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WHAT HAVE WE LEARNED ABOUT EXTENDED PRODUCER RESPONSIBILITY IN THE PAST DECADE?

A SURVEY OF THE RECENT EPR ECONOMIC LITERATURE

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What Have We Learned About Extended Producer Responsibility in the Past Decade? A Survey of the Recent EPR Economic Literature

Daniel Kaffine and Patrick O'Reilly*

Abstract

The last decade has seen a substantial increase in implementation and interest in Extended Producer Responsibility (EPR) programs. While on-the-ground implementation of EPR programs has grown, an academic literature on the economics of EPR has also developed. This document provides an overview of lessons learned from this literature. It identifies key results from the literature and possible areas for further analysis, with an eye towards informing policymakers regarding the design of EPR programs. Key insights from the literature that policymakers may want to take into consideration are as follows. First, in selecting policies within the EPR framework, multi-instrument policies, such as deposit/refund, are likely to be more efficient than single instrument policies such as an advance disposal fee. Second, while collective PROs may be attractive in terms of taking advantage of economies of scale and reducing the need to monitor individual firms, care should be taken that market power is not exploited. Third, while most EPR policies provide DfE incentives, policies that directly target product characteristics (weight, recyclability, etc.) will provide the most direct incentives. Finally, though there is evidence that EPR policies can achieve their environmental goals, empirically it is still an open question which policies will achieve those goals at the least cost.

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EXECUTIVE SUMMARY

The last decade has seen a substantial increase in implementation and interest in Extended Producer Responsibility (EPR) programs. Such programs assign financial or physical responsibility to producers for their products post-consumption, with the aim of reducing waste disposal, resource conservation, increasing recycling, and encouraging more environmentally-friendly product design. EPR is an important policy approach for the implementation of Sustainable Materials Management (OECD, 2012) and the resource preservation and whole of life-cycle approaches that it promotes. While on-the-ground implementation of EPR programs has grown, an academic literature on the economics of EPR has also developed over the past decade. This report surveys both the academic economic literature as well as the global implementation of EPR programs, with the goal of updating the 2001 OECD report "Extended Producer Responsibility: A Guidance Manual for Governments." It identifies key results from the literature and possible areas for further analysis, with an eye towards informing policymakers regarding the design of EPR programs. An important note to be made upfront is that the economics literature on EPR is heavily tilted towards theoretical studies, and thus there is a clear need for further empirical research to confirm or disconfirm the conclusions from the theoretical literature.

While the lessons learned from traditional environmental regulation have some application to EPR policies, there are important differences worthy of dedicated analysis. For many products where EPR policies have been considered, environmental damages may be felt years after the point of production, environmental damages associated with the product depend on how the good is used and ultimately disposed of, the environmental consequences of production decisions may depend on whether virgin or recycled inputs are used to make the good, and the potential for recycling creates a link between use, disposal decisions, and production decisions.

Key insights from the economic literature

Instrument Choice

- In selecting policies within the EPR framework, deposit/refund, upstream combined tax/subsidies (UCTS), and take-back requirements, eventually with tradable permits, are likely to be more efficient than single instrument policies such as an advanced disposal feel (ADF), virgin materials tax, or recycling content standards. Provided that different instruments are well aligned, the simple intuition is that multi-instrument policies are able to target different margins of adjustment by firms and households, achieving waste reductions at a lower cost. However,

- It is unlikely that uniform standards or incentives across different material types are efficient – all forms of EPR policies should reflect differences in the social costs associated with production, use and disposal of different types of goods.

Competition

- The literature on competitiveness and EPR's echoes long-standing concerns that environmental regulation can exacerbate market power concerns. The creation of collective PROs provides additional opportunities for price-gouging, entry-deterrence, and other anti-competitive activities. This suggests that in concentrated industries, individual take-back requirements may be more appropriate than collective PROs, with the caveat that potential market power concerns (or the costs of regulating market power) should be balanced against potential gains from economies of scale in collection and recycling activities as well as monitoring costs. PROs need to be monitored by regulators to ensure that they are not engaging in anticompetitive behaviour. By contrast, in markets that are closer to the perfect competition ideal, collective PROs may provide economies of scale benefits and lower monitoring costs with less risk of market power concerns. Further empirical research in this area to supplement the theoretical literature would be beneficial.

Design for Environment

- Most types of EPR policies can provide some incentives for Design for the Environment (DfE), but more empirical evidence is needed to understand the industry-wide DfE effects from EPR policies. Policies that target product characteristics such as recyclability will provide more direct DfE incentives. Market-based policies such as a deposit/refund or a UCTS per unit weight will provide more direct incentives for DfE than similar policies per unit consumed. Take-back mandates with tradable credits will provide more direct DfE incentives than take-back mandates where fees are simply based on market-share.

Environmental and Cost Effectiveness

- While EPR policies can achieve environmental goals (reduced waste and increased recycling), more systematic empirical evidence is needed. In particular, empirical assessments that carefully control for confounding factors and are able to credibly identify the effects of EPR policies are needed.

- Empirical studies of costs-effectiveness of market-based policies, generally confirm that multi-instrument policies such as deposit/refund which affect multiple choice margins are better than single instrument policies such as an advanced disposal fee (ADF). There is clearly a gap in the academic literature regarding the cost-effectiveness of policies such as take-back and recycling content standards, as well as the general equilibrium effects of EPR policies. One aspect that deserves further inspection is the general equilibrium costs of EPR policies.

EPR in practice – insights from an original survey of 395 existing EPR programs

Product Focus

- Small consumer electronics appear to be the most prevalent product covered under EPR. Packaging (including beverage containers), tires and vehicles/lead-acid batteries are the next largest groups of products covered. Less common products include used oil, paint, chemicals, large appliances, and florescent light bulbs.

Type of Instrument

- Take-back requirements are the most commonly used EPR policy (72%), and they are used for a large variety of products. Advanced disposal fees are the next most common policy (16%) and tend to have a wide range of product applications. Deposit/refund is used slightly less than the ADF (11%), though it is concentrated in the used beverage container and lead-acid battery markets. Upstream combined tax/subsidy, recycling content standards, and virgin material taxes are sparsely used, if at all.

Dynamic of EPR Adoption

- Nearly three-quarters of EPR policies sampled were implemented in the years since the 2001 OECD Guidance Manual. Thirteen per cent were implemented just in the last three years, and only 5% were implemented prior to 1990. Recently adopted

EPR policies have tended to be take-back requirements.

Regional Differences

- Regionally, the United States appears more inclined to adopt market-based EPR policies such as deposit/refund and ADF, though slightly more than half of US policies are take-back requirements. Take-back policies comprise about 80% of EPR policies outside the US. Comparing across US states, EPR policy adoption is correlated with proxies for environmental attitudes.

While the findings from the academic economic literature provide insight into a number of important questions, the existing studies in the literature are heavily tilted towards theoretical and conceptual analysis. However, there are a number of important questions that may be better answered via context-specific case studies, and as such, the report concludes with an outline of the key quantitative and qualitative information and questions that follow-up case studies will need to address.

WHAT HAVE WE LEARNED ABOUT EXTENDED PRODUCER RESPONSIBILITY IN THE PAST DECADE? A SURVEY OF THE RECENT EPR ECONOMIC LITERATURE[2]

1. Introduction

Extended Producer Responsibility (EPR) is a broad collection of environmental policies encouraging or requiring manufacturers to accept financial and/or physical responsibility for their products after the point of sale. Rather than a specific policy, EPR is better viewed as a set of policies that policymakers can select from, which can flexibly adapt to local values, legislative climates, economic contexts, or legal constraints. Within the broad set of policies that fall under the definition of EPR framework, conservation of raw materials, waste reduction and encouraging environmentally-friendly product design are a common theme, with varying emphasis on other goals such as reduction of environmental damages from the production process. EPR is an important policy approach for the implementation of Sustainable Materials Management (OECD, 2012) and the resource preservation and whole of life-cycle approaches that it promotes.

To some extent, EPR is not a new concept, in part because recycling markets can and have rewarded producer responsibility for products beyond the extract-make-use-dispose lifecycle. Reuse and recycling of disposed products has a long history, driven primarily by price signals and private incentives. However, such private incentives for treatment of disposed products may be insufficient to reach the socially desired or optimal level of producer responsibility. Thus, while recycling markets have been in existence for centuries, growing concerns regarding household waste and other sources of pollution in the 1990s led to the development of new policy options such as EPR, as summarized in the EPR Guidance Manual developed by the Organisation for Economic Co-operation and Development (OECD, 2001).

Before proceeding, it is important to note that we are adopting the definition of EPR as described in the 2001 OCED EPR Guidance Manual, as well as the goals and policies considered EPR within that document:

> "EPR is an environmental policy approach in which a producer's responsibility, physical and/or financial, for a product is extended to the post-consumer stage of a product's life cycle. There are two related features of EPR policy: (1) the shifting of responsibility (physically and/or economically; fully or partially) upstream to the producer and away from municipalities, and (2) to provide incentives to producers to incorporate environmental considerations in the design of their products."

The 2001 OECD EPR Guidance Manual also states that EPRs can be implemented by using one of three basic categories of instruments: take-back requirements, economic instruments, and performance standards. Economic instruments include incentive-based instruments such as deposit/refund, advanced disposal fees, material taxes and the upstream combined tax/subsidy.

However, it is important to recognize that there are multiple definitions and interpretations of EPR in the literature, as noted in Lifset et al. (2013). Many of these definitions are narrower than the OECD

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[2] Authored for the OECD by Daniel Kaffine (University of Colorado Boulder, USA) and Patrick O'Reilly (Colorado School of Mines, USA) with key assistance from Peter Börkey (OECD). We thank WPRPW delegates and members of the expert group on Extended Producer Responsibility for useful comments. Thanks in particular to: Arne Campen, Bruce Edwards, Nicole Koesegi, Reid Lifset, Jacinthe Séguin, Patrik Solderholm, Tomohiro Tasaki, Yoichi Toyama, and Christoph Vanderstricht.

definition and essentially identify EPR as mandatory take-back systems.[3] We focus on the OECD definitions for two reasons. First, this survey is conceived as an update to that specific document, and as such it is logical to maintain a consistent definition. Second, this is a survey of the academic economic literature on EPR, which has generally followed the OECD definition.

The remainder of the introduction provides an overview of EPR policies and goals as described in the 2001 OECD EPR Guidance Manual, as well a discussion of similarities and differences between EPR and other forms of environmental regulation. The rest of this paper updates earlier OECD efforts by surveying the academic economic literature on EPR over the last decade or so, and concludes with an overview of EPR programs in practice. Because the literature has been dominated by theoretical analysis, the paper also provides a proposal on key issues that empirical EPR case studies (to be conducted by OECD in 2013 and 2014) should address in order to exploit the information and advancements in the literature that have accumulated over the past 10 years.

1.1 EPR policies and goals – Overview

To facilitate the literature discussion that follows, this section briefly reviews the policies and goals of EPR programs.[4] The focus is on how these policies and goals have been considered in the academic economics literature on EPR. The 2001 OECD EPR Guidance Manual identified four goals of EPR policies.

- Source reduction (natural resource conservation/materials conservation)
- Waste prevention
- Design of more environmentally compatible products
- Closure of materials use loops to promote sustainable development

According to the 2001 OECD EPR Guidance Manual, policies considered under the framework of EPR include: i) product take-back with recycling targets, ii) deposit/refund, iii) advanced disposal fees (ADF), iv) virgin materials taxes, v) upstream combination tax/subsidy (UCTS), and vi) recycling content standards. While all of the alternative definitions and interpretations of EPR include product take-back as an EPR policy, there is less agreement on whether the remaining policies should also be considered EPR.[5]

[3] For example, the Product Stewardship Institute defines EPR as: "[A] mandatory type of product stewardship that includes, at a minimum, the requirement that the producer's responsibility for their product extends to post-consumer management of that product and its packaging." (http://productstewardship.us/displaycommon.cfm?an=1&subarticlenbr=231). Environment Canada defines EPR as "[A] policy approach in which a producer's responsibility, physical and/or financial, for a product is extended to the post-consumer stage of a product's life cycle." As noted by Lifset et al. (2013), despite differences in definitions, "In most policy implementation and public discourse, however, EPR signifies the assignment of responsibility for end-of-life management of products and packages to producers."

[4] Note that the shifting of the burden from local governments to producers through EPR is not listed as a goal in and of itself. As the 2001 OECD EPR Guidance Manual notes: "Therefore, EPR policy should be designed to provide incentives to encourage producers to absorb social costs from the treatment of their products. Any unavoidable costs could therefore be incorporated into the product pricing. The producer and the consumer – in lieu of the taxpayer - would pay for the social costs (externalities)." (page 59).

