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New Aspects of EPR: Extending producer responsibility to additional product groups and challenges throughout the product lifecycle

Andrew Brown,

Frithjof Laubinger,

Peter Börkey

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New Aspects of EPR: Extending producer responsibility to additional product groups and challenges throughout the product lifecycle

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By Andrew Brown (1), Frithjof Laubinger (1), Peter Börkey (1)

(1) Environment and Economy Integration Division, Environment Directorate, OECD

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Keywords: Plastics, Trade, Circular economy, Waste management

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Andrew Brown; Andrew.Brown@oecd.org
Frithjof Laubinger; Frithjof.Laubinger@oecd.org
Peter Börkey; Peter.Borkey@oecd.org

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Abstract

Extended Producer Responsibility (EPR) is a policy approach that makes producers responsible for their products at the post-consumer stage of the lifecycle. It has been widely adopted by governments and companies across the OECD membership and beyond and is currently most commonly used for electronics, packaging, vehicles, and tyres. The success of EPR in increasing material recovery rates has triggered a debate about expanding the use of EPR to additional product groups. Additionally, there is a debate about expanding producer responsibilities to additional impact categories, which go beyond the traditional use of EPR to cover end-of-life costs that occur at the domestic level.

This paper presents a discussion of relatively novel applications of EPR to additional product groups (plastic products beyond packaging, textiles, construction materials, and food waste) and to environmental impacts (design considerations, pollution and littering) that occur throughout the product lifecycle. Based on select case studies, this report evaluates the successes and challenges that early adopters of applying the EPR approach to new product groups or additional environmental impact categories have experienced. It reviews the arguments for further application of EPR, possible limitations and provides guidance on when and how to best apply an EPR.

Keywords: plastics, trade, food, textiles, circular economy, waste management

JEL Codes : F18, L65, L66, L67, Q53, Q56

Résumé

La responsabilité élargie des producteurs (REP) est une approche politique qui rend les producteurs responsables de leurs produits au stade de la post-consommation du cycle de vie. Elle a été largement adoptée par les gouvernements et les entreprises des pays membres de l'OCDE et au-delà, et est actuellement utilisée le plus souvent pour l'électronique, les emballages, les véhicules et les pneus. Le succès de la REP dans l'augmentation des taux de récupération des matériaux a déclenché un débat sur l'extension de l'utilisation de la REP à d'autres groupes de produits. En outre, il existe un débat sur l'extension des responsabilités des producteurs à d'autres catégories d'impact, qui vont au-delà de l'utilisation traditionnelle de la REP qui consiste à couvrir les coûts qui se produisent à la fin de vie des produits au niveau national.

Ce document présente une discussion sur des applications relativement nouvelles de la REP à des groupes de produits supplémentaires (produits en plastique au-delà des emballages, textiles, matériaux de construction et déchets alimentaires) et à des impacts environnementaux (considérations de conception, pollution et déchets) qui se produisent tout au long du cycle de vie du produit. Sur la base d'études de cas sélectionnées, ce rapport évalue les succès et les difficultés rencontrés par les premiers utilisateurs de la REP dans l'application de cette approche à de nouveaux groupes de produits ou à des catégories d'impacts environnementaux supplémentaires. Il passe en revue les arguments en faveur d'une application plus poussée de la REP, les limites éventuelles et fournit des conseils sur le moment et la manière d'appliquer au mieux une REP.

Mots clés : Plastiques, commerce, alimentation, textiles, économie circulaire, gestion des déchets

Classification JEL : F18, L65, L66, L67, Q53, Q56

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Executive Summary

Extended Producer Responsibility (EPR) is a policy approach that makes producers responsible for their products at the post-consumer stage of the lifecycle. It has been widely adopted by governments and companies across the OECD membership and beyond and is currently most commonly used for electronics, packaging, vehicles, and tyres. Extended Producer Responsibility has successfully contributed to (i) shifting end-of-life (EoL) management costs of products from the public sector to producers and consumers, (ii) increasing separate collection of waste that can be problematic when mixed into the general waste stream and (iii) increasing material recovery rates.

In addition, EPR has the potential to provide producers with incentives for the design of more easily recyclable or re-useable products. However, to create effective product design incentives, there is a need for careful modulation of EPR fees, something that has only emerged relatively recently.

The success of EPR in traditional product sectors has triggered a debate about expanding the use of EPR to additional product groups, including:

- Products that frequently evade public collection and cause costly environmental impacts, such as fishing gear and tobacco product filters; as well as
- High-volume or high-impact waste streams, such as food waste, textiles or construction and demolition waste.

In addition, there is a debate about expanding producer responsibilities in EPR schemes to additional impact categories, which go beyond the traditional use of EPR to cover end-of-life costs that occur at the domestic level. These impact categories include:

- Micropollutants that are released during the use-phase by textiles and tyres, which are costly to capture and treat. EPR is under consideration by some policymakers as a means to finance related mitigation measures, notably upgrades to municipal wastewater treatment plants that would enable them to retain microplastics in sewage sludge.
- Items that are frequently exported as used goods for extended use in other markets, which then fall out of the scope of traditional producer responsibility in domestic markets, such as textiles, automobiles and electronic and electrical equipment.

Some OECD countries have already started to implement EPRs for several of these novel approaches. EPR schemes exist for garments, sleeping mattresses, carpets, cooking oils, and paints, in some cases for many years. They appear to have helped generate hundreds of millions in annual funding for improvements in rates of collection and material recovery in these sectors.

Other new EPR schemes are at advanced stages of planning and will soon take effect. For example, the EU requires its member states to adopt EPRs for tobacco product filters and end-of-life fishing gear by 2023 and 2025 respectively to help cover the costs of clean up and recycling.

There are also a number of new EPR applications that are at present only at the stage of being debated and for which no concrete plans for their implementation exist. For example, this is the case of discussions around the use of EPR to help address microplastics and other pollutants in municipal wastewater.

Based on an evaluation of these cases, this report identifies the opportunities and challenges for the use of EPR in these new approaches.

The opportunities for a number of additional product groups already appear relatively clearly and are similar to those in traditional EPR applications: (i) shifting end-of-life management costs from the public sector to the producers and consumers of products, (ii) improving collection rates, and (iii) improving recovery rates in a cost-efficient way.

However, there are also challenges that need to be addressed, including:

- How to define a producer that is liable for paying the EPR fee: Usually, a producer is defined as the entity that places the product on the market. However, in some cases other options exist or may arguably be more useful, generating a policy debate. For instance, for fishing gear the "producer" could be the gear manufacturer or the vessel owner.
- Establishing a methodology for EPR fee rates that is transparent, fair and functional: For several product groups the setting and calibrating of financial responsibilities of producers is a challenge. Non-transparent and unclear methodologies for EPR fee calculation can lead to a sense of arbitrariness and provide opportunities for industry to engage in the fee-setting process.
- Allocating producer responsibility in the context of limited data availability: Data limitations can pose a barrier to the clear allocation and enforcement of producer responsibility.

Given that EPR implementation can be challenging for certain product groups or impact categories, there are merits to comparing the use of EPR to other, alternative policy approaches. EPR may not always be the best suited policy and other policy tools may be better adapted to serve certain purposes. In several cases, the primary argument for EPR is to generate revenues. However, debate remains about whether this is a sufficient justification for an EPR. A consideration for EPR should be whether producers have sufficient leverage and the specialised expertise required to reduce their products' environmental impacts.

In some instances, consumer behaviour has a primary role in the environmental impacts and thus is at least partly external to producer actions (e.g. reducing littering of single-use plastic products). In other cases, mitigation of end-of-life impacts lies too far outside of a producer's expertise to benefit from their involvement (e.g. clean-up of shed microfibres from textiles from wastewater through upgrades of wastewater treatment facilities). If consumer behaviour is the key source of environmental impacts, other policies such as more systematically enforced littering fines may incentivise behaviour change more effectively.

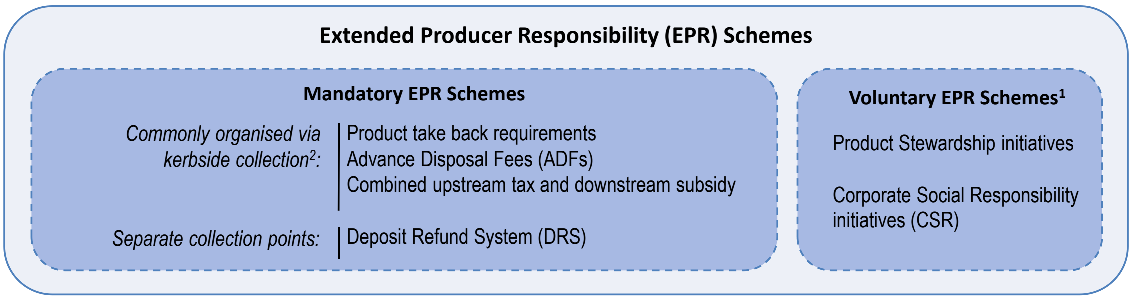
For some products a general waste collection or treatment system may be more effective than assigning the responsibility of organising (separate) collection and treatment of end-of-life products to the producer. Where EPR becomes a mere funding instrument, a waste charge or an earmarked tax that implements the polluter pays principle may have advantages. A key consideration for EPR implementation is whether producers have a specialised expertise, or a role in coordinating such expertise across the value chain to reduce EoL impacts. EPR appears to be more experimental where the case for specialised expertise or coordination is less clear, seen in tobacco product filters and fishing gear.

1 Introduction

Extended Producer Responsibility (EPR) is an environmental policy approach in which a producer's responsibility for a product is extended to the post-consumer stage of the product's lifecycle (OECD, 2016[1]). The approach seeks to increase recovery rates, and decrease waste generation and leakage by transferring end-of-life (EoL) management costs from the general public to producers and consumers of targeted products.

Different EPR policy instruments exist to assign producers with financial, and sometimes physical, responsibility of waste management (Figure 1.1). Mandatory EPR policy instruments commonly finance or organise kerbside collection of EoL products, such as product take-back requirements, Advance Disposal Fees (ADF) or upstream product taxes combined with downstream subsidies for waste management. A Deposit Refund System (DRS) is an EPR policy instrument if producers finance and/or operate the system (Laubinger et al., 2022[2]). In some markets, where mandatory EPR policy instruments do not exist, producers also commit to taking responsibility through voluntary product stewardship or Corporate Social Responsibility (CSR) initiatives.

Figure 1.1. EPR policy instruments



EPR was first proposed in academic literature as a means for increasing recycling rates and tapping the expertise and financial resources of producers, amid concerns of resource scarcity, pollution, and growing costs of public management of waste (Lifset, 1993[3]; Lindhqvist, 2000[4]) Precursors of this approach go back to the 1970s, first in the form of producer-funded deposit refund system laws for beverage containers in North America. In the 1990s countries started to adopt formal EPRs in form of ADFs and product take-back requirements.

In the 2000s, EPR adoption took off further, with the European Union implementing several Directives that required member states to implement EPRs for Waste Electronic and Electrical Equipment (WEEE), batteries, accumulators, vehicles, and encouraging adoption of EPR for packaging. As well, Japan and South Korea adopted policies for EPR for packaging and vehicles and in North America and Australia, several EPRs were implemented at the sub-national level (Kaffine and O'Reilly, 2013[5]).

In recent years EPR has gained further policy attention. The geographic scope of EPR continues to grow, with more countries implementing EPR policies for traditionally represented product sectors. For example, Chile as well as California, Oregon and Maine in the United States have recently enacted mandatory packaging EPR laws in the form of product take-back requirements with an ADF and several additional US states are currently discussing similar EPR policies (Maine legislature, 2021[6]; Oregon State Legislature, 2021[7]; California Legislative Information, 2022[8]). As well, Lithuania and Ireland have implemented a producer-financed deposit refund system, and other countries, such as the United Kingdom and Spain are considering DRS for beverage containers to increase separate collection rates. At EU-level, a fresh round of EPR adoption by the member states should occur as they begin to implement the requirements of the Single-Use Plastics Directive, which will make EPR mandatory for several additional product groups.

The use of EPR as a policy approach to increase recycling rates, is also increasingly gaining support from industry stakeholders, particularly in the packaging sector. For example, within the auspices of the Consumer Goods Forum, a group of 40 firms in the manufacturing and retailing of packaged goods, and, separately, the Ellen MacArthur Foundation together with more than 100 major businesses in the packaging value chain publicly released strong statements in support of EPR as a policy approach (The Consumer Goods Forum, 2020[9]; Ellen MacArthur Foundation, 2021[10]).

In addition to a rise of EPR in traditional product groups, there is anticipation and discussion of expanding the use of EPR to new product sectors and new uses. These “new aspects” in EPR policy development can be grouped into two categories:

- A more widespread implementation of the approach to additional product groups that are, to date, not typically covered by EPR schemes; and
- A more widespread coverage of producer responsibility, expanding the objective beyond the current traditional focus on collection and recycling rates, to additional environmental impact categories that occur outside of the end-of-life phase of a product.

This report aims to evaluate the successes and challenges that early adopters of applying the EPR approach to new product groups or additional environmental impact categories have experienced to inform the ongoing debate about when and how to best apply an EPR. The report is structured as follows: chapter 2 reviews the recent evidence for the successes of and challenges for EPR, chapter 3 provides an initial discussion of recent examples in which an OECD member country has adopted an EPR policy in a sector not traditionally covered (i.e. not in packaging, tyres, electrical and electronic equipment vehicles, batteries), and chapter 4 reviews examples in the policy debate on expanding the definition of producer responsibility outside the present focus on the post-consumer phase of the lifecycle. Chapter 5 reviews successes and challenges from the case studies and compares EPR to alternative approaches, and chapter 6 concludes with early insights.

2 Evidence of the benefits of EPR

The justification for an EPR for additional product groups and applications is based on its hypothetical ability to deliver similar benefits as EPR for traditionally covered products. This chapter reviews the recent literature on the available evidence about four theoretical benefits that are commonly associated with EPR implementation (OECD, 2016[1]):

- Cost recovery: EPR helps shift the financial responsibility of waste management from municipalities to producers of waste generating products.
- Separate collection: EPR can enable or improve separate collection of waste that can be problematic in the general waste stream
- Material recovery: EPR policies often contain targets or incentives that aim to increase collection and recycling rates. The private sector is deemed to reach these targets more cost-efficiently.
- Design for environment: By implementing the "producer pays principle", EPR incentivises producers to invest in product design that reduces downstream environmental impacts from waste treatment and/or prevents the upstream environmental impacts from resource extraction to reduce EPR fee payments.

There is evidence that EPR implementation has brought about most of these theoretical benefits. EPR facilitates increases in rates of separate collection of waste that can be problematic in the general waste stream, such as batteries and EEE. For example, a case study of the Midi-Pyrénées region (France) that compared waste data for the year of implementation of the EPR for WEEE (2007) with developments four years later (2011), found increased collection of WEEE from 2.9 to 8.8 kg/inhabitant (Bahers and Kim, 2018[11]).

Implementation of EPR schemes has also coincided with increases in recycling rates and reductions in final landfilling or incineration rates of covered materials and products. After the adoption of packaging EPR in Portugal and Spain, both countries experienced higher relative increases in recycling rates for EPR covered waste streams compared to overall municipal waste recycling (Rubio et al., 2019[12]). In South Korea, the introduction of its plastic packaging EPR helped the country to meet recycling rate targets of 70-80% depending on material (Jang et al., 2020[13]). After the introduction of Japan's EEE EPR in 2001, recycling rates increased by about 20% between 2001 and 2014 for different products (Shimada and Van Wassenhove, 2019[14]). A review of five markets (British Columbia, Ontario, Quebec (Canada), Greece, and Malta) that introduced a packaging EPR programme found that recycling rates increased between roughly 10 to 44% five years after introduction (Hesterman, Dimino and Ricchi, 2020[15]). A review of EPR programmes in Europe noted increases in recycling rates of all observed programmes (Gendell et al., 2021[16]). A review of seven markets (British Columbia, Quebec (Canada), Belgium, Spain, Portugal, the Netherlands, and South Korea) noted modest to significant increases in packaging recycling rates after introduction of an EPR for the sector (The Recycling Partnership, 2023[17]).

There is also some early evidence that EPR may instigate innovation: a difference-in-difference model study of manufacturing industries (electrical appliances, automobiles, lead-acid batteries, and packaging) in the People's republic of China from 2010 to 2018 estimates that the implementation of EPR increased the number of green patents by 23% (Zhao et al., 2021[18]).

For one of the four benefits of EPR – EPR as a tool for Design for Environment – there is only limited evidence that this has occurred so far (OECD, 2016[1]; Kunz, Mayers and Van Wassenhove, 2018[19]; Gendell et al., 2021[16]). Owing to economies of scale, most EPR schemes are organised in an industrywide way that provides only a limited link between product design and the fees paid by producers per product or per weight of material used. For example, EPR fees for packaging material have traditionally been based on a per kilo fee assessment, which incentivised some material reductions (i.e. light weighting) but did not provide incentives for other design changes to improve product circularity.

EPR implementation does not necessarily seem to lead to a large reduction in per-capita waste generation. An overall evaluation of temporal variation in EPR fees incurred by producers in 25 EU countries with EPR programmes for packaging from 1998 to 2015 found only a small reduction in packaging waste per capita and no significant substitution effect (Joltreau, 2022[20]). Joltreau conducted a regression analysis of packaging waste per capita and cost differences (by material, country, and year), which identified a 1% increase in fees incurred by producers was correlated with a 0.06% (or roughly 100g) decrease in packaging waste per capita. Joltreau also conducted regressions by material (paper/cardboard, composites, glass, PET, plastics, aluminium, and steel), and found no systemic substitution effects between materials due to variation in EPR fees.

EPR for durable products without a re-use targets or eco-modulation of EPR fees designed for this purpose may not instigate increases in re-use rates. For example, a case study of the EEE programme in the Midi-Pyrénées (France) found no growth in the reuse rate of electrical devices between 2007 (the first complete year of the programme) and 2011 (Bahers and Kim, 2018[11]).

Some countries have started to modulate EPR fees to better reflect other eco-design incentives (Laubinger et al., 2021[21]). For example, in Portugal a 10% penalty is applied to PET bottles with a PVC label or metal cap, and glass bottles with a stopper made of ceramic or steel (PRO Europe, 2021[22]). In France, fees are doubled for non-recyclable material (per national guidelines) and opaque PET with >4% mineral filler (CITEO, 2019[23]). Whilst these policies are relatively novel, some argue that the introduction of modulated fee setting can lead to stronger design incentives and significant design changes towards better recyclability of packaging.

Box 2.1. Fee modulation in EPR differentiates cost by differences in products

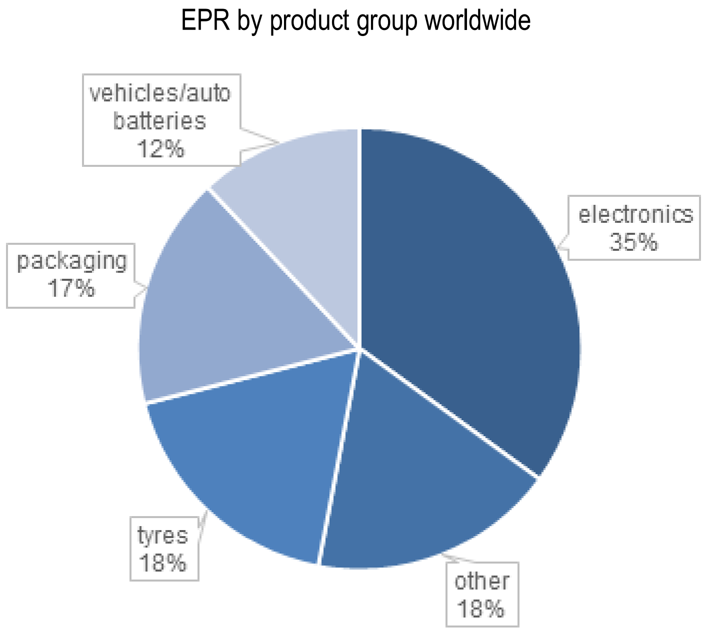
In collective implementation of EPR, the fee schedule set by Producer Responsibility Organisations (PROs) is typically quite basic. Fee differences are based on easily measurable EoL cost differences. The lack of a more granular cost differentiation provides relatively little incentive for design change by producers. A more granular and “advanced” fee modulation based on detailed product design criteria can provide producers with stronger design incentives, but can also lead to complications, such as an increased complexity and administrative burden and resulting costs (Laubinger et al., 2021[21]).

EPR has also faced some criticism from industry and other relevant stakeholders. For example, the initial push for the introduction of EPR (e.g. the German Packaging Ordinance of 1991) faced challenges from industry, principally in the form of concerns for cost and from municipalities that sought to maintain their influence on collection and sorting (Lindhqvist and Lifset, 1997[24]; Eichstädt, Carius and Kraemer, 1999[25]). Proponents of local public provision of waste management have argued that in the absence of guiding targets linked to EPR policies, such as for recycling or re-use of covered products, EPR can encourage incineration and crowd-out more circular local operations (Seldman, 2022[26]). EPR has also faced some criticism from recycling industry representatives. For example, in the United States, The National Waste & Recycling Association has argued that EPR fails to address what it views to be the core issues of insufficient demand for some recyclates, consumer behaviour, and the relative cost of recycled material (NWRA Staff, 2021[27]). The U.S.-based Institute of Scrap Recycling Industries only supports consideration of EPR policies that are temporary in nature until markets for recycling are mature (ISRI, 2021[28]).

3 Case Studies of EPR for new product groups

EPR adoption initially focused on packaging at the national level in Europe and the sub-national level in North America. In subsequent decades, adoption was instigated by policy at the EU-level on WEEE, batteries, accumulators, and vehicles. Indeed, as of 2013, EPR has been most widely adopted for electronics, tyres, packaging, and vehicle battery product groups, which together constituted roughly 82% of all EPR schemes (Figure 3.1).

Figure 3.1. EPR is widely used in several waste streams



Source: (Kaffine and O'Reilly, 2013[5])

The success of EPR in fostering recycling and ensuring funding for waste management in these sectors has instigated policy discussions about extending the use of EPR policy instruments to additional product groups. In the European Union, expansion of EPR to new product sectors is well underway. The EU Single-Use Plastics Directive will require member states to implement EPR schemes for several plastics products frequently found in litter surveys.

Construction and demolition waste (C&DW), food waste and textiles (products made of woven fabric, including garments, carpets, and mattresses) are waste streams that make up a significant portion of solid waste and currently exhibit relatively low rates of material recovery. Some first movers have implemented voluntary or mandatory EPR programmes for these waste streams. For instance, Italy and Spain have already adopted mandatory EPR programmes for cooking oils. France has an EPR in place for clothing and furniture, while several US states have EPRs for carpets and mattresses at the sub-national level.

The European Commission has placed the construction and buildings sector as a key value chain in the new Circular Economy Action Plan and is developing a strategy for a sustainable built environment (European Commission, 2020[29]). As well, the EU requires member states to implement separate collection of bio-waste by 2024 and textiles by 2025 (European Parliament and Council of the European Union, 2018[30]). The European Commission is proposing to introduce mandatory EPR for textiles in its member states (European Commission, 2023[31]). These developments will raise the question about the relevance and usefulness of EPR policies for these sectors.

At question is whether and under which conditions EPR is an appropriate policy approach for a product group as opposed to alternative policy approaches. With this question in mind, this chapter explores five sectoral case studies: tobacco product filters, fishing gear, construction and building materials, textiles, and food waste. Each case study provides a background on the current issues with the EoL management of the product sector and explores first experiences of EPR programmes in OECD member countries to address these issues. Each case study will conclude with an explanation of the merits of EPR in the given product sector compared with the benefits of EPR identified in the previous chapter. Chapter 5 and chapter 6 draw overarching conclusions from all case studies assessed.

3.1 Plastic products (beyond packaging)

Plastics are ubiquitous materials in modern economies, but their waste can cause serious environmental impacts. In 2019, 460 million tonnes of plastic were produced globally, of which only 9% was recycled at the end-of-life stage, whilst an estimated 22 million tonnes leaked into the environment (OECD, 2022[32]). By one recent estimate, the social costs of plastic-related pollution amount to hundreds of billions of dollars annually (Markl and Charles, 2022[33]). Proponents of EPR argue that it can be an attractive policy approach to address both of these aspects: to increase plastics recycling, as well as to reduce the amount of plastic being littered in the environment.

Whilst EPR programmes for packaging, a key plastics consuming sector, are widely used in the OECD, EPR for other plastic product sectors are less common. A range of durable plastics (e.g. polyvinyl chloride, polypropylene, and polystyrene) are used in non-packaging sectors, for example in construction, EEE, and households and leisure (European Environment Agency, 2022[34]). EPR programmes for some other big plastic consuming sectors are discussed in later sections, including textiles and construction. In addition to these sectors, governments are implementing EPR for several miscellaneous products composed of plastic to help increase recovery and recycling rates, including sanitary wet wipes, diapers, toys or sports equipment. Emerging EPR programmes for these product groups are discussed at the end of this section.

