EXTENDED PRODUCER RESPONSIBILITY

CASE STUDIES ON BATTERIES, END-OF-LIFE VEHICLES AND MEDICINE IN THE NETHERLANDS

Background document

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SUMMARY

This document contains three case studies on extended producer responsibility (EPR) in the Netherlands. In the first two (ex-post) case studies, we analyse the design, implementation and performance of EPR for batteries and end-of-life vehicles. In the third case study, we perform an exploratory analysis of the potential contribution of EPR to the management and prevention of leftover medicines.

Batteries

The batteries case study distinguishes between three categories of batteries: industrial, automotive and portable ones. Industrial batteries do not have explicit collection or recycling targets, but producers are still required to take back their product. These batteries are mostly part of business-to-business waste and collection seems to be on the right track. Industrial batteries also encompass traction batteries installed in electric and hybrid-electric vehicles. The second category, automotive batteries, encompasses the starting, lighting and ignition batteries used in all motor vehicles. Automotive batteries are highly recyclable, up to 99% for lead-acid ones. Because of their positive market value, they are usually not littered, landfilled or incinerated. Lithium batteries are more problematic with recycling rates of around 50%, however, larger batteries can sometimes be reused in stationary applications. Portable batteries have a 45% by weight collection target based on what has been put on the market on average in the previous three years.

Insights for policy from the batteries case study: The collection target for portable batteries is little informative in growing or declining markets, and thus a new measure based on the amount available for collection has been proposed. The Netherlands achieve a 87% collection rate with this measure with relatively few incentives for collection to consumers. Although industrial batteries are managed well, it is important to keep monitoring this into the future. As the amount of used batteries increase, opportunities for reuse may dry up and lithium batteries are generally harder to recycle. Furthermore, while collection rates are quite high already, they may increase even further with more direct incentives for consumers to separately dispose of them.

End-of-life vehicles

EPR for end-of-life vehicles (ELVs) in the Netherlands relies on two instruments: a take-back requirement, and an advance recycling fee paid at the time of purchase of a passenger car or light commercial vehicle. The combined target for reuse and recycling is currently 85% based on the weight of the vehicle. The recycling fee (currently 30 euro per vehicle) is used to cover the net costs of ELV treatment and those of data collection and reporting, and to support activities that improve ELV recyclability. A challenge is posed by the increasing uptake of electric vehicles, the most important of which is the removal and treatment of lithium-ion traction batteries. The amount of lithium-ion batteries collected by the PRO and reused in stationary applications has drastically increased in the last couple of years, but recycling remains a challenge.

Insights for policy from the end-of-life vehicles case study: In contrast to some other end-of-life products, ELVs are valuable. EPR is mostly needed to ensure that potentially hazardous materials (e.g. batteries, tyres, fluids, airbags and LPG tanks) are properly treated, and that materials with a very low or negative residual value (e.g. some plastics and glass) find a way back in the market. Additional incentives are needed to increase the reuse of parts from ELVs and the high-quality recycling of certain materials (plastics, glass). Incentives to boost the demand for used parts from

consumers are as important as incentives to increase their supply. Instruments for this purpose include public awareness campaigns, warranties on used parts, and obliging garages to provide quotes for used parts. To facilitate high-value recycling of plastics, manufacturers need to provide detailed information on the location of different types of plastics in each vehicle. A narrower definition of recycling – namely excluding backfilling from its scope – would also incentivise higher-value applications. Coupling these policies with recycled content requirements is likely to increase the effectiveness of the policy mix in promoting high-value recycling. It is also worth considering introducing mandatory EPR to other motor vehicles, namely motorcycles, scooters and heavy-duty vehicles.

Medicines

Currently, no EPR for medicines exists in the Netherlands. Medicines can cause serious environmental and health problems when improperly disposed of. Examples are antibiotic resistant bacteria but also medicine traces entering our drinking water and the food chain trough livestock or the irrigation of crops. Leftover or expired medicines can be dropped off at pharmacies, but that system is currently not financed by producers.

Insights for policy from the medicines case study: The introduction of EPR to medicines can help lower environmental pollution and health risks. EPR would make producers financially responsible for the separate collection and appropriate treatment of leftover and expired medicines. EPR could also provide producers incentives to design more environmentally friendly products. Furthermore, EPR could help finance the installation of water treatment facilities in hospitals, pharmacies, and other major suppliers of medicines (and traces), which seems to be a promising way to lower concentrations in surface water. Further research on this front is advised. Increasing the number of drop-off points for leftover or expired medicines (at supermarkets, for example) seems to be a low-cost measure to increase awareness and to make separate disposal easier.

General policy insights

From these case studies we can draw some general lessons. On the consumer side, easy access to collection points is key for high separate collection rates. High collection rates are important for lowering the amount of a product being littered. Apart from that, public awareness campaigns may be an effective way to promote proper disposal and increase collection rates, ensure more homogeneous and higher-quality streams of end-of-life products, and eventually prevent environmental damages.

On the producer side, setting realistic and insightful targets is important. This became clear in the batteries case study where the current measurement of collection is not insightful. Steering producers to more circular practices requires that targets provide incentives in the right direction. EPR would be more effective if producers were made responsible also for the share of products that is not separately collected. When their responsibility is limited to separately collected product streams, they have no incentive to collect and treat more than what is required by the targets.

Introducing EPR to other product groups, including medicines, is likely to prevent environmental damages from littering and illegal dumping. EPR shifts the financial burden of collecting and treating end-of-life products from taxpayers to producers and consumers and is consistent with the Polluter Pays Principle. The environmental and economic effects of EPR eventually depend on its design, mainly on the characteristics of the instrument mix used.

1 Introduction

This document contains three case studies on extended producer responsibility (EPR) in the Netherlands. We study the implementation and performance of EPR for batteries and end-of-life vehicles in the first two ex-post case studies. These sectors were chosen because they have been implemented EU wide and we felt that they are relatively understudied when it comes to EPR compared to Waste Electrical and Electronic Equipment (WEEE). In the third case study on medicines, we perform an exploratory analysis where we consider what the introduction of EPR could contribute to the waste management and prevention of leftover medicines. Medicines that are not disposed of properly can have serious adverse effects for health and environment, concentrations of medicines in water surfaces are increasing rather than decreasing, increasing the need for extra policy.

The case studies in this document are part of a wider research project into EPR. In a separate document we present our more general findings on EPR, the mechanisms through which it operates and the effects that it has on producers and consumers (Dimitropoulos et al., 2021). That document also contains a brief background on what EPR is and to what sectors it has been applied. The current document provides product-specific analyses from which the more general document can draw examples.

The case studies are based on a survey of the literature and interviews with stakeholders. We have consulted both the academic literature on EPR and Dutch and international policy documents. Interviews were conducted with experts from sector organisations and producer responsibility organisations (PROs), such as Stichting Batterijen, Stibat, Autorecycling Nederland (ARN) and Koninklijke Nederlandse Maatschappij ter bevordering der Pharmacie (KNMP) to retrieve insights from within the studied sectors. A full list of the organisations interviewed for the case studies is provided in the Appendix of this document. Lastly, data from Rijkswaterstaat, Eurostat, several Ministries and PROs were used to provide quantitative figures and tables to support the texts.

The report is structured as follows: Chapter 2 contains an ex-post case study on EPR for batteries in the Netherlands, and Chapter 3 contains an ex-post case study on EPR for end-of-life vehicles in the Netherlands. Following the accompanying EPR document (Dimitropoulos et al., 2021), these sections are structured consecutively around the background, EPR design and operation, and effects. Chapter 4 contains an exploratory study on EPR for medicines. This section is structured in a different way, given that for this product group EPR has not been implemented yet. In Chapter 5, we provide policy implications based on the findings of the case studies.

2 Batteries

In this case study for batteries we show how EPR (extended producer responsibility) for batteries is organised and how it performs in the Netherlands. We assess how EPR for batteries is designed and what EPR policy instruments are used to address market failures. The focus will be on portable batteries as the EPR is mostly aimed at that category of batteries. Finally, we compare the implementation of EPR in the Netherlands with other countries as well as differences in their performance.

This case study will mostly be based on insights from literature on EPR for batteries specifically and on interviews with experts and stakeholders. We do not have data or a proper counterfactual to do empirical analysis with. The texts in this case study have benefited from interviews with the following organisations: Ministry of Infrastructure and Water Management, Rijkswaterstaat, Stichting Batterijen and Stichting Stibat services.

2.1 Background

Failing to collect and recycle batteries or littering them can be very harmful for the environment. Batteries contain heavy metals and possibly toxic chemicals which can hurt plants, animals and people when littered. Collecting and recycling allows for the reuse of materials, lowering the amount of virgin materials needed and prevents them from getting incinerated together with other types of waste (Stibat, 2017).

EPR for batteries and accumulators[1] (abbreviated to batteries from now on) was introduced in the Netherlands in the year 1995. The European legislation came into effect in 2008[2], two years after the EU Directives on 'Batteries and Accumulators' and 'Waste Batteries and Accumulators' of 2006 facilitated introduction of EPR by the member states (VROM, 2008). Following the EU Directive, responsibility for battery design, manufacturing, collection, processing and recycling are specified nationally in the Dutch regulation management ('Regeling beheer batterijen en accu's 2008' in Dutch) of batteries from 2008 (VROM, 2008). The regulation translates the European Directive to the Dutch circumstances, but does not deviate from its goals or targets.

As of writing this, the EU is writing a new concept guideline for batteries (as announced in European Commission, 2020). The European Commission wants to harmonise EPR policy in EU Member States so as to reduce administrative costs for businesses. But member states prefer more control at the national level, such as being able to set their own collection targets. A number of member states have raised concerns or issues regarding the legal basis and scope of the new guidelines as well as other concerns over them (2021Portugal.eu, 2021; Ministry of Infrastructure and Water Management and Ministry of Economic Affairs and Climate Policy, 2021). The new regulation will come into force on the first of January 2022.

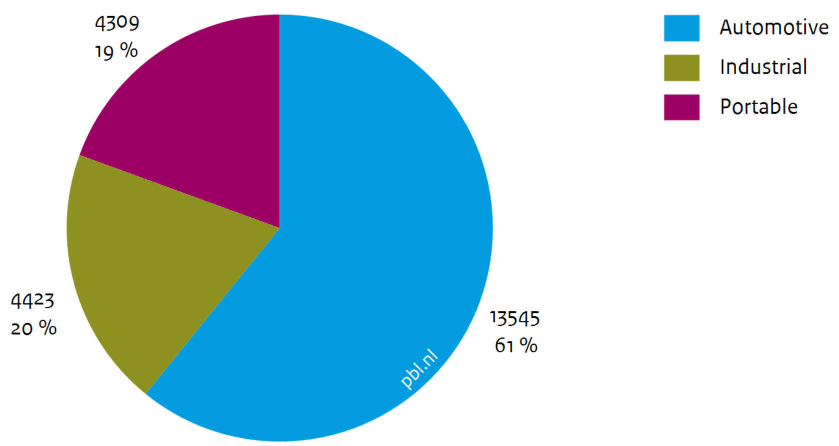
The implemented regulations distinguish between and vary over different types of batteries. There are three categories: industrial, automotive and portable. Industrial batteries are batteries that are only used for industrial or professional tasks or are used to power electric vehicles. Automotive batteries are batteries used for starting, lighting or to power the ignition of non-electric vehicles.

[1] Batteries and accumulators have the same definition in the Dutch regulation, hence the abbreviation in this document (VROM, 2008).
[2] Stichting Batterijen, the PRO (producer responsibility organisation) for EPR for batteries, was founded by producers and importers of batteries in 1995 to comply with national policy.

Portable batteries are batteries that are enclosed, can be carried by hand and are not industrial or automotive batteries (VROM, 2008). Figure 2.1 shows the share that the different categories had in the amount of batteries collected in 2018 by weight, automotive and industrial batteries (including electric car batteries) make up the majority of collected material. The share of this last category is expected to increase in the future with the rise of electric vehicles.

Figure 2.1

Collected batteries per category in 2018 (in tonnes and %)



Source: RWS

2.1.1 Battery collection and recycling

This distinction is important for EPR because only portable batteries have a legal take-back and recycling target set by the European Union. The minimum collection rate for portable batteries is 45%, moreover collected batteries also have a legal recycling target. The recycling target level varies for different battery chemistries: 65% for lead-acid batteries, 75% for nickel-cadmium batteries and 50% for batteries with other chemistries such as lithium-ion batteries (EP, 2006). The recycling percentages are calculated by dividing the output fraction (mass of resources after recycling) over the input fraction (mass of batteries). The collection target is relative to the average amount of batteries put on market in the current and previous two years. There are requirements to facilitate take back for industrial and automotive batteries, but there are no explicit targets for these categories.

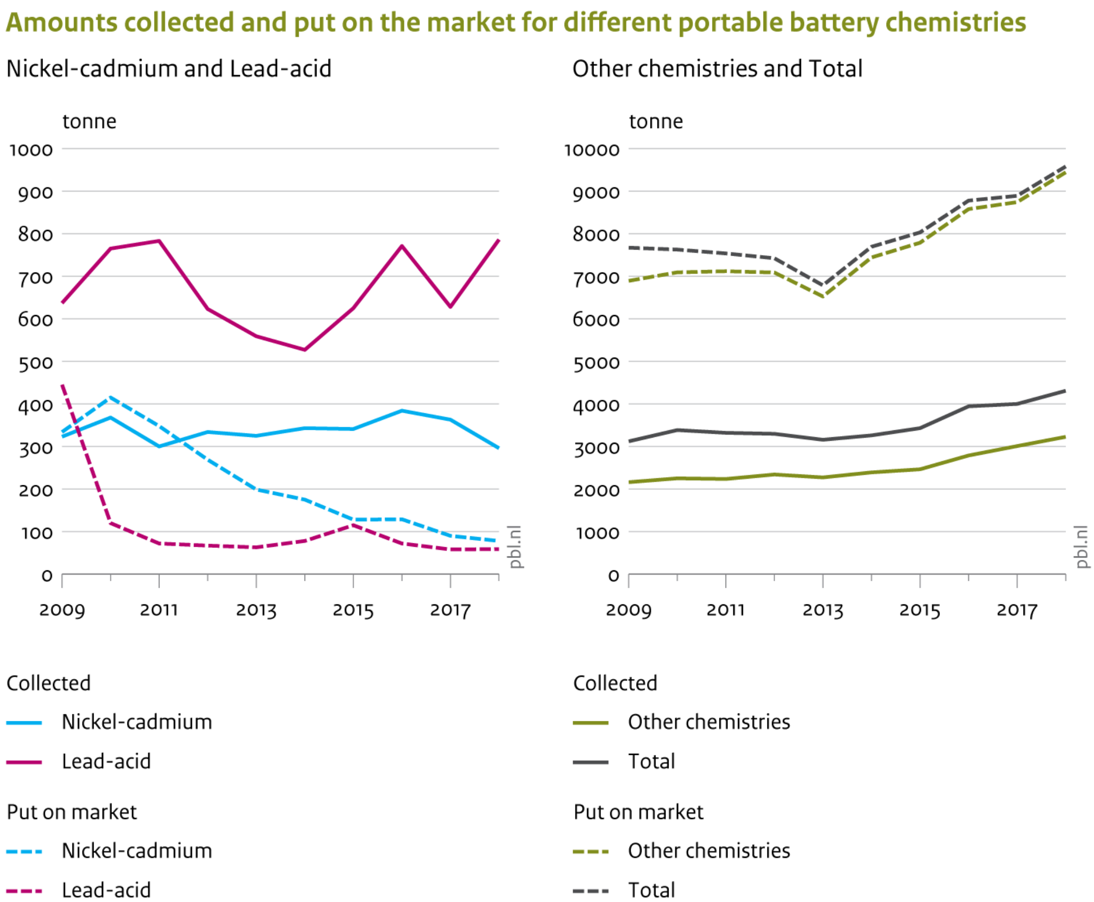
The collection of industrial batteries seems to be going fairly well, even without explicit collection targets. These batteries usually do not end up in the hands of consumers and thus are less likely to be littered or mixed with other household waste. Producers of electric vehicles (EVs) pay an upfront fee to cover the recycling costs of EV batteries so that consumers are not confronted with extra costs when discarding their old vehicle. For other automotive batteries the market is able to handle collection and recycling because these batteries have a positive market value, they are easily recycled and the metals can be sold at a profit. One problem is that non-registered companies also try to collect these types of batteries to make a profit from recycling. These companies generally do not take the correct measures to prevent environmental damages during the recycling process, such as preventing propellants from entering the atmosphere.

