EXTENDED PRODUCER RESPONSIBILITY

DESIGN, FUNCTIONING AND EFFECTS

Background document

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Contents

Summary

This report presents an overview of the design, functioning and effects of extended producer responsibility (EPR), both in theory and policy practice. EPR is a policy approach in which the responsibility of producers is extended to the post-consumer stage of a product's lifecycle. It employs a diverse policy instrument mix that aims to make producers — financially, and sometimes also organisationally — responsible for the collection, sorting and treatment of end-of-life products. It thereby aims to increase the separate collection of end-of-life products and to enable their more circular treatment.

This study analyses the effects of EPR on waste management, recycling and secondary material markets, and eco-design. To this end, we have conducted a literature review and interviews with stakeholders. Complementary to the current report, CPB and PBL publish three case studies on specific product groups, namely batteries, end-of-life vehicles and unused medicine (Tijm et al., 2021).

EPR is a widely used policy to support the transition to a circular economy, both in the Netherlands and in the rest of the European Union. EPR schemes on batteries, end-of-life vehicles, electric and electronic equipment, and packaging are implemented across the European Union. In the Netherlands, EPR also applies to car tyres, paper and cardboard, and flat glass. The Dutch government is also in the process of developing EPR policy for other product groups, including mattresses, textiles and some types of single-use plastics.

The most common EPR instruments are take-back requirements, advance disposal and recycling fees, and deposit-refund systems. Take-back requirements — the most common EPR instrument – oblige producers to collect their products at the end of their life and to organise their appropriate treatment. This is usually pursued through specific quantitative collection and/or recycling targets. Advance disposal or recycling fees are upfront payments required by consumers to cover the end-of-life collection and treatment costs of a product. In deposit-refund systems, consumers pay a deposit when purchasing a product and receive a refund (usually of the same value) upon its return to a drop-off point. This provides consumers a price incentive to properly sort and dispose of end-of-life products, and thus to reduce littering.

The effectiveness of EPR in achieving circular economy goals is determined by its design. We analyse two principal elements of EPR design: the EPR instrument mix, and whether producers fulfil their obligations individually or collectively. Beyond the type of EPR instruments used, the focus, scope, stringency and adaptability over time of the mix matters for its effectiveness, as does its coherence with other policies. In terms of organisation, firms typically choose to organise EPR collectively and establish *producer responsibility organisations* (PROs) to manage collection and treatment activities. While collective EPR offers advantages to producers, such as economies of scale and less free-riding, it dilutes eco-design incentives and may raise competition concerns.

EPR has increased the share of end-of-life products being separately collected and treated, but its impact on waste management costs is unclear. To increase collection and recycling rates, producers (usually via a PRO) set up return points where consumers can hand in their end-of-life products, organise the treatment of (hazardous) materials with negative residual value, and launch public awareness campaigns to promote proper disposal. Although all three main instruments examined in this study are likely to increase separate collection rates, take-back requirements and deposit-refund systems provide stronger incentives for this purpose. EPR has also been effective in shifting the financial burden of collection and treatment of products away from municipalities and

towards producers and consumers. It is not clear though whether producers organise waste collection and treatment more efficiently than municipalities.

Additional incentives are needed so that EPR can effectively promote reuse. Incentives for producers to carefully handle collected products so that their value is retained, to disseminate information facilitating the reuse and repair of their products, and to lead campaigns to raise public awareness of its benefits are all likely to increase reuse. Amongst the considered EPR instruments, experience with deposit-refund systems shows that they are well-positioned to promote reuse, while a relatively unexplored alternative is to develop separate reuse targets.

Moving to secondary material markets, we find that EPR increases the quantity of material supplied to recycling, but its impact on the quality of recycling output is uncertain. The use of secondary materials in production processes leads to important environmental benefits, compared to the use of virgin ones. While EPR stimulates the use of recycled materials via higher supplied quantities and reduced production costs (economies of scale), its impact on secondary material quality is open to debate. When the material provided for recycling is of low quality, a large amount of it is downcycled into products of lower value.

Deposit-refund systems and differentiated (modulated) fees can improve the quality of secondary material, as can incentives on the demand side, such as minimum recycled content requirements. Deposit-refund systems can best ensure that returned products are of homogenous and high quality. Fees differentiated according to the recyclability of the product may also lead to more homogenous inflows to recycling plants. The effects of these instruments can be reinforced by complementary policies that shift demand towards secondary materials, such as minimum recycled content requirements for new products, and pricing of the external costs of raw material extraction and processing.

There is little evidence to date of EPR instigating eco-design, but differentiated tariffs and funds on green innovation can improve its performance in this respect. To trigger incentives for eco-design, EPR would need to make environmentally damaging product attributes more expensive for producers. Indeed, collection and recycling targets based on product weight and weight-based fees already exist, but reduced weight is merely one dimension in which eco-design can help achieve circular economy goals. To increase EPR's potential to foster eco-design, the fee charged for each product by PROs should reflect the social (i.e. the sum of private and external) costs of collection and treatment at the end of its life.

A wider policy implication of our study is to make producers financially responsible also for the end-of-life products that are not separately collected. In practice, producers' financial responsibility is limited to separately collected product streams. This could be complemented by fees charged by public authorities, which would cover costs of managing end-of-life products in household waste, and of cleaning up littered or dumped products. Such an expansion of the scope of EPR would incentivise producers to constantly strive for higher collection and recycling rates, promote eco-design and relieve the financial burden on municipalities.

Well-designed EPR is a useful ingredient of the policy mix *en route* to the circular economy, but no panacea. EPR has increased collection rates, promoted recycling and shifted financial responsibility from municipalities to producers. While knowledge gaps exist, especially related to the lack of empirical research, there seems to be ample room to steer EPR instruments towards eco-design and reuse. Yet the circular economy aspires to more than what EPR can deliver on its own. Just as its effects depend on other (waste) policies, EPR will always require accompanying policies — mostly targeted at the production and consumption phases of a product's lifecycle — to facilitate the transition to a circular economy.

Samenvatting

CPB en PBL geven in dit rapport een overzicht van het ontwerp, de werking en de effecten van Uitgebreide Producentenverantwoordelijkheid (UPV), zowel in theorie als in de beleidspraktijk. UPV is een beleidsbenadering waarin de verantwoordelijkheid van producenten wordt uitgebreid tot na het einde van de levensduur van een product. Het omvat een divers palet aan beleidsinstrumenten met als doel producenten verantwoordelijk te maken — in financiële, en soms ook in organisatorische zin — voor de inzameling, sortering en verwerking van afgedankte producten. Het overdragen van verantwoordelijkheid gebeurt om een verhoogde inzameling en meer circulaire afvalverwerking mogelijk te maken.

In dit achtergronddocument analyseren we de effectiviteit van UPV op afvalbeheer, op recycling en de markt voor secundair grondstoffen, en op het milieuvriendelijk ontwerp van producten oftewel *ecodesign*. Hiertoe hebben we een beoordeling gemaakt van de beschikbare literatuur, en interviews gehouden met stakeholders. Complementair aan dit rapport publiceren drie case studies naar specifieke productgroepen, te weten batterijen, autowrakken, en ongebruikte medicijnen (Tijm et al., 2021)

UPV is een veelgebruikt onderdeel van het transitiebeleid naar een circulaire economie, zowel in Nederland als in de rest van de EU. In de gehele Europese Unie zijn UPV-systemen geïmplementeerd op het gebied van batterijen, autowrakken, elektrische en elektronische apparatuur, en verpakkingen. In Nederland geldt er ook UPV voor autobanden, papier en karton en vlakglas. Nieuw UPV-beleid in Nederland is in de maak voor andere productgroepen, waaronder matrassen, textiel, en sommige vormen van wegwerpplastic.

De meest voorkomende UPV-instrumenten zijn inzamelings- of recyclingdoelen, heffingen in de consumentenprijs en statiegeldsystemen. Inzamelings- of recyclingdoelen — het meest voorkomende UPV-instrument — verplichten de producent om een product aan het einde van de levensduur terug in te nemen en te zorgen voor een passende verwerking. De doelstelling bestaat uit een gekwantificeerd aandeel producten dat bereikt moet worden. Heffingen, zoals de verwijderingsbijdrage, verhogen de prijs voor consumenten en leveren een bijdrage aan de inzameling en verwerking van afgedankte producten. Bij een statiegeldsysteem betalen consumenten een bedrag als borg dat geretourneerd wordt indien zij het gebruikte product op juiste wijze weer inleveren. Statiegeld geeft een monetaire prikkel om bruikbare onderdelen te scheiden van restafval, en om zwerfafval tegen te gaan.

De effectiviteit van UPV in het bereiken van de doelen van de circulaire economie wordt bepaald door haar ontwerp. We analyseren de twee voornaamste elementen in het ontwerp van EPR: de aangewende instrumentenmix, en of producenten hun verplichting individueel of collectief voldoen. Behalve het type UPV-instrumenten zijn de focus, de reikwijdte, de mate van striktheid, en de dynamiek over de tijd van belang voor de effectiviteit, net als de coherentie met ander beleid. Wat betreft de ordening van UPV organiseren bedrijven zich vaak als collectief in producentenverantwoordelijkheidsorganisaties (PVO's). Collectieve UPV biedt producenten voordelen, zoals schaalvoordelen en minder free-riding, hoewel collectiviteit prikkels voor ecodesign verzwakt en mogelijk leidt tot zorgen over de mededinging.

UPV heeft tot een verhoging van het aandeel producten dat apart ingezameld en verwerkt geleid wordt heeft, hoewel het onduidelijk is of hiermee de maatschappelijke kosten van afvalbeheer zijn gedaald. De verhoging van het aandeel ingezamelde en verwerkte producten duidt erop dat de prikkels van inzamelings- en recyclingsdoelstellingen effectief zijn, en ook statiegeldsystemen geeft sterke prikkels om dit doel te bereiken. Producenten bereiken de verhoging (vaak middels een PVO) door het plaatsen van afgiftepunten voor afgedankte producten, de verwerking van (gevaarlijke) stoffen met een negatieve restwaarde, en door publiekscampagnes, bijvoorbeeld gericht tegen zwerfafval. UPV is ook effectief geweest in het verschuiven van de financiële lasten voor verzamelen en verwerken van gemeenten naar producenten. Het is echter niet eenduidig vast te stellen of producenten afvalinzameling en -verwerking efficiënter organiseren dan gemeenten.

Om ervoor te zorgen dat UPV ook effectief bijdraagt aan meer hergebruik dienen aanvullende prikkels te worden ontwikkeld. Dergelijke prikkels zouden erop gericht zijn dat PVO's zorgvuldig omgaan met ingezameld producten en daarmee hun waarde behouden, actief informatie verspreiden over hergebruik en reparatie van hun producten, en campagnes opzetten om consumenten bewust te maken van deze mogelijkheden. Wat betreft UPV-instrumenten, heeft ervaring met statiegeldsystemen aangetoond dat zij geschikt zijn om hergebruik te bevorderen. Een nog relatief onbekend terrein is het ontwikkelen van specifieke doelen voor hergebruik.

Afgezien van effecten op afvalbeheer vinden we dat UPV de hoeveelheid materiaal verhoogt dat beschikbaar is voor recycling, hoewel de impact van UPV op de kwaliteit van gerecycled materiaal onzeker is. Gebruik van secundair materiaal in productieprocessen leidt tot belangrijke milieubaten in vergelijking tot het gebruik van primaire grondstoffen. Hoewel UPV het gebruik van gerecyclede materialen stimuleert via een hoger aanbod en lagere productiekosten, is de impact op de kwaliteit van secundair materiaal nog een open vraag. Wanneer het verzameld materiaal voor recycling van lage kwaliteit is, leidt dit tot 'downcycling', oftewel verwerking in producten met een lagere waarde.

Statiegeldsystemen en gedifferentieerde tarieven kunnen de kwaliteit van secundair materiaal verhogen, aangevuld met prikkels aan de vraagzijde van de recyclingsmarkt zoals minimumeisen voor gerecycled materiaal. Statiegeldsystemen zijn het meest geschikt om te zorgen voor homogene en hoge kwaliteit van geretourneerde producten. Tarieven die differentiëren naar de recyclebaarheid van het product kunnen ook leiden tot homogenere instroom naar recyclingbedrijven. De effecten van deze instrumenten kan worden versterkt met aanvullend beleid dat de vraag naar secundair materiaal stimuleert, zoals minimumeisen voor gerecycled materiaal bij nieuwe producten, en het beprijzen van de externe kosten van winning en verwerking van primaire grondstoffen

Tot op heden is er weinig bewijs dat UPV aanzet tot ecodesign, maar gedifferentieerde tarieven en fondsen voor groene innovatie kunnen hieraan bijdragen. Om prikkels te geven voor eco-design zou EPR milieuvriendelijke producteigenschappen goedkoper moeten maken voor producenten. Inzamelings- en recyclingdoelen, en PRO-bijdragen zijn soms al gebaseerd op gewicht, maar het lichter maken van producten is slechts één manier waarop ecodesign bij kan dragen aan een circulaire economie. Om met UPV ecodesign te bevorderen zouden de bijdragen aan PVO's een uitdrukking moeten zijn van de maatschappelijke (d.w.z. de som van private en externe) kosten voor de inzameling en behandeling van producten aan het einde van de levensduur.