If EPR is to contribute more significantly to addressing plastics leakage, it will also need to be applied to product groups that are currently frequently being littered. Two such product groups include tobacco product filter and ghost fishing gear, for which the EU SUP Directive will require member states to adopt EPR programmes by 2023 and 2025 respectively. The current state of knowledge of EPR for both product groups is also discussed in this section.

3.1.1 Tobacco product filters

Background on the product sector and end-of-life issues

Tobacco product filters (a.k.a. cigarette butts) constitute a small waste stream but are one of the most commonly littered items. Euromonitor International estimates that approximately 5.2 trillion cigarettes were consumed in 2019, generating approximately 880 000 tonnes[1] of waste in cigarette butts (BAT, n.d.[35]).

[1] Waste generation estimate is based on 2019 consumption and average mass of a cigarette filter (0.12 ounces of filter per 20 cigarettes or roughly 0.17 grams per cigarette) (Register, 2000[219]).

Several studies estimate that 65% to 75% of all cigarette butts are littered (Rath et al., 2012[36]; Patel, Thomson and Wilson, 2012[37]). In a survey of 1 000 smokers, 74% of respondents reported having littered at least once in their life and 56% in the last month (Rath et al., 2012[36]).

Littered cigarette butts can cause environmental and public health impacts. Both the tipping paper and the filter are predominantly made of cellulose acetate, a synthetic plastic material. When littered, cigarette butts do not easily degrade in the environment; plastic filters can take 7.5 to 14 years to degrade in compost, and alternative cellulose filters can take 2.3 to 13 years (Joly and Coulis, 2018[38]; Bonanomi et al., 2015[39]). As well, toxic or carcinogenic chemicals captured in the used filter can leach out over time or disperse by filters breaking down into microplastics (Marinello et al., 2020[40]; Slaughter et al., 2011[41]; Booth, Gribben and Parkinson, 2015[42]).

Besides environmental and public health concerns, the clean-up of littered cigarette butts is costly. A simulation model of the total costs of cigarette waste estimates that the direct costs (including e.g. clean-up, street sweeping, capture devices, awareness campaigns, and impacts to human health) are roughly USD 15.68 per annum, per capita[2] in the largest cities of the United States (i.e. greater than 250 000 inhabitants) (Schneider et al., 2020[43]).

Collected tobacco waste is usually landfilled or incinerated and whilst there are some recycling technologies emerging, these remain niche and often not economically viable. Recycling options such as transforming tobacco waste ashes into clay bricks or insecticides are currently at laboratory or pilot scale but will require further examination before wider application (Marinello et al., 2020[40]; Murugan et al., 2017[44]). In Italy, Re-Cig has collected over 2.5 million cigarette butts as feedstock for recycling (Re-Cig, n.d.[45]). Other pilot models include voluntary support from industry to recycle plastic tobacco filters. For example, Unsmoke Canada and TerraCycle operate a nationwide voluntary cigarette recycling programme with 50 000 collection points and 1 500 receptacles. The collected waste is processed by Terracycle, who composts waste ash and recycles the plastic material in filters (Terracycle, n.d.[46]). The system has so far collected over 155 million cigarette butts (RBH Inc, 2021[47]). Santa Fe Natural Tobacco and Terracycle run a similar voluntary programme in the United States, that has collected an additional 100 million cigarette butts (CSP Daily News, 2012[48]; Reynolds, 2018[49])

Examples of EPR for tobacco product filters

In addition to the voluntary efforts in Canada, the United States, and Italy, the implementation of EPR for tobacco product filters is currently not widespread, but will be adopted by EU member states by 2025 (Table 3.2).

[2] The model adapts a previous estimate ($12.54) of the cost (of cigarette waste for communities in the West coast of the U.S. (Stickel, Jahn and Kier, 2012[216]).

Table 3.1. Examples of EPR for tobacco product filters

Country	Description
European Union (announced)	The EU Single-Use Plastics Directive will require member states to implement EPR for tobacco product filters by 2025 (EU Lex, 2019[50]).
France	Begun in 2021, it aims to reduce cigarette butt littering by 20% over the course of the first three years (Legifrance, 2021[51]).
Korea	An ADF (known as the waste charge) on difficult to recycle products, including cigarettes (Heo and Jung, 2014[52]).
San Francisco (United States)	The city' charges a cigarette litter abatement fee, an advance disposal fee, to cover it's clean up and awareness costs.
Terracycle (Unsmoke Canada and Santa Fe Natural Tobacco in the United States)	Voluntary collection and recycling programmes that have collected over 150 million tobacco product filters.

The city of San Francisco (United States) implemented a cigarette litter abatement fee in 2009, a kind of advance disposal fee. The law requires retailers to provide the city with a per-pack fee to cover the costs of litter clean up and awareness campaigns. The costs of the fee were initially based on: the 2009 litter clean-up cost accounting (estimated at USD 25 million, of which USD 5.6 million was attributed to cigarette waste); litter survey estimates from the regular street litter audit (22.5% of the count of all litter); purchase data (30.6 million packs); and projected costs for implementing awareness campaigns (USD 1.4 million annually) (Schneider et al., 2011[53]). The share (%) of cigarette waste in total abatement costs was updated to 53% in 2015 after another litter study was conducted in 2014. In 2019, the fee was increased to USD 1 per pack, then to USD 1.05 in 2022 (Office of the Treasurer & Tax Collector, n.d.[54]).

Policymakers are considering EPR for tobacco products to help reduce cigarette butt littering and shift some of the cost of the collection and clean-up of littered cigarette butts from the public to producers. For instance, an EPR scheme in France became effective in 2021[3], with the goal to reduce cigarette butt littering by 20% over the course of the first three years (Legifrance, 2021[51]).

Benefits and considerations of EPR implementation

Cost recovery

An ADF, as done in San Francisco's litter abatement fee, helps to achieve a source of funding for collection or clean-up of littered cigarette butts. In 2016, roughly 11.9 million cigarette packs were purchased in San Francisco. The city spent USD 23.5 million on litter management, of which cigarette litter was assumed to constitute between 22 and 53%[4], or USD 5.2 and 12.5 million, approximately USD 0.37 to 0.92 per cigarette pack[5]. The litter abatement fee in 2016 was USD 0.40 per pack, meaning the city recovered nearly USD 5 million to finance its litter clean-up programme (Rosenfield, 2016[55]).

Data limitations are likely to obfuscate initial fee setting. Policy would likely need to specify methods for determining litter prevention rates, collection rates, cleanliness indicators, cost allocation of clean-up costs between litter streams, and fee setting. Differences in methods for litter surveys can impact cost distribution. For example, cigarette butts constitute a greater share of count than their share of weight or volume in litter surveys when compared with other littered products such as plastic bottles (Darrah et al., 2021[56]). This is also evident in the range of estimates used in San Francisco based on the 2009 study (22%) and the 2014 survey (53%).

[3] EPR fees are introduced in a scaled approach and reduced by 50% for 2021 and 25% for 2022.

[4] Based on 2009 survey of 22% and a 2014 survey that identified 53%.

[5] Includes an adjustment of 13.8% for assumed migration of cigarettes purchased outside of the city and includes costs of litter abatement programme administration.

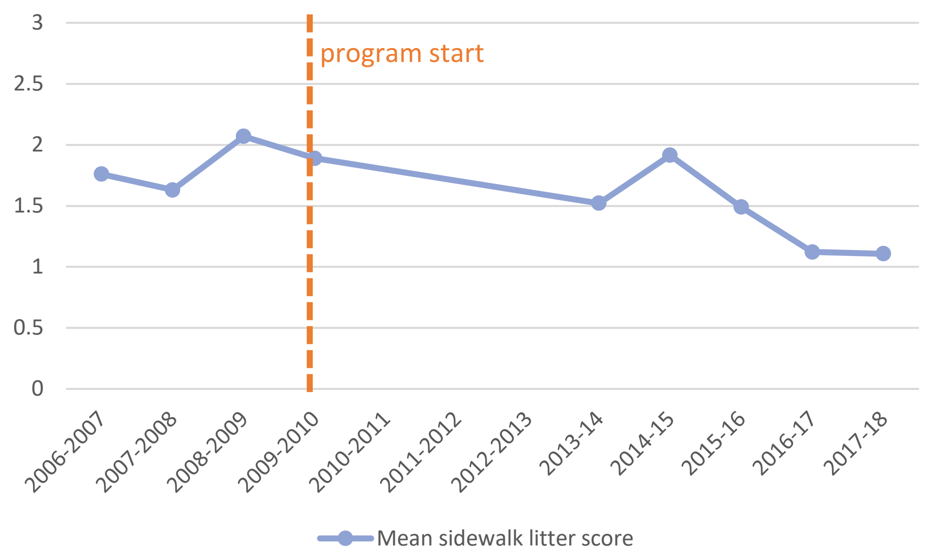
Separate collection

EPR policy instruments may help to influence littering and disposal behaviour of customers. This behaviour change can come from requirements for consumer awareness campaigns, provision of collection services and infrastructure, and incentives for proper disposal. Producer-driven awareness campaigns may be particularly effective for tobacco product groups because cigarette smokers exhibit a high level of brand loyalty (Cowie et al., 2013[57]). Proponents of EPR argue that targets for litter prevention or minimal provision of infrastructure can lead to improved access to disposal infrastructure (e.g. public ashtray dispensers) near littering hot spots such as public parks or streets with restaurants or bars.

After the introduction of its programme, San Francisco's cigarette consumption somewhat declined while both littering and street cleanliness evaluation scores showed signs of improvement from 2014 to 2018 (Figure 3.2) (Sabatini, 2017[58]; CSA City Performance, 2016[59]). However, it is possible that this relationship is confounded by other variables not considered in the scope of this paper.

Figure 3.2. Street cleanliness and littering scores are improving in San Francisco

Mean of all street littering scores in San Francisco street corridors by fiscal year



Note: Evaluation scores range from 1 (very clean- less than 5 pieces of litter per 100 curb-feet) to 3 (very dirty- over 15 pieces of litter per 100 curb feet examined)Mean is a non-weighted mean of all street corridors evaluated in each fiscal year. Litter defined as including cigarette butts, tissue paper, food wrappings, cups, plastic bags, newspapers, and loose gum .
Source: 2006 to 2010 (Stevenson et al., 2010[60]) and 2013 to 2018 (DataSF, 2019[61])

3.1.2 Commercial fishing gear and equipment

Background on the product sector and end-of-life issues

Fishing gear that is abandoned, lost, or otherwise discarded (hereafter called 'ALDFG') constitutes a significant share of marine plastic pollution. Estimates of the quantity of ALDFG are uncertain and differ by gear type, but estimates range from around 6% of nets to 29% of lines (Richardson, Hardesty and Wilcox, 2019[62]). Fishing and other marine activities contribute an estimated 3 megatonnes to macroplastics leakage (OECD, 2022[32]). Gear loss is driven by myriad reasons, and whilst some fishing gear is intentionally dumped or lost due to careless practices, a certain share is also lost unintentionally (Viool et al., 2018[63]).

ALDFG is an especially impactful type of plastic pollution on ecosystems, as it can entangle wildlife or damage habitats by settling on coral reefs or the marine floor. It also has economic impacts by affecting fishing and shipping activities and tourism (Lusher, Hollman and Mandoza-Hill, 2017[64]). Over time, ALDFG can degrade into microplastics and when particles are ingested by wildlife, hazardous substances contained in microplastic particles or adsorbed on their surface can enter the food chain (Vandermeersch et al., 2015[65]).

Besides worrying ecosystem impacts of ALDFG, material recovery of collected EoL fishing gear also remains limited and most EoL fishing gear is currently either landfilled or incinerated. Low volumes of separately collected EoL fishing gear and high transport costs of moving bulky material from dispersed ports to the few existing specialised recycling facilities currently pose economic barriers to large scale recycling (OSPAR Commission, 2020[66]). Additionally, fishers have previously identified inadequate or inconsistent waste services at ports (Nogueira et al., 2022[67]). For example, in 2016, only 1 514 or 4 443 registered ports in Norway had submitted a plan for waste reception and handling plan (The Kingdom of Norway, 2016[68]).

Nevertheless, recycling is technically feasible, as examples of voluntary EPR and product stewardship programmes show and after careful pre-sorting, recycled plastic can be used for a variety of non-food contact applications. However, gear manufacturers have previously expressed difficulty in marketing recycled gear due to perceptions of inferior quality (Nogueira et al., 2022[67]).

Examples of EPR for fishing gear

In some countries (e.g. Iceland and Norway), producers have set up voluntary responsibility schemes for EoL fishing gear, which show some success in collection and material recovery (Table 3.2). The EU Single-Use Plastics Directive will require member states to implement EPR for fishing gear by 2025. The cost implications for a mandatory EPR system for gear manufacturers, as will be introduced in the EU, can only be limitedly determined from the two existing voluntary schemes. In particular, the EU's implementing decision on monitoring and reporting will require the Member States to report on the amount of fishing gear containing plastics placed on the market, and by 2024 set minimum annual collection rates for recycling (Europen Commission, 2021[69]). In its May 2021 communiqué, the G7 Climate and Environment ministers committed to work to address ALDFG (DEFRA[70]).[6]

Table 3.2. Examples of EPR for fishing gear

Country	Description
European Union (announced)	The EU Single-Use Plastics Directive will require member states to implement EPR for fishing gear by 2025 (EU Lex, 2019[50]).
Norway (established)	Nofir is a voluntary EPR system based in Norway that collects EoL gear for recycling. The revenue generated by recycling pays for the operation costs. Nofir provides for separate collection at port facilities and collects directly from the gear owner upon request.[7]
Iceland (established)	The federation of Icelandic fishing vessel owners and fish processing plants (SFS) established a voluntary EPR system to pre-empt a national advance disposal fee policy on fishing gear. Vessel owners clean and prepare gear and pay for transportation costs to a collection centre. The costs (roughly 85-110 euros/tonne) are equivalent to disposal fees for landfill or incineration (Jauke van Nijen, 2021[71]).
Sweden	Sets national collection target of 30% of the weight of gear placed on the market, beginning in 2027 (Landbell group, n.d.[72]).

[6] For more information, see (OECD, 2021[215]).

[7] The government of Norway does not consider the programme to be an EPR because the gear manufacturers are not responsible for the costs of the programme.

Benefits and considerations of EPR implementation

Cost recovery

Proponents of mandatory EPR for fishing gear argue that shifting the cost burden of waste management from small ports and fishing operators to gear manufacturers, can help finance separate collection and treatment and increase material recovery. In Iceland, vessel owners are responsible for the costs to clean and prepare gear and pay for transportation costs to a collection centre. The costs (roughly 85 to 110 EUR/tonne) are equivalent to disposal fees for landfill or incineration (Jauke van Nijen, 2021[71]). However, there is limited evidence about the costs that could be recovered from gear manufacturers from existing EPR schemes.

Sales from the secondary material can help fund collection and recycling programmes. Nofir, the voluntary EPR in Norway, has initially received an EU innovation grant, but is now largely self-funded through secondary material sales, requiring only limited contributions from industry (OSPAR Commission, 2020[73]). Accessible and affordable separate collection in Iceland eases the costs of compliance for vessel owners to participate in the scheme. However, both schemes cover only a selection of ports and collection costs would likely increase in a mandatory system that requires covering all ports.

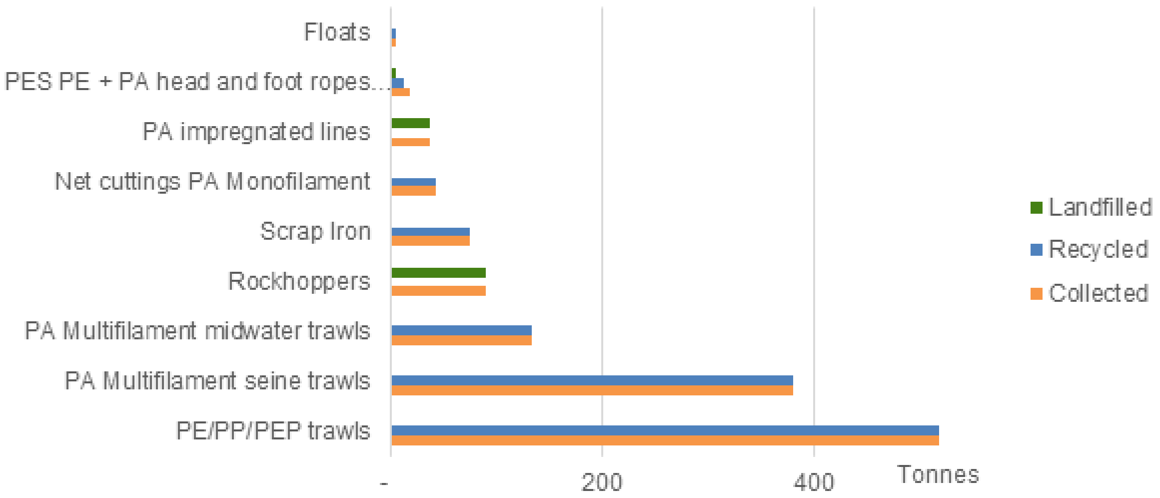
Separate collection

Proponents of EPR for fishing gear argue that the approach could instigate improved rates of separate collection of EoL gear. They argue that EPR could facilitate the development of accessible separate collection to ease participation in collection schemes for vessel owners. For example, Iceland's EPR operators reported collecting 1 297 tonnes of EoL gear in 2016 (Fisheries Iceland, 2017[74]).

Increases to recycling rates

There is some evidence that EPR can help with instigating recycling of fishing gear. For example, operators in Iceland estimate that roughly 90% of collected gear was recycled in 2016, the remaining 9% was landfilled (Figure 3.3). Recyclability appeared to vary by product, because rockhopper rubber discs and polyamide (PA) impregnated lines were both entirely landfilled, while just two-thirds of ropes and cables were reportedly recycled. Some equipment contains metallic parts that complicate sorting and recycling (Nogueira et al., 2022[67]).

Figure 3.3. End-of-life gear outcomes estimated by Fisheries Iceland in 2016



Note: Polymer acronyms include (PES) polyether sulfone, (PE) polyethylene, (PA) polyamide, and (PP) polypropylene.
Source: (Fisheries Iceland, 2017[74]).

Potential additional benefits

In addition to improved collection and recycling, proponents have argued that introduction of EPR policies for EoL fishing gear may also lead to improved designs. One frequent argument is that EPR fees, especially ones that are calibrated to loss rates, can provide incentives for manufacturers to design gear that is less vulnerable to loss during use and more recyclable or repairable (Jauke van Nijen, 2021[71]).

Data on the EoL fate of fishing gear is a remaining knowledge gap. The UN GESAMP notes that plastic pollution from ocean-based activities has not been rigorously quantified, and the burden remains poorly understood (2020[75]). Proponents of EPR for EoL fishing gear suggest that data collection and reporting in fulfilment of EPR obligations can help to improve information on the manufacture, collection and the EoL fate of fishing gear.

EPR could also potentially help extend the life of fishing gear if it included targets for collection and repair for re-use. The EPR programme could target collection of used gear that's worn out, but not yet broken.

3.1.3 Miscellaneous plastic products (beyond packaging)

Several OECD countries have announced plans to introduce EPR programmes to cover products that are made with plastics outside of the packaging sector (Table 3.3). EPR programmes do not usually specify a particular material, but rather a product category. The EU will require its member countries to introduce EPR as part of a collective effort to reduce the environmental impacts of single-use plastics commonly found in litter surveys, including balloons and sanitary wipes. France has recently (in 2022) begun to implement EPR for toys and sports equipment. Many of these programmes are either recently implemented or only announced at this point, meaning that it is too early to determine their effectiveness, but this could be a point of later research.

Sanitary products, including single-use diapers, have been the focus of several new policies. The EU will require its members to adopt EPR for sanitary wipes, Korea has an ADF in place for diapers, and the Netherlands plans to introduce an EPR programme for diapers in 2023. Recycling facilities for diapers are somewhat rare; facilities in England, Italy, and the Netherlands have a combined annual capacity to recycle 0.36 megatonnes, mostly to lower value purposes like construction aggregate and cat litter (Płotka-Wasylka et al., 2022[76]). Proponents of EPR argue that separate collection of this waste steam could facilitate recycling and provide a business case for design change to ease recovery.

Improper disposal of sanitary products can obstruct waterways when improperly. In a 2017 survey sample of sewer pipe blockage in the United Kingdom, single-use wipes composed roughly 93% by weight (Drinkwater and Moy, 2017[77]).

Table 3.3. Examples of EPR for miscellaneous plastics product (beyond packaging)

Market or country	Description
Korea	An ADF (known as the waste charge) on difficult to recycle products, including **disposable diapers PVC pipe toys, and kitchenware** (Heo and Jung, 2014[52]).
France (announced, some implementation in 2022)	The law against wastage and for a circular economy required the introduction of five new EPR programmes, including **toys, sports and leisure articles, and home improvement and gardening** (Institut National de l'economie circulaire, n.d.[78]),
European Union (announced)	The EU Single-Use Plastics Directive will require member states to implement EPR for **balloons** and **sanitary wipes** (EU Lex, 2019[50]).
The Netherlands (intention announced)	The Netherlands plans to introduce an EPR for **diapers** in 2023 or 2024. A previous study on the recyclability reviewed policy options (TAUW, 2021[79]) .

3.2 Textiles (garments, carpets, and mattresses)

3.2.1 Background on the product sector and end-of-life issues

The textiles product sector includes products composed of any fibre-based materials. These can include woven fabrics in clothing and footwear, carpets, furniture, and sheets and towels. This sub-chapter will focus on garments, carpets, and mattresses, which all form part of the textiles waste stream and for which some EPR programmes are in place or under discussion in OECD member countries.

Box 3.1. "Fast fashion" has resulted in more waste of garments

Over the past three decades, a combination of reduced prices and increased purchases resulted in shorter use phases for clothing and the emergence of "fast fashion". In the EU-27 and the United Kingdom, the real price of clothing dropped by 36% between 1996 and 2012. Whilst overall spending on clothing increased by 40% over this period, the share of household expenditures on clothing slightly declined, due to the decline in the relative price of clothing compared with other household expenses (Reichel et al., 2014[80]). In the United Kingdom garments are estimated to constitute 51% of textiles consumption (WRAP, 2014[81]). In the United States in 2018, clothing and footwear constituted roughly 76% of estimated household textiles waste (US EPA, n.d.[82]).

Textiles constitute a significant and growing share of municipal waste streams. The United States annual textile waste generation increased more than five times from 8.9 kg per capita in 1960 to 47 kg per capita in 2018. Clothing and footwear waste generation (a subset of textiles) grew from roughly 6.8 kg per capita in 1960 to 36 kg per capita in 2018[8] (US EPA, n.d.[82]). In 2018 in the EU-27 consumption of clothing and household textiles was 12.3 kg/capita, an increase of 20% compared with consumption in 2003 (European Environment Agency, 2021[83]). In Japan in 2020, 1.7 million metric tonnes (Mt) of fibre products are discarded annually, roughly 13.5 kg per capita[9].

[8] The U.S. generated 1.23 Mt in 1960 and 11.76 Mt in 2018 (U..S. EPA, n.d.[223]) with a population of 180.7 million in 1960 and 326.8 million in 2018 (The World Bank, 2020[220]).

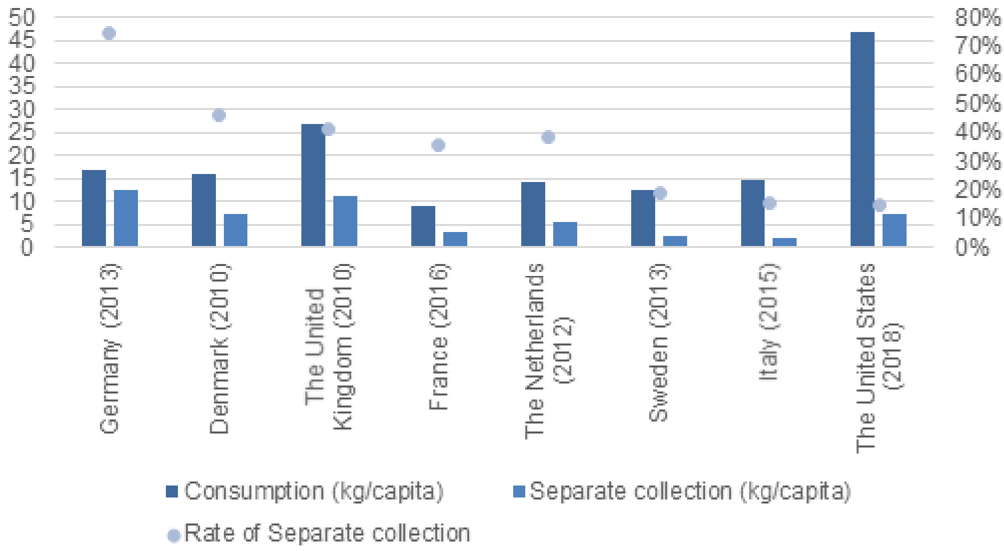
[9] 1.7 Mt of fibre products are discarded annually (TAKASHI/All, 2020[221]) with a 2020 population of 125.8 million in 2020 (World Bank, n.d.[222]).