[5] As the 2001 OECD EPR Guidance Manual notes: "Often take-back is regarded as the *purest* form of EPR." (page 41). However, it also notes that: "Whereas take-back requirements use the assignment of responsibility to the producer for the end-of-life management of their products to meet the policy objectives, economic instruments can also be used toward the same objectives." (page 41). Furthermore, it is also important to note that while the economics literature has typically considered these policies as distinct, mutually exclusive policies, in practice EPR may incorporate a mix of

The first policy, product take-back with recycling targets, mandates "producers" in a country (or region) to take back products they have put on the market for the first time in that region/country at the end of the products' life. They are often coupled with targets for reuse, collection and/or recycling. This policy can take many different forms, depending on whether individual firms or collectives of firms must organize the collection and recycling of the products they put on the market, or whether each firm is subject to a specific take-back target or if firms can trade recycling credits subject to an industry-wide target. The second policy, deposit/refund, is a market-based policy in the sense that it provides incentives that lead to waste reduction.[6] Under a deposit/refund system, a deposit is charged at the point of sale, and then redeemed if the product is brought back to a collection point for recovery or reuse.[7] Dropping the "refund" piece yields the third policy, the advanced disposal fee, whereby a fee is charged at the point of sale for the product.

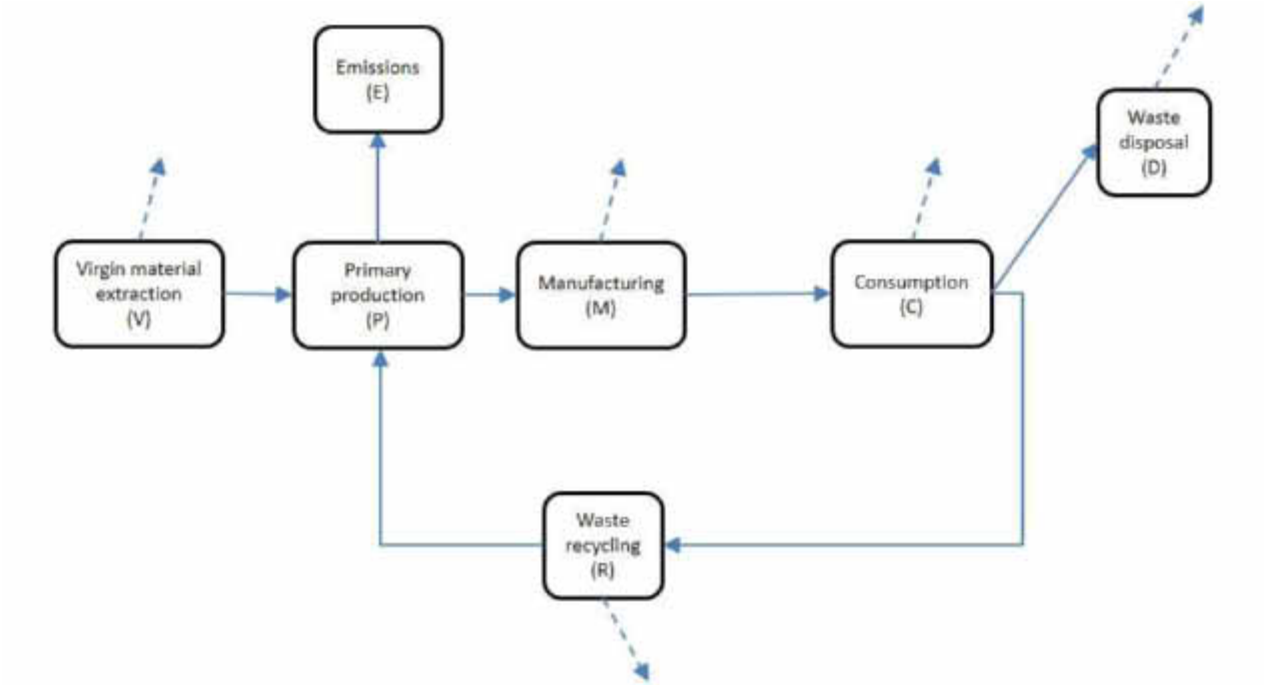
The fourth policy, a virgin materials tax, aims to incentivize the use of recycled inputs instead of virgin inputs in the production process. While the previous two market-based policies are implemented at the point of sale, this policy is implemented "upstream" in the production process (see below). The fifth policy, an upstream combination tax/subsidy works like a deposit/refund, but is levied upstream in the production process instead of at the point of sale. The tax would be a materials tax levied upstream on primary producers, while the subsidy would be paid to upstream collectors of recycled products. The final policy, recycling content standards, is simply a mandate requiring that a specific percentage of inputs used in the production process is from recycling processes. The relationship between the goals of EPR policies and the mechanisms by which they operate are discussed below.

instruments. For example, in France, EPR for batteries involves an advanced disposal fee plus downstream subsidy, prevention and recycling targets, and mandatory take-back.

[6] Economists typically differentiate between policies such as mandates and standards which require specific targets be met by firms or consumers (referred to as "command and control" policies), and those that provide incentives for firms and consumers to "voluntarily" and flexibly change their behaviour in socially beneficial ways (referred to as "market-based" or "incentive-based" policies).

[7] In practice, the refund may only be received if a new product is purchased. Also, while the economics literature on EPR has generally not examined the use of the revenue from the deposit (or advanced disposal fee), in practice it is important to consider whether it goes to the general national budget, specific waste management, or general environmental budget.

Figure 1. Product Life-cycle



Note: Solid arrows represent material flows; dashed lines indicate other potential sources of externalities (such as emissions) not discussed explicitly in the text. Source: Authors

Figure 1 is helpful in visually illustrating the simple conceptual links in the life-cycle of a product, as well as how the above goals and policies fit into the life-cycle of a product.[8] The resource extraction sector (*V*) produces virgin material inputs, which are then used by primary producers (*P*) along with recycled inputs (*R*) to produce primary materials. These primary materials are then used by manufacturers (*M*) to produce consumer goods for consumption by households and other consumers (*C*).

A well-known result from economic theory is that if the prices at each stage of the process reflect the true social benefits and costs, then the resulting private production and consumption decisions would be economically efficient. In reality of course, there are likely external costs to society that are not reflected in the private prices. Thus, consumers who choose to dispose of products may generate external waste disposal costs (*D*). Similarly, production processes may generate emissions or other forms of pollution (*E*) that generate costs for the rest of society.[9] Extraction of natural resources (*V*) may also generate external costs (potentially including the use and loss of limited natural resources) that are not reflected in the price of the natural resource. It is these external costs that provide an economic justification for EPR policies, and other environmental regulations for that matter (Walls 2004).

Turning to the goals of EPR, we can see that source reduction and waste prevention can be thought of as reducing *V* and *D*, design of more environmentally compatible products as reducing *E* (or alternatively as reducing the amount of virgin material extraction *V* and primary materials *P* used by manufacturers *M*), and closure of material loops as increasing *R*. From an economic perspective, the case for why reductions

[8] There are many versions of this basic figure in the academic economics literature. The structure of this particular figure has been selected to highlight specific features of EPR goals and policies. The discussion that accompanies this figure is intended to be general and abstracts from many of the particular considerations that exist in specific markets or for specific policies. For example, here recycling activities feed back into the production process, while in practice some recycled materials could be directly reused or prepared for energy recovery.

[9] The dashed lines in Figure 1 reflect the fact that emissions and waste can arise at all stages of the product life-cycle.

in *D* and *E* are economically justified is clear, as they represent obvious external costs. The external cost justification for reducing *V* is less clear, though may be relevant in certain contexts such as lost ecosystem services from old-growth timber harvest.[10] By contrast, increasing *R* has little direct economic justification; the economic case must rest on the indirect effects of increasing *R* which thereby reduce *D*, *E*, and *V*.[11]

In terms of policy intervention, i) product take-back with recycling targets increases the *R* channel, while altering the use of virgin and recycled inputs by *P*. Next, ii) deposit/refund policies place a charge on transactions between *M* and *C*, while altering the disposal decision by *C* to increase *R*. If the refund is not included as a policy, then iii) advanced disposal fees are simply a charge on transactions between *M* and *C*. Related to a deposit/refund, the v) upstream combination tax/subsidy levies a charge on transactions between *P* and *M*, and then subsidizes the *R* channel. Finally, vi) recycling content standards alter the use of virgin and recycled inputs by *P*.

1.2 EPR distinguished from other forms of environmental regulation

Before reviewing the literature, it is useful to consider the similarities and differences between EPR policies and other forms of environmental regulation from the perspective of the economic literature. The purpose of this section is twofold. First, to identify aspects of EPR polices and the products they regulate that are similar to other forms of environmental regulation. Given that the academic literature on other forms of environmental regulation is much larger than the literature on EPR specifically, identification of similarities may provide some insight into EPR policies in the absence of specific academic studies. Second, it is also useful to identify ways in which EPR policies and products differ from other forms of environmental regulation, as it provides motivation for the existence of studies focusing specifically on EPR policies. As a catchall term for other forms of environmental regulation, here "traditional environmental regulation" refers to policies regulating air pollution, water pollution, land pollution, etc. and could include command-and-control regulatory approaches such as emission standards and mandates, as well as market-based policies such as tradable permit systems and emission and effluent taxes.

1.2.1 Similarities

The alignment between EPR's producer-oriented approach and the targets of traditional environmental regulation is straight-forward to some extent: achieving social welfare gains through environmental improvement via regulatory intervention over firms' behaviour. Just as traditional environmental regulation is concerned with correcting negative externalities such as pollution, EPR policies can be designed to address negative externalities such as downstream waste, upstream pollution, and natural resource conservation.[12] Both EPR and traditional environmental regulation compel markets to internalize these negative externalities such that total social welfare improves. Both forms of regulation also come in many shapes and sizes, requiring breadth and flexibility that allows for solutions that are sensitive to the political, legislative, and economic demands of stakeholders affected.

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[10] In theory, if property rights over virgin materials are well-defined, then private prices reflect any costs associated with increasing scarcity. In practice, many virgin materials are extracted under imperfect property rights or in the presence of other market distortions, providing some direct justification for reducing *V*. Of course, reducing *V* may have indirect justification by virtue of the fact that reductions in raw material use may have implications for the social costs of *E* or *D*.

[11] One caveat is that it may be the case that increased recycling provides a "warm-glow" psychological benefit to those that recycle (Kinnaman, 2006), in which case there is some direct justification for increased *R*.

[12] Briefly, negative externalities reflect costs imposed on others that are not borne by the firm producing them. Such spillover costs are considered a form of market failure, whereby total social welfare could potentially be improved via intervention into markets.

It is useful to recognize that although some examples of traditional environmental regulation were not chosen with EPR specifically in mind, they are nonetheless consistent with the essence of EPR. For example, a mine-mouth or wellhead carbon price holds primary producers of fossil fuels financially responsible for negative externalities ultimately generated by consumption. The price signal from such a tax is internalized by the market, providing signals to various producers and consumers to reduce their consumption of carbon. Similarly, the Federal Excise Tax on tires in the United States was initially instituted nearly a century ago in part to encourage conservation of natural resources, and while this obviously predates the EPR framework, it is operationally similar to EPR policies such as an advanced disposal fee. Thus, to a certain extent, the lessons learned from traditional environmental regulation have some application to EPR policies as well.

1.2.2 Differences

While the above section noted that both EPR and traditional environmental regulation involve regulatory intervention to internalize externalities and thereby affect firm behaviour, there are also several key differences worth highlighting. It is these differences that necessitate a focused examination of EPR, as they introduce a number of complexities that are not present in the economics literature on other environmental regulations. For example, in the case of emissions regulation, environmental damages can be simply measured at the point of production and then regulated.[13] By contrast, for many products where EPR policies have been implemented, the measurement of total environmental damages is not so straightforward. One cannot necessarily place a meter on the production process to measure the total negative externalities that will be generated years or even decades down the road.

Expanding on the discussion above, one distinction between EPR policies as opposed to traditional environmental regulation is that they consider the whole life-cycle of products, thereby taking into account environmental damages associated with the ultimate disposal of products. By contrast, traditional policies address environmental concerns at one point in time in the life-cycle of products. This usually prevents these policies from considering externalities all along the production and consumption chain. Indeed, only in simple cases (such as the wellhead carbon tax discussed above) can we a priori determine the level of externality from the point of production.[14] Similarly, on the production side, the environmental consequences of production decisions may depend on whether virgin or recycled inputs are used to make the good. Typically, using recycled inputs results in lower energy use and pollution production, implying that two nearly-identical end products (from the consumer’s perspective) may have very different environmental footprints associated with their production.

Finally, the potential for recycling is a key feature of goods typically considered for EPR policies, a feature typically absent in traditional environmental regulation. Because recycled goods often re-enter various production processes, this creates a complicated link between use, disposal decisions, and

[13] For example, Continuous Emissions Monitoring Systems (CEMS) are used by all electricity generating plants in the United States with more than 25 megawatts of capacity to provide hourly measurements of sulphur dioxide, nitrogen oxides, and carbon dioxide to EPA regulators. Installations subject to the EU Emissions Trading System (ETS) must also have monitor and report their emissions per the European Commission’s Monitoring and Reporting Regulation. While pollutants may have long-lasting impacts, the magnitude of the pollutant emitted at any given point in time can be observed relatively easily.

