water Management, 2017). Current discussions focus on extending the approach to new streams, e.g. mattresses (Dubois et al., 2016).

Research on EPR has raised various critical issues which are either un-or-under addressed in the current EU policy (cf. Calisto Friant et al., 2021; Kunz et al., 2018). We highlight seven essential issues in the current debate: EPR, eco-design and innovation; cost allocation and incentives; targets and goals; reporting and transparency; treatment choices; and monitoring and enforcement.

One debate has broadly focused on the connection between EPR, eco-design and product innovation. Extant research has suggested that formal eco-design requirements have been more effective in influencing product design than EPR (European Commission, 2014c; Gottberg et al., 2006; Kunz et al., 2018), with only some slight evidence of the contrary (Kautto, 2006). The question of design interrelates with organisational issues of EPR, including cost allocation and incentives, mainly connected to the role of fees paid by producers. Research has examined how to set fee structures while accounting for historical and orphaned products[12] (Kalimo, Lifset, Atasu, Van Rossem, & Van Wassenhove, 2015; Mayers, Lifset, Bodenhoefer, & Van Wassenhove, 2013), whether the fees can be linked to long-term design incentives (Besiou and Van Wassenhove, 2016), or whether modulating

12 Orphaned products are those that have been taken off the market by producers, or when the original producer cannot be identified (Kalimo et al., 2015).

the fee[13] system can be more impactful on influencing product design (Hogg et al., 2020; Kalimo et al., 2015; Kunz et al., 2018).

Concerning EPR policy and law, research has illustrated the limits of the current targets and goals regime, particularly the collection and processing targets based on mass-balance, e.g. kilogrammes collected or processed, and not specific materials, quality of materials or components (Ortego et al., 2018a; Parajuly and Wenzel, 2017; Wilts et al., 2011). Alternatively, whether the targets are too stringent and unachievable (Favot, 2014), and whether they should be broadened to allow the inclusion (actor inclusiveness) of social economy actors (Bahers and Kim, 2018; Campbell-Johnston et al., 2020a). EPR systems require reporting obligations from producers or their representatives, yet, harmonising definitions and requirements of the system remain an issue (Kunz et al., 2018). Moreover, the detail and transparency of EPR reporting have received criticism, for example, the tracing of the final destinations of collected EoL products (Campbell-Johnston et al., 2020a) (reporting and transparency).

A large body of research on EPR has focused on the applications of treatment and recycling technologies (treatment choices) and standards (cf. Winternitz et al., 2019; Zhang et al., 2019). Finally, issues of transparency are interconnected with issues of monitoring and enforcement, particularly concerning non-compliance and free-riders who do not pay nor fulfil their collective obligations, or legal and illegal waste trading, which can include products under EPR schemes which are then sold second-hand abroad (Clapp, 2001; Dubois et al., 2016; Widmer et al., 2005; Wilson et al., 2011). The most pervasive free-riders appear to be "large and well-known multi-seller platforms with fulfilment centres in the EU" (Hogg et al., 2020, p. 150). The issues outlined above are not uniform and likely vary in the context, product category and conditions of the specific member state. However, they do provide a point of departure to conceiving critical issues related to EPR.

5.2.3 Circular economy and extended producer responsibility

Within the long conceptual history of CE (cf. Blomsma and Brennan, 2017; Calisto Friant et al., 2020; Ghisellini et al., 2016), EPR is argued to be an older iteration of CE-like practices that sought to increase accountability of producers and polluters through greater responsibility and visibility of externalities (Campbell-Johnston et al., 2020a; Hickle, 2017; Reike et al., 2018). Existing research has examined the complexities between material and product governance within CE, specifically the regulatory challenges of moving from pollution prevention to material recovery (Deutz et al., 2017). However, despite the growth of articles written on CE (Calisto Friant et al., 2020), comparatively few studies have focused on the interrelation between EPR and CE. Researchers have examined the role of EPR in supporting

13 Modulated fees refer to differentiating the fees paid by producers based on criteria. For example, producers in France pay a different contribution for packaging depending on whether the packaging is glued together or can be manually separated (Kunz et al., 2018)35% of e-waste and 65% of packaging waste have already been recycled (or reused in some cases.

higher waste collection and processing targets (Beccarello and Di Foggia, 2018; Rubio et al., 2019) and as a way to promote sound waste management, specifically through maximising collection rates and reducing material impurities at the point of disposal (Andreasi Bassi et al., 2020). The emphasis on CE within policy discourses coupled with the existing issues within EPR provides an opportunity to explore potential changes to EPR to support the transition to CE.

5.3 Materials and methods

5.3.1 Research design

To examine how EPR could better contribute to CE, specifically the CE goals of the Netherlands (50% reduction in primary materials by 2030 (Ministry of Infrastructure and Water Management & Ministry of Economic Affairs, 2016)), this project conducted a policy Delphi.

The Delphi originated in the 1950s as a method to obtain consensus on a specific topic by a group of 'experts' (Dalkey and Helmer, 1963; Okoli and Pawlowski, 2004). Subsequently, it has been adapted for multiple purposes and contexts, e.g. forecasting, idea generation and prioritisation or capturing existing knowledge (cf. Franklin and Hart, 2007; Yousuf, 2007). Whilst numerous variations of the methodology exist (Linstone and Turoff, 2002), at its essence, a Delphi consists of multiple questionnaire or discussion rounds with a select group of anonymised experts, with each round interspersed with controlled feedback (Yousuf, 2007). The role of the researcher(s) is to act as a facilitator by distributing surveys and tabulating responses (Gokhale, 2001).

This research adopted a policy Delphi, henceforth referred to as a Delphi or Delphi study, which is useful for idea generation on a specific topic (Franklin and Hart, 2007), by ensuring that "all possible options have been put on the table for consideration, to estimate impact and consequences of any particular option, and to examine and estimate the acceptability of any particular option" (Turoff, 1997, p. 87). Reaching consensus on a topic is therefore not the sole aim, but instead deriving thorough information that can be the basis of sound decision-making (Franklin and Hart, 2007). This Delphi was structured in five interrelated phases, following the procedural insights of Franklin and Hart, (2007), Gokhale, (2001) and Okoli and Pawlowski, (2004). These phases consisted of the selection of experts, three survey rounds and a concluding workshop.

5.3.2 Research steps and analysis

Phase 1 concerned the nomination, selection and contacting of experts. Delphi studies do not seek generalisable representativeness of the population under study; instead, they involve 'experts' who have a deep and qualified understanding of the issues presented.

Determining who is considered an expert, is, therefore, a critical requirement for the researcher (Okoli and Pawlowski, 2004).

This study followed the selection procedure of Okoli and Pawlowski (2004). First, identifying the relevant disciplines, organisations and knowledge needs related to the question. Second, assigning specific individuals to each of these disciplines. Third, nominating additional experts and ranking them based on qualifications. Finally, inviting the selected experts to the study. In total, we identified and contacted 50 experts in EPR and/or CE from government, industry and knowledge[14] connected to the following product categories: electrical and electronic equipment (EEE), flat glass, cars, and floor coverings, e.g. carpets. These experts were chosen to provide input using their experience of products with existing EPR schemes (EEE, flat glass and cars) and possible future schemes (floor coverings). The logic being that we could draw from those with longstanding incumbent knowledge and those outside of existing frameworks who could hopefully bring alternative or innovative perspectives. See Table 5.1 for an overview of the number of experts per phase.

TABLE 5.1 # expert participants per phase per expert category.

Expert category	# participated phase 2	# participated phase 3	# participated phase 4	# participated phase 5
Government	6	6	6	4
Knowledge	9	10	6	4
Producers	5	4	3	1
PRO	6	3	3	3
Processors	4	4	2	1
Total	30	27	20	13

See supplementary materials 1.A of Campbell-Johnston et al., (2021) for an overview of the experts

In Phase 2, the first questionnaire was distributed, which included 30 statements proposing *how* EPR could be changed to meet the CE goals (see supplementary materials 1.B of Campbell-Johnston et al., (2021). These statements were derived from (1) an in-depth literature review of EPR, (2) a public consultation executed by the Dutch government of the new draft decree on EPR, and (3) a science-policy workshop held in January 2020 on EPR (see supplementary materials 1.C of Campbell-Johnston et al., (2021 for the complete list of articles). The latter two inputs were used to capture more immediate perspectives and not just reiterate issues in the literature, which can be dated (Schmidt et al., 2001). The experts were asked to comment on each statements' clarity and suggest alternative ideas for transforming EPR. This round received responses from 30 experts and over 300 comments.

In Phase 3, based on the comments from Phase 2, we proposed 25 statements to the experts. These included 20 adapted statements from Phase 2 and five additional statements

14 In this study "knowledge" is defined as individuals employed in either academia, consultancy or think-tank/ knowledge institutes; "Industry" is used to signify individuals employed for either *processing*, i.e. recycling companies, PROs and producers as defined under Directive2012/19/EU.

posed by the experts (supplementary materials 1.B of Campbell-Johnston et al., (2021)[15]. Experts were asked, for each statement: (a) to signify the "likelihood" of the activities in the statement contributing to the CE goals on a 5-point Likert scale, with "1" representing "highly unlikely", "3" "neutral" and "5" "highly likely"; and (b) whether the activities in the statement were "desirable" for the sector they work on/in, with "1" representing "highly undesirable", "3" "neutral" and "5""highly desirable". Phase 3 received 27 responses.

In Phase 4, the final questionnaire was distributed, which included statements deemed the most important or controversial. The Delphi methodology has multiple approaches for consensus measurement (cf. von der Gracht, 2012). When judging which statements were important or controversial, we followed the same approach as Franklin and Hart (2007). For statements deemed important, we selected seven statements with mean scores above a 'cut-off' of 3.7 for both the "a" and "b" questions of phase 3. For those deemed controversial, we examined statements with a mean below 3.7 and compared their standard deviations, again selecting 7 (Franklin and Hart, 2007). Controversial statements were included to elicit further discussion from actors with competing positions. Experts were provided space to reflect and elaborate on the statements. This phase received 20 responses.

Phase 5 was the final phase and included a stakeholder workshop. Owing to the Covid-19 social distancing restrictions at the time of research in force in the Netherlands, a workshop with only 13 experts was organised. The results from the final questionnaire round were presented, which included a qualitative analysis of the textual responses. These were used to illustrate the competing and divergent positions of actors towards statements. The workshop then focused on discussing and reflecting on the results of the study and the challenges it raised.

5.4 Results

Transforming EPR to more effectively contribute to CE is not a straightforward task given the multitude of challenges (see Section 5.2.2). The following section presents a synthesis of the key strategies for improving EPR and the critical issues, discussions and reflections raised during the study. The results include the core strategies for improving EPR (Phase 3) with the perceptions of experts on (a) how likely each strategy would contribute to the CE goals of the Netherlands and (b) how desirable it would be for the sector they work on/in (1-5 Likert scale). The complete statements are provided in supplementary materials 1.B of Campbell-Johnston et al., (2021), and shortened versions are numbered and presented below for readability. The perceptions and critical issues raised by the experts in Phases 4 and 5 are discussed. The results are grouped under the seven critical issues for EPR (see Section 5.2.2). Specific inputs of the experts are discussed, and quotes are presented with

15 Nine of the original 30 statements we dropped or combined with other statements based on comments from the experts.

quotation marks and in an Italic script. Some quotes have been adjusted for readability purposes. The complete data set is included in supplementary material 2 of Campbell-Johnston et al., (2021).

5.4.1 EPR, Eco-design and innovation

Three measures to further spur eco-design within EPR were discussed (Table 5.2). As illustrated by the descriptive statistics, experts thought PROs introducing binding agreements would most likely contribute to the CE goals (statement 3), whilst mandatory harmonised LCAs would be least likely to do so (statement 1). Overall, none of the statements were perceived very highly: all statements were considered less than "relatively likely" to contribute to the CE goals.

The in-practice effectiveness of binding agreements was questioned (statement 3). For instance, one expert pointed out that secondary materials face more product safety legislation and tough(er) competition. This can make secondary materials less suitable and more expensive than virgin materials, which, it was argued, cannot simply be solved by establishing binding agreements. Moreover, experts were concerned that agreements do not provide enough financial incentives for stakeholders to change their behaviour fundamentally: ""*without "incentives" (e.g. financial), this will typically remain an individual rather than a collective commitment by producers*" (producer, EEE, Phase 4).

As seen in Table 5.2, experts considered differentiated EPR fee systems (statement 2) to be between "neutral" and "relatively likely" to contribute to the CE goals and neutrally desirable for the relevant sectors. The introduction of differentiated EPR fees has been suggested in various contexts (cf. European Commission, 2018b; Hogg et al., 2020; OECD, 2016). Nevertheless, how effective they are remain to be proven. Differentiated fees were discussed further during the expert workshop, where it became clear that the experts see them as a valuable and acceptable measure.

TABLE 5.2 Statements and results phase 3 concerning EPR, eco-design and innovation.

Statement & #	Likelihood of contribution to CE (1 – 5 scale): Mean & Stand. dev.	Desirable in my sector (1 – 5 scale): Mean & Stand. dev.	Source
1. All producers conduct mandatory harmonised Life Cycle Assessments (LCAs) of products	3.0; 1.2	2.9; 1.4	(Valente et al., 2019)
2. Government introduces differentiated EPR fee systems based on sustainability criteria	3.3; 1.2	3.0; 1.3	(Dubois et al., 2016; OECD, 2016)
3. PROs should introduce binding agreements on percentage of recycled content in new products	3.7; 1.2	3.7; 1.2	(Esenduran et al., 2019; Sugeta and Shinkuma, 2013)

N = 27; statements in bold are those deemed most important; statements in italics are those deemed most controversial

Considering the various clusters of experts (see supplementary materials 2.C of Campbell-Johnston et al., (2021)), it is noteworthy that experts representing PROs and knowledge institutes were least positive about these three measures (average means: 2.65 and 2.97), while producers were most positive towards them (average mean: 4.06). In particular, producers were most favourable towards differentiated EPR fee systems (statement 2) and, along with processors, towards binding agreements to work towards more circular products (statement 3). Regardless, individual producers pointed out the significance of costs and financial incentives in establishing agreements beyond current targets. One producer stated, "*establishing agreements beyond current targets normally means additional costs, which is difficult to turn into short-term competitive advantages*" (producer, EEE, phase 4).

5.4.2 Cost allocation and incentives

The statements in Table 5.3 address issues of costs and incentives for the existing EPR actors. The experts were more receptive to using the EPR fees for consumer campaigns (statement 4) and R&D (statement 5) than allowing PROs to manage these funds independently (statement 6). Interestingly, experts representing the government were very positive towards consumer campaigns (statement 4, means Qa and Qb: 5.0 and 5.0), while PROs were opposed (means: Qa and Qb: 2.38 and 2.83). However, PROs were happy to spend a percentage of the fees on R&D (statement 5, means Qa and Qb: 4.47 and 4.47), along with processors (means Qa and Qb: 4.73 and 4.73). This discussion raised further questions on designating EPR fees to R&D. For instance, one expert asked: "*Who is entitled to decide? What is the goal or aim? What to do with the outcome? Who is responsible and who is accountable?*" (EEE, producer, Phase 4).

TABLE 5.3 Statements and results phase 3 concerning cost allocation and incentives.

Statement & #	Likelihood of contribution to CE (1 – 5 scale): Mean & Stand. dev.	Desirable in my sector (1 – 5 scale): Mean & Stand. dev.	Source
4. PROs use fees for consumer campaigns	3.6; 1.4	3.6; 1.2	Based on workshop, consultation and (European Commission, 2014c)
5. Percentage of fees to PRO to be spent on R&D	3.8; 1.2	4.0; 0.9	(European Commission, 2014c; OECD, 2016), and consultation
6. R&D funds of PROs to be managed independently	3.0; 1.1	2.6; 1.5	Based on workshop
7. Government introduces VAT reduction or exemption for repair and recycling	3.6; 1.2	3.8; 1.1	(Taranic et al., 2016)

N = 27; statements in bold are those deemed most important; statements in italics are those deemed most controversial

Independent management by EPR fees for R&D by PROs (statement 6) was not considered desirable by experts. During the discussions, some mentioned the advantages of putting an independent organisation in charge of such funds, e.g. increasing transparency and reducing conflicts of interest. Yet, a majority of the experts pointed to the disadvantages: "*The money is paid by the producers, so they should decide how the funds are spent*" (producer, flat glass, phase 4) and "*there is no independent organisation which has the (practical) know-how*" (PRO, EEE, Phase 4). It is noteworthy that producers were most positive towards independent management by PROs (means Qa and Qb: 3.91 and 2.69), while PROs themselves are opposed to such new responsibilities (means Qa and Qb: 2.21 and 1.41).

5.4.3 Targets and goals

Table 5.4 outlines statements addressing EPR targets and goals. On average, producers, processors, government and knowledge experts were positive about these five statements, with average means between 3.55 and 3.96 (see supplementary materials 2.C of Campbell-Johnston et al., (2021)). PROs were the least positive (average mean: 2.98). Two statements were perceived most favourably to contribute to CE and the sectors of the participants: PROs should establish binding agreements that go beyond current targets (statement 8), and the government should make the collection and recovery targets more specific and measurable (statement 9).

Both the government and PROs were positive about the desirability of the proposal in statement 8 for their sectors. While the government was furthermore convinced about its contribution to CE (mean: 4.78), PROs did not agree that such agreements would contribute to realising CE (mean: 2.99) (see supplementary materials 2.C of Campbell-Johnston et al., (2021)). Additionally, experts again emphasised that there must be a (financial) trigger to engage in such agreements. As stated by one expert: "*Without a commercial interest, no company will go beyond targets*" (EEE, processor, Phase 4). Experts warned that producers might be unwilling to participate due to the associated extra costs: "*Not a lot of willingness from participants to do more due to costs and organisational time consumption involved*" (EEE, PRO, Phase 4).

Experts from the government were most positive towards statement 9 that addressed the specificity of targets (see supplementary materials 2.C of Campbell-Johnston et al., (2021)). Experts engaged in a discussion about the organisation of more detailed targets, moving instead to targets based on quality, components or specific materials. On the one hand, experts stated that the government should set and enforce such targets, following by industry implementation. On the other, some experts emphasised cooperation with other stakeholders in target setting: "*Differentiation of targets to product groups might be interesting, but not to be set (solely) by the government … it's also up to the EPR organisation and its stakeholders*" (Floor coverings, knowledge, Phase 4). Experts also pointed out during the workshop that the government should connect with and involve various stakeholders

from relevant supply chains. While government officials were specifically in favour of such targets, they also raised doubts about the implementation: "*Such targets are desirable, but difficult to implement, measure and enforce*" (Government, Phase 4).