Een bredere beleidsimplicatie kan zijn om producenten financieel verantwoordelijk te maken voor afgedankte producten die zij niet apart inzamelen. In de praktijk is de financiële verantwoordelijkheid die producenten dragen beperkt tot producten die ze innemen, via bijdragen aan de PVO's, maar gemeenten zouden een aanvullende heffing kunnen opleggen. Deze heffing zou dan de kosten dekken voor de inzameling en verwerking van hun producten die in het restafval terecht komen, alsook voor het opruimen van zwerfafval of gedumpte producten. Een dergelijke uitbreiding van het huidige bereik van UPV zou producenten een prikkel geven om voortdurend inzamelings- en recyclingpercentages te verhogen, ecodesign na te streven, en de financiële last van gemeenten te verlichten.

Goed ontworpen UPV is een nuttig ingrediënt in de beleidsmix op weg naar een circulaire economie, maar geen panacee. UPV heeft inzamelingspercentages verhoogd, recycling gestimuleerd, en financiële verantwoordelijkheid verschoven van gemeenten naar producenten. Er is genoeg ruimte om UPV-instrumenten in te zetten voor ecodesign en hergebruik, hoewel de huidige kennis ook hiaten bevat door het gebrek aan empirisch onderzoek. Toch ambieert de circulaire economie meer dan wat alleen UPV kan leveren. Net zoals de effecten op de afvalstroom ook van ander beleid afhangt, zo zal UPV ook altijd flankerend beleid nodig hebben — in het bijzonder gericht op de productie- en consumptiefasen in de levenscyclus van een product — om de transitie naar een circulaire economie te faciliteren.

1 Introduction

Extended producer responsibility (EPR) is an important element of the policy framework to accelerate the transition to a circular economy. In the Netherlands, it features prominently in the Circular Economy Implementation Programme 2019–2023, which presents concrete actions and projects to accelerate the transition in this five-year period (Ministery of Infrastructure and Water Management, 2019). EPR is also an important pillar of the EU's circular economy and waste management policy, as manifested in the Circular Economy Action Plan and the revision of the Waste Framework Directive (European Commission, 2020; European Parliament and Council of the European Union, 2018a).

1.1 What is extended producer responsibility?

Extended producer responsibility is an environmental policy approach in which a producer's responsibility is extended to the post-consumer stage of a product's lifecycle (OECD, 2001; 2016). This definition is very broad, but in practice EPR entails that producers[1] assume financial — and sometimes also organisational — responsibility for the collection, sorting and treatment of end-of-life products. The transfer of the financial (and organisational) burden associated with the management of end-of-life products from public authorities to producers renders EPR consistent with the *Polluter Pays principle* (OECD, 2016).

EPR is neither a single policy instrument nor a fixed policy instrument mix; it is a flexible instrument mix whose composition varies according to the context in which it is implemented. It typically involves a combination of government policy and producer initiative: the government sets specific requirements that producers should meet, and producers take various steps to fulfil their obligations. EPR schemes have in common that producers are made responsible for the destiny of end-of-life products, but vary considerably in terms of their goals and the instruments put in place to achieve them (Brouillat and Oltra, 2012).

1.2 Objectives of this report

This study focuses on the effectiveness of EPR in improving waste management, increasing recycling and the use of secondary materials, and stimulating eco-design. These objectives have been chosen as they are at the core of the definition of both EPR and the transition towards a circular economy. We investigate how effective alternative forms of EPR are in achieving these goals, and discuss whether and how EPR can be redesigned and implemented in a more effective manner. We further identify the main knowledge gaps on the performance of EPR and discuss how they can be filled by future research.

1.3 Approach

This report is part of a broader project on EPR jointly undertaken by CPB and PBL. In the first part of this work — which is presented in this document — we analyse the design, functioning and effects

[1] The term "producer" is interpreted in a broad sense in the context of EPR. For goods produced and marketed by domestic firms, responsibility lies with the producer. For products imported from abroad and sold in the domestic retail market, responsibility lies with the importer, while for products sold by online platforms abroad it lies with the retailer.

of EPR in theory and policy practice without focusing on specific product groups. Our main interest is in how different elements of EPR design, including its organisation and the instrument mix used to incentivise behaviour changes, affect environmental and economic outcomes. The second part of this work involves three case studies on specific product groups, which aim to provide a more thorough understanding of the operation of EPR in practice and its environmental and economic effectiveness. These product groups are batteries, end-of-life vehicles and unused medicine. The case studies are presented in a separate document (Tijm et al., 2021), which is complementary to this report.

The information used in our analysis is collected from the existing literature and semi-structured interviews with experts from stakeholder organisations. Our review focuses on the scientific literature, on policy reports and other documents on EPR published by national and international sources, as well as on the relevant Dutch and EU legislation. Interviews have been conducted with representatives from the national government, local authorities, EPR organisations and industry associations in the Netherlands.[2]

1.4 Structure of the report

The report is structured as follows. Chapter 2 provides a short policy background of the study, focusing on how EPR has worked in practice in the Netherlands, the European Union and elsewhere. Chapter 3 zooms in on the design of EPR and the main instruments used, and describes the mechanisms underlying their function. Chapter 4 analyses the effects of EPR on waste management, secondary material markets and eco-design. Last, Chapter 5 presents the implications of this study for the design and implementation of EPR in the future.

[2] A full list of interviewed organisations can be found in Tijm et al. (2021).

2 EPR policy in the Netherlands and the European Union

EPR was established around 30 years ago to shift the burden of managing end-of-life products from municipalities and taxpayers to producers, decrease the amount of waste destined for final disposal and promote recycling (OECD, 2016). In the early years of its development, EPR's focus was limited to waste management (Vermeulen and Weterings, 1997). From this early beginning, EPR now features prominently in European and Dutch legislation. Over time, policy attention has been shifting from EPR's role in improving waste management to its potential to support the greening of product design and development.

In the Netherlands, the recently enacted decree on extended producer responsibility (*Besluit regeling voor uitgebreide producentenverantwoordelijkheid*) is the overarching legal instrument regulating EPR. The Decree follows the EPR provisions under the Dutch Environmental Management Act (*Wet milieubeheer*) and implements the minimum operating requirements for EPR schemes set by the revised Waste Framework Directive (EU Directive 2018/851). The Decree was enacted in 2020, but regulation on the implementation of EPR for specific product groups — in the form of end-of-life management decrees — has existed since the 1990s.

Separate end-of-life product management decrees regulate EPR for batteries, end-of-life vehicles, packaging, vehicle tyres and waste electrical and electronic equipment (WEEE) (Rijkswaterstaat, 2020a). With the exception of light-duty vehicle tyres, EPR for these products groups has been introduced to all EU Member States, as a result of EU legislation. The decrees governing EPR for the product groups above are presented in Table A.1 of Appendix A, together with the relevant EU Directives.

In addition, EPR for paper and cardboard and for flat glass is implemented through 'Generally Binding Agreements' (*Algemeen Verbindend Verklaring* or AVV). EPR for these products started on a voluntary basis and was financed by contributions made by a group of producers. To ensure a level playing field, participating producers are offered the opportunity to submit a request for an AVV to Rijkswaterstaat, the executive agency of the Ministry for Infrastructure and Water Management. An AVV is a ministerial order, by which an agreement on the financial contribution for post-consumer collection, sorting and treatment is declared binding for all entities putting a product on the market. An AVV holds for a maximum of five years, after which it can be renewed.

A particularly attractive feature of AVVs is that they prevent producers from free-riding on the EPR scheme; all suppliers must participate in the scheme and pay their share of collection and treatment costs.[3] To demonstrate that broad support for an AVV exists in a sector before its implementation, the request for it must be submitted by an important majority of producers.[4]

[3] In the case of products where EPR is not accompanied by an AVV, it is left in the discretion of producers to decide whether they will participate in a collective scheme and pay the fee determined by the PRO, or they will rely on alternative approaches to discharge their EPR obligations (e.g. through an individual system). This increases the monitoring costs of the system and hampers identifying and imposing fines on producers who do not carry the costs for the collection and treatment of their end-of-life products.
[4] This majority is defined both in terms of the number of firms bringing a product to the market and in terms of the total market share of these firms. The firms submitting the request should account for at least 75% of the number of producers and/or 75% of the market share of this product. In any case, the

Beyond paper and cardboard, and flat (i.e. insulation) glass, separate AVVs apply to cars and light commercial vehicles, electrical and electronic equipment (WEEE), lamps, packaging, portable batteries, and vehicle tyres (Rijkswaterstaat, 2021). AVVs are becoming increasingly popular, with agreements on cars and light commercial vehicles, and WEEE having just been concluded, and an agreement on mattresses expected to take place later in 2021 (CBM, 2020; Rijkswaterstaat, 2021).

EPR will be introduced to new product groups soon. The currently voluntary EPR for mattresses is expected to become generally binding in the course of 2021, and a proposal for the introduction of EPR for textiles is also foreseen for the same year (CBM, 2020; State Secretary for Infrastructure and Water Management, 2020a). EPR will also be implemented for various products falling under the scope of single-use plastics, which are frequently littered. It will be introduced to tobacco filters in 2023, and to balloons, wet wipes and fishing gear in 2025 (State Secretary for Infrastructure and Water Management, 2020b). Other product groups where EPR could be introduced in the following years, including construction materials and sustainable energy technologies, are currently explored (Ministry of Infrastructure and Water Management, 2019).

Product choices for the implementation of EPR in neighbouring countries may act as a source of inspiration for the expansion of the scope of EPR in the Netherlands. EPR is implemented in some EU Member States on waste oil, construction and demolition waste, farm plastics, pharmaceutical and medical waste, chemicals and photo-chemicals, refrigerants, and pesticides and herbicides. Pharmaceutical and medical waste, oils and agricultural (plastic) film are the most common product groups where EPR has been introduced in other EU Member States (Monier et al., 2014).

EPR policy in the Netherlands and at the EU level are closely intertwined. Most of the EPR schemes currently in place have been implemented according to guidance provided by EU Directives. At the same time, experience in the Netherlands influences the design of EU EPR policy, sometimes considerably. For example, the end-of-life vehicles (ELVs) Directive was inspired by the ELV management system introduced in the Netherlands in the 1990s (Fergusson, 2007). The European Single Market and the proliferation of online sales imply that the interdependence between the Dutch and EU EPR policy will likely become even more prominent in the future.

At the EU level, EPR started taking a prominent role in the 2000s. It was mentioned as a "potential policy tool for increasing recycling" in the 2005 Thematic Strategy on the Prevention and Recycling of Waste. EPR was formally introduced in EU legislation in the 2008 Waste Framework Directive (European Parliament and Council of the European Union, 2008). Member states were further encouraged by the European Union to adopt EPR measures in 2011 through the Roadmap to a Resource Efficient Europe.

The revised Waste Framework Directive sets minimum operating requirements for EPR schemes in the European Union. [5] These requirements generally aim to reduce costs and improve performance, while also maintaining a level playing field between market actors and ensuring the smooth functioning of the internal market. They also aim at post-consumer treatment costs being incorporated in product prices, and at incentivising manufacturers to produce more reusable, repairable and recyclable products, and to avoid the use of hazardous substances. The Directive further specifies the types of costs that should be covered by producers' financial contributions in the context of EPR schemes. These comprise the costs of: (i) separately collecting, transporting and treating end-of-life products, net of any revenues from product reuse, sales of secondary materials and unclaimed deposits; (ii) informing consumers about waste prevention and management and recycling measures; and (iii) gathering and reporting data on waste collection and treatment.

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average of these two percentages should be at least equal to 65% (Ministry of Infrastructure and Water Management, 2020).

[5] Waste Framework Directive 2008/98/EC as amended by Directive (EU) 2018/851 (European Parliament and Council of the European Union, 2008, 2018a).

3 EPR design

EPR comprises important design choices on its organisation and the instrument mix used to implement it. We start with discussing the organisational form of EPR in Section 3.1 and its enforcement in Section 3.2. Subsequently, we analyse EPR instruments in more detail (Section 3.3).

3.1 Organisation of EPR

3.1.1 Individual versus collective EPR

Producers usually choose to meet their obligations through collective schemes, rather than on an individual basis. In individual schemes, each producer manages the collection and recycling of products it puts on the market. Individual responsibility schemes have mainly been established when the product market is concentrated and producers can benefit from operating a take-back system. In practice, most producers within a specific sector choose to carry their EPR obligations jointly, through third-party entities called *producer responsibility organisations* (PROs). PROs are financed by producers to manage the collection, sorting and treatment of post-consumer products on their behalf (OECD, 2016). However, collective schemes do not entail that the financial responsibility for the post-consumer stage of products is transferred to PROs: this continues lying with producers.

PROs offer advantages to producers, such as enabling economies of scale and reducing free riding (see also Khetriwal et al., 2009; OECD, 2016). Most PROs in operation collect a fee directly from producers based on a specific fee structure, and they use the revenue to pay for the costs of waste collection, sorting and treatment (OECD, 2001).[6] PROs are financed either on the basis of fixed or differentiated (variable) fees. Fixed fees are typically used by PROs for complex goods, such as electronic equipment and cars, where it is more difficult to link the fee to the product's environmental impact. Differentiated fees have mainly been used by PROs for mono-material products with a short lifetime, such as packaging and graphic paper. Fees in these cases are typically calculated on the basis of product weight, which incentivises the production of lighter products (OECD, 2016).[7] Differentiated tariffs may entail additional costs, as more detailed assessments of end-of-life treatment costs are required, but they can also incentivise producers to design more sustainable products (for more discussion on this topic, see Chapter 4).

In the European Union, operators of collective EPR schemes are obliged to make certain information publicly available. Beyond progress towards the achievement of waste management targets, PROs should report on their ownership and membership, the selection procedure for waste management operators, and the fees paid by producers per unit or tonne of product placed in the market (European Parliament and Council of the European Union, 2018a).