According to BIO Intelligence Service (2014), only a small fee from producers is needed to cover the costs for the management of portable batteries. The collection and treatment costs of industrial batteries (which includes electric vehicle batteries) can almost be covered by the recycling revenues, although this doesn't go for industrial lithium-ion batteries. Lithium-ion batteries, that are also used in electric vehicles, are now often reused in a stationary setting. The batteries may not provide the best range in an EV any more, but can still function as home batteries for people with solar panels that want to store the energy they generate during daytime. There are also commercial projects where old EV batteries are given a stationary function. This form of reuse can be seen as a better strategy than recycling from an environmental perspective. However, it is unclear whether there will be enough of these types of use cases for old batteries into the future as the supply of them will increase following the increase in electric vehicles being sold.

This case study will henceforth be focussed on portable batteries.

Figure 2.2 shows the amounts collected and put on the market from 2009 until 2018 for portable batteries of different chemistries. In the upper panel we see for both NiCd and Lead-acid batteries, that since 2012, more old batteries are being collected than new batteries are put on the market. In the lower panel we see that for both batteries of 'Other chemistries' and the total of all portable batteries, the opposite is true and more new batteries are put on the market than are being collected. We can also see that almost all batteries put on the market fall into the category 'Other chemistries', these are presumably mostly lithium batteries.

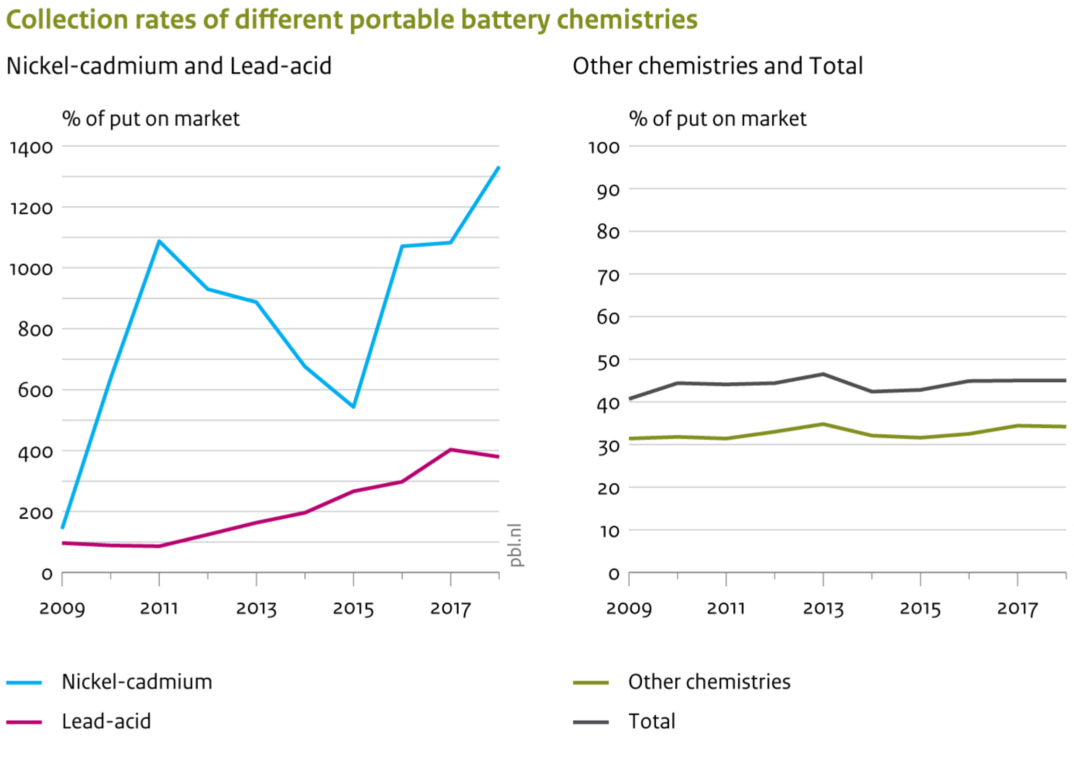
Figure 2.2



Source: RWS

Figure 2.3 shows the realised collection of batteries as a percentage of what was put on the market in the current and previous two years for 2009 until 2018. The high collection rates of over 100% are possible when a certain type of battery becomes less commonly used in new goods. If the batteries have a long lifetime, the amount put-on-market goes down before the amount collected goes down. It is then possible that in a given year, more old batteries are collected then are newly sold as is the case for Nickel-cadmium and Lead-acid batteries.

Figure 2.3



Source: RWS

The lower panel of the figure shows that the collection target of 45% for all portable batteries is being met in some years but not by a wide margin. The collection target is set as a percentage of the average amount put on the market in the last 3 years but this can be problematic for battery types that are increasing in popularity because the amount put on the market will rise before the amount with end of life status goes up. Also, batteries inside electronics cannot always be removed and are thus missed in the collection process.

2.2 EPR design and operation

In this subsection we discuss the organisation of the EPR in the Netherlands as well as different instruments that are used in the EPR scheme for batteries and how they perform. The EU has set collection and recycling targets for batteries. The PRO uses a fee that it charges to producers and importers, the fee helps financing the EPR activities of the PRO.

2.2.1 Organisation

The current EPR system for portable batteries in the Netherlands is a collective scheme with a single producer responsibility organisation: Stichting batterijen. From 1995 until 2017, an organisation called Stibat organised the waste management for portable and industrial batteries. In 2017, Stibat was split up in Stichting Batterijen, the PRO with responsibility for portable batteries[3], Stichting EPAC with responsibility for e-bike batteries and Stichting Stibat services which is the organisation that does collection, sorting, transportation and administration for both Stichting Batterijen and Stichting EPAC. Their task is to fulfil the waste management responsibilities of the portable battery producers and importers that they represent (1015 in the year 2018 (Stibat, 2018)). The non-profit organisations provide a national network of collection points for both batteries from e-bikes and portable batteries. Since there are legal collection targets for portable batteries, they also communicate towards consumers to inform them on the importance of the correct disposal of batteries.

All points of sale for batteries (such as supermarkets) must accept the return of used batteries. Stibat has around 25,000 collection point where batteries are collected. The actual collection is outsourced to two transportation companies that collect the batteries and transport them to the sorting facilities. The batteries are sorted according to their chemistry and are then shipped to recycling facilities in neighbouring countries as there are no battery recyclers active in the Netherlands. Stibat services checks these recyclers with an eco-test. This test looks at key performance indicators of the recycler with regards to toxicity-handling, circularity and recycling. The eco-test was developed with an independent organisation that tests recyclers' recycling rates, amount of material reuse, and their toxicity-handling and carbon footprint.

PROs also invest in consumer awareness on proper disposal. This helps them in achieving their collection and recycling targets and prevents littering or otherwise wrong disposal of batteries. It also lowers the environmental costs of improper disposal and thus benefits society at large.

Literature suggests that a single PRO in a country would perform better than multiple competing PROs. With multiple PROs, incentives to compete for batteries with certain chemistries, can hurt resource efficiency (Stahl et al., 2018). This will occur if recycling of some types of batteries generates more profits from selling secondary raw materials[4] than other types. From the perspective of governments and the inspection, a single PRO to work with also is more efficient.

2.2.2 Take-back requirements

With the Batteries Directive 2006/66/EC, a take back requirement was introduced for batteries. More specifically, a collection target for portable batteries of 25% in 2012 and 45% in 2016 of what has been put on the market during the last three years needs to be collected every year from 2016. Figure 2.4 shows the target and the historical performance of the Netherlands.

While the collection rate has slowly increased since the introduction of EPR in 2008, it is unclear what role EPR policy has played in this trend. The upward trend started before 2008, suggesting that either producers were anticipating the introduction of the EPR target and already started increasing their collection efforts, or that EPR has played no significant role in the increased collection rates. Because the trend has slowly continued upwards after the introduction of EPR like it had been doing before the introduction. The lack of a credible counterfactual makes it difficult for us to evaluate EPR in this regard. In the future this target is supposed to go up, the proposal for EU battery regulation is for the target to be raised to 65% in 2025 and 70% in 2030 (European Commission, 2020).

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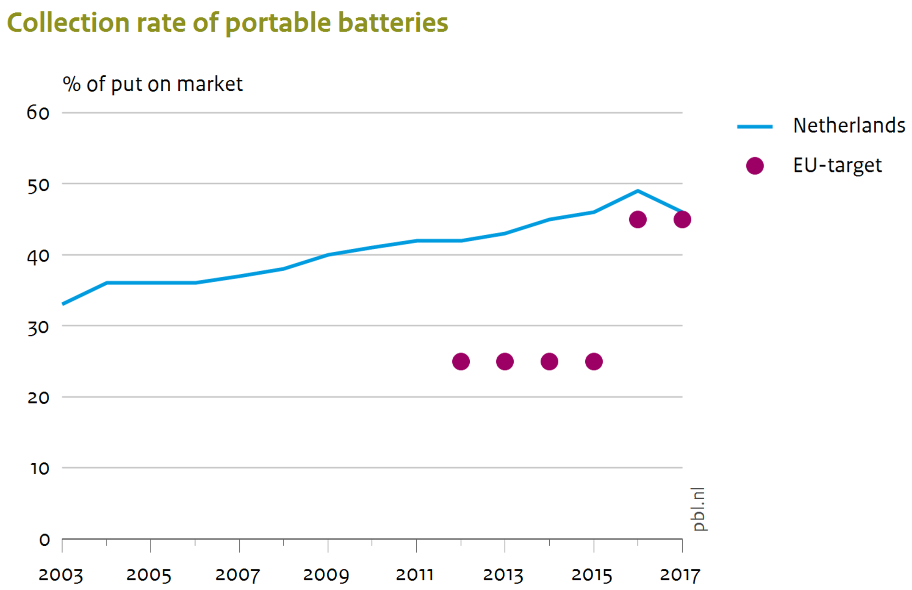
[3] Stichting Batterijen also occasionally collects industrial batteries. The collection and reuse of industry batteries are organised separately.

[4] Secondary raw materials come from recycling end-of-life products, as opposed to primary materials which come from mining, drilling or harvesting.

The sector and the Dutch Ministry of Infrastructure and Water Management would prefer a different measure. That one should be more based on the amount of batteries that are actually available for collection (Werkgroep Beoordeling Nieuwe Commissievoorstellen (BNC), 2020).

Collection rates can be misleading and their exact definition is important when evaluating the performance of EPR policies. Remember that collection rates are measured based on the amount put on market**.** If one would take a different measure, for instance the percentage collected based on batteries available for collection, a different image appears. The Netherlands then reach a collection rate of 87%, Belgium reaches 90% and France around 70% (according to EUCOBAT, 2018). The definition used for 'available for collection' is based on the quantities put on market and the known lifespan to calculate the amount of batteries that have reached 'end-of-life' status. Subtracting the amount that was exported or ended up in waste electrical appliances[5], one then reaches the quantity 'available for collection' (EUCOBAT, 2019). This mitigates the problems with high or low collection rates in declining or growing markets. The European Commission has put forward the intention to review their calculation methodology to be focused on a more 'Available for Collection' approach (European Commission, 2020).

Figure 2.4



Source: European Portable Battery Association

This performance is quite remarkable since consumers have no financial incentive to return their batteries. That suggests that these collection rates are possible mostly due to the intrinsic motivation of consumers to return their empty batteries. If collection rates fall in the future or higher

[5] Batteries that are left inside of waste electronics are currently removed and separately processed in the Netherlands (Wastenet, 2019). Still, these batteries do not count towards the collection target for Stichting batterijen to comply with the European EPR directives. A questionnaire in Japan finds that on average 76% of respondents did not remove the battery of discarded products (OECD, 2014). All built in batteries do count towards battery put-on-market and collection if their device is being discarded correctly as producers and importers have to include those batteries according to the AVV that is active.

collection rates would be preferred, direct incentives for consumers can be considered. This could be in the form of direct financial incentives or more active information campaigns for example.

To assess performance, cross-countries data have to be used. Detailed data on the cost structure of PROs is not publicly available. Hence, aggregate country data has to be used to compare performance between countries. Measuring cost effectiveness can be done by comparing collection fees and collection rates between countries as has been done by Stahl et al., 2018. Their numbers are presented in Table 2.1.

Focusing on the Netherlands first, the data show that the collection rate for portable batteries has increased over time. At the same time, the total fee paid has increased, the fee per ton collected has gone down however, suggesting that the cost efficiency went up as scale increased. We do not know if this relationship is causal as other factors may have played a role as well.

The fees vary widely between countries and so do the collection rates. Belgium and Switzerland both score around 70% in 2016 but also have far higher fees per ton collected portable batteries. For Belgium, part of the higher fee can be explained through the high investments in public awareness campaigns, the costs of which amount to over 20% of the total fee (Stahl et al., 2018). But in general it seems from this data that collection costs rise steeply when a country tries to exceed a collection rate of much more than 45%. Raising the question of whether it is socially beneficial to make this extra effort.

Table 2.1
Collection fees and rates for portable batteries in five countries in 2011 and 2016

	Year	Netherlands	Austria	Belgium	France	Switzerland
Portable batteries collected (ton)	2011	3,385	1,738	2,406	17,397	2,375
	2016	3,946		3,153	13,677	2,804
Collection rate (%)	2011	42	49	52	36	72
	2016	49.0		70.7	46.4	67.8
Collected port. batt. per inhabitant (kg/ year)	2011	0.199	0.198	0.214	0.260	0.283
	2016	0.232		0.280	0.204	0.334
Total fee (EUR millions)	2011	5.400	1.987	21.810	11.300	12.050
	2016	8.610		17.674	15.586	14.231
Fee per inhabitant (EUR/ year)	2011	0.32	0.23	1.94	0.17	1.43
	2016	0.51		1.57	0.23	1.69
Fee per ton collected port. batt. (EUR/ year)	2011	1,595	1143	9,065	650	5,074
	2016	2,182		5,605	1,140	5,075

Note: Prices are not corrected for inflation, last row corrected by CPB and PBL.
Source: Stahl et al., 2018.

2.2.3 Advance disposal fees

In the Netherlands, there is no advance disposal fee for consumers for batteries set by the government. The PRO for batteries, Stichting Batterijen, charges fees that producers and importers pay when they put batteries on the market to pay for the waste management of their batteries. These fees are implicitly payed by consumers. The fees is dependent on the type of battery (based on its recyclability) and its weight as Table 2.2 shows.

Fees for Lithium batteries are higher than fees for other battery chemistries as can be seen in the table above. This is consistent with the observation that lithium batteries are less recyclable, which means that there are less profits from selling secondary materials and that more incineration fees have to be paid. These incentives steer producers towards using easier to recycle products, lowering the environmental burden of their products. The practice of using differentiated tariffs is in accordance with recommendations in theoretical literature but not yet widely implemented in other sectors in practice. The battery fee modulation is an example where differentiation in tariffs has been successfully implemented in practice. It is a valuable example for other EPR schemes where modulation is not yet used and for governments testing which policies are most effective.

Modulating fees can be used to incentivise producers to change their product design. Eco-design is aimed at changing the design of products in such a way that the product's environmental footprint is lowered. This can be done by lowering fees for products that use more well recyclable materials or by lowering fees for products that have a higher content of secondary (recycled) materials. The fee for lithium batteries is higher than non-lithium batteries and thus stimulates the use of other battery types which generally achieve higher recycling rates as we have seen before. The PRO has an incentive to differentiate those fees because it has to be cost efficient and it is a non-profit organization. Since lithium-ion batteries are more expensive to recycle, it makes sense for them to charge a higher fee for those types of batteries.