Statements 10 and 11 were also received relatively positively by the experts. Interestingly, the introduction of product reuse targets for PROs by the government was heavily opposed by both PROs (means Qa and Qb: 2.21 and 2.45) and the government (means Qa and Qb: 2.77 and 2.77). Statement 12 relates to introducing mandatory EPR schemes for more waste streams, a proposal currently being developed at the EU level concerning single-use plastics and in the Netherlands for mattresses. This statement was received well by producers (means Qa and Qb: 4.42 and 4.22), but received substantial opposition by PROs (means Qa and Qb: 2.78 and 2.34) (see supplementary materials 2.C of Campbell-Johnston et al., (2021)).

TABLE 5.4 Statements and results phase 3 concerning targets and goals.

Statement & #	Likelihood of contribution to CE (1 – 5 scale): Mean & Stand. dev.	Desirable in my sector (1 – 5 scale): Mean & Stand. dev.	Source
8. PROs establish binding agreements beyond current targets	4.1; 0.9	4.3; 0.9	(European Commision, 2015)
9. Government makes collection and recovery targets more specific and measurable	3.8; 1.1	3.9;0.9	(Watkins et al., 2017)
10. EPR targets to be updated every 5-7 years by the government	3.4; 1.1	3.4; 1.2	(Coopman, 2014)
11. Government introduces & updates product reuse targets for PROs	3.3; 1.2	3.0; 1.2	(European Commision, 2015)
12. Government introduces mandatory EPR schemes for more waste streams	3.7; 1.2	3.4; 1.1	(Ministry of Infrastructure and Water management, 2020; Ministry of Infrastructure and Water Management, 2017) and based on workshop

N = 27; statements in bold are those deemed most important

5.4.4 Actor inclusiveness

These statements focused on the inclusiveness of relevant actors and their roles and responsibilities (Table 5.5). Of the proposed statements, experts suggested that clarification of specific roles and responsibilities of value chain actors by the government would be most likely to contribute to the realisation of the CE goals and would also be most desirable for the sectors they work on or in (statement 13). Nevertheless, none of these statements were brought forward in Phases 4 or 5.

TABLE 5.5 Statements and results phase 3 concerning actor inclusiveness.

Statement & #	Likelihood of contribution to CE (1 – 5 scale): Mean & Stand. dev.	Desirable in my sector (1 – 5 scale): Mean & Stand. dev.	Source
13. Government clarifies specific roles/responsibilities of value chain actors	3.6;1.1	3.7; 1.2	(European Commission, 2014c)
14. Social economy actors should have formal role in PROs	3.0; 1.2	3.0;1.2	Based on workshop and consultation
15. Government reduces administrative requirements of EPR schemes for SMEs	2.8;1.1	2.7;1.1	(OECD, 2019)

N = 27; statements in italics are those deemed most controversial

Experts were neutral about social economy actors having a formal role in PROs (statement 14). In their inputs, experts were mainly divided on such a role. On the one hand, "*A 'formal' position is not required; involvement is*" (EEE, PRO, phase 4). On the other hand, "*they should have their own defined responsibilities and targets to be set by government so that they are really judged on their claimed apparent positive contribution in the field of EPR*" (EEE, PRO, Phase 4). This division also became apparent within the specific groups of experts. While experts representing the government, knowledge institutes and processors were relatively positive towards a formal role, neither PROs (means Qa and Qb: 2.63 and 2.63) nor producers (means Qa and Qb: 2.49 and 2.99) thought that a formal role for social economy actors would contribute to CE or would be desirable for their sectors. One producer argued: "*producers are responsible and should therefore be able to decide whether in their specific case social economy actors should have a formal role*" (EEE, producer, Phase 4).

Finally, experts were least positive on the statement that suggested that the government should reduce administrative requirements related to EPR for small and medium-sized enterprises (SMEs) (statement 15). Several arguments were brought forward to retain the current administrative requirements: "*Administrative requirements relating to EPR are currently not high … even for SME*" (EEE, processor, Phase 4) and "*all companies have an equally balanced responsibility for products brought on the market. Why should larger companies pay for the smaller ones*?" (EEE, PRO, Phase 4).

5.4.5 Reporting and transparency

All statements in Table 5.6 concern reporting and transparency were received positively. In particular, experts believed that two statements would be more than "relatively likely" to contribute to the realisation of the CE goals: using common definitions and standards within EPR (statement 17) and PROs providing information on the collection rate, types of recycling and final destinations of products (statement 18).

While there was agreement that common definitions and standards should be used within EPR, doubts were expressed about their in-practice effectiveness. After all, establishing common definitions and standards does not necessarily mean that they will be used (effectively) in practice. Processors were most positive about common definitions and standards (means Qa and Qb: 5.0 and 5.0). PROs were more sceptical about their effectiveness (means Qa and Qb: 3.31 and 4.16). For instance, one PRO expert indicated that "*not all schemes and products are easy to compare*" (Cars, PRO, Phase 4), so it would be difficult to establish and enforce such definitions and standards. Several experts suggested various institutions to coordinate the common definitions and standards: the EU, the OECD, or a task force designated to this task.

TABLE 5.6 Statements and results phase 3 concerning reporting and transparency.

Statement & #	Likelihood of contribution to CE (1 – 5 scale): Mean & Stand. dev.	Desirable in my sector (1 – 5 scale): Mean & Stand. dev.	Source
16. Producers adopt standardized labelling of products	3.5; 0.9	3.5; 1.0	Based on workshop
17. All EPR schemes and participants use common definitions and standards	4.1;1.0	4.5; 0.7	(Kunz et al., 2018; Pouikli, 2020)
18. PROs provide info on collection rate, types of recycling and final destinations	4.2;1.0	4.4;0.9	(Pouikli, 2020; Watkins et al., 2017)

N = 27; statements in bold are those deemed most important

Providing more information on recycling activities increases transparency towards stakeholders (statement 18). This is even more crucial because globalisation and interests have led to rising imports and exports of products and waste. At the same time, experts identified the current waste trading practices as the primary obstruction for providing more information. As stated by one expert, "recycling data are (more or less) available, (reliable) export data not" (EEE, producer, phase 4). Some experts stated that such information is available but unreliable, particularly when it concerns products exported outside the EU: "*the problem is not the availability of this information, but its reliability. Many companies in emerging markets … will promise anything if it is needed to protect their production*" (EEE, processor, Phase 4). Government officials favoured the reporting of additional information (means Qa and Qb: 4.78 and 5.0), while PROs themselves expressed their concerns about such measures (means Qa and Qb: 3.13 and 2.63). As stated by a PRO expert: "*PROs can provide information about collection rates and recycling, but not on final destinations of exported products. As long as a product is a product, a PRO has no competence to regulate anything; the owner/trader is in charge*" (Cars, PRO, Phase 4).

5.4.6 Treatment choices

Statements in Table 5.7 concern treatment choices and recycling practices. Experts were neutral about the contribution of these statements to the realisation of the CE goals and their desirability for the relevant sectors. Some differences can be observed between the specific groups of experts.

Several doubts were raised about LCAs being conducted periodically by an independent organisation (statement 19), which is paid by the PROs. Amongst others, experts argued that conducting LCAs for products with a long lifetime can be challenging; conducting periodical LCAs could become quite costly, and it is uncertain whether more assessments and research would enhance circular activities. Moreover, experts from PROs and producers stated that this "*should be regulated in EU legislation*" (Cars, PRO, Phase 4) and that "*an EU tool with this methodology should become available, and the outcomes should be kept in an (open access) database*" (EEE, processor, Phase 4). Current policy developments in this area include standardised product category rules. Furthermore, the majority of experts agree that the government should incentivise greater source separation (statement 21).

TABLE 5.7 Statements and results phase 3 concerning treatment choices.

Statement & #	Likelihood of contribution to CE (1 – 5 scale): Mean & Stand. dev.	Desirable in my sector (1 – 5 scale): Mean & Stand. dev.	Source
19. Determine preferable treatment options based on independent LCAs paid by PROs	3.0; 1.0	3.1; 1.2	(Campbell-Johnston et al., 2020a)
20. Government updates EPR treatment requirements every 2-4 years	3.1; 1.2	3.2; 1.2	(European Commision, 2015; OECD, 2016)
21. Government incentivises source separation of waste streams by last users (e.g. deposit scheme)	3.2;1.3	3.0;1.3	(European Commision, 2015)

N = 27; statements in italics are those deemed most controversial

5.4.7 Monitoring and enforcement

Statements in Table 5.8 concern monitoring and enforcement. In general, all groups of experts were neutral to positive about greater enforcement (average means between 3.01 and 3.69). More specifically, experts believed that it would benefit the CE goals and their sectors if the government were to force free-riders to join EPR schemes (statement

22). However, doubts were expressed about the government's capabilities to address this issue. For instance, "*government lacks market knowledge*" (EEE, PRO, Phase 4). Therefore, a few experts recommended that authorities should have more competencies and market knowledge. Other measures to tackle free-riders were proposed, such as naming and shaming and highlighting successful EPR practices and examples (OECD, 2019).

TABLE 5.8 Statements and results phase 3 concerning monitoring and enforcement.

Statement & #	Likelihood of contribution to CE (1 – 5 scale): Mean & Stand. dev.	Desirable in my sector (1 – 5 scale): Mean & Stand. dev.	Source
22. Government forces free-riders (like internet sellers) to join EPR schemes	4.0;1.1	3.7;1.1	(OECD, 2019; Wilson et al., 2011)
23. Government intensifies enforcement of waste shipment to countries not meeting EU standards	3.5;1.2	3.8; 1.1	(EUROSAI, 2013)
24. Government fines PROs for the percentage of products they failed to collect/process	2.9; 1.4	2.8;1.3	(Leclerc and Badami, 2020)
25. Government extends EPR to recycling and circular treatment of second-hand goods, in- & outside EU	3.3; 1.3	2.7;1.5	(Kunz et al., 2018)

N = 27; statements in bold are those deemed most important; statements in italics are those deemed most controversial

Experts were unfavourable to the introduction of fines for the percentage of uncollected products by PROs (statement 24). Unsurprisingly, PROs were least positive about this measure (means on Qa and Qb: 1.68 and 2.45). It was pointed out that PROs have limited influence over producers and other stakeholders: "*it's not that the PRO's do not try to meet targets, it is that many other actors intervene*" (EEE, PRO, Phase 4) and "*there are too many factors out of direct control of an EPR organisation*" (Floor coverings, PRO, Phase 4). Furthermore, processors also expressed their discontent about this measure (means on Qa and Qb: 2.38 and 1.86). As stated by one processor: "*The statement is too black and white. In some cases, targets cannot be met as they are not realistic or for other reasons. Targets and achievement should be monitored and actions to improve should be decided upon*" (EEE, PRO, Phase 4). Exerts also offered alternatives to this measure, such as providing positive collection and processing incentives and establishing closer cooperation and structural dialogues between involved stakeholders.

5.5 Discussion

This research used a Delphi study to examine how to transform EPR to contribute to the CE goals of the Netherlands. It builds on the work of Kunz et al. (2018), exploring perspectives and ways forward for EPR but connecting it explicitly to the development of CE. Seven of twenty-five potential activities proposed in this study were delineated as having a greater agreement regarding their propensity to contribute to the CE goals (see Table 5.9). These proposals provide direction for greater economic responsibility, e.g. R&D research; informative responsibility, e.g. transparency and specify of reporting; and physical responsibility, e.g. more specific goals.

TABLE 5.9 EPR activities regarded as contributing to CE.

Statement & #	Likelihood of contribution to CE (1 – 5 scale): Mean & Stand. dev.	Desirable in my sector (1 – 5 scale): Mean & Stand. dev.
3. PROs should introduce binding agreements on percentage of recycled content in new products	3.7; 1.2	3.7; 1.2
5. Percentage of fees to PRO to be spent on R&D	3.8; 1.2	4.0; 0.9
8. PROs establish binding agreements beyond current targets	4.1; 0.9	4.3; 0.9
9. Government makes collection and recovery targets more specific and measurable	3.8; 1.1	3.9;0.9
17. All EPR schemes and participants use common definitions and standards	4.1;1.0	4.5; 0.7
18. PROs provide info on collection rate, types of recycling and final destinations	4.2;1.0	4.4;0.9
22. Government forces free-riders (like internet sellers) to join EPR schemes	4.0;1.1	3.7;1.1

Nevertheless, the transition to CE will require a profound change in the behaviour of EPR stakeholders via the assignment of new responsibilities. The majority of the activities above concern changes and modifications to the existing EPR system, except the statement to create agreements for more circular products. Yet, as indicated in the results, actors brought a wide variety of (competing and divergent) perspectives, interests and agendas to the study. Whilst experts agreed on the necessity of transitioning to CE, tensions emerged over what these responsibilities *should be* and *to whom* they should apply. Namely, who is and should be responsible for what? In the original conceptualisation of EPR, Lindhqvist (2000) proposed five means through which 'responsibility' could be extended to producers. The EU formalised two of these: economic and physical responsibility. The results from the Delphi indicate that new responsibilities for changing EPR to support CE are contested, often when this involves incumbent actors accepting new responsibilities. For instance, experts representing the PROs were less favourable towards consumer campaigns (statement 4),

providing additional data on recycling destinations (statement 18) and including new actors (statements 13-15). Whilst some actors were favourable towards measures that would not affect their own practices. For example, experts representing the producers were generally favourable to statements concerning eco-design and product innovation (Table 5.2). Yet, none of these specified new responsibilities for producers themselves. These results connect to previous writings on the complexities of dividing responsibilities within EPR systems and the necessity of a clear regulatory framework (Kalimo et al., 2015; Kunz et al., 2018).

These results echo earlier claims on how to improve EPR. For example, previous research has stressed the need for greater enforcement to prevent free-riders, harmonising legislation and responsibilities between member states and greater transparency of waste data (Kalimo et al., 2015; Kunz et al., 2018). These points were evident from the Delphi outcomes (see Table 5.9). Moreover, statement 9 related to moving beyond EPR recycling targets based on mass to more dynamic ones. The proposal to adapt EPR targets in this manner similarly reiterates previous research and proposals (cf. Ortego et al., 2018; Parajuly & Wenzel, 2017; Wilts et al., 2011). However, the results from the Delphi did not support proposals for common treatment standards for collected products (Kunz et al., 2018). Yet, the results did provide new insights not generally discussed in the literature. For example, the need for financing preferable EoL activities through the mechanism of EPR, the perspectives of some participants who pushed for balanced responsibility between all actors, including SMEs (see section 5.4.5), and the need for greater transparency in the reporting requirements of ERP systems, e.g. final destinations of collected products.

Whilst the seven activities above likely represent a starting point from which to develop EPR to further contribute to CE, they must be put into perspective. Collectively, they represent a point of agreement and negotiation between divergent groups. Consequently, they are unlikely to represent a radical point of departure for transforming EPR, given the strong evidence of interest-related positions that emerged. While actors agreed in essence on change and new responsibilities, negotiating these was different, with 'agreement' falling within areas within current policy discussions, e.g. common definitions fall within Directive 2018/851. This finding complements earlier research on material governance within CE, showing how the intention and motivation for circularity are interpreted by various actors and consequently contested based on self-interests (Deutz et al., 2017).

The necessity of pursuing CE practices is, ultimately, the reduction of material inputs (Ghisellini et al., 2016). EPR was originally conceived as a means of increasing sustainability through the entire product lifecycle (Lindhqvist, 2000), yet, regulatory approaches prioritised EoL aspects without making the connection to eco-design concrete (Deutz, 2009; Kunz et al., 2018). One measure to connect EPR more explicitly with eco-design is through fee modulation (cf. Hogg et al., 2020; OECD, 2016), with France a leading country in this area. During the workshop, eco-design and modulation of EPR fees were suggested to provide financial triggers for participants, with experts discussing how eco-modulation of EPR fees

had furthered the circularity of packaging materials. This insight was surprising given the group's neutrality towards eco-modulation (see statement 2 and supplementary materials 2.C of Campbell-Johnston et al., (2021)). This evidenced a disconnect between the experts and current policy developments. However, it can further point to a tension between visions of EPR, either as purely an EoL policy tool or one to stimulate product innovation. For example, the reluctance of the PROs to expand the scope of EPR to include additional actors (statements 13-15) could be attributed to this. This tension on the scope of EPR is still debated intensively within academic research (see Atasu, 2019).

As this study illustrates, much of the discussion on EPR is still framed within the EoL arena, reflecting earlier criticisms of CE (Gregson et al., 2015). We suggest that the limited attention to strengthening the connection between EPR and product design and materials inputs needs to be overcome for EPR to contribute to a more transformational CE vision. This statement reflects earlier scientific critiques of the limitations of EPR (Deutz, 2009), with more recent research outlining more concrete pathways for how EPR can contribute to CE, including through fee modulation and interconnection with EPR and other policy approaches (see Vermeulen et al., 2021).

This study has several limitations. First, the panel selection, which is vital to the success and outcome of any Delphi (Okoli and Pawlowski, 2004). Despite following a clear typology and selection procedure for experts, the loss of participants in each phase diminished the richness of the data. As the intention of a policy Delphi is to facilitate a discussion process, any loss of a participant reduces this debate. However, as Phases 4 and 5 contained representation from all groups, we are confident in the variety of perspectives (Table 5.1). Second, a Delphi study is, by nature, an expert study, meaning it does not seek population representativeness. Whilst such a typology of actors was necessary from a knowledge perspective, it did not mean that said actors would approach the questions from a neutral perspective, i.e. thinking beyond the interests of their organisation or sector. Consequently, whilst this study intended to explore ways of "transforming" EPR to meet the CE goals, statements that were signified collectively as important (Phase 3) were those less likely to be controversial from any of the actors, e.g. industry or government (see additional analysis of Phase 3 supplementary materials 2.B). This implies that many of the discussed outcomes were not as radical as the intention of the study assumed. Whilst the typology of experts impacted the types of results presented, we contend that the inclusion of more "controversial" statements in the latter phases provided a means of eliciting a more varied discussion in Phases 4 and 5. Finally, a critical limitation in the design of this study concerns the formulation of the statements themselves. Namely, each statement specified both an activity and a responsible actor. Despite these statements being based on the literature and reviewed during the Delphi, an underlying question relates to whether more sceptical reactions pertained to the proposed activity or the actor. Future research on CE and EPR should provide more research and support into the seven statements. For instance, how to explicitly specify EPR targets.