Management of this waste stream is costly. In the United Kingdom alone, annual management of clothing and household textiles amounts to an estimated 82 million GBP per year (WRAP, 2014[81]).

Separate collection is essential for maintaining the value of household textiles. The share of textiles that are separately collected varies from an estimated 20% in Italy to more than 75% in Germany (Figure 3.4). Charities play a significant role in accepting donations of textiles as a form of separate collection, but also some retailers operate take-back operations, and in some countries the public sector provides separate collection. Approximately one-third of textiles put on the European market each year is collected separately (between 1.6 and 2.5 Mt) (European Environment Agency, 2021[83]). In the United States, only an estimated 15% is collected separately (Adler, 2020[84]).

Figure 3.4. Rates of separate collection of clothing and household textiles vary by market

Per capita clothing consumption and separate collection (kg) of select OECD countries



Source : (Watson et al., 2018[85]; Adler, 2020[84])

Re-use and recycling[10] that replaces virgin production can help to significantly reduce the environmental footprint of the textile industry. Production of textiles requires land, water, and energy for growing of raw materials (e.g. cotton), production and transport. Proponents of textiles recycling argue that reuse and recycling of textile fibres could reduce demand for textile production and associated material footprints (Watson et al., 2016[86]; Semba et al., 2020[87]).[11]

Some textiles are in good condition at the point of EoL collection and may be re-used. In Europe, an estimated 50 to 75% of textiles collected are intended for reuse (European Environment Agency, 2021[83]).

[10] The environmental benefits of textiles recycling are mixed. Re-use is more environmentally beneficial than recycling (Sandin and Peters, 2018[217]). Recycling textiles to fabric, instead of fibre, can reduce the impacts of the additional step of creating fabric from fibres. Recycling to fibre with energy intensive processes can have negligible environmental benefits (UN Environment Programme, 2020[218]).

[11] Results of LCA studies are typically driven by the inclusion of a subsequent drop in primary resource consumption as part of the evaluation of recycling and re-use of textiles (Sandin and Peters, 2018[217]). As such, policies should seek to create a coherent set of incentives that not only drive recycling or re-use at end of life, but also encourage a shift in production to use of secondary materials or business models for re-use.

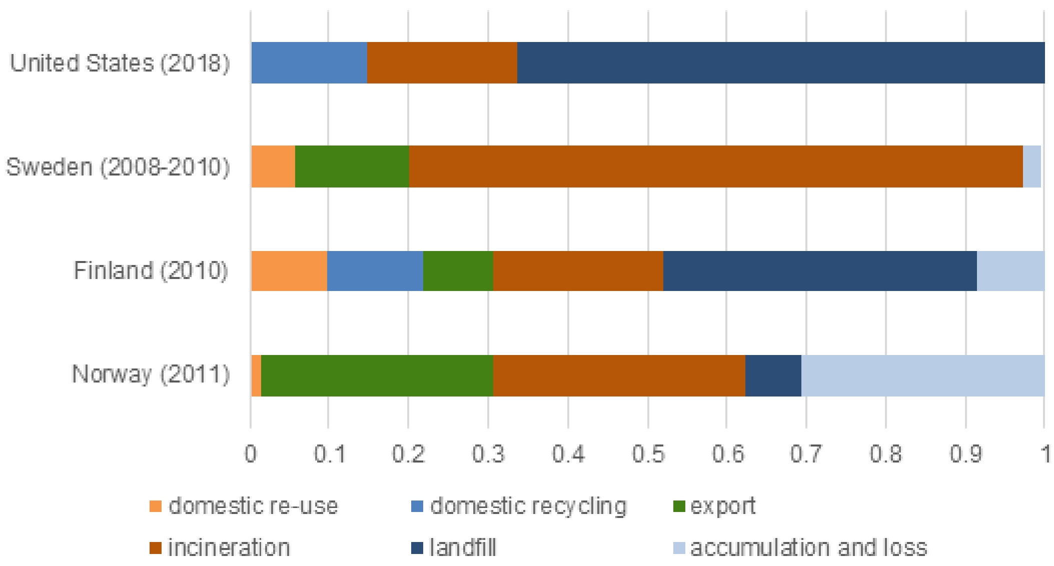
Only a very limited share of EoL textiles is currently being recycled and most recycling is for lower grade products (e.g. filling materials). The Ellen MacArthur Foundation estimated that globally 12% of EoL garments are downcycled to a less valuable use, and less than 1% is recycled to make new fibres for a textile of similar value (2017[88]). In the United States in 2018, an estimated 14.7% was recycled, mostly for use in upholstery, automotive stuffing or rags (US EPA, n.d.[82]).

There are significant barriers to enhanced circularity of textiles. EoL textiles are often multi-material fibres, requiring difficult and costly separation. Identifying and sorting materials in used textiles is difficult, especially for cellulose-polyester mixtures. Shortened fibres in recycled textiles further reduce their applicability. For these reasons there are still few mechanical recycling options for textiles (Damayanti et al., 2021[89]). Chemical and biological recycling technologies may overcome some of these barriers but remain niche or in pilot stages (Ribul et al., 2021[90]).

Incineration and landfilling of EoL textiles therefore remain prevalent. The Ellen MacArthur Foundation estimated that globally roughly 73% of EoL clothing garment waste is landfilled or incinerated (2017[88]). In Europe, an estimated 16 to 33% of collected textiles ultimately goes to mixed municipal waste and is incinerated or landfilled (European Environment Agency, 2021[83])[12]. In the United States in 2018, an estimated 18.9% was incinerated and 66% was landfilled (US EPA, n.d.[82]).

Figure 3.5 shows available estimates for Nordic countries and the United States. Noticeable is the significant proportion of EoL textiles that is landfilled, incinerated, or exported, whilst domestic recycling and re-use are relatively limited.

Figure 3.5. Fate of textiles in select countries



Note: United States data is the share by treatment type of total generation (no reuse or accumulation), Sweden, Finland, Denmark, and Norway data based on total production. Accumulation and loss represent the share of textiles generated, but not collected at end of life. United States' data does not include accumulation and loss.
Source: (US EPA, n.d.[82]; Watson et al., 2016[86]) United States data drawn from data from the American Apparel and Footwear Association, the International Trade Commission, the U.S. Department of Commerce's Office of Textiles and Apparel, and the Council for Textile Recycling.

[12] There is no requirement to report volumes of separately collected textiles, and no standard definition for whether used textiles are considered waste or not (European Environment Agency, 2021[83]).

A portion of the collected EoL garments is exported for reuse by OECD countries. Whilst trade in used textiles helps to increase their life span, imports of second-hand textiles compete with domestic textiles production (Baden and Barber, 2005[91]) and importing countries are left with the cost of managing imported textiles once they become waste (see 4.3). In response, some African countries that have been large recipients of second-hand textiles have introduced high import taxes on second-hand textiles (Anami, 2022[92]).

3.2.2 Examples of EPR for textiles

EPR schemes for textiles are starting to emerge with some success (Table 3.6). There is presently EPR for garments in France, and for carpets and mattresses at the sub-national level in the United States.

Additionally, several markets are considering whether to adopt EPR measures for textiles. At EU level, the European Commission is treating the textiles sector as a key value chain in the new Circular Economy Action Plan (European Commission, 2020[29]). The EU strategy for sustainable and circular textiles suggests that EPR has the potential to incentivise producers to reduce textile waste and increase rates of reuse and recycling. The EU Commission will propose harmonised EU EPR rules for textiles with modulated fees (European Commission, 2022[93]). Article 11(1) of the EU's Waste Framework Directive requires Member States to establish separate collection of textile waste by 2025 (European Parliament and Council of the European Union, 2018[30]). Considering this obligation, several Member States have already agreed on or are considering the introduction of EPR requirements for textiles: the Netherlands has announced an EPR programme for clothing and household textiles, Sweden plans to introduce an EPR for textiles. Additionally, Italy and England are considering an EPR for textiles waste. The European Commission is proposing to introduce mandatory EPR for textiles in its member states (European Commission, 2023[31])

Table 3.4. Examples of EPR for textiles

Country	Description	Status	Mandatory or voluntary
Australia	Koala mattress company voluntarily works with the collection service Soft Landing to provide a collection service for **EoL mattresses** at no cost to customers (Eunomia, 2020[94]).	Established	Voluntary
Belgium	An EPR scheme for **mattresses** in which mattress retailers are compensated for accepting EoL mattresses from customers upon purchase of a new replacement mattress. Producers are required to provide an environmental contribution fee (Valumat, n.d.[95]).	Established	Mandatory
France	An EPR scheme for **clothing, shoes, and household linens** began in 2007. The scheme was expanded in 2020 to include curtains (Transition, 2020[96]).	Established	Mandatory
The Netherlands	The five largest mattress producers established a voluntary EPR 'stichting matras recycling Nederland' (MRN). Participation in the **mattress** EPR scheme became mandatory after a decision of general applicability (Rijkswaterstaat, n.d.[97]). The PRO sets its own targets for recycling.	Established	Mandatory
The Netherlands	Will introduce an EPR for newly manufactured **clothing, table linen, bed linen, and household linen**. Sets targets for a share (by weight) of material put on the market the previous year to be prepared for reuse or recycled, starting at 50% by 2025 increasing to 75% by 2030 (Staatscourant, 2022[98]).	Forthcoming (2023)	Mandatory
United States (California, Connecticut, Rhode Island, Oregon)	California, Connecticut, Rhode Island, and Oregon have EPR recycling programmes for **mattresses**. Retailers collect an ADF at point of purchase that fund mattress collection and recycling programme (Bye Bye Mattress, n.d.[99])	Established	Mandatory
United States (California)	An EPR for **residential and commercial carpets** began in 2011, with a recycling rate goal of 24% by 2020. PROs or individual producers develop stewardship plans for submission to CalRecycle. An advisory committee appointed by CalRecycle works with PROs and individual producers on plans and annual reports. PROs are left to determine their own funding mechanism but must include modulated fees that consider design elements such as recycled content (CalRecycle, 2021[100]).	Established	Mandatory
United States (New York)	An EPR for **residential and commercial carpets** to be enacted in 2022. It will require producers to submit a plan to the state for a carpet collection programme by 2024. (The New York State Senate, 2022[101]).	Forthcoming (2022)	Mandatory
United Kingdom	Carpet Recycling UK is s voluntary scheme for **carpets** sponsored by large carpet companies (Eunomia, 2020[94])..	Established	Voluntary
Australia	The Australian Fashion Council is establishing a **clothing textile** waste stewardship programme by 2023. The government of Australia is providing a 1 million AUSD grant to the project (DAWE, 2021[102]).	Forthcoming (2023)	Voluntary
Sweden	Plans to establish an EPR as of 2022 to comply with the EU WFD requirement for separate collection of textile waste. The scope remains unclear.	Forthcoming (2022)	Mandatory

3.2.3 Benefits and considerations of EPR implementation

Cost recovery

EPR for several types of textiles have successfully generated revenues for EoL treatment. In terms of financing, in 2020 the French PRO provided roughly EUR 17 million for sorting, EUR 4.1 million in funding local authorities for collection, and EUR 0.7 million for research and development (Refashion, 2021[103]). In 2020, California's carpet EPR brought in USD 23.4 million in income from assessments, nearly covering its USD 24.4 million in expenses (CARE, 2021[104]). In 2021, California generated over USD 46 million in recycling charges from mattress retailers (Mattress Recycling Council, 2022[105]). The Belgian mattress PRO Valumat generated roughly EUR 7.8 million in fees in 2021.

Separate collection

Markets with existing EPRs for textiles have exhibited relatively high rates of separate collection of the waste stream. In France, the operation of the garments PRO *Re_fashion* has coincided with increased separate collection and relatively high rates of re-use and recovery. The weight of separately collected textiles per capita grew from 2 kg in 2009 to 3.7 kg in 2019 (Refashion, 2021[103]). The introduction of a mattress EPR in Connecticut coincided with a significant boost in separate collection. In the first-year collection rose from 8.7% to 63.5% (PSI, n.d.[106]). The California programme has collected 8.5 million mattresses since 2016 (Mattress Recycling Council, n.d.[107]). Rhode Island's mattress EPR programme collected 104 607 mattresses and provided access points for take back in 37 or 39 of the state's municipalities (Mattress Recycling Council Rhode Island, 2021[108]). Connecticut's programme collected 213 543 mattresses and provided tack back access points in 136 of 140 of the state's municipalities (Mattress Recycling Council Connecticut, 2021[109]). In 2021, Valumat collected 8 911 tonnes of mattresses, roughly 60% the total weight of mattresses placed in the market, well outperforming its target of 30%.

Proponents of EPR policies argue that they can help to improve collection and the circularity of textiles. They argue that mandatory take back requirements and collection targets can lead to development of a more comprehensive collection system, which would allow for new opportunities for reuse and targeted recycling. For example, California's carpet EPR programme has helped to fund a collection infrastructure throughout the state (CARE, 2021[104]).

The Nordic co-operation (Norden) commissioned a comparison of three possible policy packages to spur recycling and re-use of textiles in its member countries. The options included a mandatory EPR scheme with a tax on hazardous substances, a voluntary scheme with a raw materials fee, and a mix of policies to support new business models (e.g. leasing, repair, second-hand sales). The report suggests that EPR schemes would have a significant impact on collection and recycling of textiles and could lead to the creation of green jobs in collection, re-use, and recycling. However, support for new business models was projected to be more likely to impact pre-consumer stages of the lifecycle, encourage re-use, and create a greater share of domestic-based jobs (Ekvall et al., 2015[110]).

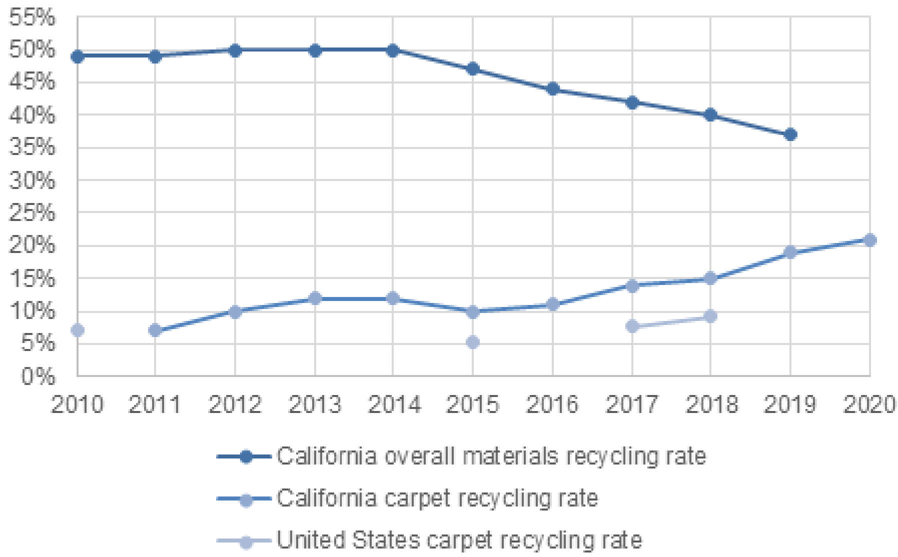
Increases to re-use and recycling rates

In France, the operation of the garments PRO *Re_fashion* has coincided with relatively high rates of re-use and recovery. The re-use rate of collected textiles is roughly 58% (Refashion, 2021[103]). From its introduction in 2009 to 2019, the share of collected garments used as material for garnetting (recycling) grew from 14% to 23%, however energy recovery also grew from nothing to 8.2% (Refashion, 2021[103]).

California's carpet stewardship programme achieved an annual recycling rate of 21% in 2020. This is below its target of 24%, but the recycling rate has continuously improved over the life of the programme (Figure 3.6). Over the same period, the overall materials recycling rate in California has decreased, suggesting that the EPR programme has helped this waste stream to buck the overall trend of declining recycling rates. The recycling rate in California has also significantly outperformed the national carpet recycling rate of 9.2% of carpet generation in 2018 (US EPA, n.d.[111]). The recycling yield (share recycled from collected material) has also grown from a historical average of 38% to 68% in 2020, indicating a growth in recyclability of textiles.

Figure 3.6. The recycling rate of carpets in California has grown over the life of its EPR programme

Comparison of California's overall materials recycling rate (%) and the carpet recycling rate (%)



Source: California data (CARE, 2021[104]) and overall United States data (US EPA, n.d.[111])

The voluntary carpet EPR in the United Kingdom coincided with an increase in the landfill diversion rate from 2% in 2007 to 70% in 2017 (Carpet Recycling UK, n.d.[112]). However, most (73%) of this recovery is waste-to-energy, 1% is for reuse, and 4% for fibre-to-fibre recycling (Eunomia, 2020[94]).

California, Rhode Island, and Connecticut have an ADF and take-back programme in place for mattress recycling (Oregon will begin one in 2024). The programme has thus far recycled more than 10 million mattresses (Mattress Recycling Council, n.d.[107]). In California, recyclers diverted roughly 77% by weight of materials for recycling, reuse, and biomass (Mattress Recycling Council, 2022[105]). Rhode Island recycled roughly 60% of collected material in its 2020-2021 reporting period (Mattress Recycling Council Rhode Island, 2021[108]). Connecticut recycled 73% of collected materials in the same period (Mattress Recycling Council Connecticut, 2021[109]). These rates compare quite impressively with the national estimate: 99.6% of furniture and furnishing (the category containing mattresses) were combusted or landfilled in 2018 (US EPA, n.d.[113]). In 2021, Valumat placed 124 716 collected mattresses for re-use, recycled 1 997 tonnes of material, sent 6 337 tonnes for energy valorisation and did not report landfilling any material in line with Belgian waste policy. The Dutch mattress PRO MRN has set itself a recycling target of 75% by 2028, though no benchmark data is available yet (MRN, n.d.[114]).

Legacy substances can limit the recyclability of EoL textiles. Old carpets and mattresses may contain persistent organic pollutants (PoPs), which should not re-enter the market in secondary content. Recycling efforts and EPR programmes should emphasise safety and work to ensure material with such PoPs are properly treated at end of life (Onyshko and Hewlett, 2018[115]).

Proponents of EPR for textiles argue that mandatory reuse, recycling, or recycled content targets in EPR policy can help to incentivise the development and roll-out of a recycling infrastructure for EoL textiles. For example, France is working to increase lifespans of textiles within the context of EPR by encouraging design for durability and re-use. A 75% reduction of EPR fees is given to clothing, home textiles, and footwear that meet certain durability design requirements (ECO-TLC, 2019[116]). As well, 5% of collected EPR fees are dedicated to support social enterprises that facilitate re-use and preparation of textiles for re-use ("Fonds pour le Réemploi Solidaire") (RReuse, 2020[117]).

3.3 Construction sector

3.3.1 Background on the product sector and end-of-life issues

This sub-chapter will consider the application of two complementary waste streams within the construction sector: materials such as residual and unused specific building material and the built environment such as construction and demolition waste (C&DW) of an entire building. The scope of the analysis on building material is limited to waste streams with an EPR programme in place in an OECD country, notably paint and flat glass windows, but insights from these schemes could potentially also be applicable to developing EPRs for other building material waste streams.

Built environment

The construction of buildings for homes, offices, commercial activities and social interaction has an enormous material and energy footprint. Global building material use is projected to double between 2017 and 2060 (OECD, 2019[118]). Globally, the construction industry accounts for roughly 30% of natural resource extraction and 25% of solid waste generation (Benachio, Freitas and Tavares, 2020[119]).

The production of new building material can be energy-intensive and generates greenhouse gas emissions, making a case for reuse and recycling. For example, in 2012 the production of concrete alone was estimated to contribute 8.6% of anthropogenic carbon dioxide emissions (Miller, Horvath and Monteiro, 2016[120]).

To date, the majority of construction and demolition waste (C&DW) is discarded or downcycled, in part due to limitations to deconstruction of buildings in a modular way to reuse individual parts. In 2018, the United States generated 544 Mt of C&DW, of which 24% was sent to landfills and 52% was used as aggregate (US EPA, 2020[121]). In 2016, across the EU-27 and the United Kingdom, C&DW was the largest single waste stream (374 Mt), while 11% was landfilled and recycling was largely low-grade recovery and aggregates used for example in backfilling (EEA, 2020[122]). The sheer volume of C&DW means it absorbs substantial space in landfills and increases the costs of tipping fees (US EPA, 2020[121]).

The disposal of C&DW creates significant costs, which can also incentivise improper disposal. In France, the removal and clean-up of illegal dumps of C&DW is estimated to cost between EUR 340 and 420 million each year (Ministère de la Transition Écologique, 2020[123]).

Building material

Residual, unused building materials can be a source of harmful constituents. For example, paints can contain volatile organic compounds and be toxic, flammable, and reactive, justifying their separate collection and disposal or specialised recycling, in particular oil and latex paints (Inglezakis and Moustakas, 2015[124]).

Separate collection or end-of-life building materials can also allow higher value recycling of some building materials, which can only limitedly be recovered when comingled in general C&DW, such as flat glass. The European flat glass trade association notes that despite a high level of recyclability, flat glass is typically crushed after comingling with other building materials and ultimately disposed of in landfill or incineration (Glass for Europe, n.d.[125]).

3.3.2 Examples of EPR for the construction sector

Several markets have adopted EPR policies for C&DW that aim to increase recovery of these materials at end of life at brownfield construction sites (Table 3.5). Demolition and construction firms are traditionally responsible for the physical and financial management of the waste generated by their work, akin to a form of individual producer responsibility. To advance material recovery ambitions, the public sector can set targets and incentives for these operations. For example, the California Green Building Standards Code (CALGreen) sets minimum recovery requirements for building and demolition projects. Several municipalities in California have also established a form of deposit-refund system in which firms pay a deposit when applying for a construction or demolition permit and receive a refund upon demonstration to have exceeded a certain recovery rate threshold of waste or debris.

Product stewardship programmes can also help with re-use and safe disposal of unused building materials like paint. Producers of the building material pay an advance disposal fee that covers separate collection for use or EoL treatment of this material. Examples include paint care in the United States and flat glass in the Netherlands

Table 3.5. Examples EPR schemes for C&DW and building material.

Country	Description
Building material	
France	An EPR scheme for marketers of construction products and materials for the building sector became effective in 2022. The measure will expand existing collection points that freely take back waste building materials from professionals and establish schemes for waste recovery from craftsmen and private individuals (Ministère de la Transition Écologique, 2020[123]).
Paintcare (United States)	Ten[13] US states and Washington D.C. have mandatory EPR programmes for architectural waste paints. The PRO provides drop-off collection sites (e.g., at paint retailers). Materials are then sorted and processed for recovery or disposal. Firms pass some the costs of the system on to consumers and customers pay an eco-fee of USD 0.35-1.99 depending on the size of the paint container purchased (PaintCare, n.d.[126]; PaintCare, n.d.[127]).
The Netherlands	A general binding agreement (Algemeen Verbindend Verklaring or AVV) for flat (insulation) glass. The programme started on a voluntary basis and is now a binding financial contribution for post-consumer collection, sorting and treatment (Dimitropoulos, Tijm and In 't Veld, 2021[128]). The Netherlands implemented a C&DW landfill ban in 1997. (Zhang et al., 2020[129]).
C&DW	
CALGreen California (United States)	CALGreen began in 2011 and requires a 65% recovery rate for non-hazardous construction and demolition waste. Firms provide a waste management plan prior to starting a demolition project that specifies materials to be diverted, diversion facilities, methods used to reduce waste generation (Government of British Columbia, n.d.[130]; CalRecycle, 2020[131]). In several municipalities in California, firms pay a deposit and receive a refund upon demonstration that the recovery rate was met. For example, this requirement exists in: – Santa Monica (70% recovery rate target) (Santa Monica Public Works, n.d.[132]); – San Diego, (65% recovery rate target) (City of San Diego, n.d.[133]); and – San Jose, (75 recovery rate target) (City of San Jose, n.d.[134]; Government of British Columbia, n.d.[135]).
Maine, Massachusetts, and Vermont (United States)	A disposal ban on asphalt pavement, brick, concrete, metal and wood, materials that were determined as having viable recycling markets (Mass DEP, 2020[136]).Massachusetts has banned disposal, combustion, and transfer of C&DW of asphalt pavement, brick, and concrete, metal, wood, and clean gypsum, Vermont has banned landfilling of S&DW from sites generating more than 40 cubic yards (0.76 cubic meters) and located within 20 miles (32 km) of a recycling facility (US EPA, 2017[137]).
Portland, Oregon and Lee County in Florida (Unite States)	Portland's demolition permits require deconstruction of historic buildings (built prior to 1916), which coincided with an increase in deconstruction from 10 to 33% of single-family homes. (US EPA, 2017[137]). Lee County charges an advance disposal fee on projects greater than 1 000 square feet (92 square meters)

[13] In July 2023, Illinois adopted a law on EPR for paint and will become the eleventh state to join the program.

