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EPR Programme Implementation: Institutional and Structural Factors

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EXECUTIVE SUMMARY

The use of EPR programmes among the OECD Member States has been increasing since the early 1990's.

In light of the growing use of extended producer responsibility (EPR) as a principle that underpins environmental policy instruments among OECD Member States since the early 1990's, this paper attempts to draw some lessons from the implementation of EPR programmes to date. The paper evaluates the results of different types of EPR programmes, analyses the institutional and structural factors that influence the results and the measures to overcome barriers, and suggests what types of products are most suitable for certain types of EPR programmes. The paper was written for the OECD Seminar on Extended Producer Responsibility: Programme Implementation and Assessment held in December 2001.

This paper evaluates the results of the implementation of the most commonly used EPR programmes, i.e.., for packaging, small consumer batteries, electrical and electronic equipment and cars.

The focus of the evaluation is on four product groups for which EPR programmes have been used widely among the OECD member states: packaging, small consumer batteries, electrical and electronic equipment (EEE) and cars. Approximately 20 programmes have been evaluated, which have been implemented f0or a relatively long time and have taken different approaches (e.g. style of enforcement, responsibilities given to the producers).

The programme results that were evaluated include: collection, reuse and recycling rates; the stimulation of innovation (e.g. design for reuse/recycle/end-of-life management, reduction of toxic substances at source, change in a product system); the costs of implementation; soft effects (e.g. capacity building, generation and diffusion of information, improved communication between the upstream and the downstream); approaches to overcome barriers such as existing and orphaned products and free riders. Because different programmes use different methods to calculate collection/reuse/recycling rates and provide for different end-of-life management steps through fees that are collected, the results are difficult to compare.

Various characteristics of the products affect the management of the EPR programmes and their effectiveness in reducing environmental impacts.

When discarded, products selected for evaluation pose threats to the environment and human health due to their high volume (e.g. packaging, cars), and/or hazardous substances in their parts and materials (e.g. batteries, cars, EEE), making it difficult and costly for conventional waste management facilities to handle them properly. EPR programmes have been used both for durable, complex products (e.g. cars, EEE) and for non-durable simple products (e.g. packaging), and compared to the latter, the former add difficulties in the management of EPR programmes. EPR programmes are most effective in reducing waste generation and increasing recycling where there is a potential for design changes of the product that can reduce the costs of recycling.

In determining the scope of an EPR programme, the consumers' ability to distinguish the difference between the products covered by an EPR programme and those not covered (e.g. different types of batteries) should be considered. An EPR programme that covers all similar products may help avoid confusion and free riders. When a product covered by an EPR programme contains products covered by other EPR programmes (e.g. tyres in cars), governments need to either co-ordinate the coverage of these products, or delay implementation of a programme to avoid an overlap.

Voluntary programmes seem to work best when the product contains a high amount of valuable resources at post consumer stage (e.g. cars), however they suffer from

Mandatory programmes give better results when the products are not profitable to recycle.

free rider problems. Involvement of governments in an EPR programme, on the other hand, reduces free rider problems, achieves higher collection/reuse/recycling results, and stimulates design for end-of-life management in the absence of consumer demand for better management. (e.g. cars, EEE). Among the programmes evaluated, there is a definite shift from voluntary initiatives of producers to the programmes where governments are involved (either mandatory legislation or negotiated agreements).

Mandatory numerical collection/reuse/recycling targets have been effective in achieving higher results.

Mandatory numerical targets have been effective in achieving high collection/reuse/recycling rates. While collection targets help increase separate collection from the rest of the waste stream and reduce littering, reuse and recycling targets drive design changes and technical improvement, leading to the reduction of environmental impacts of discarded products not only at their end-of-life, but also at during production. Due to the uncertainty as to when a product comes to its end-of-life and due to the long life of some products, it is difficult to set collection targets for durable products (e.g. EEE, cars).

Substance/landfill bans drive product re-design and development of alternative substances.

Substance/landfill bans and the mandatory achievement of recycling rates for specified materials have been powerful components of some of EPR programmes that trigger product re-design and development of alternative substances. Threat of a ban often encourages increased collection and recycling (e.g. batteries).

A successful collection system must attract consumer participation.

The establishment of a successful collection system is the prerequisite for a successful EPR programme. Achievement of high collection rates requires 1) adequate financial incentives for consumers, 2) convenience for consumers (e.g. size, weight and ease of handling of the discarded products, distance to the waste bins) and/or 3) information to consumers. Particular problems with a conventional deposit-refund system can be overcome by combining the deposit-refund system with an advance disposal fee system. Some retailers participating in the collection of old products experienced an increase in the number of customers.

Systems surrounding the product, including existing infrastructure, knowledge and skills, should be considered when establishing an EPR programme.

In introducing an EPR programme, issues such as the number of producers and distributors that exist in the market, the financial and physical capacity of the individual producers to establish and manage the end-of-life management system of their products, the number and capacity of existing end-of-life managers in the market, must be considered. Use of an existing physical infrastructure, skills and knowledge for collection and recycling (e.g. local governments, retailers, recycling facilities) facilitates fast and efficient implementation of an EPR programme. The ownership and management of existing infrastructure can be adjusted for EPR programmes.

Financial mechanisms for durable complex products pose more challenges than those for non-durable, simple products.

For non-durable, relatively simple products (e.g. packaging, some batteries), producers often organise a collective collection and recycling infrastructure. The properties of these products allow the advance fees on new products paid into a collective financial system to reflect actual collection and recycling costs of the products sold.

Individual financial responsibility presents an important opportunity to stimulate design changes.

Properties of durable, complex products (e.g. EEE, cars), on the other hand, make a collective financial system ineffective at stimulating design change. Individual financial responsibility presents an important opportunity to stimulate design changes that ultimately minimise the cost of recycling, but it fails to address orphaned products and requires an appropriate collection system where brands and properties of collected products can be easily distinguished. A last-owner-pays system, when coupled with individual physical responsibility, can be an effective measure to promote design change, but create disincentives for collection.

Further research is required for improving the evaluation of EPR programmes.

In order to improve the evaluation, consistent measuring and reporting of performance level and costs of EPR programmes is necessary, requiring additional research and/or co-ordination among the different EPR programmes. Further research is also required on the influence of different types of EPR programmes on eco-design and innovation.

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List of abbreviation

ARN	Auto Recycling Nederland
ASR	auto shredder residue
DSD	Duales System Deutschland
EEE	electrical and electronic equipment
ICT	information and communication technology
PRO	Producer Responsibility Organisation
RBRC	Rechargeable Battery Recycling Corporation

1. Introduction

Extended producer responsibility (EPR) is an environmental policy approach in which a producer's responsibility, physical and/or financial, for a product is extended to the post-consumer stage of a product's life cycle (OECD, 2001). Since the early 1990s, a number of countries have begun to incorporate the concept of EPR into their environmental policy for management of products in the waste stream. Currently, EPR programmes have been implemented for product groups such as packaging, batteries, automobiles, solvents, paper, plastics, tires, carpets, and electrical and electronic equipment ("EEE"). The range of product groups is expanding, with products such as office stationery and furniture being considered. The programmes have been implemented as mandatory legislation or regulations, negotiated agreements between the government and producers, and voluntary initiatives by producers.

Considering the growing use of EPR as a principle that underpins environmental policy instrument among OECD Member states, it is important to evaluate the results of EPR programmes that have been implemented and to understand the factors that have influenced these results. Moreover, implementation of EPR programmes can be hampered by a variety of institutional and structural barriers, and the manners in which these barriers have been overcome by existing EPR programmes can be instructive for the development of new programmes. Finally, as more types of products are considered for EPR approaches, it is important to understand how the characteristics of the products can affect the results of an EPR programme.

The purpose of this paper is to attempt to draw some lessons from the implementation of EPR programmes to date and to address the following questions:

- What are the results of the implementation of different types of EPR programmes, and how are the results influenced by the institutional and structural factors?

- In implementing EPR programmes, how have institutional and structural barriers been overcome?

- How do the characteristics of the product impact the design and implementation of EPR programmes, and which products are most suitable for certain types of EPR programmes?

In reviewing existing EPR programmes, the paper will focus on the product groups for which EPR programmes have been used widely among the OECD countries. These include packaging, electrical and electronic equipment, automobiles, and small consumer batteries[1]. Selected EPR programmes that take different approaches (e.g. style of enforcement, responsibilities given to the producers) for these four product groups will be examined. Programmes for other products, such as carpets and tyres, will be discussed briefly wherever appropriate. Aside from the programmes of individual OECD countries, EU policies and Directives for the product groups mentioned above will be assessed, based on the influence the Directives have on the formulation /revision of national legislation of the member states of the European Union, which constitute a large number of the OECD countries.

The intention of this paper is not to present or discuss results from all the OECD countries implementing EPR for the four product groups. Many of the programmes are so new that there has not been enough time to determine the results. In some cases, the results discussed are based on the

[1] Round cell and button cell batteries. In other words, batteries that are used by private households but exclude car batteries.

anticipation of such programmes, and some of the data, such as achieved collection, reuse and recycling rates and costs of implementation, are limited to estimation or to initial results. Nor does the paper describe the different programmes in details. Instead, the paper focuses on the more mature programmes and the particular factors in the design and implementation of those programmes that have influenced their results.

Different sources use different methods when calculating the collection, reuse and recycling rates. Likewise, what is covered by the fee paid by the producer differ from one programme to another. These non-uniformities, together with the question on the reliability of the data, pose severe limitation in comparing the result.

Following this introductory section, evaluation of the results of the implementation of EPR programmes for four the product groups is presented (Section 2). It is continued with the analysis of factors that affect such results (Section 3). Section 4 summarises the findings in Section 2 and 3.

2. Evaluation of EPR programmes for different products

This chapter presents the evaluation of the results of the implementation of EPR programmes for four product groups: packaging, small consumer batteries, end-of-life vehicles and electrical and electronic equipment (EEE). The evaluation focuses on the following results: the collection, reuse and recycling rates; the stimulation of innovation (e.g. design for reuse/recycle/end-of-life management; reduction of toxic substances at source, change in a product system); the costs of implementation; soft effects (e.g. capacity building, generation and diffusion of information, improved communication between the upstream and the downstream); and approaches to overcome various institutional and structural barriers, such as free riders and existing and orphan products. At the end of the chapter, development of EPR programmes for other products will be briefly discussed.

2.1 Evaluation of EPR programmes for packaging

The main environmental impacts related to packaging waste include its volume and the presence of hazardous substances, such as cadmium in plastics. Indeed, in the early 1990s, packaging waste constituted 50 to 60 % of the volume of the municipal solid waste stream in some countries (e.g. Germany and Japan), threatening to rapidly deplete the limited remaining landfill disposal capacities (OECD, 1998b; Morishita, 1997). Moreover, discarded packaging contains useful resources.

The German Packaging Ordinance, adopted in 1991, became an archetype of new type of broad-based take-back policy for waste packaging that incorporates the concept of EPR. Since the beginning of the 1990s, various other countries have adopted EPR approaches, such as Austria, Finland, France, Japan, the Netherlands, Norway, Sweden, Switzerland and the UK. Aside from this more explicit EPR legislation, there have been a number of programmes that have at their roots the same principle, such as requirements for the use of refillable beverage packaging (e.g. Denmark) and deposit-refund legislation for beverage packaging (several countries, including Korea, Canada (certain provinces) and the United States (certain states). The legislative approach has sometimes been used to mandate EPR for packaging, but often agreements have been negotiated between the producers and the governments (e.g. the Netherlands, Norway). Producers usually create a collective system to fulfil their responsibilities for collection and recycling, which is referred to as producer responsibility organisation (PRO).

Table 2-1 summarises the EPR programmes for packaging in selected countries.

2.1.1 Collection, reuse and recycling rates

Collection

Different infrastructures have been used for the collection of packaging waste. The main systems include: 1) deposit-refund system; 2) kerbside collection system; and 3) collection centre ("bring") system.

Deposit-refund systems for some packaging (e.g. glass, PET, aluminium cans) in different countries (e.g. Sweden, Germany, the Netherlands, Norway, some provinces in Canada, 10 states in the United States) have achieved very high collection rates, from 70 to close to 100% (Lindhqvist, 2000). In most cases, the amount of the refund does not have to be high, from 0.03 to 0.25 USD (Lindhqvist, 2000). In many traditional deposit-refund systems, no targets are set. A few systems (e.g. Sweden for aluminium cans and PET bottles) sets collection targets, while several others (e.g. the United States) determine the amount of deposits by law.

An example of the kerbside collection system for packaging waste is found in Germany, where, in response to the enforcement of the Ordinance on the Avoidance of Packaging Waste, industry organized a nation-wide collection system, called Duales System Deutschland AG (DSD). Among the products covered under the Ordinance, plastics, tin plate, composites and aluminium are collected at kerbside, in parallel to the municipal waste management system (OECD, 1998b). The collection rate achieved here is also high, between 80 to 95% in 1996 (OECD, 1998b). In the case of Japan, where collection and sorting lies in the hand of local governments physically as well as financially, the collection, in most cases, are done kerbside. There, steady increase of the collection rate is observed for PET bottles since the introduction of the law: from 10% in 1997 to 35% in 2000 (MOE, Japan, 2001). The rapid increase of collection causes the storage problems in some local governments (Nikkei, 2000).

With regard to the collection centre system, the result varies. A high collection rate of 83 to 93% is observed for glass (e.g. Switzerland, Austria, the Netherlands, Norway and Sweden) (ENDS, 2000e; Lindhqvist, 2000). On the other hand, fairly low collection rates have been observed in Sweden for plastics (34%), paper/carton (40%), and aluminium packaging (33%) in 1999 (ENDS, 2000h; Lindqvist, 2000).