5.6 Conclusion

This study conducted a policy Delphi to explore stakeholder perceptions of transforming EPR to contribute to CE goals of the Netherlands. Through four phases, experts reflected on improving EPR to support the development of the CE further. Seven potential areas for future improvement of EPR were outlined, which would bring existing EPR schemes above and beyond the current minimum requirements that apply to such schemes.

The transition to a CE requires a substantial change in the roles and behaviours of actors within the value chains of products. Moreover, the legal and economic frameworks surrounding the value chains ought to be reorganised as well. This research points specifically to transformational changes within EPR policies concerning targets, goals, incentives, transparency and reporting. By implementing such changes, EPR schemes would potentially incentivise the use of more eco-design principles in products at the beginning of the value chain and lead to a higher quantity and quality of recycling at the end of the chain. At the same time, however, the perceptions of experts in the policy Delphi indicate that agreement on the assignment of new responsibilities is contested. Furthermore, the experts point to various issues that may hamper such transformation of EPR schemes.

EPR has been implemented in a wide variety of context and configurations over the past 30 years. This study focused on a more narrow understanding of EPR, which is explicitly connected to responsibility (economic and physical) for the post-user phase of a product's lifecycle. The practice in the Netherlands is based on this more narrow approach to EPR, where market actors have a limited set of responsibilities. Since its inception, EPR has faced continuous criticism for failing to deliver product eco-design changes and allocating insufficient individual responsibility to producers. This study echoes those critiques, as the discussions between experts overtly focused on the EoL aspects of EPR. Given this, future research should investigate how waste practices, such as EPR, and product design requirements, such as eco-design, can be further integrated and connected. This has implications for the development of EPR within CE. Namely, the need for cross-boundary and transnational cooperation to strengthen these links.

6

This chapter is based on Campbell-Johnston, K., Roos Lindgreen, E., Mondello, G., Gulotta, T.M., Salomone, R., Vermeulen, W.J.V. Thermodynamic rarity of electrical and electronic waste: Assessment and policy implications for critical materials. *Submitted to the Journal of Industrial Ecology (under review)*.

Earlier versions of this paper were presented at the 2021 European Roundtable for Sustainable Production and Consumption in Graz Austria, and the 2021 Italian Network LCA Association in Reggio Calabria, Italy. Insights of this paper were also presented to the OECD working group on Circular Economy and Waste, the Dutch Environmental Assessment Agency (PBL) and the Dutch executive agency Rijkswaterstaat.

Thermodynamic rarity of electrical and electronic waste: Assessment and policy implications for critical materials

Abstract

The strategic relevance of extracting raw materials from waste from electrical and electronic equipment (WEEE) in the EU is increasing due to value chain risks caused by geopolitical instability, accessibility of specific minerals and decreasing reserves due to growing extraction rates. This article examines the quantities of so-called critical raw materials originating within WEEE streams from a depletion perspective. Presently, current recycling targets are based solely on mass collection and recycling rate. We examine the potential limitations of this approach using an exergy based indicator named thermodynamic rarity. This indicator represents the exergy costs needed for producing materials from the bare rock to market. The case of Italy is used to explore the application of the indicator at the macro (national) and micro (company) level for the product categories, small electronics and screens and monitors. Our results show significant differences between the mass and rarity of materials within Italian WEEE streams. While Iron accounts for more than 70% of the weight of the product categories analyzed, it accounts for less than 15% of the rarity. Similarly, several critical raw materials with a small mass have an increased rarity, e.g. Tungsten with less than 0.1% of the mass and over 6% of the rarity. The policy context is reflected upon, where it is argued that Thermodynamic rarity is a suitable indicator to support end-of-life WEEE decision-making processes, e.g. target development and recycling standards setting, to help prioritize material monitoring and recovery options.

6.1 Introduction

6.1.1 Background

The strategic relevance of extracting raw materials from waste from electrical and electronic equipment (WEEE) in the EU is increasing due to value chain risks caused by geopolitical instability, accessibility of specific minerals and decreasing reserves due to growing extraction rates (Bobba et al., 2020). Present consumption patterns raise the issue of future demand bottlenecks for certain materials, depletion and future unavailability unless actions to both conserve and increase recycling efforts are taken (Henckens, 2021; Valero et al., 2018).

Among those elements, the so-called 'critical raw materials' (CRM) are of particular interest to the EU as they represent materials with a high supply risk and a high economic value, e.g. Indium and Cobalt[16] (European Commission, 2020d). In addition to CRM, geologically scarce materials are those that are characterized by a relatively low concentration in the crust and are thus at the risk of early exhaustion, i.e. due to physical availability and demand, e.g. Copper and Antimony (Henckens, Driessen, et al., 2016; Ortego, Valero, Valero, & Restrepo, 2018). Despite the EU outlining the importance of CRM in recent years (European Commission, 2020d), they are not explicitly integrated within general EEE or WEEE legislation yet (Horta Arduin et al., 2020). In this context, (W)EEE is currently governed under two key bodies of legislation specifically related to product design and WEEE. In particular, "product design" is regulated through the Eco-design Directive 2009125/EC (European Parliament (EP) and European Union 2009) and Energy Labelling regulations 2017/1369 (EPt and The Council Of The European Union 2017), which set guidance primarily on energy labelling and in-use efficiency measures. Whereas, WEEE is governed under Directive 2002/96/EC (EP and Council 2002) and its recast 2012/19/EU (EP and Council 2012), which employs the principle of Extended Producer Responsibility (EPR) by giving market actors either organizational or financial responsibility for the collection and recycling of WEEE. Other noticeable pieces of legislation include the Waste Framework Directives (European Commission, 2008; WFD 2018/851, 2018a), which sets further requirements on EPR, and chemicals legislations e.g. the REACH regulation (European Parliament, 2006), which details, monitors and restricts the use of certain substances. Besides, the CENELEC EN 50625 Standards assist WEEE actors in fulfilling the previously mentioned Directive's requirements. Due to the focus on mass, both in the collection and recycling of WEEE, there is a broader issue of whether the end-of-life (EOL) stage appropriately considers and integrates elemental scarcity, both in monitoring and prioritization in recovery operations (Horta Arduin et al., 2020).

The ambitions of European circular economy (CE) policies center around stimulating more efficient use of natural resources and preventing resource overuse through various

16 These include geopolitical issues, e.g. country origins of specific materials. In this research, we do not examine the socio-economic constraints on CRM use and recovery, but focus on the issue of physical availability.

strategies, including EPR (European Commission, 2020). The quantities and total demand of specific elements (against their use within other products) vary depending on the product in question, e.g. 88% of Indium was used in EEE, Antimony was 41%, and Terbium was 88% (European Commission, 2018c). CRM material flows have been mapped within the EU (see Saurat and Bringezu 2008; Bobba et al. 2020). WEEE was identified as a significant source of CRM, while higher recycling efforts have been demanded to mitigate long-term bottlenecks (European Commission, 2018c; Valero et al., 2018). Conservation (either through product lifetime extension or recycling) can also mitigate the broad array of socio-environmental damages associated with mining (cf. Marcantonio et al. 2021; Tsurukawa et al. 2011). Yet, the recycling of geologically scarce and CRM differs per material due to a combination of low collection rates, varying efficiencies of suitable technologies, product design issues, e.g. dissipative losses (Ciacci et al., 2015), low concentrations in products, and a lack of secondary markets (Bobba et al., 2020).

6.1.2 Research gap and aim of this study

Van Nielen et al. (2022) studied the recyclability of CRM noting that industrial-scale recycling for most 'minor metals' (rare earth elements, precious metals, and speciality metals) is currently non-existent. Furthermore, considerable process losses might occur during the treatment of WEEE, which are not considered in the general legislation and method to calculate recycling rates put forward in Directive 2012/19/EU (Horta Arduin et al., 2020). In a study on WEEE recycling outcomes, Horta Arduin et al. (2020) showed that the recycling rates of materials varied significantly, with base and precious metals recovered more than CRMs. The authors stressed specific indicators for documenting such scarce materials. However, beyond documenting *what* elements are present in electronic waste streams, a more fundamental question of *which* elements should be prioritized (from a broader conservation perspective) for recovery and *how* this relates to WEEE policies is still lacking in the literature.

Although CRM considerations are included in the consultations of specific products, e.g. computer servers (European Commission, 2013b), the vital requirements that effects EEE are stipulated in the WEEE Directives. These include mandated collection and recycling targets for different WEEE categories. As of 2019, the current targets specify the collection of 65% of electronic products put on the market (calculated from the total weight collected and the average weight of electrical products put on the market for the previous three years). The Directive further specifies preparing for reuse recycling targets for specific product types, e.g. 70% (by weight) for screens and monitors and 55% for small equipment and information technology (IT) and telecommunications.

An alternative to mass-based targets and how to assess the priority of elements was analyzed by Ortego et al. (2018b, 2018a) using an exergy-based indicator termed *Thermodynamic rarity* (TR), applied to the recycling of automotive vehicles in Europe. TR is calculated by

assuming elements' exergy value for accounting for their relative abundance in the Earth's crust. The logic is that as demand for energy and materials increases and mineral reserves decrease, the need to conserve those materials that are more intensive and difficult to mine will grow (Valero and Valero, 2015a). The research used aggregated and estimated data. Still it allowed the authors to illustrate: i) the discrepancy between the mass and rarity of material and ii) the limitations of collection and recycling targets based on mass. The indicator illustrates *which* materials (from a conservation/depletion) perspective are rarer, and thus, valuable. Although the indicator has been applied broadly to the energy transition (Valero et al., 2021) and the composition of mobile phones (Torrubia et al., 2022), no study has it to the WEEE product category level, e.g. screens category.

In this context, the objective of this study is to apply the TR indicator to WEEE sector to evaluate the issues linked to WEEE population and recycling policies. To provide a more consistent evaluation, we opt for a dual approach to examine the WEEE generation at the macro and aggregate (national) level and further complement at the micro (company) level. In particular, in the first case, we use estimated national data, while in the latter case, we identify a sample among WEEE processed by a processing facility. Both the levels of analysis focus on the case of Italy. In particular, this research aims to (a) estimate the quantities of CRM (national and company level) and apply a TR assessment to compare the materials' mass and rarity. Based on results, we (b) reflect on the current policy practices for WEEE, including recovery targets. In essence, can TR be a useful tool to help monitor and prioritize which materials should be recovered by showing the hidden value of different materials? This study connects the technical on-the-ground flows of materials within a broader sustainability policy frame.

This study is structured as follows: after this introduction section, we expand on the selected theoretical approach of TR and the research design, including the case study and sampling approach (see Section 6.2). Then, the results section provides both the quantitative results of the analysis (mass and rarity comparison) as well as a critical policy reflection (Section 6.3) before we conclude (Section 6.4).

6.2 Materials and methods

6.2.1 Theoretical approach

This research follows the methodological approach first demonstrated by Ortego et al. (2018a, 2018b) and validated by Torrubia et al. (2022) to examine the rarity of different materials within the WEEE streams using an exergy-based indicator termed TR as proposed by Valero and Valero (2015).

As Valero and Valero (2012, 2013, 2015) outlined, *TR* provides a methodology to signify the exergy requirements needed to extract and process elements from the Earth's crust into a useful commodity or element. In this context, exergy is defined as the maximum amount of work that may be theoretically performed by bringing a resource into equilibrium with its surrounding environment through a reversible process (Perrot, 1998). All substances and elements have definable exergy against a defined external environment. In their research, Valero and Valero (2015) presented a 'dead state' of a hypothetical planet with exhausted resources termed Thanatia. Based on this reference point, they calculated the exergy costs needed to mine an element from this dead state to the current state of mineral deposits (named *exergy replacement costs),* and energy needed to extract and refine a mineral from current mineral deposits using current technologies to produce a useful commodity (named *embodied exergy costs)*. Estimates for the mineral quantities were based on Cox and Singer (1987). The sum of *exergy replacement costs* and *embedded exergy costs* is the TR. As argued by Ortego et al. (2018b, 2018a), a mineral is regarded as valuable for two reasons: 1) they are scarce (physical availability) and/or 2) they are expensive to obtain with regard to extraction costs. This latter point relates to the energy requirements to obtain and refine a specific mineral to make a useful element through beneficiation and refinement. Such costs increase when ore grades decline.

In essence, TR allocates a value (exergy) to minerals and elements based on their relative abundance in the Earth's crust and the energy intensity needed to refine them. The geological occurrence of high concentrations of elements within mines (i.e. the elements not being lightly dispersed) saves large quantities of energy during extraction (Ortego et al., 2018b; Valero and Valero, 2012). Geologically scarce or rare elements generally have a higher TR value due to their dispersed quantities and, therefore, higher extraction, beneficiation and refining costs.

In contrast to other indicators that focus on material assessments, such as Material Footprints (Sen et al., 2019; Wiedmann et al., 2015), Raw Material Equivalents (Eurostat, 2001) or Ecological footprints (Rees, 1992), TR reflects the physical criticality of mineral resources through a consistent exergy measurement, indicating which materials (from a long-term conservation perspective) are most valuable. Given the emphasis on physical rarity at the elemental level, we adopt it for this study.

TR should not be conflated with recyclability. The former allocates a value (exergy) to the elemental composition of a product based on its physical abundance in nature and the exergy needed to extract and refine them. The comparison between recycling requirements and TR has been theoretically explored by (Valero et al., 2021; Valero and Valero, 2020). However, this research does not consider the exergy requirements needed to extract specific elements from WEEE as a means of comparison was not available. Instead, we examine the application of the indicator but in the novel context of WEEE EU policies. While the technical constraints of CRM recycling are not assessed in this study (cf. CEWASTE

2021), the indicator provides insights into the rarity of the elements currently available at the point of generation and treatment, and thus enables a more dynamic view on the issues within EU policy.

6.2.2 Research design

6.2.2.1 Selected case study

WEEE recycling in the EU generally is carried out according to the following consecutive processes: after collection/sorting and transportation to a treatment plant, WEEE goes through manual and/or mechanical separation and/or dismantling (CEWASTE, 2021). The following process includes the shredding of equipment into small pieces, electromagnetic separation or optical sorting, and finally, separation of ferrous and non-ferrous metals as well as plastic and glass fractions. PCBs can undergo additional processes to recover the rare materials they contain (Dutta et al., 2018). For a clear overview of the specific steps, standards and procedures for EoL of WEEE see CENELEC (2016).

This research used a case study of Italy to examine the application of the rarity indicator in the context of CRM within WEEE. A case study is an in-depth examination of a specific case in question (Yin, 2003); while not allowing for generalizable results, it enables novel insights to be raised and explored, i.e. the impact of WEEE policies. The case of Italy is examined on two levels: macro (national) and micro (company) to compare the mass and rarity of materials within WEEE streams. The latter was selected due to the availability and accessibility of the company case. The national-level used aggregated data on WEEE generation (section 6.2.2.2), while the company level examined specific WEEE entering a processing facility. Detailed data of which elements are recovered in Italy was unavailable, meaning our macro analysis focused only on the composition of WEEE generated. This complementary approach allows a more dynamic understanding of the differences between policy and national and on the ground effects.

This study used a sampling approach that identified the products present in the waste streams at the company level. The sample was selected randomly among the boxes of WEEE that were sent to the treatment facility site in Italy. Applying these procedures was necessary because owing to the rapid innovation and varying lifespan of electronics (Bakker et al., 2014), the characterization of the WEEE population is a complex task (see Rigamonti et al. 2017). Indeed, all WEEE categories include a large number of devices characterized by a complex mixture of materials and components that changes over time and space in percentage and size in each WEEE category (e.g., fridge, lamps, monitors) as well as in similar equipment (Mählitz et al., 2020). In addition, although the collaboration with the WEEE processor allows underlining the main EoL treatment processes of EEE, only limited aggregate and detailed company inventory data were available. Therefore, the sampling approach centred around creating an accurate insight or snapshot into product flows and waste at a particular point in time.

Materials recovered from the company level treatment facility via recycling include Aluminum, Copper, Steel, Iron, Plastic and Glass. The company's recycling rate was estimated to be 96%[17]. PCBs are manually separated and sold for subsequent processing by a different company, from which no data was shared due to reservations over its sensitive nature. Thus, we assume, based on literature and research on PCB and rare materials processors in Europe, that elements recovered include: Copper, Nickel, Gold, Silver, Tin, Platinum and Palladium (Dutta et al., 2018; Sheng and Etsell, 2007). For an overview of the processes for rare element recovery from PCBs see Wang and Gaustad (2012).

6.2.2.2 Data gathering and analysis

Italy organizes and reports its WEEE under five categories, which differs to EU categories (Table 6.1). See Magalini et al. (2012) for an overview of WEEE in Italy. This study looked at *E*-R3 TVs and monitors and *E*-R4 mixed electronics product categories. These two categories were chosen due to CRMs presence and heterogeneous nature as they contain various components, e.g. base metals, PCBs (Rigamonti et al., 2017). Categories *E*-R1 and *E*-R2 were not chosen due to their inaccessibility at the processing plant. *E*-R5 was not treated at the site and was therefore not chosen. This choice of product categories informed the selection of data at the national level.

TABLE 6.1 Italian WEEE categories and how they align with those in the WEEE Directive.

Category name	Description	EU categories (Annex 4 WEEE Directive 2012)
E-R1	Cooling and freezing equipment	1 Temperature exchange equipment
E-R2	Large household appliances	4 Large equipment
E-R3	TVs and monitors	2 Screen, monitors.
E-R4	Mixed WEEE	5 & 6 Small equipment and small IT and telecommunications
E-R5	Lamps	3 Lamps

Company level data was gathered through a point-in-time sampling approach for *E*-R3 and *E*-R4. The *E*-R4 category contained a mixture of IT, consumer electronic, tools and toys, whilst the *E*-R3 category contained computer and TV screens and monitors (for a complete overview of all the specific products see the supplementary materials). Three samples were carried out at the company between December 2020 and May 2021, where the WEEE were sampled and categorized according to products, brands and models. Sampling times were partly limited and constrained by the Covid-19 restrictions in Italy during this period. In total, we collected an inventory of 680 products (90 *E*-R3 and 590 *E*-R4) with a total weight of 4285.8 kg (detailed data are in the supplementary materials).

17 This indicates the percentage of materials not sent to landfill, which is sent to national and EU reporting agencies. This number does not include further losses at subsequent processes stages.

