Unclassified **ENV/EPOC/WPNEP(2003)10/PART2/FINAL**

Organisation de Coopération et de Développement Economiques
Organisation for Economic Co-operation and Development

23-Oct-2003

English text only

ENVIRONMENT DIRECTORATE
ENVIRONMENT POLICY COMMITTEE

ENV/EPOC/WPNEP(2003)10/PART2/FINAL
Unclassified

Working Party on National Environmental Policy

PROCEEDINGS OF OECD SEMINAR ON EXTENDED PRODUCER RESPONSIBILITY: EPR PROGRAMME IMPLEMENTATION AND ASSESSMENT

PART 2: ASSESSING EPR POLICIES AND PROGRAMMES

OECD, 13-14 December 2001

For more information, please contact Claudia Fénérol
Tel: +33 (0)1 45 24 17 63, Email: claudia.fenerol@oecd.org

English text only

JT00152089

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FOREWORD

The OECD held a seminar on Extended Producer Responsibility Programme Implementation and Assessment, from 13-14 December, 2001. The main purpose of the seminar was to examine how governments are implementing EPR policies and to discuss their results achieved. The seminar created a forum for member governments to share information about their EPR programmes, discuss pressing issues and to explore ways to implement sounder and more effective EPR policies.

The seminar was divided into four sessions: two sessions consisted of panel presentations from member governments, academia, industry and other experts and two sessions comprised discussion of two consultants' papers with discussants. Over 50 participants attended the seminar, representing government, industry, academia and non-governmental organisations. The outcome of session discussions and recommendations gave OECD clear directions on where to focus its future EPR activities. In particular, participants stressed the need for further guidance on economics of EPR as well as the need for empirical data on the costs and effectiveness of EPR so as to allow for more conclusive policy and programme analysis.

The Proceedings of this seminar are divided into two parts: Part One consists of papers from Sessions 1 and 2. Part Two contains papers from Sessions 3 and 4. The Proceedings contain only papers submitted to the seminar; no presentations are included.

The opinions expressed in these proceedings are those of the authors, and do not necessarily reflect the views of individual member countries. The document is published under the responsibility of the Secretary General.

TABLE OF CONTENTS: PART 1

[The Table of Contents for Part 1 of this two volume set is included for information only.]

TABLE OF CONTENTS: PART 2

SESSION 2: ASSESSING EPR POLICIES AND PROGRAMMES

EVALUATING THE ENVIRONMENTAL EFFECTIVENESS AND ECONOMIC EFFICIENCY OF EXTENDED PRODUCER RESPONSIBILITY (EPR)

by
Michael Sturges
Pira International
United Kingdom

EXECUTIVE SUMMARY

Extended producer responsibility (EPR) is an environmental policy approach in which the producer's responsibility is extended to the post-consumer stage of the life cycle. In this way, it is intended that EPR programmes will reduce waste production and contribute to the decoupling of waste production and GDP and send strong signals to the producer to design products that are more environmentally compatible.

Within the OECD many EPR programmes are emerging for different sectors, waste streams and product groups. However, the application of EPR remains controversial, and policy makers are often challenged to prove that the economic costs of achieving EPR goals are justified in terms of the environmental benefits secured. Therefore, this paper investigates how the environmental and economic aspects of EPR can be evaluated.

Why do we need EPR and where can EPR be most effective?

The paper begins by presenting an economic rationale for EPR policies. In particular, the paper considers why EPR programmes are needed and the conditions under which they can be most effective.

Four possible scenarios for ownership of landfill and recycling operations are identified. These are investigated in a series of simplified economic models, presented in the Annex. The models show that, from both an economic and environmental viewpoint, the optimal mix of recycling and disposal may not be achieved without intervention. Further development of the models demonstrates that the optimal recycling rate is dependent on both environmental and economic influences.

A series of matrices produced at the OECD's 1999 Paris EPR Workshop further reiterate these conclusions. The matrices were developed as a screening tool to help policy makers select a course of action, and provide a way of visualising and evaluating the economic and environmental model for different situations. The matrices compare value on the y-axis and environmental impact on the x-axis. The value axis is not clearly defined, but value is considered to be a function of the internal costs of recovery, recycling and disposal compared to those of disposal. The environmental impact axis is a relative measure of the scale of the effects of the waste stream on the environment.

Where a material has a positive value at end of life – *i.e.* the costs of collecting, sorting and reprocessing the material are lower than the market value of the recyclate – market forces will generate recycling programmes regardless of the scale of environmental impact. The greater the end of life value, the more likely that development will occur. However, the models in Annex 1 suggest that the level of recycling that will be achieved with no market intervention is not the optimum recycling rate. Where the environmental impact of the waste stream is low, this is unlikely to be of significant concern. However, where the environmental impact of the material is medium to high, there may be a case for some market intervention in order to increase the recycling levels achieved.

Where the value of recovered material is negative – *i.e.* the costs of collecting, sorting and reprocessing the material are greater than the value of the recyclate produced – recycling will only occur with market intervention. Whether market intervention should be pursued or not may depend upon the relative scale of the environmental impact:

- If end of life value is low and environmental impact is low, market intervention to pursue increased recycling is unlikely to be justified;

- If end of life value is low and environmental impact is medium, a focus on increased recycling through an EPR programme may be warranted; and

- If end of life value is low but the scale of environmental impact is high, then the nature of the hazard requires that safe disposal or recycling is essential. In such instances where environmental impact is high a prescriptive approach may be required to ensure that high levels of environmental protection are achieved.

The matrices attempt to simplify the description of the environmental and economic situations into a series of discrete scenarios. In reality, the boundaries between each scenario become blurred. The availability of secondary material markets will be critical to the success of any policy objective. Changes in the supply side and the demand side may influence the economics of recycling, thereby moving the description of the situation from one matrix to another. This may be particularly important in capital intensive sectors, such as the paper and board industry. Imbalances in the supply and demand of materials driven by capital investments can lead to fluctuating virgin and recovered fibre prices. The instability generated in these situations can hinder the development of an optimal waste management strategy.

Life cycle cost benefit analysis as a tool for evaluating EPR programmes

The models and matrices investigated suggest that for an EPR programme to be justified the environmental and economic benefits must outweigh the environmental and economic costs. Therefore, an approach is required which allows environmental and economic costs and benefits to be considered on an equal footing.

The paper therefore advocates a life cycle cost benefit analysis (CBA) approach to evaluating the environmental and economic performance of EPR programmes. CBA is a developing technique that seeks to value externalities in monetary terms, and compare these against the internal costs of implementation. In this way, decision-makers and stakeholders gain insight into the trade-offs within and between economic costs and environmental benefits that are inevitably made when selecting and implementing policies.

There is no standardised methodology for performing CBA, but the following general principles should be incorporated into all studies:

- Transparency: a clear understanding of how the results are achieved, which assumptions are made, and which data are used should be provided;
- Comprehensiveness: all relevant environmental impacts should be acknowledged in some way;
- Time and spatial issues should be addressed;
- Stakeholder dialogue should be maintained to ensure that the systems modelled and assumptions made are in line with reality. Dialogue will also improve access to data;
- An evaluation of the methods applied by various researchers and consultants suggests that the following common themes apply;
- Environmental impacts are quantified using life cycle inventory analysis or life cycle impact assessment techniques; and
- Economic valuation techniques are applied to convert life cycle inventory or impact data into monetary units, which are then comparable with internal costs.