[14] A product such as a car battery that is properly recycled may generate no negative externalities from waste disposal, compared to an improperly disposed of product that can generate large externalities. By contrast, the carbon externalities from a barrel of oil are easier to asses, as carbon flow can be accurately inferred from product flow, by virtue of the same measurements that help producers determine energy content.

production decisions.[15] The complexity of EPR analysis from an economic perspective stems from the fact that there are many margins for adjustment at the production and consumption level that ultimately affect the social costs for a given product. This also gives rise to the need to consider the multiple policy instruments inherent to EPR policies, as there may be multiple objectives or sources of social cost that require multiple instruments to address.

Thus, while both traditional environmental regulation and EPR policies seek to improve social welfare and reduce environmental damages by requiring firms to internalize external costs, the above noted differences motivate the need for a focused economics literature to better understand EPR as distinct from the broader array of more traditional environmental policies.[16]

The remainder of the paper proceeds as follows. Section 2 provides an overview of economic studies of EPR, with a focus where possible on academic work published since the 2001 OECD Guidance Manual. Section 3 provides an overview and survey of EPR policies in practice, with an eye to providing a typology of various EPR programs. Finally, Section 4 provides conclusions. Guidance for further research and economic case studies of EPR programs is provided in annex 1 and an inventory of EPRs in use is provided in annex 2.

2. Economic literature overview

This section provides an overview of the economics literature on EPR, with a focus on studies that post-date the 2001 OECD EPR Guidance Manual. The intention is to provide an overview of new scholarship on the economics of EPR in the past decade that may enhance and extend our understanding of the discussion of EPR in the original Guidance Manual. We stress that the discussion below is intended to report and summarize what the literature has found, and as such the discussion is not intended to be an endorsement of these findings. We begin with the economic literature on policy analysis, including instrument choices, market power associated with producer responsibility organizations (PROs), and incentives aimed at increased design for the environment (DfE). Next we examine the literature on EPR effectiveness, both in terms of material effectiveness as well as cost-effectiveness. Finally, we discuss several potential challenges for EPR that have analogues in the broader economics literatures: free rider problems, orphan products (defunct manufacturers), and trade considerations.

2.1 Economic policy analysis

2.1.1 Instrument choice

The literature on instrument choice is primarily theoretical and can be broken down into studies examining policies with downstream and upstream externalities, recycling content standards, and product take-back mandates.[17] In the context of a single, downstream externality associated with waste, early work by Dinan (1993) and Palmer and Walls (1997) laid the basic groundwork for thinking about policy choice

[15] For example, some recyclable goods may be reincarnated as lower-grade "down-cycled" products or components (i.e. "plastic lumber" boardwalks made from an array of post-consumer plastics). Other goods are immediately recyclable as like-new refurbished or simply washed versions of their former selves.

[16] A marginal disposal fee (pay-as-you-throw) is closer to traditional environmental regulation, representing a first-best "Pigouvian" tax on waste disposal externalities. The term "first-best" denotes the fact that a tax set equal to the marginal external damage of an externality is typically the least-cost policy for achieving a given reduction in the externality. However, there are many cases where such policies are unavailable, due to technical or political constraints, or as discussed below, because they create illegal disposal incentives.

[17] The essence of this literature addresses the question: Given the many policy options under EPR, what policy or policies should we choose? It should be noted that the studies in this section are theoretical and nature, and as such are quite general and likely overlook practical considerations that influence instrument choice.

in the context of waste disposal. These papers start from the premise that unit-based waste pricing is infeasible due to illegal disposal concerns and show that a deposit/refund policy is an alternative efficient policy to unit-based pricing.[18] The basic intuition is that under a deposit/refund, a household only faces a charge if they choose to dispose products, thereby mimicking a unit-based pricing policy. Walls (2013) makes the case that deposit/refund polices may have some additional advantages over unit-based pricing in terms of monitoring and enforcement (litter, for example), as well as tax evasion (product sales taxes are difficult to avoid). One possible problem with deposit/refund programs where households receive the refund for recycling is that collectors may simply dispose of materials after collecting them from households. Ino (2011) notes that in such a case, the quantity of material collected from households does not equal the quantity of material actually recycled.[19] As such, the potential for illegal disposal by collectors of recycled goods reduces the refund that should be provided as less material is ultimately diverted from the waste stream. Alternatively, Walls (2013) notes that moving the refund "upstream" to processors or distributors who actually deliver the recycled material to processors would avoid the issue of illegal disposal by collectors of recycled material. In terms of determining the appropriate level of the deposit/refund, the standard Pigouvian prescription generally holds, in the sense that the deposit/refund should be set equal to the marginal external damage associated with disposal of the product.

The bulk of the literature described above is primarily concerned with externalities generated by waste disposal; however, as noted in the introduction, there may be "upstream" externalities associated with the production process, which in turn may be affected by waste disposal decisions. The unification of downstream and upstream externalities into a single framework was examined by Walls and Palmer (2001), who analyse policy options to address both sources of externalities. They note that multiple instruments are needed to correct for multiple externalities (such as waste disposal externalities and air pollution externalities), with the optimal policy consisting of a downstream deposit/refund and upstream taxes equal to the marginal social damage of upstream emissions. While casting some scepticism on integrated product approaches, they note that methods such as life-cycle analysis are important for determining the marginal social damages and thus the proper corrective taxes.

Acuff and Kaffine (2013) also consider upstream and downstream externalities and show that if an upstream tax on externalities (such as carbon pricing) is unavailable, downstream policy levels (policies such as deposit/refund or advanced disposal fees levied at the point of consumption) should be accordingly increased, potentially by a substantial amount. Furthermore, even if the marginal social damage of disposal is identical across products, the optimal policy level may vary substantially by material to account for differences in the upstream production processes.[20]

As a general rule, the economics literature has generally viewed regulatory standards, such as recycling-content standards, with less enthusiasm than the market-based approaches discussed above. Indeed, a long line of literature, from Helfand (1991) to Fullerton and Heutel (2010), shows that environmental standards introduce distortions and are unlikely to be efficient without additional policies. Palmer and Walls (1997) specifically consider recycling-content standards, showing that recycling content

[18] Eichner and Runkel (2005) also find that that a deposit/refund policy is efficient in the short and long-run in a model where recyclability of durable goods is considered. In contrast, Dinan (1993) shows that a virgin materials tax is *not* an efficient policy alternative to unit-based pricing, a point echoed in Fullerton and Kinnaman (1995).

[19] Ino (2011) distinguishes between the "collecting" process, whereby household residuals are gathered, and the "reprocessing" process, where those residuals are converted into usable materials. Depending on the value of the residuals, there may be an incentive to collect the residual and the associated refund (described by Ino (2011) as a "free gift"), and then simply dispose of the residual.

[20] For example, they find that the optimal deposit/refund on aluminium is USD 424 per ton compared to only USD 26 per ton for glass. This large discrepancy is driven by the fact that production of aluminium from recycled content is substantially less energy and emissions-intensive.

standards require complicated additional taxes and instruments (above and beyond the content standards) to be efficient. Furthermore, if recycling content standards are set uniformly across firms within a given industry, then potential cost-savings are missed in the sense that it may be more cost effective for low-cost firms (low-cost in the sense of ability to incorporate recycled content into their production process) to use more recycled content, and high-cost firms to use less recycled content.

Take-back requirements have also been examined in the economics literature, with Fullerton and Wu (1998) showing that a take-back requirement coupled with a disposal charge faced by the firm is as efficient as a deposit/refund policy. Ino (2011) notes that if firms are heterogeneous in terms of their production and recycling technologies, then a take-back requirement coupled with tradable credits is equivalent to a tax-subsidy scheme such as deposit/refund in the presence of illegal disposal. In other words, if firms that face high costs of meeting their take-back requirements are able to purchase "credits" from low-cost firms, the total industry take-back requirement can be met at a cost equivalent to a deposit/refund. Matsueda and Nagase (2012) explores take-back requirements with tradable credits between firms (i.e. packaging waste recovery notes [PRNs]), and show that, somewhat surprisingly, increasing landfill taxes coupled with a PRN system may actually increase landfill use.[21] Taken together, these findings would suggest that, while it is possible for take-back requirements to be designed efficiently, there may be non-obvious complicating interactions with other policies that need to be considered.

Thus, several policy-relevant results have come to light in the last decade. First, within the set of EPR policies that are market-based (deposit/refund, ADF, UCTS, virgin materials tax), multi-instrument policies such as a deposit/refund and UCTS are likely to be more cost-effective than single instruments such as an ADF or a virgin materials tax. The simple intuition is that multi-instrument policies are able to target different margins of adjustment by firms and households, achieving waste reductions at a lower cost. Second, simple command and control policies such as recycling content standards are unlikely to be efficient, though more complicated policies such as take-back requirements with tradable permits between firms may be as cost-effective as market-based policies. Finally, it is unlikely that uniform standards or incentives across different material types are efficient – both command and control and market-based policies should reflect differences in the social costs associated with disposal and production of different types of goods.

2.1.2 Market power in Producer Responsibility Organisations

In some implementations of EPR, producer responsibility organizations (PROs) carry out take-back, collection, or recycling activities of end-of-life products on behalf of a producer or set of producers. While PROs can take advantage of economies of scale and lower monitoring costs, one concern of note is market power associated with PROs. Runkel (2003) analyses several EPR policies, and finds that EPR generates welfare gains under perfect competition in the product market, but may also generate welfare losses under imperfect competition due to links between durability and output. This occurs because EPR encourages durability; however, it also induces firms to respond by reducing output, further exacerbating market power. Thus, while EPR internalizes disposal costs, it ignores market imperfections arising from market power. Fleckinger and Glachant (2010) extend this idea further to allow for the potential of collusion when producers develop a cooperative PRO. They suggest that the very flexibility PROs provide firms (for meeting EPR policy obligations) creates an additional potential for collusion and market power, which may imply a need to regulate the fees charged by PROs.[22]

[21] This counterintuitive result arises due to the fact that the landfill tax privately encourages recycling activities, lowering the cost of PRN's. By lowering input costs, total production increases by more than the increase in recycling activity, leading to an overall increase in waste.

[22] The intuition is that if firms are free to join together and design the PRO fees, these fees may be set too high, in order to reduce output and take advantage of market power. The authors conclude that collective PRO's are less socially desirable

Heyes (2009) provides an overview of anti-competitive aspects of EPR in a broad study of how environmental regulation impacts competition. The author notes that environmental regulation in general can create potential for market power concerns, and that care should be taken in evaluating and designing regulations to not exacerbate market power, echoing the points made above by Runkel (2003) and Fleckinger and Glachant (2010). Mock and Perino (2008), in their analysis of the European Directive on Waste Electric and Electronic Equipment (2002/96/EC), argue that market entry is deterred by EPR through raising common costs of disposal. Hayes (2009) identifies the Duales System Deutschland (DSD) as having raised antitrust concerns, mainly due to institutional arrangements between DSD and associated waste-recovery firms. Specifically, Lehmann (2004) notes the concern that established firms might leverage DSD to limit existing competition and extract rents from upstream firms or to exclude potential rivals, a point echoed in Walls (2006) on the potential anti-competitive impacts of PROs in the forms of price gouging and facilitating collusion.[23]

Thus, the literature on competitiveness and EPR's echoes long-standing concerns that environmental regulation can exacerbate market power concerns. Furthermore, the creation of collective PROs provides additional opportunities for price-gouging, entry-deterrence, and other anti-competitive activities. This suggests that in concentrated industries, individual take-back requirements may be more appropriate than collective PROs, with the important note that potential market power concerns (including costs of any regulation to reduce or eliminate market power) should be balanced against potential gains from economies of scale in collection and recycling activities and reductions in monitoring cost. At a minimum, PROs may need to be monitored by regulators to ensure that they are not engaging in anticompetitive behaviour.[24] By contrast, in markets that are closer to the perfect competition ideal, collective PROs may provide economies of scale benefits and lower monitoring costs with less risk of market power concerns (or costly regulation). Further empirical analysis examining effectiveness of market power regulation and whether or not collective PROs have resulted in price-gouging or entry-deterrence would clearly be beneficial.

2.1.3 Design for the environment

One frequently stressed feature of EPR policies is the extent to which it encourages "Design for the Environment" (DfE) by upstream production firms (producers involved in product and process design). The idea is that EPR policies can provide incentives for firms to engage in product redesign to minimize waste disposal costs; for example by "lightweighting" products, reducing packaging use, or enhancing recyclability. Fullerton and Wu (1998), Calcott and Walls (2000), and Eichner and Pethig (2001) show that if regulators can perfectly observe various measures of "recyclability" of a particular product, then a multiple instrument tax/subsidy system can encourage the efficient level of DfE via price signals. Because regulators can observe the characteristic "recyclability," it is somewhat straightforward to provide the correct incentives for firms to increase that recyclability. Runkel (2003) shows that other EPR polices can also lead to DfE, in this case by increasing durability of goods. As EPR policies hold firms accountable for the ultimate disposal costs of their products, they provide an incentive for firms to increase durability so as to avoid the incurrence of that cost. Calcott and Walls (2005) cast some doubt on the ability of regulators

than individual EPRs in the presence of imperfect competition; however they caveat this by noting that their model does not include potential economies of scale and lower monitoring costs that collective PROs may provide.

