

MARCH 2006 ■ RFF DP 06-08

Extended Producer Responsibility and Product Design

*Economic Theory and
Selected Case Studies*

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Abstract

A core characteristic of extended producer responsibility (EPR) policies is that they place some responsibility for a product's end-of-life environmental impacts on the original producer and seller of that product. The intent is to provide incentives for producers to make design changes that reduce waste, such as improving product recyclability and reusability, reducing material usage, and downsizing products. This paper assesses whether the range of policies that fall under the EPR umbrella can spur this "design for environment" (DfE). It summarizes the economics literature on the issue and describes conceptually how policies should affect design. It then analyzes three case studies in detail and two more case studies more briefly. The conclusion reached is that some DfE—especially reductions in material use and product downsizing—can be achieved with most EPR policies, including producer take-back mandates and combined fee/subsidy approaches. However, none of these alternative policies as they are currently implemented are likely to have a large impact on other aspects of DfE.

Key Words: design for environment, recycling, waste management, incentive-based policy instruments

JEL Classification Numbers: Q53, Q58

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Extended Producer Responsibility and Product Design: Economic Theory and Selected Case Studies

Margaret Walls*

I. Introduction

Over the 15 or so years since the term *extended producer responsibility* was first coined in Sweden and the now-famous German packaging “take-back” law was passed, the EPR concept has become an established principle of environmental policy in many countries. Although EPR means slightly different things to different people, a core characteristic of any EPR policy is that it places some responsibility for a product’s end-of-life environmental impacts on the original producer and seller of that product. In the *EPR Guidance Manual for Governments*, the Organisation for Economic Co-operation and Development (OECD 2001) defines EPR as “an environmental policy approach in which a producer’s responsibility for a product is extended to the post-consumer stage of a product’s life cycle.” The OECD goes on to say that in addition to shifting responsibility—either financial or physical—upstream to producers, it is also important that the policy “provide incentives to producers to incorporate environmental considerations in the design of their products.” In fact, the original motivation for product take-back mandates, including the German packaging program, was to provide incentives for producers to make changes that would reduce waste management costs. Those changes would include improving product recyclability and reusability, reducing material usage, downsizing products, and engaging in a host of other “design for environment” (DfE) activities.

The extent to which EPR policies lead to DfE is an open question, however. Much that is written on the topic seems to take it on faith that any form of producer responsibility will provide DfE incentives, but there is very little careful conceptual thinking on how such incentives work through the system and sparse documentation of real-world changes that have been made in response to policies. This study attempts to partially fill this gap.

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I begin Section II by describing the policy instruments that come under the EPR umbrella. EPR is a policy concept, and many different instruments—not just producer take-back mandates—can be considered EPR. However, the instruments will have different impacts on waste generation, virgin material use, and product design. I contrast EPR instruments with non-EPR policies, such as “pay-as-you-throw” waste collection charges. In this beginning section of the study, I also talk about a topic of current interest in EPR policy circles, the notion of “individual” versus “collective” responsibility. And finally, I include a discussion of transaction costs and the administrative feasibility of some of the instruments.

Section III of the study summarizes the extant economics literature on environmental policy design and the impacts of policy on producer behavior, including product design decisions. Results from recent theoretical analyses of optimal policies in a setting in which producers can make environmental product design decisions are discussed.

The penultimate section of the study includes case studies of policies in the real world. I focus on the Canadian used oil program and the packaging waste program in the United Kingdom because they are examples of economic incentive-based approaches. I then describe the Dutch waste electronic and electrical equipment (WEEE) program, a more traditional take-back program that in the early 2000s had a unique financing mechanism. The Korean WEEE program is also briefly discussed, and the concluding part of this section briefly discusses other existing EPR programs focused on packaging and vehicles. Section V provides some concluding remarks.

II. Policy Instruments

Several policy instruments, and variants of those instruments, fall under the EPR umbrella. The following list of the most common instruments is not meant to be exhaustive but includes most of the policy tools used in practice.

Product take-back mandate and recycling rate targets. With this policy approach, the government mandates that manufacturers and/or retailers take back products at the end of the products’ useful lives. Combined with such mandates is some kind of recycling or waste diversion target. For example, the government may require that each producer meet a recycling rate goal of, say, 75 percent for its products. The German packaging law works in this way; take-back is required and material-specific recycling rate targets are set. To meet these requirements, firms often form a “producer responsibility organization,” or PRO, to handle collection, arrange for recycling, and ensure that recycling targets are met. I will say more about PROs below.

Product take-back mandate and recycling rate targets, with a tradable recycling credit scheme. This approach is the same as above except that the targets apply not to each individual producer but to the industry; tradable credits are issued and firms are allowed to trade among themselves. An industry-wide recycling rate target is thus met even though some producers do better than the target and others worse. There are several ways that a tradable credit scheme could be set up, and I discuss the possibilities more in the next section. One example is the packaging system in the United Kingdom. There, reprocessors of packaging materials issue “packaging waste recovery notes,” or PRNs, which firms and PROs can trade with one another to meet their recycling obligations.¹

Voluntary product take-back with recycling rate targets. In a purely voluntary approach, firms in an industry agree to organize a take-back system for their products and set recycling goals. There are no laws or government regulations mandating compliance and no penalties for not meeting the goals. In the United States, voluntary take-back programs of this type include the Rechargeable Battery Recycling Corporation (RBRC), which represents manufacturers of rechargeable batteries who pay a fee to operate a collection and recycling system. Another is the Carpet America Recovery Effort (CARE), created by an agreement among U.S. carpet manufacturers that arose out of a 2002 memorandum of understanding between those manufacturers and several state governments and the U.S. Environmental Protection Agency. The memorandum set voluntary recycling rate goals for carpet to be reached by 2012.

Advance recycling fees. An ARF—which originally was referred to as an advance disposal fee, or ADF—is a tax assessed on product sales and often used to cover the cost of recycling. ARFs are often assessed per unit of the product sold but can also be assessed on a weight basis. ARFs may be visible to the consumer when he purchases a product—that is, as a separate line item on the bill, similar to sales tax—or they can be assessed upstream on producers and later be incorporated into the product retail price.

ARF combined with a recycling subsidy. An ARF raises money that can be used in a variety of ways. The incentive effects of the policy are highly dependent on both the type of ARF and what is done with the revenues. A “back-end” recycling subsidy—either a subsidy per unit of the product recycled or per pound of material recycled—leads to quite a different policy

¹ PROs in the U.K. program are called compliance schemes, and there are multiple compliance schemes in operation.

instrument than one in which the ARF revenues are used to cover the costs of managing waste or to cover infrastructure costs, in a lump-sum fashion. I will discuss these differences in more detail in the next section. California's used oil program, the western Canada used oil program, lead-acid battery programs in several U.S. states, and California's e-waste program are all ARF/recycling subsidy programs.²

All of these policy instruments make the producer financially or physically responsible for the end-of-life environmental impacts of his product. In this sense, all could be considered EPR. However, they have very different incentive effects and ultimately may lead to different environmental outcomes, and the costs of the instruments may differ widely.

Other policy instruments can lead to similar outcomes but do not focus upstream on producers. Four such non-EPR instruments are listed here.

Landfill bans. Many U.S. states and several countries ban disposal of particular items in landfills (or incinerators). In the United States, these bans cover white goods such as refrigerators, dishwashers, and the like; computer monitors; tires; various kinds of household hazardous wastes such as paints, fluorescent light bulbs, and batteries; and other items.

"Pay as you throw" pricing of waste collection/disposal. More than 4,000 communities in the United States charge a fee per container or per bag of trash collected at curbside (U.S. EPA 1999). This approach is in contrast to no-fee disposal or fees that do not vary with the volume of waste collected. In the Netherlands, the city of Oostzaan, as well as some others, charges a fee per kg of waste collected (Linderhof et al. 2001). In some countries, end-of-life fees are charged for specific items that are difficult to dispose of. None of these policies are aimed at the producer and so do not qualify as EPR, but they may have some of the same effects on waste generation and recycling.

Recycling subsidies. I discussed recycling subsidies in the context of ARFs above; the government also may raise funds from elsewhere and subsidize recycling. The government could make a payment per unit or per kg of material recycled, or it could make lump-sum grants to communities or recycling centers. Such grants are quite common in the United States. Whether

² The Western Canada used oil program, in operation in British Columbia, Alberta, Saskatchewan, and Manitoba, is an industry-run program in which an industry association sets the level of the ARF and the recycling subsidy (referred to as a "return incentive"). Legislation passed in each province mandates that anyone selling motor oil and motor oil containers and filters pay the fee set by the association.

the subsidy is per unit, per kg, or a one-time lump-sum payment will have different effects, as I discuss in the next section.

Recycling investment tax credits. Recycling investment tax credits are also quite common. Here, the government gives a credit on income taxes to anyone who invests in recycling infrastructure; this is like a direct subsidy to capital.

In the next section, I discuss the economic efficiency implications of some of these policies, calling on results from the literature and a simple conceptual model of producer and consumer behavior. Because a crucial component of economic efficiency, it will be shown, is optimal product design, a discussion of DfE is included in the section.

Before beginning that discussion, it is worth highlighting here a few additional considerations in evaluating these policies. These are (1) the feasibility of implementation; (2) problems that arise when there are multiple policy objectives; and (3) the issue of collective versus individual responsibility in take-back programs. I discuss each of these issues in turn.

Feasibility

Obviously, the administrative costs of designing, implementing, and enforcing compliance with a policy determine whether an approach will be cost-effective in achieving its goals. Transaction costs incurred by participants in the marketplace in the course of complying with the policy are also important. These cost concerns loom especially large in the debate about DfE. Some observers have criticized collective take-back systems and others have criticized approaches such as combined ARF/recycling subsidies or deposit-refund schemes because they do not seem to directly encourage DfE. However, alternative approaches that seem more direct—individual take-back or fees and subsidies that vary with product recyclability—may have such high administrative and transaction costs that they are essentially infeasible. This is, of course, the reason that PROs were formed in the first place and why they continue to thrive in countries with take-back programs.

Multiple Policy Objectives

In an earlier study, I argued that the goals of EPR, or any policy, need to be clearly laid out before policy instruments can be evaluated (Walls 2004). And in all cases, we should strive to achieve the given environmental objective at the lowest possible cost to society. In the case of EPR, the environmental objective is often not clear. Some objectives that have been put forward are (1) reduction in waste volumes generated; (2) reduction in waste disposed; (3) reduction in hazardous constituents in the waste stream; (4) decrease in virgin material use; (5) lowering of

pollution in the production stage; and (6) increased DfE. Some observers have argued for achievement of all of these goals. The U.S. Environmental Protection Agency describes product stewardship (the terminology commonly used in the United States) as calling on “those in the product life cycle—manufacturers, retailers, users, and disposers—to share responsibility for reducing the environmental impacts of products.” The problem here is the broad range of “environmental impacts” of products and the lack of clarity in exactly what shared responsibility means.