A general framework for performing CBA studies is put forward in this paper. It is important to recognise CBA as a useful tool rather than a decision rule. By asking the right questions it opens up the discussion and identifies key issues. Methodological and data limitations may prevent CBA from providing definitive answers to all the questions raised, but this does not render the exercise valueless. For example, it can be particularly useful to know the point at which costs of reducing a pollutant start to rise sharply. This information can then be used to make a first guess as to the level to which it is "reasonable" to reduce pollution. This approach is particularly appropriate where there is very limited information on the real risks that the pollutant poses.

In this way, CBA makes decision-making more transparent. It is an aid to the decision-making process, not a substitute for it. The decision-maker must still judge how to weigh up environmental effects, economic costs and distributional impacts of the different policy options and select the preferred option. Supplementary information from macro-economic analyses that consider the impacts of policy on economic indicators such as GDP, inflation rate, and the trade balance may also be of value to decision-makers.

Evaluating the internal costs of EPR programmes

A cost benefit analysis of public policy must focus on the total financial costs to society rather than the costs to individual groups. Affected parties may include:

1. Specific industry sectors;
2. Specific businesses;
3. Government departments and enforcement agencies;
4. Employees (in specific industries and businesses);
5. General public;
6. Special interest groups; and
7. Producer responsibility organisations.

The distribution of the costs that are incurred between these parties is important. EPR policy measures are not attractive (and are unlikely to succeed) if they imply a disproportionate burden on a particular sector. By identifying the effects on each of the stakeholders, an appreciation of the potential distribution of costs is afforded. Distribution effects may occur at a group level, at a national level, or at a regional level. However, determining the internal financial cost impact of the policy on each group is a difficult task – the financial flows between affected parties may be complex.

Available internal cost data has been collated and reviewed. Key points arising from the review are:

- Internal cost estimates show considerable variation – this may be due to inclusion of different parameters / processes / approaches to the calculations;

- There is inconsistency in the methods applied and it is therefore difficult to make comparisons between different sources;

- Often it is frequently not transparent as to whether costs are gross or net of revenues from recycling / energy recovery;

- Often it is not transparent as to whether costs include capital investment;

- Often it is not transparent who the costs apply to – industry, local authorities, government, *etc*; and

- Little is available on economies of scale, *etc*.

Although these available data can be used to quantify the order of magnitude of costs for waste management processes, detailed CBA studies to support EPR policy decisions may require considerable primary data collection in order to be justifiable to stakeholders.

Quantifying environmental benefits of EPR programmes

In CBA, the environmental impacts are quantified using life cycle assessment techniques. These environmental impacts are then converted into monetary values using economic valuation (monetisation), so that the environmental costs and benefits of the programme can be compared against the financial costs of implementation. Monetisation seeks to convert the environmental impacts of a life cycle assessment study into a single comparable unit (in this case a monetary unit such as Euros or dollars).

Economic valuation of environmental impacts can be achieved through a variety of techniques:

Contingent valuation is a state preference method which is used to ask individuals their willingness-to-pay (WTP) for the environmental good directly using a structured questionnaire.

The Hedonic pricing method (HPM) considers that the price of a good is a function of its attributes, including environmental attributes. For example, the price of a house is a function of the characteristics of the house, its neighbourhood, and environmental variables such as ambient air quality or proximity to countryside. HPM regresses house prices to statistically isolate the price differential due to the relevant environmental characteristic.

Averting behaviour method values environmental quality by looking at the expenditure people make for goods that can substitute or avoid for a decrease in environmental quality. For example, expenditure on bottled water can give an indication of people's WTP for preventing the adverse health effects from using bottles water.

There are limits to the application of monetisation. In some cases there is currently no sound understanding of the pathway between the inventory data and effects on the protection area, or no reliable economic values are available. In such cases the environmental concern cannot be quantified within the monetary valuation but must still be considered in interpretation and in making a decision based on the study.

To overcome this, methodologies should consider two types of impact category:

- Primary impact categories – those for which an economic valuation can be derived; and

- Secondary impact categories – those for which no economic valuation can be derived.

Where possible, these secondary impact categories should be quantified in traditional LCA impact assessment units. If this is not possible, the existence of the secondary impact category should at least be qualified for further discussion during the decision making process.

The different dimensions of recycling-related external effects are identified. Recycling affects the whole life cycle of materials. As a result, an EPR programme may incur a wide variety of environmental costs and benefits.

In the extraction stage, recycling may reduce demands for raw materials, thereby avoiding ecosystem damage and exhaustion of scarce resources. Transportation impacts of raw materials may also be reduced.

In the production stage, substitution of primary resources by secondary materials may reduce energy demands, air emissions and waterborne emissions. For example, the production of recycled aluminium requires 97% less energy than primary aluminium production. However, recycling can raise new environmental impacts, for example reprocessing impacts such as the decoupling of waste paper.

During the consumption stage, performance of the recycled product is critical. Secondary products may be less durable or cause operational problems. Conversely, some consumers may experience the purchase of recycled products as a positive environmental act, and thereby experience an improved feeling of well-being.

In the waste management phase, recycling prevents incineration and landfilling of solid wastes. Subsequently, emissions of pollutants to the atmosphere, surface waters and groundwater and disamenity impacts of landfill sites and incinerators are avoided, but the products of disposal (recovered landfill gas and recovered energy) are also displaced. Transport emissions associated with the collection of recovered materials and impacts associated with sorting must also be considered.

It is evident that the scope of the analysis must consider all aspects of the environmental costs and benefits across the entire life cycle.

Table 1: Main externalities for consideration in the evaluation of EPR programmes

Externality	Residual well-being effect
Global warming	Mortality and morbidity Damage to buildings, structures and materials Damage to forest resources and agricultural production
Air pollution	Chronic and acute morbidity Chronic and acute mortality Damage to buildings, structures and materials Damage to forest resources and agricultural production
Ground and surface water pollution	Safety and availability of drinking water Recreational value Biodiversity Damage to fisheries
Disamenity	Odour and Visual pollution Noise Congestion Willingness-to-recycle Convenience
Traffic accidents	Mortality Serious injury
Occupational health	Accidents Diseases
Macro-economic effects	Multiplier effects Employment

It is not the intention of this paper to evaluate all available economic valuations and how these were achieved, but for each of the possible externalities a brief discussion is provided and some possible valuations are highlighted. It is intended that the valuations presented will provide a starting point for further discussion and methodology development.

Examples of the costs and benefits of EPR programmes

Many examples of the use of CBA to evaluate EPR programmes can be found. A selection of case studies of the use of CBA is highlighted in this paper and the implications of these studies for evaluating EPR programmes and policies are assessed.

In particular, CBA studies investing three waste streams are considered, and implications on the use of CBA for evaluating EPR programmes are drawn:

General municipal solid waste

There are limitations to the methodologies and data applied in both case studies, but they both appear to lend support for the implementation of EPR programmes for household waste streams. The additional costs of EPR programmes that divert materials from disposal to recycling can be justified in terms of the reduced environmental impact of waste management and the avoided environmental emissions from virgin production.

However, it is apparent that both the costs and benefits of recycling programmes will be dependent on the specific products or materials addressed – EPR programmes for specific materials or products will require a more focused analysis of the environmental and economic costs and benefits.

Both case studies highlight the importance of the framing of the question to be addressed. A narrow focus on recycling activities without due consideration of the wider issues could lead to missed opportunity costs.

Packaging waste

A CBA study of packaging waste management in Europe seems to confirm the general inference of the waste management hierarchy that material recycling is preferable to disposal by landfill or incineration. This lends support to EPR programmes that encourage recycling of packaging waste. However, the case study also consolidates the view that the hierarchy should be applied with caution. In particular, the study highlights that the potential costs and benefits of an EPR programme are dependent on:

- The materials targeted;
- The waste streams targeted; and
- Local circumstances, including the population density and alternative MSW disposal options.