[23] On the other hand, Lehmann (2004) notes that there may be some merit to the particular institutional arrangement of DSD overlooked by those focused on antitrust concerns, though it is unlikely that it was as efficient as a hypothetical competitive market structure. Specifically, the management structure of DSD may have helped reduce transaction costs associated with the "hold-up problem" inherent when DSD was created as the sole firm engaged in the dual system of packaging waste collection and recycling.

[24] For example, policymakers are now contemplating the need to prohibit PRO's from directly or indirectly competing within the market in which they have a statutory responsibility.

to measure "recyclability," focusing instead on the costs associated with recycling a product.[25] Given the inability to precisely observe recyclability, they show that a product tax combined with a recycling subsidy and a disposal fee provides the correct DfE signals.

Walls (2006) suggests that advanced disposal fees and tax-subsidy instruments are associated with favourable DfE incentives, while PRO arrangements (with market-share based fees) provide less DfE incentives unless costly product-characteristic targeting is implemented.[26] Walls (2006) also provides a useful table of various EPR and non-EPR policies and their direct and indirect DfE incentives. Brouillat and Oltra (2012), using agent-based simulation, examine DfE incentives under a variety of policies. They find that recycling subsidies must be differentiated by "recyclability" to be effective, tax-subsidy systems encourage recycling-specific innovation by a large number of firms, while recycling standards encourage broader product innovation by a select group of firms.

Thus, most types of EPR policies (with the exception of an ADF per unit) can provide some incentives for DfE, though as Walls (2006) notes, more empirical evidence is needed to understand the industry-wide DfE effects from various EPR policies.[27] Market-based policies such as a deposit/refund or a UCTS per unit *weight* will provide more direct incentives for DfE than similar policies per unit consumed.[28] Take-back mandates with tradable credits, such as the PRN system in the United Kingdom, will provide more direct DfE incentives than take-back mandates where fees are simply based on market-share.

2.2 Environmental effectiveness of EPR

An important question of any policy intervention is whether or not the policy achieves its stated objectives. In the case of EPR, a key objective is the reduction of waste and increased recycling activities. This question of environmental effectiveness is to be distinguished from the question of whether or not the policy achieved its goals at least cost (cost-effectiveness, discussed below). Not surprisingly, the general consensus in the academic and non-academic economic literature is that mandates such as take-back requirements and recycling content standards do increase the amount of material recycled and the recycling rate (see, for example, an evaluation of Japan's WEEE Recycling Act in Tasaki et al. (2007)). Recent work also suggests that market-based policies can also be effective. Viscusi et al. (2011) show that plastic water bottle recycling increases with deposit/refund bottle bills. Similarly, Batson and Eggert (2012) find that increasing bottle-bill deposit/refund rates per container increases recycling rates in the U.S. They also note the substantial difference in recycling rates between states without a deposit/refund (roughly 30%) and those with a deposit/refund (roughly 70%), regardless of the deposit rate. Thus, one would conclude that, in general, EPR policies can accomplish waste reductions and increase recycling rates.

One important note regarding material effectiveness it that focusing on recycling rates may obscure important quality dimensions in terms of the recycling stream, starting from the sorting of materials by end

[25] While conceptually easy to model, in reality recyclability is a complex mix of product attributes which may or may not be perfectly observable or measureable by regulators.

[26] This could be individual, firm-specific take-back requirements, or fees/subsidies that vary with the difficult-to-observe "recyclability" characteristic. The additional costs associated with product-characteristic targeting would need to be weighed against the additional DfE incentives it provides.

[27] Hosoda (2004) provides a number of qualitative examples of producers altering design of container and packaging, as well as household electronic appliances, in response to EPR policies in Japan. A more quantitative approach is undertaken in Nicolli et al. (2012) who examine patent activity for end-of-life vehicles and plastic packaging and find that national regulations and voluntary agreements (of which EPR policies are a subset) stimulated technological innovation in those industries.

[28] For example, a deposit/refund per kilogram of aluminium as opposed to a deposit/refund per aluminium can.

users. One advantage of EPR policies that involve producers in the recycling process is that products may be recycled in ways that encourage their reprocessing into new products. In that sense, it engages producers in the broader efforts led by the OECD on Sustainable Materials Management (SMM) by encouraging producers to improve the lifecycle efficiency of their products and materials. In addition, growing resource scarcity and rising commodity prices encourage producers to find new ways to recover used products and to turn waste into a resource. By contrast, non-EPR policies such as commingled, curb side recycling may lead to lower quality recycling streams due to commingling and breakage of products. For example, Acuff (2013) empirically examines diversion and recycling rates from single stream versus sorted recycling at municipality-run, curb side-supplied U.S. recycling centres. She finds that while collection methods like single stream recycling have positive aspects such as increased diversion rates (more material sent from households to recycling centres), they also increase contamination rates and the amount of residual material ultimately diverted to landfills by the recycling centres.

Thus, while it seems clear that EPR policies can achieve environmental goals (reduced waste and increased recycling), more systematic empirical evidence is needed.[29] In particular, empirical assessments that carefully control for confounding factors and are able to credibly identify the effects of EPR policies are needed.[30] See below in Section 2.3 for discussion of potential empirical methods that could be utilized to better establish the environmental effectiveness of various EPR policies.

2.3 Cost effectiveness of EPR

Turning now to the economic effectiveness of policies, there are two important questions that can be asked of EPR policies: Do they achieve their objectives in the least-cost manner, and do the benefits of EPR policies outweigh the costs? A related question would be to ask: What is the optimal level of EPR policies? Several studies below attempt to address this question as well.

Although cost-benefit analysis (CBA) continues to grow as a body of economic literature, there is relatively little work on EPR policies specifically, particularly on non-market-based EPR policies such as take-back and recycling content standards. From an analytical perspective, Smith (2005) provides an excellent, thorough framework for evaluating costs and benefits of EPR policies. Costs and benefits to be considered include operating costs, environmental benefits from reduced externalities, and other side effects that might prove difficult to quantify (i.e. competition effects). Operating costs may change with the adoption of an EPR policy for two reasons: First, for a given level of waste and recycling activities, operation costs may vary by collection method (municipality versus PRO for example). Second, as the goal of EPR policies is to alter waste and recycling decisions, operating costs may vary with the level of waste and recycling activity—for example the costs associated with increasing recycling rates. These changes in costs should then be compared against the benefits associated with reductions in externalities (waste, production by-products, virgin material use – see Kinnaman (2006) for further discussion on valuation of externalities associated with waste disposal and recycling) discussed in Section 1.

Smith notes that a major challenge of evaluating EPR policies is the heterogeneity of EPR programs themselves. Furthermore, establishing a counterfactual baseline for comparison will be important to draw causal inferences—one needs to carefully establish what recycling and waste activities would have looked

[29] While the concept of "Sustainable Materials Management" has not been considered in the economics literature, the life-cycle approach at the heart of SMM would prove useful in analysing the overall environmental effects of EPR policies.

[30] For example, as Batson and Eggert (2012) note, simply comparing rates between states with and without deposit/refund may overstate the effect of deposit/refund, as states more inclined to recycle in the absence of a policy may be more inclined to adopt the policy in the first place. Similarly, comparisons of waste disposal and recycling over time in a single country or state need to credibly establish counterfactuals of what waste disposal and recycling would have looked like "but for" the policy.

like "but for" the adoption of the EPR policy. Note that many econometric tools (e.g. instrumental variables, regression discontinuity design, difference-in-difference estimators, matching estimators) have been developed and applied to a broad spectrum of environmental and other policies to carefully identify the effect of policy interventions.

Previous research provides some empirical insight into the costs and benefits of market-based EPR policies, such as deposit/refund and ADF. Palmer et al. (1997) investigate the cost effectiveness of alternative policies for waste disposal, providing a calibrated simulation model of 1990 U.S. markets for paper, glass, aluminium, steel, and plastics. They consider two EPR policies (deposit/refund and an advanced disposal fee) as well as the non-EPR policy of recycling subsidies (modelled as a simple subsidy to households per ton of material recycled). They find that for a given target of waste reduction, the deposit/refund is the least cost policy followed by the advanced disposal fee, both of which were calculated to perform better than the non-EPR recycling subsidies. Comparing the marginal costs of waste reduction against the social benefits, they conclude that modest increases in waste reduction would be efficient.[31] It is important to note that, because private recycling activities exist independent of policy interventions, imposition of polices can also affect these private recycling activities. Kaffine (2014) shows that the costs of the policies examined in Palmer et al. (1997) substantially depend on how private recycling markets (specifically scrap prices) respond to policy interventions. For small, open economies such as U.S. states or small, integrated EU countries, the costs of achieving given waste reductions with a deposit/refund will be lower, thereby justifying more aggressive waste reduction targets.[32]

In addition to benefits from waste reduction due to EPR policies, as Smith (2005) notes, other benefits should be considered as well. When upstream reductions in externalities from changes in waste disposal and recycling activities are considered (specifically CO_2 emissions), Acuff and Kaffine (2013) show that effective policy costs for a given waste disposal reduction are significantly lowered. In addition, they show that the advanced disposal fee may actually have lower net costs per ton of avoided waste disposal than the deposit/refund, due to the larger effect of advanced disposal fees in reducing upstream externalities.[33] A less-obvious source of benefits of EPR policies is found by Ashenmiller (2010) who finds that income provided by bottle bill refunds may generate positive externalities by reducing petty crime rates, by providing an easily accessible source of revenue for low-skilled people.

In any consideration of cost effectiveness or cost-benefit analysis of EPR policies, it is important to properly measure benefits and costs. For example, Kinnaman (2006) notes that many costs associated with disposal are actually internalized by the landfill operator (i.e. tipping fees). As such, the true externalities of waste disposal stem from the smaller external costs including odour, visibility, and other disamenities to the local population affected by the presence of landfills. One implication of this point is that it is not clear that shifting the financial burden of waste management from local municipalities to producers is a benefit unto itself—benefits would only exist if operating costs were less for producers than local municipalities. An additional implication of the fact that external costs of waste disposal may be rather small is that transaction costs associated with the administration of the policy may quickly diminish the net benefits of policy intervention. While such costs are notoriously difficult to model and measure precisely, one would

[31] Waste reductions beyond the efficient level would still pass a benefit-cost test, up to a point. Just as a negative externality reduces social welfare by forcing society to accept the external costs of environmental damage, policy interventions that exceed the efficient level can also reduce social welfare on the margin and potentially in total. In short, passing a cost-benefit analysis only requires that the total benefits exceed the total costs, while an efficient policy is one where the marginal social costs equal the marginal social benefits.

[32] A small, open economy in this context is one where scrap prices are determined by world markets, such that modest policy changes in the small, open economy do not affect world prices.

[33] This was only true for small waste reduction targets, and will depend on the empirical context. For example, CO_2 emissions from EU countries are already internalized by the EU-ETS carbon trading program.

anticipate that administration costs would tend to rise with the complexity of the policy. Thus, while complex policy instruments may have appealing theoretical properties, one must balance the social welfare gains against any increased administration costs. Further empirical research into this area would be beneficial.

The literature above provides some insight into the cost-effectiveness of market-based policies, generally confirming the theoretical insights from section 2.1.1 that multi-instrument policies such as deposit/refund are better than single instrument policies such as ADF. However, there is clearly a gap in the academic literature regarding the cost-effectiveness of policies such as take-back and recycling content standards. One aspect that deserves further inspection is the general equilibrium costs of EPR policies. While Smith (2005) provides some guidance in terms of looking at operating costs of waste management, EPR policies also likely create other economic costs or "deadweight loss" (in the form of lost consumer surplus or lost producer surplus) in related markets that should be investigated more fully.[34]

2.4 Free riders, orphan products and trade issues

Finally, there have been concerns raised regarding free riding behaviour by firms, orphan products, and trade considerations with EPR policies. To date there is a relatively limited economic literature specifically on these issues as they pertain to EPR. However there is a much larger literature on these issues more broadly in other contexts, which may provide some insight into these issues for EPR.

Free riding is a common problem for public goods, defined as goods that are non-rival and non-excludable.[35] In such cases, agents benefit from a public good without sufficiently contributing to its financing, maintenance, or improvement, resulting in private under-provision of that public good. Solutions to free riding are typically institutional or regulatory, to the extent such a policy is sufficiently enforceable to ensure compliance. As would be expected, this would be easier in small sectors, but more difficult in large sectors with many producers. It may also be easier for market-based EPR policies that can tap into existing tax and regulatory structures.[36] Further research into how both PROs and governmental authorities address free riding would be beneficial.

Orphan products are products created by producers who are no longer in business, leaving others to handle end-of-life responsibilities for the product. A relatively close analogy to this problem exists in extractive industries. In the U.S., extractive firms such as natural gas companies are required to post an environmental bond in escrow when extraction begins. If the firm cleans up at the end of extraction activities, they receive their bond back (with interest). But if the firm has gone bankrupt, exited the industry, or otherwise failed to clean up their extraction damages, the regulator uses the revenue in the bond pool for clean-up. The optimal bond would then be set equal to the cost of clean-up times the probability that firms fail to clean-up their damages. In an EPR context, firms could be required to

[34] For example, consumer surplus may be lost due to policies that lead to lower consumption at higher prices, and producer surplus may be lost by virgin material extractors who sell less of their product at a lower price. Focusing on changes in consumer and producer surplus can also help avoid confusing financial or accounting costs (for example, fees paid to PROs) with economic costs (deadweight loss). Such an examination would also provide insight into the question of *incidence*, or who ultimately pays the economic costs of the EPR policy. While producers may be legally responsibility for the fate of their products, who faces the economic burden of the policy will depend on supply and demand elasticities in the various markets.