A long-standing result in economics is that as many policy instruments are needed as policy goals (Tinbergen 1967). One instrument cannot efficiently accomplish all objectives. This means that if we want to reduce exposure to hazardous substances in products and also reduce volumes of waste generated from products, we are likely to need at least two instruments. For example, the European Union’s Restriction of the Use of Hazardous Substances Directive bans the use of lead, mercury, brominated flame retardants, and other hazardous substances in electrical and electronic equipment. We can compare this approach—an outright ban on the use of something—with alternative approaches to accomplishing the same end. But we cannot compare it with something like an ARF if the objective of the ARF is to reduce volumes of waste disposal. That would be an apples-to-oranges comparison. Too often, EPR debates become mired in such discussions.³

Individual versus Collective Take-Back

Although the first generation of EPR programs involved collective take-back—that is, PROs that arrange with producers to collect and recycle their end-of-life products—there has been more interest of late in individual take-back programs. In such a situation, individual producers would be responsible for collecting and recycling their own products. Interest has arisen in this approach both because collective programs may not do enough to spur DfE and because some producers in some industries have advocated it. For example, U.S.-based computer equipment manufacturer Hewlett Packard has strongly argued that any state laws using an ARF approach allow for opt-out by individual companies that can demonstrate that they have their

³ One of the most detailed published studies looking at the link between EPR and DfE is a 2002 study funded by Environment Canada (Lura Consulting 2002). Although this lengthy study contains a good deal of thoughtful analysis and information, it makes the mistake of comparing policy instruments along several dimensions—toxicity reduction, increased recycled content, extended life, and others—that those instruments are not necessarily designed to address. An “average DfE score” is calculated based on the individual scores along these dimensions.

own take-back/recycling programs. The company's arguments are based on the assumption that its own approaches will be more cost-effective than government-sponsored systems. Similar thinking was behind the collective arrangement of HP, Electrolux, Braun, and Sony to establish their own PRO to handle e-waste Europe-wide. Their express purpose was to provide competition to existing PROs in European countries while still achieving some scale economies in collection by uniting four companies' efforts (see letsrecycle.com 2004).

There are obvious trade-offs involved in a collective system versus an individual one. Although an individual system may provide more direct incentives for DfE, it may be difficult for the government to monitor and enforce the activities of many individual companies. Moreover, there should be economies of scale in collection. Many empirical studies of local waste and recyclables collection services have found that such economies exist (see Walls et al. 2005 and references therein), and individual efforts by many companies to collect their own products at end-of-life are sure to be excessively costly.⁴ On the other hand, mandating a PRO with joint collection and processing may be overly prescriptive; having the government choose the system ex ante eliminates the possibility for firms to uncover cost savings in collection and processing. Another potential disadvantage of a PRO is the possibility of anticompetitive behavior. Control of collection and contracts for processing of recyclables by one firm could lead to price gouging and other problems. Similarly, if firms in an industry cooperate to jointly arrange for collection and processing, forming a PRO on their own, there is the potential that they will collude on other things as well.

In general, if the government is going to impose take-back, it is best if obligated firms have options to come up with innovative take-back strategies on their own, since their incentives to minimize costs will help reduce the overall costs of the system. Of course, a better option might be to bypass the take-back option for something even more flexible. I will discuss the possibilities in the following sections of the paper.

⁴ For this reason, it may be costly to have local authorities collect some waste streams and PROs collect others (or contract with waste companies to do so), since they are likely to duplicate each others' efforts to some extent.

III. Economics, the Product Life-Cycle, and Incentive Effects of Policies

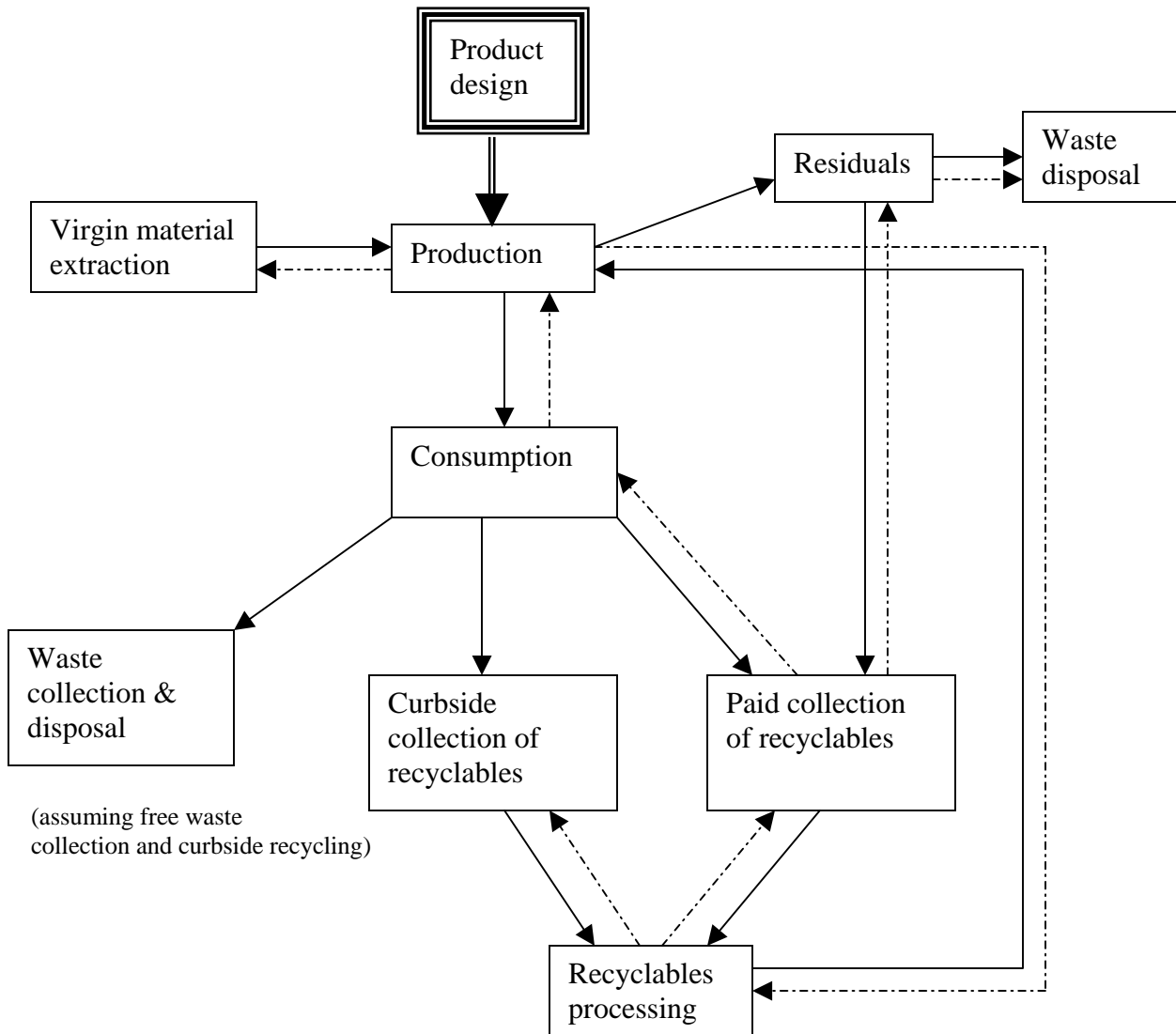
As long as markets are well-functioning, policies aimed upstream at producers or downstream at consumers will have their effects felt up and down the product chain. How these effects are felt and the incentives they provide depend on how well markets function at each stage. Figure 1 is a simple diagram of material flows through the life-cycle of a product. The flow of payments, determined by market prices, is also shown. Any raw materials that enter the system must eventually be either recycled and reused in production or end up as waste—in other words, a materials balance condition must hold. That waste could be from the production process or it could come after consumption; in both cases, it is either disposed of or sent to processors for recycling. It is assumed that producers pay for the materials they use to make their goods; either they pay primary producers for virgin materials or they pay processors for secondary materials. Postconsumption waste collection and disposal and collection of recyclables at the curbside are assumed to be free, as they often are in reality; however, this assumption could be relaxed. No illegal disposal option is shown in the diagram. If waste collection is priced, there is the possibility that dumping will occur.

Several studies by economists have modeled the system shown in Figure 1 by assuming that producers choose material inputs and the level of output to maximize profits, taking all prices as given, and that consumers choose how much to consume, recycle, and throw away so as to maximize utility, subject to a budget constraint.⁵ A few studies include recycling: recyclers purchase used products from consumers (or get them for free), process them, and resell to producers. And a study that both brings in the activities of recyclers and explicitly incorporates product design is Calcott and Walls (2005). In this study, producers can make their products more recyclable but must incur some costs to do so; the more recyclable products are, the less costly they are to process. Recyclers pay consumers more for those products that have a higher degree of recyclability. In the Calcott and Walls model, consumers can set trash and recyclables at the curbside for free, but they also can take recyclables to a recycling center where they can receive payment. Payment is higher the more recyclable the product is. Taking items to a center involves transaction costs, however; thus consumers weigh those additional costs against the

⁵ Notable studies include Dinan (1993), Fullerton and Kinnaman (1995), and Palmer and Walls (1997). None of these studies incorporate a recycler's profit-maximization condition explicitly in the model and none address issues of product design.

ease of curbside recycling, where no payment is made. In Figure 1, product design is simply illustrated by an extra box with an arrow to the production phase. Design does not involve material resources but it does involve other resources—for example, labor and capital—and costs during the production stage, and its effects are felt throughout the product life-cycle.

Figure 1. Raw Materials and Product Markets: The Flow of Materials and the Flow of Payments



Solid lines depict material flows; dashed lines the money flows.

There are three essential maximization problems in the Calcott and Walls framework, which I describe here.

1. Consumers maximize their utility, or well-being, subject to a budget constraint. They choose varieties of a consumption good that are distinguished from one another by their degree of recyclability. It is assumed that consumers don't care which variety they consume unless it affects their budget constraint. Their budget constraint can be affected because recyclers might pay them for highly recyclable products. Aggregate waste volumes affect consumers' utility negatively; this externality provides the rationale for government policy intervention in private markets.
2. Heterogeneous producers in the model are assumed to maximize profits. They choose how much to produce and how recyclable to make their products, taking all output and input prices as given. They incur extra costs to make their products more recyclable. In making their decisions, they must balance that extra cost with extra revenue they might earn on products that are more recyclable. Because such products are less costly to recycle, recyclers will pay consumers more for them at end-of-life, and thus consumers may be willing to pay more for them at time of purchase.
3. Finally, recyclers (or reprocessors) in the model collect secondary materials from consumers, reprocess them, and sell the reprocessed material to producers. Producers use the materials as inputs into production of new products. Recyclers will pay consumers for highly recyclable used products and will take some others for free; they will not accept items with recyclability levels so low that the cost of reprocessing outweighs the revenues earned. Thus, there is always some amount of waste disposal.

A mass balance condition holds in the model: all raw materials coming into the system must leave as waste disposed of in a landfill or be recycled into new products.

Calcott and Walls solve for the socially optimal level of waste, recycling, production, and product design—that is, product recyclability—and then analyze what combination of policy instruments might achieve the social optimum. Specifically, they solve for the combination of disposal fee, product tax (or ARF), and recycling subsidy that makes the private market outcome identical to the social optimum.

Because recyclability is assumed to be difficult to observe perfectly and thus the government cannot assess taxes and subsidies that vary continuously with it, the social optimum in the Calcott and Walls model is a constrained optimum. In an earlier paper with a similar model, the authors showed that when taxes and subsidies vary perfectly with recyclability, then a combined product tax/recycling subsidy, similar to a deposit-refund instrument, can achieve the full social optimum (Calcott and Walls 2000). This is akin to a finding in Fullerton and Wu

(1998). However, it is probably impractical to think that the government can set output taxes that vary with recyclability. For this reason, Calcott and Walls (2005) set up a slightly different but more realistic framework and solve for the instruments that can achieve the constrained optimum. This is the best outcome that can be achieved given the difficulty in observing and taxing product recyclability. The authors find that the constrained optimum can be implemented with relatively simple instruments: a product tax and recycling subsidy applied to all products, combined with a relatively modest waste disposal fee. The tax and subsidy are equivalent and equal the difference between unit recycling costs and virgin material costs, and the disposal fee equals the social marginal cost of waste disposal less unit recycling costs.⁶ The instruments send signals throughout the life-cycle, to producers, consumers, and recyclers. Together with the existence of markets themselves—markets that, while working only imperfectly, still send signals to producers to make products more recyclable—these instruments yield the constrained optimal amount of waste disposal, recycling, consumption, and product design.