For exploring the material composition of WEEE at the national and company level, the ProSUM dataset held by the United Nations University (UNU) (Huisman et al., 2017) was used. The UNU dataset includes an estimation of elemental composition and weight for the average product (within each EU member state) for 1980-2020. While certain elements in the dataset are characterized by low confidence due to the scattered and incomplete nature of data, it contains the most comprehensive overview of WEEE products' composition, and was thus chosen to be used (Huisman et al., 2017). Data on the national level included the estimated elemental composition for the WEEE generated in Italy (Huisman et al., 2017). We set the reference year at 2020[18], which was when this study commenced and selected the product categories based on the availability of data at the company level (see below).

After cleaning and organizing the sample data and cross-checking it with available online documents and technical reports on brands and models, the next step was to select the elements to be assessed. For this, we selected a number of elements based on the following two factors: 1) elements that the processing company extracted in the case study, e.g. Aluminum, Copper, Iron and 2) elements that are often present in electronics that are considered either 'rare and critical' or (geologically) scarce (Henckens, 2021; Horta Arduin et al., 2020). A complete overview of the selected elements and noticeable features are presented in Table 6.2.

All the elements presented above were used in the TR assessment. Due to the vast array of products and brands within the company sample, for some of which the technical reports were unrecognizable, it was assumed that year of production for products in the sample (according to Magalini et al. (2014) and Forti et al. (2018)), and therefore reference point in the ProSUM dataset, corresponded to the assumed average life of said product for Italy. The aggregated weights for each element for all R3 and R4 products are provided in the supplementary materials.

The TR assessment for each of the elements for the aggregated weights of each product category was calculated using Equation 1. The calculation procedures are provided in the supplementary materials.

$$\boldsymbol{R_A} = \sum_{i=1}^{n} \boldsymbol{m_i} \cdot \boldsymbol{R_i} \qquad \textbf{Equation 1}$$

where R_A is TR (kJ/g) of the product category *A* (*E*-R3 and *E*-R4). Instead, ***m*** and ***R*** represent respectively the mass (g) and TR - as calculated by Ortego et al. (2018b) and summarized in the supplementary materials - of the ith element assessed.

After applying Equation 1, we calculated the relative percentage of the TR of each element

18 In the dataset, the years 2016-2020 are all projected data, based on projected market shares. We decided as the focus of this research is on the usefulness of the thermodynamic rarity indicator, using projected data was acceptable, as, it was also done to compliment the more detailed company level data.

in the sample (per product category) against the total TR for the same product category. The analysis compares the differences of the relative percentages between a mass-based approach (as used in the WEEE Directive) and a TR approach for the materials in each product category. This was done at the national and company level. To provide clear communication, the results are presented into two clusters based on the element's mass share: one includes the materials below 0.1 wt% of the total weight, and the second the others. Presenting the mass and rarity of these two levels allows us to reflect on the use of the indicator and, in the case of the company level, the effect of recycling practices in the context of the EU policy regime.

TABLE 6.2 Overview of elements assessed in this study and summary of key information

Element name	Included in the 2020 EU Critical Raw Material list (European Commission, 2020d)	Ultimately Available Resources (estimated and rounded Mt) (Henckens, 2021)	Indicative exhaustion period (years after 2015) (Henckens, 2021)	End-of-Life recycling rate (estimated) (Graedel et al. 2011)
Aluminum (Al)	No	10,000,000	10,500	>50%
Antimony (Sb)	Yes	100	150	1-10%
Bismuth (Bi)	Yes	20	150	<1%
Chromium (Cr)	No	35,000	350	>50%
Cobalt (Co)	Yes	3000	1100	>50%
Copper (Cu)	No	10,000	100	>50%
Dysprosium (Dy)[2]	Yes	20,000	1200[4]	>1%
Gold (Au)	No	2	150	>50%
Indium (In)	Yes	30	250	<1%
Iron (Fe)	No	6,000,000	1100	>50%
Lithium (Li)	Yes	2000	1600	<1%
Magnesium (Mg)	Yes	3,000,000	40,000	25-50%
Molybdenum (Mo)	No	200	200	25-50%
Neodymium (Nd)[2]	Yes	20,000	1200[4]	>1%
Nickel (Ni)	No	8000	450	>50%
Palladium (Pd)[1]	Yes	3	5300[3]	>50%
Platinum (Pt)[1]	Yes	3	5300[3]	>50%
Silver (Ag)	No	20	150	>50%
Terbium (Tb)[2]	Yes	20,000	1200[4]	>1%
Tin (Sn)	No	300	700	>50%
Tungsten (W)	Yes	200	600	10-25%
Zinc (Zn)	No	30,000	400	>50%

(1) Platinum Group Metals are: ruthenium, rhodium, palladium, osmium, iridium, and platinum; (2) REE are scandium, yttrium, lanthanum, cerium, praseodymium, neodymium, samarium, europium, gadolinium, terbium, dysprosium, holmium, erbium, thulium, ytterbium, lutetium, and promethium; (3) Rare Earth Metal exhaustion rate are given collectively based on (M. L. C. M. Henckens, 2021); and (4) Platinum Group Metal exhaustion rate is given collectively for all metals. The chosen elements were selected from (M. L. C. M. Henckens, 2021; Horta Arduin et al., 2020).

6.3 Results and discussion

6.3.1 Mass and Rarity Comparison (national and company level)

The comparison between mass and rarity for both levels is provided in Tables 6.3 and 4. We present the results first on the national level. Next, we complement this aggregate analysis with more precise data at the company level.

6.3.1.1 National level

The mass and rarity results for the two product categories for WEEE generated in Italy are presented in Table 6.3 and visualized in Figure 6.1. The vast majority of the mass is accounted for by three metals: Aluminum, Copper and Iron. The major metals (those above 0.1% of the total mass share) account for 99.94% (*E*-R3) and 99.83% (*E*-R4) of the total mass share of the product category. However, applying the TR indicator reveals 14 materials with mass share below 0.1% that account for 21.07% (*E*-R3) and 4.57% (*E*-R4) of the total TR. The TR weighting for the *E*-R3 categories stands out as exceptionally high against the product category mass share of 0.06%. We examine several key differences in each product category below.

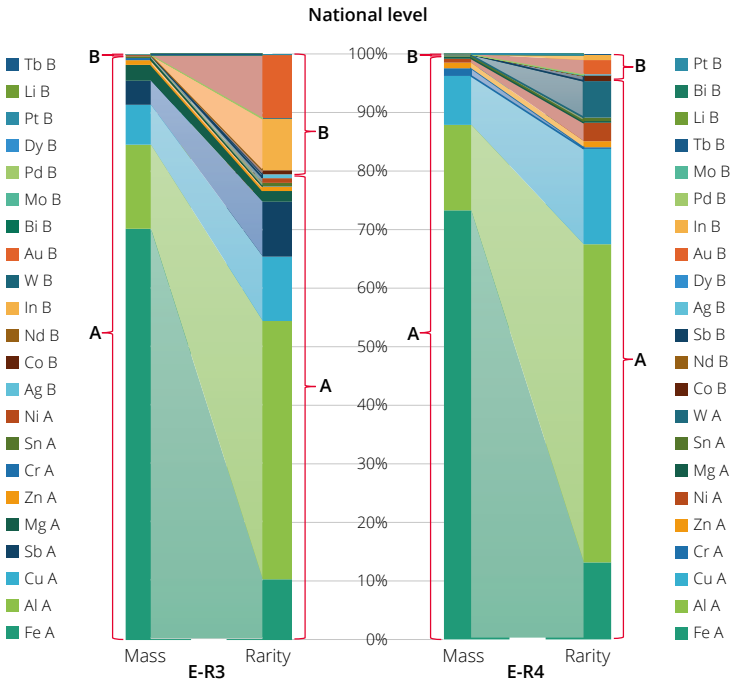

FIGURE 6.1 National level: A: Mass and rarity for product categories *E*-R3 and *E*-R4 for metals with a mass share above 0.1 wt%. B: Mass and rarity for product categories R3 and *E*-R4 for metals with a mass share below 0.1 wt%. Full names of the elements are provided in Table 6.2.

TABLE 6.3 Differences between mass and rarity at the national level. A = those elements above 0.1% of the total mass. B = those below.

National level							
E-**R3**				*E*-**R4**			
Elements	Mass	+/-	Rarity	Elements	Mass	+/-	Rarity
Fe (A)	70.2651%	-	10.3880%	Fe (A)	73.3262%	-	13.1472%
Al (A)	14.3861%	+	44.1394%	Al (A)	14.6037%	+	54.3409%
Cu (A)	6.7956%	+	10.9897%	Cu (A)	8.3260%	+	16.3297%
Sb (A)	4.1376%	+	9.3703%	Cr (A)	1.3282%	-	0.3058%
Mg (A)	2.7297%	-	1.8465%	Zn (A)	0.9263%	+	1.0269%
Zn (A)	0.8043%	-	0.7352%	Ni (A)	0.7284%	+	3.1082%
Cr (A)	0.3124%	-	0.0593%	Mg (A)	0.3425%	-	0.2810%
Sn (A)	0.2808%	+	0.5905%	Sn (A)	0.2446%	+	0.6238%
Ni (A)	0.2313%	+	0.8138%	W (A)	0.1385%	+	6.2567%
Ag (B)	0.0152%	+	0.6310%	Co (B)	0.0144%	+	0.8908%
Co (B)	0.0139%	+	0.7103%	Nd (B)	0.0086%	+	0.0325%
Nd (B)	0.0108%	+	0.0335%	Sb (B)	0.0068%	+	0.0187%
In (B)	0.0052%	+	8.7423%	Ag (B)	0.0035%	+	0.1785%
W (B)	0.0040%	+	0.1493%	Dy (B)	0.0008%	+	0.0033%
Au (B)	0.0035%	+	10.7844%	Au (B)	0.0007%	+	2.4652%
Bi (B)	0.0022%	+	0.0055%	In (B)	0.0004%	+	0.8997%
Mo (B)	0.0008%	+	0.0041%	Pd (B)	0.0002%	-	0.0000%
Pd (B)	0.0007%	-	0.0001%	Mo (B)	0.0001%	+	0.0005%
Dy (B)	0.0006%	+	0.0020%	Tb (B)	0.0001%	+	0.0003%
Pt (B)	0.0000%	+	0.0048%	Li (B)	0.0000%	+	0.0002%
Li (B)	0.0000%	-	0.0000%	Bi (B)	0.0000%	+	0.0001%
Tb (B)	0.0000%	-	0.0000%	Pt (B)	0.0000%	+	0.0901%

For *E*-R3, the most noticeable decrease in results is for Iron. Similarly, Antimony increased from around 4.1% mass to 9.3% rarity and Aluminum from 14.3% mass to over 44% of the total rarity. Several of those materials that accounted for less than 0.1% of the total increased significantly. These include Gold, which rose from 0.004% of the mass to 10.784% of the rarity; Indium, which increased from 0.005% of the mass to over 8% of the rarity; and Tungsten which increased from 0.004% of the mass to 0.149% of the rarity. Both Indium and Tungsten are CRM, which indicates their increased significance through a TR perspective. For *E*-R4, significant increases include Aluminum, 14.58% of the mass and 54% of the rarity; Nickel, 0.73% of the mass and 3.1% of the rarity; and Tungsten 0.14% of the mass and 6.25% of the rarity. Of the minor metals, Silver increased from 0.004% mass to 0.18% rarity and Gold from 0.001% mass to 2.46% rarity. Similarly to *E*-R3, Iron saw a noticeable drop from 73.23%

mass to 13.13% rarity. The results from the national level indicate a discrepancy between mass and rarity, particularly for several key materials, e.g. Tungsten and Indium.

6.3.1.2 Company level

Based on information obtained from the processing plant, Iron, Copper and Aluminum are recovered. As described earlier, we could not ascertain a definitive list of the specific materials recovered from the PCBs as a different company did these. According to previous studies (Dutta et al., 2018; Sheng and Etsell, 2007), we assume that the materials that are potentially recovered if the processes from these studies are applied are: Copper, Gold, Silver, Tin, Palladium and Platinum. The specific recovery rates of those materials and subsequent final quantities are unknown. Based on this, we can infer that the materials that are lost in this process include: Antimony, Bismuth, Chromium, Cobalt, Dysprosium, Indium, Lithium, Magnesium, Molybdenum, Neodymium, Terbium, Tungsten and Zinc. This equated to a loss of TR of 38% (*E*-R3) and 10% (*E*-R4). With the exception of Chromium, Zinc and Molybdenum all these materials lost are EU CRM. The losses of these materials raise an issue not only from preventing further resource depletion but also that many of these materials are essential for modern economies and emerging green technologies, e.g. Cobalt and Tungsten (Bobba et al., 2020; Tkaczyk et al., 2018).

The mass and rarity results of *E*-R3 and *E*-R4 categories are reported in Table 6.4 and visualized in Figure 6.2. The findings highlight that nine materials are above the 0.1 wt% mass share (Aluminum, Chromium, Copper, Iron, Magnesium, Nickel, Antinomy, Tin and Zinc). In particular, as shown in Figure 2, these major materials for *E*-R3 account for 99.95% of the mass, but 88.56% of the rarity. By comparison, the mass of the major materials for *E*-R4 total 99.82%, whereas the rarity accounts for 87.75%. The discrepancy in weighting between mass and rarity is seen clearly in Figure 2, where these 14 minor materials account for more than 11% and 12% of the total rarity of *E*-R3 and *E*-R4, but only 0.05% and 0.18% of the total mass. This highlights the increased importance and hidden value of rare materials when adopting a rarity assessment approach.

Examining the results in more detail, we observe large differences between the TR and mass of a number of specific materials. For instance, regarding waste category R3, Iron, which constitutes around 57% of the mass, translates to only 7.78% of the rarity. Additionally, Antimony which is likely not recovered in this process, accounts for 13.44% of the mass, while it shows to account for 27.95% of the rarity. Losing roughly 28% of the rarity materials present in this stream is undesirable from a CE perspective, where pressure on resource use is a major drive.

TABLE 6.4 Differences between mass and rarity at the company level. A = those elements above 0.1% of the total mass. B = those below.

Company level							
E-**R3**				*E*-**R4**			
Elements	**Mass**	**+/-**	**Rarity**	**Elements**	**Mass**	**+/-**	**Rarity**
Fe (A)	57.3220%	-	7.7833%	Fe (A)	84.6112%	-	22.7317%
Cu (A)	13.7189%	+	20.3766%	Al (A)	8.2207%	+	45.8358%
Sb (A)	13.4356%	+	27.9455%	Cu (A)	4.4800%	+	13.1659%
Al (A)	9.6731%	+	27.2585%	Cr (A)	1.0243%	-	0.3534%
Mg (A)	3.7144%	-	2.3077%	Ni (A)	0.6353%	+	4.0621%
Zn (A)	0.9374%	-	0.7870%	Zn (A)	0.4907%	+	0.8151%
Sn (A)	0.7272%	+	1.4042%	Mg (A)	0.2190%	+	0.2692%
Cr (A)	0.2161%	-	0.0377%	Sn (A)	0.1341%	+	0.5124%
Ni (A)	0.2055%	+	0.6641%	W (B)	0.0993%	+	6.7211%
Co (B)	0.0121%	+	0.5691%	Sb (B)	0.0672%	+	0.2765%
Ag (B)	0.0119%	+	0.4548%	Co (B)	0.0061%	+	0.5662%
W (B)	0.0071%	+	0.2440%	Nd (B)	0.0040%	+	0.0224%
Bi (B)	0.0061%	+	0.0143%	Mo (B)	0.0031%	+	0.0274%
In (B)	0.0041%	+	6.3980%	Pd (B)	0.0019%	-	0.0007%
Mo (B)	0.0034%	+	0.0152%	Ag (B)	0.0017%	+	0.1266%
Nd (B)	0.0028%	+	0.0080%	Dy (B)	0.0005%	+	0.0031%
Au (B)	0.0013%	+	3.7150%	Au (B)	0.0005%	+	2.5143%
Pd (B)	0.0004%	-	0.0001%	In (B)	0.0003%	+	0.8799%
Dy (B)	0.0003%	+	0.0009%	Bi (B)	0.0001%	+	0.0003%
Li (B)	0.0001%	+	0.0003%	Li (B)	0.0001%	+	0.0004%
Tb (B)	0.0000%	+	0.0001%	Pt (B)	0.0000%	+	1.1154%
Pt (B)	0.0000%	+	0.0155%	Tb (B)	0.0000%	+	0.0002%

Such distinctions between mass and rarity become more extreme when examining several materials whose mass share is below 0.01 wt% of the total of that product category stream (both for *E*-R3 and *E*-R4). For instance, while Cobalt accounted for 0.0.12% of the mass its rarity was 0.56%. At the same time, Indium accounted for 0.004% of the mass but 6.398% of the rarity. Both Cobalt and Indium are, among other materials, crucial to the energy transition due to their use in batteries (Cobalt) and PV technologies (Indium) (Bobba et al., 2020). Overall, the results indicate that the relative rarity of certain materials within a waste stream, as expressed in exergy, shows large differences compared to their mass in the products. This mass and rarity discrepancy is similarly observed in the *E*-R4 results. The major bulk metals Iron, Copper and Aluminum account for 88.61%, 4.45% and 8.22% of the mass present in this stream, respectively, compared to 22.73%, 13.17% and 45.84% of the rarity. This result is relevant since all bulk metals are recovered in the primary recycling process, e.g. Copper, Aluminum and Iron.

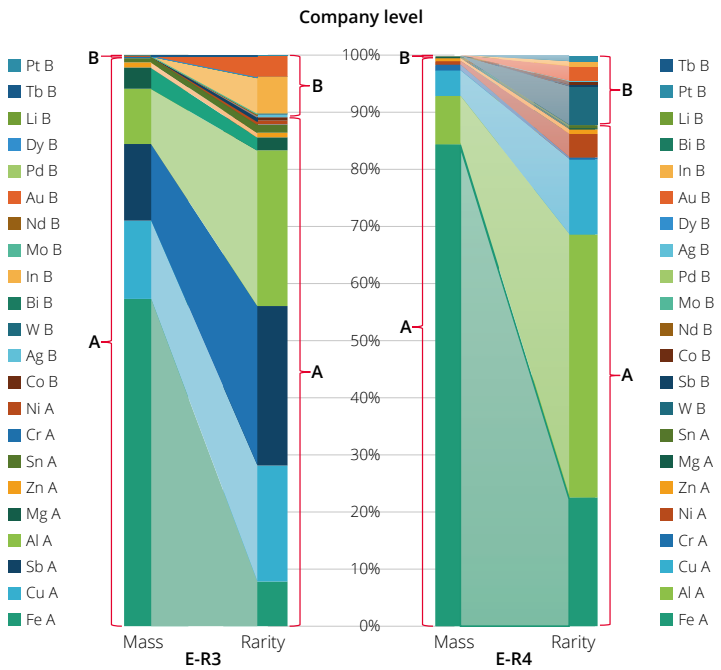

FIGURE 6.2 Company level: A: Mass and rarity for product categories *E*-R3 and *E*-R4 for metals with a mass share above 0.1 wt%. B: Mass and rarity for product categories *E*-R3 and *E*-R4 for metals with a mass share below 0.1 wt%. Full names of the elements are provided in Table 6.2.