Reuse

In the case of beverage packaging, some EPR programmes mandate use of refillable packaging. In Austria, combined reuse and recycling targets were set for beverage packaging for 1994, 1997 and 2000, differentiated among the type of beverages, and ranging from 80 to 96% (Lindhqvist, 2000). The target setting principles in Austria were changed in the revised Packaging Ordinance of 1996, which is only specifying recycling targets for the collected amounts of packaging. In Sweden, refillable PET bottles achieved the reuse rate of 91%, and

Table 2-1:Summary of EPR programmes for packaging in selected countries

Country	**Germany**	**Austria**	**The Netherlands**	**Sweden**
Legislation (timing of the enforcement)	Ordinance on the Avoidance and Recovery of Packaging Waste (1991, revised in 1998)	Ordinance on the Target Setting for Avoidance and Recovery of Waste from Beverage Packaging and other Packaging (1993, revised in 1996)	Packaging and Packaging Waste Regulation (1997, with exception on the essential requirements and material ban)	Deposit-refund legislation for aluminium cans and PET bottles since 1982. Ordinance on Producers' Responsibility for Packaging (1994, revised in 1997)
Scope	All the packaging	All the packaging	All the packaging	All the packaging
Actual implementation	PRO:DSD (Duales System Deutschland AG) since 1991. Legislation require the same obligation to the non-members.	PRO: ARA (Altstoff Recycling Austria AG) since 1993, and operated by 8 branch companies under ARA for different materials.	Negotiated agreement (covenant) between the government and the producers in 1991, revised in 1997	Returpack AB, PRO for aluminium cans started its operation in 1984 and for non-refillable PET bottles in 1994. REPA (Reparegistret) manages 4 PROs for different materials, and Svensk Glasåtervinning AB for glass since 1994.
Scope of the PRO/negotiated agreement	Household and small commercial outlets	Household, commercial and industrial	All the packaging	Returpack: aluminium cans and PET bottles REPA: household, commercial and industrial Svensk Glasåtervinning AB: glass
Recycling targets	Glass: 75% Tinplate: 70% Aluminium: 60% Paper/board: 70% Plastics: 60% Composites: 60% (from 1 January 1999)	Glass: 93% Metals: 95% Paper/board: 90% Plastics: 40% Composites: 15% (out of the collected materials)	Glass: 90% Metals: 80% Paper/board: 85% Plastics: 27% material +8% chemical Wood: 15% (objectives 2001)	Glass: 70% Aluminium other than cans: 70% Aluminium cans: 90% Steel: 70% Paper/board: 70% recovery (40% recycling) Corrugated cardboard: 65% Plastics other than PET bottles: 70% recovery (30% recycling) PET bottles: 90% (from 30 June 2001)
Rates actually achieved	Glass: 85% Tinplate: 81% Aluminium: 81% Paper/board: 92% Plastics: 68% Composites: 79% (1996)	Approx. 60% is recovered by ARA. Out of recovered materials, the recycling rate was about 90%. (2000)	Glass: 91% Metals: 77% Paper/board: 70% Plastics: 17% material +0% chemical Wood: 24% (1999)	Glass: 84% Aluminium other than cans: 33% Aluminium cans: 84% Steel: 62% Paper/board: 40% Corrugated cardboard: 84% Plastics other than PET: 34% PET bottles (reused): 91% PET bottles (recycled): 74% (1999)
Which costs do producers cover?	DSD covers the cost for collection, sorting and recycling for plastics.	Collection, sorting, recovery	Recovery and recycling	Returpack: covers the cost for the whole system. REPA, Svensk Glasåtervinning AB: collection, sorting and recycling
Collection method	Kerbside collection system for lightweight packaging, collection centre system for glass and paper/board	Collection containers (880,000) and bags (>1 mil. households) to the consumers, 1,000 recycling stations	Local governments are responsible for collection	PET bottles and aluminium cans: deposit-refund in shops (vending machines) The rest: collection centre system
Licensees/members and the number	Packers, importers and distributors 19,150 (2000)	Packaging manufacturers, dealers, fillers, packers, importers. 12,295 (2000)	250,000 signatories (1 January 2001)	Returpack: anyone that manufactures cans/PET bottles, import empty cans/PET bottles and import filled cans/PET bottles. REPA, Svensk Glasåtervinning AB: fillers and importers. approx. 10,000 (2001)
Funding mechanism	Licence fees determined by weight of materials and unit of products (determined by volume or surface area).	Licence fees determined by weight of packaging.	No system of fees, except for paper/board when the international price is below zero. (internalised in the price of the product)	Returpack: deposits combined with advance disposal fee. REPA: Licence fees determined by weight of materials. Svensk Glasåtervinning AB: Licence fees determined by volume.

Source: DSD (2001); Pro Europe (2001); ENDS (2000h); OECD (1998b); ARA (2001) REPA (2001); Svensk Glasåtervinning AB (2001); Laws in the respective countries

refillable glass bottles, 98% in 1999, with the target between 1997-2000 being 90% and 95% respectively (ENDS, 2000h).[2]

[2] The targets for refillable PET bottles and glass bottles are abandoned from the new targets enforced since 30 June 2001.

In Denmark, the government ordered that beer and carbonated soft drinks could only be sold in refillable containers, creating an effective ban on metal cans. The refilling rate achieved in Denmark is close to 100%.

In Germany, reuse of packaging is required by mandating at least 72% of beer, mineral water, soft drinks and wine to be sold in refillable containers. If this target is not met, a mandatory deposit would be imposed for the one-way packaging. The targets were met until 1996, but the percentage of refillables for a certain type of beverages fell slightly short in the following years (71.3% in 1997, 70.1% in 1998, 68.7% in 1999) (ENDS, 2000d; ENDS, 2001g). Instead of mandating the introduction of a deposit-refund system for one-way containers of a few specific types of beverages, the German government considered in January 2001 imposition of deposits on all "ecologically unfavourable" packaging, including one-way glass bottles and metal cans, as determined by a life cycle assessment of different packages (ENDS, 2001e; ENDS, 2001a). However, the proposal was not adopted by the German Bundesrat, forcing the government to consider the introduction of the deposit-refund system for only some specific types of beverages (ENDS, 2001f).

Recycling

Some countries set mandatory recycling targets in EPR legislation (e.g. Germany, Sweden, Austria), while others set targets within their voluntary agreements (e.g. Denmark for transport packaging) or in the negotiated agreement (e.g. The Netherlands). In Germany, the actual recycling rates achieved for different sales packaging in 1996 were between 68 and 92%, all of which went beyond the requirements of 60-70% in the Packaging Ordinance (OECD, 1999b). The development of recycling of plastic waste from packaging in Germany increased from close to zero in 1989 to more than half a million tons in 1997, with a dramatic increase between 1992 (less than 50,000 tonnes) and 1994 (450,000 tonnes.) (Lindhqvist, 2000). The statistics in Sweden for 1999 show that, aside from aluminium and aluminium cans which fell short of meeting the targets set for 1997-2000, all other packaging materials achieved the targets, from 34% for plastic to 84% for corrugated cardboard (ENDS 2000h, see Table 2-1). In the case of The Netherlands, where the targets set in the negotiated agreement are higher than those required by legislation, the targets in the legislation have been met, while the targets for 2001 that are set in the negotiated agreement are in the process to be met (PRO Europe, 2001).

2.1.2 Stimulation of innovation

Germany, with one of the longest track records for a broad-based EPR programme for packaging has shown that EPR can spur innovation in source reduction. The DSD has shown, for example, an increase in the use of reusable packaging, reduction in the use of composite and plastic packaging, significant design changes in packaging, and major reductions in volume and weight by alternation of container shapes and sizes (OECD, 1998b). The average yearly reduction of close to 3% in the packaging consumption in private households and small businesses (1991-97) should be compared to a projected increase of 2-4% per year based on experiences from the 1980s (Lindhqvist, 1998). The German Packaging Ordinance has also stimulated new technologies for recycling of packaging materials. Existing technologies for glass and paper have been refined to increase recycling potential and create new markets for secondary materials (e.g. development of high quality paper for drink cartons) (OECD, 1998b). New technologies, both for sorting and recycling of plastics, have been developed to meet the recycling mandate (OECD, 1998b).

DSD conducted a survey of its licensees regarding the motivation for packaging optimisation already in 1992. The PRO for Austrian packaging recycling in 1997 conducted a similar survey. In both cases, together with increased environmental awareness (ranked 2 in both), the existence of EPR legislation was ranked high (1st in Germany, 3rd in Austria) as the reason for packaging changes. Measures taken by Austrian companies included substitution of shrink wrap plastics by plastic or metal strips; a change from plastic to paper and glass; and replacement of composites by plastics and paper (Lindhqvist, 2000).

In The Netherlands, two types of innovations or improvements in packaging have been observed. First, incremental innovations that partially reduced or eliminated packaging took place (e.g. elimination of the cardboard box for the individual toothpaste tubes). The second type has involved more sophisticated innovation, such as introduction of hybrid packaging, composed of returnable and non-returnable parts. The Dutch producers have frequently used the results of life cycle assessment and material economic analysis, two integral components of the Dutch Packaging Covenant, for the optimisation of packaging (OECD, 1998a).

2.1.3 Costs of implementation

Comparison of the costs of different programmes for packaging waste recycling is difficult, because the scope of the packaging covered, the extent of the collection, transport and recycling system covered by the financing, and the collection and recycling rates for different materials achieved vary among programmes. For example, the Austrian scheme encompasses all types of packaging (household, commercial and industrial) and covers the costs for collection, sorting, recovery and public relations). The German scheme handles household packaging and packaging for small commercial outlets, and covers the costs for collection, sorting and recycling. The French scheme, which handles household packaging, shares the costs of collection and sorting with local authorities; in Sweden different schemes manage different products. Moreover, even if the coverage are the same, different PROs have different financial management and reporting methods, some of which could count the accumulation of the reserve funds as costs of the system, while others would not. Different actors involving in the system may also have different profit margins. Thus, it is difficult to compare programmes based on the overall expenditures by the recycling organisations.

A way of comparing costs is to compare the fees that the producers or brand owners (and ultimately consumers) are paying for a similar packaging collection and recycling. The fees are often set per unit of container or per kilogram of materials, which facilitate comparison.

A study done in 1998 for the beverage container recycling systems in Germany, Sweden and Switzerland failed to determine true costs. However, by using the fees paid by producer, it showed that the costs for consumers for collection, sorting and recycling of 0.33 litre aluminium can in the respective countries were: 0.0161USD (Germany), 0.0105 USD (Sweden) and 0.0345 USD (Switzerland). Each of the three countries achieved very similar recycling rates (85-90%) utilising different collection strategies (Germany has a kerbside collection system, Sweden, a deposit-refund system, and Switzerland, a bring system). In all the three countries, a PRO organises the collection and recycling operation, which is maintained by the fees paid by the producers (Vanthournout, 1998).

The example could show that normal perception of the costs of different collection system may differ from the reality. However, the results might not be generalisable, as conditions in the three countries differ. They could also be influenced by, for instance, the existence of unreasonable profit margins and under-compensation of one or more actors.

2.1.4 Soft effects

Through surveys, both Germany and Austria have determined that the enactment of EPR legislation has been one of the main drivers for packaging optimisation, and it seems reasonable to assume that the legislation may contribute to an increase in environmental awareness among the designers of packaging.

In any collection system, consumers must be motivated to do their part in delivering the packaging to the collection system. The high collection rates achieved in programmes without financial incentives for collection (deposit-refund) suggest that consumers are acting out of an increased environmental awareness. This is even more evident when consumers are obliged to sort different packaging waste (e.g. Sweden, Japan). This environmental awareness would lead to action in areas other than recycling.

2.1.5 Free riders

In the case where a licensee of an organisation that jointly carry out the tasks given to them obtains the right to put a symbol that distinguishes his/her products from non-licensees (e.g. Germany), non-licensees may put the symbol without paying the fee (OECD, 1998b). The German DSD system also experienced licensees putting the symbol on exceeded amount of products than they actually pay (OECD, 1998b). This would lead to the management of post-consumer products of the non-licensee at the expense of licensees. According to the PRO Europe, in every country there exist free riders (companies that do nothing and do not participate in a scheme, the percentage of which range from 5 to 25 (Quoden, 2001). The percentage depends on the intensity of the control of the government (Quoden, 2001).

When the recycling targets are set for the entire market, if the common scheme achieves very high recycling rate, the non-members of the scheme would get the benefit of "fulfilling" their obligation without any costs. Similar problems occur when industries carry out their obligation under negotiated agreements: those that are not participating in the negotiated agreements may get the benefit of avoiding the enforcement of legislation without any efforts.

As a way to deal with the problems, some countries (e.g. The Netherlands, France, Germany) set a legal obligation to all the affected parties, while leaving a possibility of being exempt from the obligation by establishing a negotiated agreement or joining the PRO. Enforcement of such legislation has helped reduce the free-rider problems that occur by a mere mistake of consumers who put the post-consumer products of the non-licensee in the licensee's collection scheme, as the individual legal obligation would encourage non-licensees to join the common scheme (OECD, 1998b).

The German system dealt with the problems by giving the DSD the authority to require verification that the amount of packaging with the symbol does not exceed the amount that the license fees paid by the licensee covers. Retailers voluntarily check the products with the symbol supplied by the non-licensee. Such efforts have eliminated majority of such illegal use of the symbols (OECD, 1998b).

In a system where manufacturers and importers are supposed to pay advance disposal fees and the deposit (e.g. aluminium cans in Sweden), direct imports by consumers and illegal importers as well as the imports of empty cans caused distortion of the financial system. The producers overcome this problem by putting a bar code on the cans to distinguish the cans whose advance disposal fees are paid from the rest and installing vending machines that could read the code. It cost more than SEK 50 million (USD 4.77 million) to install the new system.[3]

2.1.6 Existing and orphan products

Due to the relative short life span of the packaging, issues of the existing and orphaned products have not been perceived as an obstacle in implementing an EPR programme.

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[3] Exchange rate: SEK 1 = USD 0.095443 (Forex, 2001)

2.2 Evaluation of EPR programmes for batteries

Use of batteries has been increasing due to the growing demand for portable devices. Small consumer batteries can be divided into primary batteries (e.g. alkaline-manganese, zinc-carbon, mercuric-oxide, silver-oxide, lithium, zinc-air) and secondary (rechargeable) batteries (e.g. sealed lead-acid, nickel-cadmium, nickel-zinc, nickel-metal-hydride, lithium-ion)[4]. Several of these batteries contain hazardous substances, such as lead, mercury and cadmium. These substances, if not managed properly once coming to the waste stream, can be dispersed into the environment from landfills and incinerators and can cause serious environmental and health problems. Batteries also contain valuable resources that can displace virgin material extraction and processing if these resources are recycled.