A CBA study of packaging waste management in Australia seems to confirm that recycling of household packaging waste shows a net benefit from a combined environmental and economic viewpoint. The conclusion for Australia that recycling in urban areas delivers greater environmental benefit than recycling in rural areas is seems intuitive, but contrasts to the findings of the study for Europe. Closer analysis of the two studies demonstrates that the underlying assumptions made about the systems and the data sources used are fundamental in determining the results achieved and may influence the conclusions drawn. This demonstrates the necessity to carefully determine the goal and scope of the study.

Batteries

In a study of batteries in the UK, no attempt was made to monetise the environmental impact. As a result, the complexity of the results presented makes it difficult to draw precise conclusions. The complexity of the results could have been reduced by using economic valuation to achieve a single environmental impact cost, but this may have reduced the transparency of the results and suggested a simplicity that does not exist.

The approach applied highlights the application of cost effectiveness analysis as opposed to cost benefit analysis. In cost effectiveness analysis, the most economically efficient way of achieving a specified target is identified. Applying cost-effectiveness analysis to determine how to achieve a pre-set target makes the assumption that the target is rational and must be achieved. This contrasts with cost benefit analysis studies, which asks more fundamental questions of whether targets are justified and at what level targets should be set. This suggests that the goals and aspirations of EPR programmes should be set on the basis of CBA studies. However, where regional EPR programmes are implemented, cost effectiveness analysis approaches may be used to identify the best way to achieve these objectives at a national or local level.

Discussion and conclusions

A number of conclusions on the use of CBA can be drawn.

Conclusions on the application of EPR programmes

From a traditional economic viewpoint, conditions for achieving an optimal waste management mix do not exist without market intervention. The scale of the disparity is greatest for products with a low end-of-life value and/or disproportionately high collection and reprocessing costs.

If external environmental effects are also considered, the disparity between the achieved recycling rate and optimal recycling rate may become even more pronounced.

This would suggest that EPR programmes, which encourage a reduction in disposal and an increase in recycling, are potentially justifiable. In order to determine whether an EPR programme is appropriate for a specific product, product group or waste stream, it is necessary to evaluate the economic costs of achieving the programme against the environmental benefits delivered by the programme. This approach can also be applied to determine what the goals and aspirations of an EPR programme should be

In particular, EPR programmes appear to have a significant role to play in situations where the end of life value of products is low but the scale of environmental impact of disposal is medium to high.

Conclusions on the evaluation of EPR programmes

The matrices presented in this paper provide ground rules for making initial assessments as to which products, product groups or waste streams *might* be addressed by EPR. However, considering the scale of the potential costs involved in implementing, enforcing and realising EPR programmes it is prudent that a more thorough analysis of the environmental and economic implications is performed.

Subsequently, this paper advocates a life cycle cost benefit analysis (CBA) approach for evaluating EPR programmes. However, CBA is a developing technique, with methodological limitations. These limitations should be recognised and appreciated:

Ethical issues. Many critics of CBA question the underlying ethics of monetary valuation of environmental impacts. However, without CBA, some other form of value judgements must be made, and these may have no rational basis.

Methodological limitations of LCA. The environmental analysis that is performed prior to the economic valuation of the environmental impacts is based on life cycle assessment methodologies. Although life cycle assessment methodologies have been subject to international standardisation, there are methodological limitations:

- The nature of choices and assumptions made in LCA (e.g. system boundary setting, selection of data sources and impact categories) may be subjective;
- Models used for inventory analysis or to assess environmental impacts are limited by their assumptions, and may not be available for all potential impacts;
- Results for LCA studies focused on global or regional issues may not be appropriate for specific local applications – specific local conditions may not be adequately represented by general regional or global conditions;

– The accuracy of LCA studies may be limited by accessibility or availability of relevant data, or by data quality and data gaps; and

– A lack of spatial and temporal considerations in inventory data that are subsequently used for impact assessment may introduce uncertainty to the results.

Achieving economic valuation of environmental impacts. Not every externality can be valued in monetary terms at present. Several external effects are difficult to measure or the impacts are too site specific to be transferred from specific studies to a general methodology for a cost benefit analysis. These impact categories must still be considered as part of the evaluation, even if only a qualitative judgement is made.

***Methodological difficulties arising from attempts to perform monetisation*.** Even where monetisation can be performed, the reliability of the values derived may be questioned. A variety of techniques can be applied to derive economic valuations. In many cases, the application of different techniques results in conflicting valuations being achieved, suggesting inherent bias in valuation methodologies.

***Difficulties of identifying and isolating internal costs*.** Estimates for internal costs of waste management related activities show wide ranges. Complex financial flows make evaluation of internal costs a difficult task. In some cases, obvious surrogate data is available (for example producer responsibility organisation fees), but sometimes this data may be misleading. Increasingly, national environmental policies have attempted to internalise some aspects of external costs. For example, emission permits and landfill taxes effectively internalise some elements of pollution. However, the charges for these permits and taxes have rarely been based on a detailed evaluation of the external costs of the avoided environmental impact. In many cases, the scale of the charges is politically motivated. It is therefore impossible to accurately determine the level of internalisation. This may lead to some double counting in the methodology.

Quantifying indirect and secondary effects. The difficulties of quantifying indirect costs and secondary effects means that many studies focus only on the direct costs. This limited approach could have a significant influence on the true “social cost”. In some cases, wider effects can only be taken into account by inclusion of broad assumptions, which may be limiting

Consideration of opportunity costs. Typically CBA is used to evaluate alternative approaches for implementing a selected policy measure, rather than looking at alternative policy measures. Therefore, the opportunity costs of pursuing a particular policy measure over an alternative policy measure may be overlooked. It is possible that greater gains could be achieved with the same investment in a fundamentally different policy.

These uncertainties and difficulties do not render CBA valueless. CBA provides decision-makers and stakeholders an insight into the trade-offs within and between economic costs and environmental benefits that are inevitably made when selecting and implementing policies. In this way, CBA becomes a useful tool for informing the development of EPR policies and programmes.

It is recommended that CBA type analyses be used to evaluate the environmental and economic performance of EPR programmes. *Ex ante* analysis should be performed to inform decisions on the development of new EPR programmes, but as it is difficult to predict how individuals/organisations or markets will react (the potential implications of EPR programmes are far reaching) *ex post* studies should be performed as EPR programmes progress. The implication is that EPR programmes should be fluid and iterative. In particular, the strict application of regional policies may be misleading. The wide range of internal and external costs and benefits experienced at a national or local level suggest that CBA studies may inform the development of regional objectives, but that some discretion on how to achieve these targets must be left to national or local decision-makers. At best, ranges of targets can be established at a regional level based on CBA, then a cost efficiency approach may be more appropriate for evaluating national implementation.

Recommendations for improving future work

There are a number of research areas that need to be further addressed to improve the quality of future CBA studies:

Methodological framework. At present, there are many practitioners performing CBA studies. In order to improve the accessibility of CBA studies to policy makers and decision-makers, a common methodological framework for conducting CBA studies may be helpful. Due to the diversity of applications for CBA studies and considering the continuing developments in the field, this should not be a prescriptive methodology, but a set of ground rules for structuring studies. The framework proposed in Section 3 of this paper may provide a basis for further development.