Examining the composition of WEEE from these two levels of the case study reveals several insights. Namely, Aluminum, Copper and Iron comprise the majority of the weight. The use of the TR indicates some large discrepancies, particularly between those materials below 0.1% of the mass. These account for 21% and 4.56% (national) and 11% and 12% (company) for *E*-R3 and *E*-R4 respectively. The discrepancy between the mass and the rarity scores can be explained through the specificity of the case study sampling approach, which is not captured in aggregated reporting. Yet, the use of the indicator points to the increased significance of the minor materials in both levels of analysis, e.g. Indium, Tungsten, Cobalt and Lithium.

6.3.2 Critical policy reflection

Currently, the EU WEEE Directive and its 2012 Recast (EP and Council 2012, 2002) recycling targets that affect the product categories studied include 70% and 55% recycling/reuse rate (by weight) for screens and monitors (*E*-R3), and small equipment and IT and telecommunications (*E*-R4), respectively. Applying these targets to the materials included in the analysis, we observe the mass or rarity approach's effect has. The objective of the following reflection is to observe how the specific actors in WEEE recycling respond to - and fulfil - such targets, given the indicated policy context, and how, looking beyond this Italian case, a rarity perspective could be integrated within decision making for EU EoL policies.

Looking at the national level, the *E*-R3 70% recycling target could be met through recovering Iron. However, meeting the targets through a TR perspective would require Aluminum and a combination of another elements. For *E*-R4, meeting the 55% target via mass could be done also by only recovering Iron. However, a target based on TR would require recovering Aluminum, with a combination of other element, e.g. Tungsten or Nickel. Applying the indicator at the national level indicates the weightings between materials, and (a) signifies the significance minor materials have and (b) the varying combinations of specific materials that would need to be recovered should the targets considerate TR over mass (Table 6.5).

TABLE 6.5 Meeting the EU recycling targets through either a mass or rarity approach (based on the national case study).

Product category and target	Mass %	Rarity %
E-R3 Recycling 70%	Iron (70.2)	Aluminium (44.1), Gold (10.7), Copper (10.9), Iron (10.3), Indium (9.7) and Antimony (9.3)
E-R4 Recycling 55%	Iron (73.3)	Aluminium (54.3), Tungsten (6,2), Nickel (3.1).

Applying the targets to the company level vary against the outcomes of the national. Starting with *E*-R3, the target of 70% recycling, based on the mass of materials, could be met by focusing on Copper and Iron. Yet, from a rarity perspective, meeting the targets would require focusing on a combination of Aluminum, Copper and Antimony. Alternatively, for *E*-R4, a target of 55% recycling can be met from a mass perspective by simply collecting Iron. Alternatively, from a rarity perspective would necessitate including Aluminum, Nickel and CRM Tungsten (Table 6.6). These results are comparable to Ortego et al. (2018b) for how the EPR policies effect what is recycled in vehicles and the difference a TR approach makes.

TABLE 6.6 Meeting the EU recycling targets through either a mass or rarity approach (based on the company case study).

Product category and target	Mass %	Rarity %
E-R3 Recycling 70%	Copper (13.7), Iron (57.3)	Aluminum (27.2), Copper (20.3), Antimony (27.9)
E-R4 Recycling 55%	Iron (84.6)	Aluminium (45.8), Nickel (4), Tungsten (6.7)

These suggestions do not consider the specific technical requirements to correctly recycle each element, which is likely to be demanding for small, highly concentrated materials. It also does not include questions of economic feasibility and market conditions, which often limit the introduction of novel recycling practices. Instead, it sheds light on the value (expressed in exergy) each of these elements has against their relative abundance in the Earth's crust and, crucially, it suggests issues of current and future accessibility for these particular materials. This is essential given the increasing demands for these materials and current supply risks for many (Bobba et al., 2020).

The results provide an opportunity to reflect on the broader policy regime for the CE of electrical products in the EU. We reflect on the TR results at the national and company level, on *how* the current policy regime affects certain actors' decisions, what policy and legal structures *could* be adjusted to accommodate this reality and the underlying pressures for the CE, e.g. long-term resource depletion. Applying the TR indicator is not done to argue that national targets must be based on TR. Instead, it is used to illustrate the increased importance and significance (from a long-term perspective) that different materials have, particularly many CRM (some of which are lost). The company case illustrates the challenge of accurate WEEE sampling and how the policy conditions can result in particular decision-making outcomes for the subsequent utilization and recovery of materials, a value retention system that prioritizes mass at the expense of rarity and scarcity (Campbell-Johnston et al., 2020b).

The first point of contention is the nature of the EPR targets. Whilst there is a clear logic (both policy and market-wise) in focusing on mass-based targets, the (unintended) consequence, as illustrated in this study, is the loss of materials that are either critical or geologically scarce, e.g. 38% TR loss for *E*-R3 and 10% *E*-R4 (see Section 3.1). This is a more fundamental issue. CE policies should intend to retain materials that are likely to be exhausted or otherwise inaccessible in the not so distant future (cf. Henckens et al. 2016a). Following the implications of our results, and in line with other authors (cf. Horta Arduin et al. 2020), we propose the need for EPR policies to recognize the issue of critically and geological scarcity, either in the monitoring or target setting. For this, TR could be a useful indicator to support the decision-making process for target or recycling standard-setting, as it indicates the hidden (exergy) value of different materials.

In addition, two critical issues that emerged during this research were: (a) obtaining data on the composition of products in a specific waste stream and (b) understanding the quantities of critical and scarce materials within those products. The first issue highlights the challenge of understanding the actual complexity of WEEE streams arriving at processing facilities, which has implications both on their ability to process them, and the efficacy of EU policies aimed at reducing those streams. For the second issue, at present, current Eco-labelling requirements and product category declarations do not require acknowledging the presence and quantities of such materials, further obscuring the knowledge on what materials, including CRM, are moving through the market and the processing facilities. This could be supported by expanding the reporting requirements included in the Waste Framework Directive (2018) for the European Chemicals Agency for producers to report CRM quantities. This is currently done for specific products, e.g. computer servers, but not all. Policymakers should use this information for long-term target setting and innovation and R&D policy for CRM recovery. Producers are already required to register several hazardous substances, e.g. lead, cadmium and mercury, which includes several CRMs, e.g. Cobalt and Antimony under the REACH Directive. The requirements should also extend to all CRM. For eco-design, increasing the accessibility of information on the presence of such materials (for policymakers and recyclers), or, working towards substitutions where either recovery or increased accessibility is recommended.

6.3.3 Limitations

This approach provides novel insights into the quantities of materials with the WEEE streams. Although the results of this study allow us to observe the main discrepancies, limits of policy and usefulness of the indicator, we consider methodological and data gaps and uncertainties. First is the issue of WEEE sampling and characterisation. It is widely known that the reporting of WEEE data in the EU is characterised by many discrepancies and inconsistencies (Forti et al., 2018). Multiple variables effect WEEE over time, from the product types to the material composition. The reporting at the company involved batch tests of products, which was then scaled up to the quantities of materials entering the facilities allowing for an estimate at to whether the recycling targets were met. We opted for our sampling strategy, not for representativeness (a general issue with WEEE reporting), but to illustrate the types of products and the application of the indicator at the company level as no other data was available to us. Estimates of the elemental composition of products in the samples and those at the national level were made using the best available data (Huisman et al., 2017). The micro results could have been strengthened by conducting lab tests on the products sampled to more accurately document their elemental composition. However, such a means was not available. Applying the indicator at the national and company level allowed us to illustrate the discrepancies between data at these different levels, and how the TR indicator affected them. Nevertheless, the TR assessment was calculated having these limits in mind. Strengthening the reporting requirements and composition of products and WEEE streams coupled with greater stakeholder collaboration between producers, policymakers and recyclers is necessary to improve the potential for CRM recovery.

Another limitation relates to the scope in which the TR indicator was applied. As outlined in the methodology, this research did not consider the technical processes needed to recover specific CRM and how this relates to TR values. This work is being explored by Valero and Valero (2020), but was not possible within this study. Improving the methodology and extending its application would require an adaptation of exergy databases proposed by Ortego et al. (2018b) and information on energy and resources consumed during the treatment of and the recovery of CRM, dividing the contribution directly linked to materials and to extraction and treatment processes. Currently, information such as exergy efficiencies of the processes or technologies used is not usually available such as the exergy cost of the mechanical or manual dismantling process. As such, the proposed indicator was applied to the main CRM included in the WEEE sample. The required calculations were performed directly by the authors using exergy data as a black box.

Finally, we used the indicator to draw policy implications based on the rarity implications. We do not seek to generalize the results to the EU, instead to use the indicator to reflect on the outcomes of the policy context, and, based on the results how this context could be modified. However, in setting CRM targets and conditions, broader issues must be considered, namely, the socio-economic considerations that TR does not consider. Similarly, market and technical

dynamics for EoL materials, which greatly influence the recovery process, are not a part of the rarity assessment and should be studied separately. The commercial extraction of specific rare elements and materials is more promising for some due to the combination of favorable market conditions, collection practices and technological advancements (see CEWASTE 2021).

6.4 Conclusion and recommendations

In conclusion, growing demand for specific materials will cause increased depletion and scarcity risks. Consequently, understanding which materials are most important from a long-term perspective, particularly those at the EoL as higher recycling is needed to offset future demand bottlenecks for specific materials. This research used the TR indicator applied in a novel context to explore the differences between the mass and rarity of materials within WEEE streams. Using the case of Italy, it examines two product categories comparing their mass and rarity at the national level, and further explored at the company level to illustrate the complexity and discrepancies of the policy context and the application of the indicator. This study takes a unique interdisciplinary approach that connects a technical exergy approach with a reflexive policy analysis.

The results of company analysis indicate that the recycling process prioritizes the recovery of mass and bulk metals at the expense of minor and critical ones. A general observation is that the primary recycling process recovered three mass metals (Aluminum, Iron and Copper). Based on estimates of potentially subsequent recycling operation, the results infer that around 13 materials are lost in this process. Of these materials that are lost, three are categorized as CRM by the EU (Table 6.2). Using this indicator revealed a large difference between the mass and rarity of specific materials. For example, for the product category R3, Iron accounted for 57% of the mass compared to 7% of the rarity. Whilst Indium accounted for 0.004% of the mass but 6.398% of the rarity. This difference becomes significant when the issue of rarity is applied to the current EU EPR WEEE recycling targets. Using the indicator would result different recovery options noticeably several minor metals.

These results provide a point of reflection on the limitations of EPR within the EU policy regime. In our sample, we estimate up to 38% of the TR of one product category is lost. The logic of CE argues for closing material and energy loops to maintain the value of products and materials. Here we illustrate the need for a value retention orientation that accurately includes the issue of rarity and criticality, from which, we argue, TR is a complementary indicator to help the decision-making process for both target and standards-setting. Moreover, based on the challenges of this research, there is a need for greater knowledge of the composition of products, including their CRM, to be known by policymakers in order to help foster longer-term recovery and innovation options. Electronic products and electronic waste are projected to increase substantially in the coming decades. Thus, prospectively grappling with this question of criticality within the policy domain is essential.

Future research on TR should address two areas. First, this research has resulted in insights gained from applying the indicator in a novel case study context. Expanding the scope of this study to more companies and the EU more broadly, would result in more generalizable results. Second, research should compare the TR of elements with the exergy needed to recover it. This is proposed by Valero and Valero (2020), but not yet empirically established. This would verify the practical applicability of the indicator. We further recommend that the socio-environmental trade-offs between CRM recovery and their associated environmental impacts (or benefits) are investigated in further research. This could be done by exploring how (thermodynamic) rarity could be integrated or contrasted with other assessment methods to inform *which* materials are prioritized for recovery, e.g. developing a multi-criteria decision analysis framework for end-of-life CRM policy.

7

Reflections on the effectiveness of extended producer responsibility to contribute to the implementation of circular economy in the European Union

In the introduction, a thought exercise was given, asking the reader to consider the complexity and intricacies of the production and consumption chain. The purchase of a product is intertwined with a whole set of processes before and after this act. These can be immeasurably complex in their technical and organisational nature spanning multiple product components to a global set of value chains. This thesis is titled "*coming full circle*", where we try and connect and ground that complexity within specific organisational and policy processes, and where those processes are not separated but considered together. Where can and should the system be changed? This research focused on the EoL stage as the domain through which to consider this systematic complexity. This chapter itself comes full circle, by reflecting on this these key research and methodological insights. It looks back on the combined body of work outlined in the first chapter and presented in Chapters 2 to 6.

As outlined in Chapter 1, current production and consumption practices have driven unprecedented (and uneven) demand for materials since the Second World War (Jackson, 2009). These practices intersect and cause phenomena such as climate change, biodiversity loss, excessive waste generation and social inequalities. On the biophysical level, researchers have estimated that five of the Earth-system processes (Climate change, Nitrogen cycle, Novel entities, Land system change and Biodiversity loss boundaries) have been crossed, indicating the magnitude and also urgency for suitable responses (Persson et al., 2022; Rockström et al., 2009). Against this backdrop, and the so-called "linear" nature of current production, consumption and waste patterns, the CE has grown in popularity over the past decade framed as a solution (Ellen MacArthur Foundation, 2013; Reike et al., 2018).

As this thesis has argued, CE should not be viewed as a new phenomenon but instead a refurbishing of older concepts, which all try to mitigate human-induced material effects on the environment. Researchers have placed the current discourse of CE within a broader historical timeline (cf. Blomsma and Brennan, 2017; Calisto Friant et al., 2020; Reike et al., 2018), emphasising its evolving and evolutionary features. This thesis has followed the classification of Reike et al. (2018), which denoted three key phases: CE 1.0, CE 2.0 and CE 3.0, with the earlier incarnations resulting in policies and practices relating to product and waste management. Yet, the current discourse of CE contends the need for more transformative interventions that deal with impending societal threats of depletion, waste generation and social injustices.

Recognising this evolutionary feature of CE, this thesis examined EPR, one of the key EoL CE tools, to gauge its effectiveness in contributing to the current discursive demands of CE (3.0). Scholars have examined various aspects of CE related policies, to the broad limitations of the policy goals (Calisto Friant et al., 2021), policy mixes needed to support a CE transition (Milios, 2018), and the complexities of adjusting existing regulatory regimes for the circularity of products or substances (Backes, 2017; Deutz et al., 2017). Yet,

there is a knowledge gap regarding existing experiences and outcomes of established CE policies from a governance perspective.

This thesis presents an extensive examination and reflection on the application of EPR (as an example of an existing CE policy) within the EU to understand its effectiveness in contributing to the implementation of CE. Drawing on two bodies of literature (industrial ecology and environmental governance), this thesis analysed the application of EPR on two levels: i) by examining the instrument on its own merit and ii) from a critical perspective, drawing on the discursive claims embedded within CE and the framework by Reike et al. (2018). This research contributes to the emerging discussion on CE in two distinct ways. First, it contributes theoretically by conceptualising the complexities of socio-governance decision-making processes through outlining the cascading principle (see Chapter 3). Second, it contributes empirically through detailing specific CE EPR-related practices within the EU (Chapters 4-6). The specific research questions, answers and broader reflections are now discussed.

7.1 Reflections on the outcomes of each chapter

Chapter 2 provided a general synthesis of the literature on EPR and outlined a historical and global understanding of it. This chapter shows EPR is a policy approach or specific instrument encompassing a broad array of requirements (informative, physical, economic etc.) placed on the original product producer. These requirements predominantly take the form of physical take-back requirements at the EoL phase, e.g. post-user phase. This chapter outlined some general insights into the instrument's effectiveness over the previous 30 years and discussed its role in fostering a CE.

Chapter 3 reviewed the key literature on CE, cascading and up/downcycling and proposed a framework that considers the social and governance implications for the subsequent application and allocation of products and materials during their lifecycle. This chapter integrates the two disciplinary lenses used in the thesis by proposing a framework that enables a dual perspective for understanding product and material flows: the physical flows and the socio-governance context that facilitate decision-making processes of material outcomes. This conceptual thinking provided the reflexive and explorative rationale for considering EPR systems as the decision-making mechanism for post-user waste products.

Chapter 4 examined the governance and performance of EPR for tyres in the Netherlands. It described the key actors, policy developments and responsibilities in this specific system. This EPR system is based on single PRO with multiple collection and processing services. This chapter mapped the performance of EPR in the Netherlands, but also explored alternative material outcome strategies and associated limitations. It argued that despite high levels of material recovery (Recycling and energy Recovery), there are limited circularity outcomes.

Chapter 5 examined the perspectives of EPR and CE experts to explore how EPR could be transformed to aid the CE goals of the Netherlands (which had a target of a 50% reduction in primary raw materials by 2030). This was conducted with four sectors (electronics, cars, floor covering and flat glass) and invited 50 CE and EPR experts. A Delphi study aims to synthesize and refine expert knowledge on a subject while also observing their differences. This chapter outlined several areas for improvement for EPR to further contribute to CE. It further illustrates that whilst there was a level of agreement between actors for the need for a CE, the willingness of who should accept new responsibilities within EPR was heavily contested.

Chapter 6 examined the presence, quantities and losses of critical raw materials within WEEE, in the context of EU EPR policy. Using an indicator called thermodynamic rarity, which shows the exergy needed for mining and refining specific materials, this chapter indicates the differences between collection and recycling policies based on mass instead of rarity (which emphasises the conservation of materials based on their geological availability). This indicator illustrates the weakness within the EU's EPR targets for WEEE. These are based on prompting mass collection and recycling at the expense of other quality-related outcomes, e.g. rarity. This chapter outlines several potential recommendations for improving EPR for critical raw materials preservation.