Countries have been dealing with these problems by limiting the amount of substances used in the batteries (e.g. mercury) or by collecting and recycling end-of-life batteries. Starting as early as the 1980s, industries in some countries (e.g. Canada, Japan, The Netherlands, Switzerland, the UK) established battery collection and recycling programmes on a voluntary basis (Morrow & Keating, 1997). Due mainly to the relatively unsuccessful outcome of such voluntary programmes or to free-rider problems, some countries (e.g. Austria, Belgium, Germany, Japan, The Netherlands, Switzerland) mandated producers (manufacturers, importers and retailers) responsibility for end-of-life management of batteries in different manners (Kiehne, 1997; Raymond, 2001). Some programmes collect all the batteries, while others collect limited types of batteries (e.g. nickel-cadmium). The majority of the systems, both mandatory and voluntary, establish a collective scheme (PRO) for collection and recycling.

2.2.1 Collection and recycling rates

Collection

The countries that have mandated some form of EPR for batteries have set collection targets. Switzerland set the collection target at 80%, Belgium 75%, Austria 65% and The Netherlands 80% by 1994 and 90% by 1998 (Beaurepaire, 1997; Korfmacher, 2001; Raymond, 2001).[5] This has resulted in higher collection rates compared to previous efforts: the collection rate actually achieved in these countries was 63% in Switzerland (2000), 67% in Belgium (2000), over 50% in Austria (1999) and 52% in the Netherlands (1996) (SAEFL, 2001b; Vassart, 2001; Raymond, 2001). All of these countries require collection of all used consumer batteries, and assign manufacturers and importers responsibility for the organisation and financing of the collection.[6] Manufacturers and importers in all of these countries established a PRO, financed by the fees paid by the member companies of the PRO. Fees, set per unit of products sold depending on their size, weight and chemical composition, can be determined by the industries (e.g. Belgium and Austria) or by the authority (e.g. Switzerland). Collection points are set up at retailers (e.g. Switzerland), or both retailers and local governments (e.g. Austria and The Netherlands). In Austria, plastic bags for battery collection are provided to 2 million households twice a year since 1995 through the organisation running the PRO (Raymond, 2001).

In the United States, where there is no national (federal) law requiring EPR for batteries, battery producers and producers of battery-oriented products established a nationwide voluntary collection and recycling scheme for nickel-cadmium rechargeable batteries in 1994 after some state governments

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[4] Aside from small consumer batteries, batteries are used in, for example, cars. As the system surrounding the end-of-life management and the characteristics of the car batteries differ from the small consumer batteries, this paper focuses on small consumer batteries.

[5] In the case of Belgium, it is a combination of voluntary agreement with a threat of eco tax. Namely, the manufacturers, importers and retailers that participate in the common recycling scheme are exempt from the eco-tax so long as the common scheme achieves the collection and recycling targets of 75% (Raymond, 2001).

[6] In the case of Switzerland, obligation to accept the used batteries is given to retailers as well.

mandated producer responsibility for these batteries (Fishbein, 1997; RBRC, 2000).[7] The products covered by the scheme have been expanded to other rechargeable batteries (nickel-metal-hydride, lithium-ion and small sealed lead-acid) (RBRC, 2000). Under the programme, collection has been done by different actors: 1) retailers, 2) communities, 3) business and public agencies, and 4) the licensees of the common collection and recycling scheme (Fishbein, 1997). The programme is financed by the licence fees paid by the companies joining, and the programme finances, among other things, all or part of the collection, transportation and recycling (Fishbein, 1997). As of 1999, approximately 25% of the collection came from retail, 5% from the community, 30% from business, and 40% from licensees (Raymond, 2001). The Rechargeable Battery Recycling Corporation (RBRC) reported that in 2000, about 3.8 million pounds (1,725 tonnes) of batteries were collected. (Raymond, 2001). While RBRC reported collection rates of 15% in 1995 to 25% in 1998, the organisation is not currently reporting collection rates due to the undefined calculation method (Raymond, 2001).

In Sweden, since the late 1980s, producers of batteries with hazardous substances finance the end-of-life management of their products via advance disposal fees paid to the government (Lindhqvist, 2000). A voluntary take-back scheme of nickel-cadmium batteries by producers started in 1993 (Lindhqvist, 2000). However, despite the initial commitment of collecting 90% of nickel-cadmium batteries by the summer 1995, the actual collection rate was 35%, leading to the re-introduction of the system before 1993 (Fishbein, 1997; Lindhqvist, 2000). There also exists a law that requires consumers to separate hazardous batteries from other waste stream, but there has never been an attempt to enforce it, resulting in a very low separate collection (Lindhqvist, 2000).

Recycling

Most of the recycling programmes bring the returned batteries to contracted recyclers (Fishbein, 1997; Raymond, 2001). In programmes where all types of batteries are collected, batteries are, either manually or automatically, sorted prior to the recycling (Vassart, 2001). Today, they are typically sorted into the following categories: nickel-cadmium, primary (alkaline-manganese and zinc-carbon), button cells, and others (Vassart, 2001).

In the case of Belgium where all types of batteries should be collected, it achieves recycling rate of more than 60% for materials in the batteries in 1999 (Bebat, 2001). When using wet chemical process devoted to batteries, which is one of the three recycling processes that is commonly used in Europe, recycling rate of 70% has been achieved (Vassart, 2001).

In the case of the programme in the United Sates where all the collected nickel-cadmium batteries are shipped to one recycling plant, the cadmium is recovered with more than 99.95% purity and used in the production of new batteries (Hanewald, McComas and Liotta, 1997; Fishbein, 1997). The same figure is found for a Swedish recycling plant (Johansson, 1997). In general, once collected, recycling of nickel-cadmium batteries is relatively easy (Morrow & Keating, 1997).

2.2.2 Stimulation of innovation

Legislation restricting the hazardous substances in batteries has been the primary environmental driving forces for battery reformulation and new battery technologies. Legislation since the early 1980s (e.g. Switzerland, the European Union) restricting the amount of mercury in alkaline batteries has driven battery manufacturers to develop alternatives for mercury-containing batteries and to reduce mercury content. Recycling of mercury-free batteries can be hampered by the presence of

[7] 8 states (Connecticut, Florida, Iowa, Maine, Maryland, Minnesota, New Jersey and Vermont) have take back requirements that apply for nickel-cadmium batteries, while Minnesota and New Jersey, aside from taking rechargeable batteries back at their own expenses, require manufacturers that the rechargeable batteries be 1) easily removable from products; 2) labelled the content and method of proper disposal; and 3) banned from the municipal waste stream (Fishbein, 1997).

mercury, so producers of these batteries have developed a label for their batteries to help prevent contamination and make recycling less expensive since 2000 (Vassart, 2001; ENDS, 2000b).

The proposed ban of the use of cadmium in batteries in the European Union, as well as in some countries (e.g. Sweden), together with the general awareness of the toxicity of cadmium and the difficulties in reaching high collection rates for recycling, have helped stimulate the industry to develop rechargeable battery chemistries that eliminate cadmium. These substitutes, such as nickel metal hydride and lithium ion batteries, are being widely employed in electronic products.

Some legislation (e.g. the EU Directive on batteries in 1991) mandates to take measures to ensure that batteries and accumulators cannot be incorporated into appliances unless they can be readily removed by the consumers. Producer involvement in collection and recycling of rechargeable batteries has, by necessity, stimulated design changes to facilitate the removal of batteries from portable devices (Fishbein, 1997). Previous power tools, for instance, were designed with sealed batteries; newer models have readily removable batteries. Tools with readily removable batteries can also be fitted with a new battery once the original battery is depleted, therefore extending the life of the tool.

2.2.3 Costs of implementation

Implementation costs for some of the collection and recycling schemes for batteries have been reported for 1999: Austria: Euro 1,600 (USD 1,414) per tonne: Belgium: Euro 7,100 (USD 6,276) per tonne; the Netherlands: Euro 5,000 (USD 4,420) per tonne; Switzerland: Euro 3,300 (USD 2,652) per tonne.[8] All these programmes cover all types of consumer small batteries. Requirement of higher recycling targets (e.g. 75% in Belgium and 90% in the Netherlands) appears to lead to the higher programme costs as compared to programmes with lower targets (Korfmacher, 2001). Limited information does not allow the authors to generalise these figures.

Revision of the EU 1991 Directive on batteries is currently discussed, which contains obligation of separate collection of all used batteries, non-binding targets of 95% collection for industrial and 75% consumer batteries, and recycling rates of 55% for all the consumer batteries. The European Portable Battery Association estimated the implementation costs to be Euro 4-7,000 (USD 3,536-6,188) per tonne, with the increase in the price of battery by 30%.[9] The European Battery Recycling Association, on the other hand, stated that the producers overestimated the costs, and it would be Euro 1,500-2,000 (USD 1,326-1,768) per tonne plus the collection costs of Euro 100 (USD 88) by local authorities and retailers (ENDS, 2001b). The divergence presented here suggests the difficulties in grasping the true costs of implementation.

With regard to the system in the United States, USD 6.7million was spent for the overall cost of the collection and recycling scheme for selected rechargeable batteries in 2000 (Raymond, 2001). As the scheme collected 3.8 million pounds (1,725 tonnes) of batteries (Raymond, 2001), the cost per tonne of collected batteries is about USD 3,900.

2.2.4 Soft effects

The respective national schemes have undertaken a variety of public information campaigns (e.g. information dissemination through mass media, at retailers, local governments) to increase the awareness of consumers of the existence of the recycling programmes. For instance, RBRC, PRO for rechargeable batteries in the United States, disseminated information through televisions, radio public service announcement, media interviews, print advertising, retail point-of-sales displays, the consumer

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8 Exchange rate: Euro 1 = USD 0.88399 (Forex, 2001)

9 Exchange rate: Euro 1 = USD 0.88399 (Forex, 2001)

toll-free help line and the RBRC websites (RBRC, 2000). National survey found that 56% of selected rechargeable-powered product owners believe that the batteries in their products can be recycled (RBRC, 2000). Radio Shack, participating retailer chains in the RBRC recycling programmes in the United States, said that the business has been benefited from participating in the programme, as the consumers identify the store when calling the toll free RBRC number for recycling information (Fishbein, 1997).

High collection rates in certain countries are indicative of the success of these public information campaigns. Public information campaigns increase consumer awareness of the benefits of recycling and the hazards of improper management of hazardous substances, which can encourage recycling of other products and support for environmental legislation to address hazardous substances.

2.2.5 Free riders

Free riders are a particular problem in the implementation of voluntary programmes, where batteries from producers who are not participating in the programme by paying fees can enter the collection system. This has been one of the reasons for the development of mandatory EPR legislation in some countries that started with voluntary programmes (e.g. Switzerland, Germany) (ENDS, 1998b; Kiehne, 1997). For example, random sampling of the returned nickel-cadmium batteries conducted in October-December 1995 in Germany showed that 51.5% (25.0%: no name, 26.5%: brands of non members of the voluntary programme) of nickel-cadmium powerpacks, as well as 25% (20%: no name, 5%: brand of non member of the voluntary programme) of nickel-cadmium single cells, were produced by free riders (Kiehne, 1997). As the implementation of mandatory legislation has been recent, no empirical data showing the reduction in free riders is available.

The voluntary programme in the United States has attempted to deal with the free rider problem by licensing the use of a seal that is displayed on batteries from producers who pay fees to the programme to distinguish them from non-participants. The programme currently has more than 313 licensees, which corresponds to approximately 90% of the battery market, which indicates that the free rider problem may not be serious (RBRC, 2000: Raymond, 2001). On the other hand, it has been reported that the collection system accepts batteries that do not carry the RBRC seal, so the seal is not being used as an enforcement mechanism against free riders (Fishbein, 1997).

Other types of free riders exist in mandatory programmes. In a system where manufacturers and importers are required to pay advance disposal fees to the authorities (e.g. Sweden for hazardous batteries), some small importers have successfully avoided paying the fee, relying on weak government enforcement. Government enforcement is also ineffective against direct imports by individual consumers. Different importers have been checking the payment of the fees with each other, which contributes to the reduction of the problem.

2.2.6 Existing and orphan products

So far, the existing and orphaned batteries have not been perceived as an obstacle in implementing an EPR programme for batteries. However, some of the rechargeable batteries last for several years. As mentioned in Section 2.2.5, the random sampling conducted in October-December 1995 in Germany suggested that 25% of the returned nickel-cadmium powerpacks, as well as 20% of nickel-cadmium single cells, had no names, indicating the problems of existing and orphaned products (Kiehne, 1997). The increase in the number of rechargeable batteries used in the market indicate the increase in the problems of existing and orphaned products. Although the collective recycling programme help the physical management of orphaned products, financing of the orphaned products still remains to be a problem.

One future issue of concern is how to finance the ongoing collection and recycling of old nickel-cadmium batteries after a phase out of the sale of new nickel-cadmium batteries occurs.

2.3 Evaluation of EPR programmes for end-of-life vehicles

Automobiles are one of the central parts of the modern product based society. Currently there are approximately 700 million automobiles in the world, and 57 million cars were sold in 2000 (Bilbranschen, 2001). Cars exert significant environmental impacts throughout their life cycles, with the majority of pollutants being released during the use stage (driving). The automobile is one of the most recycled products in the world today, but the sheer number of end-of-life vehicles makes the remaining waste stream, which is primarily disposed of in landfills, a high priority for recycling efforts. The shredder waste, called auto shredder residue (ASR) or "fluff", which is composed primarily of plastics and fibres, poses threat to landfills both in terms of quality and quantity.[10] Hazardous substances contained in the car (e.g. lead, mercury, cadmium, hexavalent chromium) render ASR hazardous, and the increasing use of plastics in cars makes recycling more difficult and less economically attractive.

A number of EPR programmes have been developed with the aim of reducing the environmental and health impacts of end-of-life vehicles (ELVs). For example, the European Union, after lengthy discussion, adopted the Directive on End-of-Life Vehicles that incorporates the concept of EPR in September 2000. The Directive will be implemented by the EU member countries beginning in 2002. At the national level, voluntary systems have been implemented in Germany and in the Netherlands. In Germany, after a lengthy debate, a draft ordinance presented in 1990 was replaced by a voluntary EPR programme established by the auto industry.[11] In The Netherlands, all the actors in the chain got together and established a system for collection and recycling, with a Producer Responsibility Organisation called Auto Recycling Nederland (ARN) managing the system. In Sweden, the Ordinance on Producer Responsibility for Cars was enacted in 1997, replacing a deposit-refund system that was run since 1975. In Italy, Fiat started to explore a special collection and recycling network in 1992. In Japan, an EPR regulation for cars is under development. In the United States the system for recycling end of life vehicles relies upon market forces. One collective response from the industry in the United States to enhance recycling was the creation of the Vehicle Recycling Partnership in 1991 to promote and conduct research on technologies to recover, reuse, and dispose of materials from scrap cars (Poston, 1995). Individual companies in the United Sates have conducted pilot programs for take back and recycling of large plastic parts that have not been routinely recycled and have made major efforts to incorporate recycled materials into new cars (Davis, 1997).