Table 2.2

Stibat waste management fee for portable and industrial batteries in the Netherlands

Battery	Waste management fee in euros (VAT excluded)	Waste management fee lithium in euros (VAT excluded)	Waste management fee non-lithium, non-lead in euros (VAT excluded)
period	2011-2015	2016 - 2020	2016 - 2020
<51 gram, non-button	0.02	0.02	0.017
51<151 gram, non-button	0.06	0.10	0.09
151<251 gram, non-button	0.13	0.20	0.16
251<501 gram, non-button	0.27	0.36	0.33
501<701 gram, non-button	0.40	0.60	0.45
701<1000 gram, non-button	0.54	0.92	0.64
Button cell batteries	0.0003	0.005	0.002
Portable batteries heavier than 1 kg	0.60	2.37	1.23

Note: Prices are not corrected for inflation.
Source: Stibat 2019.

2.2.4 Reporting

Stichting Batterijen presents data on their performance to Rijkswaterstaat, a government organisation that collects data for all EPR schemes in the Netherlands. ILT, the inspection organisation, enforces the laws around EPR and can fine non-compliant businesses and underperforming EPR schemes. Even if producers organise themselves together in a PRO, the individual producers are still held accountable for the performance of the EPR scheme. Stichting Batterijen is only required to report on their EPR results through environmental performance numbers but not on other aspects of their operation such as their cost structure for example. This limits insight into their cost efficiency.

2.3 Effects of EPR for batteries

2.3.1 Waste management

A 'generally binding agreement' (Algemeen verbindend verklaring or AVV) is a unique arrangement in the Netherlands, as it can temporarily formalise a sector initiative for EPR. This means that all producers or importers of a certain product must contribute a waste management fee to the EPR scheme that successfully applies for AVV (IenW, 2018). This includes producers or importers who, prior to the AVV, contributed to a different EPR scheme. There are conditions that must be met before a sector initiative can be made generally binding by means of an AVV. The most important conditions are that the companies that wish to apply for an AVV must have at least a 75% combined market share, show that the arrangement is in the interest of effective and efficient waste management and lastly that it can ensure the continuity of the waste management system. In 2018, Stichting batterijen has successfully applied for an AVV for portable batteries, which means that producers and importers are legally required to join their EPR scheme. AVVs last for five years, after which they have to be renewed with a new application.

For most batteries, a majority of the materials can be recycled or reused. Lead-acid batteries are 99% recyclable while nickel-cadmium batteries are around 70% recyclable[6]. Lithium batteries, which have seen a large growth in their popularity and use over the last few years, are usually only 50% recyclable. For this last category, it's usually metals such as aluminium, copper and iron that are recyclable. The cathode can be reused in new batteries after going through some processing (Gaines et al., 2014). The largest difficulty in recycling stems from the wide range of materials used in battery cells which must all be separated, and the higher number of cells that a lithium-ion battery typically consists of in comparison with batteries of other chemistries (Gaines. 2014). Harper et al. (2019) give a detailed description of the recycling process of lithium-ion batteries from electric vehicles.

The Dutch ministry responsible for EPR at the time, arranged for a study on EPR in 2007 where they asked a number of PROs to share data on their financial performance for 2004 and 2005 (NVRD, 2007). The data are shown in Table 2.3 and the net result was calculated by subtracting the expenses from income. The PRO was able to cover their costs in both years with the collected fees and remaining income, this is to be expected as the PRO can set the fee themselves. Furthermore we know that Stibat spent about 40% of 'Other costs' on publicity campaigns at that time. Stibat uses these campaigns to raise awareness among consumers in order for the sector to reach its collection target. During these years 2 kton of end-of-life batteries were collected under the EPR for batteries,

[6] Note that these batteries are becoming less popular over time as can be seen by the high collection rates (over 100% of the amount put on the market).

in 2009 this was 3.1 kton while the amount grew further to 4.3 kton in 2018. More recent financial data are not available.

Besides the incentives for producers of batteries through differentiated PRO tariffs, there are no financial incentives aimed at consumer behaviour. It is quite remarkable that collection rates of 87% are reached just through the intrinsic motivation of consumers to discard them separately from other household waste and the strong network of collection points. EPR policy has led to an increase of these collection points because it made it mandatory for stores that sell batteries, to also take old batteries back from consumers. If collection rates fall in the future or if higher collection targets are brought into place, then introducing financial incentives for consumers can be a way in increase collection. A downside is that financial incentives may harm the intrinsic motivation of some consumers.

Table 2.3

Income and expenses for Stibat in 2004 and 2005 in Euros

Category	Income		Expenses	
year	2004	2005	2004	2005
PRO Fees	6,741,000	7,488,000		
Other income	355,000	273,000		
Collection			2,761,000	2,798,000
Other costs[1]			4,039,000	4,570,000
Total	7,096,000	7,761,000	6,800,000	7,368,000
Net	296,000	593,000		

[1] Examples of other costs are salaries, housing, R&D subsidies and publicity.
Data source: NVRD (2007).

Waste management of batteries abroad

The performance of EPR for portable batteries is monitored using different metrics by the European Portable Battery Association (EPBA). They provide details with data from all European EPR-schemes that they update with a yearly report. Out of the 31 investigated countries, twenty had reached or exceeded the European collection target for portable batteries of 45% while in 2012 that number was only seven (European Portable Battery Association, 2019).

The EPBA reports a 46% collection rate for batteries in 2018 in the Netherlands: a somewhat average score among European countries (European Portable Battery Association, 2019). Nine countries score above 50% while ten score below 40%. The European country with the highest collection rate of portable batteries in 2018 was Croatia with a remarkable 101%. The lowest collection rate was for Romania with 26%.

Outside of Europe, Japan was estimated to achieve a 26% collection rate by Asari et al. (2011). The Updated Guidance for EPR by the OECD (2016) contains a case study on the EPR scheme for used rechargeable batteries in Japan. Although a balance sheet for the Japanese PRO is available, information about their cost structure or individual spendings is not available. A cost efficiency analysis was therefore not carried out.

The main issue in Japan is the still low percentage of end of life batteries that are collected and sorted. The OECD (2016) case study mentions a lack of incentives for stakeholders other than producers to contribute to the collection goals. It does however mention that information campaigns and similar activities have been successful at raising the quality of collected batteries.

Problems on free riders are also mentioned. All battery producers and importers are members of the PRO for batteries but many producers of electronics are not. Because a Japanese survey (OECD, 2014) found that most people do not take the batteries out of their products when discarding them, many batteries that end up in waste, are not covered by the manufacturers. Built-in batteries do fall under the Dutch AVV and producers of electronic devices with built-in batteries do have to pay a fee to Stichting Batterijen when putting their product on the market.

Free-riding is a problem that plays on another level as well. If batteries are exported outside of the EU, either by themselves or built-in in consumer electronics, then the producer responsibility ceases to exist. This means that second-hand products containing batteries that are exported outside of the EU, may be improperly discarded and end up in the environment in the importing country.

2.3.2 Secondary materials and recycling

Environmental externalities may take place during different stages of a product's lifetime. The first stage is mining of the resources that are later needed to produce the batteries. A battery can be made with primary resources that are mined from the earth or with secondary resources which come from end-of-life products and are recycled so that the materials can be used again in new batteries. The externalities stemming from mining primary resources are generally larger than those of recycling secondary resources from waste materials, meaning that switching to the latter could lower pollution and improve social welfare. The private costs to use secondary raw materials are often larger than the costs for using primary materials impeding the adoption of more secondary materials in production processes, although the many metals used in batteries are often well reusable. EPR policies may help lowering the costs for these secondary raw materials, increasing their use (Verrips et al, 2019).

Producing batteries out of recycled materials is less environmentally damaging. Specifically for lithium batteries, currently used mining practices are water intensive and produces large volumes of waste (Flexer et al., 2018). Most lithium is produced from brines which are evaporated so that the solids, including lithium, can be collected. Extracting of lithium is chemical intensive and generates large volumes of waste material (Flexer et al., 2018). The water intensiveness of the mining process is a problem, especially since lithium mines are located in deserts, such as the Atacama Desert in Chile where vast quantities of water are needed for the industrial processes leading mining projects to increasingly locate near nature conservation areas where water is still prevalent (Romero et al, 2012). Recycling batteries could reduce the amount of virgin lithium that needs to be mined. However, lithium from batteries is still rarely recycled in practice.

There are different recycling techniques, each with their own efficiency and externalities. In a Life cycle analysis (LCA) study for portable lithium-Ion batteries, Boyden et al, (2016) compare the efficiency of different recycling processes that are employed around the world. The four recycling processes that are distinguished are: Pyrometallurgical (mostly used in Europe), Mechanical (mostly used in Europe), Hydrometallurgical (mostly used in Asia and North America) or a combination of the previous three. Not all materials are recoverable with all processes, hydrometallurgical recycling allows for recycling more types of materials than pyrometallurgical recycling but also causes more emissions and presents a stronger environmental burden (Boyden et al, 2016).

Take back requirements do not have a direct effect on the usage of recycled inputs. Secondary resources are often too expensive for their (usually) lower quality compared to virgin alternatives and sometimes supply may not be steady enough for producers to want to incorporate them in their production process. In the case of batteries, they contain many metals which generally are well recyclable, as became evident from the relatively high recycling rates that we discussed earlier. EPR through take-back requirements may indirectly contribute to increased usage of secondary raw materials as it formalises the collection and recycling processes and therefore the reliability of secondary material flows.

Scale advantages may take place when processes are organised centrally rather than when they are decentralised. As a result of more efficiency, per unit costs of recycled materials should fall. By realising scale advantages in the waste management sector, recycling could become more feasible. This is relevant because high recycling costs, are a common reason to opt for incineration of waste rather than recycling. According to Bax and Company (2019), more than 90% of cobalt, lithium, manganese, copper and aluminium can be recovered and recycled from large batteries. In practice the percentages are lower for small (portable) batteries due to the still high cost of recycling, something that the increase of scale advantages could change. For portable batteries the recycling rates vary from around 50 to around 70%.

Innovation could also help to reduce costs or increase the quality of recycled materials, but it creates positive externalities for others and is therefore generally supplied at a suboptimal level. Innovation has the potential to reduce costs for collection and recycling processes as well as to improve recycling so that the quality of the resulting product is higher (Verrips et al, 2019). For green innovation there is an extra positive externality because innovation may decrease environmental damages which are not always fully priced. This means that green innovation especially, without policy intervention, is slower than what would be socially optimal (Mot et al., 2018). PROs can set up funds to subsidise green innovation in their sector.

Forms of waste management other than recycling are available but also less environmentally friendly. These options are landfill, stabilisation and incineration. The largest environmental externalities occur with landfilling when batteries end up in the environment. Waste batteries may start leaking depending on their contents and the conditions of the landfill (Bernardes, 2004). Stabilisation is the same as landfilling but with a treatment that prevents contact of metals with the environment. Incineration of batteries can cause, depending of the batteries contents, emissions of mercury, cadmium, lead and dioxins to the environment (Bernardes, 2004).

2.3.3 Cost-effectiveness

It is difficult to give an indication of the cost-effectiveness of EPR schemes. The PROs are not required to give insight into their expenditures and cost structure in the same way that they are required to report on other performance indicators. We know that they finance the collection and waste management, as well as campaigns for consumer awareness, they may subsidise innovation into green product design and they pay their administration costs and overhead costs. We do not know the share of the total budget that PROs tend to spend on each of these different tasks however. The international literature also seems to lack cost-effectiveness analysis, possibly due to the same data limitations and difficulties with constructing a good counterfactual that we ran into.

The current collection rate of 87% for portable batteries in the Netherlands is already quite high. A 100% collection rate is probably too costly, especially given the fact that batteries do not seem to be littered very often, and it is littering where the highest damage occurs. Most of the batteries that are not collected separately will end up on other waste streams such as household waste. This means that the battery will be incinerated rather than recycled but the environmental consequences of that are lower than littered batteries. Given the fact that battery collection today is efficient and does not seem to be much more expensive than in other EU countries, it seems safe to say that increasing cost-effectiveness is not a main priority.

EPR adds administration costs and overhead. This downside of EPR is important when considering the introduction of EPR in a sector. For paper for example, Verrips and Van der Plas (2019) find that collection and recycling rates, the latter being 85%, are already high in the Netherlands, even without formal EPR. This is mainly because recycling paper is profitable due to the relatively high costs of virgin material. Introducing EPR in such a sector would add costs but may not increase collection and recycling rates by much. Like we have seen with some types of batteries (mostly automotive and industrial batteries), if recycling is profitable, then the market will do it automatically.

Whether or not EPR has led to lower waste management costs is uncertain. Our interviews with experts and stakeholders gave a mixed outcome, with some parties suggesting that EPR has indeed lowered waste management costs but others denying those claims.

2.4 Conclusions

In this case study we have seen that the Dutch EPR scheme for batteries has reached a mature status. Stichting Batterijen collects a steady share of the batteries that are put on the market. Although the EPR scheme for batteries applies to all battery types, only portable batteries have a collection and recycling target. The share of collected portable batteries has been above the European target of 25% and later 45% for multiple years now. Measuring it in an alternative way that focuses on the amount of batteries available for collection instead of what has been brought to market, gives a much higher collection rate of 87%. The recycling of automotive batteries is usually profitable as batteries contain many metals which are usually highly recyclable (up to 99% for lead-acid batteries). Economic incentives are presumed to be strong enough to achieve high collection and recycling rates for these types of batteries.

The EPR scheme has been effective in raising consumer awareness, increasing the collection of batteries, providing uniformity in collection and an increased number of collection points. It is difficult however to quantify the effectiveness due to the lack of a good counterfactual. From data by EPBA it seems that the Netherlands are scoring average compared to other European countries. The current European take-back goals are being met.

The collection target for portable batteries (45%) is based on the average amount put on the market during the last 3 years. This means that complying with the target becomes more difficult if the market for batteries is growing (sales will increase before disposal of batteries) but much easier if the market for batteries is shrinking (if sales go down, disposal will decline later). We have seen that this causes collection rates of over 100% for some types of batteries while the sector struggles to reach the 45% target for other types. A collection or recycling target based on the actual amount of available end-of-life batteries would be more insightful and more stable.

Stichting batterijen, responsible for the waste management of batteries in the Netherlands, charges waste management fees to producers of batteries when they put a product on the market. These fees are differentiated according to battery chemistry and weight. This shows that private parties (the market) see differentiated fees as an efficient way to organize their system and serves as an example to other sectors considering implementing differentiated fees for the waste management of their products.

3 End-of-life vehicles

This chapter describes how EPR for end-of-life vehicles (ELVs) works in practice in the Netherlands, assesses its performance, and identifies future directions in which it could more effectively support the transition to a circular economy. The information used in this analysis is collected through desk research, including a review of the relevant literature and online resources, and through an interview with Autorecycling Nederland (ARN), the PRO responsible for the management of ELVs.

3.1 Background

About 6.8 million tonnes of materials are collected from end-of-life passenger cars and light commercial vehicles in the European Union and the United Kingdom every year (Eurostat, 2020). This is the result of the scrappage of 6.5 million vehicles through official channels. Given the high value of the components and materials of these vehicles, and the environmental hazards that could be caused by their improper treatment, it comes as no surprise that passenger cars and vans were the first product groups where EPR – although with a narrower scope of producer responsibility than that defined in recent Directives – was introduced at the EU level. The end-of-life vehicles Directive was adopted in 2000 (European Parliament and Council of the European Union, 2000).[7]

ELVs are among the most valuable waste streams. Many of their components, such as the engine, gearbox and headlamps can be traded in the second-hand spare parts market. Parts that cannot be reused largely consist of metals – mostly iron, aluminium and copper – which are recycled and traded in secondary material markets. Advanced technologies installed in new vehicles generally entail that more value is stored in them waiting to be extracted at the end of their life.