3.2 Enforcement

Enforcement is important for achieving EPR goals. For producers to act on environmental policy, confidence that other parties are playing by the same rules is crucial. In the Netherlands, the Human

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[6] In terms of legal status, PROs can be non-profit organizations, government agencies, quasi-governmental non-profit organisations, or for-profit firms. Multiple PROs can be active in the same market.

[7] Fee differentiation is also called fee modulation or eco-modulation.

Environment and Transport Inspectorate (ILT) enforces EPR for sectors with binding legal requirements. ILT is overseen by the Ministry of Infrastructure and Water Management and can issue fines in case producers fail to meet their obligations. While enforcement is important in practice, it may not always get enough attention in policy design. What is economically optimal might be difficult to enforce in practice.

Creating a level playing field is a necessary condition for EPR to function well. Enforcing a level playing field is essential to earn producers' trust and their cooperation. Particular attention is needed for the enforcement of EPR for products sold through online platforms, especially considering the proliferation of online shopping that emerged in response to policies to contain the spread of COVID-19.

Individual EPR may provide stronger incentives for eco-design, but it may be more difficult to enforce than collective EPR. This is because more entities need to be directly monitored under an individual EPR scheme, which increases the costs of enforcement. Concentrating reporting obligations and monitoring efforts on fewer actors (PROs) per sector or product group reduces enforcement costs and is more practical for enforcement purposes.

3.3 EPR instruments

Within the flexible organisational framework of EPR, governments apply both regulatory command-and-control and market-based instruments. The most common regulatory instrument is a take-back requirement, which is typically operationalised through quantitative collection and/or recycling targets. In terms of market-based instruments, advance disposal/recycling fees and deposit-refund systems are the two most common policy instruments.

The rest of this chapter describes how different EPR instruments function and how they relate to each other. To this end, we present a conceptual framework illustrating how different economic agents interact. We look at take-back requirements (Section 3.3.1), advance disposal/recycling fees (Section 3.3.2), and deposit-refund systems (Section 3.3.3). Section 3.4 describes how these instruments are related to each other. Chapter 4 provides an analysis of the effects of alternative EPR designs.

3.3.1 Take-back requirements

***Take-back requirements* — the most common EPR instrument — oblige producers to collect their products at the end of their life, and to organise their appropriate treatment** (Walls, 2013). Governments usually operationalise such requirements through the obligation to establish drop-off points to return end-of-life products, and through collection and/or recycling targets[8]. Collection targets usually come in the form of a percentage of end-of-life goods or a percentage of the amount put on the market. This can be measured in weight (such as the targets for batteries), volume or units of a product. Producers can undertake various initiatives to meet the take-back requirements, such as introducing an information campaign in order to raise consumer awareness, or implementing a

[8] When the government sets a recycling or recovery target, this often comes with rules as to what 'useful' applications can be used to meet the targets. Useful applications include reuse (highest in the hierarchy), material recycling, and energy recovery from incineration (lowest in the hierarchy) (Ministry of Infrastructure and Water Management, 2017). In the Netherlands, practically all residual waste is incinerated; only about 2% of all waste is landfilled (Rijkswaterstaat, 2020b).

deposit-refund system (see Section 3.3.3; OECD, 2016). Targets for recycling work in a similar manner: they are usually defined as a percentage of collected products, measured in weight or volume.

In practice, producers have little if any incentive to go beyond the given targets and may actually be disincentivised to do so. The share of end-of-life material that is not separately collected does not fall under the (financial) responsibility of producers (Vermeulen et al., 2021). Municipalities still bear the costs of the collection and treatment of end-of-life products ending up in mixed waste, as well as of those being littered or dumped. Because producers do not incur the costs for the non-separately collected streams, they have incentives to collect and treat just as much as they are legally required to by the target. Every additional effort they make is most likely seen as an unnecessary increase of their costs.

A conceptual framework of the function of take-back requirements is presented in Figure 3.1. As in the other figures of this chapter, Figure 3.1 shows physical and monetary flows between economic actors, starting with producers in the upper left corner. Thick blue arrows indicate the main mechanism of physical flows that is intended by the instrument: a circular mechanism in which parts of the product return to the economy either for reuse by consumers or as inputs for producers. Thin blue arrows indicate 'leakage' from the circular economy, caused by improper disposal of end-of-life products and leading to more environmental damage. This happens when end-of-life products are littered, or by mistake end up in the residual waste. All upcoming figures are necessarily simplifications to explain the main idea of the instruments and initiatives; alternative implementations are discussed in the text.

As an example, let us apply the conceptual framework of Figure 3.1 to the case of end-of-life vehicles. In this case, the collection and sorting firm is a dismantling company that receives an end-of-life vehicle. The company removes potentially hazardous parts of the vehicle (e.g. batteries, tyres, fluids), but frequently also some valuable parts (e.g. engine, gearbox, headlights). Some of the latter are then sold and reused in other vehicles.[9] The rest of the end-of-life vehicle is forwarded to a shredder firm (the recycling firm in Figure 3.1) that grinds the vehicle into small pieces. Metals are sold to smelting firms for further processing and eventual use in new products. Other materials may be further processed and recycled in so-called post-shredder treatment facilities (another type of recycling firm in Figure 3.1). Batteries, tyres, fluids and other potentially hazardous parts removed by dismantling firms are sent to specialised recycling companies (a third type of recycling firm) for further treatment. Improper disposal — or in this case illegal dumping — is less common for end-of-life vehicles than for other products, but it still occurs frequently in several countries.

All figures in this chapter assume that producers organise EPR in a collective manner by delegating collection, sorting and recycling activities to a PRO. PROs are financed by the producers through dedicated fees, which enable them to manage — and pay for the costs of — the collection, sorting and recycling of end-of-life products. These payments are shown in Figure 3.1 with the green arrows pointing from PROs to collection and sorting firms, recycling firms, and incinerating firms. Producers may partially or fully pass on the costs for collection, sorting and recycling to consumers, who will buy less of the products having high waste management costs. Fees can therefore serve as the incentive mechanism of the policy if they take into account material characteristics of a product or, at the least, product weight (also see chapter 4 on the effect of differentiating tariffs).

In practice most EPR schemes apply uniform tariffs for a given product type, rather than differentiated ones. France is considered a successful exception in applying differentiated PRO fees

[9] In some cases, the dismantling firm sells parts directly to consumers. This is not indicated in Figure 3.1, as it less common than sales through retailers.

for packaging, electronics and graphic paper (OECD, 2016).[10] For example, packaging producers receive a fee reduction for:

+ using more than 50% of recycled content (fee reduction of 10%);
+ supplying specific sorting instructions to consumers (fee reduction of 8%);
+ reducing the weight or volume without changing the packaging material or function (fee reduction of 8% for at least a 2% reduction in weight or volume).

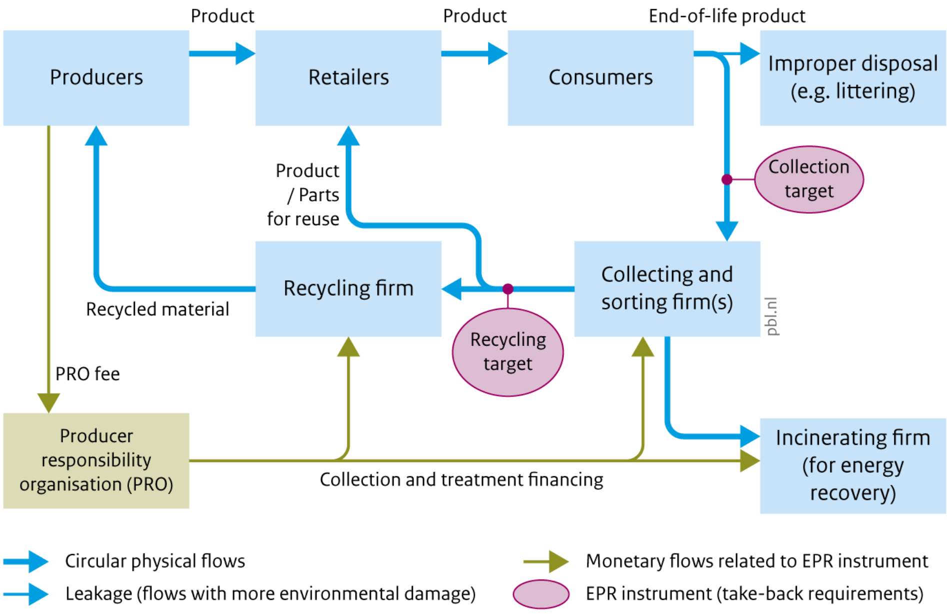
Non-recyclable packaging materials lead to a penalty on the PRO fee. For example,

- ceramic packaging leads to a fee increase of 100%.

Similarly, packaging in Portugal that disrupts the recycling process receives a penalty of a 10% increase in the PRO fee (Sociedade Ponto Verde, 2020).

Figure 3.1

Take-back requirements for end-of-life products



Source: PBL and CPB

Differentiated fees provide financial compensation for the extra effort that a producer does. It can be costly to change an input in the production process or the design of a product. The transition to more circular production requires resources and investments from producers that they cannot always earn back. PRO tariff differentiation provides a financial incentive required for producers to make these changes. In our case study on batteries we observe that the responsible PRO charges higher fees for producers that sell lithium batteries as those are more difficult to recycle. The fee thus discourages producers from producing this type of battery and stimulates them to find other materials that can be used in batteries while being easier to recycle. Since fees are passed on to consumers, they also have an incentive to look for easier to recycle batteries as they would be less impacted in their price by the fee (Tijm et al., 2021). The incentive that a fee gives, depends on the size of the fee: a small fee could have no or very little effect. The optimal fee would cover the social costs associated with the waste management of the product — including external environmental costs — not just the private costs of waste management.

While current policy is mainly focused on collection and recycling, targets can also be introduced to promote reuse. Reuse is often considered more environmentally friendly than recycling; and

[10] Differentiated tariffs have been introduced by the packaging industry in the Netherlands as of the beginning of 2019 (Ministry of Infrastructure and Water Management, 2019).

using targets could increase the share of material that is reused. Before introducing such targets, one should consider whether the social benefits of increased reuse, outweigh the additional costs. Currently no reuse targets exist in the Netherlands, but there is increased policy interest in introducing them. It can be challenging to find a good base measure, however, as products reaching end-of-life status cannot always be fully reused. If the amount of product put on the market is used as a base, there is a risk of low rates in growing markets, or rates that lie above 100% in shrinking markets, as we saw happening in the case of batteries (Tijm et al. 2021). There is also a question on the definition of reuse, there is full product reuse but also parts reuse. In Chapter 4, we discuss in more detail reuse and its effects.

A step further is reduce, which is the practice of using less products. Some producers implement this in practice via product-as-a-service schemes**.** Customers then pay a monthly fee to use a product which the company takes back and can rent out so someone else if the first customer doesn't need it anymore. This can lead to more efficient material use, lower amounts of waste and a longer lifetime for products, by taking away existing incentives for producers to continuously sell new products. Moreover, it can lead to a more homogenous waste stream as the producer receives all products back, allowing for more specialised disassembly and recycling. In practice, these kinds of schemes are usually implemented for products with a relatively long lifetime such as cars, bicycles, lighting or electronic devices (Poolen et al., 2020).

3.3.2 Advance disposal and recycling fees

An *advance disposal or recycling fee* is an instrument requiring producers and/or consumers to pay upfront the end-of-life treatment costs of a product. Consumers pay an advance disposal or recycling fee to producers when buying the product, and the revenues are used to finance the collection and treatment of end-of-life products (see Figure 3.2). An important reason to charge the fee in advance, rather than at the moment of disposal, is to reduce the risk of illegal dumping caused by the temptation of consumers to avoid paying the fee (Shinkuma, 2007).[11]

Figure 3.2 presents the functioning of an advance fee. The fee may be used to finance the collection and disposal of the end-of-life product (*advance disposal fee*) or its recycling (*advance recycling fee*). Note that a basic advance disposal/recycling fee in itself does not sustain a circular product lifecycle, as it does not give incentives for consumers to properly dispose of the end-of-life products. The problem of improper disposal is therefore not addressed by advance fees alone. Yet, the advance disposal fee is often used in a circular context, to support the fulfilment of take-back requirements, as we observed in the case studies of batteries and end-of-life vehicles (Tijm et al., 2021).[12] It works best when complementing other policy measures that do provide incentives — for either producers or consumers — but also need financing.