This is a reasonable set of policy instruments that should be comparatively easy to implement in practice for many products. And the tax/subsidy combination falls under the EPR umbrella—by paying the fee up front, producers are financially responsible for the waste generated from their products.

In the real world, however, policymakers are using different types of EPR and other instruments, and in many countries, the take-back approach is preferred. Table 1 below summarizes the incentive effects of a variety of instruments. An efficient, or cost-effective, instrument is one that exploits all the different means by which waste can be reduced. In other words, reducing material use in production, redesigning products to be more recyclable, consuming less, and recycling more are all likely to take place, to some degree, when cost-effective waste reductions occur. A policy that encourages, say, only recycling will be missing opportunities to reduce waste through source reduction. Therefore, the more “yes” answers in the cells of a particular row of Table 1, the more cost-effective that instrument will be at reducing waste. A combined ARF/recycling subsidy, for example, does a better job than an ARF alone because the former encourages both source reduction and recycling while the ARF by itself encourages only source reduction. Similarly, recycling subsidies, tax credits, or grants to fund

⁶ Recycling costs in these expressions are evaluated at the lowest level of recyclability at which recyclers will accept a product for recycling—that is, the lowest level at which they will cover their costs.

recycling centers encourage recycling but do not reduce output or material use; in fact, if they reduce the cost of an input to production, they may increase output.

Another important characteristic of an efficient policy instrument is one that is imposed “at the margin.” For example, a recycling subsidy should do a better job of encouraging recycling than a lump-sum grant for a recycling center. With the former, recyclers receive more money the more pounds of material they recycle; the lump-sum instrument does not similarly reward increases in recycling.

Some instruments may look similar along many dimensions in the table but are very different in terms of their flexibility across firms or across consumers and thus are likely to have very different costs. For example, a recycled content standard that requires all producers to make their products using a specific minimum amount of secondary material makes all producers meet the same standard even though it may be more economically efficient to have some producers use more and others use less than the standard requires.

Some policies may have unintended consequences. For example, a ban on disposing a particular product in a landfill and pay-as-you-throw pricing of waste collection services may both lead consumers to illegally dump their waste. This may not be a problem with pay-as-you-throw if the fees are low, and a ban may not be problematic for some products and in some locations, but policymakers need to be aware that some instruments inadvertently encourage dumping, thus creating a potentially more serious environmental problem than legal disposal.

Take-back mandates are somewhat difficult to compare with the other instruments because the way they are laid out in legislation and regulations is different from the way they are implemented. Instead of take-back of brand-specific items by individual obligated producers, these programs involve collective take-back arrangements whereby a PRO acts on behalf of many individual producers, jointly arranging for collection and recycling of many firms’ products. The incentive effects of such an approach depend on the financing of the system and how the PRO functions. In general, PRO fees are based on product sales and often assessed on a weight basis. Thus, a PRO fee is like a product tax, or ARF, and can provide incentives similar to that instrument. Since take-back mandates include requirements that a particular proportion of the material be recycled, they spur recycling as well. However, a recycling subsidy should have continuous impacts on the volume recycled, whereas with a standard, there is no incentive for firms to exceed the minimum. In any case, the impacts on product design are likely to be minimal with take-back programs. The PRO fees can reduce material use and encourage downsizing, but there is no direct incentive for improvements in recyclability.

Table 1. Incentive Effects of Policy Instruments Designed to Reduce Waste

<i>Policy instrument</i>	<i>Impact on production/consumption?</i>	<i>Impact on virgin material use?</i>	<i>Impact on product design?</i>	<i>Impact on recycling?</i>	<i>Comments/other considerations</i>
Advance recycling fee (ARF), \$/unit: e.g., \$10/computer*	Yes, direct	Yes, indirect, only through output reduction; no material substitution effects	Unlikely	No (recycling may even fall as output falls)	
ARF, \$/kg*	Yes, direct	Yes; larger impact than \$/unit fee; nonmaterial inputs to production substituted for material inputs	Some: downsizing and lightweighting of products	No (recycling may even fall as output falls)	
Recycling subsidy (or tax credit for recycling), \$/unit	Indirect <i>increase</i> in production/consumption by reducing cost of input to production	Yes; substitution effect reduces use, but greater output offsets the effect	Indirect; subsidy's impact sends price signal upstream to producers to make products more recyclable	Yes, direct	Funding necessary
Recycling subsidy (or tax credit for recycling), \$/kg	Indirect <i>increase</i> in production/consumption by reducing cost of input to production	Yes; substitution effect reduces use (larger effect than \$/unit subsidy), but greater output offsets the effect	Indirect; subsidy's impact sends price signal upstream to producers to make products more recyclable (more direct than \$/unit subsidy)	Yes, direct (more direct than \$/unit subsidy)	Funding necessary

Table 1 continued from previous page.

<i>Policy instrument</i>	<i>Impact on production/consumption?</i>	<i>Impact on virgin material use?</i>	<i>Impact on product design?</i>	<i>Impact on recycling?</i>	<i>Comments/other considerations</i>
Recycling lump-sum grants: e.g., grants for establishing programs, centers	No	No	Possible, but very indirect	Yes, but only indirect, since no marginal effect	Funding necessary
Recycled content standard	Indirect; increased production costs reduce output	Yes, substitution effect reduces use	Indirect	Yes	Inflexible if all producers must meet same requirement
Virgin material tax, \$/lb	Indirect; increased production costs reduce output	Yes, direct	Some: downsizing and lightweighting of products	Yes, substitution effect causes shift from virgin to recycled inputs, so more recycling	
Combined output tax (ARF)/recycling subsidy, \$/lb	Yes, direct	Yes, direct	Direct effect on downsizing and lightweighting of products; indirect impact on recyclability: subsidy sends price signal upstream to producers	Yes, direct	
Pay-as-you-throw	Yes	Yes, substitution and output effects reduce use	Indirect impact on recyclability: price signal encourages downsizing and lightweighting of products and improved recyclability	Yes	Could lead to illegal dumping

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Table 1 continued from previous page.

<i>Policy instrument</i>	<i>Impact on production/consumption?</i>	<i>Impact on virgin material use?</i>	<i>Impact on product design?</i>	<i>Impact on recycling?</i>	<i>Comments/other considerations</i>
Landfill ban	No	No	No, unless ban is specific to a particular product component or material	Yes	Could lead to illegal dumping
Product labeling	No	Possible small effect, depending on type of label	Possible, depending on type of label	Possible small effect, depending on type of label	May be more appropriate for hazardous products
Take-back mandate and recycling rate target, w/PRO setting fees based on sales	Yes; direct, from PRO fees	Yes; substitution effect reduces use	If PRO fee is weight-based (\$/lb), downsizing and lightweighting of products; no impact on recyclability	Yes	Cost-effectiveness depends on how PRO operates
Take-back mandate and recycling rate target w/PRO and tradable credit scheme (credits assigned to producers)	Yes; direct, from PRO fees	Yes; substitution effect reduces use	Yes, direct; more recyclable a firm's product, more credits it earns	Yes	Sorting requirements and administration could be costly, but credits add flexibility
Take-back mandate and recycling rate target, w/PRO and tradable credit scheme (credits assigned to recyclers)	Yes; direct, from PRO fees	Yes; substitution effect reduces use	If PRO fee is weight-based (\$/lb), downsizing and lightweighting of products; no impact on recyclability because no brand sorting	Yes	No sorting by brand so lower cost but less impact on recyclability

*Revenues raised by the ARFs are assumed to go to the general fund and not to be used to subsidize recycling; for the impacts of a combined ARF/recycling subsidy, see row 8 of the table.

IV. EPR Case Studies: Used Oil, Packaging, and Electronics

In this section of the study, I describe four very different programs that can all be classified as EPR, in that producers are made responsible for the end-of-life environmental impacts associated with their products. The first is the used oil recovery program in western Canada. This is a combined ARF/recycling subsidy program: oil sales are subject to a fee that funds collection and recycling programs. The second case study is a take-back program for packaging in the United Kingdom that includes a tradable recycling credit provision. Finally, the third and fourth case studies involve waste electronic and electrical equipment and are more traditional take-back programs with producer responsibility organizations. One is the program in place in the Netherlands, in which computer equipment was originally sorted by brand and recycling costs were apportioned to manufacturers based on individual recycling costs. This financing mechanism was changed, however, and now the program operates in the same manner as many other take-back programs. The other WEEE program that I discuss is the Korean program.

For each of the case studies, I try to identify any product design changes that may have taken place in response to the policies. In some cases, this involves describing the most likely changes that could take place with the incentives provided. In others, it involves describing outcomes and attempting to determine whether those outcomes can be attributed to the programs or might be due to other factors. In still other instances, it is impossible to find information about product changes. This section of the study ends with some overall conclusions about the merits and failings of the individual programs and how they might compare with alternative policies.

A. Western Canada Used Oil Program

In the late 1980s, the environmental problem caused by improper disposal of used oil drew attention in western Canada. At the request of the Canadian Council of Ministers of Environment, the oil industry set up a task force to address the problem. The task force devised an industry-run program in which sales and imports of oil, oil containers, and oil filters would be subject to a fee, referred to as an environmental handling charge (EHC). Authorized collectors, transporters, and processors of used oil, filters, and containers would be paid a “return incentive” for every liter of oil, container, and filter that was recycled or reused. Used oil can be rerefined into lubricating oil, reprocessed into other petroleum products, or used as heating oil for industrial plants, asphalt plants, or small space heaters. Filters are recycled at steel

recycling mills, and used plastic containers are recycled into new oil containers or other plastic durable goods.

The program is run separately in each province. The Alberta Used Oil Management Association, a nonprofit organization, collects the fees and pays out the return incentives in Alberta. The other three western provinces—British Columbia, Saskatchewan, and Manitoba—also have nonprofit associations. Laws passed in each province allow these associations to set the EHCs and require businesses selling or importing oil and oil filters and containers in each province to join the associations. Alberta and Saskatchewan's programs began in 1997, with Manitoba's following in April 1998. British Columbia was the most recent province to adopt this fee/refund approach; its program started in mid-2003. In December 2004, the provincial associations, along with the nonprofit used oil group from Quebec, joined to form the National Used Oil Material Advisory Council to study the formation of a national used oil recovery program.⁷

The EHC in Alberta in 2003 was 5 Canadian cents per liter for oil, 50 cents for filters less than 203 mm in length and \$1 for filters 203 mm or more in length, and 5 cents per liter of container size. The return incentive varies by location; transport costs can be significant in these large provinces because of the long distances to markets, so a higher return incentive is paid in the northern regions than in the areas around major cities. In 2003, the return incentive in Alberta ranged between 8 and 17 cents per liter for used oil, 87 cents and \$1.10 per kilogram for containers, and 68 cents and \$1.19 per kilogram for used filters (Vanderpol 2004). The sales volume of lubricating oil is far greater than the volume of used oil, since approximately 35 percent is burned during use. For this reason, the return incentive for used oil can be greater than the EHC on oil sales.⁸

By most measures, the programs have been a considerable success. Recovery rates have risen every year that the programs have been in operation and are now quite high, particularly in comparison with other countries and U.S. states without used oil recovery programs. The percentage of used oil collected and processed in 2004 across all four provinces was 75 percent; this amounted to 47 percent of motor oil sales in that year (Meredith et al. 2005). A study for the

⁷ In 2004, Ontario voted down a proposal for a similar program in that province.