As part of this consideration, closer attention to the presentation of results is required. The use of absolute numbers (often to several decimal places) is highly misleading, and suggests a degree of confidence that simply does not exist. Greater emphasis on the presentation of data ranges or use of alternative economic valuations is required, in order to ensure that the sensitivity and uncertainty of studies is fully appreciated.

Further economic valuation research. Further work is needed in the area of available economic valuations. Further verification of existing values is required, and development of new values for impact categories not currently valued is required. This will involve efforts to determine dose response functions, *etc*. Continued research into the issues presented by benefit transfer is required.

Further research into internal costs of waste management activities. Available data demonstrates that internal costs of waste management activities, especially collecting and sorting activities, are highly variable. Further research into the factors that determine internal waste management costs is required in order to inform *ex ante* studies in the future.

Consideration of opportunity costs. Careful attention needs to be given to the framing of the questions asked by CBA studies. A narrow focus on recycling activities may lead to opportunity costs being overlooked.

1. Introduction

In order to reduce waste production and achieve improved waste management, the financial and/or physical responsibility for the waste management of end-of-life products must be shifted away from the traditional cost bearers (the municipalities) on to the producers of the products. Only this way can there be a direct incentive for waste prevention and increased resource recovery.

"Once companies realise they are going to have to pay for waste management and recycling, they have an incentive to make less wasteful products and to design for recyclability…". [1]

This concept is known as extended producer responsibility (EPR). The OECD defines EPR as an environmental policy approach in which the producer's responsibility is extended to the post-consumer stage of the life cycle. There are two related features of EPR policy:

- The shifting of responsibility upstream from municipalities towards producers; and
- The provision of incentives to producers to incorporate environmental considerations into the design of their products.

Increasingly, OECD governments are implementing EPR programmes for a broad range of products, product groups and waste streams. For example, the European Commission considers producer responsibility as a key principle of its waste management strategy.

Instruments implementing EPR for packaging continue to maintain a high profile. For example, the Beverage Container Stewardship Programme 1997 in British Columbia is the culmination of developments dating back to the 1960's. Several "bottle bills" exist in the US. In Japan laws have been enacted to encourage the utilisation of recycled plastic containers. National regulations in Europe implementing the requirements of the EU Directive on packaging and packaging waste (94/62/EC) have been the focus of considerable debate.

Legal instruments also exist for a diverse range of non-packaging products. British Columbia, Canada has an EPR programme for products that contribute to household hazardous waste, such as lubricating oil and paint. Comprehensive EPR legislation for waste electronics has been implemented in Japan and in several European countries. The US is negotiating a voluntary sector wide agreement on an electronics take back programme. Further voluntary programmes have been implemented by individual companies and by industrial sectors. Sector schemes include end-of-life cars in Sweden and Ni Cd batteries in the US.

The implementation of these programmes has demonstrated that EPR can be an effective instrument for meeting specified waste management targets (for example, achievement of packaging waste targets in Europe and recovery of products that contribute to household hazardous waste in British Columbia).[2,3] However, despite these successes the application of EPR programmes remains

1 Gertrude Lubbe-Wolff, University of Bielefild, Germany, as quoted in "Is Extended Producer Responsibility Effective?", Carola Hanisch, April 1, 2000 (available from http://pubs.acs.org/hotartch/est/oo/apr/hanis.html)

2 European Policy for Packaging Waste Management, Michael Sturges and Carolynn Royce available at www.bt.com.

controversial. Affected parties frequently challenge policy makers to prove that the economic costs of achieving EPR goals are justified in terms of the environmental benefits secured.

Therefore it is the aim of this paper is to investigate how the environmental effectiveness and economic efficiency of EPR programmes can be evaluated. This is investigated considering two aspects:

- The economic rationale for EPR, demonstrating the situations under which EPR has greatest potential to deliver benefit; and

- The state of the art of techniques available to evaluate the environmental and economic costs and benefits of specific EPR programmes.

It is hoped that the findings of this paper will provide a stimulus for debate and serve to inform the development of the next phase of the OECD's EPR research programme, more specifically possible activities to develop an analytical framework for addressing EPR.

2. Why do we need EPR and where can EPR be most effective?

2.1 Economics of waste management

The Annex to this paper investigates the economics of waste management using a series of simplified economic models, and attempts to demonstrate why EPR is necessary and for which types of products and waste streams EPR can be most effective. The key findings of this investigation are:

- For a specific material, a theoretical optimum recycling recycling rate may exist, but in practice this may not be achieved; and
- Four basic scenarios for ownership of recycling and landfill operations exist, and these influence the recycling rate that can be achieved. These are shown in Figure 1.

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[3] Product Stewardship in British Columbia, Kelly Lease, October 2000, presented as Facts to Act On Release #39, Institute for Local Self-Reliance, Washington DC.

Figure 1: Influence of ownership of recycling and landfill operations on recycling rate achieved

RECYCLING \ LANDFILL	Private Ownership	Owned by municipality
Private Ownership	If separate companies operate landfill and incineration, recycling will only be pursued by the recycling company to the point where the marginal net revenues are zero. This may not be the optimum solution from an economic viewpoint. If one company operates both landfill and incineration facilities, an optimum solution may be achieved if the company operations efficiently.	Recycling will only be pursued by the private recycling operations to the point where the marginal net revenues are zero. This may not be the optimal solution from an economic viewpoint.
Owned by municipality	In this situation regulations could be introduced to ensure the public recycler strives to achieve the optimum recycling rate, even though it may provide negative revenues – a 'planning solution'.	In this scenario, a full planning solution can be implemented to achieve the optimum balance of disposal and recycling.

- For products with a reduced end of life value (*i.e.* where the recycled material has a low value and/or the reprocessing costs are disproportionately high), both the achieved and optimal recycling rates will be reduced. In some case, the optimal recycling rate may be zero.
- Traditional economics only considers the financial aspects of recycling. If environmental and social aspects (the externalities) are considered, then the optimal recycling rate will change. In instances where recycling has a net environmental benefit (i.e. the environmental benefits of offset virgin production outweigh the environmental cost of collection, sorting and reprocessing) the optimal recycling rate is increased.

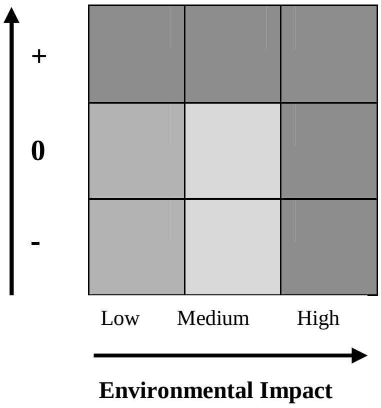
2.2 When should EPR be applied?

The models presented in the Annex demonstrate that both economic and environmental factors may be influential in determining whether an EPR programme should be applied to a selected product, product range or waste stream. This conclusion is reiterated by the decision-making matrices or guides developed at the 1999 Paris EPR Workshop as a screening tool to help policy makers select a course of action. [4] These matrices provide a way of visualising and evaluating the economic and environmental model for different situations.

The key criteria for consideration are set out in Matrix A. Two axis are identified – value and environmental impact. The value axis is not clearly defined, but value is considered to be a function of the internal costs of recovery, recycling and disposal compared to those of disposal. The environmental impact axis is a relative measure of the scale of the effects of the waste stream on the environment.

[4] Source: Michael Bennett, Engineering, TM Australia, Pty Ltd, presented at OECD EPR and Waste Minimisation Workshop, "Towards Sustainability", 4-7 May 1999, Paris.