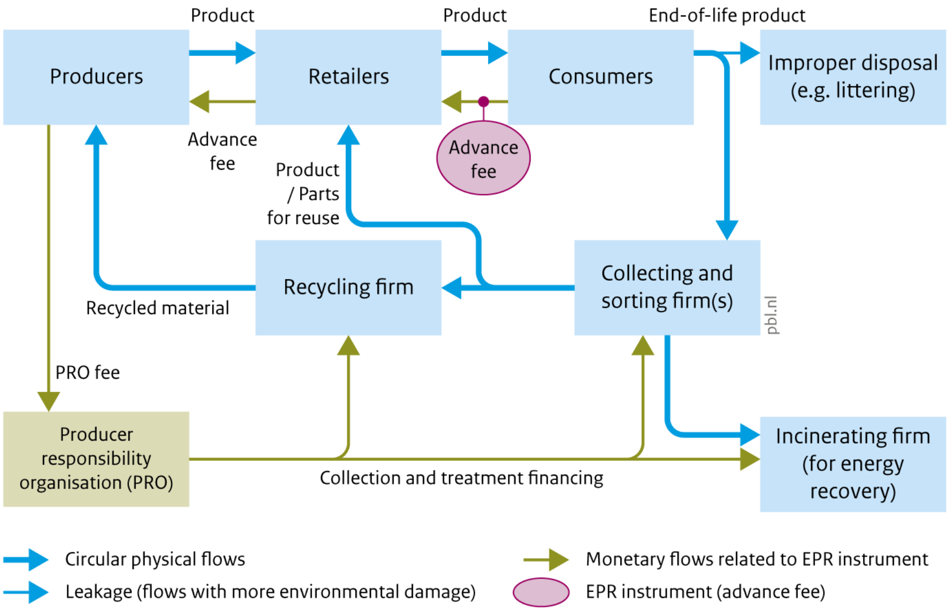
A differentiated advance fee may promote eco-design. If the fee is the same for all products of a specific sector, say cars, then there is no incentive for car manufacturers to change the design of their cars as there is nothing for them to be gained. A differentiated advance fee that is lower for cars that either contain more recycled materials or that are easier to disassemble and recycle, incentivises producers to take these design aspects into account. Doing so would lower the fee that consumers have to pay at the time of purchase, making the car relatively cheaper compared to other cars that have not seen any design changes. This differentiated fee does not change the lack of effect on the choice of manner of disposal. In Box 3.1, we touch upon promising ongoing research by the Dutch

[11] For durable goods with long lifetimes (e.g. motor vehicles and large household appliances), an advance disposal fee ensures the proper collection and treatment of products from producers who have ceased operations by the time the product reaches the end of its life.
[12] An output tax levied by the government is sometimes also called an advance disposal fee (Walls, 2013), but this does not fall under our definition of it.

Ministry of Infrastructure and Water Management, where finely differentiated fees are used to finance a reserve fund for the collection and treatment of end-of-life products.

Figure 3.2

Financing collection and treatment through an advance fee



Source: PBL and CPB

Box 3.1. A 'pension fund' for goods

The Dutch Ministry of Infrastructure and Water Management is working on an idea to encourage producers to develop a reserve fund which would cover the costs of collecting and treating their products at the end of their life. When a product is disposed of, the payment that has been made to the fund would be used to finance the post-consumer management of that product. The fund would be financed by advance fees that consumers pay at the time of purchase. Fees could be highly differentiated and reflect as accurately as possible the social – i.e. private and external – costs that producers need to incur for the collection and treatment of products. For durable goods, producers would have to take into account alternative scenarios for the evolution of these costs in the future. In general, producers who sell more durable, repairable or reusable goods would have to keep aside lower reserve funds than producers whose products reach the end of their life at an earlier stage.

This is an idea worth further exploration. While it does not solve all issues identified in our analysis, such as the lack of incentives to consumers to return their products, it could imply an improvement of the functioning of EPR in various ways. One of the most promising aspects is the mechanism of a product-specific reserve, which is based on the waste management costs of a product. This allows for fine tariff differentiation, that is likely to promote eco-design.

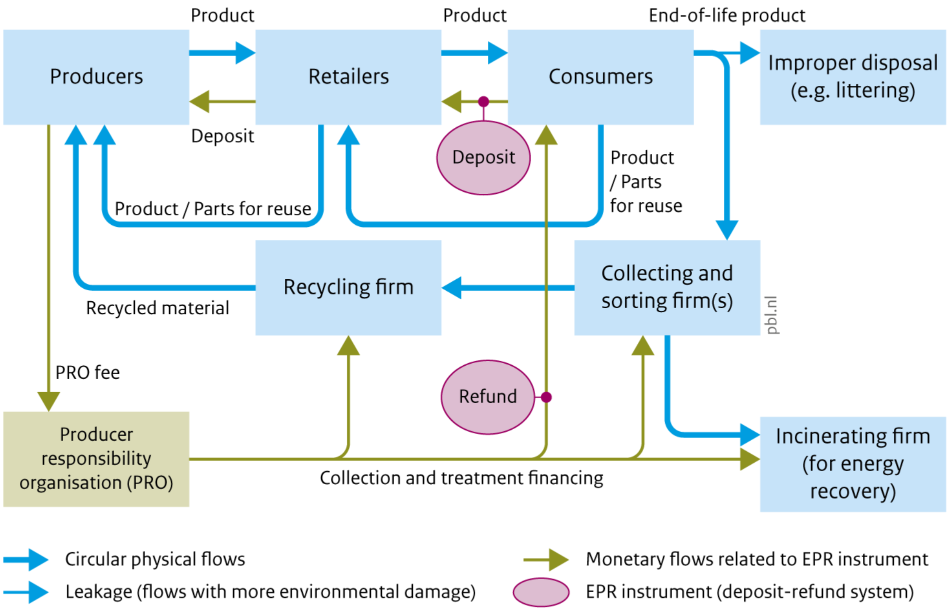
3.3.3 Deposit-refund systems

Under a deposit-refund system, consumers pay a deposit to producers when purchasing a product and receive an equal[13] amount as a refund upon return of the product (see Figure 3.3). The refund provides consumers with an incentive to sort and return the product, as well as to prevent any further damage to it at the end of its lifetime (Fullerton and Kinnaman, 1995). Deposit-refund systems are mainly aimed at reducing improper disposal, after which the returned products can be either reused or recycled.

Figure 3.3 explains the main mechanism of a deposit-refund system. Having bought and used a product, consumers can choose to return it at the end of its life to the retailer in order to collect the refund. Alternative options, such as throwing the product in the waste bin or even littering, are financially less attractive for consumers (and more environmentally damaging) compared to participating in the collection scheme. The refund can be seen as a compensation for the extra effort that it takes a consumer to properly return their product. It even works for products that have already been littered: in principle, anyone could retrieve the refund for already littered products. In the ideal situation, the deposit-refund scheme leads to a fully circular system of producers reusing all products, providing refunds through the distributors to consumers.

Figure 3.3

A deposit-refund system for end-of-life products returning to producers via retailers



Source: PBL and CPB

The infrastructure of the deposit-refund system, such as drop-off points, is financed by unclaimed refunds and/or by producer contributions via the PRO (Bergsma et al., 2017). Unclaimed refunds (not shown in Figure 3.3) arise not only when bottles are not returned to the distributor, but also when refund tickets are not redeemed, e.g. when consumers forget their refund ticket at the cash desk. PRO contributions by producers to finance the deposit-refund infrastructure are, at least partially, derived from cost savings from reusing products or parts of products, or from profits from selling collected materials.[14]

[13] In practice, the refund may be smaller than the deposit. Such an EPR policy is equivalent to the combination of a deposit-refund system and an advance fee under our terminology.
[14] Some authors consider alternative definitions of the deposit-refund scheme that do not fall under the scope of EPR. See Appendix B for a further discussion.

3.4 Conclusion

Each of the previously discussed instruments provides a different incentive for producers and consumers and/or helps to finance waste management. Instruments do not have to be used in isolation but can also be combined. In practice, we see that targets are mainly set by governments and that PRO's use fees to be able to finance the required collection and recycling measures. In Chapter 4, we further discuss what can be expected from combining multiple EPR instruments.

Take-back requirements ensure that producers arrange for the collection and proper treatment of end-of-life products[15], but do not provide direct financial incentives for consumers to separate their waste and return the relevant products to dedicated collection points. Admittedly, producer initiatives such as installing drop-off points for lamps or batteries do encourage consumers to separate and return those products. Yet, if financial incentives are considered necessary to influence consumer behaviour, governments could combine a take-back requirement with an additional incentive scheme for consumers, such as a deposit-refund system.

From the perspective of producers, advance fees provide a direct way to finance the obligations specified by take-back requirements.[16] In the case of batteries, it is the relevant PRO that imposes the advance disposal fee. Another example of this are end-of-life vehicles in the Netherlands, where an advance recycling fee paid per vehicle is used to ensure that recycling and recovery targets set at the EU level are met. In deposit-refund systems, the deposit could be higher than the refund, effectively adding an additional advance fee to the product price. But also refunds that are not collected help producers with financing EPR. In the absence of explicit advance fees, producers may decide to pass on PRO fees to consumers as well, which may lead to a similar price raise.

From the perspective of consumers, advance disposal/recycling fees and deposit-refund systems increase the price to be paid at the time of purchase of the product. It is possible to design the fees in such a way that the highest fee is charged for the most polluting good, therefore discouraging consumers from buying those goods (in line with the polluter pays principle). For deposit-refund systems to be effective, the level of the deposit needs to create sufficient financial incentive to return the product. In either case, the additional payment makes consumers aware of the costs of collection and treatment of end-of-life products.

[15] When failing to fulfil take-back requirements, producers may face financial penalties. In the Netherlands, take-back requirements are enforced by the Human Environment and Transport Inspectorate (*Inspectie Leefomgeving en Transport*, ILT), see e.g. State Secretary of Infrastructure and Water Management (2020).

[16] Note that an advance disposal fee by itself will probably have no impact on waste separation and return of end-of-life products by consumers.

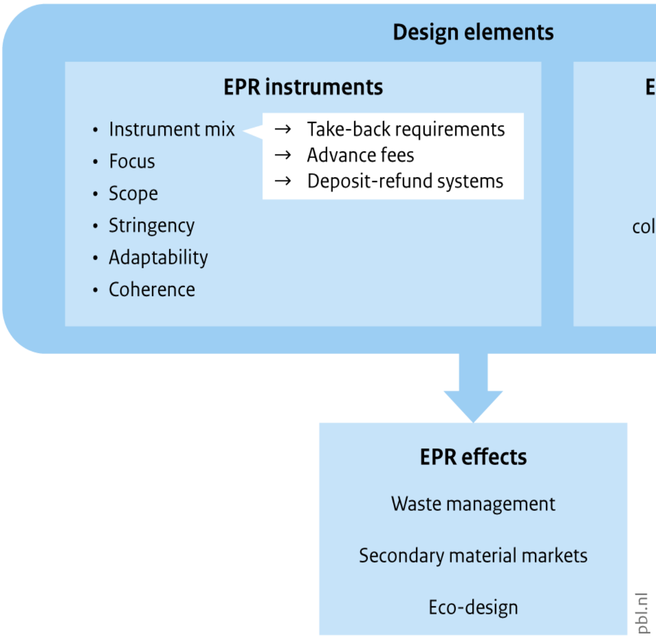
4 Effects of EPR

This chapter synthesises the academic and policy literature to provide insight into the environmental and economic effects of EPR. In terms of environmental effects, we narrow down the scope of our analysis to damages from littering, and to the redirection of waste flows from incineration and energy recovery towards higher R-strategies[17], namely reuse and recycling. We also focus on environmental damages from material extraction and processing and product manufacturing, insofar as they can be addressed by a more circular product design. In terms of economic effects, we focus on the costs of waste collection and treatment, and those of producing secondary materials.

The environmental and economic effectiveness of EPR hinges upon the incentives it provides to change consumer and producer behaviour. Such incentives are largely determined by the EPR design choices discussed in the previous chapter, namely on instruments used and how EPR is organised. The design elements of EPR affecting the outcomes of interest are sketched in Figure 4.1.

Figure 4.1

The design elements of *extended producer responsibility* (EPR) that determine its effects



Source: PBL and CPB

The effectiveness of EPR is influenced by the type of instrument mix used, as well as by its focus, scope, stringency, adaptability and coherence with other policies (van der Werf et al., 2021)**.**

- *Focus* is to be understood as the (intermediate) operational objective that the instrument mix is aiming to achieve. For example, such an objective for EPR could be to prevent littering, promote reuse, or stimulate recycling and the use of secondary materials.

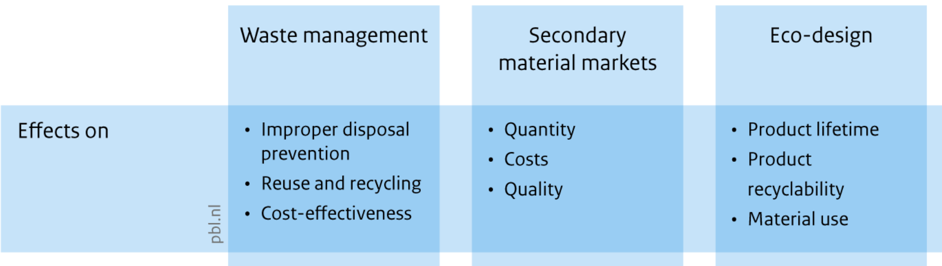
[17] See Campbell-Johnston (2020) for an explanation of the different R-strategies.

- *Scope* denotes the extent to which the instrument mix makes producers responsible for financing the totality of (net) costs of collection and treatment for all units of a product reaching an end-of-life stage. Various aspects of scope are key for the effectiveness of EPR, such as the extent to which: (i) producers cover the costs of the clean-up of littered products, and of the treatment of end-of-life products mixed with other waste streams; (ii) end-of-life products from both household and business users are covered by the scheme; and (iii) the external costs of waste management are covered by producers, including the damage costs associated with emissions of greenhouse gases from waste treatment, and with air, water and soil pollution from (partially) untreated waste.
- *Stringency* demonstrates the strictness of the instrument mix, such as the level of a fee or the ambition of a target. More stringent policies employ higher fees or targets.
- *Adaptability* indicates the ease with which the scope or stringency of a policy instrument mix can change. Adaptable policies can retain their effectiveness under changing circumstances. An adaptable EPR policy may for example contain built-in elements facilitating the periodic review and adjustment of instruments in response to future conditions and needs. In some other cases, the stringency of EPR instruments is designed to change — typically increase — over time, often in regular time intervals (e.g. every two years).
- *Coherence* shows the extent to which the EPR instrument mix is in alignment with other waste and environmental policies, as well as with trade and other relevant policies. A coherent instrument mix is complementary to other policies and avoids overlaps to the extent possible.