Based on the empirical insights gathered in the research, we can now reflect directly on the sub-research questions.

1. How has EPR been implemented and organised within EU member states?

As outlined in Chapter 2, EPR has generally been organised in four distinct modes: 1) one single PRO with commercial and/or multiple collection and processing services; 2) multiple PROs with a clearing house system; 3) government tradable credits structure and 4) government-run EPR system. Systems 1,2 and 4 are generally used within the EU for various product categories, e.g. cars, tyres, electronics, batteries. As shown in Chapter 5, the general requirements (at the EU level) for EPR are outlined in the Waste Framework Directives and accompanied by specific EPR product-related laws, e.g. the 2012 WEEE Directive. A key feature of this organization is the requirement for producers to organize the take-back of EoL products (either physically or financially) either individually or collectively. Building on this, Chapter 4 provides a more in-depth review of one specific form of the organization for tyre recycling in the Netherlands. It described the particular history and division of responsibilities between key actors: market and government. This system is based on a single PRO model, where authority for executing the collection of recovery requirements is delegated from the PRO to a third-party actor. The insights from these desk and empirical studies indicate that the implementation of EPR in the EU has resulted in a varying array of organisations where different levels of responsibility and authority are divided between the government and market, in what we understand as a form of public-private governance, i.e. where cooperation is mainly between market actors and government.

2. How do EPR strategies vary in respect to their effectiveness, limitations and outcomes?

EPR systems have pushed collection and recovery targets as their key measurable outcome (Chapter 2). The levels of collection and recycling vary depending on the specific product category and country in question (see Table 2.2). Chapter 2 further outlined a general summary of the effectiveness and limitations of EPR systems. These include 1) organising recycling, resulting in directing products away from landfill (strength) but leading to unidentified exports (weakness); 2) efficiency, which results in low operating costs for actors involved (strength), but can also lead to issues of free-riding and low-quality outcomes (weaknesses); and finally, 3) stimulating eco-design, for which there is limited evidence of EPR having affected (see Table 2.1).

EPR systems *are* effective on their original design purpose, e.g. promoting high levels of collection and recycling (Chapter 4). Yet, when applying a CE framework and analysing the potential value retention options available (R-strategies), EPR systems result in limited CE outcomes. More specifically, they can result in problematic recycling outcomes (Chapter 6). Chapter 5 further details several broader issues related to EPR. These are the lack of innovation, limited targets and goals, actor inclusiveness and reporting and transparency requirements. Based on this, we can understand that EPR has been effective in many of the original functions in which it was designed. However, several operational issues remain related to the material outcomes and inability to affect upstream producer practices as the EoL stage and actors are not integrated. These limitations, in the context of CE, are more acute.

3. What (EPR) mechanisms (and value decisions) are used to identify the trajectory and use of products and materials post-collection?

As outlined in s-RQ 2, EPR systems generally push and result in the collection and recovery of materials (R7 and R8). S-RQ 3 bring together the various mechanisms, i.e. the means through which the material outcomes of products collected within EPR are directed. The general EPR 'mechanism' usually takes the form of take-back requirements, which, as specified in Chapters 4 – 6, broadly manifest as collection and recovery targets (usually based on mass). The conceptual thinking for comprehending EPR mechanisms was to understand not just the material and physical outcome, i.e. *how* the material was used, but to consider the decision-making contexts that facilitate this particular outcome (or set of outcomes). This necessitated looking closer at what governance structures were in place and the effects they had. Within the targets, the mechanism can take the form of a threshold value, where a specified treatment option, e.g. Recycling (R7) or incineration with energy recovery (R9), is given a specific monetrary value (Chapter 4). If the said threshold value of a higher treatment option is projected to be exceeded, then a lower treatment level can be pursued, e.g. energy recovery over recycling. This mechanism emphasizes cost and efficiency over the quality and higher circular outcomes. A mechanism, such as this, in which the value system emphasises cost and efficiency can also result in distinctly problematic outcomes

(from a CE perspective). As observed from Chapter 6, the mass-based targets promoting the collection and recycling of electronic waste are based on quantitative targets, which omits material (and product) quality as a critical issue for EoL practices in a CE. The combination of EPR targets and various markets mechanisms in place results material outcomes that promote efficiency at the expense of other quality of sustainability outcomes, particularly critical materials.

4. *How can current approaches to EPR be further developed and strengthened to contribute to CE?*

The s-RQs above outlined the prevailing issues within EPR approaches within the EU using specific case studies. Specific operational improvements to the organisation and outcomes of ERP strategies are detailed in the specific chapters. Chapter 5 presents the synthesised views of various EPR and CE practitioners and arrives at a set of recommendations for improvement. Nevertheless, some general insights from all the research can be outlined. These relate to the need to develop the material outcomes of EPR systems based on a quality system, e.g. issues with the recycling mechanisms (Chapter 6) and actors involved, e.g. issues pertaining to the inclusion of other R-actors (Chapter 5). The varying insights generated from this thesis on *how* to specifically develop and strengthen EPR as a policy to contribute to CE are further detailed in additional policy briefs (see Appendix).

Reflections on the effectiveness of EPR (as a form of public-private governance) in contributing to CE

In bringing this together, we can reflect on how these results relate to the main research question: **to what extent can extended producer responsibility (as a form of public-private governance) act as an effective mechanism in contributing to the implementation of CE in the European Union?**

As outlined in the introduction, this question was considered on two levels: on its own merit and from a CE perspective.

To speak to the first part. When considering the instrument's effectiveness, we must first understand the broad rationale for which it was introduced and then what the results from this study reveal. The broad rationale for introducing EPR was dealing with the EoL stage, shifting the responsibility of waste away from municipalities, whilst aiming to promote cleaner production practices and product design changes. Broadly we observe varying levels of organisational effectiveness. Chapter 2 provides a snapshot of EPR results, while Chapter 4 deep-dived into one case study. EPR as a policy approach has contributed to shifting the burden for the EoL stage away from municipalities (in selected product categories) while promoting higher R strategies (recycling R7 and energy recovery R8). PROs (or their representatives) who assume the financial and/or organizational responsibility for the EoL products run efficient and lean organisations. In this sense, the experiences and scope of

EPR as explored in this research provides some, yet limited, support for the implementation of CE in the EU. They are limited because the scope of EPR is still confined to the post-user recycling phase, with little to no evidence of EPR affecting higher R-strategies and product design changes.

However, as outlined, the second level of analysis considered understanding and evaluating the instrument from the perspective of CE. This explorative and evaluative combination of reveals several weaknesses regarding the governance of EPR. These reflections are complemented by broader policy and theoretical insights from the literature.

First, these relate to the decision-making processes within, and the organisation for, the outcomes of the EoL products, e.g. what happens and why. As indicated in Chapter 4, these mechanisms can result in low-quality materials outputs or, as shown in Chapter 6, the loss of geologically scarce and critical raw materials. This organisational element often relates to the standards for recycling and the price mechanisms built to facilitate it that emphasizes efficiency over 'best' application, e.g. offsetting virgin material inputs, or highest social value (see Chapter 3). Additional issues with the organisation of EPR systems relates to the transparency, monitoring and final destinations of products (the documentation for these often being unknown). In taking the broader perspective of CE, one that emphasizes the entire production and consumption value chain, the (theoretical) link between EPR and the contemporary understanding of CE is clear and compatible, e.g. responsibility for the entire lifecycle matches and respective R strategies. Yet, we observe that i) the scope of EPR is constrained to the higher and lower quality Rs, meaning the stress on repair and altering consumer habits, e.g. reducing consumption, is missing, and ii) the connection of EPR to product design is (still) lacking, despite this being a continued rationale for its use at the EU level. Thus, from a design and organisational perspective, we can infer the limitations of using EPR to contribute to the implementation of CE in the EU *unless* further changes are taken to improve its effectiveness. Proposals connected to this research that have provided solutions to these issues are outlined in the policy briefs listed in the Appendix. These include proposals to increase the stringency of requirements on actors in the EPR system, include a broader array of actors and further integrate EPR with Eco-design policy.

Beyond these operational insights and conclusions, it is important to reflect on the theoretical aspects derived from this research. This is done in two parts.

Reflections on EPR as a form of pubic private governance

This research considered and conceptualised EPR as a form of public-private governance. This form of actor and policy arrangements emerged in the 1990s, where the market and market actors became a mechanism to solve societal problems, e.g. dealing with pollution and following the polluter pays principle. As is shown above,

varying responsibilities (financial and/or physical) connected to the post-user phase of a product's lifecyle were transferred to market actors. In reflecting on the *effectiveness* of this type of organisational relations (as found in this study), it is hard to definitively state if this form of governance, compared to other organisational arrangements, e.g. top down, forms of governance, is the most effective or not. Instead, *what can* be observed is the limited effectiveness, relating to the operational issues outlined earlier, such as the conditions in which market actors operate, i.e. the limited scope and function of EPR. This relates not per se to the effectiveness of the policy approach but more to the need to redefine the ambitions and function of EPR in the context of CE. As argued in Chapter 3, in terms of understanding *effective* outcomes, these must be connected to a strong integrated sustainability agenda. This requires a more active and dynamic role of the state as a pivotal actor in shaping and monitoring these processes. This claim echoes those examining CE policy approaches more broadly. Next, we reflect on the notion of effectiveness of EPR to contribute to CE in the context of the industrial ecology literature.

Reflections on EPR as a means to contribute to the material claims of CE

This research was inspired by the work of industrial ecology, which relates to analysing the flows of energy and materials through the economy. This thesis has approached the contemporary phenomenon of CE from an evolutionary historical perspective, taking a broadly interdisciplinary lens. It argued that CE could be broadly classified into three historical phases: CE 1.0, CE 2.0 and CE 3.0. These phases resulted in specific regulatory responses to societal problems related to consumption and waste. Concerning the EoL phase, this broadly resulted in regulatory measures to prevent pollution from waste CE 1.0 (1970s) and measures to promote the increased circulation of secondary raw materials in CE 2.0 (1990s), when EPR was introduced. In this research, we argue that these regulatory and organisational conditions that emerged around EPR must adjust to the new material demands of CE, i.e. dealing with waste and resource scarcity.

Yet, on the empirical basis, there is still this notion that emerged from an earlier phases, e.g. providing secondary raw materials, or promoting recycling as a goal in and of itself. From a policy perspective, the EoL stage and EPR are stuck in this paradigm. See, for example, the rationale of the Dutch 2014 "From Waste to Resources plan" and the EU's "Waste Framework Directive" (Ministerie van Infrastructuur en Milieu, 2014; WFD 2018/851, 2018a). However, instead of promoting recycling as the main solution to CE, we argue, concerning the material basis and outcomes of EoL and EPR policy, the function, i.e. the purpose, needs to be redefined and reconfigured. As shown in Chapter 6, the governance and decision-making within EPR systems can result in a loss of geologically scarce materials. As scarcity represents a key aspect underpinning the discursive drive for CE, this finding reveals an organisational limitation within the design of EPR and EoL systems. The connection between the EoL stage and the whole of the products lifecycle needs to be strengthened from this regard. In essence, policies for the EoL stage needs

to evolve from simply preventing pollution and providing secondary raw materials through material recycling (as R7) (CE 1.0 and CE 2.0) towards documenting product design flaws and promoting the recovery of materials most likely to be exhausted, e.g. geologically scarce materials, in effect, becoming a diagnostic tool to inform upstream product design changes. This also necessitates behaviour change related to the technical activities of product design and recycling.

The thesis title is "*Coming full circle*" and represents two key elements from this research. First is the notion of interconnectivity of a product, its material and components to each aspect of its lifecycle and the need to close the loop throughout. Second the notion of reviving the original intentions of EPR to create and inform product design changes. "*Coming full circle*" implies coming back to this original intention more strongly by designing out waste and connecting the EoL stage to the product design stage (see Figure 7.1).

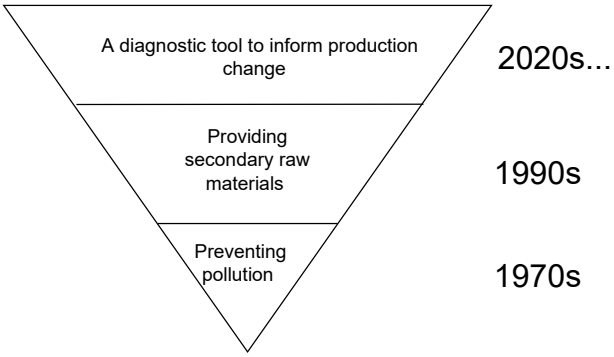

FIGURE 7.1 The evolution of the rationale for EoL policy, including EPR and the proposal for a new approach within a CE.

This research has, in effect, illustrated the limited nature of the organisation of EPR in the EU and called more a more proactive and balancing role of the state and governments in re-shaping the conditions of the market. In this sense, following the call of Ayres et al. (2001) of stepping out of the conventional market framework to develop *stronger* sustainability outcomes. Research to support such proposals is needed. However, whilst a disciplinary approach, e.g. industrial ecology or economics, can help reveal and understand specific issues at specific scales, developing appropriate responses requires a more sustained and integrated set of knowledge across the disciplines. This research project took a broadly interdisciplinary approach and was inspired by two disciplines. However, it was also embedded within a larger EU interdisciplinary project called Cresting that varuous topics from business models, policy approaches and assessment methods.

Such broader projects are needed to provide broader answers to the interrelated set of challenges facing the world.

7.2 Methodological reflections

This research used a variety of methods, that are detailed in Chapters 2-6. This section reflects more broadly on the two approaches that informed the research approach: grounded theory (GT) and transdisciplinarity (TD). Critically reflecting on the process and decisions makes the research more transparent and rigorous (Palaganas et al., 2017).

GT, as a broad approach to theory building, was followed in this research due to the inductive and reflexive nature needed in this method. GT emphasises field experience and an iterative process between data collection and theory instead of deductive theory testing (Strauss and Corbin, 1990). This research followed such a non-linear approach. It used insights from three different case studies, exploring the research to different ontological and contextual environments. While the latter was not used in the research, the interviews and experiences examining the case study very much informed insights in other cases and contexts[19]. This process was challenging to navigate in terms of the diverse contexts and approaches taken and hugely enriching and rewarding to gain such a diverse set of insights and perspectives. In particular, these insights formed intricate connections between specific projects. For example, the focus on the loss of critical raw material value (Chapter 6) arose from company field visits and a critical review (Chapter 3) and stakeholder conversations in France and the Netherlands.

However, in a practical sense, the empirical data collection was completely disrupted by the onset of the Covid-19 pandemic from 2020. This meant that almost all in-person field visits and in-person interviews were impossible after March 2020. Consequently, much of the research was forced to change from field to desk research. Chapters 2, 5 and 6 were designed and written during this period. Nevertheless, these projects hugely benefited from the insights and experiences of earlier case study work and the interdisciplinary international project (Cresting) in which it was embedded that informed much of the practical and theoretical reasoning.

The second aspect that informed this broad inductive approach was TD. Because sustainability is related to many ongoing and wicked societal developments, there is a need for continuously developing multi-disciplinary theories. More collaborative forms of scientific engagement and knowledge production with society are becoming the norm. This type of science-society engagement is particularly important for sustainability science and sustainability research. As outlined in Chapter 1, this research did not follow a strict TD approach but built on the notion of useful knowledge

19 The empirical insights of France are being developed into a policy brief.

production. This general approach, coupled with the interactive approach of GT, meant the research was shaped in collaboration with and by its participants. Certain issues and discussions with stakeholders informed interest in a specific research direction or the use of a particular method. Yet, the broad TD approach allowed the research to contextualise these individual discussions within broader EPR and CE policy discussions. This work is ongoing beyond the completion of this thesis. This constructivist approach emphasizes the notion of multiple competing ontologies, with the role of the researcher to engage with, understand and synthesise these perspectives. This rationale underpins the research process and approach and should be seen as part of the context in which specific chapters were conceived and explored.

On the one hand, these types of stakeholder engagements have illustrated the challenge of conducting more participatory forms of scientific knowledge production. Question of what is useful knowledge and for whom were continuous questions throughout the research. On the other hand, this has emphasised the importance and critical role research can take in challenging concepts and activities as they emerge in practice. In that sense, this research could have benefitted from employing critical political economy theories (see Calisto Friant et al., 2020; Hobson, 2016; Newell, 2014) or theories on actors and power to complement the empirical cases on EPR.

A final consideration of the methodological approach concerns the choice of cases. The selection of cases was set due to the broader research project this research was embedded in. These cases are illustrative of the complexities and challenges of EPR for a CE in the EU. However, the research could have benefited through a broader array of cases within the EU. For example, in countries that more recently joined the union, or with a lower purchasing power, for example, Romania or Hungary. The comparative aspect of this research could have been further highlighted directly in this research.

7.3 Recommendations for future research

To further build theories on EPR in the context of CE, recommendations for future collaboration and the development of the insights in this thesis are outlined in four areas.

First is the nature of theory building on EPR in the context of CE. This research took a specific focus looking at the EU, driven by historical and contemporary policy developments. Nevertheless, to build more rich and diverse theories and insights that account for contextual and temporal insights, future research should focus on further comparative EPR studies. This should include systematic studies across the EU and beyond to global systems, particularly in the Global South. Collaboration between

different research and university institutions could help foster a rich understanding of it. Such research should consider the varying products and product categories that EPR schemes have affected beyond the ones studied in this thesis (OECD, 2016).

The second recommendation takes a more practical perspective. As outlined, the perennial claim of EPR is the connection and modification of product design through greater responsibility. In this area, several existing practical developments are currently unfolding related to the new responsibilities and fee modulations (cf. Laubinger et al., 2021; Vermeulen et al., 2021a). However, in extending the line of argumentation outlined in 7.1, future research could focus more extensively on integrating the EoL stage more concretely with eco-design and product design requirements. This should be done from an interdisciplinary approach combining technical, policy and legal researchers. This connection has been provisionally explored by Campbell-Johnston et al. (2022), but requires more extensive and robust empirical validation.

Third, future research should consider the combination of policy instruments, including EPR as one approach in a broader mix in fostering circular and sustainable outcomes. Research on the vein is already on-going (Milios, 2018), yet, the connection between policy instruments with behavioural and reducing consumption practices must be explored.