ELVs can pose significant environmental and health risks, however, if not properly treated. These risks stem from various vehicle components and vary in their nature and severity. According to the European Commission's Guidance document on the ELV Directive, about 25% (by weight) of an ELV was characterised as hazardous waste in the early 2000s. This implies that ELVs were responsible for about 10% of the hazardous waste generated annually in the European Union (European Commission, 2005).

Environmental policy was needed to correct the market failure arising from the environmentally irresponsible treatment of ELVs. In the absence of targeted policy action, dismantlers would have no incentive to remove and discard in an environmentally sound manner potentially hazardous elements of ELVs, such as fluids, used oil or batteries. Furthermore, vehicle components with negative residual values would not have been recycled. ELVs would have been stripped of their valuable parts, and hazardous waste would likely have been left behind.

The ELV Directive and national legislation put in place to implement it in EU countries have been the policy response to ensure that ELVs are treated in an environmentally sound manner and that the net economic burden of their treatment is covered by producers and consumers.

[7] Note that the term 'extended producer responsibility' is not used in the text of the Directive.

3.1.1 End-of-life vehicle collection and treatment

The procedure followed for the treatment of an end-of-life vehicle consists of four main stages: collection, dismantling, shredding and post-shredder treatment. In the Netherlands, it starts with the ELV being handed in by the last owner to an authorised vehicle dismantling company, which deregisters it from the Rijksdienst voor het Wegverkeer (RDW). The dismantling company then strips the vehicle of its battery, fluids and tyres. Fuel tanks are also removed from cars running on LPG.[8] Dismantling firms can also opt for removing parts including the motor, gearbox, dynamo and headlamps, as well as glass from the vehicle. These parts are generally removed if considered to have a positive residual value. Mechanical equipment, tyres in good condition and headlamps are stored by dismantling firms to be sold for reuse. Batteries, fluids, glass and unusable tyres are transported to specialised firms who are responsible for their further processing (ARN, 2020a).

The remainder of the ELV is sent to a shredding company, which crushes it and grinds it into fine pieces of metals and other materials. Recovered iron, aluminium, copper and other metals are sent to the metallurgy industry for further processing in order to be used in new products. Other materials continue their trip to post-shredder treatment.

The post-shredder treatment facility used in the Netherlands separates shredder residuals in four streams: metals, plastics, fibres and minerals. Metals follow the same route as those recovered from shredding. Minerals – primarily sand and glass – are mostly used in road asphalt. A fraction of fibres is used as insulation material. Light plastics, such as polyethylene and polypropylene, are sent for recycling. Part of the recycled material is used to manufacture plastic components of new vehicles, such as bumpers, dashboards and hubcaps. The recycled plastic content of the bumpers of some car models reaches 33%.[9] Heavier types of plastics separated by post-shredding treatment replace coke as a reducing agent in blast furnaces. Some types of recovered plastics are used in construction applications. Other types, such as PVC, are incinerated with energy recovery (ARN, 2021a).

3.2 EPR design and operation

The Dutch EPR scheme on end-of-life vehicles is organised collectively and relies on two instruments: a take-back requirement and an advance recycling fee. The take-back requirement is operationalised through two targets: one for reuse (of vehicle parts) and recycling, and another for reuse, recycling and recovery. The definition of recovery encompasses various applications, among which the most widely used is incineration with energy recovery. An organised system for the recycling of motor vehicles has existed in the Netherlands since 1995, although binding EU and national legislation for the implementation of EPR for this product group did not come into force until 2002.

3.2.1 Legislation

The collection and treatment of ELVs is regulated at the EU level through Directive 2000/53/EC (European Parliament and Council of the European Union, 2000)**.** The ELV Directive provides an EU-wide policy framework aiming at the prevention of waste from vehicles, and the reuse, recycling and recovery of ELVs and their components. To achieve these objectives, it requires that systems for the collection and proper treatment of ELVs are put in place by the sector, bans the use of specific hazardous substances in vehicles, sets quantitative targets for ELV component reuse, recycling and

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[8] Dismantling firms must remove all potentially hazardous components of an ELV, including liquids, batteries, oil filters, airbags and LPG tanks, within 10 days from receipt (ARN, 2021c). The rest of the ELV can then remain with the dismantling firm up to a period of three years (ARN, 2018).

[9] This percentage applies to some Renault models.

recovery, and stipulates the provision of information facilitating ELV treatment and its monitoring by producers and other entities involved in the ELV chain. The Directive was implemented in 2002 in most EU countries.

The Directive transfers responsibility for the establishment and operation of systems for the collection, treatment and recovery of ELVs to producers and firms active in the ELV chain. ELVs should be brought to authorised treatment facilities free-of-charge for consumers; producers are responsible for covering all or a substantial part of the costs associated with ELV collection. Producers are also required to provide information about different vehicle components and materials, and about the location of all hazardous substances in the vehicles they put on the market, to facilitate dismantling at end of life. Producers and firms involved in the treatment of ELVs are responsible for achieving the quantitative targets set by the Directive. On this point, the Directive deviates from the definition of EPR, as responsibility does not rest solely with producers, but is instead shared with dismantling and shredding firms and other actors in the ELV chain.

The Directive also attempts to promote eco-design by encouraging the manufacturing of vehicles which are easier to disassemble, reuse and recycle, make increasingly higher use of recycled materials, and contain as little hazardous substances as possible. No quantified requirement is attached to these objectives, but producers are banned from using lead, mercury, cadmium or hexavalent chromium in vehicles put on the market as of July 2003, with the exception of specific applications.

The ELV Directive only applies to a subset of motor vehicles: passenger cars and light commercial vehicles. [10] Motor caravans with a gross vehicle weight up to 3.5 tons also fall within its scope (European Commission, 2005). For other end-of-life motor vehicles, including mopeds, motorcycles and heavy-duty vehicles, no special EU regulatory framework currently exists for their collection and treatment; they are instead covered by the Waste Framework Directive (European Parliament and Council of the European Union, 2018). The original motivations for limiting the scope of the ELV Directive to cars and light commercial vehicles include the relatively low volume of sales of other motor vehicles, the higher presence of small and medium-sized enterprises in their manufacturing, and their longer lifetimes which often end outside the European Union (Williams et al., 2020).

At the EU level, ELVs are generally defined as vehicles discarded or intended to be discarded by their owners (European Commission, 2005). This definition does not provide, however, an objective way to distinguish an ELV from a used vehicle, when the vehicle holder has no intention to discard it. Additional national guidelines exist in the Netherlands for this purpose: a vehicle is considered an ELV if its market value is lower than the costs that have to be incurred in the Netherlands to return the vehicle to a sufficient driving condition (Rijkswaterstaat, 2020a). This definition can be particularly useful to determine whether vehicles prepared for export to other countries fall under the definition of an ELV or of a second-hand vehicle, and has been considered a best practice at the EU level (Williams et al., 2020).

The Directive stipulates annual reuse, recycling and recovery targets for ELVs as a percentage of vehicle weight, whose stringency slightly increased over time. The first set of targets came into effect in 2006: at least 80% of average vehicle weight should be reused or recycled, and 85% recovered. These targets were increased to 85% and 95% respectively as of 2015, as specified by the Directive. Member states are required to collect all vehicles reaching the end of their lifespan, a collection target of 100%. While the ELV Directive also requires member states to ensure that as many of the end-of-life parts removed from repaired vehicles as technically feasible are collected, this requirement is not fixed in national legislation. The methodology that should be used by

[10] Light commercial vehicles are vehicles used for the transport of goods with a maximum gross weight of 3.5 tonnes.

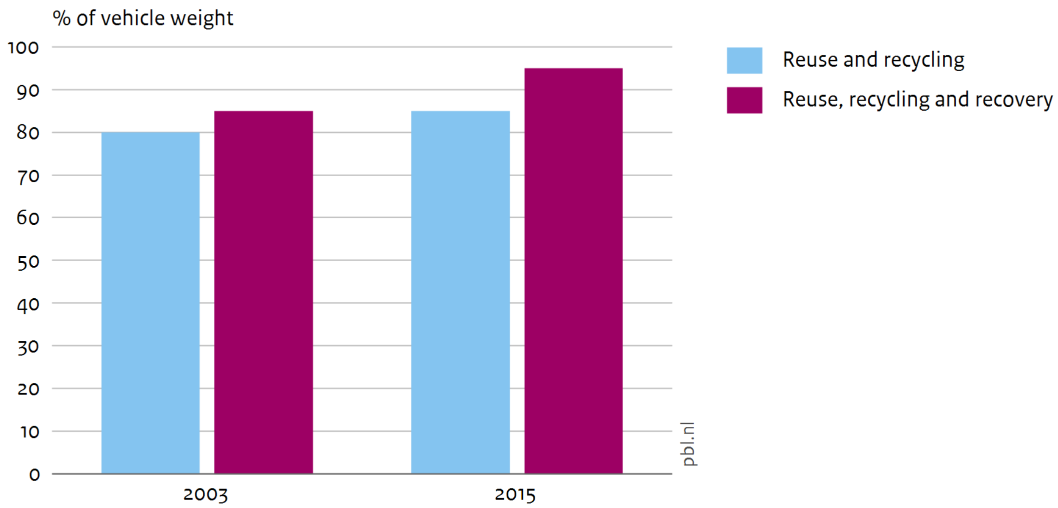
Member States for calculating ELV reuse, recycling and recovery performance is described in European Commission's Decision 2005/293/EC (European Commission 2005)**.**

In preparation of the revision of the ELV Directive, the European Commission recently conducted an evaluation of it (European Commission, 2021)**.**[11] The purpose of this evaluation was to assess the effectiveness, efficiency, coherence, relevance and EU added value of the Directive, and the extent to which it is well suited to handle emerging trends in the car market, including the growing uptake of electric and hybrid vehicles, and the increasing deployment of electronics and materials that are more difficult to recycle, such as plastics and carbon fibre. The consultant report supporting the evaluation of the Directive was published last summer, concluding *inter alia* that the Directive has been effective and relevant, and that it remains a necessary framework for the treatment of ELVs. However, the report also identifies several points for further consideration, including new challenges raised by the widespread uptake of electric vehicles, the identification and treatment of missing vehicles, and the scope of vehicle types covered by the Directive (Williams et al., 2020).

The ELV Management Decree – *Besluit beheer autowrakken* or Bba (Ministerie van VROM, 2002)**, which enacts the ELV Directive into law in the Netherlands, extends producer responsibility for ELV collection and treatment a bit further.** The Decree specifies that producers are responsible for the establishment and operation of a system for the nationwide collection and treatment of ELVs, as well as for meeting the quantitative targets for ELV reuse, recycling and recovery. This implies that the concept of EPR is more precisely introduced in the Bba than in the ELV Directive, where responsibility for the activities above is shared between producers and other economic actors in the ELV chain. The Bba also introduced the annual 80% target for reuse and recycling and 85% for recovery in 2003, three years earlier than the deadline imposed by the ELV Directive. The 2003 and 2015 targets for ELV reuse, recycling and recovery in the Netherlands are shown in Figure 3.1.

Figure 3.1

End-of-life vehicles' reuse, recycling and recovery targets in the Netherlands



Source: VROM, 2002

An organised system for the collection and treatment of end-of-life vehicles has been existing in the Netherlands since 1995, much earlier than the introduction of EU legislation. In fact, the EC's proposal for the ELV Directive was inspired by the system introduced in the Netherlands in the 1990s

[11] The revision of the ELV Directive is scheduled for 2022 (European Commission, 2021)**.**

(Fergusson, 2007). The system was operating on a voluntary basis before the adoption of the Bba. The targets for reuse, recycling and recovery were also achieved by the system much earlier than required by the Dutch legislation: a recovery rate of more than 85% was already reached by 1999 and a rate of more than 95% in 2010 (vs. 2003 and 2015 required by Dutch law respectively).

3.2.2 Organisation

The Dutch EPR system for the collection and treatment of ELVs is organised in a collective manner with a single PRO, *Autorecycling Nederland* (ARN). ARN is the only PRO for end-of-life vehicles in the EU (Williams et al., 2020). It was founded in 1995 by four associations representing the automotive industry: *BOVAG*, *RAI Vereniging*, *Stiba* and *Vereniging FOCWA Schadeherstel*. The four associations together cover a very broad range of operations, including vehicle manufacturing, import, rental, repair and dismantling. ARN is organised as follows: Stichting Auto Recycling Holding is a foundation with four executives, one from each of the four associations. This foundation oversees a foundation on the recycling of ELVs, Stichting Auto Recycling, and a foundation on the recycling of their batteries, Stichting Autobatterij Recycling. Stichting Auto Recycling manages ARN Holding B.V., a private limited company directing the activities of ARN Auto Recycling B.V., which oversees the operations performed to fulfil the obligations arising from the ELV Management Decree (ARN, 2021b).[12]

Not all companies active in ELV treatment are affiliated to ARN. However, ARN's partners include more than 220 vehicle dismantling companies, 13 shredder firms and several other companies that contribute to the achievement of the national and European recycling and recovery goals. ARN's partners treat about 84% of ELVs in the Netherlands.

ARN is also responsible for the management of most end-of-life vehicle batteries. Both traction batteries, such as those powering electric vehicles, and starting, lighting, and ignition (SLI) batteries, fall under ARN's responsibility. As we already saw in the previous chapter, the former category is currently classified as *industrial* batteries in European and Dutch legislation, while the latter as *automotive* ones (see *Regeling beheer batterijen en accu's* (Ministerie van VROM, 2008)). However, traction batteries for electric and hybrid vehicles are expected to be allocated to a separate new category – with its own quantitative targets – in the new Battery Regulation, forthcoming in July 2023 (ARN, 2021c).

For traction batteries, some car manufacturers and importers have chosen to assume individual responsibility instead of participating in ARN's collective scheme (ARN, 2020b)**.** This renders traction batteries a particularly interesting case, where a number of producers organise EPR in a collective manner, while others do so on an individual basis.[13]

3.2.3 Advance recycling and management fees

End-of-life vehicle treatment is financed by an advance recycling fee paid at the time of purchase of a passenger car or light commercial vehicle. In 2021, the recycling fee amounts to € 30 per vehicle (including VAT), down from € 35 per vehicle in 2020 (ARN, 2021d).[14] The same fee applies to all vehicles, regardless of whether they are passenger cars or commercial vehicles, of their weight or other characteristics. In the past few years, the revenues from the advance recycling fee amounted to about € 16-17 million annually. In 2020, revenues were significantly lower (about € 12 million), due

[12] Another firm overseen by ARN Holding B.V. was responsible for the operation of the post-shredder technology plant until 2020. This firm was sold to a shredder company in March 2020, following ARN's plans to assume a more managerial role in the ELV treatment chain. The sales agreement ensures that other shredder companies are able to use the post-shredder technology plant (ARN, 2020e).
[13] Some of the top selling manufacturers of electric vehicles, including Nissan, Renault, Tesla and Volvo, assume individual responsibility for traction batteries (ARN, 2020f).
[14] The fee is planned to be further reduced to € 25 per vehicle in 2022. However, the base on which the fee is charged has been substantially increased since the implementation of the AVV in April 2021.

to a large decline in vehicle sales associated with the financial implications of the COVID-19 pandemic.

The main costs covered by the recycling fee are the net costs of ELV treatment and those of data collection and reporting. The revenues from the fee are used to finance the mandatory removal and recycling of hazardous vehicle components with a negative residual value, mainly airbags, batteries, fluids, tyres and LPG tanks. Furthermore, part of the revenues is used to reimburse vehicle dismantling firms for the costs of collecting and reporting data to ARN and shredder companies for monitoring costs. Another part of the revenues covers ARN's operating costs to direct and monitor the recycling chain and to meet the reporting obligations laid down in the Bba and the ELV Directive. Net treatment costs had also been covering the operating costs of the post-shredder technology plant until March 2020, when the PST plant was sold to a shredder company.