[35] Non-rival means that one agent's consumption of the good does not affect the consumption of another. Non-excludable goods are those where it is impossible to prevent others from consuming the good.

[36] A small note on terminology is in order. The 2001 OECD Guidance Manual lists a number of examples of "free-riding" behaviour in EPR systems; however, many of the examples listed involve producers not fulfilling their obligations under EPR (under paying fees, illegal disposal, etc.). Such activities may be better described as *non-compliance*, as opposed to free-riding on the provision of a public good

annually post an orphan product bond, with the bond returned with interest at the end of the year if the firm is still in business and fulfilling its end-of-life management responsibilities. If the firm goes bankrupt and orphan products are created, the bond pool revenue can then be used to finance end-of-life product management.[37] For longer-lived products, the bond would be posted when the product is placed on the market (similar to the extraction analogy) and only returned if the firm is still operating when the product reaches end-of-life.

From an economic perspective, one concern regarding EPR policies is that they may create distortions in international trade markets. If EPR raises the costs for domestic firms relative to foreign firms, they may be at a disadvantage in world markets. Trade theory can shed some light on this issue. Focusing on just the external costs of waste disposal and ignoring recycling activities, if the optimal policy would be a consumption tax (i.e. a household unit-based disposal charge), a domestic production tax such as an advanced disposal fee combined with an import tariff of equal magnitude would be equivalent to the optimal consumption tax. If recycling activities are also considered, then both domestic and importing firms should pay a deposit charge and both domestic and importing firms should receive a refund for their products when recycled.[38] Similarly, take-back requirements and recycling standards applied equally across both domestic and importing firms should not create economic distortions, though they may be more prone to legal challenges.[39]

In sum, the academic economic literature on EPR over the past decade provides a number of insights that policymakers should take into consideration, with the caveat that more empirical research is needed to supplement the heavily theoretical literature. First, in selecting policies within the EPR framework, deposit/refund, UCTS, and take-back requirements, eventually with tradable permits, are likely to be more efficient than single instrument policies such as an ADF, virgin materials tax, or recycling content standards. Second, while collective PROs may be attractive in terms of taking advantage of economies of scale and reducing the need to monitor individual firms, care should be taken that market power is not exploited. Third, while most EPR policies provide DfE incentives, policies that directly target product characteristics (weight, recyclability, etc.) will provide the most direct incentives. Finally, though there is some evidence that EPR policies can achieve their environmental goals, empirically it is still an open question which policies will achieve those goals at the least cost. As noted above, carefully measuring the effectiveness of policies and the costs associated with them is a key challenge in determining the cost effectiveness of EPR policies.

3. EPR in practice

In this section we provide an overview of EPR policies in practice.[40] While not intended to be completely comprehensive, the list of 395 EPR policies in Table 1 (see Annex) provides some insights into a) the types of products covered, b) the types of EPR policies typically used, c) trends in EPR implementation, and d) regional trends. Note that with respect to the individual policies, there is enormous

[37] Of course, other revenue sources such as an advanced disposal fee or general taxation could be used to finance end-of-life management of orphan product, but requiring firms to post an in-advance bond for their products is more in the EPR spirit of holding producers responsible for their products. Also, firms who do not create orphan products are also no-worse-off under this mechanism, as they receive their bond with interest at the end of the year. Only those firms which create orphan products face the consequences of losing their bond.

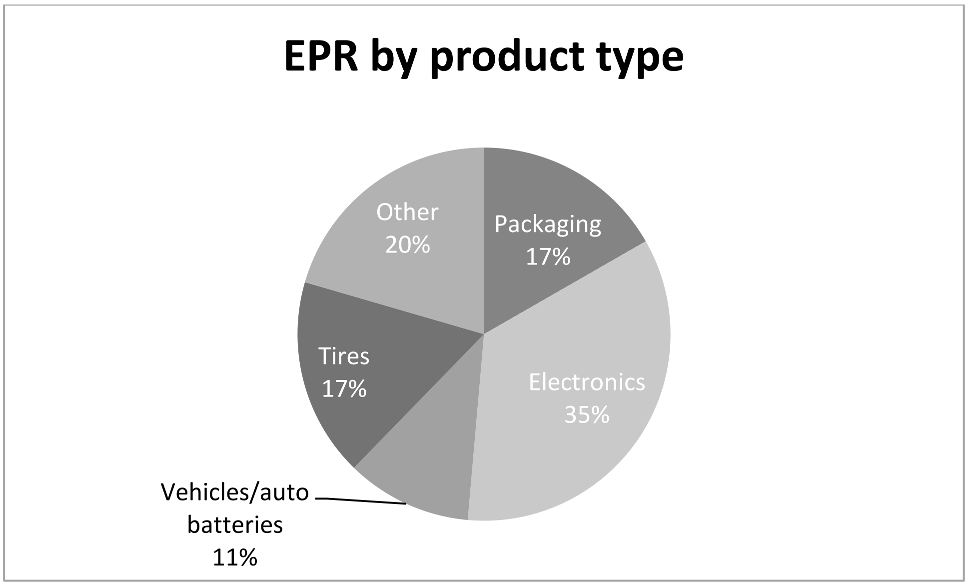
[38] In addition to minimizing distortions, such a policy would also be consistent with WTO rules.

[39] For example, if the costs of certifying that a certain percentage of content is recycled material varies substantially between domestic and foreign firms.

[40] A wide range of sources were utilized for this overview, including legal documents, PRO websites, Watkins et al. (2012), The Product Stewardship Institute, Environment Canada's program inventory, and other web-based sources.

variation in the details and specifics, particularly with take-back programs.[41] EPR policies have also been updated over time, with the date in the table representing the initial implementation data.[42] Furthermore, some policies may contain aspects of multiple EPR policies. The purpose of the table is to match the existing policy as close as possible to one of the six EPR policies discussed in the introduction. The following discussion provides figures and discussion of summary statistics of the information in Table 1 (see Annex).

Figure 2. EPR by product type



Note: “Electronics” includes mobile phones, renewable batteries, thermostats and automobile switches; “packaging” includes beverage containers; “other” includes used oil, paint, pesticides and chemicals, appliances and other less common products. Source: Authors

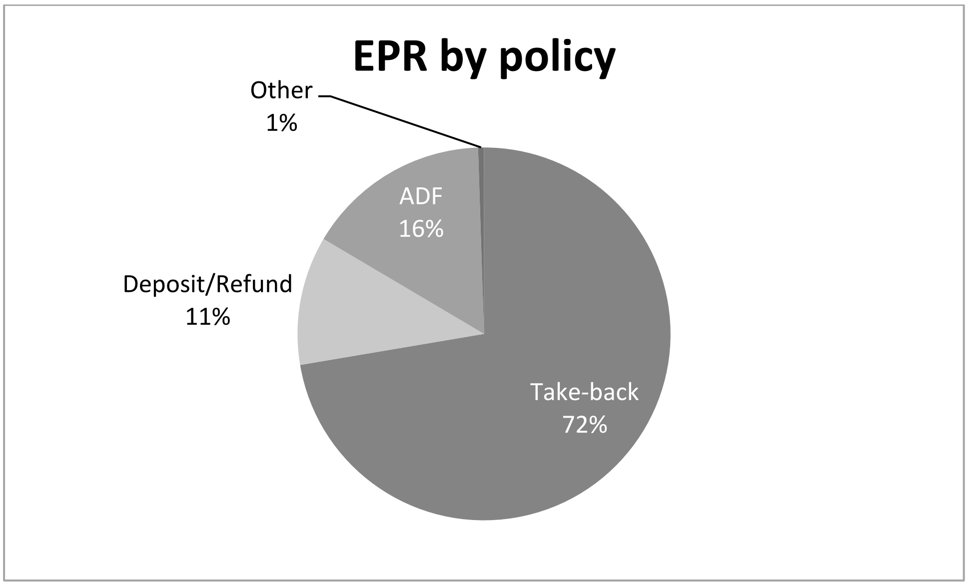
In terms of products frequently covered under EPR, small consumer electronics appear to be the most prevalent (see Figure 2). Including mobile phones, renewable batteries, thermostats and auto switches, this accounts for 35% of all EPR policies globally. Packaging (including beverage containers) (17%), tires (17%) and vehicles/lead-acid batteries (7%/4%) are the next largest groups of products covered.[43] The remaining 20% of polices cover less common products including used oil, paint, chemicals, large appliances, and florescent light bulbs. The prevalence of EPR policies covering electronics, phones, tires, rechargeable batteries, thermostats and switches is consistent with the literature in Section 2 in that they are products with potentially large social costs of disposal. They are also products with a relatively large level of consumption, which may explain the prevalence of EPR policies for them relative to other products with large social costs such as used oil, paint and chemicals.

[41] In particular, take-back policies varied in terms of collective versus individual requirements (with collective being the most common), voluntary versus mandatory requirements (there is a fair “grey” area between voluntary and mandatory, but mandatory appears to be the most common), and fee structure (for example, the tradable package notes in the UK). Further research into the specific institutional set-up of various take-back policies would be valuable.

[42] In some cases, it was difficult to precisely determine implementation date, and passage date or effective date was used instead.

[43] Vehicles and auto batteries are included together because end-of-life vehicle recovery in the EU includes auto batteries.

Figure 3. EPR by policy

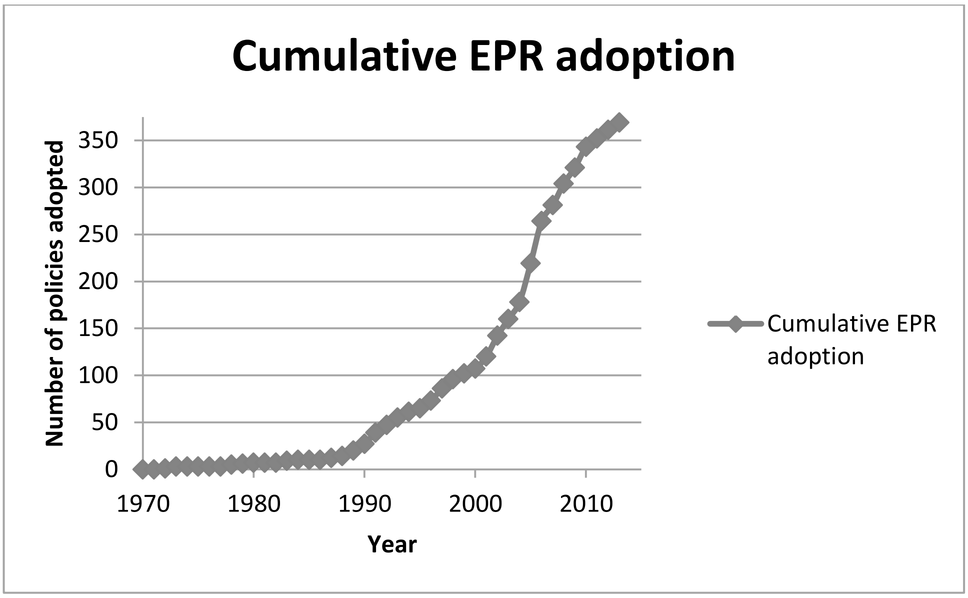


Note: "Other" includes UCTS and recycling content standards. Source: Authors

Looking at the policies put in place (Figure 3), various forms of take-back requirements are the most commonly used EPR policy (72% globally), and they are used for a large variety of products. Advance disposal fees are used the next most frequently (16%) and also have a wide range of product applications. Deposit/refund is the next most common policy (11%), though it is concentrated in the used beverage container and lead-acid battery markets. The final three EPR policy instruments (upstream combined tax/subsidy, recycling content standards, and virgin material taxes) are sparsely used, if at all.[44] On the one hand, the prevalence of take-back and deposit/refund policies is consistent with findings in the academic literature reviewed in Section 2 in that they are likely more efficient than other EPR polices such as virgin material taxes and recycling content standards. On the other hand, the popularity of the ADF suggests that it may have advantages in terms of ease of implementation, despite its theoretical inferiority to deposit/refund.

[44] The breakdown for the materials in Figure 1 by policy type is as follows: Electronics are 95% take-back, 4% ADF, and 1% deposit/refund; packaging is 48% take-back, 12% ADF, and 39% deposit/refund; tires are 41% take-back, 57% ADF, and 1% deposit/refund; vehicles/auto batteries are 49% take-back, 21% ADF, and 30% deposit/refund (policies specifically for auto batteries tend to be deposit/refund, while policies for the entire vehicle are a mix of take-back and ADF).

Figure 4. Cumulative EPR policy adoption over time



Source: Authors

Examining the implementation dates, we see a marked increase in EPR adoption over the last decade (Figure 4). Of the 369 EPR policies included in Table 1 with dates, over 70% were implemented since 2001, when the OECD Guidance Manual was initially published. Breaking it down by decade, 5% of policies were instituted prior to 1990, 22% were instituted between 1990 and 1999, 59% were instituted between 2000 and 2009, and 13% were instituted from 2010 to the present. Of the types of policies adopted within the last decade, take-back policies appear the most popular, while many of the deposit/refund and ADF policies were implemented in past decades.[45] The overall increase in EPR policies is consistent with a general increase in interest in waste management and EPR in particular over the last few decades.