Finally, this research processes further theoretical development and detailing and empirical exploration of the cascading principle (Chapter 3). In particular, how this relates to the decision-making processes for the outcomes of materials and products through examining specific sectors, individual firms or governmental policymaking procedures, and how cascading approaches can be fully assessed (from an integrated sustainability perspective). The work on broader CE and sustainability approaches and assessment tool is taking off (see Lindgreen et al., 2020; Walker et al., 2021b), but the connection to cascading requires further detailing. Moreover, future theoretical work should consider research on power regarding how this informs decision-making processes and outcomes. This research could be empirically explored through ethnographic and document analysis within organisations connected to EPR, e.g. government and PROs.

As the challenges of the 21st century continue to rise, meeting them requires ambition and vision. The systems and organisations that worked for yesterday's problems will not be automatically solve the problems of tomorrow. Old systems need to be reformed and new ones developed. In the words of Albert Einstein "we cannot solve our problems with the same thinking we used when we created them".

Additional materials

Publications of Kieran Campbell-Johnston in this book

Vermeulen, W.J.V., Campbell-Johnston, K. Extended Producer Responsibility. Handbook of Recycling 2nd edition *forthcoming (2022)*

Campbell-Johnston, K., Vermeulen, W.J.V., Reike, D., Bullot, S., 2020b. The circular economy and cascading: towards a framework. Resour. Conserv. Recycl. X 100038. https://doi.org/10.1016/j.rcrx.2020.100038

Campbell-Johnston, K., Calisto Friant, M., Thapa, K., Lakerveld, D., Vermeulen, W.J.V., 2020a. How circular is your tyre: Experiences with extended producer responsibility from a circular economy perspective. J. Clean. Prod. 270, 122042. https://doi.org/10.1016/j.jclepro.2020.122042

Campbell-Johnston, K., de Munck, M., Vermeulen, W.J.V., Backes, C., 2021. Future perspectives on the role of extended producer responsibility within a circular economy: A Delphi study using the case of the Netherlands. Bus. Strateg. Environ. submitted, 1–20. https://doi.org/10.1002/bse.2856

Campbell-Johnston, K., Roos Lindreen, E., Mondello, G., Gulotta, T.M., Salomone, R., Vermeulen, W.J.V. Thermodynamic rarity of electrical and electronic waste: Assessment and policy implications for critical materials. *Submitted to the Journal of Industrial Ecology (under review)*.

Appendices

A: List of additional publications connected to this thesis

Campbell-Johnston, K., de Wall, I.M., Roos Lindgreen, E., Gulotta, T.M., Mondello, G., Salomone, R., Vermeulen, W.J.V., 2022. POLICY BRIEF on Critical Raw Materials and their integration in Extended Producer Responsibility and Eco-design Policy Key message. https://doi.org/10.5281/zenodo.6444189

Campbell-Johnston, K., Pruijsen, J., Vermeulen, W.J.V., Dermine-Brullot, S. Report on the Governance of Extended Producer Responsibility in the Transition to a Circular Economy.; 2022. doi: 10.5281/zenodo.6597508

Vermeulen, W., Campbell-Johnston, K., Thapa, K., 2022. Extended Producer Responsibility and Circular Economy: Three Design Flaws. Ökologisches Wirtschaften - Fachzeitschrift 37, 21–23. https://doi.org/10.14512/oew370121

Vermeulen, W.J.V., Backes, C.W., Munck, M.C.J. de, K.Campbell-Johnston, Waal, I.M. de, Carreon, J.R., Boeve, M.N., 2021a. WHITE PAPER on Pathways for Extended Producer Responsibility on the road to a Circular Economy. Utrecht. https://doi.org/10.13140/RG.2.2.11527.93602

Videos connected to this project:

On EPR and critical raw materials:

On organising EPR:

B:Overview of interviewees and workshops conducted in this research

The following tables outlines a summary of the various interviews, workshops and presentations conducted during the course of this research. Many interviews contributed directly or indirectly to the research, either in formal interviews for a specific research project, or in terms of general information gathering about extended producer responsibility, or, as a place for presenting results.

Interviews

#	Type of organisation	Country	Position (if known)	Date(s) of interview(s)	Chapter connected to	Additional material in Appendix A
1	Gv	BE	Policy officer for CE	April 2021	7	
2	Gv	BE	Policy officer for CE	April 2021	7	
3	Gv	FR	NA	May 2021	7	Report
4	Pr	FR	Consultant	May 2021	7	Report
5	PRO	FR	NA	March 2021	7	Report
6	Pr	FR	NA	March 2021	7	Report
7	Gv	FR	NA	March 2021	7	Report
8	Pr	FR	NA	March 2021	7	Report
9	Pr	FR	NA	March 2021	7	Report
10	Gv	FR	NA	March 2021	7	Report
11	Pr	FR	NA	June 2021	7	Report
12	Gv	FR	Expert on CE policy	January 2020	6,7	Policy brief
13	Gv	FR	WEEE monitoring agenda	January 2020	6,7	Policy brief
14	Pr	FR	Legal advisor on CE	January 2020	7	
15	Pr	FR	Advisor on CE	January 2020	7	
16	PRO	FR	Researcher on recycling options	January 2020	7	Report
17	Ts	FR	NA	January 2020	7	Report
18	PRO	FR	Researcher on recycling options	February 2020	7	Report
19	Pr	FR	NA	April 2021	7	Report
20	Pr	FR	NA	April 2021	7	Report
21	Ts	FR	NA	April 2021	7	Report
22	Ts	FR	NA	April 2021	7	Report
23	Ts	FR	NA	April 2021	7	Report
24	Pr	IT	Consultant	October 2019	7	
25	PRO	IT	Waste management coordinator	October 2019	7	
26	PRO	IT	Waste management coordinator	October 2019	6, 7	
27	PRO	IT	Director PRO for WEEE	October 2019	7	Report
28	Gv	IT	Policy officer for circular economy	October 2018	6, 7	

#	Type of organisation	Country	Position (if known)	Date(s) of interview(s)	Chapter connected to	Additional material in Appendix A
29	Pr	IT	Director WEEE recycling company	November 2019	7	Report
30	PRO	IT	NA	March 2021	7	Report
31	Pr	IT	NA	March 2021	7	Report
32	Pr	IT	NA	March 2021	7	Report
33	Un	IT	NA	March 2021	7	Report
34	Gv	IT	NA	March 2021	7	Report
35	Gv	IT	NA	March 2021	7	Report
36	Gv	IT	NA	March 2021	7	Report
37	Gv	IT	NA	March 2021	7	Report
38	Gv	IT	NA	March 2021	7	Report
39	Gv	IT	Researcher	January 2019	6, 7	
40	Pr	NL	Consultant	October 2018		
41	PRO	NL	NA	November 2021		
42	PRO	NL	NA	November 2021		
43	Pr	NL	Director of recycling facility	May 2020	7	Report
44	PRO	NL	Secretary	May 2019	4, 5, 7	
45	PRO	NL	CEO	May 2019		
46	PRO	NL	Advisor on CE	March 2022	7	Report
47	PRO	NL	Researcher on CE	March 2022	7	Report
48	Pr	NL	Lead consultant on CE	March 2021	7	
49	Pr	NL	NA	March 2020	7	Report
50	PRO	NL	NA	March 2020	7	Report
51	Pr	NL	Lead consultant	March 2020	5,7	White paper
52	Gv	NL	Policy officer for circular economy	March 2019	5, 7	
53	PRO	NL	CEO	March 2019	4	
54	Pr	NL	Sustainability officer	June 2019	5, 7	
55	Pr	NL	Engineer	June 2019	5, 7	
56	Pr	NL	Lead consultant on WEEE	January 2022	7	Policy brief
57	Gv	NL	Monitoring agent EPR	January 2020	5, 7	
58	Pr	NL	Director of company	January 2020	5, 7	
59	Un	NL	Researcher	January 2020	5, 7	White paper
60	Gv	NL	Researcher	January 2020	5, 7	White paper
61	Gv	NL	Researcher	January 2020	5, 7	White paper
62	Gv	NL	Monitoring agenda EPR and plastics	January 2020	5, 7	
63	Un	NL	Researcher	January 2020	5, 7	White paper
64	Un	NL	Assistant professor	January 2020	5, 7	White paper
65	Un	NL	Researcher	January 2020	5, 7	White paper

#	Type of organisation	Country	Position (if known)	Date(s) of interview(s)	Chapter connected to	Additional material in Appendix A
66	Un	NL	Researcher	January 2020	5, 7	White paper
67	Un	NL	Researcher	January 2020	5, 7	White paper
68	Un	NL	Researcher	January 2020	5, 7	White paper
69	Gv	NL	Policy officer for CE	January 2020	5, 7	White paper
70	Gv	NL	Monitoring agent EPR	January 2020	5, 7	White paper
71	Gv	NL	Monitoring agent EPR	January 2019	5, 7	
72	Gv	NL	Monitoring agenda EPR and plastics	January 2019	5, 7	
73	Gv	NL	Monitoring agent WEEE	January 2019	5, 7	
74	Gv	NL	Monitoring agent WEEE	January 2019	5, 7	
75	Gv	NL	Policy officer for textiles	January 2019	5, 7	
76	Gv	NL	Director of department	January 2019	4, 5, 7	
77	Gv	NL	Monitoring agent EPR	January 2019	4, 5, 7	
78	Pr	NL	CEO	January 2019	4, 5, 7	
79	Pr	NL	Engineer	January 2019	4, 5, 7	
80	Un	NL	Researcher	January 2020	5, 7	White paper
81	Gv	NL	Monitoring agenda for all waste flows in the NL	January 2019	4, 5, 7	
82	PRO	NL	CEO	January 2019	4	
83	PRO	NL	Advisor on CE	February 2021	7	Report
84	PRO	NL	Researcher on CE	February 2021	7	Report
85	Gv	NL	Policy officer for EPR	February 2020	5, 7	
86	Gv	NL	Policy and legal officer for circular economy	February 2020	5, 7	
87	Gv	NL	Monitoring agenda, Dutch waste flows	February 2019	5, 7	
88	Gv	NL	Monitoring agenda EPR and plastics	February 2019	4, 5, 7	
89	PRO	NL	NA	April 2020	7	Report
90	Pr	NL	NA	April 2020	7	Report
91	Pr	Switzerland	Researcher	January 2022	7	Policy brief

Governmental = Gv; Private = Pr; University = Un; Third sector = Ts

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Company and site visits

#	Type of company	Country	Date
1	Energy provider	United Kingdom	September 2018
2	Tyre recycling company	Netherlands	January 2019
3	Tyre pyrolysis company	Netherlands	January 2019
4	Food waste prevention company	Netherlands	January 2019

#	Type of company	Country	Date
5	Car recycler	Netherlands	January 2019
6	Tyre PRO	Netherlands	March 2019
7	Tyre manufacturer	Netherlands	March 2019
8	Wine producer	Portugal	September 2019
9	Industrial symbiosis network	France	January 2020
10	Research institute on Bioeconomy and Biotechnology	France	January 2020
11	WEEE refurbishing company	France	January 2020
12	WEEE recycling facility	Italy	November 2020
13	WEEE recycling facility	Italy	December 2020
14	WEEE recycling facility	Italy	April 2021
15	Packaging manufacturer	Italy	April 2021 (online)
16	WEEE recycling facility	Italy	May 2021
17	WEEE PRO	Netherlands	November 2021
18	WEEE sorting facility	Netherlands	November 2021

Stakeholder workshops

#	Workshop description	Number of attendees	Date
1	Science policy discussion, with colleagues from Rijkswaterstaat Environment. Title presented: Moving to circular economy 3.0: upcycling in the context of resource scarcity. This was followed by a roundtable discussion on the challenges and knowledge gaps for circular economy in the context of the current organization of waste management in the Netherlands.	12	May 2019
2	Science policy workshop titled: Extended Producer Responsibility & Circular Economy: Refurbishing the Instrument. Purpose: to discussion possibilities with practitioners from science and policy for improving the instrument extended producer responsibility. I presented the insights from Chapter 4 and facilitated a group discussion. This workshop contributed to some of the conceptual insights in starting Chapter 5.	14	January 2020
3	Workshop with practitioners discussing the preliminary results from the Delphi study (Chapter 5), and reflecting upon them. More detailed description in Chapter 5.	13	September 2020
4	Workshop with colleagues from Rijkswaterstaat and the Dutch Ministry of Infrastructure and Water Management. Title presented: Developing extended producer responsibility for circular economy. I presented insights from the policy brief based on Chapter 5, followed by a roundtable and discussion on how to develop the instrument extended producer responsibility.	10	March 2021

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#	Workshop description	Number of attendees	Date
5	Workshop with OECD working group on circular economy and extended producer responsibility Title presented: Extended producer responsibility and critical raw materials. Purpose: to discuss broader issues related to the design, and functioning of extended producer responsibility within the European Union.	5	November 2021
6	Workshop with colleagues from Rijkswaterstaat and the Dutch Ministry of Infrastructure and Water Management. Title presented: Extended producer responsibility and critical raw materials. Purpose: to discuss broader issues related to the design, and functioning of extended producer responsibility within the European Union.	8	November 2021

Project workshops

#	Project description	Location	Date
1	Purpose: Introducing the Circular Economy; Research design for multidisciplinary and interdisciplinary projects; Ethics; Career Development Planning; Local Circular Economy Showcase. This workshop contributed classroom and field content for the first year of training within the Cresting project. Activities comprised a combination of speakers, discussion, and site visits. Topics covered were: 1. Stakeholder perspectives on the circular economy; 2. Employment in the circular economy; 3. Career development planning; 4. Research methodologies: transdisciplinary research, interdisciplinary research and critical realism; 5. Ethics in research; and 6. Investigating the circular economy in practice.	Hull, UK	September 2018
2	Purpose: Circular Economy Policy; Cross-cultural Research; Observing, participating and reflecting as research methods and training for employment; Local Circular Economy Showcase. This second workshop built on the first workshop of September 2018 in Hull, and was a part of the first year of training within the Cresting project. Activities comprised a combination of speakers, group discussion, and field visits. Topics covered were: 1. Sharing the progress made and identifying areas of common interest and needs for mutual recognition, integration or alignment; 2. Identifying stakeholder's knowledge needs and required capabilities of academic professionals in the field of circular economy; 3. Research methodologies: translating transdisciplinary research and critical realism into effective methods for researcher's engagement in secondments; and 4. Investigating the circular economy in practice.	Utrecht, NL	January 2019

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#	Project description	Location	Date
3	Purpose: Public Sector and the Circular Economy; Academic publishing; Local Circular Economy Show case. The central theme of this workshop was "Public Sector and the Circular Economy (CE)" and aimed to highlight the achievements and challenges of the Portuguese public sector regarding CE related policies, projects and practices. Topics covered were: • Exploring the links between public and private sector CE initiatives; • Demonstration real life cases of CE policy-to-practice; • Participatory research and design thinking; and • Academic publishing.	Lisbon, PT	September 2019
4	Purpose: Building a Regional Circular Economy; Communicating and Disseminating Research in an Open Access World; Local Circular Economy Employment Showcase. This workshop is the second workshop of the second year of PhD training for the Early Stage Researchers. This workshop focussed on the following objectives: 1. To provide knowledge and understanding of issues involved in building a regional circular economy; and 2. To introduce practices and implications of communication and dissemination, including in an open source access content.	Troyes, FR	January 202
5	Purpose: Business and the Circular Economy; Exploitation of Research; Building Relationships with Stakeholders; Gender in the Workplace; Local Circular Economy Employment Showcase; Stakeholder Event – valorising results. This workshop is the first workshop of the final year of training and focused on the following objectives: 1. Business engagement with the CE; 2. Exploitation of research: Gender in the workplace; and 3. Local employment show case and stakeholder event for valorisation of CRESTING findings. The workshop explored the relationships established between private sector actors collaborating in the context of circular economy projects and reflected on the relationships we have established with our respective project partners during our research. We used the tools of ideation and analysis; business case and decision-making used in company settings to give circular solutions for a sustainability challenge; explored ways of increasing the impact of our research both in terms of content and communication and to understand how to navigate the ever-changing labour market scenarios regarding hard and soft skill development.	Graz, AT	September 2020 (online)

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#	Project description	Location	Date
6	Purpose: Measuring the Circular Economy; Local Circular Economy Employment Showcase; Stakeholder Event – valorising results. This workshop is the final workshop for the CRESTING Early Stage Researchers and had the following objectives: 1. Measuring the circular economy 2. Local employment showcase and stakeholder event for valorisation of Cresting research. The workshop from 12-16 April was titled "Assessing the Sustainability of the Circular Economy", thus mainly revolving around the question, whether a circular economy would also be a more sustainable one. Two panel sessions, one on the connection of circular economy and sustainability assessment, and one on sustainable and circular finance, brought together a broad group of stakeholders. The ESRs were further trained how to assess circularity and to relate the findings to life cycle assessment results. Another crucial aspect of the workshop was co-authoring with academics from other fields and the private sector, and targeted dissemination of the created knowledge to audiences outside academia, such as businesses and policy makers. Given this was the last workshop, one session also focused on how to write a grant proposal for future research.	Pescara, IT	April 2021 (online)

Conference presentations

#	Conference	Title of presented work
1	25th International Sustainable Development Research Society Conference, University of Nanjing, China, 26 - 28 June 2019	The failing success of rubber tyre recycling in the Netherlands
2	25th International Sustainable Development Research Society Conference, University of Nanjing, China, 26 - 28 June 2019	Downcycling, upcycling and organizational recycling: mechanisms and practices within the EU, a research framework
3	10th International Conference on Industrial Ecology, Tsinghua University, Beijing, China, 7 - 11 Jul 2019	Preferential material allocation for a circular economy: the challenges and possibilities of applying the cascading principle
4	26th International Sustainable Development Research Society Conference, University of Budapest, Hungary 15-17, July 2020 (online)	Future perspectives on extended producer responsibility: a Delphi study
5	27th International Sustainable Development Research Society, Hosted by Mid Sweden University, July 2021(online).	Resident perspectives of design for sustainability: insights from the European Union.
6	28th International Sustainable Development Research Society, Stockholm, July 2021.	Lessons on extended producer responsibility in the context of circular economy

Secondment stay description

The secondment for the project is a placement with **Rijkswaterstaat,** based in the Netherlands. Rijkswaterstaat is the executive body of the Dutch Ministry of Infrastructure and the Environment. **Rijkswaterstaat Environment** provides knowledge in relation to various environmental areas including soil remediation, protection and management; best practices for environmental legislation; transportation networks; local climate policy and waste and materials.