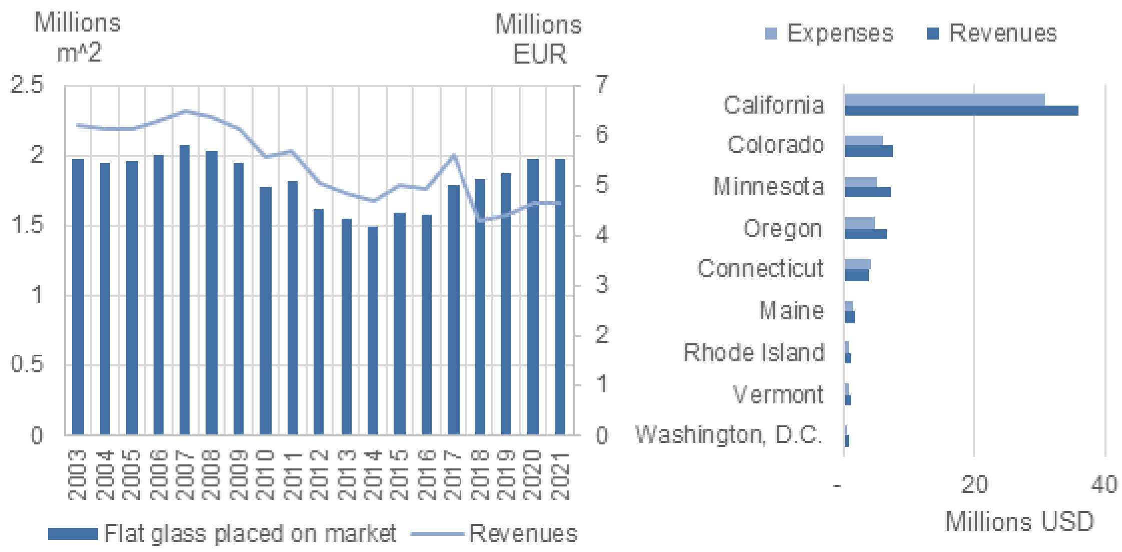
3.3.3 Benefits and considerations of EPR implementation

Cost recovery

EPR for building materials helps to collect fees from producers to support the sperate collection and treatment of these materials at EoL. For example, Vlakglass Recycling Netherlands recouped an estimated EUR 1.6 million in producer fees to finance recycling of the 5.5 million meters squared of glass sold in 2021 (Vlakglass Recycling Nederland, 2022[138]). Since 2003, the programme has generated over EUR 36 million from producer fees to finance collection and recycling of EoL insulation glass (Figure 3.7). Eight paintcare states and Washington D.C. collected nearly USD 65 million in producer fees in 2020, covering the USD 53 million in expenses for collection and disposal in these states.

Figure 3.7. Fee collection by building material EPR schemes

Historical fee collection of flat glass in the Netherlands (left) and revenue and expenses of paintcare states in 2020 (right)



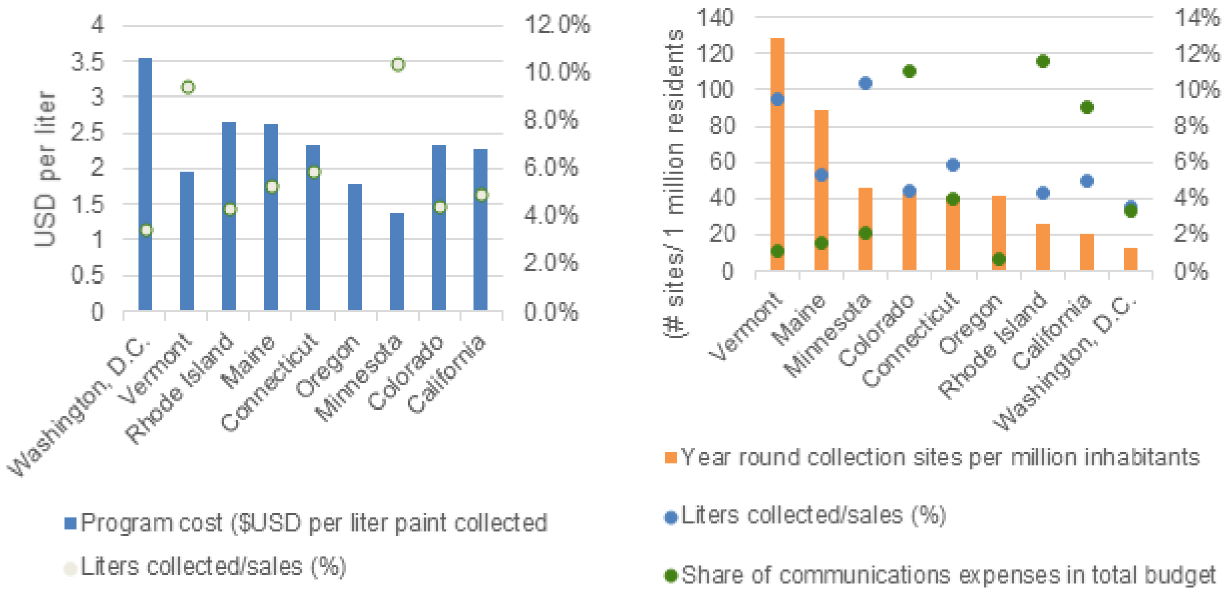
Note: (right) Data collected from annual reports by state. All data from fiscal year 2020 annual reports. PROs in paintcare states are not-for-profit organisations and differences in revenues and expenses are used as reserves to absorb future price fluctuations.
Source: left (Vlakglass Recycling Nederland, 2022[138]), right (PaintCare, n.d.[127])

Proponents of EPR for residual building materials, such as excess material, cut offs, or unused material, argue that EPR can instigate some waste reduction. Compared with an alternative collection mechanism, a take back approach could incentivise the private sector to adjust product sizing and recovery.

Separate collection

EPR for building materials can help to facilitate separate collection of harmful constituents and reduce the presence of harmful materials in general waste. The paint stewardship programme in the United States has collected over 268.2 million litres of paint. Doing so has helped to divert this potentially hazardous material from general waste. Paintcare has achieved collection rates that range from 3.5% in Washington, D.C. to 10.4% of the volume of paint sold in Minnesota (Figure 3.8) (Paintcare, 2021[139]; Paintcare, 2020[140]). For comparison, the Product Stewardship Institute estimated that 10% of purchased household paint goes unused, representing a possible upper limit for collection rates (2015[141]) Higher collection rates are associated with lower costs of programme implementation per litre, suggesting that the programmes achieve some economies of scale.

Figure 3.8. Comparison of programme expenses and collection rates in US paintcare states



Source: (PaintCare, n.d.[127]), population data (US Census Bureau, 2022[142])
Note: Data collected from annual reports by state. All data from fiscal year 2020 annual reports.

Increases to recycling rates

Some building materials can be isolated for recycling or reuse. For example, about 80% of paintcare paint is latex-based paint, most of which is recycled and blended into paint products, and some is used for concrete mixtures, as a component of daily cover for landfill or other downcycling applications. However, the remaining 20% is oil-based paint that is predominantly incinerated while a small percentage can be recycled into new paint products (PaintCare, n.d.[143]). From July 2020 to July 2021, 69% of the 12.9 million litres of latex paint collected in California was recycled to make new paint products (Paintcare, 2021[144]).

Proponents of EPR for the built environment (buildings) argue that recovery targets and incentives on this waste could encourage the recovery of materials from demolition projects. In several cases where a ban on landfilling of C&DW waste has been implemented, the policy has increased material recovery, but primarily as use for aggregate and low-value material applications. For example, Maine (United States) has a disposal ban on materials with a viable recycling market, including asphalt pavement, brick, concrete, metal and wood, (Mass DEP, 2020[136]). Additionally, The Netherlands implemented a C&DW landfill ban in 1997 and achieved a near 100% recycling rate by 2010, though material is primarily used in low-value applications such as road foundations, concrete, and site elevations. (Zhang et al., 2020[129]). Therefore, the ban has helped to create a saturation of low-quality road aggregate (Di Maria, Eyckmans and Van Acker, 2018[145]). Similarly, C&DW recycling facility operators in Massachusetts (United States) have reportedly struggled to find markets for some materials (e.g. polystyrene foam commercial roofing material) and many local operators have been acquired by more vertically-integrated regional or national companies (Rosengren, 2023[146]).

3.4 Food loss and waste (Cooking oils, and commercial food producers)

3.4.1 Background on the product sector and end-of-life issues

This sub-chapter will focus on food loss and waste streams with EPR programmes in place or under discussion in OECD member countries: cooking oil and commercial food waste.

Cooking oils

Cooking oil is a significant source of waste. A 2008 study compiling reported cooking oil waste statistics, identified at least 16 Mt of annual waste (Gui, Lee and Bhatia, 2008[147]).

Disposal of cooking oil is environmentally and economically costly. When leaked into environmental systems, via landfill leachate or direct disposal in water bodies, it can impact wildlife and habitats and contaminate drinking water sources. Its introduction to water can cause oxygen overload and result in dead zones.

Improper discharges in sewage impact these systems leading to blockages followed by maintenance issues and costs (Hosseinzadeh-Bandbafha et al., 2022[148]). Disposal of cooking oils is a common source of sewage maintenance issues. For example, the United States EPA estimated that "grease" was the most common cause (responsible for 47%) of reported sewer blockages (2007[149]).

Separate collection of waste cooking oils is rather limited in most cases. For example, in the EU, the per capita consumption of vegetable oil was roughly 21.9 kg from 2010 to 2012, but separate collection was about 1 kg per capita. Otherwise, disposal via the general waste stream or in sewage are common disposal methods (Ortner et al., 2016[150]).

The high calorific value of waste cooking oils means that they may be converted to other forms of energy use, including biodiesel, thermal energy, electric energy, or biogas. Conversion and utilization of vegetable oils in place of fossil energy sources can mean reductions in greenhouse gas emissions (Ortner et al., 2016[150]).

Commercial food waste

Separate collection of food waste helps to divert these wastes from landfills (where it causes leachate and methane gas emissions) or incineration (where its high-water content reduces the efficiency of waste-to-energy plants). Separate collection of food waste can also generate high-quality secondary raw materials and by-products, such as compost, biogas and digestates, commonly used as fertiliser .

To date, a large share of bio-waste, including food waste, is landfilled, or incinerated. In 2017, the average rate of separate collection of bio-waste across the 32 EEA member and cooperating countries was approximately 50% (Van Der Linden and Reichel, 2020[151]). However, these collection rates are likely to increase rapidly in the EU, since the revised EU Waste Framework Directive will require its member states to separately collect bio-waste by 31 December 2023 (European Parliament and Council of the European Union, 2018[30]). In the United States, only an estimated 4% of food waste from municipal solid waste was composted, but 56% was landfilled in 2018 (US EPA, 2022[152]). Separate collection and treatment increases costs of waste management.

3.4.2 Examples of EPR for food waste

Cooking Oils

Several countries require producers of cooking oils to manage the waste stream from their products. The EU's Waste Framework Directive states that the European Commission will consider the feasibility of adopting quantitative targets on the regeneration of waste oils (European Parliament and Council of the European Union, 2018[30]).

Table 3.6. Examples of EPR for cooking oils

Example	Description	Source
Belgium	Edible oils are covered by an EPR scheme in which producers contribute to EoL costs through contracts with municipalities.	(European Commission, 2014[153])
France	Requirement to sort bio-waste and to treat it through suitable channels. Producers of 60 litres of edible oils per year.	(French Ministry of the Ecological Transition, 2021[154]).
Italy	A dedicated consortium for the collection and treatment of used vegetable oils (CONOE) manages the collection and treatment or recycling of used oils and fats Most of the collected material (90%) is processed to make biodiesel.	(Ibanez et al., 2020[155])
Spain	The National Association of Waste and Edible Oil and Fat By-Product Managers (GEREGRAS), is the PRO collecting and processing its target of 60% collection from total production by 2030.	(Ibanez et al., 2020[155])

Commercial food waste generators

In France, and at the subnational level in the United States there are requirements for collection and treatment of food loss and waste generated by commercial firms. These policies may be considered EPR because they make the commercial entity that has produced food waste responsible for meeting recovery targets for these products at the EoL stage.

Table 3.7. Requirements of commercial food waste

Examples of food waste requirements

	Regulated actors	Threshold	Description
France	Large producers of bio-waste or food waste	10 tons of bio-waste/year	Requirement to sort bio-waste and to treat it through suitable channels (composting or anaerobic digestion). (French Ministry of the Ecological Transition, 2021[154]).
California (United States)	Firms and public entities	4 cubic yards/week of bio-waste	Requires actors to subscribe to service, processing on-site or donating surplus food
Connecticut (United States)	Food wholesalers and distributors, manufacturer or processors, supermarkets, and resort or conference centres. Educational facilities are exempted.	52 US tons per year.	Requires food waste be sent to compost or AD facility or animal feed operation, donation, on-site treatment
Maryland (United States)	Entities generating food residual within 30 miles (48 km) distance from a processor	2 US tons per week, then one ton per week in 2024	Requires separation of food residual for diversion of food waste.
Massachusetts (United States)	Any entity producing commercial organic material	1 US ton per week	Requires food waste be sent to compost or AD facility or animal feed operation, donation, on-site treatment
New York (United States)	Businesses, non-profits, public entities	2 US tons per week	Requires transporting food waste to an organics recycler or processing on-site. Must separate edible surplus food for

			donation.
Rhode Island (United States)	Commercial food wholesaler or distributor, industrial food manufacturer, supermarket, resort of conference centre, restaurant, religious institution, prison, hospital, casino, or covered educational facility	104 (52 for covered educational facilities) US tons per year	Requires food waste be sent to compost or AD facility or animal feed operation, on-site treatment
Vermont (United States)	Essentially all actors, including individuals	18 US tons per year, and any amount of food scraps from landfill	Requires donation for human consumption, agricultural use, composting, AD, or energy recovery, or onsite treatment
Washington, D.C. (United States)	Retail food stores and universities.	Retail food stores with a floor area of at least 10 000 square feet (929 square meters), and colleges and universities with at least 2 000 students.	Sets goals for the separation, donation, and diversion of commercial food waste.

Source: (Bolden et al., 2019[156])

3.4.3 Benefits and considerations of EPR implementation

Cost recovery

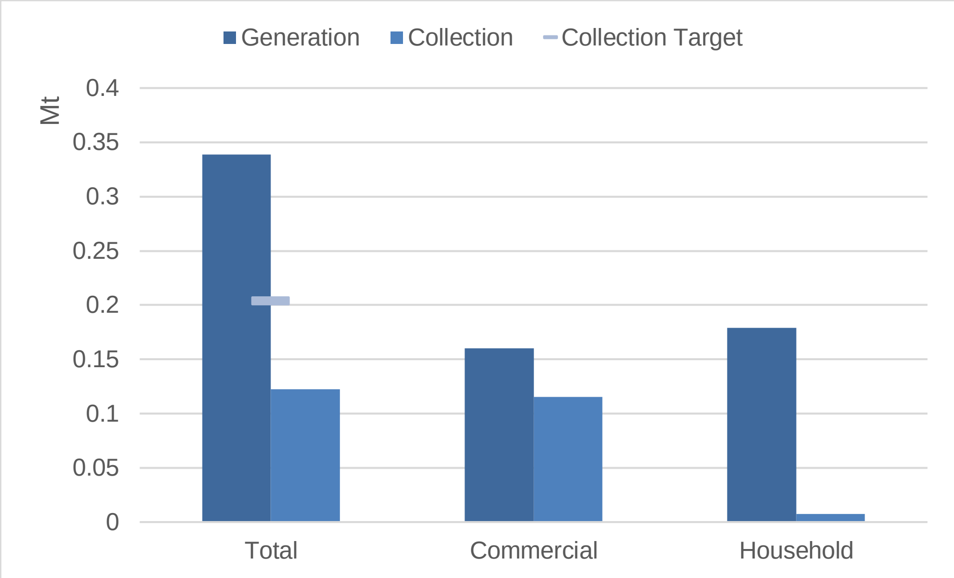
Proponents of EPR for foodstuffs argue that it could be an attractive policy approach to shift some of the costs of managing volumes of food waste from the public sector to producers and instigating a shift in treatment of food waste. An EPR system can oblige producers to finance the collection and processing of food waste to meet recovery targets, increase awareness of and provide access to households to separate collection. Advance disposal fees for items that generate problematic food waste could help to recover the costs of managing this waste. For example, in Italy, law 154/2016 establishes a fee schedule for charges to producers of food oils and animal and vegetable fats, which help to fund collection services, awareness campaigns and administration (CONOE, 2019[157]). In addition to producer fees, the cooking oil waste stream is high-value and the sale of collected material as a feedstock helps fund the collection service (Fondazio per lo Sviluppo Sostenible, 2021[158]).

Recovery targets for large commercial food waste generators do not aim primarily to generate revenues for the public sector, because this waste sector is typically organised by the private sector itself. However, there is some evidence that recovery targets on commercial food waste can stimulate demand for these waste collection and recovery services, ultimately generating revenues for private separate waste collection. For example, an ex-post input-output model assessment of Massachusetts' landfill ban on commercial bio-waste estimates that the ban increased employment in waste hauling, processing, and rescue organisations, while these sectors generated USD 5.4 million in state and local tax revenue (Bolden et al., 2019[156]).

Separate collection

Countries with EPR for cooking oil waste have exhibited high collection rates, especially for commercial waste. In Italy, the PRO CONOE has increased its collection from 15 000 tonnes in 2001 to 73 000 tonnes in 2017, an estimated 27% of total collectable waste in 2017 (Ibanez et al., 2020[155]; ENI, 2017[159]). The PRO Renoils has collected 75 161 tonnes from its creation in 2016 to 2021 (RenOils, n.d.[160]). In total, PROs in Italy collected 83 000 tonnes in 2019, about 30% of collectable waste (Fondazio per lo Sviluppo Sostenible, 2021[158]). In Spain, the National Association of Waste and Edible Oil and Fat By-Product Managers (GEREGRAS) collects an estimated 36% of edible oil waste, primarily from the commercial sector, which lies below its target of 60% by 2030 (Figure 3.9).

Figure 3.9. Edible oil collection in Spain by sector



Source: (Ibanez et al., 2020[155])

Recovery targets can also stimulate increases in separate collection of food waste. For example, Massachusetts organic waste haulers and processors reported handling 6 to 8 times more food tonnage in 2015 compared to 2010 (Bolden et al., 2019[156]).

Increases to recovery rates

EPR-collected waste cooking oils can help to meet some of the demand for the material as feedstock for renewable energy generation. Demand for waste oils is expected to increase in the coming years, possibly to more than 3 billion litres in the EU alone by 2030 (Ibanez et al., 2020[155]). In Italy, CONOE partners with biorefineries to supply them with feedstock to produce biofuels. The feedstock can help to displace some demand for primary materials like palm oil. The current agreement is for up to 1 Mt, which will likely cover all its collected oil (ENI, 2017[159]). From 2016 to 2021, RenOils sent roughly 94% of its collected material for recovery (RenOils, n.d.[160]). RenOils also partners with biorefineries to provide its collected material as feedstock for the generation of biofuels (ENI, 2019[161]). In Italy, roughly 90% of collected waste was used in biofuel production, with smaller shares used to produce soap, cosmetics, lubricant oils, and industrial fats. The average value of collected oils and fat in Italy is approximately EUR 620 per tonne (Fondazio per lo Sviluppo Sostenible, 2021[158]).

Proponents of recovery targets argue that this EPR approach can stimulate waste reduction, food rescue, and ultimately demand by the private sector for compost and anaerobic digestion services. Recovery targets typically provide exemption to actors that fall below a specified waste generation level, providing a possible incentive for waste reduction. They also aim to generate demand for reuse (via donation). For example, Massachusetts experienced a 22% increase in donated or rescued food from 21 300 in 2014 to 25 900 in 2017 (Bolden et al., 2019[156]). Proponents of these policies argue that they generate economic benefits such as increasing employment in food waste services as well as tonnage for separate treatment (ICF, 2016[162]; Bolden et al., 2019[156])

3.5 Early lessons from product sector case studies

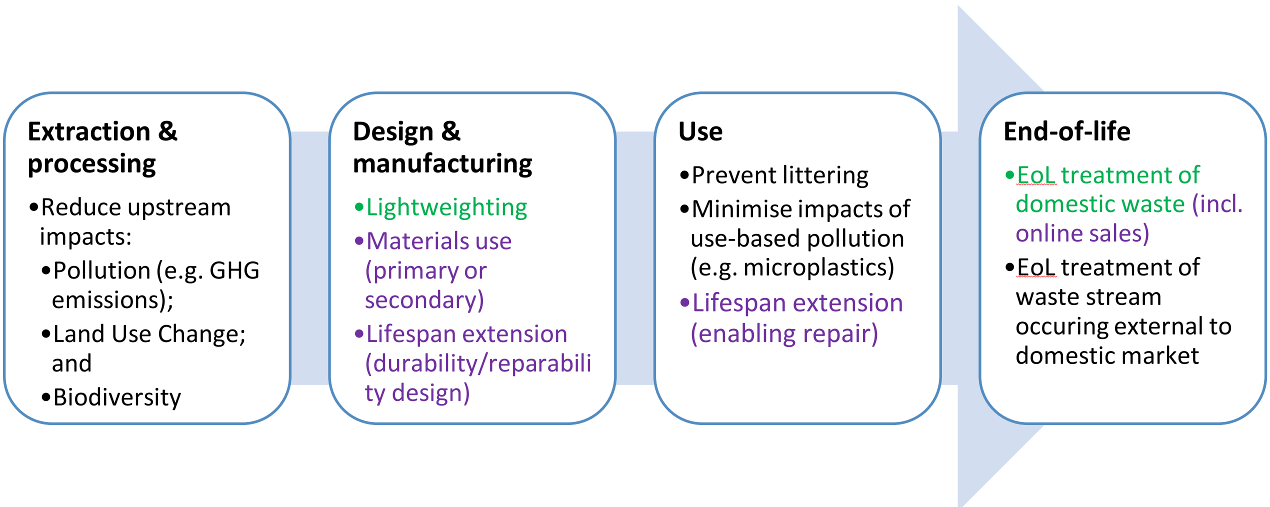
Throughout the preceding case study analysis, there is evidence that EPR schemes generate funding for the collection and recovery of targeted products. For several products there is a strong argument that producers have a specialised expertise to improve collection and recovery, or to improve design of their products, generating benefits in addition to funding. For example, producers with dedicated retail space, such as textiles and paint appear to be well placed to increase consumer access to separate collection services, providing a clean source of material for recovery. There is a strong argument that EPR targeting products used by businesses can help to incentivise waste reduction, for example in C&DW and food waste generation. Indeed, in these product sectors, EPR schemes exist at some scale and several additional schemes are forthcoming. EPR for products that frequently leak to the environment appear to have a less strong argument for a producers' specialised expertise, due to a consumer's influence in leakage (e.g. tobacco product filters) or a missing consensus about what would constitute design for the environment (e.g. fishing gear). EPR is just one policy approach and can be complemented with other policy approaches in these cases, or alternative policies may prove more appropriate in some contexts (see section 5.2).

4 Case studies of EPR to address impacts throughout the lifecycle

EPR has traditionally served to internalise the EoL management costs of a product. Indeed, the OECD has previously defined EPR as an environmental policy approach in which a producer's responsibility for a product is extended to the post-consumer stage of the product's lifecycle (OECD, 2016[1]). However, environmental impacts occur throughout a product's lifecycle. In its original conception, EPR was not limited to the EoL phase, but rather its progenitors argued that producers should bear responsibility for the impact of their products throughout the lifecycle, both "upstream" in design and "downstream" in waste management (Lindhqvist and Lifset, 1997[24]). Lindhqvist originally defined EPR as making the manufacturer of the product responsible for the entire lifecycle of the product, especially for the take-back, recycling, and final disposal (2000[4]).

Producers are well placed to reduce the environmental impacts of products, for instance through improved product designs. Part of the argument for EPR by its proponents is the hope that by making producers responsible for lifecycle costs it would incentivise producers to change their product design in order to reduce environmental or human health impacts (Figure 4.1).

Figure 4.1. Possible objectives of EPR to reduce environmental impacts throughout a product's lifecycle



Note: **Green** marks the actions traditionally covered by an EPR. **Purple** marks the actions that are currently explored or implemented in some EPR systems for some product categories. **Black** mark actions to avoid but are currently not considered in EPR systems.
Source: Authors own.

Some EPR systems have started to address externalities that occur outside the EoL phase through advanced EPR fee modulation (i.e. differentiating the cost paid by producers to the collective PRO to fulfil their EPR obligations based on product design criteria) (see Box 2.1). Fee modulation based on criteria for design-for-repair or recyclability, or the use of secondary materials are now in use in several EPR systems (Laubinger et al., 2021[21]). Signalling some producers' embrace of this approach, the Consumer Goods Forum includes "eco modulation" of EPR fees as an approach to setting incentives for packaging sustainability (The Consumer Goods Forum, 2022[163]). There is ongoing debate about the extent to which the modulation should correspond only to EoL costs. For example EXPRA has argued in its guiding principles for modulated fees that the fees should reflect the real and measurable management costs (EXPRA, 2019[164]).