Comparing internationally, the United States appears slightly more inclined to adopt market-based policies such as deposit/refund and ADF (slightly less than half of US policies) relative to the rest of the world (79% take-back, 21% market based-policies), though there are many examples of take-back within the US and deposit/refund and ADF policies outside the US.[46] In terms of products covered, there is some variation across regions, but electronics is the most common product under EPR. Within the EU 34% of policies cover electronics, 18% cover packaging, 14% cover tires, and 20% cover vehicles/auto batteries, while in the US, 50% of policies cover electronics, 8% cover packaging, 24% cover tires, and 7% cover vehicles/auto batteries. Comparing across states within the US, EPR policy adoption is correlated with proxies for environmental attitudes ("greenness"), with greater adoption in states such as California and Vermont (and much of the Northeast).[47] This suggests that the environmental attitudes of citizens may be a

[45] This is of course not universally true, as there are a few cases of take-back policies implemented in the late 1980's and early 1990's, and a several cases of deposit/refund or ADF implemented recently. Specifically, the policy mix for policies implemented prior to 1990 is 10% take-back, 70% deposit/refund, and 20% ADF; for 1990-1999 it was 50% take-back, 20% deposit/refund, and 30% ADF; for 2000-2009 it was 79% take-back, 6% deposit/refund, and 15% ADF; and for post 2010, it was 92% take-back, 2% deposit/refund, and 2% ADF.

[46] Within the EU, 87% of EPR policies are take-back, with ADF at 12% of EPR policies and deposit/refund at only 1% of EPR policies.

[47] The number of EPR policies for each state was regressed against a variety of state-level proxies for environmental attitudes, including attitudes in 1974-1998 towards increased government spending on environmental protection (Brace et al.

strong driver of EPR adoption, though further work on determinants of EPR adoption would be necessary to disentangle "greenness" from other potential factors such as income.

4. Conclusions

The previous sections have provided an overview of the economic literature on EPR as well as EPR programs in practice. The use of EPR has increased exponentially over the past 20 years, with about than three-quarters of EPR policies implemented over the past 10 years. Most of these focus on electric and electronic equipment, packaging materials, tires and vehicles/lead-acid batteries. The vast majority of EPRs use take-back requirements as their key policy instrument.

Key insights from the literature that policymakers may want to take into consideration are as follows. First, in selecting policies within the EPR framework, deposit/refund, UCTS, and take-back requirements with tradable permits are likely to be more efficient than single instrument policies such as an ADF, virgin materials tax, or recycling content standards. Second, while collective PROs may be attractive in terms of taking advantage of economies of scale and reducing the need to monitor individual firms, care should be taken that market power is not exploited. Third, while most EPR policies provide DfE incentives, policies that directly target product characteristics (weight, recyclability, etc.) will provide the most direct incentives. Finally, though there is evidence that EPR policies can achieve their environmental goals, empirically it is still an open question which policies will achieve those goals at the least cost.

However, more work remains to be done in a number of areas. In particular, there is a lack of careful empirical research on the effects of many types of EPR policies, particularly non-market based policies such as take-back and recycling content standards. More generally, while the findings from the academic economic literature provide insight into a number of important questions, the existing studies in the literature are heavily tilted towards theoretical and conceptual analysis. But there are a number of important questions that may be better answered via context-specific case studies (see annex 1 for an outline of the key quantitative and qualitative information and questions that case studies need to address).

2005), attitudes in 1985-1987 towards the environment (Johnson et al. 2002), a 2007 Forbes study of the "greenest" states http://www.forbes.com/2007/10/16/environment-energy-vermont-biz-beltway-cx_bw_mm_1017greenstates.html and 2008 vote shares for the Democratic Party. All proxies were positively correlated with the number of EPR policies adopted, with statistically significant correlations for attitudes towards environmental spending ($p = 0.04$), Forbes' greenness ($p < 0.01$) and Democratic vote share ($p < 0.01$). Note these should be treated as pure correlations.

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ANNEX 1: CASE STUDY GUIDANCE

The following sections lay out key questions and quantitative and qualitative information that should be provided in a series of case studies that will be developed to provide empirical material for an up-date of the 2001 OECD EPR Guidance Manual. The information items below build on the insights from the above literature review as well as on discussions with the OECD Secretariat and the OECD Expert Group on EPR. While the set of information items was primarily developed to collect information about take-back systems, the most frequently used type of EPR (72%), many questions are also relevant for EPRs that are set-up around advance disposal fees or deposit/refund systems.

1 Legal aspects

- A summary of the applicable legislation regarding the definition of key concepts, the role and respective obligations of manufacturers and importers, distributers(retailers), municipalities, citizens/consumers, and other interested parties in the value chain (downstream and upstream), and the definition of the financial obligations;

- A summary of the objectives and targets assigned by law and/or independent and proactive initiative to the producers (and/or retailers), how producers (and/or retailers) will comply with the obligations, and the sanctions foreseen in case of non-compliance (see Section 2.4 for discussion of non-compliance) The definition of targets such as recovery rate and collection rate – e.g., whether recycling rate includes energy recovery, what is the denominator of the collection rate - should be clearly explained;

- If take-back polices are in place, how are contributions to be paid by producers determined (e.g., import/manufacturing volumes, possible thresholds determining liability etc.)? What is their level, the manner of collection and relationship with recyclability and true cost? (see Section 2.1.3 for discussion of recyclability);

- In the case that an ADF/ARF (Advanced Disposal/Recycling Fee) is levied, what is the level of the fee and who manages the collected ADF? Is the collected ADF/ARF itself taxed?

- What is done to ensure that procurement related to collection, sorting and recycling is achieved in a transparent and non-discriminatory way? What is done to ensure a level playing field in the market? (see Section 2.1.2 for discussion of market power in PROs, and Section 2.4 for discussion of trade issues);

- How is dialogue organized between the involved entities (producers, national/regional authorities, municipalities, private waste collectors, sorters and recyclers)?

- What efforts are being developed as part of the EPR to inform and educate consumers?

- What are the possible supporting measures taken at national/regional/local level to support producer responsibility objectives (pay as you throw schemes, landfill/incineration taxes, mandated separate collection, etc.)? Are there standards, codes of practice or other guidance that have been used/considered to guide the environmental, health and safety and other aspects of the EPR activity? Are those supporting measures compliments or substitutes to EPR? (see section

2.1.1 for discussion of how landfill taxes may actually weaken environmental performance take-back mandates with tradable credits).

- Are there obligations for PROs to accept products from other producers, particularly smaller producers unable to construct their own collection systems?

2 Governance of the system

- A description of the governmental oversight over the EPR system. Is there a public oversight and if yes, what type of institution is tasked with this role and what are the means at its disposal? What is the allocation and balance of authorities between the producers and government? How many people are involved? What is the level of enforcement executed? Is there a public certification/accreditation for the producer responsibility organization, and what are the criteria? Was a clearing house established? What are its tasks? What is its legal status? What is done to ensure that the funds collected through the EPR are used appropriately, that targets are met and that the problem of free riders is addressed? Are there fees/contributions levels, or is collection subject to the approval by public authorities? What type of quality control systems are in place at each step of the value chain (including collection, sorting, recycling, and exports)? What is done to enforce the EPR law to ensure a level playing field and that fair competition is guaranteed?

- A stakeholder analysis of the EPR environment, including their interests and expectations, the legal status of the producer responsibility organization (private, not for profit, public-private partnership etc.), the position of the producers (and other stakeholders) in the producer organizations, and the position of the government in the producer organizations.

- How is the transparency of the EPR system ensured? Is data and information on the performance of the system easily and publicly accessible? What are the legal requirements to ensure transparency of the scheme? How are the producer declarations about the products that are put on the market verified and by whom? What additional monitoring is put in place by the producer organizations (auditing, control of exports, etc.)? What requirements are foreseen for audits and auditors?

3 Environmental effectiveness

- What is the current state of policy implementation (qualitative description)?

- What has been improved by applying EPR? What issues remain unsolved? What issues have newly emerged after applying EPR (qualitative description)?

- What are the collection amount and/or rates achieved and how do they compare to the targets that were set for the scheme? (See section 2.2 and 2.3 for discussion of assessing environmental effectiveness.[48])

- What are the impacts on prevention of waste, natural resource use, and on design for the environment? (see section 2.1.3 for discussion on design for the environment)

[48] As Smith (2005) notes, the key element in this assessment is establishing the counterfactual—what would waste, recycling and other externalities have looked like in the absence of the policy? As discussed above, there are econometric techniques designed to carefully address similar issues in other contexts which could be applied to the effects of EPR policies.

- Have the actors in the value chain, such as the consumers and national/local governments, assumed their environmental responsibilities?

- What has been the role of other existing policies and independent and proactive initiative by the actors in the value chain in generating the observed environmental results (i.e., other policies and independent and proactive initiative that provide economic actors with incentives to improve collection and recycling rates, for instance landfill taxes, information campaigns, support for research and development, introduction of PAYT schemes, deregulation, and independent and proactive initiative by industry)?

4 Coverage and quality of waste collection and treatment

- A description of the organization of separate waste collection schemes, as well as the level of standardization of sorting and collection methods. A description of the systems that have been set-up to monitor whether waste that is collected and sorted is effectively treated in the appropriate facilities;

- What is the quality of the collection, sorting and recycling operations (residue rates)? What proportion of the territory/population is covered by the EPR system?

- A description of the removal and treatment of hazardous substances and those requiring proper treatment. How has it improved since the introduction of EPR?

- An assessment of the proportion of waste/products that are exported.

5 Cost effectiveness

- A summary of information about the use of the funds collected, structured into different categories (expenses for information and public awareness, for collection/sorting/ recycling/incineration/disposal, general expenses, reserves, etc.), as well as the main revenues (contribution of producer with breakdown per relevant category, sales of the materials, etc.);

- What is the overhead cost of the system? What are their financial liabilities in relation to their obligations? What is the economic/financial sustainability of the schemes? What is their strategy to ensure coverage of the financial liabilities of the producers?

- An analysis of overall costs and benefits of the EPR scheme:

  – What are the marginal external costs of waste disposal? Are there other unregulated emissions or other externalities either in the production stage or the extraction stage affected by the policy?[49]

  – What are the total social benefits of EPR from avoided waste disposal and other externalities?[50] (see section 2.3 for discussion of benefits of EPR policies);

[49] Note that reductions in *regulated* emissions (such as carbon emissions in EU countries) should not be included as a benefit of EPR, assuming the level of regulation is reasonably close to optimal.

[50] The social benefits of the EPR policy can then be approximated by taking the marginal external damage of the relevant externalities times the change in the levels of the externalities. In other words, if the marginal external damage of waste disposal is 8 USD per ton, and the EPR policy reduced waste disposal by 1 million tons annually, then the benefit of the EPR policy (purely in terms of waste reduction) would be 8 million USD annually.

- What other benefits or costs (besides the environmental ones) have been anticipated or achieved (e.g., jobs, economic growth, contributions to other policies goals such as climate change mitigation for example, etc.)?

- What are the economic costs of the EPR policy?[51] (See section 2.3 for discussion of assessing the costs of EPR policies).

- Has the EPR generated net benefits and if so, has there been a comparative assessment of the costs and benefits of alternative policy options that identified EPR as the most efficient option?

6 Competition and market barriers

- A summary of the details, impacts or characteristics of competition within the EPR and/or between EPR schemes. If possible, it will also provide information about the trade-off that exists between economies of scale and the establishment of competition between different Producer Responsibility Organisations. (See section 2.1.2 for discussion of market power in collective PROs);

- What is the state of the domestic recycling industry? Is recycling capacity a barrier to increased recycling?

- Are there issues with market access and competition for producers that may result from the EPR? Have these issues been addressed in the scheme?

[51] It may be tempting to conclude that if a PRO collects 2 million USD in fees, then that represents the cost of the policy. However, this is misleading, as this represents a transfer between producers and the PRO. The true costs to society are reflected in the changes in *behaviour* by all of the agents in the model (virgin material producers, consumers, waste managers, etc.) in response to the policy. A proper assessment of alternative EPR policy costs would require careful considerations of the various margins of adjustment and interlinkages between all agents involved in the life-cycle of the product. While this is a daunting task, our understanding of the costs of EPR policies would be enhanced greatly by such an undertaking.