In addition to the net costs of ELV treatment, the recycling fee is used to support activities that improve ELV recyclability, especially in light of recycling challenges posed by the growing penetration of electric vehicles. ARN uses part of the revenues to support R&D on innovative recycling techniques. Revenues have further been used to develop protocols for the safe disassembly of electric vehicles and to provide relevant training courses to dismantling firms.

The fee has also been instrumental in enabling ARN to make large investments in recycling performance, whose short-term yields would not have been attractive to private investors. The best example of such an investment is the construction of the post-shredder treatment plant in 2010. Despite its crucial role in achieving the statutory recycling and recovery targets for ELVs, the plant remained a negative business case for many years after its construction. Another example of such investments is those in drainage installations which are lent by ARN to dismantling companies.

The fee used to be € 45 per vehicle, but it has been declining on an annual basis since 2017. Three developments are mainly responsible for this decline: (i) the average age of end-of-life vehicles has been steadily growing; (ii) recycling has become more efficient; and (iii) revenues from the sales of materials and used vehicle components have been increasing (ARN, 2021d). We will come back to the two last developments later in our analysis, but it is useful to draw on the evolution of the average age of ELVs at this point. The average age of an ELV has increased by 2.5 years in the past decade, from 16.3 to 18.8 years (ARN, 2020c). This 15% increase implies that revenues from advance recycling fees can be invested for longer time periods, resulting in higher returns for ARN.

The payment of the recycling fee was earlier this year declared generally binding through an AVV agreement (State Secretary for Infrastructure and Water Management, 2021)**.** Since late April 2021, the recycling fee is levied at the time a passenger car or light commercial vehicle is first registered in the Netherlands, regardless of whether it has been registered before in another country. The purpose of the AVV was to extend the payment of the advance recycling fee to second-hand vehicles imported by individuals or firms, which were estimated to account for an important share of vehicle registrations (about 33% in 2019). It was not possible to charge the recycling fee on these vehicles before the conclusion of the AVV, despite many of them being eventually scrapped in the Netherlands, and therefore entailing the same ELV collection and treatment costs as other vehicles. The AVV holds until the end of 2025, but it can thereafter be renewed. Each agreement is valid for a period of maximum five years.

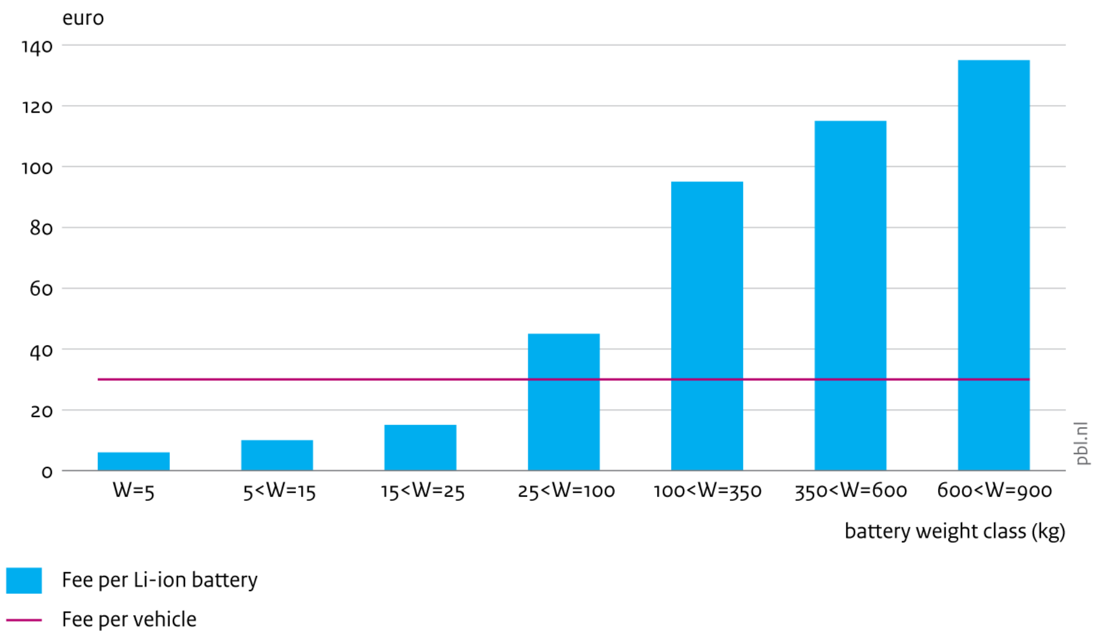
Distinct advance fees are also charged by ARN for the end-of-life treatment of vehicle batteries under its collective system. In contrast with the recycling fee, charging the battery advance management fee is on a voluntary basis: producers assuming individual responsibility for batteries are not obliged to do so. Advance management fees vary by battery chemistry and – for lithium-ion batteries – by battery weight. Three battery chemistries are mainly used in automotive applications: lead-acid, nickel-metal hydride, and lithium-ion. Lead-acid is the most common chemistry used for starting-lighting-ignition batteries, nickel-metal hydride has until recently been the standard chemistry for the traction batteries of hybrid vehicles, while lithium-ion is used in traction batteries of full electric and plug-in hybrid vehicles. The fee for the end-of-life management of lead-acid and nickel-metal hydride batteries is € 0.05 excluding VAT. For starting-lighting-ignition batteries installed in *new* vehicles, that fee is incorporated in the total amount (€ 30) of the recycling fee.

The fee for lithium-ion batteries is a stepwise increasing function of the battery weight, and in 2021 varies from € 6 to 135 per pack. The values that the management fee takes for different battery weight categories is plotted against the recycling fee in Figure 3.2. The figure provides a first impression of how the net treatment costs of a Li-ion battery compare to those of an ELV: the recycling fee for a car amounts to ⅔ of the fee paid for the battery of a typical plug-in hybrid electric vehicle (€ 45), and about 26% of that paid for the battery of a long-range electric vehicle model (€ 115).

In contrast to other battery chemistries used in vehicles, end-of-life lithium-ion batteries have a negative residual value (ARN, 2014). This explains to a large extent the relatively high management fees charged on them. Another driver of lithium-ion batteries' high management costs is that their transport should adhere to strict rules to prevent safety, health and environmental hazards.[15] Unstable or damaged batteries need to be transported in boxes specially certified for this purpose (ARN, 2017). Lithium-ion batteries that are not suitable for reuse are recycled in neighbouring countries with efficiencies of up to 70% (excluding casing). As there are no facilities for recycling lithium-ion traction batteries in the Netherlands, they are transported to Belgium, France or Germany to be recycled, something further increasing their treatment costs.

Figure 3.2

Advance management fee for Li-ion batteries by weight class vs. recycling fee for vehicles in the Netherlands



Source: ARN

Management fees for lithium-ion vehicle batteries have also been rapidly declining in the past few years. The 2021 fees are on average 38% lower than those four years ago, in 2017. This has been

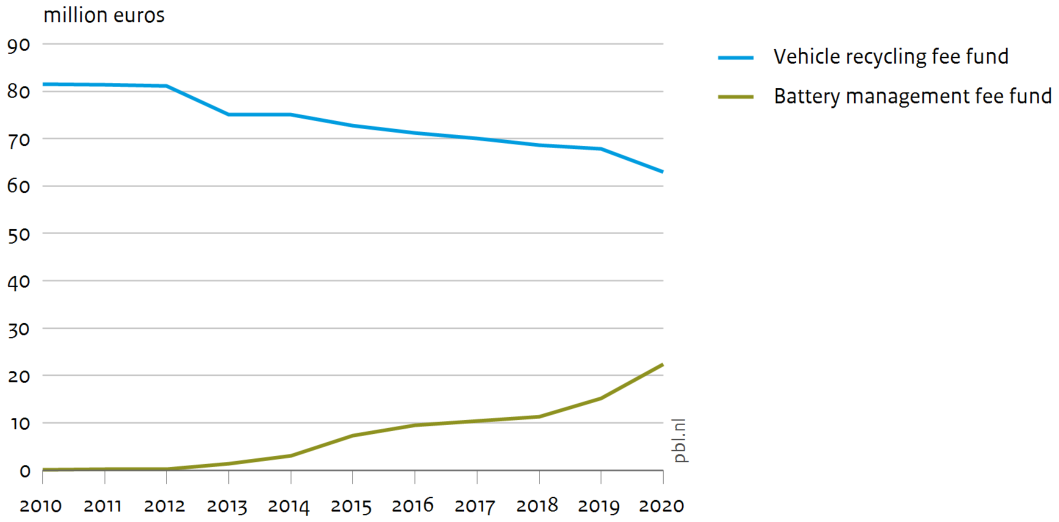
[15] These rules are laid down in the European Agreement concerning the International Carriage of Dangerous Goods by Road (ADR).

possible due to the growing number of batteries being reused in stationary applications (e.g. storage of solar energy), economies of scale in battery treatment and technological progress (ARN, 2020d). ARN is currently working on a prediction model aiming at setting battery management fees that more accurately reflect the future costs of battery collection and treatment. The output of the model is determined by various factors, including the number of exports of electric and hybrid vehicles, the expected lifetime of traction batteries, average battery pack weight, as well as forecasts of future collection and processing costs (ARN, 2021c).

The size of the battery management fee fund has seen a remarkable increase in the past few years, due to the growing penetration of hybrid and electric vehicles. By contrast, the size of the vehicle recycling fee fund has been steadily decreasing. The evolution of the size of the two funds is depicted in Figure 3.3. The value of the battery management fee fund has grown to become about 35% of the value of the vehicle recycling fee fund at the end of 2020, up from an almost negligible amount in 2010. In the same period, the number of Li-ion batteries to which the fee is levied annually exploded from a few dozens to more than 100,000.

Figure 3.3

ARN's vehicle recycling fee fund vs. battery management fee fund in the Netherlands



Source: ARN

3.2.4 Reporting

ARN publishes annually a sustainability report, which contains key information and data on physical and financial flows related to car recycling in the Netherlands. Among others, the report contains information about the number of vehicles processed by ARN and its partners, the reuse, recycling and recovery rates achieved, and the number of vehicles paying the recycling fee annually. It also presents the annual balance sheet and combined profit and loss account for ARN companies.

ARN reports on an annual basis data to the RAI Association enabling the monitoring of the compliance of the Dutch car recycling sector with the requirements of the Bba and the ELV Directive (ARN, 2017).

3.3 Effects of EPR for end-of-life vehicles and vehicle batteries

3.3.1 ELV treatment and recycling

In 2020, about 98.3% of an ELV's weight was reused, recycled or recovered by ARN and its partners. This corresponds to 88% reuse and recycling, and 10.3% recovery.[16] Taken together, these shares are well above the targets set by the ELV Directive and the Bba: 85% for reuse and recycling, and 95% including recovery. The 2020 combined share of reuse, recycling and recovery was very similar to that of 2019 (98.4%), but reuse and recycling (87.2%) have gained weight compared to recovery (11.2%). To provide a rough indication of the material use implications of these numbers, the total weight of material reused and recycled in 2019 was about 163 kilotons. An additional 21 kilotons were directed to other useful applications, mostly the production of energy (ARN, 2020e). The remaining 1.7% of ELV weight – which corresponds to a total of about 3 kilotons of waste – was landfilled.

In general, vehicle dismantling firms are responsible for the parts of an ELV being reused, while shredder firms for the lion's share of material recycling. Dismantling firms remove on average about 25% of the weight of an ELV, 90% of which concerns spare parts for potential reuse. However, many of these parts end up as input for recycling, as it is often more profitable for dismantling firms to sell vehicle parts, such as the motor block and the gearbox, for recycling than for reuse. Shredder firms retrieve almost all metal content of vehicles and send it for further processing in the metallurgy industry. On average, 60% of vehicle's weight is recycled or (to a very small extent) recovered by shredder firms. Post-shredder technology is responsible for the treatment of the remainder 14-15% of ELV weight: 40-45% of this is recycled, and the rest recovered or landfilled (ARN, 2018, 2017).

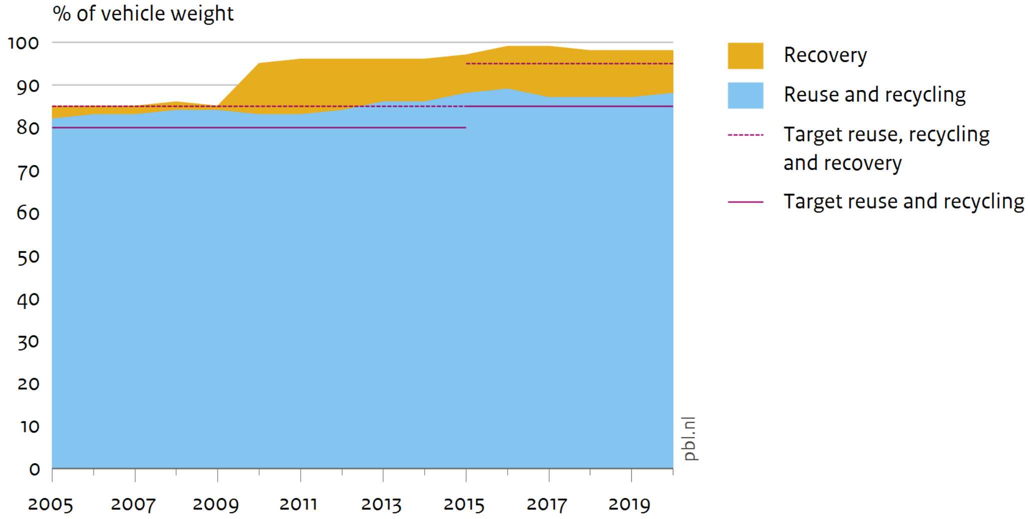
Looking back to the past 15 years, the reuse, recycling and recovery targets set by the ELV Directive and the Bba have consistently been achieved. Figure 3.4 presents the ELV reuse, recycling and recovery rates achieved in the Netherlands since 2005, and juxtaposes them against the corresponding Bba targets. As can be observed in the figure, two major changes have occurred during this period: a drastic increase of 10 percentage points in the recovery rate in 2010,[17] and a gradual increase in the recycling rate in the period 2011-2016. These changes allowed reaching the 2015 targets much earlier than required by legislation: two years earlier for the reuse and recycling target, and five years for the recovery one.

While recycling and recovery have seen noticeable growth in the examined period, reuse of vehicle parts has overall remained stable. As shown in Figure 3.5, while the (potential) reuse rate had slowly been increasing in the years leading to 2015, a shift from reuse to recycling from 2016 onwards has brought reuse rates back to their initial levels. A likely cause of this shift is the substantial increase of steel scrap prices in 2016.[18] The lower recovery rates in the period 2013-2017 were due to increased gate fees for shredder and PST waste, as incineration plants preferred incinerating household waste imported from other EU countries instead.

[16] The recycling performance of ARN is calculated based on data from three sources: (i) weight data submitted from dismantling and shredder companies; (ii) estimates of recycling performance from recent shredder trials/campaigns; and (iii) mass balance data from shredder companies in Belgium and Germany, for the Dutch ELVs exported for shredding to these two countries (ARN, 2017).
[17] The reason for the considerable increase in the energy recovery rate is that in 2010 it became feasible to incinerate shredder waste in waste-to-energy plants (ARN, 2011).
[18] Prices of futures for steel scrap increased by about 40% between March and December 2016 (London Metal Exchange, 2021).