⁸ Worldwide, approximately 55 percent of lubricating oil sales goes toward automotive uses, with the remainder used for industrial purposes, as process oils, for metalworking, and as greases (Fitzsimons 2005).

Used Oil Management Association comparing the western Canada results with those in 14 other countries and U.S. states found that the Canadian program achieved higher recovery rates than any other program with the exception of the program in the United Kingdom (Meredith et al. 2005). The U.K. recovery rate was 76 percent compared with 75 percent in Canada. A significant fraction of Canadian used oil is rerefined—30 percent in 2004—whereas used oil in the United Kingdom is all burned for energy recovery.⁹ The primary externalities associated with used oil come from illegal disposal, so either energy recovery or rerefining should be beneficial; however, life-cycle studies have shown that rerefining has overall higher benefits because of the air pollution from burning waste oils (Fitzsimons 2005). The study also surveyed a large number of Canadian stakeholders about their experiences with the used oil program and found widespread support for it.

Table 2 below shows Alberta's recovery rate for used oil for 1997 through 2004. As the table shows, in the most recent year, Alberta's recovery rates were 77 percent for oil, 84 percent for filters, and 50 percent for containers. Although the recovery rates are not the same in each province, the upward trend in Alberta has been replicated across the other provinces.

Results in British Columbia are particularly revealing about the success of the fee/refund approach. Prior to July 2003, British Columbia had a different type of program—a more traditional take-back system in which retailers were required to act as return facilities and accept used oil from any consumer at no cost. They were also required to arrange for and pay to have a waste management company collect the used oil and transport to a refinery or processor. Although this take-back approach as well as the new fee/refund approach can both be considered EPR, results in British Columbia from the new program far surpass results from the take-back program. The main problem experienced with retailer take-back was enforcement: many retailers simply did not accept used oil. Since they were forced to incur the high costs of both take-back and waste management, many retailers simply opted for noncompliance. In 2002, the recovery rates for used oil, filters, and containers in British Columbia were, respectively, 61, 17, and 12 percent. In 2004, those rates had risen to 72, 82, and 42 percent (BCUOMA 2005).

⁹ There is a significant tax advantage in the United Kingdom for heating oil made from waste oil (see Fitzsimons 2005).

Table 2. Recovery Rates for Used Oil, Filters, and Containers in Alberta

	<i>Recovery rate</i> <i>(percentage of used oil, filters, and containers collected)</i>		
	<i>Used oil</i>	<i>Used oil filters</i>	<i>Used oil plastic containers</i>
1997	52	58	12
1998	66	70	19
1999	67	78	34
2000	67	78	38
2001	71	86	43
2002	73	89	45
2003	75	82	51
2004	77	84	50

Sources: Vanderpol (2004); Alberta Used Oil Management Association (2003, 2004, 2005).

An interesting result of the western Canada program is that even though the refunds are paid to authorized collectors, the generators of used oil, including do-it-yourself oil changers, farmers, and others, are benefiting, too. Collectors in Alberta have paid generators up to 37 percent of the return incentive for turning in their used oil and 35 percent of the return incentive for oil filters (McCormack 2000). This system thus provides incentives to downstream consumers even though the return incentive payment created by the system is paid only to collectors.

Good information is available on costs of the Canadian program, since the nonprofit associations in each province must file annual reports with balance sheets and other financial information.¹⁰ Vanderpol (2004) reports that preprogram industry estimates of the annual costs of removing used oil materials were approximately \$13 million in Alberta. This is an estimate of the costs that generators incurred to have used oil legally removed and managed. Total EHC payments collected are around \$13 million per year; thus the program's costs are no greater than preprogram costs incurred by generators. More importantly, of course, less used oil is dumped illegally into the environment, so pollution damage costs are far below preprogram levels. Administrative costs are also low. In Alberta, administrative costs are currently around 4 percent of total EHC collections, well below the estimates made when the program began (Vanderpol 2004).

¹⁰ Such information is not as readily available for government-run programs.

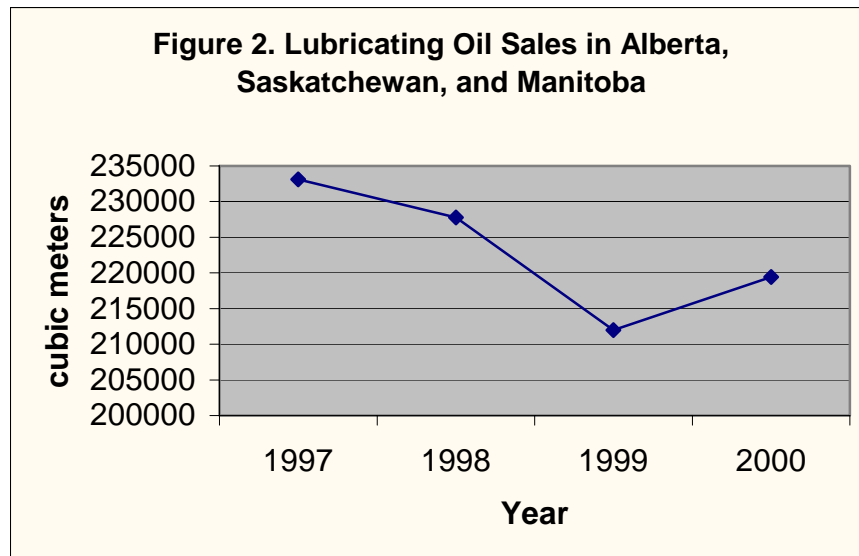
The potential for product design changes in response to the fee/refund system for oil and oil products is limited. Four possibilities seem to exist: (1) oil filters and containers could be made more recyclable; (2) filters and containers could be made lighter and less material intensive; (3) lubricating oil could be reformulated to make it easier and less costly to rerefine; and (4) oil could be made to last longer, and oil changes would thereby become less frequent.

Oil filters are recycled by steel recyclers into products such as rebar, wire, and nails. However, they must be cleaned of all remaining oil, by being crushed and shredded, before they are acceptable to a recycler. It is possible that an improved degreasing system could be designed or that the oil could be made easier to remove, or both. In addition, a shift to plastic or paper filters may be on the horizon for some kinds of filters; if these are more difficult to recycle, an EPR policy could provide some disincentives for this shift. Plastic containers are currently recycled into new oil containers and into a variety of other kinds of plastic products. As with filters, the oil must first be completely removed; thus any improvements to the product to make this process easier would be considered DfE. Finally, for lubricating oil itself, there are two issues. The first is the ease with which it can be used for combustion purposes or rerefined into new oil. The cost and difficulty of rerefining is related to the extent of oil additives, such as lead, chlorinated hydrocarbons, and sulfur compounds. The greater the amount of these contaminants, the more energy the rerefining process uses and the higher the costs. The second issue is the wear of the oil in its intended use. For example, in a motor vehicle, the longer the motor oil lasts, the greater the time between oil changes, and the less used oil generated over the life of the vehicle.

It is difficult to address those questions with available data. Experts who monitor and manage the program cite some design changes that have been made but hasten to add that the changes are unlikely to be attributable to the used oil program. Hambleton (2005) finds recent examples of oil companies that have removed lead from lubricants, switched to thin-wall plastic containers, and replaced paper labels with stenciling. However, he believes that these changes may be attributed as much to “competitive positioning in the marketplace” as to the used oil requirements. Driedger (2005) seems to concur, writing that the “drive for competitiveness in the marketplace” is the greatest factor, by far, behind product changes.

If drivers are waiting longer between oil changes or if oil is more efficient at performing its functions, then it is possible that less oil is produced and sold as a result. In addition, the fee on oil sales should have some dampening effect on demand. Statistics on lubricating oil sales in

the western provinces is available from Statistics Canada. Unfortunately, the data from 2001 forward are unavailable for public use. Figure 2 shows a decline in oil sales for the years 1997–2000 for the provinces of Alberta, Saskatchewan, and Manitoba¹¹: oil sales in 2000 were 6 percent lower than sales in 1997. This decline occurred at a time when gross domestic product (GDP) in those provinces was rising. Baldwin et al. (2004) report that over the 1997–2003 period, Alberta’s real GDP per capita rose 1.4 percent, Manitoba’s rose 2.6 percent, and Saskatchewan’s 4.5 percent. Combined with rising populations, these per capita figures lead to even higher total GDP growth. Of course, attributing the falling oil sales figures to the used oil recovery program is pure speculation. Many other factors could contribute to the decline. The 2001–2005 figures, if available, would reveal whether the trend has continued.



Source: Statistics Canada. Table 128-0001, Demand of lubricating oils and greases in natural units.

B. The U.K. Packaging Waste Program

Section II above introduced the concept of tradable recycling credits as a possible economic incentive-based instrument that might be a flexible and cost-effective way of reducing waste while promoting DfE. No program exists in which producers of the targeted product generate the credits and products are sorted by brand at end-of-life. Although perhaps feasible for some groups of products, it would be difficult for others. To my knowledge, the United Kingdom has the only tradable credit scheme in operation, a system applied to packaging waste.

¹¹ Since British Columbia’s program started only in 2003, we omit B.C. oil sale numbers from the graph.

In this section, I describe the system and its achievements to date. The performance of the credit market is analyzed, along with the attainment of recycling goals. To the extent possible, I discuss information available on changes to packaging design in response to the regulations.

Program design. The United Kingdom adopted the EPR concept in its 1995 Environment Act. The government then passed the Producer Responsibility Obligations (Packaging Waste) Regulations in 1997 and the Packaging (Essential Requirements) Regulations in 1998. These regulations, which came into force on March 6, 1997, require producers to recover and recycle a specific percentage of their packaging waste each year, with the percentage rising over time. The targets are shown in Table 3.

Table 3. U.K. Packaging Recovery and Recycling Targets

	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008
Total recovery	38%	43%	45%	56%	59%	59%	63%	65%	67%	69%	70%
Recycling	7%	11%	13%	18%	19%	19%	59%	61%	63%	66%	66.5%

Sources: Salmons (2002), ENDS Environment Daily (2002), ENDS Report (2002), U.K. DEFRA (2003).

There are also separate targets set for each material. In 2004, the recovery targets ranged from 18 percent for wood to 21.5 percent for plastics and up to 65 percent for paper. The goal of the program is to meet the European Union packaging waste requirements. In 1994, the European Parliament passed Council Directive 94/62/EC on Packaging and Packaging Waste, which required member states to pass national legislation to reduce packaging waste; it also set minimum recovery and recycling targets that member states must meet. These EU targets are revised every five years. The most recent ones, passed in 2004 (2004/12/EC), established 2008 targets of 60 percent for overall recovery and 55 percent for recycling, with material-specific recycling rates that range from 15 percent for wood up to 60 percent for glass and paper. As can be seen from a comparison with Table 3, the U.K. targets are set slightly above the EU targets. This is because not all packaging is captured in the system. Obligated companies in the United Kingdom are only those earning more than £2 million per year. A recent analysis by the government suggests that the U.K. material-specific recycling targets may be altered somewhat to meet the 2008 EU goals. It has been proposed, for example, that the glass target be raised to 74 percent while the aluminum and paper targets be lowered slightly; the overall recovery target may be reduced to 68.5 percent (U.K. DEFRA 2005b).