Matrix A – Application of EPR Key Criteria

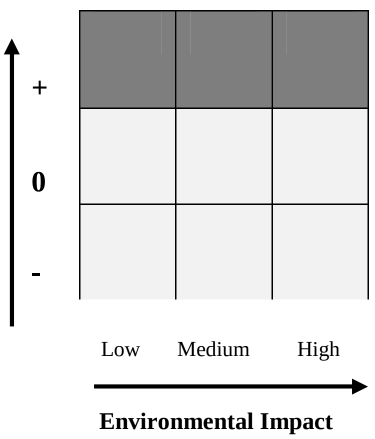


Value:
- Cost of recovery /recycling/disposal compared to revenue from recovered components
- Will vary depending on market and country dynamics

Impact:
- Specific hazards and impacts need to be assessed
- Magnitude of hazards and impacts will vary due to country and socio-geographic dynamics

Matrix B illustrates a situation where the material has a positive value at end of life – the costs of collecting, sorting and reprocessing the material are lower than the market value of the recyclate. Under these conditions, market forces will generate recycling programmes regardless of the scale of environmental impact. The greater the value the more likely that development will occur. The models in the Annex suggest that the level of recycling that will be achieved with no market intervention is not the optimum recycling rate. Where the environmental impact of the waste stream is low, this is unlikely to be of significant concern, but where the environmental impact of the material is medium to high, there may be a case for some market intervention in order to increase the recycling levels achieved.

Matrix B – Market Driven Programme



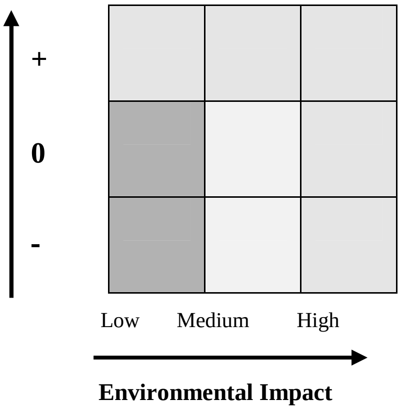
- Market forces will generate programmes where positive value exists
- The greater the value the higher the likelihood of development
- Intervention by governments carefully considered to prevent monopolistic practises and other market distortions
- Intervention may be required where the size of the market prohibits profitable operations for high impact materials

Positive value – Post Consumer Phase

Now consider Matrices C, D and E. These illustrate situations where the value of recovered material is negative – the costs of collecting, sorting and reprocessing the material are greater than the value of the recyclate produced.

Matrix C – Voluntary Programmes

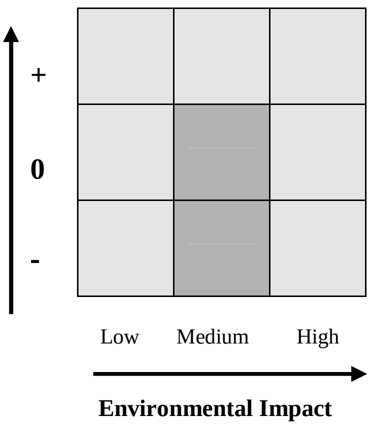
Negative value – Low Impact



- Intervention by governments not required
- Programmes will be established due to corporate responsibility policies and for marketing purposes
- Potential for product differentiation and competitive advantage
- Success of programmes will be dependent on consumer commitment

Matrix D – Negotiated Programmes

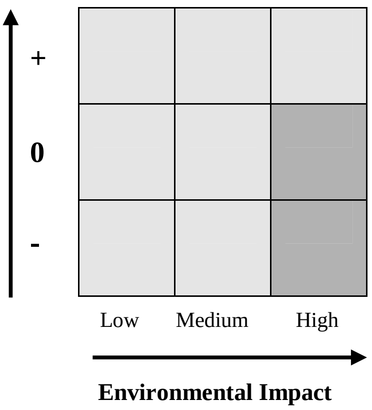
Negative Value – Medium Impact



- Hazard/impact warrants focus
- Industry may not be capable of effectively establishing and maintaining a voluntary programme
- Intervention through outcome oriented legislative support may be required
- Broad supportive framework to ensure participation by all actors
- Enforcement is critical
- National consistency is critical

Matrix E – Mandatory Programmes

Negative Value – High Impact



- Hazard/impact such that safe disposal/recycling is essential
- Emphasis on outcome oriented legislation but likely to be more prescriptive
- Environmental/disposal costs may be displayed in pricing to assist with consumer awareness and decision making
- Enforcement critical
- National consistency critical

Source: OECD, 2001

Under these circumstances, recycling will only occur with market intervention. Whether market intervention should be pursued or not may depend upon the relative scale of the environmental impact.

In Matrix C, end of life value is low and environmental impact is low – market intervention to pursue increased recycling is unlikely to be justified. This situation may occur in instances where the environmental costs of collecting and reprocessing a material outweigh the environmental benefits of virgin offset production and avoided disposal. Examples of this situation may include the recovery of mixed plastics packaging from the municipal solid waste stream or the recycling of consumer batteries.[5,6]

In Matrix D, end of life value is low and environmental impact is medium. This situation may warrant a focus on increased recycling. As no market revenue will be available to support recycling activities, it is unlikely that industry would be capable of effectively establishing and maintaining a voluntary programme. A good example of this situation in practice may be PET bottle recycling. The sheer quantity of PET bottles placed on the market may warrant action, but costs of collection, sorting and reprocessing are greater than the revenues that can be received for materials, even though demand for RPET is significant.[7]

In Matrix E, end of life value is low but the scale of environmental impact is high. The nature of the hazard requires that safe disposal or recycling is essential. An extreme example of this situation is a decommissioned nuclear power station. The potential environmental impacts are high, the costs of decommissioning are high. It is expected that operators have built these into the operational costs of the installation. In reality, these costs are often subsidised by governments.

In such instances where environmental impact is high a prescriptive approach may be required to ensure that high levels of environmental protection are achieved.

The matrices attempt to simplify the description of the environmental and economic situations into a series of discrete scenarios. In reality, the boundaries between each scenario become blurred. The availability of secondary material markets will be critical to the success of any policy objective. Changes in the supply side and the demand side may influence the economics of recycling, thereby moving the description of the situation from one matrix to another. This may be particularly important in capital intensive sectors, such as the paper and board industry. Imbalances in the supply and demand of materials driven by capital investments can lead to fluctuating virgin and recovered fibre prices. The instability generated in these situations can hinder the development of an optimal waste management strategy.

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5 Evaluation of the costs and benefits of reuse and recycling targets for materials in the frame of the packaging and packaging waste directive (94/62/EC), RDC/Pira International, draft report, May 2001, available from EC DG Environment.

6 Analysis of the Environmental Impact and Financial Costs of a Possible New European Directive on Batteries, ERM for the UK Department of Trade and Industry, November 2000.

7 Evaluation of the costs and benefits of reuse and recycling targets for materials in the frame of the packaging and packaging waste directive (94/62/EC), RDC/Pira International, draft report, May 2001, available from EC DG Environment.

2.3 Implications of the models and matrices for evaluating EPR policies

In order to focus on which products or waste streams are likely to be best addressed through EPR and which type of instruments may be appropriate, it is necessary to think carefully about the factors that will influence the economics and environmental impacts of that product or waste stream.

The type of product, its durability, composition, availability of end use markets and market distribution will all play a role in defining the situation. Other factors to consider are the number of products, the degree of homogeneity within a product category and the size and scope of the product distribution network.

In essence, the matrices and the models in the Annex suggest that in order for an EPR policy to be justified the environmental and economic benefits of the policy must outweigh the environmental and economic costs. To evaluate if this is the case, we need to be able to quantify economic costs and benefits and environmental costs and benefits on an equivalent basis. This is best achieved through applying a life cycle cost benefit analysis approach, as described in Section 3.