In addition, there are discussions to use EPR schemes to prevent littering and finance clean-up of littered items. These discussions are particularly present for littered tobacco product filters in the EU and the UK, but also include chewing gums and various single-use plastic (SUP) items. The EU SUP Directive states that member states should introduce EPR to cover the necessary costs of waste management and clean-up of litter from single-use plastics with no readily available alternative as well as the costs of awareness campaigns to prevent and reduce such litter (EU Lex, 2019[50]).

At question is whether and under which conditions producers should be made responsible for circular ambitions or environmental impacts of products throughout the lifecycle. With this question in mind, this chapter explores three case studies where there are current examples or an active policy debate about using an EPR approach outside the EoL phase: cost recovery of pollution mitigation and clean-up activities, instigating design for the Environment (DfE) with fee modulation, and extending the geographical scope of second-hand goods. Each case study will review the debate about the application and outline current examples where they exist. The succeeding chapter will identify early insights about applying an EPR approach to these new applications.

4.1 Cost recovery of mitigation efforts: littering, pollution and clean-up costs

4.1.1 Littering

Improper disposal and littering of products can generate various environmental, economic and public health externalities. For instance, littered single-use plastic items, tobacco product filters (see section 3.1) and chewing gum can cause habitat destruction or have eco-toxicological health effects. Littering can also cause economic impacts on tourism and fisheries. In 2019, an estimated 22 million tonnes of plastic waste leaked into the environment. Plastics leakage is largely due to mismanagement of waste and littering. Around 2% of municipal waste is littered, varying across product groups (OECD, 2022[32]). By one recent estimate, the social costs of plastic-related pollution amount to hundreds of billions of dollars annually (Markl and Charles, 2022[33]).

Litter management and clean-up is costly. For instance, the United Kingdom spends an estimated GBP 662 million annually on litter management (see Box 4.1). Incorporating impacts from littering in mandatory EPR for products that are frequently found in litter surveys is one possibility for recovering some of these costs.

Box 4.1. Expansion of littering into the Packaging EPR in the United Kingdom

The United Kingdom is considering proposed amendments to broaden the scope of EPR packaging to include cost obligations on producers for litter management. The annual costs of prevention, provision of receptacles and clean-up is roughly GBP 662 million, of which GBP 212 million is spent on managing littered packaging that is currently covered by an EPR. Roughly 65% of costs are for clean-up efforts. After consultation, the Government has decided that litter should be included in the full net costs of packaging waste to better implement the polluter pays principle. The government has proposed a fund, in which producers would contribute, based on prevalence of their products in the litter stream. The fund would support litter management activities by the public sector, NGOs, and other duty bodies.

Source: (Darrah et al., 2021[56]; DEFRA, n.d.[165])

In addition to the provision of financial resources, proponents of applying an EPR approach to littering argue that a producer may be able to impact littering outcomes through their impacts on collection rates via awareness campaigns, especially in product sectors with high brand loyalty, the availability of appropriate collection facilities or design of products that minimise littering (e.g. see section 3.1). As well, in some cases, producers may be uniquely positioned or be particularly effective at improving recycling efforts. Several markets will soon be implementing requirements for littering coverage in EPR (Table 4.1).

Table 4.1. Examples of EPR covering littering

	Examples
European Union	The SUP Directive states that Members Countries should introduce EPR to cover the necessary costs of waste management and clean-up of litter from single -use plastics with no readily available alternative as well as the costs of awareness campaigns to prevent and reduce such litter (EU Lex, 2019[50]).
Germany	Manufacturers of SUPs will pay into a fund to cover the costs of waste collection, clean-up of litter, and transport and treatment of such litter (Ellinghaus and Neumann, 2022[166]).
United Kingdom (legislative proposal)	The United Kingdom is considering proposed amendments to its EPR packaging policy to broaden scope of EPR to include cost obligations for litter management. Producers will be responsible for contributing, based on prevalence of their products in the litter stream. for support of litter management activities by the public sector, NGOs, and other duty bodies. (The exact methodology remains to be defined).
Norway	Beverage Producers pay an environmental tax of NOK 6.2 (0.7 USD) per can or glass bottle and NOK 3.75 (0.4 USD) per plastic bottle. The environmental fee declines depending on the collection rate of a product (The Norwegian Tax Administration, n.d.[167]): The system means that a high littering rate leads to a higher fee paid by producers and incentivises DRS participation in order to ensure high collection rates and minimise littering.

There is an ongoing debate about whether mandatory EPR is the right approach to finance litter reduction and clean-up efforts. Littering strongly depends on consumer behaviour and is thus to some extent external to a producer's influence. One argument expressed by industry is that this application of an EPR approach to littering does not perfectly align with the polluter pays principle. For example, The Consumer Goods Forum (CGF) does not include responsibility for littering costs in its view from industry on optimal EPR for packaging (The Consumer Goods Forum, 2020[9]). There is an argument that financial EPR alone is likely insufficient to impact consumer behaviour. Complementary or alternative policies that can potentially be uniquely tailored to the consumer include product taxes on frequently littered items to fund clean-up efforts. Additional policies may be needed to discourage littering, as customers could miscomprehend the fee as a condoning of their improper disposal.

The composition of litter has brought expression of concern for fair obligations in terms of the range of products covered by EPR. For example, in the United Kingdom, an EPR for packaging will make packaging producers responsible, but chewing gum and cigarette producers are not yet fully responsible for the costs of the litter of their products. The Foodservice Packaging Association is calling for impartiality in coverage (Chadwick, 2021[168]).

Differences in litter survey results have also been observed, such as in San Francisco for the surveys to determine the share of cigarette waste in total littering costs that ranged from its 2009 study (22%) and its 2014 survey (53%). These differences can lead to different cost liabilities for producers and create a sense of unfairness or arbitrariness in EPR fee setting (see section 5.1.2) .

4.1.2 End-of-pipe capture of micropollutants: synthetic microfibres and tyres

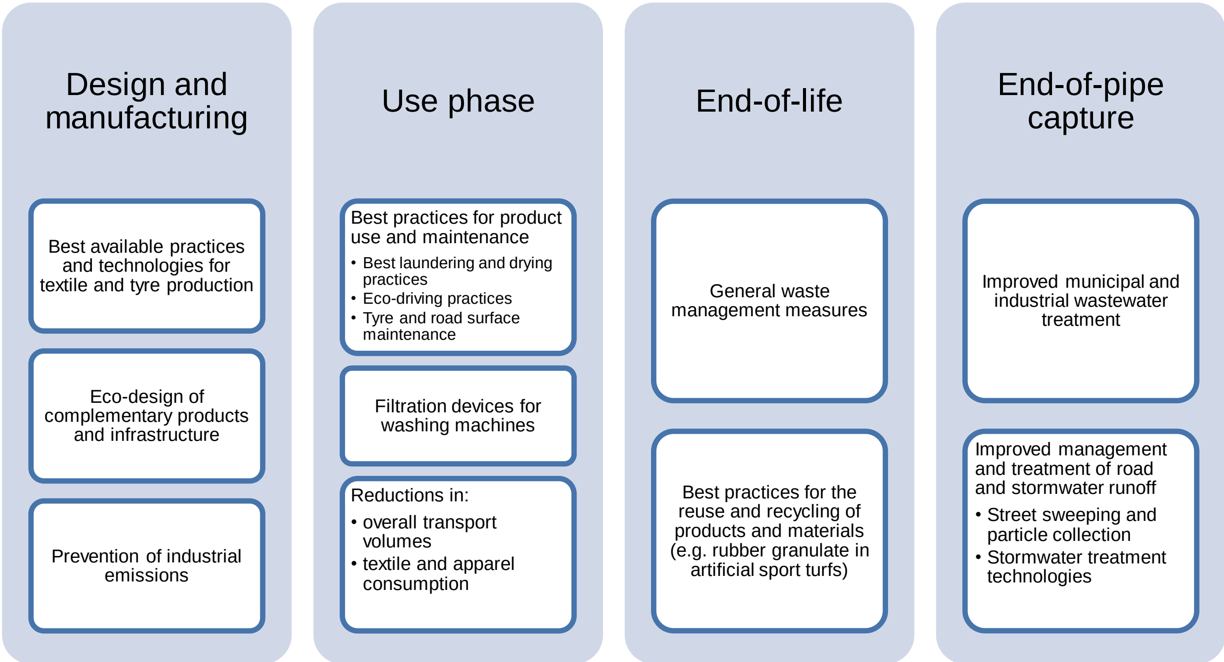
The production and use phases of the product lifecycle can generate pollutants that, if not properly captured and treated, are released to the natural environment. Micropollutants are natural and synthetic contaminants that make their way into ground and surface waters largely due to human activities. They can consist of natural and synthetic substances, including microplastics, pharmaceuticals, antibiotics, personal care products, but also pesticides and industrial chemicals. This section focuses on EPR for use-based microplastic emissions, a central part of micropollutants.

Two of the biggest sources of microplastic release into the environment are through washing of synthetic textiles and tyre abrasion on roads. The manufacturing and use (including wear and wash) of synthetic and blended textiles can result in the release of synthetic microfibres that may pose a risk to ecological and human health. Friction between vehicle tyres and pavement results in the abrasion of vehicle tyres and the agglomeration of tyre material and pavement, known as tyre and road wear particles (TRWP).

There are several actions that producers, consumers, and waste managers can take throughout the lifecycle to reduce the release and impact of these micropollutants (Figure 4.2). At the use phase, laundering and driving practices can reduce the release and increase capture of the pollutants. At ‘end of pipe’, regulatory requirements, and economic instruments can help to increase the uptake of mitigating measures.

Infrastructural improvements via upgrades in wastewater treatment plants (WWTP) can mitigate the leakage of micropollutants and microplastics into freshwater systems, however these upgrades are costly, and operation is energy intensive (OECD, 2021[169]).

Figure 4.2. Overview of synthetic microfibre and tyre particle pollution entry points and mitigation opportunities:



Source: (OECD, 2021[169]).

In several OECD Member Countries, there is an ongoing discussion about whether to apply a financial EPR to producers for the upgrading of WWTPs to mitigate micropollutants at the end-of-pipe. Proponents argue that it can help the public sector to recover the costs of providing end-of-pipe microplastics mitigation and for treatment costs for waste-water treatment. Indeed, the EU Commission is considering stakeholder arguments about the feasibility of an EPR scheme on products that release micropollutants and microplastics in the aquatic environment, including synthetic fibres (In Extenso and Deloitte, 2020[170]).

Some in the policy debate have pointed out that the diffuse nature of substances complicates the assignment of responsibility. There are data gaps on flow and impacts of each substance, making it difficult to assign a fair financial obligation to each product sector, let alone producer. If EPR is not applied to all sources that contribute to micropollutant emissions in sewage, then they argue that the costs of upgrades would be unfairly distributed among only a few key emission sectors, leaving others to free-ride on the clean-up benefits (In Extenso and Deloitte, 2020[170]).

More generally, there is an argument that an overemphasis on costly end-of-pipe solutions foregoes opportunities to lower either the flow or impact of micropollutants through design changes.

Others argue that the administrative burden of an EPR is too costly if it is only a means for revenue generation. For example, they argue a tax on households or product tax on tyres could serve the same purpose of raising funds and require less administrative burden.

4.2 Incentivising design for the environment: favourable product characteristics

4.2.1 Recycled content in products

Products with high recycled content can lower a product's environmental footprint[14] and boost demand for secondary materials, thus stimulating recycling. Naturally, producers are well placed to influence product design, such as through the inclusion of secondary materials. Whilst environmentally preferable, using secondary materials is often not the preferred choice of producers, due to added costs related to adjusting production methods and barriers such as contamination, supply risks, and demand uncertainty (Wijayasundara, 2021[171]). Modulation of EPR fees, based on the share of secondary materials in a product can incentivise the use of secondary materials. Some PROs have started to experiment with incentives to increase recycled content (Table 4.2).

Table 4.2. Examples of EPR fee modulation according to recycled content criteria

Country (State/Province)	Examples of recycled content criteria
Belgium	Bonus of EUR 50 per tonne for all industrial plastic packaging made from a minimum of 30% post-consumer recycled material.
France	EPR fees for packaging include a: • 10% fee reduction for cardboard and graphic paper (in publications) with > 50% recycled content, • 5% fee increase for using primary fibres from forests without eco-management labels, • 50% fee reduction for textiles and shoes with 15% recycled fibres/materials (EY, 2016[172]), • Reduction in charges for plastic packaging: ○ PET (EUR 0.05 per kg and an additional EUR 0.35 if the packaging is made exclusively with post-consumer recycled material). ○ PE flexible (EUR 0.40 per kg and an additional EUR 0.15 if made exclusively of recycled household packaging) ○ PE rigid [mainly HDPE] (EUR 0.45 per kg) ○ BD (EUR 0.40 per kg and EUR.15 for post-consumer recycled material). ○ PP (EUR 0.45 per kg); and ○ PS (EUR 0.55 per kg) (CITEO, 2021[173]).
Germany	The 2019 Packaging Ordinance requires PROs to provide incentives for sustainable packaging design and to modulate EPR fees accordingly. PROs are required to design fees that include differentiating fees along criteria of among others recyclability (given existing technologies) and recycled content and content of renewable materials (BMJV Germany, 2019[174]).
Canada (Quebec)	EPR fee modulation is applied to packaging, inspired by the French bonus/malus scheme system. This involves a 20% bonus for producers who entirely manufacture packaging with recycled content and who use at least 50% to 80% of recycled content for printed materials (e.g. magazines and other publications) (EEQ, 2020[175]).
United States (California)	• State law establishing EPR for carpets requires a difference in fees for the presence of post-consumer recycled content (California Legislative Information, 2020[176]). • State law establishing EPR for packaging requires that fees be adjusted using malus fees or credits for the percentage of post-consumer recycled content (California Legislative Information, 2022[8])
Chile	Collective management systems for packaging must modulate fees with bonus or malus based on recycled content, if the secondary material is derived from waste generated in Chile (Ministerio del Medio Ambiente Chile, 2021[177]).

One challenge to this approach is that an emphasis on recycled content may give mixed signals to producers on priorities for design. For example, inclusion of recycled content can compete with lightweighting because recycled materials tend to have a higher weight-to-strength ratio than virgin materials. Recycled content can impact the mechanical recyclability of a product or packaging. Further, the environmental benefits of recycled content are based on the displacing of virgin production and should complement policies that aim to prevent material consumption and waste generation.

[14] Recycled materials often have a lower environmental footprint than virgin materials.

Another question that has appeared in the policy debate is whether alternative policy instruments are better suited to incentivise recycled content targets. Regulatory standards for product composition can complement an EPR scheme that is focused on the EoL costs of the product. For example, the EU Single-Use Plastics Directive requires plastic bottles to be made of at least 25% recycled content by 2025 and 30% recycled content by 2030 (European Parliament, 2019[178]). The policy will likely have a strong impact on demand for secondary materials, but a minimum target does not provide a dynamic incentive to increase recycled content beyond the required minimum share. Primary material taxes are another option to stimulate the substitution of these materials with secondary equivalents. In contrast with an EPR scheme, the revenues generated do not stay with the PROs, but instead go to the public sector.

4.2.2 Lifespan extension: Design for durability, reparability, and re-use

Increasing the lifespan of products slows material throughput in the economy and, assuming that it reduces the overall demand for new products it also avoids waste. EPR can contribute to a longer product lifespan by differentiating its fees per product based on reusability, reparability or durability criteria. For example, EEE products in France are subject to fee increases (malus) and decreases (bonus) based on reparability and availability of spare parts (Table 4.3).

Table 4.3. Examples of EPR fee modulation according to lifespan extension

<table>
<tr><th>Country (PRO)</th><th>Examples of durability, reparability, reusability, or waste prevention criteria</th></tr>
<tr><td>Belgium (Valipac)</td><td>Reusable industrial packaging is exempted from EPR fees entirely.</td></tr>
<tr><td>Estonia</td><td>Reusable packaging does not need to be declared, if reused.</td></tr>
<tr><td>Portugal</td><td>Tyres placed on the market from the national retreading programme are not charged an ‘ecovalor’ fee (VALORPNEU, 2018[179]).</td></tr>
<tr><td>France</td><td>20% fee increase for:
• Refrigerators, vacuum cleaners and drills without technical documentation for reparation – OR – unavailable spare parts.
• Game consoles without technical documentation of reparation – OR – absence of spare parts – OR – presence of brominated flame retardants in the plastic hull.
20% fee decrease for:
• Washing machine or dish washer with spare parts available up to 11 years – OR – post-consumer recycled content > 10%.
• Coffee machines and kettles with spare parts available up to 5 years – AND – availability of technical documentation for reparation.
• Computers with standard peripherals including memory card and readers, absence of paints and covers that complicate recycling and reuse and recycled content of post-consumer plastics > 10%.
• Printers that can be fully dismantled with standard equipment – AND – availability of spare parts for up to 5 years.
• ‘Eco-modulation’: Lower rate for rechargeable batteries as compared with single use batteries (SCRELEC, 2019[180]).
• Eco-mobilier: scale-able furniture products are given a bonus modulation intended to reduce furniture waste. Supporting documentation examples include assembly instructions, such that duration of use can be extended (Eco-Mobilier, 2018[181]).
• ECO-TLC: a 75% bonus is given for clothing, home textiles, and footwear that meet durability requirements (ECO-TLC, 2019[116]).
A one-time 8% bonus to producers that achieve weight reductions or that reduce the number of packaging units from the prior year, whilst maintaining ISO-material and functionality standards (CITEO, 2019[23]).
• For re-use and repair, the Fonds pour le Réemploi Solidaire will take 5% of collected EPR fees to support social enterprises that facilitate re-use and preparation for re-use of textiles (RReuse, 2020[117])</td></tr>
</table>

Whilst a product's lifespan can be influenced by the producer's design choices, it is an ongoing policy debate whether EPR is the most effective policy tool to increase product lifespans. Proponents of lifespan extension in EPR argue that:

- EPR eco-design criteria are easier to change than legislation. Therefore, EPR may be more dynamic compared to regulation.
- Producers can actively participate in the setting of the criteria and therefore ensure that the financial incentives are relevant and in line with the best available technology.
- Building a platform to collect data and sharing insights from recycling and production can also contribute to better design.

The extent of this incentive to change design is dependent on the magnitude of the fee modulation as a share of the product's price (fee to product price ratio). In most durable product groups, such as electrical and electronic and electrical equipment, the EPR fee tends to be relatively small compared to the product price and incentive by modulating the fee is thus limited. Regulatory standards and economic incentives are alternative policies that could be used to increase durability, repair and spare parts availability. Increasing product lifespans does not only imply a change in product design, but also a change in consumer behaviour, for which alternative approaches could be used. For instance, an EPR approach like deposit refund schemes can support the use of reusable packaging and products.

4.3 Extended geographic scope of EPR

Value chains are globally interconnected, and some products are frequently traded for repair and reuse and exported to other markets. As such, some products that are purchased in one market, can eventually become waste in another market. For example, a substantial volume of used vehicles, used textiles, and EEE is exported to developing and emerging economies for repair, refurbishment, and further use.

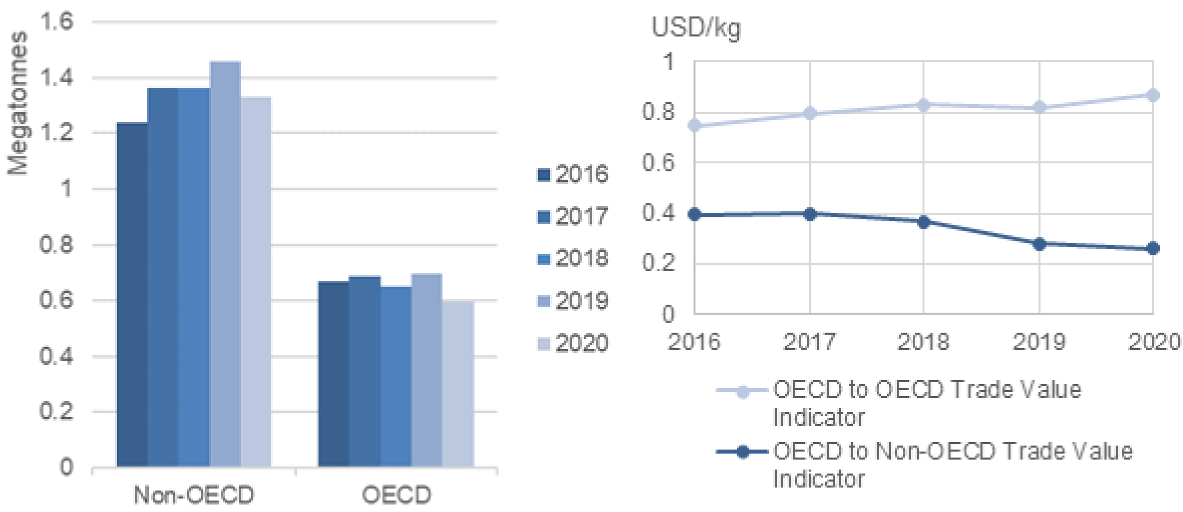
Whilst trade in second-hand goods can extend the lifespan of these products and thus be environmentally preferable, it raises concerns about producer responsibility and producer requirements at the end of the product's life. As these products become waste in foreign markets, they are not captured by the collection and recycling requirements of the EPR system in the purchase market, creating externalities in financing the collection and treatment in the market, where they eventually become waste (Yamaguchi, 2021[182]). Uncertainties about the environmentally sound management (ESM) of this waste in destination countries can also lead to environmental concerns. These uncertainties are particularly pronounced for trade of used goods to least developed countries' (LDCs).

Between 2015 and 2018, 14 million used vehicles were exported worldwide from Europe, the United States and Japan, with 80% going to low and middle-income countries and more than half destined for Africa (UNEP, 2020[183]). Looking ahead, with electrification of vehicle fleets and assuming that trade flows of EVs follow similar patterns of internal combustion engine vehicles, EoL batteries of these electric vehicles (EVs) may end up in markets with limited environmentally sound treatment facilities. EoL batteries contain valuable minerals and materials that should be recovered and without proper treatment, these batteries can cause substantial environmental harm and public health impacts (Pearce, 2020[184]). The European Environmental Bureau (EEB) has estimated that the export of end of life vehicles out of the European Union corresponds with between 294 and 409 million EUR in fees collected in EPR programs that do not follow the flow to the importing markets (Apoorva and Bhutani, 2023[185]).

Electronic and electrical equipment (EEE) is another example of a product group that often receives an extended life in a jurisdiction outside of its initial purchase, as reflected in trade data. For instance, U.S. exports of used EEE were valued USD 1.45 billion in 2011 and almost half of the U.S exports were shipped to non-OECD countries, with India, Hong Kong, and China, together receiving about 31% (U.S. International Trade Commission, 2013[186]). Exports of used EEE from the EU increased from 2008 to 2013, with the fastest growth in exports to Northern and Western Africa and Southern and Central Asia. Once these exported used goods become waste, they can constitute up to 10 to 15 percent of the e-waste share in the importing country's market (Baldé, Wang and Kuehr, 2016[187]). A 2020 study estimated that annually 0.3 kg per inhabitant from Northern Europe and 1 kg per inhabitant from Western Europe of used-EEE for re-use are exported to Western Africa (Baldé et al.[188]). The EEB has estimated that 4.3 megatonnes of WEEE is exported out of the European Union, which corresponds with between 340 and 380 million EUR in fees collected in EPR programs that do not follow the flow of the waste products to the importing markets (Apoorva and Bhutani, 2023[185]).

Worn textiles is a third product group frequently exported for reuse. Many garments are discarded after little wear and tear, meaning they could be reused or recycled. Whilst higher quality garments are typically kept for domestic re-use, a portion of worn clothing is traded, with roughly one-third of OECD exports traded within the OECD and two-thirds destined for non-OECD countries. Trade data shows that worn textiles traded among OECD countries tend to be of higher value than worn textiles exported to non-OECD countries, with the values diverging further in recent years (Figure 4.3).

Figure 4.3. OECD exports of worn textiles by destination type



Note: Left is reported export weight by OECD member countries destined for other OECD member countries and exports to non-members Right is the Value (USD) per weight (kg) of trade between OECD member countries and exports to non-members. Data is from 25 March 2022 of HS Code 6309 worn textiles and other worn articles.
Source: (UN Comtrade, n.d.[189])

The export of used goods for repair, refurbishment and further re-use raises policy questions on how to address and possibly incorporate such goods into producer responsibilities laid out in EPR schemes. Usually, EPR policies like product take-back requirements and recycling targets only target the recycling of products that become waste in the home country and producer responsibility ends at the point of export. Proponents of an extension of the geographic scope of EPR argue that producers should bear the ultimate responsibility for their products, also when exported. Proponents have argued that this responsibility could include finance, technical assistance or other support to the countries that ultimately handle the product at EoL (Thapa et al., 2022[190]; Thapa et al., 2022[191]). In theory, funding could be collected by the EPR scheme in the domestic market to finance collection and end-of-life management in the destination country.