The U.K. regulations divide producer responsibility into four categories—manufacturer, converter, packer/filler, and seller—and then apportion the recycling obligations to each group. The definition for each category is listed in Table 4, along with the percentage of the recycling requirements for which companies in each category are responsible. A packer/filler, for example, is responsible for 37 percent of the requirements; thus, if the minimum material recycling requirement for a given year is 18 percent (as it was for 2001), packers and fillers must ensure that 6.7 percent (0.37×0.18) of the packaging they handle gets recycled.¹²

Table 4. Definitions of Producers in the Packaging Chain and Recycling Obligations

		<i>Recycling obligations (effective Jan. 2000)</i>
Manufacturer	A person who manufactures raw materials for packaging	6%
Converter	A person who uses or modifies packaging materials in the production or formation of packaging	9%
Packer/filler	A person who puts goods into packaging	37%
Seller	A person who supplies packaging to a user or a consumer of that packaging, whether or not the filling has taken place at the time of the supply	48%

The United Kingdom began phasing in obligated companies in 1998, beginning with the largest companies—those with annual turnover of more than £5 million—and later, in 2000, incorporating companies with an annual turnover of more than £2 million. The £2 million threshold includes 88.6 percent of all packaging handled by U.K. companies (EC 2001).

Obligated companies can show that they have met recycling obligations themselves—usually by contracting with a reprocessor—or they can join a “compliance scheme” that will fulfill the obligations for them. Compliance schemes essentially serve the same functions as PROs in other European countries. For a fee, a scheme takes on all of a company’s requirements,

¹² Importers are responsible for the “rolled up” obligations of the packaging they import—that is, the percentages attributable to all activities such as manufacturing and converting that take place prior to importing. See U.K. DEFRA, The Scottish Executive, and The National Assembly for Wales (2003).

including agency fees, data submission, recycling/recovery, and certificates of compliance. In 2004, there were 23 registered compliance schemes operating across England, Scotland, and Northern Ireland, representing approximately 9,200 businesses (U.K. DEFRA 2005a).

Most companies join a compliance scheme rather than registering individually. In 2004, there were only 1,076 individually registered companies, and the number has fallen since the program began—from 22 percent of all obligated companies in 1998 to 17 percent in 2002 and approximately 10 percent in 2004 (U.K. DEFRA 2002, 2005a). Of the 23 compliance schemes, the largest is Valpak, which was the first operating compliance scheme in the United Kingdom. In 2004, 52.7 percent of all businesses registered with a compliance scheme were with Valpak.

To demonstrate compliance with the regulations, compliance schemes or obligated companies show that they hold enough packaging waste recovery notes (PRNs) to meet their recycling obligations. These notes are generated every time a tonne of packaging material is recycled. PRNs are material-specific; there are PRNs for glass, paper, aluminum, steel, plastics, wood (since 2004), general recycling (which covers textiles, cork, and miscellaneous other packaging materials), and energy recovery. The use of the notes was not prescribed in the regulations themselves; instead, it arose from Valpak's decision, in the early years, to contract out the collection of packaging waste to reprocessors. As evidence that a reprocessor had recycled packaging on behalf of Valpak, the government came up with the idea of PRNs. Though originally only for documentation, PRNs soon provided the "common currency" necessary for trading, an idea devised by Valpak (Salmons 2002).

PRNs are traded among obligated companies, reprocessors, and compliance schemes. An organized marketplace, the Environment Exchange, exists to handle trades, and an active spot market has developed. Although compliance schemes often sign long-term contracts with reprocessors to recover and recycle a specific amount of packaging, in many cases, they contract for some of their obligation and then purchase PRNs on the spot market to satisfy the remainder. PRNs serve a particularly useful function as a hedge against the risk that a firm contracted for an insufficient amount of recycling.

The mechanics of the system are as follows. Accredited reprocessors must submit quarterly reports to the government that state how many tonnes of packaging were reprocessed in the quarter. The government then issues blank notes to these reprocessors, who fill them out and issue them to obligated companies or compliance schemes. One PRN means that 1 tonne of material has been recycled. The reprocessors can issue PRNs only up to the amount of recycling that has been reported to the agencies. The tradable credits are thus generated by recyclers, not

by producers, and they are not brand-specific. Obviously, for a good such as packaging, it would be prohibitively costly to have a system where brands are sorted.

Packaging waste that is exported for recycling is issued a packaging export recovery note (PERN). PERNs are also considered acceptable evidence of compliance, equivalent to PRNs; only accredited exporters can issue them. In 2000, PERNS accounted for 10 percent of total recycling, 46 percent of steel, and 19 percent of plastic (Salmons 2002). Recently, exports have risen sharply; they accounted for 29 percent of total recycling in the second quarter of 2005 (Cartledge 2005).

PRN prices and trading volume. The extent to which obligated companies and compliance schemes rely on PRN purchases on the open market to meet their obligations varies across companies and across time. PRN trading volumes on the Environment Exchange have increased since trading began in 1998. Figure 3 shows the years 1998–2004; the number traded has risen every year with the exception of 2003. In 2003, the government somewhat unexpectedly left the recycling and recovery targets the same as in 2002. As a result, many firms and compliance schemes had PRNs to carry over from 2002 to 2003. Trading was lighter in 2003 as a result. There was also an impact on prices, as shown below.

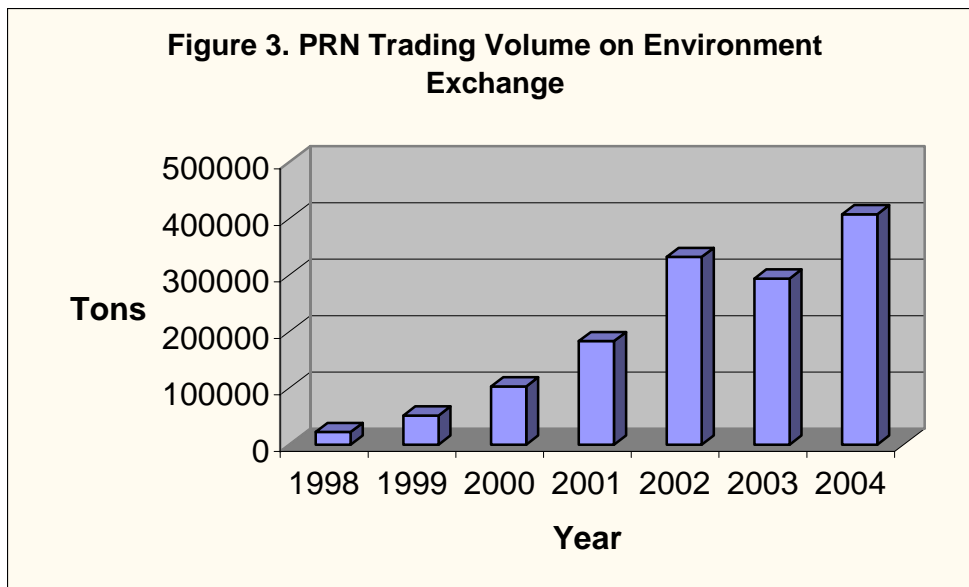
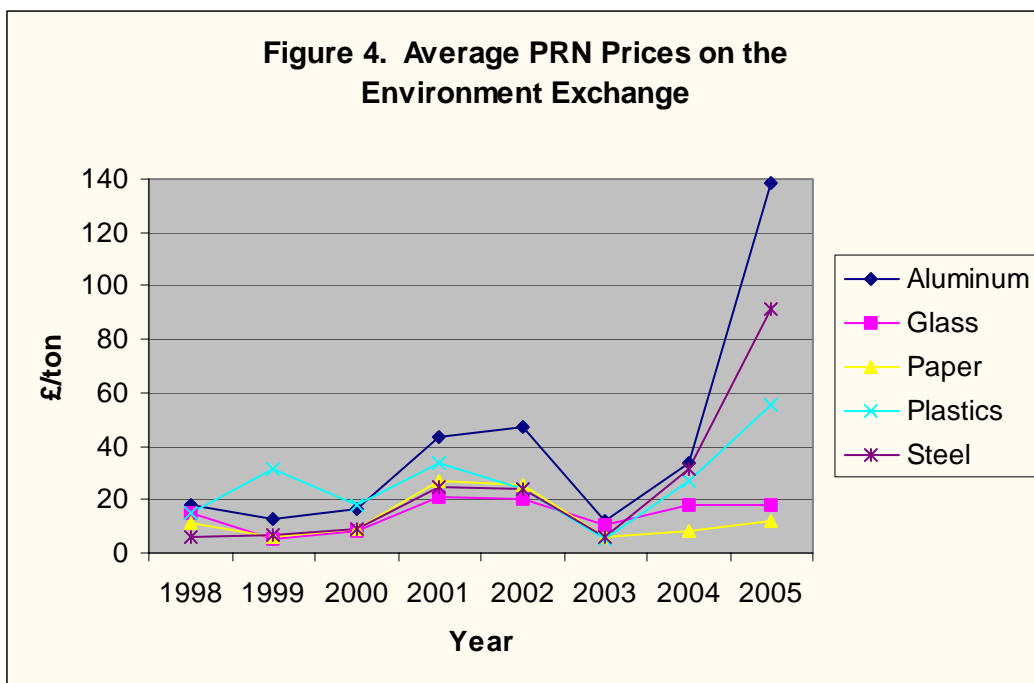


Figure 4 shows average PRN prices for 1998 through 2005. These are average spot market prices for trades made on the Environment Exchange.¹³ With the exception of plastics and steel, PRN prices dropped in the first year that PRNs were traded. Prices rose slightly in 2000, followed by a sharp rise in 2001. The EU deadline for meeting its packaging waste recovery requirements was 2001, and the U.K. material-specific targets increased significantly that year. Prices for all materials dipped sharply in 2003, when, as explained above, the government did not increase the recycling and recovery targets; with a carryover of PRNs from the previous year, there was less demand for PRNs. This led to a decline in prices for all materials. Prices rebounded in 2004, and aluminum, steel, and to some extent plastic PRN prices spiked even higher in 2005. The average aluminum PRN price for 2005 (through August) was \$138 compared with only \$34 in 2004; steel PRNs averaged \$32 in 2004 and \$91 in 2005.



Source: Environment Exchange website (<http://www.t2e.co.uk/>). Accessed September 16, 2005.

Although the Environment Exchange is open to all obligated companies and compliance schemes, most of the trading on the exchange—78 percent of all trades in 2000—has been by individual companies. These trades represented 11 percent of the companies’ total combined

¹³ These are averages, weighted by sales volumes; the 2005 prices are the average through August (see <http://www.t2e.co.uk/default.asp>).

obligation for that year. The 22 percent completed by compliance schemes in 2000 represented only 2.3 percent of their total combined obligation (Environment Exchange 2001), with the remainder met through contracted recycling.

There has been some concern about fraud in the PRN markets. The number of PRNs issued for wood and plastic grew substantially in 2002 with no apparent increase in the collection of these materials. Some reprocessors were suspected of issuing illegitimate PRNs, and this activity has been cited as contributing to the fall in plastic PRN prices in 2002. The government Advisory Committee on Packaging looked into these allegations of fraud and published two reports, one for wood and one for plastic. It was found that recycling of both wood and plastic was overstated. The figures reported in Table 5 below are the adjusted figures; original percentages were significantly higher. The committee concluded that the fraud in the plastic market was limited to a small number of reprocessors. For both wood and plastic, changes were made to reporting requirements and enforcement to prevent future occurrences. One compliance scheme, Valpak, believes that fraud is still an issue (see Valpak 2005).¹⁴ With the value that PRNs have on the open market, the government will almost certainly have to be constantly diligent in preventing fraud and enforcing compliance within the system.