2.3.1 Collection, reuse and recycling rates

Collection

Collection rates for ELVs are typically high in industrialized nations with well-developed dismantling and recycling infrastructures, although reliable statistics are hard to come by that take into account abandoned cars and second-hand cars exported to other countries (Kincaid, Wilt, Davis, Lumley, Stoss & Carnes, 1996). Even a relatively small number of ELVs abandoned by the side of the road are considered unacceptable in most countries, and EPR programmes have been developed to encourage the last vehicle owners to turn them in for recycling. Sweden, for instance, instituted a deposit-refund system in 1975 to decrease the number of abandoned cars. The problem was nearly eliminated until

[10] For example, In Europe, approximately 2 million tonnes of shredder waste is generated from end-of-life vehicles per year, which constitutes approximately 60% of the overall weight of shredder waste. Commission of the European Communities. (1997)

[11] The German government is currently working on the implementation of the EU Directive (ENDS, 2001i).

recently when dumping of cars in nature revived due to the inadequate level of the refund given to the last owner. With the increase of the refund from SEK 500 (USD 48) to SEK 700-1,700(USD 67-162) in July 2001,[12] the dumping problem disappeared (Lindhqvist, 2001).

None of the EPR programmes that have been implemented for ELVs (The Netherlands, Sweden, Germany) have collection targets; neither does the EU ELV Directive. Targets focus instead on the percentage of cars collected that are recycled. In order to create incentives for collection the EU Directive requires producers to insure that the last owner can turn in ELVs free of charge and requires member states to continue to collect registration fees on a vehicle until the last owner presents a certificate showing that it has been recycled by an authorized recycler.

In the United States it has been estimated that 94 percent of the cars and trucks at the end of their useful lives are currently returned to dismantling and shredding facilities for recycling (Curlee, Das, Colleen & Schexnayder, 1994). This high collection number in a country without an EPR programme for cars is attributable to a profitable ELV dismantling and recycling infrastructure that maintains a positive value for ELVs. If higher recycling performance is required of this system, however, ELVs end up with a negative value that would hamper collection efforts.

Reuse and recycling

The predominant method of dealing with end-of-life vehicles involves dismantling, shredding, and recycling of steel and aluminium. Dismantlers remove high-value parts for reuse and reconditioning. Shredders shred the auto hulks to recover ferrous and non-ferrous metals, which are sent to recycling mills (Kincaid, Wilt, Davis, Lumley, Stoss & Carnes, 1996).

Most reports have previously estimated the percentage of the vehicle reused and recycled in modern dismantling and shredding systems to be around 75% by weight. These materials are primarily steel, iron, and non-ferrous metals, such as aluminium. These percentages have generally been accomplished under market-based systems because of the value of the metals. According to the new way of calculating the recycling rate adopted by the European Union, approximately 81-82% of the vehicle, by weight, is currently reused and recycled.

The EU 2000 Directive mandates the achievement of 85% reuse and recovery by 1 January 2006, 80% of which should be achieved by reuse and recycling. The rate will be increased to 95% and 85%, respectively, by 1 January 2015.

The ARN in the Netherlands established its own recycling goal of 86% by the year 2000, which it accomplished by 1997. The recycling rate achieved in 2000 remained 86%. ARN aims to achieve 95% recycling rate, suggested by the European Union Directive, before 2015. When calculating the recycling rate, no distinction has been made between reuse, recycling and energy recovery (ARN, 2001).

2.3.2 Stimulation of innovation

Although there could be some difference depending on the size and structure of the cars, with the current calculation models, 81-82% of the cars are recycled on commercial basis. Therefore, the challenges for the car manufacturers lie on the achievement of additional 13-14% by 2015 to fulfil the requirement set in the EU Directive, which will have to be primarily achieved through the recycling of

[12] Under the new ordinance, the premium for the car less than 7 years old is SEK 700 (USD 67), 7-16 years old SEK 1200 (USD 115) and more than 16 years old SEK 1700 (USD 162), respectively (Regeringskansliet, 2001). Exchange rate: SEK 1 = USD 0.095433 (Forex, 2001)

glass and plastics. Producers are making significant efforts to incorporate plastics in cars that can be easily recycled. Due to the global market of cars, such efforts have been made not only as a response to national legislation, but also to legislation in other parts of the world, including the EU Directive.

In order to increase the recyclability of the plastic portion of the car, some manufacturers have reduced the variety of plastics used for different parts of the car, and are using plastics that are commonly used among a wider range of industries (Nissan, 2000). One manufacturer succeeded in developing specific plastics that can be recycled for exactly the same purpose without degrading the quality (Toyota, 1998). Recycling of plastic bumper covers, which are one of the largest plastic parts, both bumper-to-bumper and down-cycling to other car parts has been practiced by a number of companies (Nissan, 2000; Toyota, 1998). Individual companies in the United States have also conducted pilot programs for collection and recycling of certain plastic parts to increase recycling percentages and have utilized recycled plastics in automobile production to increase markets for recycled materials (Davis, 1997).

Different design initiatives that ease the removability of components (e.g. fuel tank) at the dismantling process have been taking place. Besides the design change of its own products, one manufacturer in Japan, in anticipation of the national EPR regulation, has been developing tools that could facilitate the dismantling and scrapping process (Nissan, 2000).

Toxic substance restrictions are also an important driver for design changes. The EU Directive restricts the use of cadmium, lead, mercury and hexavalent chromium in vehicles. Development of alternative substances that will allow producers to achieve these bans has been progressing under the cooperation with the material and components suppliers (Tojo, 2001).

2.3.3 Costs of implementation

Under the Swedish programme, manufacturers and importers of cars established individual internal funds that are set aside for future recycling (Lindhqvist, 2001). One of the manufacturers gave the figure of SEK 1,300 (USD 121) as the amount the company set aside for recycling when selling a new product (Tojo, 2001).[13]

ARN, the PRO for the Dutch programme, used in total NLG 76.4 million (USD 30.3 million) for the recycling and handling (also includes research and development expenditure) of 286,595 cars that were handled under ARN in 2000 (ARN, 2000; ARN, 2001). This means that it costs approximately USD 106 to recycle a car with the achievement of current recycling rate (86%). The fee paid by the industry has been reduced from NLG 250 (USD 100) between 1995 and 97, NLG 150 (USD 60) between 1998 and 2000, and will become Euro 45 Euro (USD 40) (ARN, 2001).

As discussed in the case of packaging, direct comparison of the costs of different programmes is difficult and misleading unless there are similar standards (environmental, as well as others), similar recycling levels, similar markets for spare parts, equally enforced taxation systems, similar coverage of collection costs, and the like.

For example, the cost that is set aside by the manufacturers under the Swedish programme will cover the recycling costs and the refund of the car that is presently sold, that is it should cover the costs when the recycling targets goes up to 95%. The fees paid in the Dutch programme, on the other hand, will cover the cost of the car that ends its life now, with the recycling requirement of 86%. The money reserved in the Swedish system has the possibility to yield interest, while there is little room for

[13] Exchange rate used in this section is the following: SEK 1 = USD 0.095443; NLG 1 = USD 0.396507; Euro 1 = USD 0.88399. (Forex, 2001)

the fee in the Dutch system to increase with interest. Under the Dutch system, the difference in the number of cars presently sold and those coming back now should also be considered. The cars that are exported in the second hand markets will not be covered by the domestic recycling system. Furthermore, the cars that go to scrappers that have no contract with the producers would not be included in the calculation of the recycling rate.

2.3.4 Soft effects

As mentioned earlier, EPR programmes have encouraged manufacturers started to co-operate with actors both upstream (material and component suppliers) and downstream (dealers, dismantlers and scrappers). One Japanese manufacturer sent its personnel to more than 300 recycling facilities both within and outside Japan, to familiarise itself with the existing practices. The company has created a network with the dismantlers and scrappers. It distributes newsletters with information on new tools and technologies for dismantling and scrapping, which the company itself developed. It also communicates with the car dealers, who, under the current practice and anticipated Japanese EPR legislation for cars, will be the first party to receive end-of-life cars (Nissan, 2000; Tojo, 2001).

One of the Swedish manufacturers sent its design personnel to dismantling plants so that issues regarding the end-of-life management can be directly communicated. The European car industries developed a common manual for dismantlers and scrappers, and provide it to more than 2,200 dismantlers in Europe in the form of a CD-ROM (ENDS, 1999; Tojo, 2001).

In the United States, Ford has responded to EPR legislation in the EU and elsewhere by setting out to become the world’s largest automobile recycler. Ford has been purchasing automobile recyclers in the United States and Europe and has a goal of recycling more than 90% of its cars and trucks and generating USD one billion in revenues for the company (Hoffman, 2000).

2.3.5 Free riders

It has been feared that for the cars which are directly imported by consumers, no one would set aside the funds for the future recycling. As a solution to this problem, the Swedish system demands that consumers who import cars contribute to the state-administered fund for future recycling (Regeringskansliet, 2001).

2.3.6 Existing and orphan products

Due to the long life span of cars, objections to retroactive legislation have been raised for the application of EPR requirements for existing cars, especially in the development of the EU Directive. As a result, the Directive includes phased timing for the free take back requirement, phased timing for recycling requirements, and reduced recycling requirements for older cars. The free-of-charge take back from consumers would be required only for the new cars placed on the market from 1 July 2002, which will be expanded to all the cars by 1 July 2007. Also, for cars produced before 1 January 1980, the recovery and recycling targets is not 85%, but 75%.

Use of insurance as a financial mechanism for producers to cover future recycling costs of cars has been discussed in Sweden, and has been utilised by one importer since April 2001 (Olle Olsson Bolagen & Länsförsäkringar AB, 2001). An insurance arrangement would help eliminate the problems of orphaned end-of-life vehicles. A mutual fund that is separated from the producers’ own accounting system is also considered. Moreover, in order to fund the recycling of the existing products while establishing a funding mechanism that encourages design for end-of-life management, two different systems for existing and new products are used in Sweden. In the case of new products, fees for the future recycling costs are managed independently by each manufacturer, whereas the recycling costs for the existing products that are presently coming back are paid for by the common car scrapping funds collected from the sale of new cars. Thus, the price of a new car will bear both the recycling costs for an old car which should be reserved in the common car scrapping funds and the recycling costs of its own (Lindhqvist, 2001).

2.4 Evaluation of EPR programmes for EEE

Electrical and electronic equipment (EEE) include a wide range of products, from large and small home appliances (refrigerators, microwaves, air conditioners, toasters, shavers), telecommunication and ICT (information and communication technology) equipment (telephones, computers) to toys, lighting equipment and medical equipment. EEE are considered priority products for diversion from landfills and incinerators because of their increasing overall volume and because they contain hazardous substances, such as lead, cadmium, mercury, and brominated flame retardants. EEE are problematic for traditional municipal collection and recycling infrastructures because of the rapid advancement of technology increases the variety of and complexity of products. The situation gets worse when adequate information is not transferred from the manufacturers to the treatment facilities. These interrelated features of EEE make EEE waste problematic both in quality and in quantity.[14] Improper management of the discarded products also leads to the abandonment of useful resources in the landfill.

In Europe, the European Union has been discussing the development of the Directive on Waste Electrical and Electronic Equipment at length.[15] Among the EU member states, the Netherlands and Sweden enacted national legislation prior to the completion of the Directive. Italy also started a programme for a limited scope in November 1997, when producers established a collection/recovery network for refrigerators, based on the comprehensive waste management decree on 1996 (Product Stewardship Advisor, 1998). Denmark also enforced legislation for the end-of-life management of EEE in December 1999, but this legislation does not incorporate EPR. Among the non-EU member states, Switzerland and Norway enforced their national EPR regulation for EEE in July 1998, and in July 1999, respectively. Some others, such as Germany and Austria, started developing the national legislation, but are waiting until the EU Directive is finalised (Dworak & Kuhndt, 2000; BATE, 2000).

In Asia, Taiwan enforced take back legislation for four large home appliances (TV sets, air conditioners, washing machines and refrigerators) in 1998 (Tanaka, 2000). In Japan, the Specified Home Appliance Recycling Law, covering the same products as the one in Taiwan, was enforced in April 2001. Moreover, take back of computers is being discussed in a separate regulation. Korea has a deposit-refund system for the same product categories (MOE, Korea, 2001).

Some states in the United States as well as provinces in Canada, have been discussing legislative measures incorporating EPR, while some industries initiated voluntary take back systems (Sustainable Business Insider, 2000). Moreover, in the United States, different stakeholders, such as state and local governments, federal governments, manufacturers, retailers, recyclers and environmental groups are gather together to come up with an optimal system for end-of-life management of EEE, while giving incentives to manufacturers for design change (NEPSI, 2001).

2.4.1 Collection, reuse and recycling rates

Collection

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[14] For example, among the 15 EU nations, the average annual increase rate of the volume of waste EEE is expected to be 3-5%. AEA Technology (1997) p. 24.

[15] The proposal by the European Commission in June 2000 consists of two separate directives, the Directive on waste EEE and on the restriction on the use of certain substances in EEE.

The EPR programmes for EEE do not have mandated collection targets, and the amounts of EEE collected vary considerably. In the Netherlands, the manufacturers and importers of large and small consumer EEE, aside from the ICT equipment, established a collective system to fulfil the responsibility assigned to them (NVMP, 2001). Under the Dutch EPR legislation, it is the retailers, local governments and repair shops that collect the waste EEE. As of 2000 when the Decree came into force in full scale, the total annual collection through the collective programme amounted to 57 million kg, or 3.6 kg per person per year, slightly below the non-binding separate collection target set by the proposed EU Directive (4 kg per inhabitant per year from private households) (NVMP, 2001). In the inspection in mid 2000, it was found that not all the retailers are accepting the end-of-life EEE from consumers free of charge (ENDS, 2000a).