3. Life cycle cost benefit analysis as a tool for evaluating EPR programmes

The matrices and models presented provide ground rules for making initial assessments as to which products, product groups or waste streams *might* be addressed by EPR. However, considering the potential costs involved in implementing, enforcing and realising EPR programmes it is prudent that a more thorough analysis of the environmental and economic implications is performed.

The presentation of the economic theory leads to the conclusion that we should advocate a life cycle cost benefit analysis (CBA) approach for evaluating environmental policies. Indeed, some environmental economists believe that this is the only rational approach to decision-making.[8] To achieve this, we need to consider the environmental and economic aspects on an equivalent basis. Converting environmental impacts into a monetary unit provides a foundation for achieving this. It is an approach that is most compatible with the thinking of economists, policy makers and politicians.

This section of the paper proposes a methodological framework for CBA and discusses the practicalities of performing CBA studies and using CBA results to evaluate EPR policies and programmes.

3.1 The dilemma for policy makers

Environmental policies such as EPR are pursued in order to provide environmental protection and environmental improvement. These are the benefits of environmental policies. Such benefits may be measured in terms of environmental protection provided or environmental improvements made. For example, policies to reduce emissions of global warming gases may be quantified in terms of avoided emissions of kgs of CO_2 equivalents. These benefits are known as externalities, as they are external to the traditional economic model.

However, in addition to these environmental benefits, an environmental policy decision will also incur implementation costs. Different policy options will incur different cost implications. These are the internal costs of implementation. As a society, we should pursue policies that provide good

8 See "Integrating cost-benefit analysis into the policy process", Professor David Pearce and EFTEC, included as Annex II of Valuing the benefits of environmental policy: The Netherlands, RIVM, March 2001.

value for money by achieving environmental objectives / benefits without incurring disproportionate internal costs.

Looking at and comparing the implementation costs of different policy options in isolation would be misleading, as the scale and type of benefits secured may vary widely across the different options. It is necessary to consider the costs of implementation within the context of the benefits that they provide. Thus, a quantitative comparison of the costs and benefits of environmental policies can only be achieved if the internalities and externalities are measured in a common unit.

3.2 Using CBA to aid the decision-making process

CBA is a developing technique that seeks to value externalities in monetary terms, and compare these against the internal costs of implementation. In this way, decision-makers and stakeholders gain insight into the trade-offs within and between economic costs and environmental benefits that are inevitably made when selecting and implementing policies.

It is important to recognise CBA as a useful tool rather than a decision rule. By asking the right questions it opens up the discussion and identifies key issues. Methodological and data limitations may prevent CBA from providing definitive answers to all the questions raised, but this does not render the exercise valueless. For example, it can be particularly useful to know the point at which costs of reducing a pollutant start to rise sharply. This information can then be used to make a first guess as to the level to which it is "reasonable" to reduce pollution. This approach is particularly appropriate where there is very limited information on the real risks that the pollutant poses.

In this way, CBA makes decision-making more transparent. It is an aid to the decision-making process, not a substitute for it. The decision-maker must still judge how to weigh up environmental effects, economic costs and distributional impacts of the different policy options and select the preferred option. Supplementary information from macro-economic analyses that consider the impacts of policy on economic indicators such as GDP, inflation rate, and the trade balance may also be of value to decision-makers.

3.3 Ex ante and ex post analysis

Given the potential costs involved in implementing, enforcing and realising EPR programmes some form of ex ante analysis is essential to determine the potential environmental and economic implications. Looking at futures to determine how a sector or individuals within a sector will react to an EPR policy is inherently uncertain, but the process allows us to identify implications and issues that may arise, and to identify gaps in our knowledge that need to be addressed.

Further *ex post* studies will provide insight into the key responses affecting the environmental and economic effectiveness without the limitations of scenario forecasting assumptions necessary for *ex ante* analysis. In particular, *ex post* studies provide evidence to assess the effects of policies, given the actual responsiveness and motivations of firms and individuals. Organisational inefficiency or other considerations not included in the optimising analysis may mean that some firms do not respond as predicted. This will be reflected in the *ex post* analysis. The implications are that EPR policies must be responsive and iterative, developing over time according to the reactions to implementation.

3.4 A proposed framework for performing CBA

There is no standardised approach for performing CBA, but the following general principles should be incorporated into all studies:

- Transparency: a clear understanding of how the results are achieved, which assumptions are made, and which data are used should be provided;

- Comprehensiveness: all relevant environmental impacts should be acknowledged in some way;

- Time and spatial issues should be addressed; and

- Stakeholder dialogue should be maintained to ensure that the systems modelled and assumptions made are in line with reality. (Dialogue will also improve access to needed data.)

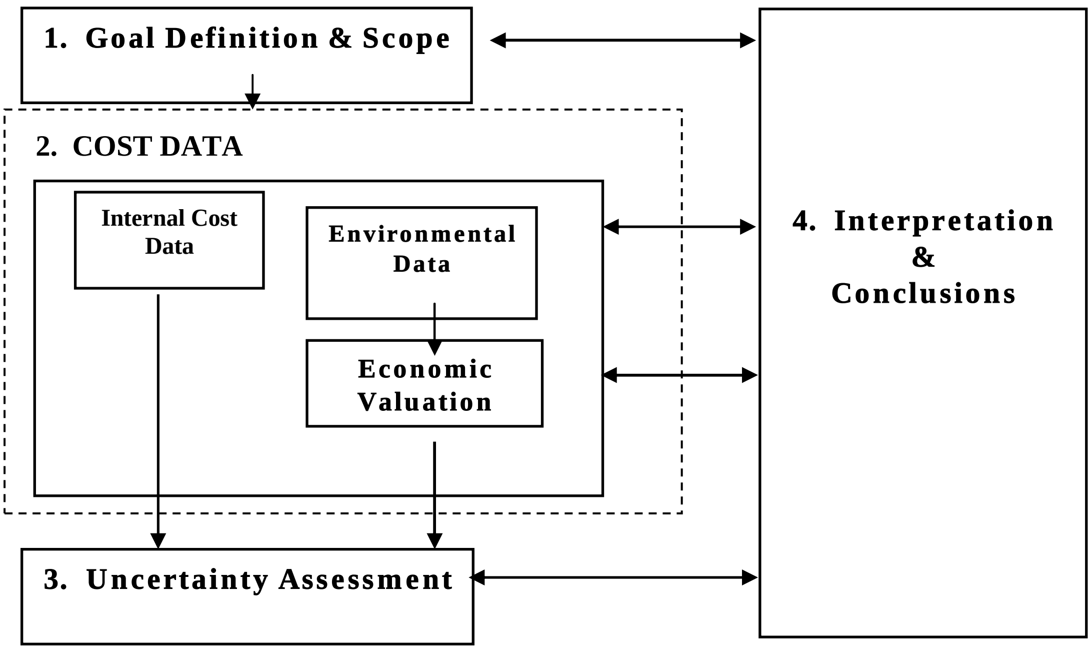
An evaluation of the methods applied by various researchers and consultants suggests that the following common themes apply:

- Environmental impacts are quantified using life cycle inventory analysis or life cycle impact assessment techniques; and

- Economic valuation techniques are applied to convert life cycle inventory or impact data into monetary units which are then comparable with internal costs.