The secondment lasted from January to April 2019, and consisted of 1 – 3 days per week. The primary aims were:

- Explore using the secondment placement as a site for research;
- Make connections to develop research projects and share future results;
- Locate initial research subjects;
- Explore the current governance approach to secondary materials management and recycling, including current policy approaches, organisation, implementation and monitoring.

The primary outputs and results from this stay included:

- A large network of policy officers, and civil servants working on circular economy (many attended presentation briefings and stakeholder workshops);
- A comprehensive overview of waste policy in the Netherlands, including main waste streams and instruments used;
- A place to present the output of for the research of this project.

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Summary

The current production and consumption patterns have created unprecedented ecosystem destruction. In the past 20 years, circular economy (CE) has been embraced as a tool to reconcile both environmental and economic practices. Through altering to so-called linear nature of the economy through closing material and energy loops, two dual aspects underpinning environmental degradation are argued to be mitigated: resource depletion and waste generation. CE has been argued to be a more concrete means to contribute to sustainable development. In the midst of this excitement, there is a need to critically reflect on existing governance practices and experiences that already contribute to CE. Instead of understanding CE as a purely contemporary phenomenon, this PhD thesis puts it in a historical context and argues it should be seen as the third iteration of policies and practices that sought to deal with resource use and waste generation. These earlier phases brought issues related to the environment into public discussion and resulted in distinct governance practices that shape the flow of materials through the economy.

To contribute to the theoretical and empirical understanding of the governance of CE, this PhD thesis focused on experiences and practices of extended producer responsibility (EPR) policies in the EU. EPR policies emerged in the 1990s in North-Western Europe and involved shifting the end-of-life (EoL) responsibility for a product away from municipalities to producers. EPR policies and practices represent an older iteration of CE-like policies and are also a concrete focus of the EU's CE agenda, thus, making it an ideal research area. This research drew on grounded theory and transdisciplinarity approaches to research whilst employing various mixed methods to explore and analyse the governance of EPR within the EU. These methods focused on conceptualising, describing and critically analysing the organisation, performance and outcomes of several EPR case studies. The case study outcomes revealed the core organisational attributes, mechanisms and weaknesses that pertain EPR's propensity to contribute to the CE implementation. Chapters 2-6 of this thesis address different aspects and perspectives of EPR that pertain to the following sub research questions:

1. How has EPR been implemented and organised within EU member states?
2. How do EPR strategies vary in respect to their effectiveness, limitations and outcomes?
3. What (EPR) mechanisms (and value decisions) are used to identify the trajectory and use of products and materials post-collection?
4. How can current approaches to EPR be further developed and strengthened to contribute to CE?

Chapter 2 provided a general synthesis of the literature on EPR and outlined a historical and global understanding of it. This chapter shows EPR is a policy approach or specific instrument encompassing a broad array of requirements (informative, physical, economic

etc.) placed on the original product producer. These requirements predominantly take the form of physical take-back requirements at the EoL phase, e.g. post-user phase. This chapter outlined some general insights into the instrument's effectiveness over the previous 30 years and discussed its role in fostering a CE, which requires i) a stronger emphasis on extension of the lifetime of products in use; ii) applying far more options of value retention than the recycling of mixed streams of post-consumer waste; and iii) stronger emphasis on producers developing new circular business models.

Chapter 3 reviewed the key literature on CE, cascading and up/downcycling and proposed a framework that considers the social and governance implications for the subsequent application and allocation of products and materials using their lifecycle. This chapter integrates the two disciplinary lenses (governance and industrial ecology) by proposing a framework that enables a dual perspective for understanding product and material flows: the physical flows and the socio-governance context that facilitate decision-making processes of material outcomes. This conceptual thinking provided the reflexive and explorative rationale for considering EPR systems as the decision-making mechanism for post-user waste products.

Chapter 4 examined the governance and performance of EPR for tyres in the Netherlands. It described the key actors, policy developments and responsibilities in this specific system. This EPR system is based on a single producer responsibility organisation (PRO), i.e. the collective representation for all producers of that product, with multiple collection and processing services. This chapter mapped the performance of EPR in the Netherlands but also explored alternative material outcome strategies and associated limitations. It argued that despite high levels of material recovery (Recycling and Energy Recovery), there are limited circularity outcomes. This chapter proposes several critical points to improve EPR in the context of CE, namely on issues related to transparency and actor integration.

Chapter 5 examined the perspectives of EPR and CE experts, exploring how EPR could be transformed to aid the CE goals of the Netherlands (which had a target of a 50% reduction in primary raw materials by 2030). This focused on four sectors (electronics, cars, floor covering and flat glass) and included 50 CE and EPR experts. A participatory method (a Delphi study) was used to synthesize and refine the expert knowledge on a subject while observing the differences between participants. This chapter outlined several areas for improvement for EPR to further contribute to CE. This chapter illustrates that whilst there was a level of agreement between actors for the need for a CE, the willingness of who should accept new responsibilities within EPR was heavily contested.

Chapter 6 examined the presence, quantities and losses of critical raw materials within WEEE recycling processes in the context of EU EPR policy. Using an indicator called thermodynamic rarity, which shows the exergy needed for mining and refining specific materials, this chapter indicates the differences between collection and recycling policies based on mass instead of rarity (which emphasises the conservation of materials based

on their geological availability). This indicator illustrates the weakness within the EU's EPR targets for WEEE, which are based on prompting mass collection and recycling at the expense of other quality-related outcomes, e.g. rarity. This chapter outlines several potential recommendations for improving EPR for critical raw materials preservation.

Collectively, these chapters provide insights on answering the main research question: to what extent can extended producer responsibility (as a form of public-private governance) act as an effective mechanism in supporting the implementation of circular economy in the European Union?

When answering this question, we can consider the effectiveness of EPR based on the original intention the policy was designed for, e.g. relieving pressure from municipalities and giving post-user responsibility to producers. In this sense, EPR systems can be viewed as effective across the various product categories studied. However, on a broader level, when connecting EPR more closely to the CE debate, the extent of the effectiveness is more limited. As outlined in the various chapters, the current organisation of EPR has led to several operational and governance issues, e.g. transparency, material outcomes and waste leakages. Other limitations pertain to the lack of integration of other CE actors, e.g reuse, repair etc, within EPR systems.

Beyond these operational conclusions, we observe that while EPR have been efficient as public-private organisations in fulfilling their original intention, these intentions are themselves not suitable for the contemporary demands of CE, i.e. reducing the demand of new materials. Based on this, we argue that the state needs to become more active in redefining the roles and responsibilities within EPR systems to contribute to more circular and sustainable outcomes, including behavioural and consumption pattern changes. We further argue that these outcomes must result in the evolution of EPR end-of-life policy, from promoting collection and recycling to becoming a diagnostic tool to inform product design changes, and in this sense, connecting EPR back to the original intentions in which it was designed.

Samenvatting

Moderne productie- en consumptiepatronen hebben geleid tot een ongekende vernietiging van ecosystemen. In de afgelopen 20 jaar is de circulaire economie (CE) omarmd als een manier om de ecologische en economische praktijken met elkaar te verbinden. Door het lineaire karakter van de economie te veranderen in materiaal- en energiekringlopen, kunnen twee ontwikkelingen die grote invloed hebben op het aantasten van ecosystemen worden verzacht: de uitputting van hulpbronnen en de productie van afval. Volgens sommige wetenschappers is CE een concrete aanpak is in het proces naar duurzame ontwikkeling. Bovendien is er behoefte aan een kritische reflectie op bestaand beleid en praktijkervaringen gerelateerd aan CE. Dit proefschrift ziet CE niet als een puur hedendaags fenomeen, maar plaatst het in een historische context. De stelling wordt ingenomen dat CE gezien kan worden als een derde vorm van beleid en praktijken rond het gebruik van hulpbronnen en de productie van afval. De eerdere vormen brachten de milieukwesties aan het licht in de publieke discussie en resulteerden in verscheidende bestuurspraktijken die de stroom van materialen door de economie vormgeven.

Dit proefschrift draagt bij aan de theoretische en empirische kennis over beleidsvorming en uitvoering van de circulaire economie, door zich te richten op ervaringen en praktische toepassingen van "extended producer responsibility" (EPR) in de Europese Unie. Het EPR-beleid, ontstaan in de jaren '90 in Noordwest-Europa, heeft als doel om de verantwoordelijkheid voor product-levensduur (End-of-life of "EoL") te verleggen van gemeenten naar producenten. Het ERP en de uitvoering daarvan komt voort uit een eerdere vorm van CE-achtig beleid. Het is echter ook een concreet aandachtspunt binnen de huidige CE-agenda van de EU. Hierdoor is het een ideale onderzoekcasus. Dit onderzoek is gebaseerd op "grounded theory" en transdisciplinariteit, en gebruikt verschillende methodieken om de beleidsvorming en uitvoering van EPR binnen de EU te verkennen en te analyseren. Dit onderzoek gebruikt deze methodieken voor het conceptualiseren, beschrijven en kritisch analyseren van de organisatie, prestaties en uitkomsten van verschillende EPR-casestudies. De uitkomsten brengen de belangrijkste organisatorische kenmerken, onderliggende mechanismes en belangrijkste zwakke punten aan het licht die invloed hebben op de bijdrage van EPR-beleid aan de implementatie van de CE. De hoofdstukken 2 tot 6 van dit proefschrift behandelen de verschillende aspecten en perspectieven van EPR die betrekking hebben op de volgende deelvragen:

1. Hoe is EPR geïmplementeerd en georganiseerd in de EU-lidstaten?
2. Hoe variëren de effectiviteit, beperkingen en uitkomsten van EPR-strategieën?
3. Welke (EPR-) mechanismes (en morele beslissingen) worden gebruikt om het traject en gebruik van producten en materialen nadat ze zijn verzameld vast te stellen?
4. Hoe kan het huidige EPR-beleid en de EPR-toepassingen verder worden ontwikkeld en zodat het de bijdrage aan de Circulaire Economie versterkt?

Hoofdstuk 2 bevat een synthese van de literatuur over EPR en schetst een globaal historisch beeld van EPR. Uit dit hoofdstuk blijkt dat EPR een beleidsbenadering of een specifiek instrument is dat een breed scala van eisen (informatief, fysiek, economisch enz.) bevat waar de producent van het oorspronkelijke product aan moet voldoen. Concreet zorgen deze criteria voornamelijk voor fysieke terugname-eisen in bepaalde EoL-fases, zoals in de fase na gebruik. Dit hoofdstuk schetst verder enkele algemene inzichten in de doeltreffendheid van het instrument in de afgelopen 30 jaar, en bespreekt de rol van EPR in de bevordering van CE, die 1. Een grotere nadruk legt op de verlenging van de levensduur van producten; 2. Meer opties biedt voor waardenbehoud dan enkel via recycling van afval na consumptie; en 3. Een sterke nadruk legt op dat producenten nieuwe circulaire bedrijfsmodellen ontwikkelen.

In hoofdstuk 3 wordt een literatuurstudie gepresenteerd over CE, cascading en up/downcycling. Er wordt een nieuw denkkader voorgesteld dat ruimte biedt voor de maatschappelijke en beleidsimplicaties voor mogelijke toepassingen van producten en materialen in verschillende fasen van de levenscyclus van het oorspronkelijke product. In het hoofdstuk worden twee perspectieven geïntegreerd (die uit bestuurskunde en industriële ecologie) in een denkkader dat een tweeledige invalshoek mogelijk maakt om product- en materiaalstromen te begrijpen: de fysieke materiele stromen en de sociaal-bestuurlijke context die besluitvormingsprocessen rondom materiaal uitkomsten versimpelt. Dit denkkader biedt een reflexieve en exploratieve logica die EPR-systemen helpt te begrijpen als besluitvormingsmechanismes voor afvalproducten na gebruik.

In hoofdstuk 4 wordt de beleidsvorming en het succes van EPR-beleid rondom de productie van autobanden in Nederland uiteengezet. Eerst worden de belangrijkste actoren, beleidsontwikkelingen en verschillende verantwoordelijkheden in het specifieke systeem beschreven. Dit specifieke EPR-systeem maakt gebruik van een enkele vertegenwoordiger van alle producenten van dit specifieke product, ook wel een "producer responsibility organisation" (PRO) genoemd. In dit hoofdstuk wordt het effect van EPR in Nederland in kaart gebracht, en worden alternatieve strategieën en hun beperkingen onderzocht. Het hoofdstuk betoogt dat ondanks een relatief hoog niveau van terugwinning van materiaal (via recycling en energie terugwinning), de resultaten ter bevordering van de circulariteit beperkt zijn. Het hoofdstuk stipt ook verschillende reflectiepunten aan, met name met gerelateerd aan transparantie en integratie van verschillende actoren, die het EPR-beleid in de context van CE kunnen verbeteren.

In Hoofdstuk 5 zijn de perspectieven van EPR en CE-experts onderzocht, met het doel om te achterhalen hoe EPR getransformeerd kan worden om de Nederlandse CE doelstelling (een 50% reductie van primaire grondstoffen voor 2030) te ondersteunen. Het hoofdstuk richt zich op vier sectoren (de productie van elektronica, auto's, vloerbedekking en vlakglas). Een participatieve methodiek (Delphi studie) is gebruikt om de kennis van 50 EPR en CE-experts Te analyseren om overeenkomsten en verschillen helder te krijgen. In het hoofdstuk worden verschillende manieren uiteengezet hoe EPR veranderd kan worden om beter bij te dragen

aan CE. Daarnaast illustreert de studie dat terwijl er een zekere mate van overeenstemming bestond over de noodzaak van een circulaire economie, de bereidheid om nieuwe verantwoordelijkheden binnen de EPR op zich te nemen beperkt was.

In hoofdstuk 6 worden de aanwezigheid, de hoeveelheden en het verlies van cruciale grondstoffen via afgedankte elektrische en elektronische apparatuur (AEEA) onderzocht in de context van het EU-beleid voor terugwinning van energie. Met behulp van de indicator "thermodynamic rarity" wordt in dit hoofdstuk de verschillen in inzamelings- en recyclebeleid geanalyseerd. Thermodynamic rarity geeft aan hoeveel exergie nodig is voor de winning en verwerking van specifieke materialen. In de analyse staat de hoeveelheid materiaal centraal in plaats van de zeldzaamheid daarvan. De nadruk wordt hierbij gelegd op het behoud van materialen op basis van hun geologische beschikbaarheid. Met deze studie wordt de zwakte van de Europese EPR-doelstellingen voor AEEA blootgelegd, namelijk dat die zijn gebaseerd op het stimuleren van massale inzameling en recycling ten koste van andere kwaliteitskenmerken zoals de zeldzaamheid.

Gezamenlijk bieden deze hoofdstukken inzichten die helpen om de hoofdvraag te beantwoorden: In hoeverre kan EPR (als een vorm van publiek-privaat beleid) fungeren als een effectief mechanisme ter ondersteuning van de implementatie van circulaire economie in de Europese Unie?

Bij het beantwoorden van deze vraag moet de effectiviteit van EPR in het behalen van de originele doelstellingen waarvoor het beleid is ontworpen meegenomen worden. Die doelstellingen zijn onder andere het verlichten van de druk op gemeenten en het leggen van verantwoordelijkheid na gebruik bij producenten. Vanuit dat perspectief kunnen de EPR-systemen in de verschillende productcategorieën die onderzocht zijn als effectief worden beschouwd. Echter, wanneer EPR-beleid verbonden wordt aan het CE-debat, is de mate van effectiviteit een stuk beperkter. Zoals aangetoond in verschillende hoofdstukken, heeft de huidige organisatie en structuur van EPR geleid tot verschillende beleids- en uitvoeringsproblemen, zoals beperkte transparantie, matige materiaalresultaten en afvallekken. Ook zijn er verscheidene beperkingen door het gebrek aan integratie van andere CE-aspecten, zoals hergebruiken en repareren, in de EPR-systemen.

Naast deze organisatorische conclusies, concludeert dit proefschrift dat EPR als publiek-private organisatie weliswaar efficiënt is gebleken in de oorspronkelijke doelstellingen. Deze doelstellingen zijn echter niet toereikend voor het bereiken van de doelstelling voor een circulaire economie, namelijk het terugdringen van de vraag naar nieuwgewonnen grondstoffen. Op basis hiervan is te stellen dat de overheid actiever moet optreden en de rollen en verantwoordelijkheden binnen de EPR-structuren moet herdefiniëren. Op die manier kan bijdragen aan circulaire en duurzame uitkomsten, inclusief verandering van gedrags- en consumptiepatronen. Daarnaast moeten deze veranderingen resulteren in een transformatie van het EPR EoL beleid van een bevorderingsinstrument voor verzameling en recycling tot een diagnostisch instrument om verandering in productontwerp te onderbouwen. Op die manier wordt EPR-beleid teruggevoerd naar de originele bedoeling waarvoor het werd ontworpen.

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About the author

Kieran Campbell-Johnston (1993) is originally from Norfolk in the United Kingdom. He studied a bachelor's Degree of Arts in Contemporary History at the University of Sussex from 2013 – 2016, where he worked on topics including US-Middle Eastern relations post-World War Two, and female destitution and labour unions in Britain in the 19th century. During his studies he became increasing interested in issues of environmentalism and environmental policy, after volunteering as a coordinator for the Green Party of England and Wales in Brighton.

After working as a secondary school teacher in Hong Kong, he moved to Amsterdam to study his masters in Social Geography with a focus on Environmental Governance. During this period, he became interested in concept and application of the circular economy, and wrote his thesis (later published) on circular economy policies in urban areas. During this time, he was also sustainability advisor for the Amsterdam-based social housing cooperative de Nieuwe Meent. Since 2018, he has been working as a PhD at the Copernicus Institute of Sustainable Development at Utrecht University. Kieran is an avid hiker, reader and climber as well as an perennial politics enthusiast.

Research context

This research was one of 15 individual research projects, all involved in an Horizon2020 ITN research programme called Cresting. The Cresting project looked at the broad sustainability implications of the CE. This included five distinct work packages, which examined current practices and discourses, corporate engagement, public sector engagement, capturing the benefits of CE and measure the impacts of CE. These projects were spread across 8 European universities, with partners in Nigeria and China. The PhDs were partnered with two or three universities, with which they held research visits. During the three years of the project, meetings and trainings were held in Hull, Utrecht, Lisbon, Troyes, Graz (online) and Pescara (online). This diversity of project partners, institutors and experiences have allowed the project and researchers to benefit from a diverse and interdisciplinary array of perspectives.

The Cresting Early Stage Researchers and supervisors Pauline Deutz and Andy Jonas at the first Cresting meeting in Hull, September 2018.