ANNEX 2: TABLE 1. BROAD SAMPLE OF EPR POLICIES

Location	Material stream	Policy instrument	Instituted
Australia – North. Terr.	Beverage Containers	Deposit/Refund	2011
Australia - South Australia	Beverage Containers	Deposit/Refund	1993
Australia - National	Used Oil	UCTS	2001
Australia - National	Televisions/Electronics	Take-back	2012
Australia - National	Mercury Bulbs	Take-back	2010
Australia – National	Packaging	Take-back	2010
Australia - National	Ag/Vet Chemicals	Take-back	1993
Australia - National	Mobile Phones	Take-back	1999
Austria - National	Batteries	Take-back	2006
Austria – National	Electronics	Take-back	2005
Austria - National	Packaging	Take-back	1993
Austria - National	Vehicles	Take-back	2002
Austria – National	Tires	Take-back	2002
Austria - National	Used Oil	Take-back	
Austria – National	Medicine	Take-back	
Belgium - National	Batteries	Take-back	2006
Belgium - National	Electronics	Take-back	2005
Belgium - National	Packaging	Take-back	1997
Belgium – National	Vehicles	Take-back	1999
Belgium – National	Tires	Take-back	1998
Belgium – National	Used Oil	Take-back	2003
Belgium – National	Medicine	Take-back	
Belgium - National	Batteries/Capacitors	Take-back	1995
Brazil - National	Tires	Take-back	2002
Bulgaria – National	Batteries	Take-back	2008
Bulgaria – National	Electronics	Take-back	2006
Bulgaria – National	Packaging	Take-back	2004
Bulgaria – National	Vehicles	Take-back	2002
Bulgaria - National	Tires	Take-back	
Bulgaria - National	Used Oil	Take-back	2006
Canada – National	Mobile Devices	Take-back	2008
Canada - National	Rechargeable Batteries	Take-back	2000
Canada - National	Mercury Auto Switches	Take-back	2008
Canada - National	Pesticides	Take-back	2010
Canada - National	Pesticide containers	Take-back	1989
Canada - National	Refrigerants	Take-back	2000

Table 1 (continued from previous page)

Location	Material stream	Policy instrument	Instituted
Canada - Alberta	Beverage containers	Deposit/Refund	1997
Canada - Alberta	Electronics	ADF	2010
Canada - Alberta	Paint	ADF	2008
Canada - Alberta	Medicine	Take-back	2011
Canada - Alberta	Used Oil	Deposit/Refund	2007
Canada - Alberta	Tires	ADF	1992
Canada - British Columbia	Beverage Containers	Deposit/Refund	2004
Canada - British Columbia	E-waste	Take-back	2007
Canada - British Columbia	Auto Batteries	Take-back	1991
Canada - British Columbia	Milk Containers	Take-back	
Canada - British Columbia	Medicine	Take-back	1999
Canada - British Columbia	Solvents	Take-back	2004
Canada - British Columbia	Used Oil	Take-back	2003
Canada - British Columbia	Tires	Take-back	2007
Canada - Manitoba	Auto Batteries	Take-back	2011
Canada - Manitoba	Packaging	ADF	2008
Canada - Manitoba	Used Oil	Take-back	1997
Canada - Manitoba	Tires	Take-back	2006
Canada - New Brunswick	Beverage Containers	Deposit/Refund	1999
Canada - New Brunswick	Paint	Take-back	2009
Canada - New Brunswick	Tires	Take-back	2008
Canada - Newfoundland	Beverage Containers	Deposit/Refund	1997
Canada - Newfoundland	Used Oil	Take-back	2003
Canada - Newfoundland	Tires	ADF	2002
Canada – Northwest Terr.	Beverage Containers	Deposit/Refund	2005
Canada – Nova Scotia	Beverage Containers	Deposit/Refund	1996
Canada – Nova Scotia	Electronics	Take-back	2008
Canada – Nova Scotia	Milk Containers	Take-back	
Canada – Nova Scotia	Paint	Take-back	2002
Canada – Nova Scotia	Medicine	Take-back	1995
Canada – Nova Scotia	Sharps	Take-back	
Canada – Nova Scotia	Used Oil	Take-back	1996
Canada – Nova Scotia	Tires	ADF	1997
Canada – Ontario	Beer Containers	Take-back	2002
Canada – Ontario	Electronics	Take-back	2009
Canada – Ontario	Household Haz. Waste	Take-back	2006
Canada – Ontario	Mercury Bulbs	Take-back	2010
Canada – Ontario	Thermostats	Take-back	2006
Canada – Ontario	Packaging	ADF	2002
Canada – Ontario	Tires	Take-back	2009
Canada – Prince Edward I.	Beverage Containers	Deposit/Refund	2008

Table 1 (continued from previous page)

Location	Material stream	Policy instrument	Instituted
Canada – Prince Edward I.	Electronics	Take-back	2010
Canada – Prince Edward I.	Auto Batteries	Deposit/Refund	2009
Canada – Prince Edward I.	Medicine	Take-back	2004
Canada – Prince Edward I.	Used Oil	Take-back	
Canada – Prince Edward I.	Tires	ADF	1991
Canada – Quebec	Beverage Containers	Deposit/Refund	1984
Canada – Quebec	Packaging	Take-back	2010
Canada – Quebec	Paint	Take-back	2001
Canada – Quebec	Medicine	Take-back	
Canada – Quebec	Used Oil	Take-back	2004
Canada – Quebec	Tires	ADF	2009
Canada – Saskatchewan	Beverage Containers	Deposit/Refund	1973
Canada – Saskatchewan	Electronics	Take-back	2007
Canada – Saskatchewan	Milk Containers	Take-back	
Canada – Saskatchewan	Paint	Take-back	
Canada – Saskatchewan	Medicine	Take-back	1997
Canada – Saskatchewan	Used Oil	Take-back	1996
Canada – Saskatchewan	Tires	Take-back	1998
Canada – Yukon	Beverage Containers	Deposit/Refund	1992
Canada – Yukon	Tires	ADF	2003
Chile – National	Beverage Containers	Deposit/Refund	1998
Chile – Regional	Tires	Take-back	2004
Chile – National	Used Oil	Take-back	2007
Chile – Regional	Auto Batteries	Take-back	2013
Chile – National	Electronics	Take-back	2008
Chile – National	Pesticides Packaging	Take-back	2001
China - National	Large Appliances	UCTS	2012
Colombia – National	Batteries	Take-back	2010
Colombia – National	Beverage Containers	Deposit/Refund	1998
Colombia – National	Computers	Take-back	2010
Colombia – National	Fluorescent Lamps	Take-back	2010
Colombia – National	Medicine	Take-back	2010
Colombia – National	Pesticide Containers	Take-back	2007
Colombia – National	Tires	Take-back	2010
Cypress – National	Electronics	Take-back	2003
Cypress – National	Batteries	Take-back	2003
Cypress – National	Packaging	Take-back	2006
Cypress – National	Vehicles	Take-back	
Cypress – National	Tires	Take-back	
Cypress – National	Used Oil	Take-back	
Czech Republic – National	Batteries	Take-back	2006

Table 1 (continued from previous page)

Location	Material stream	Policy instrument	Instituted
Czech Republic – National	Electronics	Take-back	2005
Czech Republic – National	Packaging	Take-back	2002
Czech Republic – National	Vehicles	ADF	2003
Denmark – National	Batteries	Take-back	2006
Denmark – National	Electronics	Take-back	2005
Denmark – National	Packaging	Deposit/Refund	2001
Denmark- National	Vehicles	ADF	2002
Denmark- National	Tires	ADF	2002
Denmark- National	Used Oil	Take-back	2000
Estonia- National	Batteries	Take-back	2006
Estonia- National	Electronics	Take-back	2005
Estonia- National	Packaging	Take-back	2004
Estonia- National	Vehicles	Take-back	2005
Estonia- National	Tires	Take-back	2006
Finland - National	Batteries	Take-back	2006
Finland - National	Electronics	Take-back	1993
Finland – National	Packaging	Take-back	1998
Finland – National	Vehicles	ADF	2005
Finland – National	Tires	Take-back	1995
Finland – National	Medicine	Take-back	
France – National	Ag Waste	Take-back	2001
France – National	Batteries	Take-back	2001
France – National	Electronics	Take-back	2005
France – National	Furniture	Take-back	2012
France – National	Gas Bottles	Take-back	2013
France – National	Graphic Paper	ADF	2006
France – National	Med. and Haz. Waste	Take-back	2012
France – National	Packaging	ADF	1993
France – National	Spec. Dif. Household Waste	Take-back	2012
France – National	Vehicles	Take-back	2006
France – National	Textiles	ADF	2004
France – National	Tires	Take-back	2004
France – National	Medicine	Take-back	1993
Germany – National	Batteries	Take-back	2006
Germany – National	Electronics	Take-back	2005
Germany – National	Packaging	Take-back	1991
Germany – National	Vehicles	Take-back	1998
Germany – National	Used Oil	Take-back	1998
Greece – National	Batteries	Take-back	2006
Greece – National	Electronics	Take-back	2005
Greece – National	Packaging	Take-back	2003

Table 1 (continued from previous page)

Location	Material stream	Policy instrument	Instituted
Greece – National	Vehicles	Take-back	2004
Greece – National	Tires	Take-back	2004
Greece – National	Used Oil	Take-back	2004
Hungary – National	Batteries	Take-back	2008
Hungary – National	Electronics	Take-back	2005
Hungary – National	Packaging	ADF	1996
Hungary – National	Vehicles	ADF	2005
Hungary – National	Tires	ADF	2003
India – National	E-waste	Take-back	2010
India – National	Auto Batteries	Deposit/Refund	2001
Ireland – National	Batteries	Take-back	2006
Ireland – National	Electronics	Take-back	2005
Ireland – National	Packaging	Take-back	2007
Ireland – National	Vehicles	Take-back	2006
Ireland – National	Tires	Take-back	2008
Italy – National	Batteries	Take-back	2006
Italy – National	Electronics	Take-back	2005
Italy – National	Packaging	Take-back	1997
Italy – National	Vehicles	Take-back	2003
Italy – National	Tires	Take-back	2009
Italy – National	Used Oil	Take-back	1992
Japan - National	Packaging	ADF	1997
Japan - National	Computers	ADF	2001
Japan - National	Large Appliances	Take-back	2001
Japan - National	Vehicles	ADF	2005
Japan - National	Rechargeable Batteries	ADF	2001
Latvia – National	Batteries	Take-back	2006
Latvia – National	Electronics	Take-back	2005
Latvia – National	Packaging	Take-back	2002
Latvia – National	Vehicles	Take-back	2001
Latvia - National	Tires	ADF	2006
Latvia – National	Used Oil	Take-back	
Lithuania – National	Batteries	Take-back	2006
Lithuania – National	Electronics	Take-back	2005
Lithuania – National	Packaging	Take-back	2002
Lithuania – National	Vehicles	Take-back	2004
Lithuania – National	Tires	Take-back	
Luxembourg – National	Batteries	Take-back	2006
Luxembourg – National	Electronics	Take-back	2005
Luxembourg – National	Packaging	Take-back	2000
Luxembourg – National	Vehicles	Take-back	

Table 1 (continued from previous page)

Location	Material stream	Policy instrument	Instituted
Malta – National	Batteries	Take-back	2006
Malta – National	Packaging	Take-back	2004
Malta – National	Electronics	Take-back	2007
Mexico – National	Auto Batteries	Deposit/Refund	1998
Netherlands – National	Batteries	Take-back	2006
Netherlands – National	Electronics	Take-back	2005
Netherlands – National	Packaging	ADF	2008
Netherlands – National	Vehicles	ADF	2002
Netherlands – National	Tires	Take-back	2004
Netherlands – National	Used Oil	Take-back	
Netherlands – National	Electronics	Take-back	1999
Norway – National	Packaging	Take-back	1997
Norway – National	Tires	Take-back	1995
Philippines - National	Manufactured Goods	Deposit/Refund	2000
Poland – National	Batteries	Take-back	2002
Poland – National	Electronics	Take-back	2005
Poland – National	Auto Batteries	Deposit/Refund	2002
Poland – National	Packaging	Take-back	2002
Poland – National	Refrigerators	Take-back	2002
Poland – National	Vehicles	Take-back	2005
Poland – National	Tires	Take-back	2002
Poland – National	Used Oil	Take-back	2002
Portugal – National	Batteries	Take-back	2006
Portugal – National	Electronics	Take-back	2005
Portugal – National	Packaging	Take-back	1996
Portugal – National	Vehicles	ADF	2003
Portugal – National	Tires	ADF	2002
Portugal – National	Used Oil	Take-back	
Portugal – National	Medicine	Take-back	
Romania – National	Batteries	Take-back	2008
Romania – National	Electronics	Take-back	2005
Romania – National	Packaging	Take-back	2005
Romania – National	Vehicles	Take-back	
Romania – National	Tires	Take-back	2004
Singapore - National	Packaging	Take-back	2007
Singapore - National	Used Ink Cartridges	Take-back	2011
Slovakia – National	Batteries	Take-back	2006
Slovakia – National	Electronics	Take-back	2005
Slovakia – National	Packaging	Take-back	2003
Slovakia – National	Vehicles	ADF	2006

Table 1 (continued from previous page)