Figure 3.4

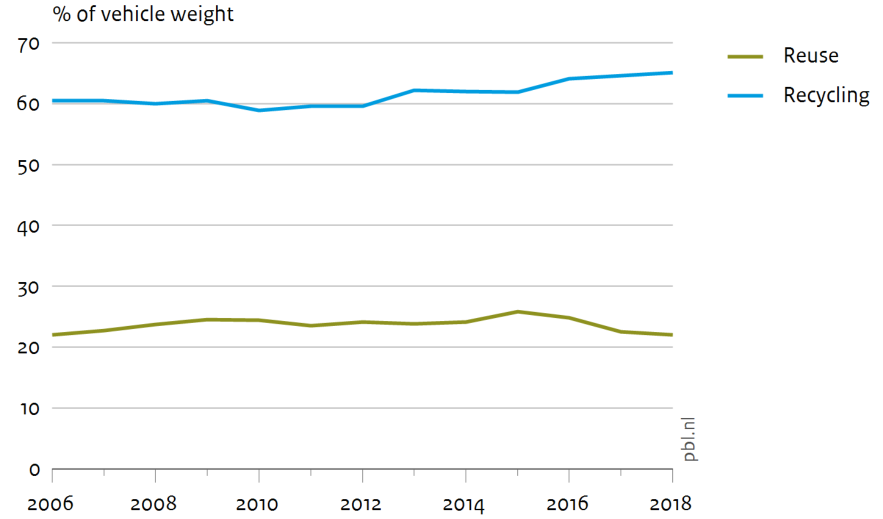
Reuse, recycling and recovery of end-of-life vehicles in the Netherlands



Source: ARN, RWS

Figure 3.5

Reuse and recycling of end-of-life vehicles in the Netherlands

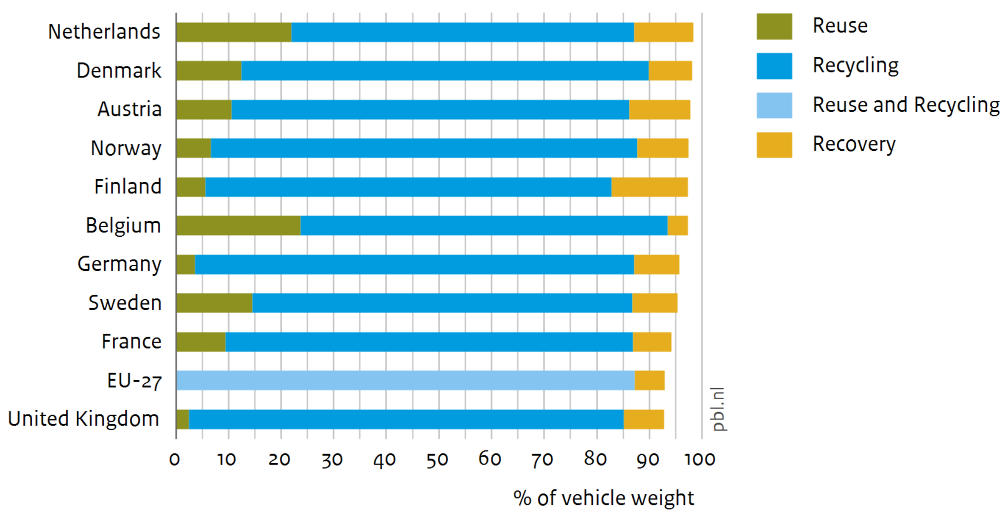


Source: Eurostat

The ELV reuse rate in the Netherlands is one of the highest in Europe, but exploring the potential for further increasing reuse of vehicle parts is important for the transition to a circular economy. Figure 3.6 compares the rates observed in the Netherlands with those of neighbouring countries and the EU-27 average in 2018. With 98.4% of an ELV recovered, the Netherlands has the highest ELV combined reuse, recycling and recovery rate among the examined countries. This is the result of a high rate of reuse – second highest after Belgium – and a high recovery one. Material recycling rates observed in the Netherlands are below those of neighbouring countries and the EU average.[19]

Figure 3.6

Reuse, recycling and recovery rates of ELVs, various EU countries, 2018



Source: Eurostat

The lower recycling rates observed in the Netherlands are to a large extent due to differences in reporting between countries. Methods to calculate or estimate the ELV share sent for recycling vary among EU countries, and so does the stringency of legislation as regards the scope of recycling. The scope used in the Netherlands is likely narrower than that used by some other EU countries. For example, the use of plastics from ELVs in blast furnaces as a reducing agent is considered as material recycling by German legislation, whereas as energy recovery by Dutch legislation (ARN 2014).

The post-shredder technology (PST) plant, whose operation started in 2011, has been instrumental in the increase of the recycling and recovery rates and the achievement of the 2015 targets. The PST plant is responsible for recycling and recovering about 15% of the weight of an ELV. However, part of this 15% is not additional to the reuse, recycling and recovery rate that would have been reached without the plant, as the treatment of some parts and materials has simply been shifted later in the chain. For example, glass from windscreens was until late 2014 manually removed and sent for recycling by dismantling companies, while now glass granules are mostly separated from other material streams at the PST plant.

3.3.2 Cost-effectiveness

The net costs of ELV treatment have been declining in the past few years. As we have already seen, the recycling fee charged at the time of purchasing a vehicle in the Netherlands has steadily been declining in the past five years. Two of the reasons put forward by ARN for the reduction of the fee are improvements in recycling efficiency and the growth in the revenues from the sales of recycled material. We will now discuss these channels in turn.

[19] This is the reason why the combined reuse and recycling rate in the Netherlands is slightly below the EU average, despite the high reuse rate. This can be seen by comparing the sum of the values underlying the green and dark blue bars with that of the light blue bar for the EU-27 in Figure 3.6.

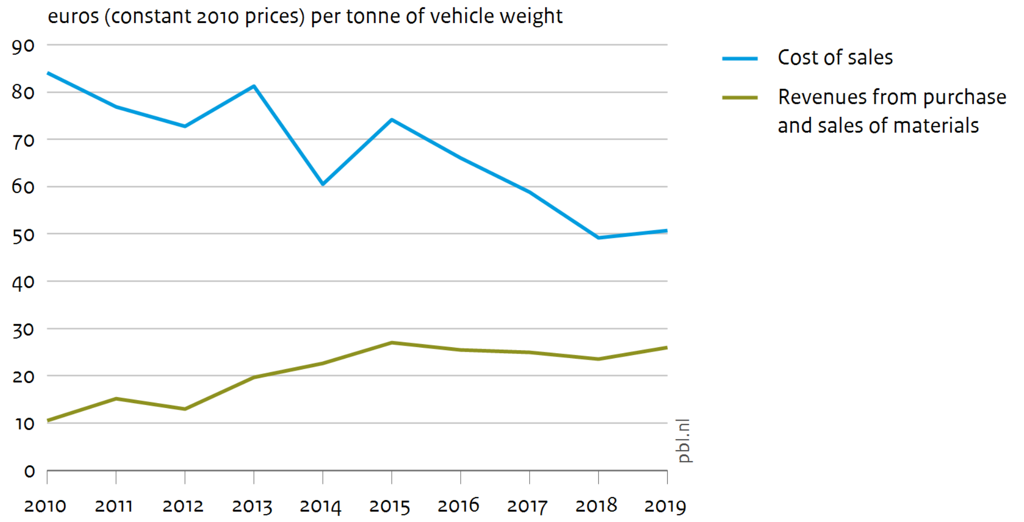
The first channel through which net costs have been decreasing is improvements in recycling efficiency. While recycling and recovery targets are reached, and recycling rates have been increasing, the treatment costs incurred by ARN have been falling. Figure 3.7 plots the evolution of the annual cost of sales incurred by ARN over time against the evolution of the revenues generated from the purchase and sales of materials. This information is obtained from ARN's annual financial reports and normalised in Euro per ton of processed ELV (i.e. ELV delivered to ARN-affiliated shredders). All values have been converted to 2010 prices to reflect real changes in costs and revenues using the price index from CBS (2021). The cost of sales captures the sum of the disposal fees paid by ARN to vehicle dismantling companies, the costs of collection and processing of hazardous materials (mostly batteries, tyres and fluids), and the production costs of the PST plant.

Between 2015 and 2019, the cost of sales per ton of ELV has declined by 32%. As shown in Figure 3.7, the cost of sales has declined from €84/ton in 2010 and €74/ton in 2015 to €51/ton in 2019. This is a sizeable reduction, revealing significant improvements in the efficiency of ELV treatment.

An important part of this cost reduction is to be attributed to the lower fees paid by ARN to vehicle dismantling companies. Dismantling fees have been reduced several times since the construction of the PST plant. These cuts were made on the grounds that various vehicle parts that had to be manually removed from the vehicle by dismantling firms were now kept on the vehicle and processed by shredders and the PST plant. A first reduction of dismantling fees was implemented in 2011 and a second one in 2014, also coupled with a termination of payments for the dismantling of glass and large plastic parts (ARN, 2014, 2012). At the same time, the basis for the reimbursements provided to dismantling companies shifted from material quantity to quality. Further reductions of payments to dismantling firms followed in later years: while they were receiving € 20 per ELV in 2017, their reimbursement was reduced to € 15 in 2018, and to € 10 in 2019 (ARN 2018).

Figure 3.7

Cost of sales and revenues from sales of materials from ELVs delivered to shredders



Source: ARN, CBS

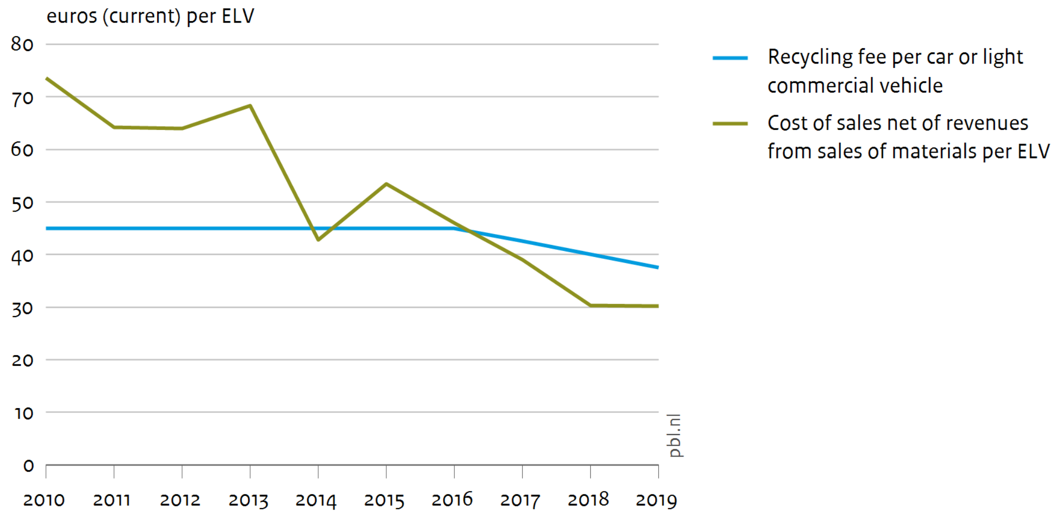
Efficiency improvements at the post-shredder treatment plant and economies of scale in its operation have been responsible for a further reduction of costs. Investments in equipment and machinery made in the past few years have improved the technical efficiency of the plant. Furthermore, the volume of material sent to the plant has considerably increased over time, enabling the development of economies of scale in recycling. As of 2016, the plant received shredder residues from almost all Dutch ELVs, regardless of whether they were dismantled by firms affiliated to ARN or unaffiliated with it.

The growing revenues from the sales of materials is the second channel through which net costs have been decreasing. Figure 3.7 reveals that the revenues from the purchase and sales of materials in 2018 are about 2.5 times those of 2010.[20] They have increased from € 10 to € 26 per ton in the examined period, with a peak of € 27 per ton in 2015. The PST plant has probably played a crucial role in this development too, by substantially increasing the quantity of materials that can be recycled or recovered, as well as by offering them at competitive prices.

The gradual reduction of the recycling fee coincides with the reduction of the net marginal costs of treatment below the initial fee levels. Figure 3.8 juxtaposes the evolution of the advance recycling fee for cars and light commercial vehicles against the per ELV difference between cost of sales and revenues from the purchase and sales of materials (expressed in current prices). We consider this difference a proxy for the net marginal costs of ELV treatment. While this difference used to comfortably exceed the recycling fee, the converging trends observed in the previous figure (Figure 3.7) have considerably reduced it over the past decade. From 2016 onwards – the year on which the recycling fee started declining – we observe a structural fall of the net marginal costs of treatment below the level of the recycling fee.

Figure 3.8

Recycling fee and difference between cost of sales and revenues from sales of materials from ELVs delivered to shredders



Source: ARN

Coming up with a precise estimate of the impact of EPR targets on recycling and the efficiency of ELV treatment remains difficult. Our graphical analysis and insights from ARN's reports are strongly suggestive of a positive impact of the 2015 targets on recycling rates and on the cost-effectiveness of ELV treatment. However, it is hard to estimate the evolution of these outcomes in the absence of the 2015 targets and in turn the impact of EPR for them. Furthermore, given the limited availability of data before 2002, when EPR legislation came into effect, it does not seem possible to estimate the effect of the introduction of EPR to this end-of-life product stream. As we have also seen in other parts of this document, one of the major contributions of EPR is that it comes together with a

[20] The data used to construct these series are referred to as "Other income" in ARN reports. This category mainly concerns revenues from the purchase and sales of materials (ARN, 2020e).

requirement for systematic collection and reporting of data on end-of-life product flows. However, this also implies that systematic efforts to collect such data have rarely taken place before the introduction of EPR, which poses an important challenge for rigorous empirical evaluations of its effects.

3.3.3 Addressing challenges posed by the uptake of electric vehicles

About 298 000 electric cars and light commercial vehicles had been registered in the Netherlands by the end of 2020 (Rijksdienst voor Ondernemend Nederland, 2021)**.**[21] The number of electric vehicles has rapidly increased in the past couple of years, with the fleet in 2020 being more than double that in 2018. These developments signal that expertise in the treatment of end-of-life electric vehicles and their components is already essential for the recycling chain.

The main recycling challenge posed by the increasing uptake of plug-in electric vehicles is related to the removal and further processing of lithium-ion traction batteries. This is an activity carried out by dismantling firms for both full electric vehicles and (plug-in) hybrids. Dismantling firms should first ensure the safe removal of the high-voltage battery from the vehicle, then test it to determine whether it can continue its life in stationary or other second-use applications, and finally send it for reuse or recycling by specialised companies.[22]

In the past, the EPR system developed and offered training programmes on the safe dismantling of electric and hybrid vehicles, thereby improving the recyclability and reusability of their components. These trainings were first provided in 2013 and focused on ensuring safety when working on electrical installations. They have been followed by at least 147 firms, implying a high representation of the vehicle dismantling sector (more than 60% of ARN's partners). As the number of hybrid and electric vehicles reaching their end-of-life is increasing, knowledge and skills on their dismantling will be key for ensuring high reuse and recycling rates and for preventing the release of hazardous substances to the environment.

The lifetime of traction batteries may be much shorter than that of the electric and hybrid vehicles they power. This is particularly the case for defect batteries, which might need to be replaced in the first years of operation of an electric vehicle. Following their removal, these batteries need to be safely transported to specialised companies for further processing.

The amount of lithium-ion batteries collected by ARN and reused in stationary applications has considerably increased. As shown in Figure 3.9, out of the approximately 48 tonnes of lithium-ion vehicle batteries collected in 2016, only 6% were reused (ARN, 2017). By contrast, with more than double that weight of batteries collected in 2019 (102 tons), a 75% reuse rate was achieved. However, the picture has been completely inversed in 2020, with only 19% of the 121 tonnes of batteries collected being directed to reuse. A main reason for this considerable reduction of the percentage of reuse is that an important share of batteries was removed because they were (likely to be) defect. These batteries were not considered safe for reuse. Battery reusability depends on at least two main factors: the state of batteries removed from electric and hybrid vehicles, and demand for used batteries by stationary applications. EPR has very little influence on the quality of removed batteries, but can help promote reuse through strengthening the relationship between the ELV recycling chain and companies providing batteries a second life in stationary applications.