Program results. Table 5 lists the recycling rates achieved in each of the years between 1998 and 2004. The table clearly shows that recycling and recovery have increased over the years since the regulations were passed. The overall recovery rate has risen 68 percent, and material-specific recycling rates have jumped between 45 percent and 137 percent. The overall recovery rate, however, has fallen short of both the U.K. and the EU targets in every year. In 2001, for example, the EU recovery target was 50 percent and the U.K. target 56 percent; actual recovery was 47.9 percent. The EU recycling target of 25 percent, on the other hand, was easily reached in most years, and many of the material-specific recycling rates surpassed the EU's previous 15 percent targets. The new 2008 EU goals—60 percent recovery and 55 percent recycling—may prove more difficult, however.

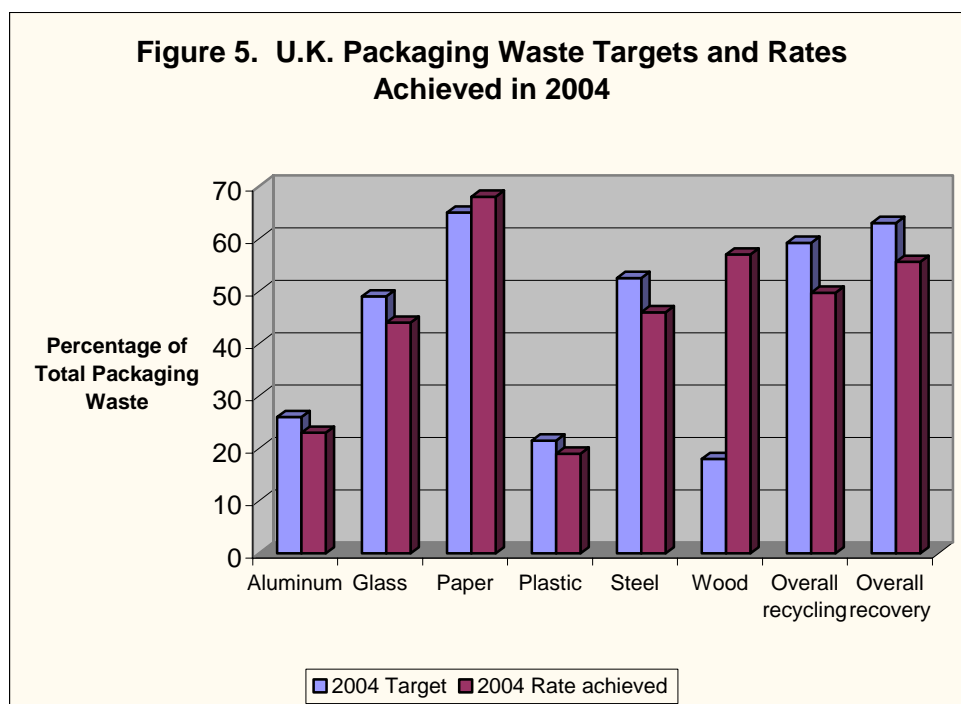
¹⁴ Only accredited reprocessors can issue PRNs. The Valpak (2005) report also contends that the government needs to accredit more small reprocessors who are legitimately recycling but not allowed to be part of the system.

Table 5. U.K. Packaging Generation, Recovery, and Recycling Rates

	<i>Recycling and recovery rates (percentage of total waste stream)</i>						
	<i>1998</i>	<i>1999</i>	<i>2000</i>	<i>2001</i>	<i>2002</i>	<i>2003</i>	<i>2004</i>
Material-specific							
Aluminum	13%	14%	15%	24%	22%	23%	23%
Glass	23	27	33	33	34	37	44
Paper	47	47	49	53	59	65	68
Plastic	8	12	12	16	20	18	19
Steel	25	30	32	37	42	44	46
Wood	—	—	—	57	54	54	57
Total recycling	29	33	36	42	44.5	46.8	49.7
Total recovery	33	38	42	47.9	50.4	52.7	55.6

Sources: U.K. DEFRA (2001), Meacher (2002). Material-specific rates are rounded to nearest whole number. Wood was not covered in the system until 2001.

Figure 5 shows the U.K. packaging waste targets and rates achieved in 2004. The target for total recovery was 63 percent, with 59.2 percent recycling; actual rates achieved were 55.6 percent (recovery) and 49.7 percent (recycling). With the exception of paper and wood, none of the material-specific recycling targets were met.



Source: U.K. DEFRA (2005c).

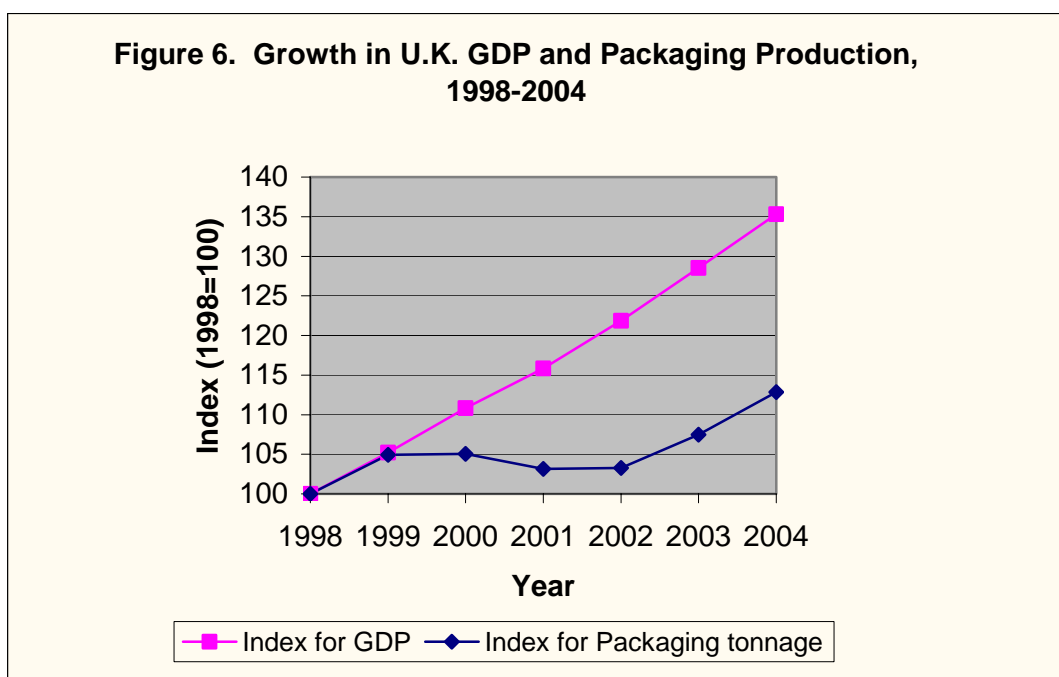
The extent to which the tradable credit system for packaging in the United Kingdom is able to spur DfE is limited: no incentive is provided for firms to make their packaging more recyclable, since they would incur the cost of those design changes while the benefit would be reaped across the marketplace. With no brand sorting and no tracing of products' recycling costs to producers, there can be no strong incentives for producers to increase recyclability. Moreover, only limited design changes can be made to improve packaging recyclability. Packaging is a relatively simple product; changes would be limited to material substitutions, labeling changes, less mixing of materials in a given package, and so forth.

The U.K. system does provide incentives for downsizing, some material switching, and dematerialization, however. The lower the packaging tonnage an obligated business has in a given year, the lower its recycling requirement and the fewer PRNs it must hold. Thus, we should see some reductions in packaging tonnage. For materials that are more difficult to recycle, and for which the PRN price is therefore higher, we should expect to see the largest reductions. It is difficult to draw conclusions from the data, however, because there is no counterfactual. In other words, one does not know what packaging would have been in the absence of the regulations. Producers make changes for many reasons, including cost savings and consumer preferences, and thus it is difficult to attribute changes solely to the regulations.

Figure 6 shows an index for gross domestic product in the United Kingdom for each of the years 1998 to 2004, along with an index of packaging volumes over the same time period. The indexes are constructed for both series using 1998, the first year the regulations were enforced, as the base year. The figure shows that the economy grew and packaging production grew with it over the seven-year period. However, the growth in packaging was far less than the growth in GDP overall. In addition, total packaging volumes fell between 1999 and 2002, rebounding in 2003—the year that the government did not increase the recycling targets—and 2004.

According to government officials and other observers of the system, anecdotal evidence suggests that there have been changes in some packaging in response to the system. Biott (2005) reports that reduction in the weight and size of yogurt containers is due to the regulations and also points out that because the regulations promote reuse of packaging—the recovery obligation only applies to the first use of a package—there has been a switch from one-use cardboard containers at supermarkets to reusable plastic pallets and crates. A recent study by Sturges et al. (2004), funded by the Packaging Federation, a compliance scheme, and several companies representing the waste management and packaging industries, includes five case studies of packaging material and energy use. The authors find that although U.K. demand for soft drinks

increased 20 percent over the 1997–2002 period, packaging per liter of soft drink consumed decreased 21 percent. They also establish that material use for cat food containers fell 38 percent between 1993 and 2002 at the same time that cat ownership in the United Kingdom increased sharply. And in a third case study, of laundry detergents, the authors found that the amount of detergent packaging per wash dropped from 11.4 grams in 1999 to 10.6 grams in 2002, a reduction of 7 percent. Much of this last reduction could be attributed to a shift in the type of product on the market: detergent capsules, for which packaging is minimal compared with standard liquids and powders, were introduced in 1998, and their market share has greatly increased.



The Sturges et al. study and the other information provided above do not necessarily mean that the regulations are directly responsible for the changes observed. It is possible that some of the changes would have taken place anyway. The best one can hope to do is document the changes and emphasize the need for better data collection efforts to provide more information on this topic.

Moreover, even if the changes reported here are due, at least in part, to the regulations, it is difficult to know how much the tradable credit aspect of the program—the feature that distinguishes the U.K. EPR program from many others—has contributed to the changes. The fewer tonnes of packaging material used, the smaller the recycling obligation. The smaller the recycling obligation, the lower the financial cost of meeting the program requirements. The PRN

market gives obligated companies and compliance schemes an alternative means of meeting requirements—they can contract with a reprocessor but they can also simply purchase PRNs on the open market. It is possible that not having to lock into a contract for a specific amount of recycling would provide a further impetus for firms to reduce material use, but it is extremely difficult to know whether this is the case. Economic theory tells us that the tradable credit aspect of the program should reduce the overall program costs because it gives producers flexibility to meet their obligations. However, offsetting these cost reductions could possibly be some increased transaction costs on the part of businesses and extra enforcement costs incurred by the government. It is beyond the scope of this study to address the overall issue of program cost-effectiveness in comparison with an alternative system without trading, but this would be a useful topic for future analysis.

C. The Dutch Waste Electronics and Electrical Equipment (WEEE) Program

The Netherlands became the first country in Europe to introduce the EPR principle for a wide range of electronic and electrical equipment when the Management of White and Brown Goods Decree was passed in 1998. This decree mandated that retailers take back old electronic and electrical goods in exchange for new ones and that manufacturers accept those products from retailers and arrange for transportation and recycling. Municipalities were also required to take products back free of charge. Recovery and reuse targets were negotiated with industry and varied across products, ranging from a high of 75 percent for refrigerators down to 45 percent for small appliances.