In Switzerland, where two take-back systems had been developed by the industries prior to the enactment of the ordinance in 1997, collection reached 36,000 tonnes, a collection rate per person per year of 5.1 kg, in 1999 (Türk, 2001). However, it should be noted that the amount of discarded EEE is estimated to be 110,000 tonne per year, which indicates that only one third is covered by the existing programmes (SAFEL, 2001a). Disposal of small EEE in the municipal waste stream and exports of second-hand products are some of the explanation for the gap between the collected end-of-life EEE and the estimation of the total (Türk, 2001).

In Norway, where EPR legislation was enforced in July 1999, it was found in December 2000 that 3 distributors have not fulfilled their responsibility of accepting the end-of-life EEE on one-for-one, old-for-new basis, while 5 major retail chains failed to inform consumers of their collection responsibility (ENDS, 2000f). Despite the 80% collection targets within the voluntary agreements in 1998, the annual collection of mobile phones was 25,000, while annual sales were about 1.5 million (ENDS, 1998a; ENDS, 2000g). It was also mentioned that small appliances, such as toothbrushes, drills, toys, alarm clocks and hair dryers, suffer low collection rates (ENDS, 2000g).

In the German state of Lower Saxony, the separate collection only reached to 2.7 kg per person per year in 1999, although a 60% increase by weight was observed compared to 1995 levels. According to the survey, the programme achieved 100% collection of large household appliances (e.g. washing machines, stoves and dishwashers) and close to 100% collection of entertainment equipment (e.g. TV sets, computers, videos and stereos). However, for smaller items (e.g. electric razors, mobile phones and pocket calculators), only 30-40% collection was achieved (ENDS, 2001d).

In Japan, manufacturers and importers are physically responsible for recycling. They built their own recycling plants, and securing enough end-of-life EEE is vital for the efficient management of these recycling facilities. Since the enforcement of the legislation in April 2001, a total of approximately 2.5 million products have been collected (more than 95% by retailers) in the first 3 months, leading to a projection of 10 million per year (METI, 2001). This would represent a decrease of approximately 3 million, or 25% compared to the annual collection (both by retailers and local governments) before the new programme was implemented (METI, 2001). As the end-users pay for the collection and recycling at the time of disposal, an increase of illegal dumping has been reported. Exports of second hand products and components continued as well (Miyasaka, 2001).

It should be noted that a considerable amount of discarded products are probably not accounted for in official statistics, and determination of the number of such “grey-zone” products and their final disposition is very difficult.

Reuse and recycling

Minimum reuse and recycling rates are mandated by most of the EPR programmes for waste EEE. One of the collective systems in The Netherlands, achieved a recycling rate of 86% for refrigerators and freezers, 75% for large home appliances, 78% for TVs, and 64% for other small appliances

(NVMP, 2001). All these figures exceed the targets set in the advance notification sent to the Ministry of Environment by producers (75%, 74%, 69% and 53%, respectively) (NVMP, 2001). The Dutch legislation allows the inclusion of recycling by energy recovery.

In Switzerland, about two-thirds of the large and small electrical appliances collected by one PRO in 1999 was materially recycled. The PRO handling the information and communication technology (ICT) equipment achieved material recycling rate of roughly 75% by weight. The Swiss law does not set numerical targets (Türk, 2001).

In Japan, although data regarding the attainment of the minimum recycling rate is not available so far, the initial recycling requirement of 50 to 60% by weight (fulfilled by product reuse, component reuse and material recycling) was considered to be relatively easy to achieve (Bizen, 1999). However, the manufacturers strive to achieve higher recycling rates in anticipation of the introduction of higher recycling requirement in coming years (Tojo, 2001). It should be noted that under the Japanese law, the requirement must be fulfilled with material with no or positive monetary value.

2.4.2 Stimulation of innovation

Although not enforced, the discussion and anticipation of the EU Directive that restricts the use of heavy metals (lead, mercury, cadmium, hexavelent chromium and brominated flame retardants, with some exemption depending on the application) promoted vigorous efforts in the development lead free solders.

Similarly, the producers strenuously search for ways to meet with the increased reuse and recycling targets that are anticipated (e.g. Japan), with the reuse and recycling requirement of the anticipated EU Directive, and with landfill restriction (e.g. The Netherlands). With regard to material use, measures that are taking place include uniformity of the type of plastics, marking of the type of plastics, development of recyclable plastics, replacement of plastics with magnesium alloy, reduction of use of hazardous substances, and development of refrigerants that have less impacts for ozone depletion and climate change (Mitsubishi, 1999; NEC, 1999; Matsushita, 1998; Sony, 1999; Tojo, 2001). With regard to structure, measures such as the reduction of the number of components, standardisation of screws, and uniformity of the direction of the screws are taking place (Mitsubishi, 1999; Tojo, 1999). All of these lead to the reduction of the cost for recycling.

In Japan where manufacturers and importers are physically responsible for take back and recycling of discarded products, some manufacturers, in cooperation with some existing end-of-life management plants, conducted different projects for improved recycling technology (Sony, 1999; Matsushita, 1998).

In the United States, the Design for the Environment (DFE) Program of the Environmental Protection Agency has conducted major research and demonstration projects aimed at improving the design of EEE. One project evaluated the life-cycle environmental impacts of liquid crystal display computer monitors as compared to cathode-ray tube monitors to assist the industry in determining whether substitution of cathode-ray tubes by liquid crystal display would constitute an environmental benefit, including reductions in leaded glass disposal from cathode-ray tubes (USEPA, 2001). The DFE Program is planning a life-cycle study of substitutes for tin-lead solder to assist producers in minimizing the environmental impacts of solder disposal.

2.4.3 Costs of implementation

Table 2-2 presents the end-of-life management fees in selected countries for some of the EEE, as well as what is covered by the fees, the type of producer responsibility and the financial mechanisms used.

In the Netherlands, where both visible and invisible fee systems are used, some of the producers that use both systems have complained that the system with the visible advance disposal fee was charging five times more than the actual recycling costs in February 2001 (ENDS, 2001c).

In Japan, consumers pay both the collection and the recycling costs at the time of disposal. The manufacturers as well as the retailers and local governments must announce the costs of the operation for which they are responsible. The initial costs that prominent manufacturers announced turned out to be the same, although the fees announced for the four products are perceived to be far less than the actual costs (Tokyo Metropolitan Government, 2000; Tanaka, 2001).

In Sweden, most of the producers are joining a collective system for recycling. Although the methods of treatment of certain components have been determined by the law, and are to be carried out in certified treatment plants, the Swedish law does not require any recycling rate achievement.

Comparing the costs that are born by consumers in the three countries, relatively close figures are observed between Switzerland and Japan, while the fees in the Netherlands and Sweden are significantly lower. Just as any other products, a simple comparison is difficult due to the difference in the range of responsibility, in achieved recycling results, in the recycling methods, and the like.

Table 2-2: Fees for some of the EEE in selected countries and financial mechanism (2001)

	Switzerland USD (in SFR)	**The Netherlands** USD (in Euro)	**Japan*** USD (in JPY: recycling fee + collection fee)	**Sweden** USD (in SEK)
Refrigerators/freezers	45 (75)	15 (17)	37 + 3 to 24 (4,600 + 500 to 3,000)	Handled by local governments
TVs	12-42 (20-70)	4.4-15 (5-17)	22 + 3 to 24 (2,700 + 500 to 3,000)	3-8 (30-80)
Large home appliances	9-26 (15-43)	4.4-15 (5-17)	Air conditioners: 19 + 3 to 24 (3,500 + 500 to 3,000) Washing machines: 19 + 3 to 24 (2,700 + 500 to 3,000)	4-8 (45-85)
Small home appliances	0.2-1.3 (0.30-2.20)	0-0.9 (0-1)	Not covered by the legislation	0.5-2 (5-20)
ICT equipment	Determined by the price of the new products	Internalised in the price of the new product	Not covered by the legislation	Special agreements between producers and the PRO
What is covered by the cost?	Collection from the retailers and recycling	Collection from collection points, retailers and repair shops, recycling	Manufacturers: Establishment of collection points, collection from collection points and recycling Retailers and local governments: collection	Collection from collection points and recycling
Type of producer responsibility	Brand related	From retailers: old-for new until 2005, brand-related from 2005 From local governments and repair shops: brand related	Brand related	Old for new
Financial mechanism	Refrigerators/freezers, ICT equipment: visible advance disposal fee Others: the last owners pay	ICT equipment: invisible to the consumers Others: visible advance disposal fee	The last owners pay:	Manufacturers pay the fees invisible to the consumers to the PRO.

*Unit: USD, ( ) in local currency. Exchange rate: SFR 1 = USD 0.601575; Euro 1 = USD 0.88399; JPY 1 = 0.008036USD; SEK 1=USD 0.095443 (Forex, 2001) *The recycling costs announced by prominent manufacturers and collection cost announced by retailers or local governments.*

Source: SWICO (2001), S.EN.S (2001),NVMP (2001), Tanaka (2001), El Kretsen (2001)

2.4.4 Soft effects

In Japan where the manufacturers are directly involved in the development and management of recycling plants, enhancement of the communication between the upstream and the downstream actors has been reported (Tojo, 2001). Concrete measures taken include designers' study visits to recycling plants, information exchange via intranets, seminars held for designers by the personnel in charge of recycling, and designers' participation in dismantling exercises (Mitsubishi, 1999; Fujitsu, 2001; Tojo, 2001). Similarly, anticipating the EU Directive, prominent manufacturers of mobile phones conducted pilot projects for take-back and recycling, involving both retailers, transporting companies and recyclers (ECTEL, 1997).

In order to survive in the market, component as well as material suppliers also started to develop alternatives with less hazardous substances (Nakanishi, 2000).

Emergence of new businesses are observed, such as second-hand markets, collection service by convenience stores, and even monitoring equipment to film people that are engaged in illegal dumping (e.g. Japan) (Tanaka, 2001).

2.4.5 Free riders

With a purely voluntary program (without government enforcement of responsibility for all producers), the classical free rider problem, where products enter the recycling system from companies that have not paid for the programme, can jeopardize the financial health of and producer support for an EPR programme, unless there is near universal participation from producers and importers. None of the programmes evaluated in this paper are purely voluntary, however.

In EPR programmes where producers exercise their responsibility collectively and do not have brand-related responsibility, producers pay flat fees, most typically depending on the type of the product and its weight. Namely, as most of the common schemes deal with a wide range of products, with fast development of new technologies and materials, it is impractical to set up a differentiated fee among the same type of products depending on design for end-of-life management. As a result, the "green" companies subsidize the recycling of the products made by companies that have not redesigned their products to facilitate reuse and recycling. This issue has been debated heavily in the development of the anticipated EU Directive, and the opinion of the industry is split. The problem is one of the reasons why Denmark decided not to establish an EPR system. (Christiansen, 1999).

2.4.6 Existing and orphan products

In dealing with management of the existing EEE that have a relatively long life span, EPR programmes allocate the responsibility differently. For example, in The Netherlands, up until 2005, producers are required to take back discarded products from retailers on an old-for-new basis, and take back their own products from take-back sites (aggregation points) and repair companies (brand-related responsibility). From 2005, all the take-back responsibility will be brand-related. The current proposal for the EU Directive requires collective responsibility for the existing products, while assigning either collective or individual responsibility after 2005.

Some EPR programmes (e.g. Sweden) make producers responsible for taking back an old product when selling a similar new product regardless of the brand (old-for-new, one-for-one). In some others (e.g. Japan and partly Switzerland), the last owners are to pay at the time of disposal, and together with the back up system by local governments and designated legal entities (e.g. Japan), all the existing and orphaned products would be covered. However, non-brand related responsibility has a problem of not effectively giving incentives to the producers to change their products and product system, and illegal dumping or disposal in the municipal waste strea, is a serious concern with regard to the last-owner pays system.

Use of an insurance premium as discussed in Sweden would solve the problem of orphaned products, but would not help the situation with existing products. The Dutch system has another approach for dealing with orphaned products: producers or importers must notify the financial scheme when withdrawing from business in the country. However, the feasibility of the enforcement of such a system is not clear.

2.5 EPR programmes for other products

Aside from the four products mentioned above, EPR programmes have been in place for, for example, household hazardous wastes (e.g. used oils) in British Columbia, Canada (mandatory), fluorescent tubes in Austria (mandatory), tyres in some countries (e.g. Sweden and the Netherlands: mandatory, the United States: voluntary), carpets (the United States: voluntary).

Although the implementation of these programmes will not be evaluated here, experiences from these schemes will be mentioned in the following sessions whenever appropriate.

3. Factors affecting the results of the programmes

Based upon the experienced of the EPR programmes for the four products presented in Section 2, this Section assesses the factors that have affected the results of EPR programmes. Factors discussed include: the characteristics of the products themselves; the type of EPR programmes; the manner in which different types of responsibility are allocated in the product chain; the financial mechanism used for funding the programmes; the use of targets for collection, reuse and recycling; the systems (infrastructure) surrounding the products for each stage of their life cycle; and the awareness and perception of actors in society affected by the programmes.

3.1 Characteristics of the products

3.1.1 Environmental and health impacts of discarded products

The potential environmental and health impacts of discarded products have affected the impetus for developing EPR programmes and the manner in which EPR has been implemented. As discussed, the primary environmental problem related to the post-consumer packaging is the volume it occupies in landfills, while the content of hazardous substances that is the most problematic in the case of batteries. In the case of EEE and cars, both the volume and hazardous substances content result in disposal problems and increased recycling and disposal costs. These increased waste management costs worsen the problems of illegal dumping and the exports to other countries of waste under the name of second-hand products leading to the dispersal of environmental and health problems.

Depending on the types (quality and/or quantity) and magnitude of the waste problems that each country is facing, different products have become the focus for EPR. For example, in countries where the scarcity of disposal sites poses a major threat to society (e.g. Germany, the Netherlands, Japan), post-consumer products that have large volume such as packaging and large EEE have come first under EPR programmes. In other countries the emphasis is the environmental and health effects of hazardous substances in the products (e.g. Canada, Sweden).

A high level of concern about these product disposal problems in the country, as well as recognised limitation in the capacity of conventional waste management systems to handle the problems, create an environment where the government responds by establishing an EPR programme for these products. Further, general awareness of the problems promotes the co-operation of consumers in EPR programmes, which is a prerequisite for the successful outcome of the EPR programmes (e.g. improved collection rates for batteries and packaging).