This chapter is structured around the effects of EPR on waste management (Section 4.1), secondary material markets (Section 4.2), and product design (Section 4.3). Figure 4.2 presents the outcomes analysed within each of these three categories of effects. Regarding waste management, we examine the effects of EPR on the separate collection of waste, the reuse and recycling of discarded products, and the costs incurred for these activities. We then analyse the effects of EPR on the quantity, costs and quality of the secondary material supplied to the market. Note that recycling is on the intersection of waste management and secondary material markets, and therefore features in both Section 4.1 and 4.2. The analysis of the effects of EPR on product design focuses on its potential to extend product lifetime, increase recyclability, and reduce the amount of material used in production. Furthermore, interactions between EPR and municipal waste management policies are considered in Section 4.1, while trade-offs between eco-design attributes are discussed in Section 4.3. Possible effects of EPR on competition between firms are analysed in Section 4.4.

Figure 4.2

Effects of EPR on waste management, secondary material markets and eco-design



Source: PBL and CPB

4.1 Effects of EPR on waste management

This section focuses on the effects of EPR at the post-consumer stage of a product. We first analyse the effects of EPR on preventing improper waste disposal, including littering and dumping, as well as the mixing of recyclable streams with non-recyclable ones (Section 4.1.1). Our attention then shifts to the effectiveness of EPR in promoting reuse and recycling (Section 4.1.2). Less is known about the cost-effectiveness of EPR in achieving waste management goals (Section 4.1.3). Instead of studying a particular effect of EPR, the last subsection discusses how EPR interacts with existing municipal policies that affect the incentives for households to separate waste (Section 4.1.4).

A qualitative overview of the effects of the three EPR instruments — take-back requirements, advance fees and deposit-refund systems — on waste management, is provided in Table 4.1. We evaluate the effectiveness of EPR instruments compared to a counterfactual without EPR, where responsibility for the management of end-of-life products rests with local authorities. As discussed in Chapter 3, EPR in practice entails a combination of instruments and initiatives. Following the literature, we discuss here their effectiveness when implemented in isolation. The overview presented in this table, as well as similar overviews in the following sections, are limited by the form of EPR instruments studied in the literature; in practice, the instruments may be implemented in a different way. The table also abstracts from the influence of other instrument characteristics, such as stringency and adaptability, as well as of the context in which the instruments are implemented (e.g. product group, market structure), in order to focus on the impact that the type of instrument has on waste management. These other aspects are further analysed in the case studies of certain product groups (Tijm et al., 2021).

Table 4.1

Effectiveness of EPR instruments in improving waste management

	Take-back requirements	Advance fees	Deposit-refund systems
Improving waste management	**+ / ?**	**+ / ?**	**+ / ?**
- Improper disposal prevention	+	0 / +	+
- Reuse and recycling	+	+	+
- Cost-effectiveness	?	?	?

Note: + indicates an improvement, 0 indicates negligible effect, and ? uncertain or unknown.

4.1.1 Improper disposal prevention

A key determinant of EPR's environmental effectiveness is the extent to which it stimulates the separate collection of end-of-life products. Increasing collection rates is important from an environmental point of view, as higher rates imply that fewer end-of-life products are illegally dumped, littered, or simply mixed with other waste, eventually ending up in incinerator plants or landfills. Moreover, separately collecting waste streams containing hazardous materials, such as batteries, tyres and oils, prevents risks posed by a possible leakage of these materials to the environment. Separate collection also facilitates directing end-of-life product streams to reuse or recycling.

EPR increases the share of end-of-life products being separately collected and sent for further treatment (OECD, 2016; Walls, 2006). An increase in the collection rate is achieved for multiple reasons. First, producers (or PROs) set up return points where consumers can hand in their end-of-life products. Smaller items, such as packaging, portable batteries and lamps, can be returned to special containers installed in retail shops, offices, or public space. Larger items are usually received by specialised firms: for example, vehicles are returned to dismantling firms (Tijm et al., 2021) and household appliances to electronics stores. In addition, producers inform consumers about their EPR activities and encourage consumers to return their products, often through public awareness campaigns.[18]

All three instruments examined in this chapter are likely to increase collection rates, but take-back requirements and deposit-refund systems provide more direct incentives for this purpose. Collection targets set in the context of take-back requirements do so by obliging producers to collect a certain share of their end-of-life products. In practice, it is often difficult to determine the total amount of products that have been discarded by consumers, so the amount of products reaching the end of their life is approximated by the amount *put on market* in the past one or more years. While this may be a reasonable approximation for products with a very short lifetime, such as packaging, it can be very inaccurate for products with longer lifetimes, such as portable batteries (Tijm et al., 2021). Coming up with accurate estimates of the amount of products reaching the end of their life is key for the design of effective collection targets; inaccurate estimates provide a misleading picture of environmental risks and EPR policy outcomes. Deposit-refund systems incentivise consumers to bring their products back by rewarding them with a refund. This way they can ensure that more end-of-life products are collected. Advance fees provide the necessary finances for collection activities: more funds directed to collection should in principle lead to higher collection rates, but there are no reward (for consumers) or penalty (for producers) mechanisms to promote this outcome.

4.1.2 Reuse and recycling

The environmental effectiveness of EPR is also determined by its potential to promote reuse and recycling. Reuse of products or components is at the core of circular economy objectives, as it prevents environmental damages occurring both at the production and at the post-consumer phase of a product's lifecycle.[19] While less environmentally beneficial than reuse, recycling reduces the environmental burden from the extraction and processing of natural resources, and prevents the depletion of scarce materials. In addition, recycling causes less emissions of greenhouse gases, and air, water and soil pollution than conventional waste treatment options, such as incineration. These environmental costs are often not included in waste management prices.

Reuse is increasingly acknowledged as a focal point for new EPR design (RReuse, 2013). EPR policy has historically focused on the collection and recycling of end-of-life products. However, reuse of discarded products or their parts ranks higher in the hierarchy of circular economy strategies, with more potential for environmental improvements. EPR schemes can promote reuse in a number of ways, for example by: (i) handling end-of-life products in ways to preserve the potential for reuse, prior to any further treatment; (ii) ensuring that reusable products and parts find their way back to the market; (iii) providing guidance on product reuse and repair to consumers, and reuse and repair organisations; and (iv) helping raise public awareness about the importance of reuse (see also RREUSE 2013).

Reuse can be stimulated through dedicated targets. Take-back requirements may come with specific targets on the share of products or components — in terms of e.g. product weight — that should be reused. Such targets would directly incentivise producers to promote the reuse of their products. Separate reuse targets have rarely been implemented in EPR schemes: in one of the few

[18] An example of such an information campaign in the Netherlands is 'Plastic Heroes' (2009-2013), which aimed to incentivise plastic waste separation.
[19] The impact of reuse at the consumption phase of a product's lifecycle varies by product group. For motor vehicles, electric and electronic appliances, and other devices where energy (or water) consumption is a key source of environmental damages, the reuse of an older product generally implies higher damages at the consumption stage than the adoption and use of a new one.

examples, the French EPR system for furniture stipulated a 50% increase of product reuse compared to a baseline situation by 2017 (RReuse, 2013). Reuse targets can be especially useful for products whose value when discarded is very low — and therefore where second-hand markets are inexistent or underdeveloped — but whose condition is frequently good enough to allow reuse. In addition to furniture, durable types of packaging (e.g. glass, wood, and certain plastics) may also fall under this description.

Deposit-refund schemes can promote reuse, while advance fees can help finance activities encouraging it. Deposit-refund systems incentivise consumers to return end-of-life products in good condition. This promotes reuse, as the higher quality of returned products makes them more suitable for use by other consumers. Advance fees can be used to finance activities promoting reuse, including the identification of and separation of reusable products in waste streams, the provision of instructions on product repair, and information campaigns on the benefits of reuse.

EPR can promote recycling through targeted instruments, such as recycling targets and advance recycling fees. Recycling targets oblige producers to recycle a certain share of the amount of collected end-of-life products. Increasing the stringency of recycling targets over time — as implemented in the case of end-of-life vehicles — provides strong incentives for producers to invest in new recycling techniques to increase the amount of recycled material (see also Tijm et al., 2021). Advance recycling fees enable raising the necessary funds to finance recycling activities and investments in innovative recycling processes.

To allow more flexibility to producers, combined reuse-recycling targets can be set. Such targets can take various forms, implying different incentives for reuse. For example, the combined reuse-recycling targets long applying to end-of-life vehicles stipulate that a certain percentage of vehicle weight (currently 85%) should be reused or recycled (Tijm et al., 2021). Likewise, the new 'circular targets' applying to packaging as of 2021 in the Netherlands determine for different material types (glass, paper and cardboard, plastic, wood and metal) the minimum share of packaging material to be reused or recycled (State Secretary for Infrastructure and Water Management, 2020c). Despite rewarding reuse, such combined targets do not reveal a clear preference of reuse over recycling, and thus do not by themselves stimulate a shift from recycling towards higher R-strategies. Incorporating incentives to prioritise reuse over recycling in reuse-recycling targets would lead to EPR schemes that better promote reuse.

Reuse and recycling targets can be designed in a way that increases environmental benefits and does not discourage higher R-strategies. Several elements of the design of a target matter for its environmental effectiveness. First, while targets expressed as percentages of product weight may work well for mono-material goods, additional incentives may be needed to stimulate the reuse of products and parts or the recycling of materials with the highest environmental impact in more complex product groups, such as consumer electronics. Second, targets would ideally be expressed as percentages of products reaching their end-of-life. By contrast, the standard practice in EPR schemes is that targets are set against the amount of products being collected, which is in most cases only a fraction of those reaching the end of their life. This implies that targets do not provide further incentives to prevent littering, collect littered products, or properly separate them from other waste streams. Attention should also be paid to possible side-effects of targets on higher R-strategies. To this end, targets could be accompanied with incentive mechanisms promoting the adoption of higher R-strategies.

4.1.3 Cost-effectiveness

EPR entails a transfer of waste collection and treatment costs from municipalities and taxpayers to producers and consumers. This is a direct consequence of shifting the financial responsibility for waste management from municipalities to producers. Beyond leading to a fairer distribution of the burden of waste collection and treatment and creating the necessary incentives for producers to take the associated costs into account in their decisions, this transfer may also lead to more cost-effective waste management. Cost-effectiveness might increase, for example due to economies of scale in the collection, sorting and recycling of waste, especially when these activities are collectively carried out by producers.

Producers only pay for the costs of management of the share of end-of-life products that is collected separately. For example, the management of batteries that end up in mixed waste is not financed by battery producers. This means that municipalities still need to finance these streams. If EPR would make producers responsible for the costs of managing end-of-life products mixed with other waste, littered or dumped, they would be incentivised to increase collection and recycling rates. They would do so insofar as the management of separately collected products is cheaper than the management of mixed waste. If the incineration costs of mixed waste were to include external costs, producers (or PROs) would likely collect higher amounts of waste than those prescribed by take-back targets.[20]

The overall effect of EPR on waste management costs is unclear. This is one of the most important knowledge gaps in the EPR literature. Both from a theoretical and from an empirical point of view, this topic has received relatively little attention. Amongst other factors, empirical research in this area is hampered by the unavailability of detailed data on the EPR costs incurred by producers and PROs. These data are considered commercially sensitive and PROs are generally unwilling to share them with external parties.

Policies targeting multiple behavioural responses are likely to be more cost-effective. Dubois (2012) stresses that static collection and recycling targets used in the context of EPR do not incentivise additional effort by producers once the target is reached. To increase its efficiency, a target can be combined with a fee charged to producers for the share of end-of-life products that have not been collected. In one of the few empirical studies in this field, Palmer et al. (1997) investigate the cost-effectiveness of three instruments in US markets for recyclable waste: a deposit-refund system, an advance disposal fee, and a benchmark subsidy to households for recycled material. They find that for a given target of waste reduction, the deposit-refund system is the least-cost policy, followed by the advance disposal fee.

Collective EPR systems are more cost-effective than individual systems. The reason for this is economies of scale in waste management. Especially when the market for waste management is not perfectly competitive, PROs may be preferable to individual EPR systems (Fleckinger and Glachant, 2010). Also, collective EPR systems are less costly to implement for the producer, and easier to monitor for the government (Walls, 2006). However, as we will see in more detail in Section 4.3, collective schemes dilute incentives for waste prevention and eco-design.

4.1.4 Interaction between EPR and municipal waste policies

Although EPR makes producers responsible for the collection of end-of-life products, direct incentives for consumers to return or appropriately dispose of these products also follow from municipal policies. In many cases, municipalities act as operators of the separate collection systems for certain end-of-life products, such as packaging. In these cases, PROs pay municipalities to collect and sometimes also treat these waste streams. In the Netherlands, for example, paper and cardboard are collected by, or on behalf of, municipalities, and the PRO compensates them for the collection costs they incur (Verrips and van der Plas, 2019). In such cases, municipalities influence household disposal behaviour through various levers, including the frequency of collection, and the availability and accessibility of containers.

[20] This assumes that treatment costs are set at the socially optimal level, which may be difficult to do in practice.