Figure 2 proposes a basic methodological framework for applying CBA as an analytical tool for evaluating the environmental effectiveness and economic efficiency of EPR programmes and policies. It is not the aim of this paper to provide a detailed description of every step necessary to perform a CBA study, but some key methodological aspects are outlined below. The critical aspects of determining internal costs and achieving external economic valuations are discussed later in Section 4 and Section 5 respectively.

Figure 2: Cost benefit analysis methodology [9]



[9] Adapted from "The multiple pathway method: A guide to the application of the methodology developed through the research project: A combined methodology to evaluate recycling based on life cycle assessment (LCA) and economic valuation analysis (EVA)".

3.4.1 *Goal and scope definition*

The first stage is to determine the goal and scope of the analysis. Two important questions must be addressed.

How extensive will the analysis be?

The extensiveness of the analysis is dependent on:

- The likely significance of the costs (both internal and external) to be incurred by the policy;
- The importance of the issues being investigated;
- How sensitive the results may be to the assumptions made; and
- The resources available to complete the analysis.

In some instances it may be a sufficient and useful achievement to provide a rough order of magnitude for the costs. Data shortages may limit what can be achieved, but the process of asking the right questions is useful in itself. CBA highlights the issues involved, where more information is needed and where the uncertainties lie. An analysis with considerable uncertainties is not necessarily an ineffective study.

What scenarios will be considered?

A "business as usual" scenario provides the point of departure for assessing the costs of the changes induced by the policy and for determining the environmental effects.

For *ex ante* analysis, the baseline scenario should incorporate possible future changes not related to the policy measure being investigated (*i.e.* those changes that will occur regardless of the policy). These may include industry trends, technological developments or the effects of any other policy measures. Defining the baseline reveals the extent of our knowledge of the current situation. The presumed response to the implementation of the policy is compared against the baseline to determine the incremental changes in costs and benefits. Where a number of possible responses could occur or where the responses are difficult to predict a series of alternative responses should be defined.

For *ex post* analysis, the business as usual scenario is presumed to be the situation prior to the implementation of the EPR policy. The only response scenario that needs to be considered is the situation now, but only the changes attributable to the introduction of the policy should be considered. Changes that would have occurred due to industry trends, technological developments or the effects of any other policy measures should not be included in the evaluation.

3.4.2 *Generating cost data*

For a cost benefit analysis of public policy, the emphasis of the data collection must focus on the total costs to society rather than the costs to individual groups. The following steps provide an outline for identifying and collecting the internal and external cost data necessary to complete the cost benefit analysis:

1. Identify and list all parties affected.

2. Identify the internal cost impacts of the policy on each group (expressed in monetary terms).

3. Determine the environmental costs and benefits of the policy using life cycle techniques, and convert the environmental costs and benefits into monetary values.

4. Where necessary, convert the internal and external costs and benefits into comparable financial terms, taking into account the time scales of investment.

Internal and external cost impacts may occur over different time frames. The costs must be considered in comparable financial terms. Possible techniques that can be applied include discounting or annualisation. Time considerations are important because a policy measure can generate one-off capital costs, operating costs, or a combination of the two. It is therefore necessary to understand the time profile of the costs rather than just determine a single value, but ultimately the time profile data must be aggregated to provide an assessment of the relative merits of the alternative options, and to compare these against the *internalities*.

The simplest approach would be to directly aggregate the internal costs in the time profile. However, resources available now are worth more than a similar amount of resources available at some time in the future, as resources available now can be put to work earlier. A good analogy is investing money. The earlier one has money available the sooner one can invest it and begin to earn interest or profit. The same applies to all other economic resources. Therefore, costs that arise at different points in time have to be weighted before they can be aggregated.

The weighting procedure that is conventionally used is called 'discounting'. The weighting factor is called the “discount rate”. It is a rate of interest that describes how much a unit of resources in the future is worth in terms of today's money. By determining the present value of future investments, the stream of costs over time can be converted into a single figure for the total cost of a policy in terms of today’s money. This allows us to compare different policy options with different time-investment profiles. It is suggested that a range of discount rates is applied to determine the sensitivity of the results to the discount rate applied. Typical discount rates used in CBA are 2%, 4% and 6%.

An alternative to using present values derived from discounting is to use annualised costs. Using present values is an adequate approach for comparing the costs of policy options provided that the period of time is equivalent. However, some options for meeting objectives may require capital investments which have different operating lifetimes. In this instance, annualised costs should be used.

3.4.3 Sensitivity and uncertainty analysis

Uncertainties and sensitivities are the general case in CBA. It is rare that information and data is adequate for the full analysis. Surrogate data and assumptions will be inevitable. Care must be taken when defining assumptions and filling data gaps.

The process of sensitivity and uncertainty analysis:

- Gauges the robustness of the results achieved and conclusions drawn;
- Determines where knowledge gaps exist; and
- Identifies those factors that determine the outcome of the analysis.

There are some general rules and activities that can be performed as part of the sensitivity analysis, but sensitivity analysis is an iterative process, dependent on the nature of the assumptions made and results generated.

Analysis of the internal costs and benefits:

The “key drivers” should be identified. These are the assumptions and gaps which are most important in determining the scenarios chosen and the costs that are calculated. Key drivers may be assumptions used to fill data gaps or economic assumptions (such as the discount rates applied, the exchange rates for currencies, and the effects on behaviour of market price).

The influence of the key drivers is evaluated using available alternative data values or estimated ranges or alternative scenarios. If a range of estimates is plausible for a given variable, the final results achieved should be presented as a range.

Ideally the analysis provides a quantitative evaluation of the sensitivities, but a lack of data may prevent quantitative analysis. In such cases a qualitative assessment should be made. This introduces subjectivity but it at least recognises the potential uncertainties and sensitivities that exist.

Sensitivity analysis of the external costs and benefits:

Sources of uncertainties should be identified. These may include:

- Assumptions made in modelling the system life cycle (including the chosen system boundaries, the unit processes chosen for study and knowledge of the flows between unit processes);
- Surrogate data, poor data or data gaps in the life cycle inventories for individual unit processes;
- The environmental impact assessment methods use; and
- The economic valuations applied.

Theoretically, complex statistical analyses can be applied, such as Monte Carlo simulation. In reality there is rarely enough data available to complete a detailed statistical analysis of the uncertainties. Sensitivity analysis of the externalities becomes an iterative process, using available or estimated surrogate data or alternative assumptions and scenarios to gain an understanding of the potential impact of identified uncertainties. Where a quantitative assessment of uncertainties is not possible due to unfillable data gaps, or inadequate system information the limitations imposed on the study should be made explicit.

3.4.4 Interpretation

The results must be presented in comparable and clear units. For example, if alternative waste management options are being investigated, the results should be presented per tonne of waste material treated. The presentation of results must be consistent. Summary tables and graphs improve the understanding of the results and conclusions drawn. All conclusions drawn must be validated through reference to specific numerical results. If any subjective or qualitative statements are made this should be stated explicitly.

When presenting the results of a study, some general rules should be applied:

- All data sources must be transparent and referenced within the bounds of confidentiality. If possible, key data should be reported in an annex to the main study;

- All assumptions underpinning the baseline scenario and the predicted response to the policy measure should be documented and justified;
- Any key economic assumptions should be clearly stated and justified;
- All economic calculations should be presented explicitly; and
- All uncertainties and sensitivities should be made explicit, and where possible the results of any sensitivity analysis should be presented.

The objective is to make the data sources and assumptions as clear as possible. In effect, an "audit trail" should be established that allows the reader to repeat the analysis.

Any issues not addressed by the study should be identified and discussed, for example, consideration of social motivation of the population to participate in recycling schemes. All limitations of the CBA approach should be discussed.