3.1.2 Useful resources in the products

Recycling of cars, some EEE (e.g. old computers) and some packaging (e.g. aluminium cans, paper and glass) began prior to EPR programmes, without the significant involvement of producers, purely on a commercial basis due to the value of the materials that the products contain. These pre-existing recycling systems can make it easier to set up an EPR programme for the products. However, if the recycling of a product is not economically profitable, or if an increased level of recycling is necessary that cannot be achieved on a market basis, there must be some direction provided by governments to change the market. While not the only means of encouraging recycling where the market has not adequately responded, EPR is a strong incentive because of its involvement of producers, both financially and in product design to make products more economically recyclable.

3.1.3 Potential for design changes

EPR programmes are most effective in reducing waste generation and increasing recycling where there is a potential for product design changes that can reduce the costs of recycling. Example of such design change include elimination of hazardous substances (e.g. mercury and cadmium in batteries, lead in the components of EEE) or of unnecessary material (e.g. optimisation of packaging), increased reuse (e.g. transport packaging, refillable bottles, component reuse for some EEE), increased use of recyclable materials (e.g. change from plastic to metal and development of recyclable plastics in cars and EEE), and promotion of design for disassembly (e.g. bumpers and fuel tanks in cars, some components of EEE). For some products producer responsibility for take back and recycling may not send sufficiently strong signals to producers to implement design changes. This is typically the case with hazardous substances. EPR programmes for these products are often supplemented with hazardous substance restrictions.

3.1.4 Complex products versus simple products

At present, EPR programmes exist for both complex products (e.g. cars, EEE) and relatively simple products (e.g. packaging). For complex products programme implementation can be more difficult than for simple products. For instance, the measurement of recycling rates is more difficult for complex products because they have multiple components and multiple reuse and recycling pathways. Moreover, advance calculation of costs for recycling for complex products that truly reflect the environmental impacts is more difficult than for simple products, which hinders giving incentives to the producers to consider the end-of-life management of products at the design phase (discussed further in Section 3.3.2, 3.5).

3.1.5 Durability of the products

Compared to products with relatively short life span (e.g. packaging, non-rechargeable batteries), products with long life span (e.g. cars, EEE, rechargeable batteries) add to difficulties in the management of EPR programmes. For instance, for products that are used for several years, a financial mechanism relying upon fees on the sale of new products generates revenue years before the product is recycled, making it difficult to predict ultimate recycling costs. A new EPR programme for durable products inevitably receives a backlog of existing products in the beginning that may have not been designed with recycling in mind and that may far exceed the ultimate volume of products projected to be handled by the programme. Durable products also increase the possibility of orphaned products coming into the system. It is also more difficult to evaluate the collection rate of durable products, because current production rates have little to do with the volume of products being discarded that were produced years ago (discussed further in Section 3.3, 3.4 and 3.5).

3.1.6 Size, weight and ease of handling

Generally speaking, the smaller the size of a product, the less likely that it will be sorted from the waste stream unless sorting and collection is made very convenient and/or supported with financial incentives. This is illustrated in the low collection rate of small EEE in comparison to the large EEE (e.g. Switzerland, Norway), of batteries (e.g. Sweden), and of aluminium and plastics other than cans and PET bottles (e.g. Sweden).

When products are heavy and difficult to handle (e.g. large home appliances) or perceived to be a direct harm when mixed with the rest of the waste stream (e.g. glass) unless the old products are picked up by the retailer when a new one is purchased, it often takes a while for consumers to bring the old products to established collection points. This is especially the case when they can easily store the old products in their homes. However, once consumers do decide to discard such products, they often bring them to the appropriate collection points, at least partly because the products are usually not accepted and easily introduced in the municipal garbage collection.

For large appliances that are readily identified by brand, individual producer take back is possible which results in more direct participation by the producer and greater opportunities for individual design changes and competition among producers for reducing recycling costs.

When the size of the individual products is small (e.g. packaging, batteries), producers tend to establish a collective system to maximise the efficiency of collection and recycling.

3.1.7 Similar products and confused consumers

If the scope of an EPR programme is limited to certain types of products among the wide range of products that have a similar function (e.g. nickel cadmium batteries among batteries), as is common in voluntary EPR programmes, products that are not covered by the programme would still come into the system. For example, collection stations for nickel-cadmium batteries in different retailers in France ended up receiving all types of batteries, primarily because consumers were unable to distinguish different types of batteries (Beaurepaire, 1997). This creates problems and added expense for the collection and recycling system and results in free riders if the erroneously collected products are recycled by the EPR programme without payment by the producers.

Even if consumer information campaigns are undertaken or labels are created for participating products, consumers still often place non-participating products in the collection scheme. This has been experienced in the case of packaging (e.g. Sweden for glass bottles for wines with no labels, kerbside collection of packaging not licensed by the DSD system in Germany) and in many collection systems for batteries.

Mandatory EPR schemes are more likely to include in their scope all similar products and, as a result, avoid the problem of consumer confusion and free riders. Furthermore, in countries where all types of batteries are covered under mandatory EPR programmes (e.g. Switzerland, Austria, the Netherlands), relatively high collection rates have been demonstrated as compared to having separate collection paths for different batteries. While an EPR programme may not be a high priority for certain types of batteries that are relatively non-hazardous, their collection and recycling in a universal battery programme still provides environmental benefits from recycling of non-renewable resources.

3.1.8 Overlapping EPR programmes for components in complex products

In some countries, in the case of EPR programmes for complex products (EEE and cars), some of the components within the products (e.g. batteries in EEE, batteries, tyres and EEE in cars) are covered in other EPR programmes. On one hand, this may create a synergy in developing an efficient logistics for take-back and treatment. On the other hand, it may cause confusion on the side of producers and consumers, and delay the implementation of the programmes.

Some governments have delayed programmes to avoid overlapping producer responsibilities. In Austria, where concerns about the hazardous substances in the batteries in discarded mobile phones were raised, the government decided to wait to implement any measures until the proposed EU Directive for waste EEE is enforced. In Sweden, an EPR programme for tyres was introduced prior to the enforcement of the EPR programme for cars. It was decided that the tyres attached to the cars would be exempt from the EPR programme for tyres.

3.2 Voluntary or Mandatory EPR programmes

One major difference in the implementation of EPR programmes is whether the programmes are voluntary or mandatory. For the products evaluated, there is a definite shift from voluntary initiatives of producers to the introduction of mandatory programmes by governments, or a combination of both. The primary reasons for the shift include the free rider problem and the apparent ability to achieve higher collection, reuse and recycling rates with mandatory programs, particularly those with targets for collection and recycling. In general, mandatory EPR programmes with mandatory numerical targets achieve higher collection, reuse and recycling rates as compared to voluntary programmes (e.g. battery collection in Switzerland, the Netherlands, Austria). Setting targets for the operation of collective systems with the threat that an eco tax or retailers' collection responsibility will be imposed is also effective in meeting the targets (e.g. Belgium for batteries, Germany for DSD system).

Almost all the voluntary programmes suffer from free riders, caused either by deliberate abuse by non-members or by confusion of consumers who return the products of non-members to the members-only collection system. To avoid the free rider problem legislation typically mandates that all producers of the product must either be individually responsible or participate in the collective system. (e.g. Germany for packaging, batteries for Belgium).

Aside from re-enforcing the existing voluntary scheme, the threat of the introduction of a mandatory EPR programme often leads to the establishment of a voluntary one (e.g. batteries in the United States where different states start to enforce EPR programmes at the state level).

Mandatory programmes could have another benefit. People in charge of EPR programmes and end-of-life management in a company have found it easier to communicate the necessity to allocate resources for the fulfilment of EPR requirements if the programme is mandatory (Tojo, 2001).

3.3 Allocation of responsibility

The types of responsibility allocated to producers in different programmes can be divided into physical responsibility and financial responsibility. The level of co-operation among the producers distinguishes the form of responsibility as either collective or individual.

3.3.1 Physical and financial responsibility for collection

Different EPR programmes take different approaches in allocating physical and financial responsibility for collection. In some cases, producers (manufacturers, importers) and retailers have jointly created their own collection infrastructure and manage the system (e.g. packaging in Germany, aluminium cans and PET bottles in Sweden). This means that producers and retailers bear both physical and financial responsibility.

Many programmes allocate partial or full responsibility for collection to retailers (distributors) (e.g. EEE in Japan, The Netherlands, Norway and Switzerland). Retailers are typically responsible for collection of a discarded product when they sell a similar new product (old-for-new) (in all the EEE programmes mentioned above). In addition, some programmes (e.g. Japan, Norway, Switzerland) make retailers responsible for collection when a consumer wishes to bring back a discarded product that the retailers themselves sold before. Aside from the Japanese system where the last-owner-pays for the collection, retailers in other systems are both financially and physically responsible for collection. In the case of EPR programmes for batteries in the United States, the collection by retailers is fully financed by the licence fees paid to the PRO by producers.

Retailer responsibility for collection provides the possibility of reverse logistics, where trucks for new product delivery can be used for take back of old products. Retailer responsibility also takes advantage of the consumers' tendency to bring back a used product to make sure that they can get the correct new product (e.g. batteries) (Beaurepaire, 1997). Finally, some retailers have experienced that participating in the programme help consumers identify their shops (e.g. batteries in the United States). A pilot project for the recycling of used electronics in one state in the United States found that 7.5% of the consumers go to the retailers participating in the pilot project for the first time, because of the collection service (Minnesota, 2001).

Local governments are often involved in collection. In some cases, local governments bear both financial and physical responsibility (e.g. household packaging in the Netherlands, Japan). In others, manufacturers and importers cover the full costs of collection (e.g. batteries in Sweden) or partial costs (e.g. packaging in France; batteries in the United States), and local governments bear physical responsibility for establishing collection points (e.g. batteries in the United States) and for collection of discarded products that are not covered under the producer or retailer take back responsibility (e.g. EEE in the Netherlands, Norway and Japan). Sometimes public, as well as private institutions co-operate in collection (e.g. batteries in the United States).

One of the common reasons why local governments are involved in, or responsible for, collection is that they have an existing solid waste collection infrastructure that need not be duplicated. This is particularly the case for products that have been traditionally collected and recycled by local governments, such as packaging. Although a shift in financial responsibility may be called for, local governments often wish to retain their roles for end-of-life management for employment reasons and to ensure that products are collected and managed properly. However, producer responsibility for at least part of the collection of discarded products not only reduces the financial burden of local governments, but also stimulates innovation in transportation and collection logistics, and in design of products for easy collection and sorting.

One of the challenges in the collection of post-consumer products is the availability of space. Small retailers and local governments with limited storage space may not be able to store sufficient quantities of discarded products for long periods to make transport more efficient. As a solution, many EPR programmes for large products (e.g. EEE in Norway, Japan, Sweden) mandate that producers set up take-back sites, where the parties responsible for collection can bring the collected products. In the case of deposit-refund systems for aluminium cans and PET bottles in Sweden, compensation has been provided to the retailers for accepting each aluminium can and PET bottle to help pay for storage space and other collection expenses (Lindhqvist, 2000).

When producers are responsible for sorting collected products (e.g. batteries), they have incentives to introduce measures to facilitate sorting, such as labels and codes to distinguish mercury and non-mercury batteries and for distinguishing different types of plastics.

3.3.2 Responsibility for recycling and collective versus individual responsibility

In most of the EPR programmes, producers bear financial responsibility for recycling (and environmentally sound treatment of the residues), while delegating the physical responsibility to a third party by establishing a collective system together with other producers. The limitation of the capacity and resources of individual producers, the difficulty and inefficiency of establishing multiple recycling infrastructures, and the inefficiency of individual producers negotiating with different actors for end-of-life management are the main reasons for the establishment of a collective system.

In the case of packaging and batteries, almost all the EPR programmes are run collectively by a third party organisation, or Producer Responsibility Organisation (PRO). Typically, the members of the PROs pay fees to the organisation that are based on the material (packaging), the types of the products and the substances contained in the product (batteries) and on the weight of the products. The fee structure usually reflects the cost of the collection and recycling of such materials, and as mentioned before, encourages a producer to work on design changes and substitutions of materials, so that the total fee paid decreases.

In the case of complex products, such as cars and EEE, the setting of the fee in a collective system is not as straight forward. In a system where the fee is determined merely by the type of the product and its weight/size, the environmental characteristics of the products (e.g. easy to disassemble, reuse and recycle and containing less hazardous substances) that affect the costs for end-of-life management would not be reflected. This would result in an overpayment by manufacturers that work hard on design for end-of-life management to support the recycling of products from manufacturers who do not. Consequently, there is less incentive for design changes in collective systems where flat fees are charged. The anticipation of the problem of not being able to set differentiated fees that reflect the environmental characteristics of the product was one of the reasons why the Danish system did not incorporate the concept of EPR in their waste EEE regulation (Christiansen, 1999).

Collective systems have been developed not only under an EPR programme where producers are responsible for the take back and recycling of an old product when they sell a similar new product (old-for-new, one-for-one responsibility, e.g. EEE in Sweden), but also under the programme where the producers are responsible for their own discarded products (brand-related responsibility, e.g. Switzerland for all the EEE covered under the Ordinance, the Netherlands for cars and EEE except for ICT equipment).

In contrast to collective systems, where it is difficult to achieve design incentives through collective fee structures, an EPR programme where producers receive brand-related responsibility and the responsibility is carried out individually (e.g. cars in Sweden, four home appliances in Japan) can provide greater incentives to manufacturers to establish communication between designers and end-of-life managers and to strive for design for end-of-life management. Producers have a strong incentive to determine the real costs of recycling their products in order to negotiate with recyclers. In the case of the Japanese EPR programme for EEE, for instance, the individual physical responsibility allocated to producers has resulted in direct communication between designers and recyclers (Section 2.4.4).

3.3.3 Existing, orphan and new products

In the case of products with long life span (cars, EEE and rechargeable batteries), the allocation of responsibility for new, existing and orphaned products is essential. Various measures have been taken for cars (differentiated enforcing timing for existing and new cars, and differentiated recycling rates) and EEE (applying old-for-new rules, making the last owners pay, or making all the existing producers responsible for existing products). There are no differences between new and existing products with the last-owner-pays system, nor are there orphaned products. But the disincentive for collection created by the payment requirement has resulted in continued disposal of products in the municipal waste stream and illegal dumping (discussed further in Section 3.4).