Some voluntary EPR programmes demonstrate how producers could fulfil physical responsibility for products in LDCs (Table 4.4). In addition to provision of financial and technical resources, the programmes have demonstrated some success in collection. For example, the French telecommunications company

Orange has collected 2 million mobile phones and the Closing the Loop programme has collected a further 3 million mobile phones (Orange, n.d.[192]; Closing the loop, n.d.[193]). Japan's circular economy implementation plan calls for investigating the launch of an international resource circulation network.

Table 4.4. Examples of extended geographic scope of EPR

Country (PRO)	Examples
Japan	The 2018 fundamental plan for establishing a sound material-cycle society states that Japan will form an international resource circulation network for its recycling industry to accept waste with resource potential that is difficult for developing nations to properly dispose (Ministry of Environment Japan, 2018[194])
Closing the Loop (Europe)	A voluntary procurement programme in which the IT equipment purchaser gives a fee that is used to collect EoL EEE in Africa to be sent to recovery facilities (Closing the loop, n.d.[193]).
Orange (France)	Orange (mobile phone company) has partnered with local partners Emmaüs International and Ateliers du bocage, to operate mobile waste collection sites in Burkina Faso, Benin, Niger, Cameroon and Côte d'Ivoire. Collected waste is returned to France for environmentally sound treatment and recovery.
Recupel, (Belgian)	Recupel, the Belgian WEEE PRO, has partnered with and helped to finance the non-profit Worldloop in establishing a recycling centre in Nairobi, Kenya (Worldloop, n.d.[195]). They have also partnered with UNIDO to provide technical assistance and training for EEE recycling officials in the Democratic Republic of Congo, Ethiopia, Kenya, Morocco, Senegal, Tanzania, Uganda, and Zambia (UNIDO, n.d.[196])
Sustainable Materials Management Electronics Challenge (United States)	The United States' EPA's Sustainable Materials Management Electronics Challenge encourages EEE producers to send collected used EEE products to third party certified recyclers (U.S. EPA, n.d.[197]).

However, data limitations about the pathways of products exports for reuse likely impact the feasibility of an ADF in which funds travel with products across borders. For example, there is a difference of 22.4 Mt between reported European plastic demand (52.4 Mt) and post-consumer plastic waste (29.1 Mt). This gap could in part be explained by a net addition to stocks, but also by plastic embedded in exports of secondary goods, such as vehicles or EEE (Katainen, Ragaert and Shiran, 2022[198]). Due to data uncertainties, accountability for the use of EPR funds would thus likely be an issue.

In the absence of funding support from the geographic expansion of EPR, some policymakers have alternatively turned to official development assistance to address particularly harmful stemming from the mismanagement of used goods traded for the purpose of repair, refurbishment, and re-use. Official Development Assistance to least developed countries (LDCs) helps LDCs build capacity for ESM of waste. Regulation of exports of second-hand goods can help to stymie the flow of these goods to markets without the capacity to manage the subsequent waste. However, annual ODA currently provides less than 2% of the financial needs for basic waste management in LDCs (OECD, 2022[32]).

5 Considerations for new applications of EPR

5.1 Early evaluation of EPR in the case studies

The reviewed case studies show that the use of EPR as a policy approach is expanding to additional product groups and this trend is likely to continue in the coming years. For instance, the EU requires its member states to adopt EPR for tobacco product filters and fishing gear by 2023 and 2025 respectively. It also requires member states to implement separate collection of food waste by 2024 and textiles by 2025. As well, several US states have adopted novel application of EPR, including EPRs for textiles (e.g. carpets and mattresses), construction and demolition waste and paint.

EPR is also being expanded to enlarge producer responsibility to additional lifecycle impacts. For instance, the EU is requiring a more detailed fee modulation of collective EPR schemes that better reflects impacts of individual products to incentivise sustainable product design and also incorporates fees to cover clean-up costs of littered single use plastic items. The United Kingdom is considering inclusion of litter clean-up costs in its forthcoming EPR for packaging.

Whilst there is momentum on many levels to enlarge the use of EPR and examples of early adopters have been successful in securing funding, improving collection and recycling infrastructure and also to some extent incentivising design changes, there remain challenges in the conceptualisation and implementation of EPR for some of these novel applications.

5.1.1 Opportunities

The recent literature on EPR consistently identifies three benefits to EPR implementation: (i) shifting EoL management costs from the public sector to the producers and consumers of products, (ii) improving collection rates, and (iii) improving recovery rates in a cost-efficient way (see chapter 2). Examination of the early adopters of the case studies identified confirms that these benefits remain relevant.

EPR has helped to generate significant amounts of revenue to finance the collection and management of building materials (e.g. paint and flat glass), textiles (e.g. garments, mattresses, and carpets), and cooking oils. EPR programmes for these relatively novel product groups have generated hundreds of millions annually to finance their collection and recovery. For commercial food waste and C&DW, where private entities are frequently physically responsible for their waste, there is also some early evidence that recovery targets helped to instigate private spending on waste hauling, processing, and recovery. In addition, the implementation of an EPR for tobacco filters in San Francisco (United States) has helped to recoup some of the costs of public provision of clean-up activities and awareness campaigns.

Early adopters of EPR for building materials (e.g. paint and flat glass), textiles (e.g. garments, mattresses, and carpets), and cooking oils have exhibited relatively high rates of separate collection. This can be particularly helpful for products that are hazardous or problematic when comingled with the general waste stream, such as, paints, motor and cooking oils. Voluntary EPR schemes in place for fishing gear indicate that improvements that ease access of fishing vessel owners to collection can increase rates of separate collection.

EPR implementation has coincided with increases in material recovery rates of building materials (e.g. flat glass and latex paints), cooking oils for production of biofuels, and commercial fishing gear. EPR programmes for construction and demolition waste and textiles have coincided with increases in recovery rates, but material is predominantly used in low-value applications, such as insulation material for recovered textiles or backfill or road aggregate for recovered C&DW.

There is some indication that a more refined modulation of EPR fees can incentivise more sustainable design of products. However, due to the novelty of this policy development the evidence to date is limited. In France, an early adopter of EPR fee modulation, there is some evidence of growth in the share of producers in the EEE, packaging, furniture, and textiles product sectors eligible for an EPR fee bonus due to adjusting their product designs. However, for some EEE products (laptops, cell phones, and vacuum cleaners) the share of producers receiving a penalty fee also grew over time, indicating the fee liability had little impact on design change for some products (Joltreau, 2019[199]).

5.1.2 Challenges and considerations

The review revealed several challenges that policymakers likely face when applying EPR to some novel product groups or when enlarging the scope of producer responsibility to additional impact categories along the lifecycle.

Defining the "producer" liable for paying the EPR fee

For several of the product sectors considered in the case studies the definition of the "producer" who is liable for the EPR fee payment is not always clear. Frequently, a producer is defined as the entity that places the product on the market, for example the manufacturers or importers of cooking oils in Italy. In its original conception, manufacturers were identified as actors well placed to affect change (Lifset, 1993[3]). However, there is an argument present in the policy debate that for some products, producers may not necessarily be the "actor" causing the environmental damage and that is best placed to affect change and as such it may be more effective to charge fees to other actors. This is particularly relevant in the debate on tobacco product filters, fishing gear, and cooking oils, where collection and recovery rates strongly depend on the consumer disposal behaviour. As such, producers have limited leverage in influencing disposal. EPR detractors claim alternative policies, such as public provision or taxes may be better suited.

The progenitors of the EPR approach noted a difference between an EPR and a product-oriented environmental policy which aims to reduce impacts by targeting the impacts themselves throughout the lifecycle. They argued that a product orientation risked leaving no particular actor responsible for impacts, whereas an EPR approach pragmatically aims to access producers' resources and change producer or consumer behaviour at EoL (Lindhqvist and Lifset, 1997[24]). For example, a review of the possibility of applying EPR to fishing gear identified the high number of stakeholders that impact EoL fishing gear as a barrier to any one of them taking responsibility for facilitating EoL management (Nogueira et al., 2022[67]).

Establishing a methodology for EPR fee rates that is transparent, fair and functional

For several product groups the setting and calibrating of financial responsibilities of producers is a challenge. Some in the policy debate argue that EPR fees should correspond only to observable EoL costs, such as collection, recovery, and disposal. Others argue that observable EoL costs alone may not fully capture external costs of an EoL product, particularly in cases when mismanaged and littered waste that escaped formal collection and waste management systems causes environmental damage (e.g. littered tobacco filters or abandoned fishing gear).

The challenge is to establish a methodology, that is transparent and “fair” for individual producers and product sectors, but also remains operational. For example, litter composition surveys can be used to assess and distribute the costs of clean-up between commonly littered product groups (e.g. packaging and cigarette butts), however, survey methods and thus survey data can vary. Different methodological approaches can also lead to different conclusions. For instance, distributing costs based on either count, weight, or potential environmental impact and toxicity of items found in litter surveys can each lead to substantially different cost distributions and fee estimates. There is also a potential of noise between multiple surveys, even when using consistent methodology.

Similar challenges exist for establishing methodologies to enlarge producer responsibilities to additional impact categories that lie outside of the traditional coverage of EPR programmes. Where fee adjustments aim to include externalities beyond observable EoL cost differences, there is also a risk of diversion from the polluter pays principle if these externalities are only partially subject to producer’s actions and also influenced by e.g. consumer behaviour (Laubinger et al., 2021[21]).

Non-transparent or unclear methodologies for EPR fee calculation can lead to a sense of arbitrariness and provide a reason for industry to engage in the fee-setting process. Overly complex methodologies reduce their functionality.

The benefits of fee modulation, instigating design change and arguably a fairer distribution of fees, should be compared with the administrative burden and other costs. The additional complexity requires data collection and management capacity that may be more available in mature EPR systems. Therefore, it may be advisable to add this feature once an EPR has been established and is already accomplishing its basic targets (Laubinger et al., 2021[21]).

Allocating producer responsibility in the context of limited data availability

Data limitations can pose a barrier to a clear allocation of producer responsibility. Limited data on littering of specific items obscures the ability to assign producer responsibility of litter clean up among different product groups and producers within a product group.

Similarly, data limitations also pose a barrier to enlarge producer responsibility to products that are exported from the domestic market before their end-of-life and enter the waste stream in other markets. For producer responsibility to be enlarged to these exported goods and eventually EPR funds to travel with a product, detailed information about transboundary movements of these products is needed. Whilst some information exists about the trade in used goods, this is currently not detailed or sufficient enough to assign specific producer responsibilities to these trade flows.

What is the added value of EPR beyond revenue generation?

The original intent and rationale for EPR is not only to raise funds for end-of-life treatment, but to make use of producers’ specialised expertise or position in the value chain to organise EoL treatment cost-efficiently and possibly improve recyclability through sustainable design of products. However, for some product groups it is questionable whether producers can (1) ensure cost-efficient waste treatment, or (2) change product designs to mitigate EoL impacts.

For example, for littered cigarette butts there is limited gains in recycling, but the primary gain is to avoid negative impacts of littering through improved collection and clean up. As well, there is little room for producers to change product design to reduce littering or the impact of littered cigarette butts. A similar situation is apparent for EPR programmes that aim to assign producer responsibility to micropollutants caused by synthetic microfibre shedding of textiles. Whilst there is some opportunity for textile producers to adjust fabric designs to reduce microfibre shedding, much of it is dependent on consumer behaviour (e.g. washing behaviour), or on upgrades to filter technologies at the end-of-pipe or in wastewater treatment plants.

Where producers are unlikely or unable to influence the cost-efficiency of EoL treatment, there is a question about the usefulness of EPR. Alternative policy approaches such as product charges or taxes can equally raise revenues and implement the polluter pays principle. Where the externality does not lie exclusively with the producer, there is also a question whether other policy tools may be more targeted and better suited to address the impacts.

5.2 EPR in a policy mix with alternative and complementary policy approaches

Based on the challenges and considerations discussed in the preceding section, EPR is not necessarily an ideal policy approach to solve all environmental issues throughout a product's lifecycle. This sub-chapter aims to reflect the usefulness of EPR in the context of a comprehensive policy mix with complementary policy approaches or compared against alternatives, highlighting relevant examples from the policy debate identified in the case studies.

5.2.1 General Treasury provision of collection and processing services

Public provision and financing of EoL management is commonly observed as a status quo for several of the case studies. For example, the public sector is commonly responsible for provision of tobacco product filter litter clean-up activities, the provision of wastewater contaminant clean-up, and the collection and treatment of mixed post-consumer waste which frequently contains food waste, textiles, and sometimes hazardous wastes improperly sorted like cooking oils and paints.

Part of the initial promise of EPR is to access the specialised expertise of producers who might be able to find more (cost-)efficient solutions in the provision of EoL management (Lindhqvist and Lifset, 1997[24]). Proponents of EPR have argued that organisation by industry can mean more efficient collection than by the public sector, for example due to the private sector's ability to optimise costs or the use of reverse supply chains, which entails operational gains and upscaling. The applicability of this argument may depend on whether the producers of the product sector are well placed to facilitate improved recovery. For example, the paint EPR in the United States makes use of retailers' stores to ease customer participation in the programme.

Public provision of services can be complementary of producers' efforts to service isolated or otherwise underserved communities. EPR programmes typically set targets for collection but may not specify collection services by location. For example, the United States' paint care programme often targets a share of the population within a specific distance of a year round collection facility. This could incentivise retailers to focus collection on urban centres, where there is a greater opportunity for economies of scale, but at the expense of more isolated communities. Here, public provision or support of services for some communities could be complementary to private EPR efforts. The California packaging EPR programme will include an innovative mix of these concepts, requiring a future PRO to fund USD 500 million per year for 10 years for public provision of plastics pollution mitigation, of which 60% must offset impacts on low-income or rural areas (California Legislative Information, 2022[200]).

A shared operational responsibility may help to increase recovery rates. Croci et al conducted a pooled regression analysis of 21 European PROs to identify drivers of differences in recycling rates. They found higher recycling rates when local authorities lead or share in operational responsibility for collection (2022[201]). Cahill et al's case study analysis of 11 packaging and WEEE PROs concluded that the involvement of local authorities in design and implementation results were significantly more positive (2011[202]).

The debate surrounding public or private financing of EoL management is particularly relevant for product sectors in which there is an ambition or requirement for separate collection or a target for environmental quality. For example, EU member states will need to determine whether separate collection of textile and bio-waste, as the EU Waste Framework Directive will soon require, should be financed and managed through public provision or through producer contributions in the form of an EPR. California (United States) is an informative example of public provision of separate waste collection of organic waste. It is seeking to reduce its landfill disposal of organic waste by 75% by 2025 compared with 2014 without using an EPR approach, but instead using a combination of public funding programmes and requirements of municipalities (see Box 5.1).

Box 5.1. California's (United States) organic waste disposal reduction programme

To achieve its bio-waste disposal reduction targets, California requires its municipalities to facilitate separate collection of organic waste for all residents and businesses, which is estimated to cost a total of USD 21.1 billion from 2019 through 2030, an increase to households of roughly USD 1.40 per month and USD 55 per month for regulated businesses (State of California, 2018[203]). California has established a USD 348 million grant fund (168 in 2022 and 183 in 2023) for local organic waste recycling projects (Quinn and Rachal, 2022[204]; Rosengren, 2022[205]). However, some localities have experienced or anticipate a need for double digit rate increases to pay for the collection programme and infrastructure development (Roengren, 2022[206]; League of California Cities, 2022[207]). Some localities, such as San Jose, that do not have direct usage fee charges, will need to absorb the costs and finance this provision through other means such as its general treasury.

There is a risk that producers will pass EPR fees to consumers through increased product prices. The size of the effect is dependent on the share of EPR costs in total price, the elasticity of demand by consumers for the targeted product, and the costs of tailoring price by market. For example, Bose estimated an upper bound increase in costs due to introducing EPR at a national level for packaging in the United States could amount to roughly 0.69% of household grocery spending or USD 4 monthly per household (2022[208]). This low increase is in part due to the low share of packaging costs in product prices. Valpak estimated that the introduction of EPR policies to the United Kingdom could result in GBP 8.33 per household per month (2021[209]). Hesterman et al found no link between packaging EPR policies and prices offered online by national retailers in Canada (2021[210]). Their finding indicates the reticence of national retailers to tailor prices due to subnational policies and that additional costs of EPR compliance are not necessarily passed through to customers directly in higher prices in all cases.

Incorporating EoL costs into essential goods introduces an equity consideration because low-income households would be more highly impacted by EPR fees that are passed on through product prices. Additionally, independent retailers, often found in low income neighbourhoods, can exhibit relatively low competition enabling these retailers to pass along a higher portion of cost increases compared with national retailers (Ma et al., 2019[211]). Public provision, ultimately charged through progressive council taxes could ensure a more equitable way of revenue raising with less distributional impacts.

5.2.2 Incentives (taxes) to incentivise behaviour change by consumers

For some of the product groups there is debate about whether consumers may be better placed than producers to impact the resource productivity or environmental impacts of a product group. For example, tobacco product filter littering is done by consumers. Similarly, some fishing gear loss is intentional and gear owners can change their behaviour to lower the risk of unintentional gear loss (Viool et al., 2018[63]). Consumers are also responsible for separating waste streams at source, for example by separating textiles, cooking oils, and paints from the general, co-mingled residual waste. Several policy options that directly target consumer behaviour are in use for the products considered in the EPR case studies, including product taxes, "pay-as-you throw" waste collection systems and littering fines.

Whilst an EPR can recoup costs of managing impacts caused by products from producers, there is a debate as to whether cost recovery alone is a sufficient justification for the application of an EPR approach compared with a charge or an earmarked tax which is arguably simpler to set and to administer. A tax on households can be designed to target household behaviour more effectively. A product tax can deviate from the end-of-life costs and incorporate other external costs, such as littering or environmental and health costs, as opposed to an Advance Disposal Fee (or ADF, an EPR policy instrument) that aims to primarily to internalise the EoL costs of products. When tax revenues are specifically earmarked for improving waste management infrastructure, it may have a similar outcome as an ADF, but arguably with reduced complexity. However, taxes could be susceptible to being diverted for other public priorities.

Pay-as-you-throw (PAYT) policies can also effectively influence consumer behaviour and incentivise waste reduction and improve sorting of targeted materials. For example, in 2005, Korea banned landfilling of food and implemented a PAYT system on household food waste, which now helps to pay for about 60% of collection and processing costs and reduced food waste by 10 to 30% (Kim, 2019[212]) (Yu, 2017[213]).[15] Similarly, Portland, Maine offers its residents free access to drop-off collection points for food waste and simultaneously is increasing PAYT fees for residual waste to incentivise waste reduction and participation in the separate food waste collection programme and several municipalities in Belgium, the Netherlands and Luxembourg have also implemented PAYT schemes (Card and Schweitzer, 2016[214]). As a complementary policy to EPR, PAYT can potentially help to incentivise consumer behaviour at separating at source and improve EPR outcomes.

5.2.3 Product design regulations to ensure design for environment

There is an argument present in the policy debate as to whether regulatory measures may be better suited for achieving desired design changes as compared to modulated EPR fees. Regulatory measures can help to establish minimum design requirements for producers. Examples of these design requirements include bans on hazardous chemicals, eco-design minimum requirements for durability, reparability, and recyclability or recycled content targets. As a complementary policy to EPR, a regulation or tax could help to instigate design change while the EPR focuses on provision of collection and recovery.

Proponents of regulation have argued that environmental regulations can help to establish a level playing field and reduce uncertainty about investment in environmental design. There is also an argument that recycled content requirements could be a more straight-forward policy approach to ensure design changes by producers as compared to fee modulation.

The EPR fee (product price to EPR compliance fee) should be large enough to incentivise design changes. The higher the difference in the EPR fees between design choices, the greater the incentive. Design changes can be costlier than EPR fees. Therefore, to provide a sufficient incentive, fee modulation may need to be greater than the difference in observable EoL costs. This difference is susceptible to criticism of arbitrariness in the fee setting methodology. A tax or regulation on design criteria may be more transparent and functional and ultimately more effective.

Design regulations are used in several of the product groups reviewed in the case studies. For fishing gear, there are requirements for gear marking (methods to identify or locate the owner of fishing gear) in the United Kingdom and at the sub-national level in the United States and Canada (OECD, 2021[215]). Indeed, the Norwegian Seafood Federation has argued that measures other than EPR should take precedence, including requirements on documentation of waste practices and promoting eco-design (Nogueira et al., 2022[67]). In the EU, the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH) bans the use of some chemicals in textiles, such as phthalates and formaldehyde, which improves the recyclability of textiles. Also, the inclusion of lead in paint products, once nearly ubiquitous, is now widely banned. These regulatory methods can be complementary of EPR programmes that focus primarily on the observable EoL costs of targeted products.

[15] Some of this weight reduction may be due to citizens draining organic waste of water to lower the weight of the food waste (Kim, 2019[212]).

6 Conclusion

There is significant momentum to apply EPR to additional product groups, as well as to enlarge producer responsibility to additional impact categories beyond its traditional focus at the post-consumer phase of products. In light of this momentum, it is worthwhile to consider the arguments present in the policy debate and to critically reflect under which circumstances EPR seems to be a useful approach, to identify for which applications EPR design and implementation may be more challenging and to consider when other policy tools may be better suited to achieve the desired outcome. Based on several case studies, this report aimed to capture the arguments present and to provide insights for this debate.

As a basic principle, EPR assigns financial responsibility of EoL management to the producer. Doing so implements the “polluter pays principle”. In addition to financial responsibility, some EPR programmes assign organisational responsibility to producers. Doing so targets more efficient solutions, because, as the argument goes, producers can use their specialised expertise and supply chain network to, for instance, organise collection and material recovery more effectively or change product designs to reduce certain lifecycle impacts.

Whilst assigning financial responsibility is a compelling argument for EPR for most of the product groups and impacts reviewed in the case studies, the benefit of assigning the organisational responsibility is less clear in some of the cases. For instance, producers of textiles are not necessarily well-positioned or in possession of a specialised-expertise to upgrade end-of-pipe filtering to capture shed microfibres. If there is an unclear advantage for organisational responsibility, the primary argument for EPR is to generate revenues. In these cases, there is a debate whether this is a sufficient justification for an EPR approach or whether a charge or an earmarked tax, that is arguably easier to administer, can be used instead to generate these revenues. However, EPR should not necessarily be dismissed in cases where it is primarily a financial tool. For example, producers could still contribute financially to the research and development or the construction of appropriate waste management infrastructure. As well, in developing countries exhibiting a developing tax base, collection or administration, EPR may present an attractive solution for financing waste management.

There is also an ongoing debate about whether EPR is the best approach to address impacts caused by products that frequently evade collection, such as littered tobacco product filters and fishing gear. Producers have some influence in changing product designs or improving collection infrastructure to disincentivise littering and improper disposal. However, for example with cigarette littering, consumer behaviour is the primary cause of impacts and some in the policy debate argue that other policies may be more effective in influencing this behaviour, such as targeted fines, behavioural interventions, or informational campaigns.

In addition, there is a debate whether EPR is the best policy approach when the primary aim is to incentivise product design changes. The magnitude of modulation of EPR fees may need to be substantial to provide meaningful incentives and thus the fee is likely to be higher than the measurable costs of EoL management. Higher fees risk relying on methodologies for EPR fee calculation that may create a sense of arbitrariness. Some argue that other policy tools, such as minimum design requirements can more directly compel changes by producers. For example, the EU’s REACH programme effectively bans the inclusion of some chemicals in textiles to compel design change instead of relying on EPR fee modulation.

Finally, EPR implementation may be challenging where data limitations restrict the clear allocation of producer responsibility. As such, assigning producer responsibility and EPR funds to items leaving domestic markets, for instance, may be challenging in light of insufficiently granular data of trade flows of used goods. Voluntary approaches, that do not require such a robust information base may be more functional.

The reviewed evidence and case studies show that EPR is the right policy approach when (i) it allows governments to better implement the polluter pays principle and thereby secure financial resources for EoL management, and (ii) when there is an opportunity for producers to use their expertise or position in the value chain to improve collection of the EoL products and increase material recovery and/or to significantly reduce impacts and improve circularity through product design changes.

However, even in these cases it is worthwhile to critically assess the benefits of EPR compared to other policy approaches. The unique challenges and ambitions for each individual case and market context suggests that in-depth individual analyses will be needed to determine whether an EPR approach is appropriate. This report reviewed the early evidence of the successes and challenges observed in EPR schemes applied to product groups that are not traditionally covered by such schemes, but there remains a research gap in the effectiveness of some of the potential expansions of EPR with little current application. As well, conditions may differ by market for some products depending on local conditions and recovery ambitions. There is a need for in-depth research, informed by local circumstances. Ex-ante analysis for prospective policymaking and additional ex-post analysis on the outcomes on early adopters will help to further inform this ongoing policy debate.