Choosing between separation of waste streams at source and post-separation is another channel through which municipal policies directly affect the effectiveness of EPR. This choice is only relevant for some of the end-of-life product streams where EPR is implemented, most notably packaging made out of metal or plastic. The two methods of separation imply different costs for municipalities and consumers (in terms of time required to sort and bring to the appropriate container) and have different effects on the quantity and quality of separated streams. The choice between the two methods therefore matters both for the environmental effectiveness, that is to say, how much material will be collected and recycled, and for the cost-effectiveness of EPR. Dijkgraaf and Gradus (2020) argue that post-separation of plastic packaging and aluminium tins is better than sorting by households from a financial and environmental point of view, because a larger amount of material is separated from residual waste[21]. On the other hand, mixed kerbside collection leads to lower-quality recycling streams, due to commingling and breakage of products (Kaffine and O'Reilly, 2015).

Local waste policies also influence the effectiveness of EPR by providing indirect incentives for households to sort and appropriately dispose of their waste. Around half of Dutch municipalities do not rely on fixed charges — or charges varying only by household size — to finance waste collection (Wal, 2019). By contrast, they charge their citizens for the actual amount or weight of residual waste they produce through unit-based pricing. In 2017, such systems were deployed in slightly less than half (46%) of Dutch municipalities. Unit-based pricing (also known as 'pay as you throw' (PAYT)) for residual waste acts as an indirect incentive for households to separate waste, and therefore save on waste disposal costs.

EPR for products disposed of frequently by households (e.g. paper and packaging) may be more effective in municipalities with unit-based pricing for residual waste. Table 4.2 shows that the amount of separately collected paper, glass, plastic, metals and drinks packaging is higher in municipalities with a unit-based pricing system.[22] Households in municipalities with unit-based pricing dispose of 12% less total waste per capita,[23] and separate 12% more paper and cardboard, and 39% more plastic, metal and drinks packaging than households in municipalities using other types of waste collection charges.[24] These differences are significant, but do not necessarily imply a causal relationship between the policy (unit-based pricing) and collected amounts of waste per capita. For example, inhabitants of municipalities with unit-based pricing might be intrinsically more motivated to separate waste, due to e.g. being more environmentally aware, or having better accessibility to recycling containers. Indeed, these might be some of the reasons why these municipalities have opted for unit-based pricing in the first place. Empirical analysis controlling for other factors influencing waste separation per capita is needed to confirm or reject a causal relationship between unit-based pricing and recycling per capita.

[21] This study ignores littering, if sorting by households has an effect on littering then their reasoning may not hold.
[22] This pattern is not observed in streams that are not collected by municipalities, but should instead be brought to special collection points (tyres, flat glass and e-waste). These streams are less affected by unit-based pricing of residual waste also because households dispose of them much less frequently and separating them from other streams costs households less time.
[23] Another positive side-effect of unit-based pricing is that it discourages the (illegal) usage of household waste containers by businesses (van der Wal, 2019).
[24] This provides, however, no insight into the quality of the material thrown in recycling containers. In their effort to reduce the amount of residual waste they produce, households may be inclined to throw non-recyclable items – especially items about whose recyclability they are unsure – to recycling containers or dump them in other places. Contamination of recycling containers generally deteriorates the quality of collected material and increases recycling costs, therefore reducing the potential of producing high-quality secondary material. Unfortunately, there is no data available at the municipal level on illegal dumping or the contamination of separate waste streams, so this link cannot be further investigated.

Table 4.2

Waste by type, kg per capita, for Dutch municipalities with and without unit-based pricing (UBP) in 2017

	Total household waste	Paper and cardboard	Glass packaging	Plastic, metal and drinks packaging
Municipalities with unit-based pricing	497.84	64.01	23.91	27.52
Absolute difference between municipalities with and without unit-based pricing	-60.46*	7.64*	1.19	10.70*
Percentage difference between municipalities with and without unit-based pricing	-12.0%	11.9%	5.0%	38.9%

Source: Calculations by CPB and PBL on data provided by CBS and Rijkswaterstaat.

Note: The number of observations used for each type of waste is as follows: (i) Total household waste: 312, out of which 145 municipalities use UBP; (ii) Paper and cardboard: 320 (151 with UBP); Glass packaging: 320 (152 with UBP); and (iv) Plastic, metal and drinks packaging: 276 (136 with UBP). The asterisk (*) denotes that the difference is statistically significant at the 99% confidence level.

4.2 Effects of EPR on secondary material markets

The use of secondary (i.e. recycled) materials in production processes has important environmental benefits compared to the use of virgin materials. In the Netherlands, as well as in several other European countries, the highest environmental damage in a product's lifecycle occurs in the phase where natural resources are processed to usable materials and intermediate goods (Brink et al., 2020; Vollebergh et al., 2017). The use of secondary material also reduces environmental damages from raw material extraction, which can be considerable for fossil fuels and some other resources. An example of this is the virgin lithium used in most rechargeable batteries. The extraction of lithium from evaporated brines requires vast amounts of water, which is particularly problematic in the desert areas where this often occurs (Tijm et al., 2021) .

Therefore, we now turn to the effects of EPR on secondary material markets, focusing on its impact on the supplied quantity (Section 4.2.1), costs (Section 4.2.2) and quality (Section 4.2.3) of recycled material. Taken together, these three attributes largely determine the potential of secondary materials to replace raw ones in product manufacturing. The three attributes are interrelated: for example, producing secondary material of higher quality often requires an increase in recycling costs, while an increase in secondary material output usually entails a reduction in per unit costs.

While EPR stimulates the use of recycled materials by increasing the quantity supplied and reducing production costs, its impact on secondary material quality is rather uncertain. Table 4.3 provides a qualitative overview of the effects of the three EPR instruments on the three aspects of secondary material explored in this section. The table shows that while the overall effect of EPR on secondary material markets is likely to be positive, the strengths and weaknesses of different instruments lie with different characteristics of secondary material output. Care needs to be taken at the EPR design stage, so that material downcycling is avoided as far as possible.

Table 4.3

Effectiveness of EPR instruments in improving secondary material markets

	Take-back requirements	Advance fees	Deposit-refund systems
Improving secondary material markets	**+**	**+ / o**	**+**
- Quantity	+	o / +	+ / –[25]
- Costs	+	+	+
- Quality	o / –[26]	o	+

Note: + indicates an improvement, – indicates a deterioration, o indicates negligible effect, and ? uncertain or unknown.

4.2.1 Quantity

As EPR increases the quantity of end-of-life products separately collected and sent for recycling, recyclers have more input at their disposal to produce secondary material. For example, collection and recycling targets set in the context of take-back requirements help ensure a constant and adequate supply of material to recycling plants. As advance fees are used to finance collection and recycling activities, they also lead to an increase in the quantity of material collected and treated. Advance *recycling* fees are put in place with the explicit aim to finance recycling activities: insofar as they are effective, they should increase the quantity of secondary material produced.[27]

Deposit-refund systems provide consumers strong incentives to return their end-of-life products in good condition, therefore ensuring that materials will continue their life in other products. Systems targeted at recycling lead to a high and rather homogeneous material inflow, which is a necessary condition for the production of large quantities of high-quality secondary material. By contrast, less material reaches recycling plants when returned products are reused, as is the case with glass or plastic bottles. While this is desirable from a circular economy point of view — reuse is a preferred option to recycling — it may entail that recycling plants have a harder time to secure the material supply necessary for them to operate in an efficient way. This implies higher costs for recycled material, which in turn reduces its financial attractiveness relative to virgin material.

4.2.2 Costs

EPR lowers the costs of secondary material production through at least two channels: increasing recycling efficiency, and helping attain economies of scale in production. Recycling efficiency can be increased by investments in innovation in recycling processes, which can be financed by advance recycling fees, PRO fees or unclaimed deposits. In the case of EPR for end-of-life vehicles in the Netherlands, for example, part of the revenues from advance recycling fees are used to support R&D

[25] The net effect depends on the share of returned products directed to reuse and that directed to recycling. When returned products are mostly reused, material input to recycling plants declines, and so does secondary material output. The opposite outcome occurs when most returned products are directed to recycling facilities.
[26] Too stringent take-back requirements based on e.g. material weight or volume, may have a negative effect on quality of secondary materials in the absence of differentiated tariffs.
[27] The effect of advance (disposal) fees directed to other waste management activities (e.g. waste collection) is more uncertain and depends on the value of secondary material: while more valuable materials, such as basic metals, are likely to be recycled, such advance fees may have no noticeable effect on the quantity of secondary plastics or textiles produced.

on new recycling techniques and develop protocols for safer vehicle disassembly (Tijm et al., 2021). The positive effects of investments in recycling efficiency are usually visible in the medium to long term.

EPR ensures a higher and more stable supply of input to recycling markets, which helps achieve economies of scale in the production of secondary materials (OECD, 2016; Verrips et al., 2019)**.** The flow of materials supplied to recycling plants is often low or unstable, which implies less efficient production processes and higher per unit costs of secondary material production. The higher the production costs of secondary materials, the more difficult it is for them to compete with virgin materials in the market. Hence, it is important that the scale of recycling operations increases, leading to economies of scale — i.e. declining costs per unit of output — and better market prospects for secondary materials.

Economies of scale can be promoted through take-back requirements, advance fees or deposit-refund systems in different ways. While both minimum collection and recycling targets could in principle promote economies of scale, recycling targets are likely to be more effective than collection ones, because they ensure that a minimum percentage of collected waste will be delivered to recycling facilities. Advance fees also promote economies of scale through financing the collection and sorting of end-of-life products and thereby increasing the supply of material offered to recycling plants. Deposit-refund systems provide strong incentives for consumers to return their products, which promotes economies of scale if these product flows are directed to recycling. Furthermore, deposit-refund systems are best suited to ensure that recycling input is of sufficiently homogeneous and high quality — another enabler of economies of scale — as consumers can only get a refund for products returned in an acceptable condition.

4.2.3 Quality

A large amount of collected material is currently downcycled into products of lower value. For example, recycled textile fibres from various products are used as insulation material, and rubber from car tyres in artificial sports fields (ARN, 2020; Campbell-Johnston et al., 2020). However, the environmental benefits of recycling are largest when secondary materials substitute virgin ones in high-value applications, where higher costs for resource extraction and processing, and product manufacturing are involved.

Targeted instruments are needed for EPR to have a positive impact on the quality of recycled material. The impact of EPR on the quality of recycled material is uncertain. Amongst the EPR instruments considered, a deposit-refund system is the one most likely to have a positive impact on recycled material quality, as it can best ensure that returned products are of homogenous and high quality (see also Tijm and Verrips 2019). Differentiating fees according to the recyclability of the product, as already implemented in the case of batteries and packaging (State Secretary for Infrastructure and Water Management, 2020c; Tijm et al., 2021), could also lead to improvements in the quality of secondary material. Fee differentiation can promote the use of more recyclable materials, increasing thus the homogeneity and purity of the input entering recycling plants, and accordingly the quality of their output. Using PRO fees or advance recycling fees to finance innovation in recycling processes can also improve secondary material quality in the longer term.

Recycling targets induce producers to recycle more material, but complementary policies are necessary to improve secondary material quality. EPR is often operationalised through recycling targets, whose attainment is financed by weight-based PRO fees. Without complementary policies in place, such forms of EPR design can lead to a high percentage of products being downcycled. For example, producers may decide to supply light-weight plastic bottles of which the end-of-life material can easily be used in lower-quality applications, but not back into new plastic bottles. Complementary policies directed to the production side can assist in further incentivising high-quality recycling. Such policies include minimum recycled content requirements for new products, such as those recently introduced at the EU level for plastic bottles (Parliament and Council of the European Union, 2019), and the full pricing of the external costs of primary material extraction and processing.

4.3 Effects of EPR on product design

One of the objectives of EPR is to promote eco-design, also called Design for Environment (OECD, 2016, 2001). Eco-design is a broad concept denoting the design and development of products with a lower environmental impact over their lifetime, namely from production until end of life (OECD, 2016). EPR provides eco-design incentives for producers who seek to minimise the costs of collection and treatment of end-of-life products. By developing products with lower environmental impacts at the end of their life, producers can save on these costs. This section focuses on how EPR affects product development, and on changes in EPR schemes that can further promote eco-design.

We focus on three objectives of eco-design: extending product lifetime (Section 4.3.1), increasing product recyclability (4.3.2), and reducing material use in production (4.3.3). A longer *product lifetime* implies that fewer units of a product can cover the needs of a given group of consumers in the long term. Environmental benefits can thus be achieved by the reduction of the total number of units produced and consumed.[28] A longer product lifetime can be achieved by developing more durable products and/or products that are easier to reuse and repair. Higher *product recyclability* entails lower post-consumer environmental damages, as more end-of-life products are directed to recycling instead of incineration or landfilling. It also leads to a higher, more stable, and more financially attractive supply of secondary materials, which results in lower environmental damages from raw material extraction and use. Another way to reduce these damages is through developing products that *use less material* to provide a given level of service to consumers. Such a form of higher material resource efficiency also implies lower environmental damages throughout the lifecycle of a product. While increasing material resource efficiency reduces environmental damage per product unit, extending product lifetime reduces damage from the total number of units produced intertemporally.

For EPR to have noticeable effects on eco-design, the contribution paid by each producer should reflect as closely as possible the social costs of collection and treatment of its own products. This implies that certain organisational structures of EPR and policy instruments are better suited to promote eco-design than others.

Individual EPR systems are generally more likely to promote eco-design than collective systems, as each producer is responsible for the end-of-life management of the products it develops. This is the most direct way to link producers with the end-of-life environmental impact of their own products and incentivise them to make changes in the product development phase to reduce it. In practice, however, EPR is usually implemented through collective systems, which are advocated for their cost-effectiveness and their potential to prevent free-riding.