3.4 Financial mechanism

In all EPR programmes, it is ultimately the consumers who bear the costs. The questions are when, and in what manner consumers pay? From this perspective, there are three basic financial mechanisms: 1) visible advance disposal fee systems; 2) invisible advance disposal fee systems; and 3) last-owner-pays systems. Among the visible advance disposal fee systems, a deposit-refund system, with its unique characteristics of the deposit, will be discussed separately.

3.4.1 Visible advance disposal fee systems

In a visible advance disposal fee system, the consumer is made aware that a specific amount of the purchase price of a product goes to support an end-of-life management system for that type of product. Examples include some programmes for EEE (e.g. Netherlands except for ICT equipment, Switzerland for refrigerators, freezers, air conditioners and ICT equipment) and for cars (e.g. Netherlands).

One purpose of visible fee systems is to make consumers aware that they are paying for the end-of-life management of the products. While this may have a general educational effect, the true advantage would come from a differentiated fee that would reflect the individual product's recycling costs or design for recycling characteristics (e.g. hazardous substances in the product, ease of disassembly).

However, so far, advance disposal fees have not been differentiated within the same type of products. The flat disposal fee does not give any signal to the consumers as to which products are more recyclable or less environmentally harmful at the end of their lives. Moreover, due to the uncertainty of the development of future recycling technology and the market for recycled materials, it is very difficult to calculate the costs of future recycling, especially for products with a long life span.

3.4.2 Invisible advance disposal fee systems

Most of the EPR programmes for packaging and batteries completely internalise the costs of end-of-life management within the price of the product and make it invisible for consumers. Producers of ICT equipment in the Netherlands also chose the invisible advance disposal fee system scheme. The advance fees can be collected either directly from the consumer at the point of sale or can be collected from producers based upon their total sales.

One advantage of an invisible fee is that the consumer does not perceive the added cost of the product as a government-imposed tax. Instead, it is part of the cost of production, like labour or materials. An invisible fee leads to efforts by producers to reduce the costs for end-of-life management so that the final price of the products is as low as possible.

In the case of packaging and batteries, as discussed in Section 2 and 3.3, the cost for implementation is covered by the fees collected by the common scheme or by the government. The fees are set depending on the type and weight of the packaging and batteries that are sold on the market.

3.4.3 Deposit-refund systems

A traditional deposit-refund system, which has been used for packaging in many countries, has consumers paying the deposit at the time they purchase the product and receiving the same amount as a refund when they return the used product to the collection system. Most deposit-refund systems achieve very high collection rate because of the financial incentive for return by the consumer. The high collection rate, in turn, encourages producers to maximise reuse opportunities, to improve the recyclability of the materials and to make the recycling as economically efficient as possible.

The traditional deposit-refund system has three financial sources for operation of the system of collection, reuse and recycling: 1) unredeemed deposits from products that are not returned; 2) sales of materials that are reused or recycled; and 3) interest gained from the deposits while they are pooled in the fund. If most of the products are returned for refunds, there is a risk that limited funds are available for the administration of the system.

A financial mechanism used for aluminium cans and PET bottles in Sweden is a combination of deposit refund system and invisible advance disposal fee system. In this system, in addition to the deposit, which is the same amount as the refund, administration fees are added in the price of the product. This hybrid system makes it possible to give a reasonable financial incentive to the consumers while financing the collection and recycling system in a sustainable fashion.

3.4.4 Last-owner-pays systems

The last-owner-pays system has been used for four home appliances in Japan and for certain EEE in Switzerland. In Japan, the producers must announce in advance the fees that are charged for take back and recycling of their products that are currently collected, and the retailers, for their collection programmes. In Switzerland, the collective scheme determines the flat fees to be charged to the last owner, depending on the weight and type of the products. The system has the advantage of making the price as close to the actual recycling costs as possible. Moreover, as long as the infrastructure for end-of-life management exists, the problem of existing and orphaned products is eliminated.

However, as discussed earlier, mixed disposal in the municipal waste stream and illegal dumping, including the export of discarded products as second-hand products, have been experienced in these systems.

3.4.5 Management of the fees

For advance disposal fees, the management of the collected fees also affects the implementation of the EPR programme. In cases where fees are handled collectively, they often function like a pension system for retirement. Fees are collected for new products sold and used to pay for recycling of old products that are discarded now. For the products that have a long life span (e.g. EEE, cars, rechargeable batteries), fluctuations in recycling costs, changes in revenue from recycled materials, and changes in the number of new products sold create uncertainties in the financial management of the scheme over the long term. This is especially the case if the sale of a certain product (e.g. nickel-cadmium batteries) is banned, while the existing products are still collected for recycling.

A new approach is for the individual producer to collect the recycling fee when the product is sold, but to set it aside to be used for the recycling of that particular model of product when it reaches end of life (e.g. cars in Sweden). This approach provides the producer with a higher incentive for design for end-of-life management than does the "pension" approach, because the fees collected are actually paying for the recycling of the product model being sold. However, this approach can result in a long delay in programme implementation for durable products, without an interim programme with a different financing mechanism, as the recycling of products sold today will not take place for years. Orphaned products can be a problem if companies cease to operate without protection of the fees collected from creditors.

Use of insurance as a financial mechanism for EPR programmes for products with long life spans has been offered as an alternative and has started to take place in the Swedish auto industry and has been discussed for EEE in Sweden. Use of insurance, among other things, will eliminate the problems of orphaned products. However, considering all the variables (future recycling costs, hazardous substances in the product, products' life time, the reinsurance costs, the estimated capital yield), it would be difficult to differentiate premium costs depending on the environmental characteristics of the products, as has been advocated. Moreover, the existence of the third party in the middle may hinder the communication between producers and recyclers.

3.5 Establishment of requirements

3.5.1 Use of targets for collection, reuse and recycling

As seen from the evaluation of the EPR programmes for packaging (e.g. Germany, Austria) and batteries (e.g. Switzerland, Belgium, Austria, Netherlands), establishment of mandatory targets by the government has been effective in achieving high collection, reuse and recycling rates compared to the EPR programmes that do not have the targets (e.g. batteries in Sweden). The high targets also trigger design changes for increased recyclability (e.g. uniformity of type of plastics, development of recyclable plastics found both in EEE and car industries), as achieving the targets with conventional product design would be either impossible or very costly. Similar design change have been observed from use of refillable quota for beverage packaging in Germany, where the threat of enforcement is the imposition of a deposit-refund system if the quota is not achieved.

High collection and recycling rates have been achieved without an EPR programme when there is a strong commitment by a government that has both the collection and recycling infrastructure and financial resources to achieve high targets (e.g. Denmark). However, it is often the lack of physical capacity and/or financial resources that lead to the introduction of an EPR programme.

Separate collection of discarded products from the rest of the waste stream requires the participation of consumers. Factors that affect the level of co-operation of the consumers include financial incentives (e.g. deposit-refund system), convenience (e.g. proximity of the collection points, marking of products to easily identify materials for recycling), and the awareness of consumers. In general, it is difficult to discard a large product in a regular waste bin, so the consumer is motivated to think about using an established collection point before discarding it. The smaller a discarded product is, the more incentives and convenience are needed for the product to be collected separately.

Collection targets have a different purpose and effect than recycling targets. Collection targets drive development of collection infrastructure, and recycling targets drive recycling technology. The use of one or the other (or both) depends on the status of the collection and recycling infrastructure and on the availability of information with which to set a realistic target and monitor performance of the system. Most of the EPR programmes for packaging and cars, which have mature collection programs, set recycling (and sometimes reuse/refillable) targets, whereas most of the programmes for batteries set collection targets instead. In the case of EEE, with an immature collection system, the proposed EU Directive has established an absolute target for collection (4 kg/person, instead of a percentage), but existing EPR programmes have not attempted to use collection targets but set recycling (and sometimes reuse) targets to drive the recycling technology.

For products with a long life span, establishment of a collection rate target is challenging. The difficulty in setting a percentage target based on sales of a product is that the products sold in one time period are not the same as the products discarded in that time period, and sales can vary dramatically while the collection numbers stay the same. Even with an absolute target, like the collection target in the proposed WEEE Directive, gaining reliable statistics to estimate the total amount of discarded products for setting and updating the target is challenging. As a result, EPR programmes tend to emphasize recycling targets, despite the need for collection targets to improve collection infrastructure, such as for EEE.

For short life span products, such as packaging, a combination percentage target can be used. While it may be called a “recycling” target, the numerator of the target usually includes collection and recycling, and the denominator of the target is the amount of the product sold. It is a reasonable assumption that the products sold in a given time period is the same as the products available for discard during that time period.

In the case of batteries, the difficulties lie in the collection, not the recycling (Morrow & Keating, 1997). Meanwhile, the long life span of some of the batteries makes it difficult to establish the appropriate denominator. Percentage collection targets for rechargeable batteries have been controversial for this reason (e.g. rechargeable batteries in the United States) (Raymond, 2001).

3.5.2 Use of substance and landfill restrictions

While producer responsibility for recycling and the use of high recycling rate targets should provide incentives to producers to redesign products to remove hazardous substances to make them easier to recycle, some programmes have gone further and have directly restricted the use of certain hazardous substances in the product as the most effective way of reducing these substances in the waste stream (e.g. cars, EEE, batteries). Substance bans have played a significant role in triggering product redesign and material substitution, although the bans have been criticized as ignoring the potential impacts of the substitutes and life-cycle tradeoffs that may create health and environmental problems in other life-cycle stages while reducing problems in waste management. Even the threat of a ban in proposed legislation has helped trigger development of alternatives (e.g. rechargeable batteries without cadmium, lead-free solder) and, in some cases, increased collection and recycling activity (e.g. nickel-cadmium batteries) to avoid the ban. Similarly, a ban on landfilling or incineration of a product containing hazardous substances (e.g. nickel-cadmium batteries) can also stimulate the development of alternatives. Many EPR programmes are complemented by substance restrictions (e.g. proposed Restriction of Hazardous Substances Directive for EEE, EU Directive on end-of-life vehicles) or restrictions on landfill (e.g. EEE in The Netherlands, Sweden, batteries in Switzerland).

3.6 Systems surrounding the products

3.6.1 Existing infrastructure around reuse and recycling

With regard to recycling and environmentally sound treatment of the discarded products, manufacturers either contract with the existing recycling firms and delegate their physical responsibility, or co-operate with existing recyclers that have high skill and establish their own recycling facility. In cases where an established collection and recycling infrastructure there exists (e.g. dismantlers and scrappers for cars in most countries, some recycling facilities for EEE in Sweden), establishment of a new infrastructure by the producers may pose a threat to the existing business, and is not necessarily welcome. In such cases, producers often strive to establish a network with existing end-of-life managers and contract with them.

Where a large number of recycling facilities exist (e.g. cars), producers may prefer individual responsibility as compared to products for which a limited number of recycling facilities exist, where a collective system is usually used (e.g. nickel-cadmium batteries). An existing collection and recycling infrastructure for one product covered by an EPR programme could serve another product where the products and materials are similar (e.g. use of battery collection and recycling infrastructure for certain EEE). In cases where there are few existing facilities (e.g. EEE in most countries), the involvement of manufacturers in establishing new facilities has been necessary.

3.6.2 Structure of the market

The greater the number of producers and distributors of a product and the more dispersed the distribution network is, the more difficult it becomes to coordinate and control their actions. Among the four product groups discussed in Chapter 2, packaging has the most dispersed distribution networks, as virtually all the products available on the market use some type of packaging in one or several part of their life cycle.

When the packaging is used for the same purpose between the same actors (e.g. between a material supplier and a component supplier, a component supplier and a manufacturer, a manufacturer and a wholesaler, a wholesaler and a retailer), it can be both economically efficient and logistically possible to use reusable packaging instead of one-way packaging.

Between the interface of the final distributor and the consumer, however, one consumer receives different packaging materials from numerous distributors. In such cases, except for some of the commonly used local distributors (retailers), it becomes economically inefficient and logistically impractical that a consumer brings back each packaging to all the distributors. Therefore, the separate collection takes place by the types of materials, not by where the package comes from. Most of the successful collection programmes use a deposit-refund system and/or kerbside collection system (see Section 2.1). The bring system for glass have achieved very high collection rate. The reason may be the consumers' reluctance to put the glass in normal waste bins.

In the case of batteries, just as packaging, a large number of distributors have direct contact with the consumers of primary batteries (particularly alkaline batteries), although the type of distributors for some primary batteries (e.g. button cells) and rechargeable batteries is more limited (e.g. retailers for tools, EEE, light, security equipment). Many of the rechargeable batteries are including the bodies of other products (e.g. EEE, cars).

For those batteries where consumers have the natural tendency to bring back the used ones to the retailers to exchange with the new ones (e.g. button cells, lead-acid batteries in cars, rechargeable batteries within EEE), use of such reverse logistics is useful for collection. As consumers do not distinguish the difference in type, the rest of the batteries are often collected at the universal collection points set up mostly by retailers and local governments.

Concerning large EEE, the number of the distributors is limited to the local retailers and large discount shops. Most of the EPR programmes utilize these distributors for the collection of old products when they sell new products and deliver them to the consumers.

With regard to small EEE, the number of the distributors is higher compared to large EEE (e.g. electric shavers can be sold not only at the same places as TVs, but also at super markets, kiosks, and the like). However, EPR programmes that cover both small and large EEE (e.g. the Netherlands, Norway, Sweden, Switzerland and the proposed EU Directive) allocate the same collection responsibility to the distributors. The difference in these distribution networks, and the corresponding difference in the difficulty of establishing the infrastructure for collection, was one of the reasons that the Japanese scheme covered only the four large EEE at the initial stage, and the Dutch scheme set a differentiated starting point between the large home appliances and the small home appliances.

Regarding cars, the limited number of distributors (dealers), and in most countries, the established practice of accepting an old car when selling a new car, facilitates the collection of a recycling fee at the purchase of a new car, and the collection of an old car (which may be used in the second hand market or scrapped).

The large number of packaging producers and fillers (see Table 2-1), as well as the diversified distribution network, makes it necessary to establish a PRO to fulfil the responsibility of the producers in an economically feasible way. The battery producers under an EPR programme also established a PRO.

When a PRO begins to have a dominant power within the collection and recycling market, the issue of fair competition may be questioned. Cases that have been observed include the DSD system for packaging, and the Swedish collective programme for EEE (ENDS, 2001h; Miljörapporten, 2001).