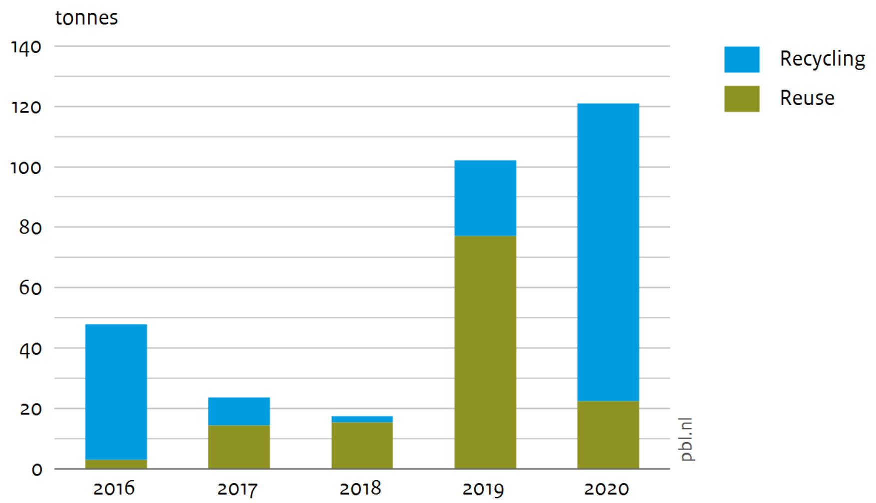
The reuse of electric vehicle traction batteries in other applications entails a transfer of responsibility from car manufacturers to companies reusing them. To be allowed to receive these batteries and reuse them in other applications, companies should have obtained an end-of-waste

[21] This number includes plug-in hybrid electric vehicles, but not standard hybrid ones.
[22] Before being sent for recycling to other EU countries, all end-of-life batteries are collected by a company acting as the sorting centre for batteries in the Netherlands (ARN, 2021c).

legal judgement (rechtsoordeel) by the Dutch government. ARN collaborates with three such companies in the Netherlands, which mostly use the batteries in renewable energy storage applications (ARN, 2020b).

Figure 3.9

Weight of collected lithium-ion vehicle batteries by type of treatment, the Netherlands



Source: ARN

3.4 Conclusions and looking ahead

The Dutch EPR scheme on ELVs has been effective in achieving the quantitative targets set by the EU and Dutch legislation and in reducing net treatment costs. The reuse, recycling and recovery targets set by the ELV Directive and the Bba were achieved years before their entry into force. ELV net treatment costs have been decreasing over time, both due to an increase in recycling efficiency, and to higher yields from the sales of recycled materials. A main driver of both these developments has been the post-shredder treatment plant which was constructed by the PRO and came in operation in 2011. The decrease in net treatment costs has enabled a reduction of the advance recycling fee by a third in the period 2016-2021.

The scheme has also been contributing to the development of secondary material markets. This is particularly the case for emerging material markets, such as those of secondary plastics. For example, the Dutch EPR scheme has been supplying car manufacturers with recycled plastic that is used in new vehicles. The impact of the Dutch EPR scheme on eco-design is generally limited, but efforts on this front are likely more effectively pursued and evaluated at the EU level. The Dutch car market is relatively small, and few vehicles are produced or assembled in the country, so it is hard for Dutch EPR for ELVs alone to influence producers' manufacturing choices.

The scheme has been getting prepared for the challenges posed by the growing uptake of electric vehicles. It has done so by offering trainings for the safe dismantling of hybrid and electric vehicles and by establishing partnerships for the reuse of traction lithium-ion batteries. Given the importance of the transition to electromobility in achieving climate and air quality objectives in the Netherlands and the EU, it is important that the ELV collection and treatment system is well-prepared for it.

Looking ahead, it is important from a circular economy perspective to investigate options to increase the reuse of ELV components. Despite the high reuse rates of vehicle parts achieved in the Netherlands, these rates have lately been declining. Trying to bring some components back to dismantling firms, so that reuse is increased, while also maintaining the desired level of efficiency of the post-shredder treatment plant may look like a difficult balance to strike, but it would be worth further exploration.

Efforts to increase the supply of used parts should be coupled with incentives to boost their demand. For instance, car repair shops may be required to offer a quote for original second-hand parts for vehicles whose warranty has expired, a policy already implemented in France with encouraging results (ARN, 2021c). Consumer awareness campaigns financed by producers and targeted at the financial and environmental benefits of opting for second-hand parts, such as the campaign launched by the Dutch association of certified dismantling companies (Stiba) in 2020, could also increase demand for them. Extending the length of warranty offered on used parts would also help in convincing consumers to choose them over new ones.

Sub-targets for the reuse and recycling of materials with low residual value, such as glass and some types of plastics, would help in shifting flows towards these applications. However, it is important that the design of such targets and the policies accompanying them disincentivise the downcycling of these materials.

Incentives are also needed for the high-value recycling of plastics, glass and other materials in ELVs. New cars contain on average 40-50 kilograms of 39 different types of plastic (ARN, 2021c). This implies an annual inflow of roughly 20 kilotons of plastics in the Dutch economy and of 750 kilotons of plastics in the EU. This inflow is only expected to grow in coming years, as plastics are increasingly used in the manufacturing of new vehicles. The diversity of plastics in cars poses important challenges for recycling: in practice, less than half of the plastics in ELVs is recycled, and most of this part is downcycled. To stimulate high-value recycling, EPR policy could be accompanied with recycled content requirements for plastics per new vehicle or vehicle part. That would increase demand for recycled plastics and incentivise the ELV chain to further invest in the separation and recycling of different types of plastics. This effort should be facilitated by detailed information on the location of different types of plastics in each vehicle, provided by manufacturers in the context of their EPR obligations (ARN, 2021c). Glass is also largely downcycled. Incentives promoting the removal of glass by dismantling firms are likely to increase its potential for high-value recycling. It might also be useful to reconsider the list of applications falling under ELV recycling: for example, backfilling is not classified as recycling by the Waste Framework Directive (Williams et al., 2020).

Given the environmental risks and circular economy opportunities implied by other types of end-of-life motor vehicles, it seems useful and relevant to introduce EPR to them too. A discussion about the expansion of the ELV Directive to heavy-duty vehicles, motorcycles and scooters took place in the context of the Directive's recent evaluation (Williams et al., 2020). End-of-life vehicles in these categories are currently covered by the Waste Framework Directive. While the industry is not generally opposed to the introduction of EPR to additional vehicle categories, it would prefer a separate piece of legislation for this purpose, tailored to the characteristics of the product and the size of the firms producing it. Introducing EPR to motorcycles and scooters is more straightforward than introducing it to heavy-duty vehicles, as most of the latter are exported and continue their life outside Europe. The Dutch industry has introduced voluntary EPR for scooters since 2013 and has established a PRO, Scooter Recycling Nederland, for this purpose. Scooter recycling is financed through an advance recycling fee, currently amounting to € 5. For more details on the Dutch scooter recycling scheme, see Box 3.1.

At the EU level, the greatest concern with the implementation of the ELV Directive is the large number of missing vehicles. There were close to 3.8 million vehicles deregistered from the EU fleet in 2017, without them being reported as ELVs or as exports to third countries. This amounts to slightly less than 60% of the ELVs treated in the EU in that year (Williams et al., 2020). While it is unclear how many of these vehicles were illegally dismantled (and not e.g. exported without this export being reported), this is a sizeable amount potentially posing considerable environmental risks. The vehicle (de-)registration system of the Netherlands is among the most advanced ones, so missing vehicles is much less of a problem there than in other countries.

Co-ordinated action at the EU level is also needed to ensure the environmentally responsible treatment of used vehicles exported abroad. Neither obligations arising from EU and national legislation on EPR nor advance fees paid for ELV treatment are currently carried over to third countries. EPR for ELVs only applies to the share of cars scrapped in the Netherlands or other European Economic Area countries. Vehicles reaching an end-of-life stage in countries where no EPR or related environmental policies are implemented are unlikely to be treated in an environmentally sound manner. To prevent such environmental damages, the EU can work together with governments and vehicle importers in third countries to help them set up their own EPR systems. PROs and other participants in the EU ELV recycling chain can actively support this effort, by e.g. sharing best practices and know-how with local stakeholders, and by providing training to local workers.

Box 3.1: Scooter and moped recycling

A voluntary scheme for the recycling of scooters and mopeds exists since 2013. The scheme is managed by Scooter Recycling Nederland (SRN), a PRO established by two of the founding members of ARN, the RAI Association and BOVAG (ARN, 2014).

The process for scooter and moped recycling resembles the one followed for car recycling. This recycling scheme is also financed through an advance recycling fee, which currently amounts to € 5 per scooter or moped. The fee was initially set at € 10 per vehicle, but was later reduced to its current amount (ARN, 2013).

4 Medicines

4.1 Introduction

In this case study, we take a look at a sector where EPR has not been implemented. We explore what the introduction of EPR to medicines and their packaging can achieve in terms of promoting a more circular economy. We take the instruments that have been successful in the other case studies and consider what their contribution to the waste management of medicines could be. Even if reuse or recycling of medicines is not possible due to hygiene concerns, EPR may be able to help with the collection of unused medicines.

Since medicines are consumable and perishable, unlike the products from the earlier case studies, EPR for medicines might need to be designed with different intermediate objectives in mind than those employed for other product groups. Still we can take the lessons that we have learned from other EPR schemes on what effects can be expected from different EPR instruments to help us suggest what could be done for the waste management of medicines. Improperly disposed of unused medicines and their packaging can be harmful to the environment and public health medicines. This is also the case for medicines that have been consumed and enter the sewage system without having been broken down by the human body.

4.2 Background

Many pharmaceutical companies producing medicines have moved out of the Netherlands over the past few years[23]. Although the country possessed a substantial number of producers in the past, many of them have moved out of Europe, mainly to India and China. Some companies moved their production abroad themselves and others have been taken over by foreign companies that were interested in the intellectual property, but closed Dutch production facilities after the takeover.

Production of medicines also takes place in pharmacies and hospitals but that is only a small fraction of total medicines production. Pharmacies are mainly distributors. Still, when designing EPR for medicines, these smaller producers should also be considered. They also have a role in securing medicine safety and should consider the environmental consequences of their production processes. A good example of the latter is the purchase by the Academic Medical Center (AMC) in Amsterdam of an advanced wastewater treatment plant to treat wastewater of the hospital, removing medicine residues and harmful bacteria (Dutch Water Sector, 2018).

4.2.1 Current waste management

The use of medicines results in solid waste and water pollution (OECD, 2019). Solid waste stems from left-over medicines that need to be disposed of when patients receive more than what they needed. It also includes its packaging because medicine traces in packaging can also be harmful to the environment. In the remainder of this chapter we abbreviate 'medicines and their packaging' to 'medicines'. In the Netherlands, consumers can dispose of medicines by returning them to a pharmacy. Pharmacies collect the leftover medicines and deliver them to the municipal waste

[23] As of writing, one producer has received funds from the Dutch government to make a fresh start, after it had been taken over by an Indian company that decided to close the factory and to move production to India (FD, 2020).

treatment services that then usually incinerate it. Recycling of left-over medicines is generally not allowed due to hygiene and health concerns. Most municipalities provide this service for free to pharmacies, as households are also not charged extra when they dispose of left-over medicines themselves at the municipal waste service. Municipal waste management is financed by municipal taxes. Some municipalities still charge pharmacies for their collected waste medicines however (KNMP, 2020).

Pharmacies in the Netherlands have introduced a system called PharmaSwap. This allows them to buy and sell unused and unopened medicines between each other, reducing waste. This can be useful if a customer passes away and the pharmacy had built up some stock. It does not need to wait for another patient that requires that same medicines or until they expire. The pharmacy can sell these leftovers to other pharmacies, usually at reduced prices. Because these medicines have only been stored by other pharmacies, the buyer can be assured that they have been kept under the correct conditions and that they will still meet the inspections' criteria (PharmaSwap, 2020).

Improper disposal of medicines but also medicine residue from the human body in the form of undigested traces may cause water pollution. Improper disposal such as flushing left-over medicines through the toilet or down the sink, is a behavioural problem. The other source of medicines in our sewage is the human body. Medicines in the form of cremes can also be washed away when showering. Proper waste management limits improper disposal of medicines, but it cannot effectively prevent medicines from ending up into the environment through the human body or through washing away. Eco-design could help in this regard, although medicines that break down naturally may not be as effective if it is broken down too quickly, presenting a possible trade-off.

Environmental effects of pollution through increased medicine concentration is substantial. In the Netherlands 2272 tonnes of medicines are prescribed annually. About 509 tonnes of medicine traces end up in surface water. About 190 tonnes of medicines cannot be broken down by water treatment plants properly. The costs of cleaning up this remaining fraction are about 10 euro per capita per year (Vereniging Innovatieve Geneesmiddelen, 2021). Under EPR, producers of medicines could contribute to these costs.

4.2.2 Waste management of medicines abroad

Various waste management schemes are in place to manage unused medicines in Europe and North America (Alev et al. 2021). A first approach is to reduce the amount of medicine waste by discouraging excessive production and consumption though introducing costs to producers. The Canadian province of British Columbia has an EPR scheme for medicines in place in which producers fund the collection system with fees based on the amount prescribed. Producers in Portugal are also subject to a collection fee based on the number of products they put on the market

Some EU countries and some US counties have introduced a policy instrument that makes producers contribute to the take-back programmes for unused medicines (Alev et al., 2021). This type of fee is very different from what we have seen from EPR schemes that have been put in place in other sectors. There is no take-back requirement in the form of a collection target in place, nor an advance disposal fee nor a deposit-refund system. There are no guidelines from the European Commission for the implementation of such schemes and information on implementation and efficiency of collection is missing (Amaral and Fop, 2013). This waste management policy for medicines can only be focused only on the unused fraction to avoid throwing away of unused medicines. Given this limited scope of the policy, no effects on product design should be expected. However, it does appear to be an effective policy to reduce incentives for overprescription (Atasu, 2019).

In several states and counties in the United States, some form of EPR for medicines exists. In Alameda County in California and King County in Washington, pharmaceutical manufacturers are responsible for funding disposal programs for medicines (Craver, 2016). All manufacturers that make their products available in these two counties are required to contribute to their drug disposal program. With the raised funds, separate collection and disposal of unused medicines is financed. Manufacturers estimate the costs of complying to the Alameda county scheme to be $1.2 million per year on a revenue of $950 million per year from that same county. The introduction of this policy was not without controversy: producers filed a lawsuit as they did not feel responsible for the costs of the initiative. However, they were not successful in holding back the EPR policy. Other counties and states have introduced voluntary or mandatory programs as well (OECD, 2021).

Comparison of data between countries is difficult due to differences in the way that unused medicines are measured. What is considered household medicines waste varies by country (or state). In some countries, the residual medicines and all (inner and outer) packaging are considered, while in others only inner packaging (unless it is empty) is counted. For example, Amaral and Fop (2013) claim that while 53 g of medicines waste was collected per capita in Belgium and 78g in Italy in a given year, this does not necessarily mean that Italians disposed of more left-over medicines. The difference is instead largely driven by the standard of measurement in each country.

4.3 Waste medicines and market failures

In a survey by Dutch media, 84% of a total of 31,511 respondents indicated that they sometimes have leftover medicines (Radar, 2020). Of those with leftover medicines, about half indicated that they still have some medicines at home. About 93% of respondents indicated that leftover medicines should be returned to pharmacies; while 74% of people with leftover medicines do actually return them to their pharmacy. The main reasons not to return medicines was that it would seem wasteful as someone (in the household) might want to use the medicines at a later moment (28%) and that it is simply too much effort to return medicines to the pharmacy (27%).

A specific characteristic of the 'market' for medicines is that doctors decide for their patients on what and how much to consume. Consumers of medicines do not choose themselves how much medicines they purchase and which medicines they need. It is doctors that determine demand for medicines as they choose what medicines to prescribe and the quantity that a patient should take. Therefore, there could be a role for EPR for medicines to aim at producers' and doctors' behaviour, as without policy intervention neither of them has incentives to opt for more environmentally friendly medicines and improved end-of-life management. At the same time, providing incentives and information on environmentally friendly disposal methods to consumers is essential for the effectiveness of waste management policies. We continue this section with three main market failures that are relevant to the medicines sector to see what potential problems there are for EPR to resolve.