Location	Material stream	Policy instrument	Instituted
Slovakia – National	Tires	ADF	2001
Slovenia – National	Batteries	Take-back	2006
Slovenia – National	Electronics	Take-back	2005
Slovenia – National	Packaging	Take-back	2003
Slovenia – National	Vehicles	ADF	2003
Slovenia – National	Tires	ADF	2003
Slovenia – National	Used Oil	Take-back	
Slovenia - National	Medicine	Take-back	
South Korea - National	Solid waste	Take-back	2003
South Korea - National	E-waste	Deposit/Refund	1992
Spain – National	Batteries	Take-back	2006
Spain – National	Electronics	Take-back	2005
Spain – National	Packaging	Take-back	1996
Spain – National	Vehicles	Take-back	2004
Spain – National	Tires	Take-back	2005
Spain – National	Used Oil	Take-back	2007
Spain – National	Medicine	Take-back	
Sweden – National	Batteries	Take-back	2005
Sweden – National	Electronics	Take-back	2005
Sweden – National	Packaging	Take-back	1994
Sweden – National	Vehicles	Take-back	1998
Sweden – National	Tires	Take-back	1994
Sweden - National	Medicine	Take-back	
Switzerland - National	Beverage Containers	Deposit/Refund	1990
Thailand - National	Electronics	Take-back	2011
Turkey – National	Packaging	Take-back	1992
Turkey - National	Tires	Take-back	2010
UK - National	Batteries	Take-back	2006
UK - National	Electronics	Take-back	2006
UK - National	Packaging	Take-back	1997
UK- National	Vehicles	Take-back	2005
UK- National	Tires	Take-back	2004
UK- National	Packaging	ADF	1997
US - Arizona	Auto Batteries	Deposit/Refund	1990
US - Arkansas	Automobile Switches	Take-back	2005
US - Arkansas	Auto Batteries	Deposit/Refund	1992
US - Arkansas	Tires	ADF	1997
US - California	Carpet	Take-back	2011
US - California	Paint	Take-back	2012
US - California	Thermostats	Take-back	2009

Table 1 (continued from previous page)

Location	Material stream	Policy instrument	Instituted
US - California	Pesticide Containers	Take-back	2006
US - California	Batteries	Take-back	2006
US - California	Cell Phones	Take-back	2006
US - California	Toxic Substances	Content Standards	2012
US - California	Electronics	ADF	2005
US - California	Beverage Containers	Deposit/Refund	1987
US - California	Tires	ADF	1993
US - Colorado	Tires	ADF	1988
US - Connecticut	Electronics	Take-back	2011
US - Connecticut	Paint	Take-back	2013
US - Connecticut	Thermostats	Take-back	2013
US - Connecticut	Beverage Containers	Deposit/Refund	1980
US - Connecticut	Auto Batteries	Deposit/Refund	1990
US – Connecticut	Mattresses	Take-back	2013
US - Delaware	Tires	ADF	2007
US - Florida	Rechargeable Batteries	Take-back	1989
US - Florida	Tires	ADF	1988
US - Georgia	Tires	ADF	2005
US - Hawaii	Electronics	Take-back	2010
US - Hawaii	Beverage containers	Deposit/Refund	2005
US - Hawaii	Tires	ADF	1994
US - Idaho	Auto Batteries	Deposit/Refund	2001
US - Illinois	Electronics	Take-back	2008
US - Illinois	Automobile Switches	Take-back	2007
US - Illinois	Thermostats	Take-back	2010
US - Illinois	Tires	ADF	1991
US - Indiana	Automobile Switches	Take-back	2006
US - Indiana	Electronics	Take-back	2007
US - Indiana	Tires	ADF	1990
US - Iowa	Thermostats	Take-back	2009
US - Iowa	Automobile Switches	Take-back	2006
US - Iowa	Batteries	Take-back	1996
US - Iowa	Beverage Containers	Deposit/Refund	1979
US - Kansas	Tires	ADF	1990
US - Kentucky	Tires	ADF	1998
US - Louisiana	Automobile Switches	Take-back	2007
US - Louisiana	Tires	ADF	1992
US - Maine	Electronics	Take-back	2006
US - Maine	Thermostats	Take-back	2006
US - Maine	Automobile Switches	Take-back	2003
US – Maine	Batteries	Take-back	1996

Table 1 (continued from previous page)

Location	Material stream	Policy instrument	Instituted
US – Maine	Fluorescent Lamps	Take-back	2006
US - Maine	Beverage Containers	Deposit/Refund	1978
US - Maine	Auto Batteries	Deposit/Refund	1989
US - Maine	Tires	ADF	1990
US - Maryland	Electronics	Take-back	2005
US - Maryland	Primary Batteries	Take-back	1994
US - Maryland	Rechargeable Batteries	Take-back	1994
US - Maryland	Automobile Switches	Take-back	2009
US - Maryland	Tires	ADF	1991
US - Massachusetts	Automobile Switches	Take-back	2006
US - Massachusetts	Beverage Containers	Deposit/Refund	1983
US - Michigan	Electronics	Take-back	2008
US - Michigan	Beverage Containers	Deposit/Refund	1978
US - Minnesota	Electronics	Take-back	2008
US - Minnesota	Rechargeable Batteries	Take-back	1991
US - Minnesota	Auto Batteries	Deposit/Refund	1989
US - Minnesota	Paint	Take-back	2013
US - Mississippi	Tires	ADF	1992
US - Missouri	Electronics	Take-back	2008
US - Missouri	Tires	ADF	1991
US - Montana	Thermostats	Deposit/Refund	2009
US - Nebraska	Tires	ADF	1990
US – Nevada	Tires	ADF	2008
US - New Hampshire	Thermostats	Take-back	2009
US - New Jersey	Electronics	Take-back	1991
US - New Jersey	Batteries	Take-back	2005
US - New Jersey	Automobile Switches	Take-back	2005
US - New Jersey	Tires	ADF	1987
US - New York	Electronics	Take-back	2010
US - New York	Rechargeable Batteries	Take-back	2010
US - New York	Beverage Containers	Deposit/Refund	1983
US - New York	Auto Batteries	Deposit/Refund	1991
US - New York	Tires	ADF	2003
US - North Carolina	Electronics	Take-back	2007
US - North Carolina	Automobile Switches	Take-back	2006
US - North Carolina	Tires	ADF	2002
US – Ohio	Tires	ADF	1999
US - Oklahoma	Electronics	Take-back	2008
US - Oklahoma	Tires	ADF	1989
US - Oregon	Electronics	Take-back	2009
US - Oregon	Electronics	Take-back	2010

Table 1 (continued from previous page)

Location	Material stream	Policy instrument	Instituted
US - Oregon	Paint	Take-back	2009
US - Oregon	Beverage Containers	Deposit/Refund	1972
US - Pennsylvania	Thermostats	Take-back	2008
US - Pennsylvania	Electronics	Take-back	2010
US - Pennsylvania	Tires	ADF	1997
US - Rhode Island	Automobile Switches	Take-back	2005
US - Rhode Island	Electronics	Take-back	2009
US - Rhode Island	Paint	Take-back	2012
US - Rhode Island	Thermostats	Take-back	2011
US - Rhode Island	Tires	Deposit/Refund	1989
US - South Carolina	Automobile Switches	Take-back	2006
US - South Carolina	Electronics	Take-back	2010
US - South Carolina	Auto batteries	Deposit/Refund	1991
US - South Carolina	Tires	ADF	1991
US - Tennessee	Tires	ADF	1994
US - Texas	Electronics	Take-back	2008
US - Utah	Automobile Switches	ADF	2007
US - Utah	Tires	ADF	1991
US - Vermont	Fluorescent Lamps	Take-back	2012
US - Vermont	Thermostats	Take-back	2009
US - Vermont	Batteries	Take-back	1993
US - Vermont	Automobile Switches	Take-back	2006
US - Vermont	Electronics	Take-back	2011
US - Vermont	Beverage containers	Deposit/Refund	1973
US - Vermont	Paint	Take-back	2013
US - Virginia	Electronics	Take-back	2009
US - Virginia	Automobile Switches	Take-back	2007
US - Virginia	Tires	ADF	2008
US - Washington	Electronics	Take-back	2009
US - Washington	Fluorescent Lamps	Take-back	2013
US - Washington	Auto batteries	Deposit/Refund	2005
US - West Virginia	Electronics	Take-back	2008
US - Wisconsin	Electronics	Take-back	2010

Note: The policy instrument listed is the closest match to the six policy types listed in the 2001 OECD Guidance Manual

ANNEX 3: ABBREVIATIONS AND TERMINOLOGY

Abbreviations and terminology as used in the document are listed below. Note that the terminology used generally follows the academic economic literature, which may not always reflect terminology use in other contexts – important discrepancies are noted below

Abbreviations

ADF – Advanced Disposal Fee

ARF – Advanced Recycling Fee

CEMS – Continuous Emissions Monitoring System

CBA – Cost Benefit Analysis

DfE – Design for the Environment

DSD – Duales System Deutschland

EPR – Extended Producer Responsibility

ETS – Emissions Trading System

EU – European Union

OECD – Organisation of Economic Cooperation and Development

PAYT – Pay As You Throw

PRN – Packaging waste Recovery Notes

PRO – Producer Responsibility Organization

SMM – Sustainable Materials Management

UCTS – Upstream Combined Tax/Subsidy

UK – United Kingdom

US – United States

WEEE – Waste Electrical and Electronic Equipment

WPRPW – Working Party on Resource Productivity and Waste

WTO – World Trade Organization

Terminology

Advanced Disposal Fee – a product charge/fee levied at the point of sale. The literature generally does not consider how the fees are collected or put to use, and as such does not differentiate between an Advanced Recycling Fee (ARF – where the fee used to fund recycling efforts) and an Advanced Disposal Fee (ADF). For example, the fee charged on purchases of new vehicles in Portugal would be considered an ADF by the literature, though the revenues are used to finance recycling activities.

Command and Control – policies that mandate specific activities or standards to be met by firms or consumers to achieve a desired policy outcome, generally enforced by fines.

Deadweight loss – the economic cost of market distortions (including policy interventions), measured as the sum of lost consumer surplus (monetary equivalent of consumer utility) and lost producer surplus (profits).

Deposit/Refund – the deposit is payment made at the point of sale of the product, while the refund is only received if the product is returned to an authorized recipient. For example, in British Columbia, sellers of beverage containers charge a deposit at the point of sale, and refund the amount if the container is returned.

Downstream – generally refers to consumer and firm activities near the point of sale of products (purchase and end-of-life).

Extended Producer Responsibility – EPR is an environmental policy approach in which a producer's responsibility, physical and/or financial, for a product is extended to the post-consumer stage of a product's life cycle.

Externalities – any cost or benefit borne by an agent who receives no compensation for the cost nor pays for the benefit. Related to the idea of "external costs" – uncompensated costs imposed on others.

First-best – in a market with a single market failure, a first-best policy would correct the market failure at the least-cost (for example, a Pigouvian Tax). Because markets often contains many market failures, the "Theory of the Second-best" is often invoked to analyse least-cost policies when other market failures are present.

General/Partial equilibrium – a partial equilibrium model holds prices in other markets fixed, focusing on a single market of interest. General equilibrium relaxes this assumption, such that changes in one market can affect other markets.

Instruments – a policy tool that can be used to influence behaviour towards a desired outcome. A single instrument policy targets only one margin of behaviour, while a multi-instrument policy can target several margins of behaviour.

Life-cycle – the "life" of a product from "cradle-to-grave," encompassing extraction, production, consumption, disposal, recycling, and any other intermediary steps.

Market-based instruments/policy– synonymous with "economic instruments," market-based instruments seek to achieve a desired policy outcome by using financial incentives (taxes, subsidies, tradable permits) to influence economic behaviour.

Market power – a market in which particular agents can influence price through their actions (price-makers), generally to the detriment of consumers. Concerns generally arise in concentrated industries, or where collusion between multiple firms is possible.

Pigouvian Tax – a tax set equal to the marginal external costs of an externality. If waste disposal creates an external cost of $8/ton, then a pay-as-you-throw tax of $8/ton would be considered a Pigouvian tax.

Producer Responsibility Organization – an organization controlled and funded by producers to set up and manage the infrastructures that satisfy product take-back obligations on behalf of individual producers.

Recycling Content Standard – a performance standard that requires that products contain a mandated percentage of recycled content. For example, requiring all aluminium cans to incorporate 50% recycled content would constitute a recycling content standard.

Social cost – the sum of private costs and external costs. Private costs are internalized by decision makers, while external costs are not (giving rise to "externalities").

Take-back – a policy where producers are given responsibility for end-of-life products, either individually or collectively, and generally paired with mandated targets for collection and recycling. For example, the German Packaging Ordinance implemented in 1991 requires take-back of packaging products.

Upstream – generally refers to firm activities prior to the point of sale (extraction, production, design).

Upstream Combined Tax/Subsidy – a policy similar to deposit/refund, but rather than levied at the point of sale, the charge and refund is placed further "upstream" in the production process. For example, a tax levied by weight on aluminium ingots coupled with a subsidy by weight to collectors of aluminium cans would constitute a UCTS.

Virgin Materials Tax – a tax levied on extraction or use of virgin materials. For example, a tax on the use of virgin wood pulp would constitute a virgin materials tax.

Waste – unwanted post-consumption residuals. A point of confusion is that some of the economics literature uses "waste" as a synonym for "disposal" or "garbage."

Waste Disposal – the portion of the waste stream destined for disposal (generally assumed to be landfill in the economics literature).

Waste Recycling – the portion of the waste stream not destined for disposal. As used in the economics literature, it generally encompasses activates such as re-use and recovery.