The legislation allowed visible up-front fees to be charged on products to cover recycling costs. In the case of household appliances, stereos, and televisions—“white and brown goods”—the PRO that has managed collection and recycling for most producers, NVMP, has charged fees ranging from zero for small household products up to \$17 for refrigerators and some televisions (Future Energy Solutions 2003). Computer equipment, or “gray goods,” has been handled by a separate PRO, ICT-Milieu, formed by computer equipment manufacturers and importers. The financing of the computer system was different. Instead of an up-front fee covering recycling costs, computer equipment recycling costs prior to 2003 were covered by payments made in arrears by manufacturers and importers after the products were processed. Processors sorted computer equipment by brand name and assessed differential fees based on recycling costs. The fees charged were weight-based. Because sorting and invoicing based on brand-specific recycling costs were quite expensive and there was a high percentage of “orphan” products on which fees could not be assessed, ICT changed the financing arrangement in January 2003

(Veerman 2005; Future Energy Solutions 2003). A third-party organization was hired to obtain confidential sales data from member companies and issue invoices based on market share of sales. This is the usual PRO financing method that has been applied to packaging and other products in several countries. The costs associated with orphan products were then apportioned to members based on market share.

Dutch retailers faced with storage difficulties and the high costs of handling a wide range of used products often offered discounts to customers buying new products if they did not ask to drop off an old one. As a result, more than 80 percent of returned equipment was collected by municipalities and only about 20 percent in the retail take-back schemes. Collection and sorting depots exist around the country for drop-off services; the Dutch PROs share the costs of operating these facilities.

Volumes of WEEE collected have risen since the program began, and recycling rates achieved have been high, at least for white and brown goods. Table 6 shows annual volumes of WEEE collected per capita for each year since the program began, 1999, through 2003. The EU target of 4 kg/person was reached in 2001 and has been maintained since then. Information on preprogram recycling rates for WEEE appears to be unavailable, but rates for some years are available through the PRO that handles the bulk of the white and brown goods collection and management, NVMP. Figure 7 below shows recycling rates for selected white and brown goods for the years 2000 and 2001. The targets were exceeded in both years. For example, the recycling rate for refrigerators was 86 percent in 2000 and 85 percent in 2001, compared with a target of 75 percent. Small appliances were recycled at a 60–64 percent rate in the two years, compared with a target of only 45 percent.

Table 6. Waste Electronic and Electrical Equipment Collected for Recycling in the Netherlands, Per Capita

<i>Year</i>	<i>Average WEEE collected per inhabitant (in kgs)</i>
1999	2.26
2000	3.94
2001	4.66
2002	4.82
2003	4.69

Source: Dutch Industry Association for Audio and Visual (2004).

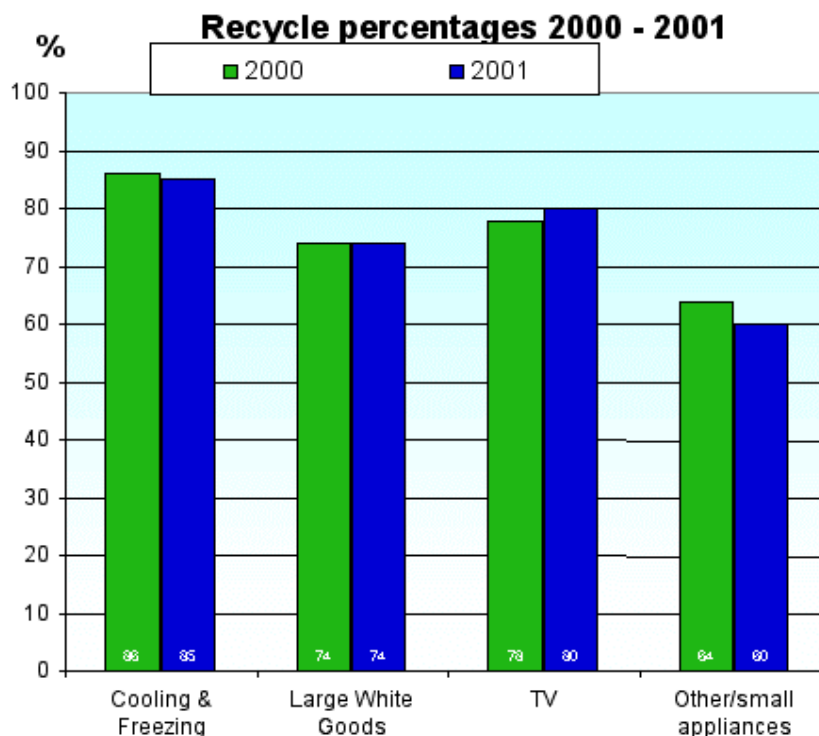
Less information is available about information technology (IT) equipment—computers and other gray goods—than about white and brown goods, but it appears that the white and

brown goods sector achieves higher recycling rates and is responsible for bringing up the collection figures reported in Table 6. Future Energy Solutions (2003) reports that although collection of IT equipment by ICT-Milieu increased between 1999 and 2002—the total amount collected in 2002 is three times the 1999 level—the per capita figure for 2002 is only 0.58 kg. Recycling rates calculated by ICT-Milieu may not be comparable with those from other programs and other countries. An 89 percent rate for 2002 apparently was obtained by subtracting materials that cannot be recycled from the total amount of available materials before making the percentage calculation (Future Energy Solutions 2003). I was unable to locate any reliable recycling rate figures for IT equipment from the Dutch government.

No good information is available on product design changes, such as improved recyclability or dematerialization, that might have taken place in the Netherlands in response to the WEEE programs. The financing of the IT program before 2003 could have revealed some impacts, since costs were apportioned to producers based on costs of recycling. However, the program was phased out because it was too costly—a telling result in itself—and was probably not in place long enough to provide any incentives to producers. A Dutch government official who has monitored the WEEE EPR program for a number of years has stated publicly that although he considers the program a success in terms of increasing recycling and reducing waste disposal, it provides little, if any, incentive for DfE (Veerman 2004, 2005). One reason for his conclusion comes from the apparently small differential in recycling costs across brands, particularly of televisions.

The White and Brown Goods Decree was supplanted in 2005 by new requirements embodied in the WEEE Management Regulations. These new Dutch regulations primarily enforce the EU's WEEE directive (2002/96/EC and Amendment 2003/108/EC). Retail take-back and free municipal collection are still in place, since a principle of the EU directive is free-of-charge options for end users. New higher recovery and recycling targets are set for most products in the Netherlands to meet the EU targets. Large household appliances must meet an 80 percent recovery rate and a 75 percent recycling rate (by weight). Computer equipment must meet a 75 percent recovery rate and 65 percent recycling or reuse rate. Small household appliances, toys, tools, and other miscellaneous WEEE are subject to slightly lower rates: 70 percent recovery and 50 percent recycling (by weight). The WEEE directive also includes a collection target. By the end of 2006, an average of at least 4 kg of waste electrical and electronic equipment per inhabitant per year must be collected from private households. A new target rate to be met by the end of 2008 has yet to be established.

Figure 7. Dutch Recycling Rates for Selected Products in 2000 and 2001



Reprinted from NVMP (2005) Facts and Figures: Recycle Percentages. Available at http://www.nvmp.nl/html/10_english/05_fr_ff.htm.

The financing of the system has also changed. In keeping with the EU directive requirements and the spirit of EPR, producers are responsible for covering the waste management costs of equipment they put on the market after August 13, 2005. Equipment put on the market prior to that date must be paid for by current producers in proportion to their current market share. Thus, for the next few years, as the backlog of equipment in use gradually becomes obsolete, the waste management costs will be covered by producers in proportion to their current sales. In a long-run equilibrium situation, producers would cover the costs of waste management for their own products. Visible fees are not allowed on items put on the market after August 13, 2005.

There is some debate among policymakers and analysts, and also among members of industry, as to whether a visible fee is a good idea. The fee serves two purposes. First, it provides incentives to consumers to reduce consumption and thereby reduce waste; also, it can lead producers to reduce product weight and material content if it varies with product weight. Second,

it provides revenues to cover the costs of collection and recycling. Although the first point is more important from a policy cost-effectiveness and economic efficiency standpoint, the latter is often the focus of discussions among policymakers and industry.

From a purely neoclassical economics standpoint, a higher price for a product should lead to reductions in the quantity demanded regardless of whether the higher price results from a visible or an invisible tax. Thus, the fee itself—visible or invisible—will help reduce waste. However, there could be some value in terms of information provision if the fee is visible. By highlighting that particular component of overall product cost—that is, the end-of-life management costs—showing the fee could provide a stimulus for getting costs down over time. Also, fees that varied across products or brands could lead to differential purchasing decisions that could be beneficial for reducing waste.

Although the principle of the new Dutch requirements is individual responsibility for the waste generated from products, in actuality, PROs are likely to continue to manage collection, transport, and recycling for obligated producers. The important thing, then, is how the PROs choose to charge producers for these services. As it currently stands, NVMP sets separate fees for classes of products but not for individual brands. And a producer's obligation is based on its current sales volume, not on the cost of managing its products at end-of-life. ICT-Milieu operates in a similar manner.

D. The Korean Waste Electronics and Electrical Equipment (WEEE) Program

Korea applied the EPR concept to several different waste streams beginning in 2003. The products covered include tires, lubricants, fluorescent lights, metal cans, glass and PET bottles, and 15 types of electrical and electronic equipment.¹⁵ Additional computer equipment will be added to the system in 2006. Available documentation about the program suggests that it operates like many of the European take-back programs—that is, the government sets mandatory take-back and recycling requirements for each product, and producers pay fees to join a PRO that handles all of the collection and recycling obligations. Financial penalties are assessed on producers that do not meet their obligations. Park (2005) reports that about 60 percent of used

¹⁵ See <http://www.epr.or.kr/eng/produce/products.asp> for the full list and more information about the program. See also the Korean Ministry of Environment website and its discussion of the program: http://eng.me.go.kr/user/policies/policies_view.html?msel=b5&seq=5&filename=5_waste_01.html&table_name=me_new_waste.

“large-sized” electrical and electronic equipment is collected by retailers and about 30 percent by local authorities.¹⁶ In addition to the EPR program, “waste dues” are levied on particular products and the revenues used to pay for local government collection and recycling efforts. Yeo (2005, 7) explains that the EPR system replaced a kind of deposit-refund approach in which “products subject to the system [were] selected based on the degree of environmental pollution stemming from their mass production, the facility of recycling, and the probability of motivation to actually collect and dispose them.” Yeo further reports that “reimbursable amounts [were] calculated on the basis of collection and disposal performances.” A deposit-refund program for reusable glass alcoholic beverage and soft drink containers is still in operation; however, the approach was considered ineffective for most products, since recycling rates were relatively low.

The Korean EPR program has not been in operation long enough to allow full assessment of its performance. However, a recent presentation by the director of the Resource Recycling Division of the Korean Ministry of Environment reports recycling rates and other information for WEEE (Park 2005). In the first year of the program, 2003, the recycling targets were exceeded for televisions and personal computers but were just missed for refrigerators, washing machines, and air conditioners. Figure 8 below reprints the graph of recycling volumes from Park (2005).