7 Works Cited

Adler, M. (2020), *Textile Recovery in the U.S. a roadmap to circularity*, RRS, http://recycle.com/wp-content/uploads/2020/09/2020-Textile-White-Paper-sept-15-2020.pdf (accessed on 29 March 2022). [84]

Anami, L. (2022), *East Africa wrestles with proposals to ban imports of used clothes*, https://www.theeastafrican.co.ke/tea/business/region-wrestles-with-proposals-to-ban-imports-of-used-clothes-3854868 (accessed on 14 December 2022). [92]

Apoorva, A. and A. Bhutani (2023), "Study on items shipped for reuse and Extended Producer Responsibility fees". [185]

Baden, S. and C. Barber (2005), *The impact of the second-hand clothing trade on developing countries*. [91]

Bahers, J. and J. Kim (2018), "Regional approach of waste electrical and electronic equipment (WEEE) management in France", *Resources, Conservation and Recycling*, Vol. 129, pp. 45-55, https://doi.org/10.1016/j.resconrec.2017.10.016. [11]

Baldé, C. et al. (2020), "In-depth review of the WEEE Collection Rates and Targets", http://www.bureauopdekaart.nl. (accessed on 20 April 2022). [188]

Baldé, C., F. Wang and R. Kuehr (2016), *Transboundary movements of used and waste electronic and electrical equipment Estimates from the European Union using trade statistics*, United Nations University, Bonn. [187]

BAT (n.d.), *The global market: Trends affecting our industry*, https://www.bat.com/group/sites/UK__9D9KCY.nsf/vwPagesWebLive/DO9DCKFM# (accessed on 9 September 2021). [35]

Benachio, G., M. Freitas and S. Tavares (2020), *Circular economy in the construction industry: A systematic literature review*, https://doi.org/10.1016/j.jclepro.2020.121046. [119]

BMJV Germany (2019), *Gesetz über das Inverkehrbringen, die Rücknahme und die hochwertige Verwertung von Verpackungen (Verpackungsgesetz-VerpackG)*, http://www.gesetze-im-internet.de/verpackg/VerpackG.pdf (accessed on 9 September 2020). [174]

Bolden, B. et al. (2019), *Bans and Beyond: Designing and Implementing Organic Waste Bans and Mandatory Organics Recycling Laws*, Food Law and Policy Clinic and the Center for EcoTechnology, http://www.chlpi.org/flpc/. (accessed on 15 November 2021). [156]

Booth, D., P. Gribben and K. Parkinson (2015), "Impact of cigarette butt leachate on tidepool snails", *Marine Pollution Bulletin*, Vol. 95/1, pp. 362-364, https://doi.org/10.1016/j.marpolbul.2015.04.004. [42]

Bose, S. (2022), *Economic impacts to consumers from extended producer responsibility (EPR) regulation in the consumer packaged goods sector*. [208]

Bye Bye Mattress (n.d.), *FAQs - Bye Bye Mattress | A Program of the Mattress Recycling Council*, https://byebyemattress.com/faqs/#toggle-id-7 (accessed on 28 March 2022). [99]

Cahill, R., S. Grimes and D. Wilson (2011), *Review Article: Extended producer responsibility for packaging wastes and WEEE - A comparison of implementation and the role of local authorities across Europe*, https://doi.org/10.1177/0734242X10379455. [202]

California Legislative Information (2022), *Bill Text - SB-54 Solid waste: reporting, packaging, and plastic food service ware.*, https://leginfo.legislature.ca.gov/faces/billTextClient.xhtml?bill_id=202120220SB54 (accessed on 15 July 2022). [8]

California Legislative Information (2022), *SB-54 Solid waste: reporting, packaging, and plastic food service ware.*, https://leginfo.legislature.ca.gov/faces/billTextClient.xhtml?bill_id=202120220SB54 (accessed on 28 November 2022). [200]

California Legislative Information (2020), *PUBLIC RESOURCES CODE-PRC DIVISION 30. WASTE MANAGEMENT [40000-49620] (Division 30 added by Stats*, https://leginfo.legislature.ca.gov/faces/codes_displayText.xhtml?lawCode=PRC&division=30.&title=&part=3.&chapter=20.&article=. [176]

CalRecycle (2021), *Carpet Stewardship Law*, https://www.calrecycle.ca.gov/Carpet/Law/. [100]

CalRecycle (2020), *2016 & 2019 CALGreen Construction Waste Management Requirements*. [131]

Card, D. and J. Schweitzer (2016), “Pay-As-You-Throw schemes in the Benelux countries”, https://ieep.eu/uploads/articles/attachments/84782562-17b9-4a16-b496-95dca4183fcf/BE-NL-LU%20PAYT%20final.pdf?v=63680923242 (accessed on 13 December 2022). [214]

CARE (2021), *CARE California Carpet Stewardship Program: 2020 Annual Report*, https://carpetrecovery.org/wp-content/uploads/2021/09/2020_CA_AnnualReport_v15_2021-08-27b_FINAL.pdf (accessed on 8 September 2021). [104]

Carpet Recycling UK (n.d.), *Promoting the use of textile flooring as a sustainable resource*, https://carpetrecyclinguk.com/ (accessed on 29 November 2022). [112]

Chadwick, P. (2021), “FPA calls for industry fairness over litter clean-up costs”, *Packaging News*, https://www.packagingnews.co.uk/news/249321-06-09-2021 (accessed on 10 March 2022). [168]

CITEO (2021), “La grille tarifaire 2021”, http://www.citeo.com/info-tri (accessed on 5 May 2022). [173]

CITEO (2019), *The 2020 rate for recycling household packaging*, https://www.citeo.com/le-mag/le-tarif-2020-pour-le-recyclage-des-emballages/ (accessed on 7 October 2020). [23]

City of San Diego (n.d.), *Construction & Demolition (C&D) Recycling*, https://www.sandiego.gov/environmental-services/recycling/cd (accessed on 17 September 2021). [133]

City of San Jose (n.d.), *Construction & Demolition Diversion (CDD) Program*, https://www.sanjoseca.gov/your-government/environment/recycling-garbage/construction-demolition-debris (accessed on 17 September 2021). [134]

Closing the loop (n.d.), *Greener Procurement Safe, Solid and Engaging | Closing the Loop*, https://www.closingtheloop.eu/greener-procurement-safe-solid-and-engaging (accessed on 15 November 2022). [193]

CONOE (2019), *PROCEDURA PER L'APPLICAZIONE DEL CONTRIBUTO CONOE*. [157]

Cowie, G. et al. (2013), "Cigarette brand loyalty in Australia: findings from the ITC Four Country Survey", *Tobacco Control*, Vol. 23/suppl 1, pp. i73-i79, https://doi.org/10.1136/tobaccocontrol-2013-051071. [57]

Croci, E. et al. (2022), *Screening the efficiency of packaging waste in Europe*, Universita Bocconi. [201]

CSA City Performance (2016), *Street & Sidewalk Maintenance Standards: Fiscal Year 2015-16 Annual Report*, Office of the Controller, City & County of San Francisco. [59]

CSP Daily News (2012), "Santa Fe Natural Tobacco Launches Butt Recycling Program", *CSP Daily News*, https://www.cspdailynews.com/tobacco/santa-fe-natural-tobacco-launches-butt-recycling-program (accessed on 18 March 2022). [48]

Damayanti, D. et al. (2021), *Possibility routes for textile recycling technology*, MDPI, https://doi.org/10.3390/polym13213834. [89]

Darrah, C. et al. (2021), *Financial Cost of Packaging Litter-Phase 2-Final Report*, WRAP, https://wrap.org.uk/resources/report/financial-cost-of-packaging-litter-phase-2 (accessed on 9 March 2022). [56]

DataSF (2019), *DPW Street & Sidewalk Evaluation Results, 7-1-2013 to Present*, https://data.sfgov.org/City-Infrastructure/DPW-Street-Sidewalk-Evaluation-Results-7-1-2013-to/83ki-hu3p. [61]

DAWE (2021), *Clothing textiles waste*, https://www.awe.gov.au/environment/protection/waste/product-stewardship/textile-waste-roundtable (accessed on 21 March 2022). [102]

DEFRA (2021), *G7 Climate and Environment: Ministers' Communiqué, London, 21 May 2021 - GOV.UK*, https://www.gov.uk/government/publications/g7-climate-and-environment-ministers-meeting-may-2021-communique/g7-climate-and-environment-ministers-communique-london-21-may-2021 (accessed on 5 August 2022). [70]

DEFRA (n.d.), *Litter payments*, https://consult.defra.gov.uk/extended-producer-responsibility/extended-producer-responsibility-for-packaging/user_uploads/9.-litter-payments.pdf (accessed on 9 March 2022). [165]

Di Maria, A., J. Eyckmans and K. Van Acker (2018), "Downcycling versus recycling of construction and demolition waste: Combining LCA and LCC to support sustainable policy making", *Waste Management*, Vol. 75, pp. 3-21, https://doi.org/10.1016/j.wasman.2018.01.028. [145]

Dimitropoulos, A., J. Tijm and D. In 't Veld (2021), *Extended producer responsibility design, functioning and effects*, PBL Netherlands Environmental Assessment Agency, http://www.cpb.nl/en. (accessed on 25 May 2022). [128]

Drinkwater, A. and F. Moy (2017), *Wipes in sewer bloackage study*, Water UK. [77]

Eco-Mobilier (2018), *Eco-fee pricing scale with eco-modulation*, https://www.eco-mobilier.fr/wp-content/uploads/2018/09/bareme_eco-modulation-uk-2018_v3.pdf (accessed on 28 September 2020). [181]

ECO-TLC (2019), *NOUVEAUX CRITERES DURABILITE et PERIMETRE PRODUITS CONCERNES Eco Modulation 1 «durabilité » (bonus de 50 %)*, https://refashion.fr/pro/sites/default/files/fichiers/Crite%CC%80res%202020%20Eco%20modulation%201_durabilite%CC%81%20%281%29.pdf (accessed on 28 September 2020). [116]

EEA (2020), *Construction and demolition waste : challenges and opportunities in a circular economy.*, https://www.eea.europa.eu/publications/construction-and-demolition-waste-challenges (accessed on 6 April 2022). [122]

EEQ (2020), *Credit for post-consumer recycled content*, https://www.eeq.ca/en/faq/prepare-report/am-i-entitled-to-the-credit-for-post-consumer-recycled-content/ (accessed on 28 November 2018). [175]

Eichstädt, T., A. Carius and R. Kraemer (1999), "Producer responsibility within policy networks: the case of German packaging policy", *Journal of Environmental Policy & Planning*, Vol. 1/2, pp. 133-153, https://doi.org/10.1080/714038530. [25]

Ekvall, T. et al. (2015), *EPR-systems and new business models: Part II: Policy packages to increase reuse and recycling of textiles in the Nordic region*, TemaNord, Nordic Council of Ministers, Copenhagen K, https://dx.doi.org/10.6027/TN2015-514. [110]

Ellen MacArthur Foundation (2021), *Extended Producer Responsibility is a necessary part of the solution to packaging waste and pollution: A position paper endorsed by key stakeholders across the packaging system*, https://plastics.ellenmacarthurfoundation.org/epr#Statement. [10]

Ellen MacArthur Foundation (2017), *A new textiles economy: Redesigning fashion's future*, https://www.ellenmacarthurfoundation.org/publications/a-new-textiles-economy-redesigning-fashions-future. [88]

Ellinghaus, U. and A. Neumann (2022), *Germany: Government intends to introduce extended producer responsibility requirements for single-use plastics - Global Compliance News*, https://www.globalcompliancenews.com/2022/05/24/germany-government-intends-to-introduce-extended-producer-responsibility-requirements-for-single-use-plastics030522/ (accessed on 8 July 2022). [166]

ENI (2019), *Deal between Eni and RenOils to boost collection of used food oil and oil used for frying*, https://www.eni.com/en-IT/media/press-release/2019/02/deal-between-eni-and-renoils-to-boost-collection-of-used-food-oil-and-oil-used-for-frying.html (accessed on 3 August 2022). [161]

ENI (2017), *Eni and CONOE sign agreement for the transformation of used vegetable oils into biofuels*, https://www.eni.com/en-IT/media/press-release/2017/05/eni-and-conoe-sign-agreement-for-the-transformation-of-used-vegetable-oils-into-biofuels.html (accessed on 3 August 2022). [159]

EU Lex (2019), *Directive (EU) 2019/904 of the European Parliament and of the Council of 5 June 2019 on the reduction of the impact of certain plastic products on the environment*, https://eur-lex.europa.eu/eli/dir/2019/904/oj. [50]

Eunomia (2020), *How to reduce waste and carbon emissions caused by mattresses A review of global Extended Producer Responsibility schemes*, Zero Waste Scotland, https://www.zerowastescotland.org.uk/sites/default/files/How%20to%20reduce%20waste%20and%20carbon%20emissions%20caused%20by%20mattresses%20-%20A%20review%20of%20global%20Extended%20Producer%20Responsibility%20schemes%20report.pdf (accessed on 29 November 2022). [94]

European Commission (2023), *Circular economy for textiles: taking responsibility to reduce, reuse and recycle textile waste and boosting markets for used textiles*, https://ec.europa.eu/commission/presscorner/detail/en/IP_23_3635 (accessed on 11 July 2023). [31]

European Commission (2022), *Textiles strategy*, https://ec.europa.eu/environment/publications/textiles-strategy_en (accessed on 31 March 2022). [93]

European Commission (2020), *Circular economy action plan* -, Publications Office of the EU, https://op.europa.eu/en/publication-detail/-/publication/45cc30f6-cd57-11ea-adf7-01aa75ed71a1/language-en/format-PDF/source-170854112 (accessed on 11 March 2022). [29]

European Commission (2014), *Development of Guidance on Extended Producer Responsibility*, https://ec.europa.eu/environment/archives/waste/eu_guidance/pdf/report.pdf. [153]

European Environment Agency (2022), *Managing non-packaging plastics in European waste streams*, https://circulareconomy.europa.eu/platform/en/knowledge/managing-non-packaging-plastics-european-waste-streams-missing-part-plastic-puzzle?_cldee=8bi58ZKnuQqtvGHn5HAGSiPNp2d4Edv8Ut2D6KQa50TwCW29OqNqx9YnLv3aDc2C&recipientid=contact-79cb7abaa8bfeb11811b005056a033d5-3f97bdf5ba744ef5a45a4305ee00ed80&esid=612cd320-2997-ed11-8123-005056a043ea (accessed on 23 January 2023). [34]

European Environment Agency (2021), *Progressing towards waste prevention in Europe – the case of textile waste prevention — European Environment Agency*, https://www.eea.europa.eu/publications/progressing-towards-waste-prevention-in (accessed on 4 March 2022). [83]

European Parliament (2019), *Parliament seals ban on throwaway plastics by 2021*, European Parliament, https://www.europarl.europa.eu/news/en/press-room/20190321IPR32111/parliament-seals-ban-on-throwaway-plastics-by-2021. [178]

European Parliament and Council of the European Union (2018), *Directive (EU) 2018/851 of the European Parliament and of the Council of 30 May 2018 amending Directive 2008/98/EC on waste (Text with EEA relevance)*, https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex:32018L0851 (accessed on 5 October 2020). [30]

Europen Commission (2021), *Harmonised application of Single-Use Plastic rules*, https://ec.europa.eu/commission/presscorner/detail/en/IP_21_2710 (accessed on 26 July 2022). [69]

EXPRA (2019), “Towards a common approach to modulated fees”, http://www.expra.eu/downloads/expra_position_paper___towards_a_com.pdf (accessed on 28 September 2020). [164]

EY (2016), *Exploration of the Role of Extended Producer Responsibility for the circular economy in the Netherlands*, https://kidv.nl/exploration-of-the-role-of-epr-for-the-circular-economy-in-nl (accessed on 16 November 2018). [172]

Fisheries Iceland (2017), *Resource Utilisation and Environmental Footprint*, https://sfs.is/wp-content/uploads/2018/09/Environmental_report_2017.pdf (accessed on 16 March 2021). [74]

Fondazio per lo Sviluppo Sostenible (2021), *L’Italia del Riciclo 2021 Sintesi del rapporto CON IL PATROCINIO*. [158]

French Ministry of the Ecological Transition (2021), *Biodéchets*, https://www.ecologie.gouv.fr/biodechets. [154]

Gendell, A. et al. (2021), *Extended Producer Responsibility for Packaging: Elements and Outcomes*, Eunomia, http://www.eunomia-inc.com (accessed on 8 February 2023). [16]

GESAMP (2020), *Our Work: Sea-based sources of marine litter*, http://www.fao.org/3/cb0724en/cb0724en.pdf. [75]

Glass for Europe (n.d.), *Recycling of end-of-life building glass - Glass for Europe*, https://glassforeurope.com/recycling-of-end-of-life-building-glass/ (accessed on 26 July 2022). [125]

Government of British Columbia (n.d.), *CR&D Case Study 10: Waste Diversion, Mandatory Mechanism Linked to Building Code*, https://www2.gov.bc.ca/assets/gov/environment/waste-management/zero-waste/case-studies/cs_californiagreen.pdf (accessed on 17 September 2021). [130]

Government of British Columbia (n.d.), *CR&D Case Study 8: Waste Diversion, Mandatory Mechanism*, https://www2.gov.bc.ca/assets/gov/environment/waste-management/zero-waste/case-studies/cs_sanjose.pdf (accessed on 17 September 2021). [135]

Gui, M., K. Lee and S. Bhatia (2008), “Feasibility of edible oil vs. non-edible oil vs. waste edible oil as biodiesel feedstock”, *Energy*, Vol. 33/11, pp. 1646-1653, https://doi.org/10.1016/j.energy.2008.06.002. [147]

Heo, H. and M. Jung (2014), *Case study for OECD project on extended producer responsibility Republic of Korea Contact person*. [52]

Hesterman, B., R. Dimino and K. Ricchi (2020), *Impact of EPR for PPP on recycling rates*, RRS, https://www.epa.gov/sites/production/files/2019-11/documents/2016_and_2017_facts_and_figures_data_tables_0.pdf (accessed on 8 February 2023). [15]

Hesterman, B. et al. (2021), “Impact of EPR Fees for PPP on Price of Consumer Packaged Goods”. [210]

Hosseinzadeh-Bandbafha, H. et al. (2022), “Managing the hazardous waste cooking oil by conversion into bioenergy through the application of waste-derived green catalysts: A review”, *Journal of Hazardous Materials*, Vol. 424, p. 127636, https://doi.org/10.1016/j.jhazmat.2021.127636. [148]

Ibanez, J. et al. (2020), "European Union Legislation Overview about Used Vegetable Oils Recycling: The Spanish and Italian Case Studies", *Processes*, Vol. 8/7, p. 798, https://doi.org/10.3390/pr8070798. [155]

ICF (2016), *Massachusetts Commercial Food Waste Ban Economic Impact Analysis*, Massachusetts Department of Environmental Protection, https://www.mass.gov/doc/massachusetts-commercial-food-waste-ban-economic-impact-analysis/download (accessed on 16 November 2021). [162]

In Extenso and Deloitte (2020), *Study on the feasibility of applying extended producer responsibility to micropollutants and microplastics emitted in the aquatic environment from products during their life cycle: Module 3-Assessment of stakeholder positions on EPR*, https://www.eureau.org/resources/publications/4309-deloitte-eureau-report-extended-producer-responsibility-module-3/file (accessed on 7 May 2021). [170]

Inglezakis, V. and K. Moustakas (2015), "Household hazardous waste management: A review", *Journal of Environmental Management*, Vol. 150, pp. 310-321, https://doi.org/10.1016/j.jenvman.2014.11.021. [124]

Institut National de l'economie circulaire (n.d.), *La REP, quésaco ? La responsabilité élargie du producteur et la loi AGEC LA REP QUÉSACO ? LA RESPONSABILITÉ ÉLARGIE DU PRODUCTEUR (REP) ET LA LOI AGEC 2*. [78]

ISRI (2021), *ISRI Updates Product Stewardship Position*, Waste 360, https://www.waste360.com/recycling/isri-updates-product-stewardship-position-reflect-ever-evolving-industry (accessed on 22 July 2022). [28]

Jang, Y. et al. (2020), "Recycling and management practices of plastic packaging waste towards a circular economy in South Korea", *Resources, Conservation and Recycling*, Vol. 158, p. 104798, https://doi.org/10.1016/j.resconrec.2020.104798. [13]

Jauke van Nijen (2021), *How to come to a more circular (management) system of fishing gear in the OSPAR-region*, Rijkswaterstaat WVL/BN REM. [71]

Joltreau, E. (2022), "Extended Producer Responsibility, Packaging Waste Reduction and Eco-design", *Environmental and Resource Economics*, https://doi.org/10.1007/s10640-022-00696-9. [20]

Joltreau, E. (2019), "Pricing products' negative externalities at end-of-life using eco-modulation: Discussion from case studies", *ECONOMICS AND POLICY OF ENERGY AND THE ENVIRONMENT* 1, pp. 149-172, https://doi.org/10.3280/efe2018-001007. [199]

Joly, F. and M. Coulis (2018), "Comparison of cellulose vs. plastic cigarette filter decomposition under distinct disposal environments", *Waste Management*, Vol. 72, pp. 349-353, https://doi.org/10.1016/j.wasman.2017.11.023. [38]

Kaffine, D. and P. O'Reilly (2013), "What Have We Learned About Extended Producer Responsibility in the Past Decade? A Survey of the Recent EPR Economic Literature", *OECD Papers*. [5]

Katainen, J., K. Ragaert and Y. Shiran (2022), *Reshaping plastics: Pathways to a circular, climate neutral plastics system in Europe*, SYSTEMIQ, https://plasticseurope.org/reshaping-plastics/ (accessed on 6 April 2022). [198]

Kim, M. (2019), "The Country Winning The Battle On Food Waste". [212]

Kunz, N., K. Mayers and L. Van Wassenhove (2018), "Stakeholder Views on Extended Producer Responsibility and the Circular Economy", *California Management Review*, Vol. 60/3, https://doi.org/10.1177/0008125617752694. [19]

Landbell group (n.d.), *Another country introduces EPR for fishing gear - Landbell Group*, https://landbell-group.com/news/another-country-introduces-epr-for-fishing-gear/ (accessed on 3 August 2022). [72]

Laubinger, F. et al. (2022), "Deposit-refund systems and the interplay with additional mandatory extended producer responsibility policies", *OECD Environment Working Papers*, No. 208, OECD Publishing, Paris, https://doi.org/10.1787/a80f4b26-en. [2]

Laubinger, F. et al. (2021), "Modulated fees for Extended Producer Responsibility schemes (EPR)", *OECD Environment Working Papers*, No. 184, OECD Publishing, Paris, https://dx.doi.org/10.1787/2a42f54b-en. [21]

League of California Cities (2022), *Solid waste and recycling costs on the rise in Clifornia*, https://pub-chulavista.escribemeetings.com/filestream.ashx?DocumentId=14982 (accessed on 24 June 2022). [207]

Legifrance (2021), *Arrêté du 5 février 2021 portant cahier des charges d'agrément des éco-organismes de la filière à responsabilité élargie du producteur des produits du tabac*, https://www.legifrance.gouv.fr/eli/arrete/2021/2/5/TREP2027629A/jo/texte. [51]

Lifset, R. (1993), "Take it back: Extended producer responsibility as a form of incentive-based policy", *The Journal of Resource Management and Technology*, Vol. 21/4. [3]

Lindhqvist, T. (2000), "Extended Producer Responsibility in Cleaner Production Policy Principle to Promote Environmental Improvements of Product Systems". [4]

Lindhqvist, T. and R. Lifset (1997), "What's in a Name: Producer or Product Responsibility?", *Journal of Industrial Ecology*, Vol. 1/2, https://doi.org/10.1162/jiec.1997.1.2.6. [24]

Lusher, A., P. Hollman and J. Mandoza-Hill (2017), *Microplastics in fisheries and aquaculture*. [64]

Maine legislature (2021), *130th Maine Legislature, An Act To Support and Improve Municipal Recycling Programs and Save Taxpayer Money*, https://legislature.maine.gov/legis/bills/display_ps.asp?paper=HP1146&snum=130&PID=1456 (accessed on 7 March 2022). [6]

Ma, M. et al. (2019), "Market Concentration, Market Shares, and Retail Food Prices: Evidence from the U.S. Women, Infants, and Children Program", *Applied Economic Perspectives and Policy*, Vol. 41/3, pp. 542-562, https://doi.org/10.1093/aepp/ppy016. [211]

Marinello, S. et al. (2020), *A second life for cigarette butts? A review of recycling solutions*, https://doi.org/10.1016/j.jhazmat.2019.121245. [40]

Markl, A. and D. Charles (2022), *The price of plastic pollution: Social costs and corporate liabilities*, Minderoo Foundation, https://www.minderoo.org/no-plastic-waste/reports/the-price-of-plastic-pollution/ (accessed on 17 January 2023). [33]