3.6.3 Impacts on the market

With regard to materials used for packaging a well-established recycling market exists for the metals (e.g. aluminium, steel), glass and paper. Due to a variety of uses in different products, the metals recycled from packaging have not created any major changes in the market for recycled materials. The increase in the recycling of glass, paper and cardboard, however, has in some cases saturated and the market and caused a fall in the market price of the secondary materials (e.g. Germany, Japan) (OECD, 1998b; Tojo, 1999). Problems occur when collection requirements are instituted before recycling capacity is online, such as the case of PET bottles in Japan, where the development of the recycling facility could not meet the rapid increased collection of PET bottles, causing storage problems in some local governments (Nikkei, 2000).

Some argue that in order for the recycling system to work properly, it is important to develop the market for the secondary materials first. Others argue, however, that after a certain time passes since the market for recycled materials become saturated, a new market will occur. Moreover, it is only after the saturation of the materials occurs, that industry starts to invest to utilise such materials. Without knowing if there are abundant materials available in the market, it is difficult for a private industry to invest in technologies or products that enable utilisation of such materials.

For instance, in some countries (e.g. Japan), the drop in the price of recycled metals was one of the reasons that triggered the development of EPR legislation (Automotive Recycling, 2001).

Concerning EEE, due to the relatively short time of the implementation, changes in terms of material and component demand and supply have not been clear. However, it is anticipated that the raw material suppliers (e.g. plastics) will start to participate in the recycling business to survive in the market. On the other hand, one of the computer manufacturers mentioned that despite their will to use recycled plastics and pay prices higher than the virgin materials, they struggle with lack of recycled plastics in the market (IBM, 1999).

Under the last owner pays system (e.g. large EEE in Japan), development of the second-hand market as well as repair shops has been remarkable, reflecting the consumers' reluctance to be the last owner of the product paying the fee (Tanaka, 2001; Kaneko, 2001).

3.6.4 Structure of the companies

EPR can affect the business models of companies. In cases where the manufacturers participated in the establishment of the recycling plants (e.g. Japan), they established either one department or a separate company that is in charge of the management of the plants.

When a manufacturer sells the function of a product while retaining the ownership of the product (e.g. copier machine), it naturally considers end-of-life management of their products. For example, as of 1999, Fuji-Xerox which rent approximately half of their products directly to their customers, achieved the average inclusion rate for recycled parts on a volume basis across all the products of 19%, and the total reuse ratio for collected parts of 43% (Fuji Xerox, 2000; Miyasaka, 2001).

3.7 Awareness and perception of affected actors in the society

Although consumers are more and more aware of the environmental and health impacts of waste, their awareness does not necessarily affect their purchasing behaviour. When consumers purchase products such as batteries, EEE and cars, their generalized concerns about waste are rarely reflected in their decisions. Consequently, customer demand, which is among the strongest driving forces for a company to invest, is not a strong driving force for design for end-of-life management.

This low demand from the consumers is one reason for the development of an EPR programme to provide companies with incentives to consider the environmental impacts of their products at the post-consumer stage. The implementation of the programme, in return, raises the awareness of the consumers. If the consumers have problems with waste separation, for instance, they may communicate their frustration to the producers. This would help establish a communication path between consumers and manufacturers as well. The communication from consumers to manufacturers may lead to the improved design of packaging (e.g. elimination of unnecessary packaging, or reduction of different types of materials used) (Tojo, 2001).

1) Convenience, 2) financial incentives, and 3) information on the scheme, are the factors that determine the consumers' willingness to separate the discarded products from the rest of the waste stream and bring it to the collection points. Collection systems that achieve high result seems to fulfil one or more of the above criteria. However, a battery collection pilot project that took place for 6 months (November 1987 – May 1988) on an island in Denmark, with massive information efforts (after the intensive campaign, 92% of the population were aware of the programme), achieved only low collection results (Lindhqvist, 2000). These results illustrate that even when there is ample information, mere information cannot overcome inconvenience and lack of financial incentives.

4. Summary

Now that EPR programmes have been operating for at least a few years in a number of political settings and for a number of different types of products, we should be able to begin to evaluate EPR in practice rather than principle. Unfortunately, without a major research effort or much greater cooperation among EPR programmes in measuring and reporting of performance and cost data in common formats, the information base for a thorough evaluation of this new policy will remain lacking. Furthermore, it is difficult to compare EPR-based policies with other policies that attempt to achieve the same goals, because there are so many different elements to EPR programmes in practice, (e.g., financing mechanism, PRO structure, adjunct recycling rate goals and product material bans) that it will always be difficult to say whether the producer responsibility core of these programmes is responsible for their performance, good or bad. Some of the same elements found in EPR programmes, particularly recycling rate goals and product material bans, are part of other policy instruments that attempt to achieve the same results without producer responsibility.

While a thorough evaluation of EPR as a policy principle cannot yet be performed, we believe that there are some tentative conclusions that can be drawn from the examination of the programmes that have been operating to date. These conclusions, drawn mostly from the evaluation of the major programmes for the four products focused on in this paper, are more qualitative than quantitative and are proposed in this working paper as hypotheses that are in need of further testing.

EPR programmes generally increase collection and recycling rates significantly by making resources available that governments, by themselves, through taxpayer funding, are typically unable to commit.

The often-asked question, whether EPR programmes are more effective at increasing collection and recycling rates for products in the waste stream than other policies, does not have a straightforward answer. It is conceivable that governments could devote the same resources, through taxpayer funding, to establishing the necessary collection and recycling infrastructure as have been created through EPR programmes. In fact, Denmark appears to have decided that, for some products, it is preferable to have governments responsible for collection and recycling in order to ensure high environmental standards and to avoid export of waste as second-hand products. But most governments have resorted to EPR because they are confronted with growing quantities of used products in the municipal waste stream without the possibility of raising the resources to create the necessary infrastructure and develop the necessary expertise for sorting complex products and handling hazardous materials in the products. They have also seized upon the idea that EPR provides incentives to producers to design cleaner products that are easier and more cost-effective to reuse and recycle.

The producer responsibility element of EPR programmes, when not diluted by too many intermediaries, has resulted in an apparently effective feedback loop from waste managers to producers for stimulating changes in product design.

The small amount of research and reporting that has been performed on this subject suggests that EPR programmes can create effective feedback to product designers to design cleaner products that are easier and more cost-effective to reuse and recycle. Of course, product design changes often have several different drivers, and pressures for recycling may come from many sources (public perception, customer demands, government procurement), so there is no definitive way to correlate design changes to the shift in responsibility for end-of-life management to producers. As discussed further below, different types of EPR programmes provide greater design incentives, and some programmes, such as uniform advance disposal fees collected by the government to fund government-run recycling programmes, provide little or no feedback to producers for design changes. Also discussed further below, certain products lend themselves better to establishing the design feedback loop through EPR programmes than others.

EPR programmes appear to work for a variety of products, both durable and non-durable, simple and complex, but the focus has been on products that are high-volume, difficult to manage, and contain hazardous substances.

Are there some products for which EPR is not suitable at all? It is possible, but not desirable, to have an EPR programme for any identifiable product that can be sorted out of the waste stream. Most EPR programmes, however, focus on products that have high volume in the waste stream, are large or difficult to manage, and/or contain substances that are potentially damaging to human health or the environment. The collection and recycling infrastructures of local governments have generally not been designed to accommodate these types of products as they increase in the waste stream, and it is generally feasible for the producers to reduce the impacts of these products in the waste stream through redesign, given the proper incentives.

The programmes evaluated showed significantly enhanced collection and recycling rates for non-durable, simple products, such as packaging, and for durable, complex products, such as automobiles. But, as discussed further below, it is more difficult to create an incentive for product redesign through financial responsibility for durable, complex products.

Voluntary EPR programmes are best suited for products that have higher value at end of life and where consumer demand for design for better end-of-life management can differentiate the participating brands in the marketplace.

Clearly, there are some products separated from the waste stream and recycled on a routine, profitable basis without any involvement by the producer or the government. At some level of profitability, producers may want to enter the recycling market to cash in on the residual value of their products or to take advantage of remanufacturing opportunities that can save a large percentage of manufacturing costs. There are also other intangible reasons for voluntary adoption of EPR, such as enhanced customer loyalty and green marketing, but these factors are not enough by themselves unless the value of the end-of-life product can make EPR at least close to a break-even proposition. A subset of products that cannot be readily distinguished from other similar products can create difficult free rider problems for voluntary EPR programmes, as in the case of programmes for nickel-cadmium batteries, where consumers place other types of batteries in the collection system.

EPR programmes with government involvement in enforcement against free riders (either mandatory legislation or negotiated agreements) appear to produce higher collection and recycling rates than purely voluntary programmes.

Without some government involvement in enforcement against free riders, voluntary EPR programmes have suffered from both low participation from producers in financing the programmes and the inability to prevent non-participating products from entering the system. Mandatory legislation eliminates the major portion of the free rider problem by requiring all producers to take responsibility for a particular class of products. Often, a negotiated agreement between producers and the government will provide for government enforcement against free riders.

EPR programmes with goals or mandates set by government for collection and recycling are able to produce higher results than those without such goals, unless there are other significant incentives for consumers to participate.

Establishment of numerical targets that are either mandatory or supported by a trustworthy threat of regulatory intervention is effective in attaining high collection, reuse and recycling rates. Collection targets are typically set to increase the separate collection from the other waste stream or to reduce littering problems, while recycling targets are set to drive design changes and technical improvements, leading to the reduction of environmental impacts of products from the post-consumer stage not only at its end-of-life, but also at source.

While product design changes may be driven by the subtle shift of end-of-life responsibility to producers, the blunt instruments of recycling rate mandates and material bans that are part of certain EPR programmes account for much of the attention to product redesign.

Experience has shown that mandated requirements encourage manufacturers to explore the possibilities of redesigning products. Such requirements are frequently put on the content of toxic chemicals but also on the achievement of recycling rates for specified materials (e.g. minimum recycling requirements for the specified packaging materials).

Anticipated or existing mandatory EPR programmes that mandate individual responsibility to producers and demand specified performance (or if the system is set in such a way that the costs that industry are paying reflect the design change) give definitive incentives for design change.

The establishment of a successful collection system is the prerequisite for a successful EPR programme. Different types of collection systems can produce high collection results as long as the resources are available to provide: 1) financial incentives to consumers, 2) convenience for consumers and/or 3) information for consumers.

The most difficult part of the collection system is to motivate the consumers to actively participate in an EPR programme and separate the end-of-life product in accordance with the system requirements. The clear financial incentives for collection provided by a deposit-refund system can only be substituted by very high level of convenience if the same collection rate is aimed for. The problem with the financial management of a traditional deposit-refund system can be overcome by combining the deposit refund system and advance disposal fee system.

The necessity to provide consumers with financial incentives, convenience and information increases when the size and the weight of the products become smaller, and when only a fraction of products that have similar appearance and function are covered by an EPR programme.

The scope of products covered by an EPR programme can be an important factor in the success of the programme.

An EPR programme must be planned in a way that it is easy for the consumer to understand which products are included and not included. Experience shows that consumers would not likely distinguish various types of batteries or products with relatively subtle differences. The message to the consumer must be straight-forward and immediately understandable.

When developing an EPR programme for complex products which include components that are covered by another EPR programme or a collection system, the existing and new programmes should be coordinated so that producers of the respective products are given clear and undisputable responsibilities.

When there is an existing physical infrastructure for collection and recycling prior to the introduction of an EPR programme, it is more efficient and results in faster implementation to further develop the existing system utilizing the available skills and knowledge as much as possible.

In developing an EPR programme, issues such as: the number of producers and distributors that exist in the market; the financial and physical capacity of the individual manufacturers to establish and manage the end-of-life management of products; the number and capacity of existing end-of-life managers in the market; and the size of the individual products must be considered.

Taking advantage of these available skills and knowledge does not necessarily mean the operation is governed and managed by the same juridical entities prior to the introduction of an EPR programme. Producers may shift the ownership of the end-of-life management facilities by, for example, purchasing the dismantlers and recyclers, while retaining the existing capacity and enhancing it further.

The financial mechanism that works best for promoting the aims of EPR programmes depends, to a certain extent, on the type of product. For non-durable, relatively simple products, collective financing schemes with advance fees on new products can more accurately reflect the costs of collection and recycling of old products than for durable, complex products.

Most of the EPR programmes for packaging and batteries are financed by fees paid by the producers, based on the materials (packaging), types and substances (batteries) and the amount of products put in the market by the respective producers. The limited number of the materials used in these products as well as the relatively short life span of the products (except for the rechargeable batteries) allows the amount of fees to reflect the costs for the end-of-life management. The fees send direct price signals to producers to use less materials, or materials that are easy to identify, separate and recycle.

For durable, complex products, systems with individual financial producer responsibility for collection and recycling present an important opportunity to stimulate design changes that ultimately minimize the costs of recycling, but such systems fail to address orphan products, require more sophisticated collection systems, and make the enforcement of collection and recycling goals more difficult.

The properties of durable, complex products make collective financing ineffective at stimulating design changes. Individual producer responsibility offers an opportunity for competitive advantage to be gained by reducing recycling costs through product redesign. However, recycling of orphan products must also be financed, which is best handled by a collective system. Difficulties in calculating the cost of end-of-life management of a product that reflects all these variables pose challenges in establishing a collective physical arrangement with individual financial responsibility.

It is often difficult to set the collection targets in percentage, due to the difficulties in determining the basis for the calculation of the return rate. Providing the target in absolute number (e.g. kg of products) is an alternative, but the experience of such implementation is still limited.

A last-owner pays financing system, when coupled with individual physical producer responsibility for recycling, can be an effective incentive for changes in product design to reduce the costs of reuse and recycling, but creates a disincentive for consumer participation through the imposition of the fee.

Charging a collection and recycling fee to the last owner of a product is clearly a disincentive for consumers to participate in a recycling programme. This incentive may be overcome when there are no other convenient ways to discard the products, as in the case of large appliances. Where governments or private recyclers collect and recycle the products and charge the fee, these programmes have little to do with EPR. But when producers have physical responsibility for recycling the products, and for setting the fee, as in the Japanese large appliance programme, a strong incentive for changes in product design to reduce the costs of reuse and recycling is created as a result of competitive pressures to keep the fee low.

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