4.3.1 Unpriced environmental externalities

We can distinguish between unpriced environmental externalities in three different stages. The production stage mainly takes place abroad, the consumption and disposal stages take place in the Netherlands. EPR policy has the potential to address all these three stages when implemented according to theory, whereas current waste policy in the EU and US mainly has an effect on the consumption stage by making overprescription less attractive. This may partially address both social (drug abuse) and environmental (water pollution) problems (Atasu, 2019).

The production stage mainly takes place in China and India where environmental regulations are less strict than in the EU. Unregulated water, ground and air pollution are some of the negative external effects of production of medicines on the environment and the health of the local population. One of the greatest concerns is the dumping of antibiotics by producers through wastewater in local water streams, creating antibiotic-resistant bacteria, as reported for example by Lübbert et al. (2017).

The consumption stage is not associated with direct environmental externalities but there are indirect effects. Traces of medicines may be contained in the metabolic waste excreted by patients. They may place a large burden on the environment. Even if alternatives are available that are less damaging to the environment, doctors currently have no incentive to prescribe more environmentally friendly alternatives. They choose the medicines they think work best to help the patient. From a social perspective, the environmental aspects should also be considered when choosing a particular treatment.

In the waste stage, environmental damage may arise from burning pharmaceuticals that have been collected as waste or from flushing medicines through a sink or toilet. There exists an incineration fee in the Netherlands for burning waste but this fee is not differentiated according to the type of waste. Medicines that are littered or flushed entail a number of environmental risks. Water treatment plants cannot usually filter out all medicine traces and some small amounts may end up in drinking water, concentrations in the Netherlands are still at safe levels though (RIVM, 2017). In the Netherlands, 140 tonnes of medicine traces flow into the sewers every year (about 8 grams per capita per year). Medicines that are disposed of in the environment may also end up in crops or animals and thus can enter the food chain, putting human health at risk (Zenker et al., 2014).

Medicines in household waste that are incinerated create hardly any environmental costs. Cook et al. (2012) argue that emissions in the waste stage may be lowest when leftover medicines are disposed of in household waste by consumers themselves. Leftover medicines that are collected at pharmacies are usually incinerated afterwards. Medicines that were disposed of without any sorting, are also usually burnt. The study therefore argues that the extra effort that people need to take, to separately dispose of their leftover medicines, creates extra emissions (extra trips and transportation) compared to when they would have just disposed of them with their other household waste. The argument also holds for the extra logistics that are introduced for separate collection. They do note that disposal via the toilet leads to much higher emissions from water treatment than both other options for disposal. The same probably holds for littering medicines in the environment. If the use of separate collection points lowers the amount of medicines flushed (or littered), it can still lower emissions overall.

This creates a communication issue towards consumers. Disposing of medicines with other household waste does not generate a larger burden on the environment than separate collection does, but flushing medicines through the sink or toilet *does* put a heavy burden on the environment. Ideally, all consumers would return their leftover medicines to the pharmacy but the actual challenge is to prevent people from flushing their medicines. Even disposing of them with other household waste may be a good alternative. When designing EPR, this aspect needs to be thought out well, it seems that one, not two goals should be pursued at the same time to keep communication towards consumers clear.

4.3.2 Missing incentives

Besides doctors not currently having incentives to consider the environmental impact of their choices, consumers also lack financial incentives to properly discard their unused medicines. Consumers that return their leftover medicines do that because of their intrinsic motivation. EPR for medicines could help to increase the number of drop off points to supermarkets for example, where other types of waste are also collected separately.

4.3.3 Shortage of innovation

Biodegradable medicines could be a solution to the environmental burden that conventional medicines pose. Currently, producers do not have incentives to innovate in this direction. Innovation is aimed at finding new medicines to cure new diseases or to lower side effects. This is of course the main purpose of medicines. Lowering the negative (side-)effects for the environment does not directly benefit the producers but it does benefit society. Innovation in environmentally friendly medicines generates positive externalities for society and therefore deserve to be subsidized, and EPR scheme could help to fund such subsidies.

4.3.4 Incomplete information

It is not consumers who choose which medicines to take, it is the doctors who do so. Doctors do not have full information about the environmental impact of the medicines that they prescribe. They prescribe the medicines that they think will help the patient best. This is their main task, but in doing so they do not necessarily consider other factors such as the environment and the health of others. This means that they prescribe medicines that are less environmentally friendly than optimal from a social welfare perspective. In other words, society would be better off if doctors were also provided information about the environmental impact of medicines and thus could take this information into account in their prescription decisions.

4.4 EPR policy instruments

When it comes to EPR policy instruments for medicines, we focus on take-back requirements and advance disposal fees (ADF) to tackle the market failures mentioned above[24]. These are the two most used instruments in current EPR schemes. The deposit-refund schemes make less sense for medicines as that would charge a deposit on a good that is meant to be consumed, creating an unwanted incentive to use less medicines so that a larger refund could be gained. An option that could work is a deposit-refund scheme for only the packaging, the costs of implementing that are probably high and the environmental benefits probably low[25]. While we have not explored other policy instruments, they may still prove useful and efficient when implementing EPR for medicines.

4.4.1 Take-back requirements

Setting take-back requirement a target, works differently for medicines than for goods where this instrument has been implemented before. That is the case because medicines are consumable. It is not supposed to reach the waste status in the first place. The only medicines that do so are, its packaging with medicines traces (e.g. squeezed tubes or empty pill strips), leftovers and expired ones [26]. One cannot know beforehand how much of the sold medicines remains as leftover. Governments cannot therefore set a take-back target based on the amount of medicines put on the market. They should rather focus on the environmental problem more directly, which is water and soil pollution.

The target of EPR policy could be to lower the amount of medicines traces in soil, surface, and drinking water. A target for the concentration of medicines traces for those medicines that are most harmful in surface water gives direct incentives for producers to lower this value. The target should be dynamic and move according to the latest insights on environmental effects over time.

[24] Unpriced environmental externalities, missing incentives, shortage of innovation and incomplete information.
[25] If the packaging is discarded in household waste, then separate collection does not provide further environmental benefits. If packaging is also frequently flushed, a deposit-refund scheme could help to prevent that.
[26] This reasoning does not hold for its packaging which already falls under the packaging directive.

Introducing static rather than dynamic targets, means that producers have no incentives to put in more effort once the target has been reached. The maximum concentration that the target should move towards does not necessarily have to be zero. Once the concentration is so low that the additional environmental and health risks prevented do not exceed the incremental costs of further reducing concentration levels, an optimum is reached and the target can become fixed. A periodic evaluation is still needed as technological progress may make further reduction possible with lower costs, or previously unknown environmental and health risks may be identified.

The traditional take-back target incentivises the producer to take action to change consumer behaviour. As we have seen in Chapter 3 of CPB and PBL (2021), producers will use instruments themselves to induce consumers to change their behaviour so that they are able to comply with the collection target. The producers may introduce their own measures such as an information campaign for consumers on how and where to dispose of their waste or use other instruments. This type of EPR does not generally prescribe to the producer how they should reach their target. Producers are free to come up with the best way to comply with their responsibility.

For producers of medicines, such a target may imply that they will work together with pharmacies, the distributors of medicines. In the Netherlands, pharmacies already collect leftover medicines and their packaging. Currently, municipalities collect these from the pharmacies and producers are not involved in the waste management. When producers become responsible for the waste management phase of medicines, they will have to finance such schemes instead of municipalities, introducing a direct financial incentive for them to keep collection costs down and organise their waste collection efficiently. The producer responsibility also ensures a sustainable source of funding.

With these funds, information campaigns aimed towards consumers can help to raise awareness about the proper disposal method for medicines waste. Although we have seen that a large majority of people seem to know how to dispose of leftover medicines correctly, under EPR it would be the duty of producers to keep that value up into the future. Awareness can also be raised under doctors that prescribe medicines.

This financial responsibility may introduce incentives for eco-design. As producers have more waste management costs in the form of incineration fees for medicines that are not treatable in any way and packaging that is not recyclable, they have an incentive to reduce the volume of medicines that needs to be burnt. This can be achieved by working together with doctors and preventing them to prescribe too large quantities but also by innovating more towards medicines that are biodegradable in such a way that it does not pose an environmental threat for example. For packaging, separate collection may make recycling of the used materials possible after washing them. Part of producers' R&D budget would be going to researching more environmentally friendly medicines to address the social cost of the current medical waste.

Another way to lower medicines concentrations in surface water is to set targets for concentrations in wastewater from hospitals and pharmacies. As we have seen with the Amsterdam Medical Centre, it is possible to filter out a lot of residue. A cost effectiveness analysis could determine if wider adoption of such wastewater treatment systems is feasible.

4.4.2 Advance disposal fees and deposit-refund systems

An advance disposal fee (ADF) can be used for medicines packaging to finance its waste management. An ADF in combination with a take-back requirement or a deposit-refund system for packaging could make retrieval and subsequent recycling of the packaging materials possible. It is likely that most non-used medicines are collected simultaneously. Maybe, complementary instruments are needed however to prevent medical waste flushing.

An EPR scheme that uses a differentiated fee per type of packaging also gives indirect incentives for producers to produce their packaging in more environmentally friendly ways (eco-design). This differentiation could happen on the basis of the materials used, recyclability, or the amount of recycled materials used in the production. By introducing such a differentiated fee, packaging that is less easy to recycle receives a higher fee to cover its disposal costs, while packaging that is easier to recycle and generates more recycled materials that can serve as an input for new production, receives a lower fee. This stimulates producers to better think about their packaging design. Inner packaging may be impossible to recycle regardless of the design of the packaging because of the chance of leftover medicines (or traces), especially in the case of creams and ointments. However, for environmental reasons it is crucial that they are collected simultaneously.

Deposit-refund systems do not seem to be able to contribute much to the waste management of medicines themselves as most medicines are consumed and the deposit could never be refunded. This would essentially disincentivize consumers to take their prescribed medicines. For packaging a deposit-refund system could work but the instrument is usually costly to introduce. It is therefore mainly applied to products that are often found to be littered which is not the case for medicines packaging.

4.4.3 Information campaigns

It seems that with the right information, many consumers are willing to dispose of leftover medicines in an environmentally friendly way. Information campaigns aimed at consumers, funded by an EPR scheme, could have significant effects on the amount of medicines flushed through the sink or toilet. Likewise, information campaigns aimed to inform doctors on the environmental effects of the medicines that they prescribe, may lead to lower concentrations of problematic medicines in the environment. If combined with an increased number of collection points, such as supermarkets, collection rates may go up significantly as we have also seen with battery collection. These campaigns could be a very cost-effective way to increase social welfare by lowering medicine concentrations in the environment.

4.5 Conclusions

Introducing EPR to medicines could help improving their waste management and possibly create incentives for eco-design. While the traditional instruments, such as take-back requirements, advance disposal fees and deposit-refund systems cannot be implemented in the way they usually are, with some adaptations there may be possibilities like we have seen abroad, or they can be aimed at packaging.

Through a formal EPR, producers can be made responsible for some of the practices we are already seeing today. Giving producers responsibility for collection and incineration and financing thereof as well as of information campaigns, makes the system more reliable towards the future and ensures financing of the system. It also adds transparency about who is responsible for what costs. Best practices from other countries can be considered when designing EPR for medicines, ideally harmonized EU policy levels the playing field and helps producers in setting up the logistics in a consistent manner.

Medicines are meant to be consumed and not to reach the waste stage, only leftover or expired medicines need to be handled. This means that the traditional instruments that are aimed at getting the product back may introduce adverse incentives by persuading the consumer to return the product instead of consuming it. However, focusing on crucial packaging elements (pill strips, tubes) may solve this problem. Rather than using a collection target, an EPR for medicines could target other outcomes such as environmental pollution in the form of medicines traces in soil and water. An advance disposal fee for medicines alone would not work because the leftover fraction is unknown at the time of purchase, but the instrument could work for packaging.

Effects on eco-design are uncertain. In theory, targets on medicines traces could also incentivise producers to innovate towards medicines that are biodegradable, but it is unknown if those types of medicines can reach the same effectiveness as their existing counterparts. If medicines would degrade too fast in the human body, then that could affect their effectiveness.

Hospitals and pharmacies that produce medicines themselves could install water treatment facilities. To prevent medicines (and traces) from entering our surface water and to prevent growing resistance to antibiotics, existing techniques can be employed today. Cost effectiveness analyses of these water treatment facilities are therefore a recommended direction for more research.

It seems that consumers are generally very willing to dispose of their leftover medicines at pharmacies or other collection points. Information campaigns aimed at informing consumers of these collection points could be very effective in increasing collection rates. Increasing the number of collection points might also be very effective, considering the relatively high collection rates of batteries which can be discarded at almost all stores.

5 Policy implications

This chapter presents the main policy implications from the joint analysis of the three case studies. We organise them around incentives for consumers (for proper disposal), incentives for producers (for collection and treatment of all end-of-life products), and EPR for new product groups.

To incentivise consumers to properly sort and return their end-of-life products, easy access to separate collection points is key. Achieving high rates of separate collection is important for the environmental effectiveness of the policy. High rates ensure that only a limited number of end-of-life products are littered, illegally dumped or mixed with other household waste. The wide coverage of collection points has been an important factor for the increased collection rates in the case of batteries, and is likely to lead to similar outcomes also for other product groups, such as leftover medicines.

Public awareness campaigns financed by producers in the context of their EPR obligations may be an effective way to promote proper disposal, reuse and recycling. Campaigns informing consumers about how to properly dispose of their waste and about the environmental problems associated with improper disposal can help increase collection rates, ensure more homogeneous and higher-quality streams of end-of-life products, and eventually prevent environmental damages. For batteries, campaigns and similar activities have been successful at raising the quality of collected products. Campaigns directed to the environmental and financial benefits of buying used products, as already done to promote the use of parts from end-of-life vehicles, may also contribute to increasing reuse.

For take-back requirements to give the right incentives, targets should be defined in a way that producers can direct efforts towards a better outcome. From the batteries case study, it became apparent that collection targets as a share of the quantity of a product put on the market, are not so insightful in markets that are quickly growing or shrinking. In a growing market with a relatively long product lifetime, collection rates will always be low because the amount of product available for collection lags behind what is being put on the market. Basing a target on what is available for collection — using average product lifetime — circumvents this issue and gives more insightful outcomes and targets.

A key issue for the environmental effectiveness of EPR policy is how producers carry (financial) responsibility for end-of-life products they do not collect. This concerns both products that are improperly disposed of in the EU, but also used products exported to third countries where EPR is not implemented. Regarding the former, producers could be required to reimburse public authorities for the costs incurred to collect and treat products in mixed waste and products which have been littered or illegally dumped. That would be consistent with the EPR and Polluter Pays principles. With regard to used products that are exported to third countries where no EPR applies, such as vehicles and electronic equipment, the EU can join forces with producers to help these countries in setting up EPR systems. Such help can come in financial support for investments in separate collection and recycling infrastructure, but also in the form of sharing knowledge and expertise gained from the long experience with EPR in the EU.

Introducing EPR to other product groups, including medicines, is likely to prevent environmental damages from littering and illegal dumping, and to direct more flows to reuse and recycling. EPR shifts the financial burden of collecting and treating end-of-life products from taxpayers to producers and consumers, and is consistent with the Polluter Pays Principle. The environmental and economic effects of EPR eventually depend on its design, mainly on the characteristics of the instrument mix used. Ex-ante cost-benefit analyses would help determine the composition, scope and stringency of an efficient EPR policy mix for each product group considered.

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Appendix – Interview partners

Interviews were conducted with the organisations presented in Table A.1.

Table A.1

Interviewed organisations

National government	Ministry of Infrastructure and Water Management (Ministerie IenW) Human Environment and Transport Inspectorate (ILT) National Institute for Public Health and the Environment (RIVM) Rijkswaterstaat
Local authorities	Association of Dutch Municipalities (VNG) Union of Water Authorities (Unie van Waterschappen)
EPR organisations	Association of Producer Responsibility Organizations (VPN) Autorecycling Nederland (ARN) Stibat Stichting Batterijen
Industry associations	Dutch Association of Pharmacists (KNMP)