Mass DEP (2020), *Frequently Asked Questions About the Massachusetts Construction and Demolition Materials Waste Bans*, https://www.mass.gov/doc/frequently-asked-questions-construction-demolition-materials-waste-ban/download (accessed on 7 April 2022). [136]

Mattress Recycling Council (2022), "2021 California Annual Report". [105]

Mattress Recycling Council (n.d.), *Our Impact - Mattress Recycling Council | Recycling Programs in California, Connecticut & Rhode Island*, https://mattressrecyclingcouncil.org/our-impact/ (accessed on 28 March 2022). [107]

Mattress Recycling Council Connecticut (2021), *Connecticut Annual Report*, https://mattressrecyclingcouncil.org/wp-content/uploads/2021/10/MRC-20-21-CT-Annual-Report.pdf (accessed on 11 August 2022). [109]

Mattress Recycling Council Rhode Island (2021), "SUBMITTED BY Mattress Recycling Council", https://mattressrecyclingcouncil.org/wp-content/uploads/2021/10/MRC-2021-RI-Annual-Report-final-web.pdf (accessed on 11 August 2022). [108]

Miller, S., A. Horvath and P. Monteiro (2016), "Readily implementable techniques can cut annual CO2 emissions from the production of concrete by over 20%", *Environmental Research Letters*, Vol. 11/7, https://doi.org/10.1088/1748-9326/11/7/074029. [120]

Ministère de la Transition Écologique (2020), *The Anti Waste Law in the Daily Lives of the French People: What does that mean in practice?*, https://www.ecologie.gouv.fr/sites/default/files/en_DP%20PJL.pdf (accessed on 5 May 2021). [123]

Ministerio del Medio Ambiente Chile (2021), *Decreto-12: Establece Metas de Recolección y Valorización y Otras Obligaciones Asociadas de Envases y Embalajes*, Ley Chile, https://www.bcn.cl/leychile/navegar?idNorma=1157019 (accessed on 7 June 2021). [177]

Ministry of Environment Japan (2018), *Fundamental Plan for Establishing a Sound Material-Cycle Society*, https://www.env.go.jp/en/recycle/smcs/4th-f_Plan.pdf (accessed on 30 November 2022). [194]

MRN (n.d.), *About us - MRN*, https://mrn.nl/about-us/?lang=en (accessed on 29 November 2022). [114]

Murugan, K. et al. (2017), "Managing wastes as green resources: cigarette butt-synthesized pesticides are highly toxic to malaria vectors with little impact on predatory copepods", *Environmental Science and Pollution Research*, Vol. 25/11, pp. 10456-10470, https://doi.org/10.1007/s11356-017-0074-3. [44]

Nogueira, L. et al. (2022), "What would it take to establish a take-back scheme for fishing gear? Insights from a comparative analysis of fishing gear and beverage containers", *Journal of Industrial Ecology*, https://doi.org/10.1111/jiec.13296. [67]

NWRA Staff (2021), *NWRA Issues Policy Position on Extended Producer Responsibility - National Waste & Recycling Association*, https://wasterecycling.org/press_releases/nwra-issues-policy-position-on-extended-producer-responsibility/ (accessed on 22 July 2022). [27]

OECD (2022), *Global Plastics Outlook: Economic Drivers, Environmental Impacts and Policy Options*, OECD Publishing, Paris, https://doi.org/10.1787/de747aef-en. [32]

OECD (2021), *Policies to Reduce Microplastics Pollution in Water: Focus on Textiles and Tyres*, OECD Publishing, Paris, https://dx.doi.org/10.1787/7ec7e5ef-en. [169]

OECD (2021), "Towards G7 action to combat ghost fishing gear: A background report prepared for the 2021 G7 Presidency of the United Kingdom", *OECD Environment Policy Papers*, No. 25, OECD Publishing, Paris, https://doi.org/10.1787/a4c86e42-en. [215]

OECD (2019), *Global Material Resources Outlook to 2060: Economic Drivers and Environmental Consequences*, OECD Publishing, Paris, https://dx.doi.org/10.1787/9789264307452-en. [118]

OECD (2016), *Extended Producer Responsibility: Updated Guidance for Efficient Waste Management*, OECD Publishing, Paris, https://dx.doi.org/10.1787/9789264256385-en. [1]

Office of the Treasurer & Tax Collector (n.d.), *Notice to cigarette retailers regarding cigarette litter abatement fee increase*, City and County of San Francisco, https://sftreasurer.org/sites/default/files/2019-12/CIG%20%20Rate%20Increase%20Letter%2012-5-19%20Final.pdf (accessed on 10 September 2021). [54]

Onyshko, J. and R. Hewlett (2018), *Toxics in Carpets in the European Union*, https://circulareconomy.europa.eu/platform/sites/default/files/knowledge_-_toxics_in_carpets_eu_review_anthesis_final_study.pdf (accessed on 29 November 2022). [115]

Orange (n.d.), *How Orange is reducing its environmental footprint in Africa and the Middle East | Corporate*, https://www.orange.com/en/how-orange-reducing-its-environmental-footprint-africa-and-middle-east (accessed on 6 April 2022). [192]

Oregon State Legislature (2021), *SB582 Relating to modernizing Oregon's recycling system*, https://olis.oregonlegislature.gov/liz/2021R1/Measures/Overview/SB582 (accessed on 7 March 2022). [7]

Ortner, M. et al. (2016), "Environmental assessment of three different utilization paths of waste cooking oil from households", *Resources, Conservation and Recycling*, Vol. 106, pp. 59-67, https://doi.org/10.1016/j.resconrec.2015.11.007. [150]

OSPAR Commission (2020), *OSPAR scoping study on best practices for the design and recycling of fishing gear as a means to reduce quantities of fishing gear found as marine litter in the North-East Atlantic*, https://www.ospar.org/documents?v=42718 (accessed on 7 May 2021). [66]

OSPAR Commission (2020), *OSPAR scoping study on best practices for the design and recycling of fishing gear as a means to reduce quantities of fishing gear found as marine litter in the North-East Atlantic*. [73]

PaintCare (n.d.), *District of Columbia - PaintCare*, https://www.paintcare.org/states/district-of-columbia/ (accessed on 17 September 2021). [126]

PaintCare (n.d.), *FAQs*, https://www.paintcare.org/faqs/ (accessed on 2 August 2022). [143]

PaintCare (n.d.), *Our Story - PaintCare*, https://www.paintcare.org/our-story/ (accessed on 17 September 2021). [127]

Paintcare (2021), *District of Columbia paint stewardship program 2020 annual report*. [139]

Paintcare (2021), *Paintcare in California*, https://www.paintcare.org/wp-content/uploads/docs/ca-AnnualRpt-Infographic.pdf (accessed on 2 August 2022). [144]

Paintcare (2020), *Minnesota paint stewardship program FY2020 annual report (July 1, 2019-June30, 2020)*. [140]

Patel, V., G. Thomson and N. Wilson (2012), "Cigarette butt littering in city streets: a new methodology for studying and results: Table 1", *Tobacco Control*, Vol. 22/1, pp. 59-62, https://doi.org/10.1136/tobaccocontrol-2012-050529. [37]

Pearce, F. (2020), *Getting the Lead Out: Why Battery Recycling Is a Global Health Hazard*, https://e360.yale.edu/features/getting-the-lead-out-why-battery-recycling-is-a-global-health-hazard. [184]

Płotka-Wasylka, J. et al. (2022), "End-of-life management of single-use baby diapers: Analysis of technical, health and environment aspects", *Science of The Total Environment*, Vol. 836, p. 155339, https://doi.org/10.1016/j.scitotenv.2022.155339. [76]

PRO Europe (2021), *Participation Costs Overview 2021*. [22]

Product Stewardship Institute (2015), *Paint stewardship: effects on household hazardous waste programs*, https://www.paintcare.org/wp-content/uploads/docs/psi-report-effects-on-HHWs-Dec2015.pdf (accessed on 23 November 2022). [141]

PSI (n.d.), *Mattresses - Product Stewardship Institute (PSI)*, https://www.productstewardship.us/page/Mattresses (accessed on 27 July 2022). [106]

Quinn, M. and M. Rachal (2022), "WasteExpo highlights big ideas for solving food waste, recycling, data and infrastructure problems", *Waste Dive*, https://www.wastedive.com/news/wasteexpo-big-ideas-on-solving-big-problems-food-waste-recycling-data/624055/?utm_source=Sailthru&utm_medium=email&utm_campaign=Issue:%202022-05-19%20Waste%20Dive%20Newsletter%20%5Bissue:41873%5D&utm_term=Waste%20Dive (accessed on 20 May 2022). [204]

Rath, J. et al. (2012), "Cigarette litter: Smokers attitudes and behaviors", *International Journal of Environmental Research and Public Health*, Vol. 9/6, https://doi.org/10.3390/ijerph9062189. [36]

RBH Inc (2021), *Unsmoke Canada and TerraCycle Team Up to Reduce Cigarette Waste Nationwide*, https://www.rbhinc.ca/home/media/media-detail/2021/03/01/unsmoke-canada-and-terracycle-team-up-to-reduce-cigarette-waste-nationwide (accessed on 8 March 2022). [47]

Re-Cig (n.d.), *re-cig.it •*, https://www.re-cig.it/ (accessed on 28 March 2022). [45]

Refashion (2021), *Key performances & Financial situation - Annual Report 2020*, https://en.calameo.com/read/0035113755e68def61fb6?authid=bgcD0pVRtrx3 (accessed on 8 September 2021). [103]

Register, K. (2000), "Cigarette Butts as Litter—Toxic as Well as Ugly", *The Underwater Naturalist*, Vol. 25/2, http://www.longwood.edu/cleanva/ciglitterarticle.htm (accessed on 17 March 2022). [219]

Reichel, A. et al. (2014), *Environmental indicator report 2014 : environmental impacts of production-consumption systems in Europe*. [80]

RenOils (n.d.), *Home*, https://www.renoils.it/ (accessed on 3 August 2022). [160]

Reynolds (2018), *Santa Fe Natural Tobacco Company Announces Major Recycling Milestone*, https://www.reynoldsamerican.com/press/2EKHT7zcBPleKPqb21AyGj (accessed on 18 March 2022). [49]

Ribul, M. et al. (2021), "Mechanical, chemical, biological: Moving towards closed-loop bio-based recycling in a circular economy of sustainable textiles", *Journal of Cleaner Production*, Vol. 326, https://doi.org/10.1016/j.jclepro.2021.129325. [90]

Richardson, K., B. Hardesty and C. Wilcox (2019), "Estimates of fishing gear loss rates at a global scale: A literature review and meta-analysis", *Fish and Fisheries*, Vol. 20/6, https://doi.org/10.1111/faf.12407. [62]

Rijkswaterstaat (n.d.), *Staatscourant van het Koninkrijk der Nederlanden*, https://zoek.officielebekendmakingen.nl/stcrt-2021-49806.html (accessed on 29 November 2022). [97]

Roengren, C. (2022), "Ambitious policy makes California a crucible for investments in organics collection and recycling | Waste Dive", *Waste Dive*, https://www.wastedive.com/news/california-sb-1383-part-one-organics-recycling-crucible/624725/ (accessed on 13 June 2022). [206]

Rosenfield, B. (2016), "Cigarette Litter Abatement Fee Adjustment, Effective January 1, 2017", https://sfcontroller.org/sites/default/files/Documents/November%202016%20Cigarette%20Abatement%20Fee%20Letter%20-%20Final%20Signed.pdf (accessed on 27 July 2022). [55]

Rosengren, C. (2023), "The fragile ecosystem of Massachusetts C&D recycling", *Waste Dive*. [146]

Rosengren, C. (2022), "California's local governments grapple with financial and logistical demands of organics recycling law | Waste Dive", https://www.wastedive.com/news/sb-1383-part-three-california-local-government-budget-pandemic/625818/?utm_content=ad-INTRO&utm_term=43683&utm_source=Sailthru&utm_medium=email&utm_campaign=Issue:%202022-06-23%20Waste%20Dive%20Newsletter%20%5Bissue:42617%5D (accessed on 24 June 2022). [205]

RReuse (2020), *France to create a Solidarity Re-use Fund (and other re-use friendly measures)!*. [117]

Rubio, S. et al. (2019), "Effectiveness of extended producer responsibility policies implementation: The case of Portuguese and Spanish packaging waste systems", *Journal of Cleaner Production*, Vol. 210, pp. 217-230, https://doi.org/10.1016/j.jclepro.2018.10.299. [12]

Sabatini, J. (2017), "SF's cigarette 'litter' fee on packs sold increases to 60 cents", *The San Francisco Examiner*, https://www.sfexaminer.com/news/sfs-cigarette-litter-fee-on-packs-sold-increases-to-60-cents/ (accessed on 13 September 2021). [58]

Sandin, G. and G. Peters (2018), *Environmental impact of textile reuse and recycling – A review*, https://doi.org/10.1016/j.jclepro.2018.02.266. [217]

Santa Monica Public Works (n.d.), *Construction and Demolition Waste FAQs*, https://www.smgov.net/Departments/PublicWorks/ContentRecycling.aspx?id=8137 (accessed on 17 September 2021). [132]

Schneider, J. et al. (2011), "Tobacco litter costs and public policy: A framework and methodology for considering the use of fees to offset abatement costs", *Tobacco Control*, Vol. 20/1 SUPPL, https://doi.org/10.1136/tc.2010.041707. [53]

Schneider, J. et al. (2020), "Online Simulation Model to Estimate the Total Costs of Tobacco Product Waste in Large U.S. Cities", *International Journal of Environmental Research and Public Health*, Vol. 17/13, p. 4705, https://doi.org/10.3390/ijerph17134705. [43]

SCRELEC (2019), *BARÈME SCRELEC 2020 Applicable au 1 er Janvier 2020*. [180]

Seldman, N. (2022), "EPR: The good, the bad and the ugly | Waste Dive", *Waste Dive*, https://www.wastedive.com/news/epr-good-bad-ugly/519582/ (accessed on 22 July 2022). [26]

Semba, T. et al. (2020), "Greenhouse gas emission reductions by reusing and recycling used clothing in Japan", *Sustainability (Switzerland)*, Vol. 12/19, https://doi.org/10.3390/su12198214. [87]

Shimada, T. and L. Van Wassenhove (2019), "Closed-Loop supply chain activities in Japanese home appliance/personal computer manufacturers: A case study", *International Journal of Production Economics*, Vol. 212, pp. 259-265, https://doi.org/10.1016/j.ijpe.2016.11.010. [14]

Singer, A. (ed.) (2015), "Cigarette Butt Decomposition and Associated Chemical Changes Assessed by 13C CPMAS NMR", *PLOS ONE*, Vol. 10/1, p. e0117393, https://doi.org/10.1371/journal.pone.0117393. [39]

Slaughter, E. et al. (2011), "Toxicity of cigarette butts, and their chemical components, to marine and freshwater fish", *Tobacco Control*, Vol. 20/1 SUPPL, https://doi.org/10.1136/tc.2010.040170. [41]

Staatscourant (2022), *Voorpublicatie ontwerpbesluit uitgebreide producentenverantwoordelijkheid textiel*. [98]

State of California (2018), *Standardized regulatory impact assessment summary*. [203]

Stevenson, P. et al. (2010), *Street and Sidewalk Maintenance Standards FY 2009-10*, http://www.sfgov.org/controller (accessed on 21 March 2022). [60]

Stickel, B., A. Jahn and B. Kier (2012), *The cost to west coast communities of dealing with trash, reducing marine debris*, Prepared by Kier Associates for U.S. Environmental Protection Agency, Region 9, https://www.resource-recovery.net/sites/default/files/kier_costs_marinedebris_2012.pdf. [216]

TAKASHI/AII, S. (2020), *New Technology for Recycling Used Clothes | August 2020 | Highlighting Japan*, https://www.gov-online.go.jp/eng/publicity/book/hlj/html/202008/202008_03_en.html (accessed on 21 March 2022). [221]

TAUW (2021), *Verkenning van knelpunten en oplossingen voor het sluiten van de keten voor luiers en incontinentiemateriaal*. [79]

Terracycle (n.d.), *UNSMOKE Cigarette Waste Recycling Program · TerraCycle*, https://www.terracycle.com/en-CA/brigades/cigarette-waste-en-ca (accessed on 8 March 2022). [46]

Thapa, K. et al. (2022), “Ultimate producer responsibility for e-waste management–A proposal for just transition in the circular economy based on the case of used European electronic equipment exported to Nigeria”, *Business Strategy & Development*, https://doi.org/10.1002/bsd2.222. [191]

Thapa, K. et al. (2022), “Policy Brief: Blueprint for Ultimate Producer Responsibility”, https://doi.org/10.5281/ZENODO.5957809. [190]

The Consumer Goods Forum (2022), *Guiding Principles for the Ecomodulation of EPR Fees for Packaging*, http://www.tcgfplasticwaste.com (accessed on 26 April 2022). [163]

The Consumer Goods Forum (2020), *Building a Circular Economy for Packaging: A View from the Consumer Goods Industry on Optimal Extended Producer Responsibility*, https://www.theconsumergoodsforum.com/wp-content/uploads/EPR-Building-a-Circular-Economy-for-Plastic-Packaging-cgf-plastic-waste.pdf (accessed on 11 May 2021). [9]

The Kingdom of Norway (2016), *Failure by an EFTA State to fulfil its obligations – Directive 2000/59/EC on port reception facilities for ship-generated waste and cargo residues)*, https://eftacourt.int/download/35-15-judgment/?wpdmdl=1559 (accessed on 11 August 2022). [68]

The New York State Senate (2022), *NY State Senate Bill S5027C*, https://www.nysenate.gov/legislation/bills/2021/S5027 (accessed on 9 May 2022). [101]

The Norwegian Tax Administration (n.d.), *Beverage packaging*, https://www.skatteetaten.no/en/business-and-organisation/vat-and-duties/excise-duties/about-the-excise-duties/beverage-packaging/ (accessed on 24 September 2021). [167]

The Recycling Partnership (2023), *Increasing Recycling Rates with EPR Policy*, https://recyclingpartnership.org/eprreport/ (accessed on 8 February 2023). [17]

The World Bank (2020), *Population, total - United States | Data*, https://data.worldbank.org/indicator/SP.POP.TOTL?locations=US (accessed on 21 March 2022). [220]

Transition, F. (2020), *Textiles usagés*, https://www.ecologie.gouv.fr/textiles-usages. [96]

U..S. EPA (n.d.), *Nondurable Goods: Product-Specific Data*, https://www.epa.gov/facts-and-figures-about-materials-waste-and-recycling/nondurable-goods-product-specific-data#ClothingandFootwear (accessed on 13 December 2022). [223]

U.S. EPA (n.d.), *Sustainable Materials Management (SMM) Electronics Challenge | US EPA*, https://www.epa.gov/smm-electronics/sustainable-materials-management-smm-electronics-challenge (accessed on 11 March 2022). [197]

U.S. International Trade Commission (2013), *Used Electronic Products: An Examination of U.S. Exports*, https://www.usitc.gov/publications/332/pub4379.pdf. [186]

UN Comtrade (n.d.), *Un Comtrade*, United Nations Statistical Office, https://comtrade.un.org/ (accessed on 5 October 2020). [189]

UN Environment Programme (2020), *Sustainability and Circularity in the Textile Value Chain - Global Stocktaking*. [218]

UNEP (2020), *Global Trade in Used Vehicles Report*, https://www.unep.org/resources/report/global-trade-used-vehicles-report. [183]

UNIDO (n.d.), *UNIDO and WorldLoop partner with Recupel and the Belgian recycling industry to advance sustainable e-waste solutions in Africa | UNIDO*, https://www.unido.org/news/unido-and-worldloop-partner-recupel-and-belgian-recycling-industry-advance-sustainable-e-waste-solutions-africa (accessed on 6 April 2022). [196]

US Census Bureau (2022), "2020 Population and Housing State Data", https://www.census.gov/library/visualizations/interactive/2020-population-and-housing-state-data.html (accessed on 25 August 2022). [142]

US EPA (2022), *Food: Material-Specific Data | US EPA*, https://www.epa.gov/facts-and-figures-about-materials-waste-and-recycling/food-material-specific-data (accessed on 23 November 2022). [152]

US EPA (2020), "Advancing Sustainable Materials Management: 2018 Fact Sheet Assessing Trends in Materials Generation and Management in the United States". [121]

US EPA (2017), *The State of the Practice of Construction and Demolition Material Recovery*, https://nepis.epa.gov/Exe/ZyPDF.cgi/P100SSJP.PDF?Dockey=P100SSJP.PDF (accessed on 30 November 2022). [137]

US EPA (2007), "Summary National Pretreatment Program (40 CFR 403) Controlling Fats, Oils, and Grease Discharges from Food Service Establishments What is the environmental problem with FOG discharges into sewers?", https://yosemite.epa.gov/OA/RHC/EPAAdmin.nsf/Filings/360737AD61D13C6E8525764E0068D812/$File/CWA0820090031ATTACHMENTS.pdf (accessed on 26 July 2022). [149]

US EPA (n.d.), *Durable Goods: Product-Specific Data | US EPA*, https://www.epa.gov/facts-and-figures-about-materials-waste-and-recycling/durable-goods-product-specific-data (accessed on 9 May 2022). [111]

US EPA (n.d.), *Durable Goods: Product-Specific Data | US EPA*, https://www.epa.gov/facts-and-figures-about-materials-waste-and-recycling/durable-goods-product-specific-data (accessed on 11 August 2022). [113]

US EPA (n.d.), *Textiles: Material-Specific Data*, https://www.epa.gov/facts-and-figures-about-materials-waste-and-recycling/textiles-material-specific-data (accessed on 1 September 2021). [82]

VALORPNEU (2018), *TABELA DE ECOVALOR*, https://www.valorpneu.pt/tabela-de-ecovalor/. [179]

Valpak Ltd. (2021), *Extended Producer Responsibility: what you need to know*. [209]

Valumat (n.d.), *FAQ*, https://valumat.be/en/faq/what-are-the-environmental-contributions-for (accessed on 15 November 2022). [95]

Van Der Linden, A. and A. Reichel (2020), *Bio-waste in Europe — turning challenges into opportunities — European Environment Agency*, European Environment Agency, https://www.eea.europa.eu/publications/bio-waste-in-europe (accessed on 15 November 2021). [151]

Vandermeersch, G. et al. (2015), "A critical view on microplastic quantification in aquatic organisms", *Environmental Research*, Vol. 143, https://doi.org/10.1016/j.envres.2015.07.016. [65]

Viool, V. et al. (2018), *Study to support impact assessment for options to reduce the level of ALDFG*. [63]

Vlakglass Recycling Nederland (2022), *Flat glass recycling Netherlands - Overview recycling contribution*, https://www.vlakglasrecycling.nl/index.php?page=overzicht-verwijderingsbijdrage-nl (accessed on 1 August 2022). [138]

Watson, D. et al. (2018), *Used Textile Collection in European Cities*, ECAP, http://www.wrap.org.uk (accessed on 2 September 2021). [85]

Watson, D. et al. (2016), *Exports of Nordic Used Textiles: Fate, benefits and impacts*, TemaNord, Nordic Council of Ministers, Copenhagen K, https://dx.doi.org/10.6027/TN2016-558. [86]

Wijayasundara, M. (2021), *5 barriers to using recycled materials to boost the circular economy*, The European Sting, https://europeansting.com/2021/12/13/5-barriers-to-using-recycled-materials-to-boost-the-circular-economy/ (accessed on 22 March 2022). [171]

World Bank (n.d.), *Population, total - Japan | Data*, https://data.worldbank.org/indicator/SP.POP.TOTL?locations=JP (accessed on 1 April 2022). [222]

Worldloop (n.d.), *Our Partners: Recupel*, https://worldloop.org/?partners=recupel (accessed on 6 April 2022). [195]

WRAP (2014), *Evaluation of the end markets for textile rag and fibre within the UK*, https://wrap.org.uk/sites/default/files/2021-03/WRAP-evaluation-of-end-markets-final.pdf. [81]

Yamaguchi, S. (2021), "International trade and circular economy - Policy alignment", *OECD Trade and Environment Working Papers*, No. 2021/02, OECD Publishing, Paris, https://doi.org/10.1787/ae4a2176-en. [182]

Yu, K. (2017), *Volume Based Waste Fee(VBMF) System for Municipal Solid Waste*, Seoul Solution. [213]

Zhang, C. et al. (2020), "Upgrading construction and demolition waste management from downcycling to recycling in the Netherlands", *Journal of Cleaner Production*, Vol. 266, p. 121718, https://doi.org/10.1016/j.jclepro.2020.121718. [129]

Zhao, Y. et al. (2021), "Does the extended producer responsibility system promote the green technological innovation of enterprises? An empirical study based on the difference-in-differences model", *Journal of Cleaner Production*, Vol. 319, https://doi.org/10.1016/j.jclepro.2021.128631. [18]