The effectiveness of collective EPR systems in promoting eco-design can be increased through targeted incentives, such as a differentiation of the contributions paid by producers according to specific product attributes. An overview of the effects of take-back requirements, advance fees and

[28] For some durable goods, trade-offs between environmental damages caused at the production and end-of-life stage and those caused at the consumption stage intensify as products get older. For example, old internal combustion engine vehicles emit significantly more CO_2 and air pollutants than their newer counterparts; thus, there comes a moment when net environmental gains can be achieved by scrapping and replacing them with new vehicles. Similar arguments hold for various old large appliances, such as washing machines or refrigerators, where substantial energy and water savings can be achieved by replacing them with new ones.

deposit-refund systems on the three objectives of eco-design presented above are provided in Table 4.4. The table reveals strong uncertainty about the effects of EPR on product life, recyclability and material use, which has two sources: first, theoretical and empirical studies of the relevant effects of EPR have been scarce (Brouillat and Oltra, 2012); and second, in the few research efforts to identify the effects of EPR on product design, its impact has been found to be limited (OECD, 2016). However, this does not imply that EPR cannot have a more noticeable effect on product design; instead, it means that targeted incentives need to be introduced in the EPR policy mix for this purpose. A promising instrument in this context is a differentiation of producers' contributions according to the eco-design attribute that the policy aims to influence (OECD, 2016).

International cooperation — especially at the EU level — is key for the effectiveness of differentiated fees and other instruments targeted at promoting eco-design. As producers sell their products across the EU Single Market, fee differentiation needs to be based on the same attributes across the European Union for it to be effective. It is thus essential that EU countries agree on the bases of fee differentiation and provide producers with strong incentives for eco-design.

Producers' contributions can be differentiated regardless of the instrument mix chosen. In the case of take-back requirements, this would be operationalised through a differentiation of the fees paid by producers to the PRO, as also suggested by Hogg et al. (2020). In deposit-refund systems, consumers would have to pay a higher deposit — and receive a higher refund — for products with more environmentally damaging attributes. Advance fees would also be higher for products entailing higher social costs at the end of their lifecycle.

The benefits yielded by differentiated contributions should be traded off with higher implementation and monitoring costs. Eichner and Pethig (2001) show that differentiated fees based on material content are more efficient to improve product design, but costs of implementation and monitoring increase. Differentiated fees are desirable when the environmental and financial gains from their use are expected to outweigh the administrative costs of their implementation (OECD, 2016).

Innovation funds, financed by part of producer contributions to the EPR system, advance fees or unclaimed deposits, can be an effective means to stimulate eco-design. Producers may be encouraged to set part of their contributions aside in an innovation fund, whose role is to finance innovation in the design of more durable, repairable, reusable and recyclable products. The effectiveness of innovation funds in promoting eco-design heavily depends on their size: innovation is costly, and thus large funds are more likely to be effective than small ones.

Table 4.4

Effectiveness of EPR instruments in promoting eco-design

	Take-back requirements	Advance fees	Deposit-refund systems
Promoting eco-design	**?**	**o* / ?**	**?**
- Product life	?	o* / ?	+ / ?
- Product recyclability	+ / ?	o* / ?	o / ?
- Material use	?	o* / ?	o / ?

Note: + indicates an improvement, – indicates a deterioration, o indicates negligible effect, and ? uncertain or unknown.
*: an improvement (+) is possible if the fee is differentiated based on a relevant product attribute: e.g. a higher fee is levied on a less durable or recyclable product, or a product where more virgin material is used.

4.3.1 Product lifetime

EPR can theoretically incentivise producers to develop products with longer lifetimes, but these incentives may be limited compared to those of selling more products. Prolonging the lifetime of products, by making them e.g. more durable, reusable or repairable, implies that less products will be needed to cover a given set of needs in the long term. This entails lower end-of-life costs from the totality of products consumed. As EPR induces producers to take end-of-life costs into account in their decisions, it incentivises them to extend product lifetime (Runkel, 2003). This incentive will be effective in prolonging product lifetime insofar as it outweighs producers' short-term incentives to sell more — cheaper — products with a shorter lifespan. The empirical literature offers no insights into whether EPR has led to the design of more durable, reusable or repairable products.

Amongst the EPR instruments considered in this chapter, deposit-refund systems targeted at reuse are those most likely to incentivise producers to design products with a longer lifetime. Deposit-refund systems can effectively promote reuse — when it is a cost-effective option — as they provide strong incentives for consumers to return end-of-life products, and the quality of returned products is routinely monitored. This emphasis on reuse may incentivise producers to design more durable and reusable products, so that it is easier for consumers to return them in an appropriate condition. A deposit-refund system on beverage containers may for example induce producers to utilise more durable bottles. Take-back requirements and advance fees do not provide direct incentives for producers to develop products with longer lifespans, unless fees are differentiated according to product durability and lower fees are charged for products that last longer.

4.3.2 Product recyclability

Recyclability depends on multiple factors, including the number of materials used, the recyclability of each of these materials, and the ease of disassembling the product. Other things being equal, products containing a larger number of materials are more complicated to recycle. Some materials are less suitable for recycling than others: for example, black-coloured plastic is less recyclable than plastics of other colours (Dvorak et al., 2011). Specific additives can also pose problems to recycling. The ease of disassembly is an important factor for electronic appliances, motor vehicles, large battery packs, textiles and some types of packaging. A more difficult disassembly of a product or its components increases its treatment costs, and therefore reduces its recyclability.

Higher recyclability implies lower treatment costs, but in practice this incentive may be too small to outweigh the costs of making a product more recyclable (OECD, 2016)**.** Different materials have different properties, and the switch from the use of a non-recyclable material to a recyclable one in the manufacturing of a product may be particularly costly. Additional incentives may thus be needed in the context of EPR to increase product recyclability.

Differentiated tariffs would stimulate the design of more recyclable products. For example, products made from plastics that are easier to recycle could be charged a lower tariff than products composed of other types of plastics. This is already the case for plastic packaging in Italy and the Netherlands, where lower fees are charged to packaging that is more easily recycled (Tijm and Verrips, 2019). Other factors that influence recyclability, and may thus be used as a basis for fee differentiation, are the combination of materials used in a product, and the ease of disassembly. Mono-material goods are in general easier to recycle than multi-material ones; this holds also for goods which are easier to disassemble. Fees differentiated according to the recyclability of products, i.e. to the number of materials used, the recyclability of materials, or the ease of disassembly, provide a way to reward the design of more recyclable products (Brouillat and Oltra, 2012). A similar incentive is derived from deposits differentiated according to product recyclability (Calcott and Walls, 2005).

4.3.3 Material use

EPR can affect material use through a shift from raw to recycled materials, through the manufacturing of more lightweight products, or through the substitution of more harmful substances by less harmful ones.[29] What effects EPR eventually has on material use depends on the product attributes targeted by the policy instrument mix. If the basis used to set quantitative targets or fees is weight, then the policy stimulates the production of lighter products. By contrast, when targets and fees are a function of the share of recycled materials in the content of a product, EPR promotes a shift from raw to recycled materials in production. EPR instruments penalising the use of particular (e.g. hazardous) substances induce producers to seek less harmful alternatives.

Although collection and recycling targets based on product weight and weight-based fees lead to lighter products, they do not trigger other forms of eco-design. In Europe, EPR is usually operationalised through collection and recycling targets expressed as a percentage of a product's weight. The cost allocation between producers for the financing of a PRO is often also weight based. These EPR instruments stimulate the downsizing or light-weighting of products (Walls, 2006). For example, various forms of packaging have become lighter in Europe, such as PET bottles and aluminium tins (OECD, 2016). Even though it has not been possible to quantify the contribution of EPR to the light-weighting of various forms of packaging in Europe, it is believed to have helped in this direction. However, weight is not always directly related to the environmental burden of a product, and weight-based policies do not give incentives to change other factors, such as the use of recycled inputs, of less harmful materials or of cleaner production processes.

Setting targets or differentiating fees according to the recycled content of a product can promote a shift from the use of raw to secondary materials. Recycled content recently started being used as a basis for the setting of targets or fees. The EU single-use plastics Directive introduced specific recycled content targets for plastic bottles, amounting to 25% in 2025 and 30% in 2030 (European Parliament and Council of the European Union, 2019). In France, producers of packaging receive a 10% fee reduction, when more than half of the content of their product is made from recycled materials (OECD, 2016). Using recycled content as a basis for targets or fees can be an effective means to reduce the use of specific virgin materials. The effectiveness of weight-based targets and fees in achieving this objective can be further improved if the focus is shifted from *product* weight to *material* weight. While this would make no difference for mono-material goods (e.g. some forms of packaging), it could have important implications for multi-material ones. Such a policy would imply that more stringent targets and/or higher fees would apply to products using virgin materials that cause higher environmental damages.

4.3.4 Trade-offs between eco-design aspects

EPR policy needs to take into account possible trade-offs between different elements of eco-design. For example, weight is not always a good measure of environmental impact, especially in the case of multi-material products. Light-weighting products may reduce material use, but not necessarily lead to more environmentally friendly product designs in the end. Eichner and Pethig (2003) suggest that durability is often achieved by making a product heavier (e.g. designing a car tyre with a more robust casing and tread). Furthermore, there are trade-offs between durability and the functioning of secondary material markets. Higher product durability implies less materials supplied for recycling. There are also trade-offs between recyclability, durability and the use of hazardous materials, well manifested in the case of battery production. Nickel batteries are more recyclable, but

[29] In theory, EPR should also have an effect on output. EPR alters the optimal production decision for producers: they decrease the quantity put on the market (Fleckinger and Glachant, 2010). This output effect also helps reduce the amount of materials used. The decrease in output, however, typically reduces consumer welfare: consumers derive less utility from a lower amount of products put on the market. Therefore, there is a trade-off between reduction in environmental externalities (welfare-increasing) and reduction in output (welfare-decreasing).

have a shorter lifetime. Lead-acid batteries can be 99% recyclable, but require the use of lead, a toxic metal (Tijm et al., 2021). In developing EPR policies, it is important to consider such trade-offs between eco-design attributes and choose an instrument mix which steers product design towards the attribute combination with the maximum net social benefit.

4.4 EPR and market structure

EPR can also have an impact on competition between firms. The relation between EPR and competition is complex, and a full review is out of the scope of this report.[30] We focus here on competition in products to which EPR is applied (Section 4.4.1) and competition in waste management (Section 4.4.2).

4.4.1 Competition in products under EPR

One of the more general benefits of EPR systems is that the international competitiveness of domestic producers is not affected. Other instruments that correct for environmental externalities, such as a pollution tax, increase domestic producers' costs both for the products directed to the domestic market and for those they export. This distorts competition with foreign competitors, who are not liable to the tax (in the absence of cross-border adjustment mechanisms). EPR systems hold for all quantity put on the domestic market. This increases prices for domestic consumers, but does not create distortions between domestic and foreign producers.

PROs often take shape as monopolies, although some form oligopolies or competitive markets. Three main arguments are often put forward for the monopoly status of PROs (OECD, 2016). First, managing producer responsibility leads to economies of scale. Second, the PRO can be more easily overseen by regulators. Third, a single PRO can more effectively address the free-riding problem.[31] These arguments all favour concentration of PRO activities, with a monopoly amounting to the highest possible market concentration. A counterargument against a monopoly follows from diminished incentives for efficiency. Unfortunately, the question under what conditions monopoly PROs are more efficient than competitive ones has not been answered empirically, due to too many differences between cost factors in waste streams and countries.

The formation of monopoly PROs may have spill-over effects, harming competition in product markets (OECD, 2016). The exchange of market information within a PRO may facilitate collusive behaviour. Examples include agreements to pass the PRO fee on to consumers, market allocation, or creating barriers to the entry of rival producers. Collusion reduces welfare compared to competitive markets.

To safeguard competition, PROs need to be monitored by regulators, and PRO agreements should be assessed by external parties. Competition agencies should consider whether the level of information that is to be exchanged is not too extensive. If so, a solution could be an independent firm that collects all the information and distributes to each firm only the information that it needs to fulfil its requirements (OECD, 2016). Monitoring is also required to ascertain that PRO do not create barriers to entry or discriminate amongst producers.

[30] For example, a relatively unexplored topic is the impact of AVVs (see Chapter 2) on competition (State Secretary for Infrastructure and Water Management, 2020d). We do not discuss competition in the market for PRO services, because costs of PRO services are low compared to waste management costs (OECD, 2016).
[31] In this context, free riders are defined as "those producers who benefit from EPR systems without contributing their share of the costs" (OECD, 2016).

The competition concerns of PROs imply that in concentrated markets individual EPR systems may be more appropriate (Kaffine and O'Reilly, 2015). As usual, the potential competition concerns should be balanced with gains from economies of scale in collection and recycling activities as well as monitoring costs. In any case, monopoly PROs should not be the default.

4.4.2 Competition in waste management

Some markets for waste collection and sorting are *natural monopolies* that are most efficiently served by a single entity. After responsibility has shifted from municipalities, producers may decide to install a (new) monopolist for this job. Economies of population density and economies of scale imply that kerbside collection should be a regulated private monopoly or a municipal monopoly. Other collecting and sorting markets, such as the collection of recyclable waste from business, tend not to be natural monopolies (OECD, 2016). This type of waste is collected from fewer collection points, so that economies of population density apply to a much lesser extent. Also, the markets for collection from business may be geographically larger, not just local but regional or national.