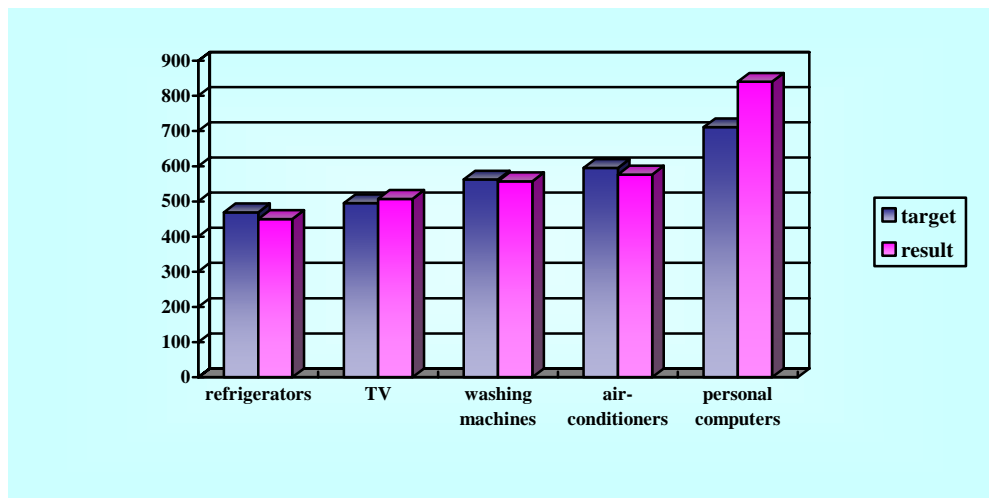
Producers in the Korean system have collected and recycled much more waste since the EPR program was enacted, with the largest increase occurring for personal computers. Producers collected and recycled only about 30,000 computers in 2002; the number exceeded 250,000 in 2003 (Park 2005). Of the total WEEE collected and recycled in 2003, 59 percent was handled by producers and 30 percent by local authorities. That compares with 2002 figures of 49 percent and 38 percent, respectively.¹⁷ Finally, the sheer volumes of WEEE recycled have increased significantly. In 2002, 1,418,000 units of electronic and electrical equipment were recycled; in 2003, 1,965,000 units were recycled (Park 2005).¹⁸ That represents an increase of 39 percent, and the 2002 figures in turn were substantially higher than those of the three previous years.

¹⁶ The remaining 10 percent is attributable to reuse after repairs.

¹⁷ The remaining percentage is handled by associations for product reuse.

¹⁸ Using units as the performance metric, as Park (2005) does, rather than pounds of material may be problematic. Electronic products are made up of many differential materials and components, and it is unusual for the entire item to be fully recycled. Individual products are likely to differ in the extent to which they are recycled, and thus adding units could be misleading. Also, items differ greatly from each other in their weight and thus in the amount of material diverted from a landfill, even if each item is fully recycled. Performance of the Korean system would be more effectively assessed with a weight-based metric.

Figure 8. Korean Recycling Targets and Outcomes in 2003



Unless producers foresaw the EPR system on the horizon well before 2003, it is unlikely that the program could have had a discernible effect on product design in such a short period. Yeo (2005) describes some changes that electronics producers have made to products that have reduced their environmental impact, but it seems unlikely that these changes can be attributed to the EPR program. Yeo reports that Samsung and LG Electronics have reduced the weight and material content of their televisions, though over what time period these accomplishments were made is unclear. In the same study, Yeo states that the number of equipment circuit parts in hard disk drives has been reduced by 11.7 percent and the weight of the product reduced by more than 30 percent. However, these decreases occurred between 2001 and 2002, before the EPR program was enacted. Improvements in vacuum cleaners are also documented in Yeo, but these also took place before the EPR program was in place. Such examples simply highlight the difficulty in attributing product design changes to government policy. They also highlight the problems with looking at anecdotal information for individual companies. Ideally, one would like industry-wide measures of weight reductions and product recyclability metrics for time periods well before government policy was enacted and for a reasonable length of time after. In addition, a wealth of other control variables would be needed to isolate the changes due to EPR policy.

E. Other EPR Programs: Packaging in Germany, Vehicles in Japan

As mentioned in the introduction, Germany began the EPR revolution in 1991 with its now-well-known packaging ordinance. The law required that producers and retailers take back

the packaging associated with products and ensure that specified recycling rates be met for each material. These rates range from 60 to 75 percent. Early in the program, producers formed a PRO to jointly handle collection and recycling. This PRO, the Duales System Deutschland (DSD), has been in operation ever since and has spread its operations to other countries. As of January 1, 2006, the licensing fees it charges its members range from €0.076/kg (\$0.093/lb) for glass to €0.18/kg (\$0.22/lb) for paper and paperboard up to €1.35/kg (\$0.75/lb) for plastic.¹⁹

The DSD and the German government, among others, have written many testimonials to the success of the German packaging program. Recycling rates are high and material use has dropped dramatically since the program began. The Öko-Institut (2002) reports that packaging volumes in Germany declined 4 percent over the 1990–1999 period, while packaging volumes in the Netherlands, which relied on a voluntary recycling program during that period, rose 15 to 20 percent. Packaging recovery rates were reported to be 37.3 percent in 1991 at the time the law was passed; by 2000, the rate had risen to 76.7 percent with the material-specific recycling targets of 60 to 75 percent routinely met (GVM, 2003a). By contrast, the 2003 recycling rate for containers and packaging in the United States was 39 percent (U.S. EPA 2005).

In terms of DfE, the primary outcome from the German program is the reduction in packaging volumes, documented above, and thus material use. No other significant design impacts should be expected from the program. These reductions arise from the fees, which are weight-based and thus encourage producers to use less material. It is worth pointing out that it is highly likely that the same result would have been achieved with any other fee-based system and not necessarily one that relies on producer responsibility for collection and recycling. In U.S. states with deposit-refund systems for beverage containers, recycling rates for these containers average 62 to 95 percent (Palmer and Walls 2002). German studies have estimated that packaging for beverages declined 6 percent between 1991 and 2001 while beverage consumption rose 22 percent (GVM 2003b). However, this is not out of line with some U.S. findings; the American Beverage Association reports that the weight of soft drink containers has declined 30 percent since 1972 (American Beverage Association 2005). And as was pointed out in the discussion of the U.K. packaging program, material reductions for beverage containers have been documented in other countries. The point is that the Germany results are consistent with other findings in the case studies above: EPR leads to more recycling, and the fees that PROs

¹⁹ Fees are available on the DSD website (see http://www.gruener-punkt.de/Current_licence_fees.133+B6Jkw9MQ_0.html).

charge members generally lower material use. However, other policies besides producer take-back mandates—many of which fall under the EPR mantle—can yield the same outcomes.

Finally, it is worth mentioning that motor vehicles are an increasing focus of EPR legislation. In July 2002, Japan passed its Law for the Recycling of End-of-Life Vehicles, whose requirements went into effect in January 2005. Passenger cars and commercial vehicles are covered by the law, which requires manufacturers to collect and recycle shredder residue—the material left over after all recycling of metals and other materials—and collect and properly dispose of fluorocarbons and air bags.²⁰ Shredder residue must meet a recycling rate of 30 percent by 2005, 50 percent by 2010, and 70 percent by 2015. To help fund the system, deposits are collected from vehicle owners upon purchase of new vehicles and inspection of used vehicles.

On its Web site, Honda reports several efforts to reduce automotive shredder residue and meet the recycling rate requirements, as well as multiple DfE efforts for automobiles in general. For the first three months the law was in effect—January through March 2005—Honda maintains that it achieved a 53.2 percent shredder residue recycling rate (see Honda 2005), well above what was required by the law. A portion of this recycling rate was reached by generating less shredder residue in the first place; in other words, Honda recycled more of the materials in its vehicles, leaving less residual to manage. Other DfE efforts reported by Honda include using more easily recycled materials in instrument panels and bumpers, making bumpers easier to dismantle, developing markets for reuse of bumpers, and downsizing side protector braids (see Honda 2006). Many of these activities were taking place well before passage of the vehicle recycling law and thus cannot be attributed directly to the EPR policy. Moreover, possible cost savings from many of these changes could motivate Honda to undertake them. For one thing, automobile shredder residue disposal costs are quite high in Japan, particularly in comparison with Europe and North America (Kanari et al. 2003).

The European Union issued its End-of-Life Vehicle Directive in October 2000. Beginning in 2006, each member country must have its vehicles meet an 85 percent recycling rate, with 5 percent allowed as energy recovery; in 2015, the recycling rate target rises to 95

²⁰ Materials in vehicles are highly recycled, with overall recycling rates in most countries around 75 to 80 percent. However the 20 to 25 percent of material left over—auto shredder residue—is difficult to deal with, sometimes hazardous, and has few secondary uses. The new laws in Japan and elsewhere are attempting to force manufacturers to address this problem.

percent. The directive also mandates that car owners be able to return vehicles for recycling free of charge. In contrast to the Japanese law, the EU directive mandates that a specific percentage of vehicles be recycled; in Japan, the law applies to the shredder residue.

As with waste electronics laws and regulations, vehicle policies are too recent for us to see much impact yet on waste volumes, recycling, or DfE. Both products will be interesting to study in the future, however. Unlike packaging and some other products, these are complicated, highly engineered products made with multiple materials. The potential for DfE is therefore much greater than with other products, and thus EPR could have its greatest impact on these products.

V. Conclusions

In this study, I have described the features of EPR policy instruments and compared and contrasted specific instruments that might be used to achieve EPR goals. These instruments range from economic incentive-based approaches, such as a combined ARF and recycling subsidy, to product take-back mandates. I first discussed findings from the economics literature about the impacts that should be expected, in theory, from these instruments and their likely costs. The second part of the study analyzed several existing EPR programs in detail. To the extent possible, I tried to identify any product design changes that have taken place in response to the programs. It is very difficult to identify and document DfE in practice, and nearly impossible to definitively attribute to EPR policy any changes that are identified. Moreover, anecdotal evidence from individual firms, while interesting, does not provide industry-wide evidence that policies are having a widespread DfE impact. And given the multiple policy instruments in play, targeting hazardous substances, waste reduction, and other outcomes, one needs to take care in comparing observed outcomes across a range of dimensions. In this study, I have considered the design changes that could take place given the incentives provided by the individual instruments and then tried to determine whether such changes actually occurred. Unfortunately, evidence is limited. Better documentation of industry-wide impacts on material use and product recyclability over time is needed.

Despite those limitations, I am able to draw some conclusions from the research and determine the need for more work in the future. The following are the main conclusions from the study.

1. EPR programs have proven themselves in terms of reducing waste associated with consumer products and increasing recycling rates; in all countries with such programs, documented increases in recycling have occurred.
2. A limited form of DfE has already taken place in response to policy in many instances: reductions in material use and product/packaging downsizing.
3. These accomplishments can be, and have been, achieved through several means. Traditional product take-back mandates combined with recycling rate standards are one way, and in virtually all of these programs, collective implementation with a PRO is the rule, but combined ARFs/recycling subsidies lead to the same outcomes and may be more cost-effective.
4. It is too early to say whether more complex forms of DfE for highly designed and engineered products such as electronics and motor vehicles can be encouraged with EPR policy, but it seems unlikely that large changes will result from the types of policies we currently see in place. In particular, PROs, as they currently operate, provide very little incentive for members to engage in DfE.
5. Policies that directly target DfE—individual firm take-back programs and/or fees and subsidies that vary with product characteristics—are likely to be very costly and difficult to implement and enforce. Nonetheless, research into, and experimentation with, such policies may be useful.
6. Policymakers need to keep in mind that multiple policy instruments are necessary for efficiently accomplishing multiple environmental goals. One instrument cannot, for example, efficiently reduce the hazardous constituents of products and also reduce waste volumes, and comparisons between such instruments should not be made.
7. As more experience is gained with electronics and vehicle programs, more data and information should be systematically collected from all participants so that the programs can be better evaluated. This information should include data on material use (both types and volumes), product weight, ease of dismantling, labeling, and more. It will always be difficult to attribute observed changes to policy, but a first step is to collect better data.

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