Low competition in waste treatment markets may result in high costs of recovery and disposal. As with waste collection and sorting, waste treatment markets can be concentrated. Markets in which EPR is introduced may see the exit of small-scale waste treatment, especially if the PRO restricts the trade of collected waste outside its members. Also, high non-competitive treatment costs reduce incentives for producers to recycle waste.

The design of procurement by PROs can increase competition in waste management markets (OECD, 2016). Many PROs procure waste collection and sorting services. Transparent tender rules and procedures that do not discriminate, and attract sufficient bidders, significantly reduce collection costs. For example, the German packaging PRO DSD (*Duales System Deutschland*) managed to reduce costs of collecting and sorting by 20 to 30% between 2003 and 2005, after modifying tender conditions to attract more bids from smaller enterprises. In the first call for tender in 2003, about half of the contracting areas had received only a single bid.

4.5 Conclusions

Collective EPR systems — which is the most common form of organising EPR in the Netherlands and other EU Member States — have advantages and disadvantages when compared to individual ones. The advantages of collective EPR systems can be summarised in that they:

+ are more cost-effective;
+ are less costly to implement for the producer and to monitor for the government;
+ more easily achieve economies of scale in waste collection and treatment; and
+ more effectively address free-riding problems.

On the other hand, the main disadvantages of collective EPR systems are that they:

- provide weaker eco-design incentives; and
- may increase collusive behaviour in product markets.

However, it is not only the choice between a collective and an individual EPR system that matters; the number of PROs being active in a collective system also influences the effects of EPR. The more PROs operating in a collective system, the higher administrative and monitoring costs will be, but also the less likely that market competition is affected by the system. It is not a priori clear what the effect of the number of PROs on eco-design would be.

Regardless of how the EPR system is organised, what matters for environmental effectiveness is that each producer remains responsible for its own products when they reach an end-of-life stage. This requires a closer monitoring of waste flows to gain a good understanding of their composition in terms of the type of products being discarded and the producers putting them on the market. Alternatively, producers could be required to provide accurate estimates of the durability, repairability, reusability and recyclability of their products and contribute to collection and treatment costs accordingly.

Deposit-refund systems and differentiated fees can provide targeted incentives for the return of end-of-life products and for eco-design, while targets can stimulate specific R-strategies. Beyond the type of policy instrument mix used, the scope, stringency, adaptability and coherence of the implemented mix is key for its effectiveness. EPR policies with a wider scope — i.e. covering as many products used by households and firms as possible — higher stringency, easier adaptability and better coherence with other policies are generally more likely to help achieve circular economy objectives.

5 Policy implications and future research

We now turn to the policy implications of our analysis and our suggestions for further research. Our attention remains fixed on the design, functioning and effects of EPR, without focusing on opportunities and challenges related to specific product groups. Our case studies on batteries, end-of-life vehicles and medicine, presented in a separate background document (Tijm et al., 2021), provide product-specific insights complementary to those offered in this study.

5.1 General implications for policy

The effectiveness of EPR in achieving circular economy goals depends on the composition, focus, scope, stringency, adaptability and coherence of the instrument mix used. These characteristics of the instrument mix determine the incentives that EPR provides for producers and consumers. The scope of EPR should cover all sources of relevant environmental impacts, i.e. all end-of-life product units, regardless of how they are disposed of or by whom. The stringency of the instrument, that is, the level of the target or the fee, also determines the effectiveness of EPR. More stringent policies are likely to lead to greater environmental benefits, but also higher costs. Stringency may also need to evolve over time. Incremental and predictable increases of targets or fees incentivise producers to keep improving their practices. Coherence with other policies and avoidance of overlaps is also key for the effectiveness of EPR.

Wider use of market-based instruments can make EPR policy more efficient. Market-based instruments, such as fees and deposit-refund systems, provide incentives for producers to continuously improve their practices, while static regulatory requirements do not. Price mechanisms may help increase the use of recycled inputs up to a socially optimal level. Even if collection or recycling targets become more stringent over time, it is difficult for policy makers to decide on the optimal timing to change these targets and the optimal level of collection or recycling, and there is no direct incentive for producers to move beyond these targets if they have the opportunity.

The effectiveness of collective EPR systems in promoting eco-design can be increased through differentiation of the contributions paid by producers according to specific product attributes. Some PROs (e.g. for batteries and packaging) already differentiate fees according to the weight of the product and/or its recyclability. Extending this practice to other product groups and other product attributes, such as the content of recycled material, can incentivise producers to design products with a lower environmental impact. Governments may consider setting minimum requirements to tariff differentiation for products whose environmental impact can be clearly linked to specific attributes. Producers' contributions can be differentiated under any of the three main EPR instruments. When there are trade-offs between eco-design attributes, differentiation could be based on multiple criteria, weighted according to their importance for welfare.

A wider policy implication of our study is to make producers financially responsible also for the end-of-life products that are not separately collected. In practice, producers' financial responsibility is limited to separately collected products. This could be complemented by fees charged by public authorities, which would cover the costs of managing end-of-life products in mixed waste, and the costs of cleaning up littered or dumped products. Such an expansion of the current scope of EPR would incentivise producers to constantly strive for higher collection and recycling rates, promote eco-design, and relieve the financial burden on municipalities.

Expected benefits from implementing any EPR policy, including fee differentiation, should be weighed against possibly higher costs. Even if a complete welfare analysis is not possible (due to data limitations), it is useful to make trade-offs between environmental benefits and administration and implementation costs.

We conclude that well-designed EPR is a useful component of the policy mix for the transition to a circular economy, but no panacea. Without EPR, municipalities might still be responsible for waste management, producers would be less likely to internalise the costs of waste management and incentives to recycle or reuse would be weaker for the majority of end-of-life products. Nevertheless, there is ample room to steer EPR towards promoting eco-design and reuse. More generally, the transition to a circular economy aspires to more than what EPR can deliver on its own. Just as its effects depend on other waste policies, EPR will always require accompanying policies — mostly targeted at the production and consumption phases of a product's lifecycle — to facilitate the transition to a circular economy.

5.2 Knowledge gaps and directions for future research

Experience with EPR has been long both in the Netherlands and abroad, but much remains unknown about its effectiveness in achieving policy goals. A solid knowledge base on its economic and environmental effects and their implications has not yet been developed for at least two reasons. First, EPR has been applied with substantial heterogeneity to different product groups and countries, which hampers the generalisability of findings to other contexts. Second, existing studies are mostly of a theoretical or descriptive nature. While the lack of empirical analyses does not mean there are reasons to argue against the environmental effectiveness or cost-effectiveness of EPR, policy makers should consider that the evidence basis underpinning the effectiveness of EPR is not yet robust.

A first promising avenue for future research on EPR is the empirical evaluation of its environmental effects. The lack of empirical evidence on the effectiveness of EPR schemes can mainly be explained by challenges posed by data availability, and by methodological difficulties in disentangling the effects of EPR from the effects of other policies or factors (see also Kaffine and O'Reilly, 2015). Such a methodological challenge is the difficulty in constructing convincing counterfactuals. Empirical tests on the effectiveness of EPR require the development of a counterfactual scenario, based for example on data on the period before the EPR policy was introduced, or on other countries without EPR. This may be difficult because the collection of data on collection and recycling rates typically started with the introduction of EPR, so there is very little data available in the absence of the policy. More econometric research is needed to overcome this problem, for example by exploiting a phased implementation of different EPR instruments.

The second area where more research is needed concerns the cost-effectiveness of alternative EPR designs and their impact on welfare. EPR has improved the way that end-of-life product and material flows are monitored, but more efforts are necessary to gather the fruits of monitoring. Policy makers and researchers would need better access to data on monetary flows, in order to study the economic effects of alternative EPR designs. Data on the cost structure of PROs are now very difficult to access, mostly because they are considered commercially sensitive. EPR policies could include the requirement to report on financial flows in addition to physical flows. Making EPR schemes more transparent with regard to the financial performance of PROs would help policymakers obtain more insights into the welfare effects of EPR and its cost-effectiveness.

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Appendices

Appendix A – Overview of EPR legislation

The management decrees regulating EPR for batteries, end-of-life vehicles, packaging, vehicle tyres and waste electrical and electronic equipment in the Netherlands, as well as the relevant EU legislation, are presented in Table A.1.

Table A.1: Product groups on which EPR has been implemented

Product group	Dutch legislation	Year of first implementation	EU legal framework
Batteries and accumulators	1995-2008: Besluit beheer batterijen 2008 onwards: Besluit beheer batterijen en accu's 2008 Regeling beheer batterijen en accu's 2008	1995	Directive 2006/66/EC[32]
End-of-life vehicles	Besluit beheer autowrakken	2002	Directive 2000/53/EC[33]
Packaging	Until 2006: Regeling verpakking en verpakkingsafval 2006-2014: Besluit beheer verpakkingen en papier en karton 2015 onwards: Besluit beheer verpakkingen 2014 Regeling beheer verpakkingen	1997	Directive 94/62/EC as amended by Directive (EU) 2018/852[34]
Vehicle tyres	Besluit beheer autobanden	2004	No separate legislation on tyres. Removal of tyres from end-of-life vehicles is regulated by Directive 2000/53/EC.

[32] Directive 2006/66/EC of 6 September 2006 on batteries and accumulators and waste batteries and accumulators and repealing Directive 91/157/EEC (European Parliament and Council of the European Union, 2006).
[33] Directive 2000/53/EC of 18 September 2000 on end-of life vehicles (European Parliament and Council of the European Union, 2000).
[34] Directive 94/62/EC of 20 December 1994 on packaging and packaging waste, and Directive (EU) 2018/852 of 30 May 2018 amending Directive 94/62/EC on packaging and packaging waste (European Parliament and Council of the European Union, 2018b, 1994).

Waste electrical and electronic equipment (WEEE)	Regeling afgedankte elektrische en elektronische apparatuur	2014	Directive 2002/96/EC recasted in Directive 2012/19/EU[35]

Sources: (European Parliament and Council of the European Union, 2018b, 2012, 2006, 2003, 2000, 1994; Rijkswaterstaat, 2020a).

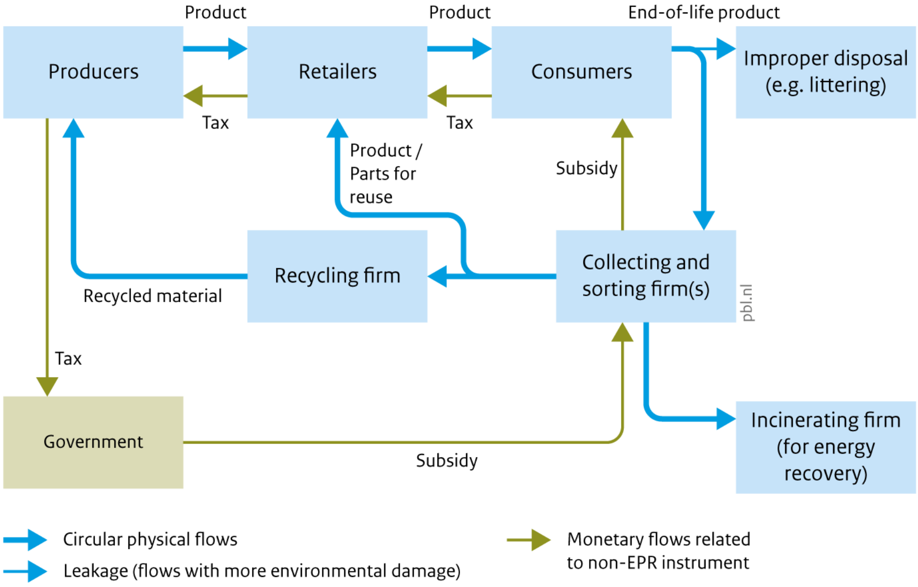
[35] Directive 2002/96/EC of 27 January 2003 on waste electrical and electronic equipment (WEEE), recasted in Directive 2012/19/EU of 4 July 2012 on waste electrical and electronic equipment (European Parliament and Council of the European Union, 2012, 2003).

Appendix B – Other definitions of deposit refund

In Chapter 3, we describe three different instruments that are often used in EPR schemes. Regarding the deposit-refund scheme, this appendix points to differences between our definition and others in the literature.

Figure B.1

Combination of product tax and recycling subsidy (not falling under the definition of EPR)



Source: PBL and CPB

Many economists (Fullerton and Wu, 1998; Kinnaman, 2014; Walls, 2013) use other definitions of the term 'deposit-refund', which do not fall within the scope of EPR. We define a *deposit-refund scheme* as a scheme in which consumers pay a deposit to the producers (or equivalently to distributors or retailers) and receive a refund after returning the product. The government does not have an active intermediary role in this process. By contrast, if consumers pay a tax to the government and recyclers receive a subsidy from the government, this is a *combined product tax and recycling subsidy* (Fullerton and Wolverton, 1999). [36] Figure B.1 shows the physical and monetary flows under this two-part instrument, which does not fall under our definition of EPR. The reason for this is that responsibility for the end-of-life stage of the product is not transferred to producers under this instrument; it remains with the government. One important difference is that unclaimed deposits in the case of deposit-refund systems flow back to the producers, while in a tax-subsidy scheme this advantage is with the government.[37] The tax-subsidy scheme might also be more costly to implement, with higher administration costs (for the government and the recycling industry), than a deposit-refund scheme.

[36] From the perspective of the consumer, the combination of a tax and recycling subsidy gives the same incentives as a deposit-refund scheme.
[37] However, in some deposit-refund systems in the United States (Michigan and Massachusetts), unclaimed deposits must be returned to the state as well (Walls, 2013).