4. Evaluating the internal costs of EPR programmes

4.1 Identifying financial flows

A cost benefit analysis of public policy must focus on the total financial costs to society rather than the costs to individual groups. In the US, this approach is known as Full Cost Accounting.[10] Affected parties may include:

- Specific industry sectors;
- Specific businesses;
- Government departments and enforcement agencies;
- Employees (in specific industries and businesses);
- General public;
- Special interest groups; and
- Producer responsibility organisations.

The distribution of the costs that are incurred between these parties is important. EPR policy measures are not attractive (and are unlikely to succeed) if they imply a disproportionate burden on a particular sector. By identifying the effects on each of the stakeholders, an appreciation of the potential distribution of costs is afforded. Distribution effects may occur at a group level, at a national level, or at a regional level.

Determining the internal financial cost impacts of the policy on each group is a difficult task. The financial flows between affected parties may be complex. Figure 3 illustrates the complexity of financial flows that may be instigated by the introduction of an EPR policy.

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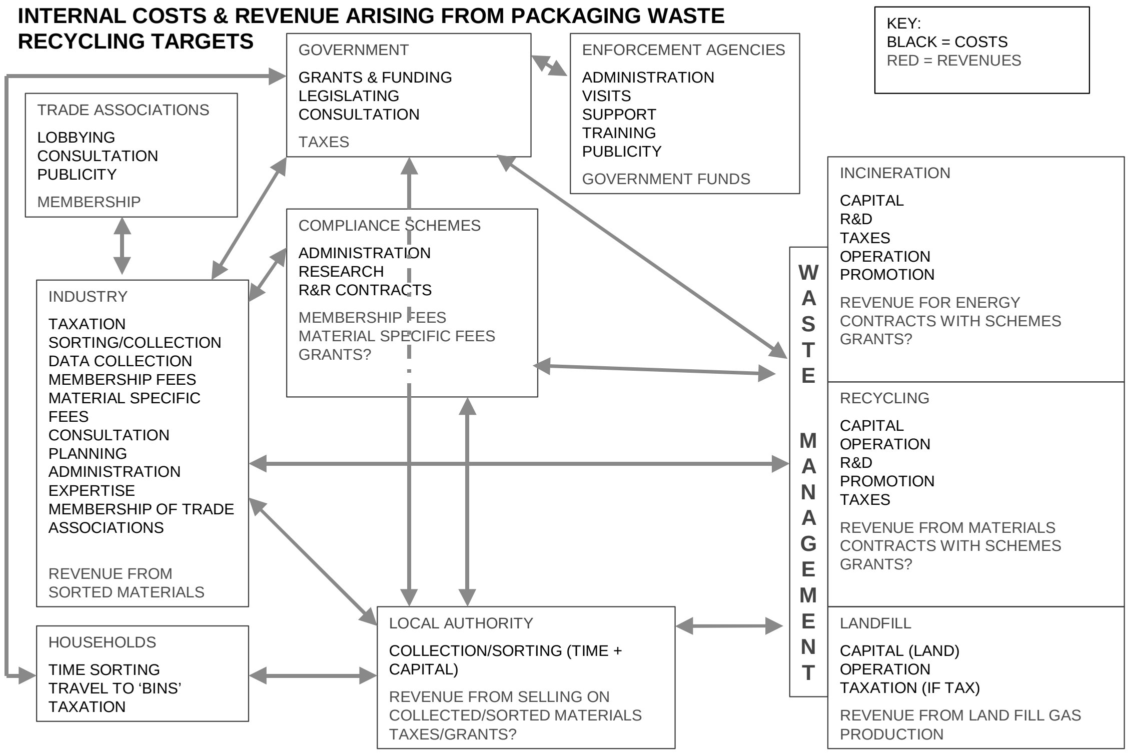
10 http://www.epa.gov/epaoswer/non-hw/muncpl/fullcost/index.htm.

Figure 3 Financial flows for packaging waste producer responsibility



NB: In some cases Industry and Recyclers are the same organisation

4.2 Categorising and quantifying financial flows

The internal financial costs of achieving an EPR programme can be categorised as administration costs or compliance costs.

Administration costs relate to the administrative and managerial cost burden imposed on the bodies responsible for applying the instrument (government departments responsible for drafting and consulting on the legislative instruments and enforcement agencies responsible for ensuring that legal obligations are fulfilled) and the economic agents subject to the instrument.

Compliance costs relate to the tangible financial burden imposed on the economic agents subject to the instrument. These may manifest as increased waste collection and management fees, subsidy payments to recyclers, or fees for membership of a Producer Responsibility Organisation (PRO).

The characteristics and relative complexity of policy instruments influence this cost burden. For example, claims of cost efficiency made on behalf of the UK implementation of packaging waste legislation may be exaggerated. The UK legislation is based on a complex shared producer responsibility approach. Compliance costs appear low compared to other countries (although recycling achievements are also low).[11] However, the administrative costs in the UK have not been calculated, but

11 Cost Efficiency of Packaging Recovery Systems – The Case of France, Germany, The Netherlands and the United Kingdom, TN Sofres for DG Enterprise, February 2000.

it is likely that these are considerably greater than the compliance costs and negate UK claims for cost efficiency.[12]

The costs considered must be the net costs to society. It should be remembered that an EPR programme may also generate revenues. For governments and enforcement agencies, levies or taxes may offset some or all of the administration costs, and any savings through reduced municipal disposal costs should be incorporated. For economic agents responsible for achieving EPR goals and targets reduced waste collection/disposal fees and revenues received for materials must also be included.

Theoretically, additional costs imposed by EPR are ultimately passed on to the final consumer in the form of price increases or increased taxes. The increased tax burden represents to costs imposed on government departments, local authorities and regulatory authorities, whilst price increases can be taken as a measure of the cost of compliance. In reality, these aspects cannot easily be used to determine financial burden imposed by EPR. The proportion of costs that can be passed on to the final consumer depends on the elasticity of demand for the product and the state of market competition. The remainder of the costs must be achieved through cost savings or reduced dividends. In the short-term, it is likely that the majority of costs will be borne by industry, with a gradual shift to the final consumer. Similarly, implementing EPR policies may only divert tax revenue from one activity to another, or it may be difficult to identify what proportion of a tax increase is attributable to the EPR programme. Therefore, we must try to quantify these costs through other routes.

For *ex post* studies, government departments and enforcement agencies should be able to quantify the labour effort put into designing, implementing and enforcing an EPR policy. This time can be valued to gain an insight into the administrative costs incurred by these organisations. Design and implementation costs may be one-off costs, or incurred at regular intervals as programmes are reviewed (for example, the EC Directive on packaging and packaging waste (94/62/EC) commits the Commission to an extensive review of the achievements and targets of the directive every five years). In addition to these time-related costs, any financial costs of enforcement (such as court fees and expenses) must also be considered. The revenues received from licences, fees and taxes will also be readily quantifiable from organisation and department accounts. For *ex ante* studies, it is necessary to make estimates of the likely scale of these costs based on experience of implementing EPR programmes in other countries or for other product and waste streams.

It may not be easy to obtain accurate estimates of the administration costs incurred by economic agents subject to the instrument. Administrative and managerial activities are often lost as part of the overhead burden of a company. Broad assumptions may be required in order to quantify this aspect for both *ex post* and *ex ante* studies.

For *ex post* studies, compliance costs incurred by economic agents subject to the instrument should be quantifiable through company accounts information (for changes in waste collection costs or subsidies paid to recyclers) or through reference to PRO membership fees. Revenues received by PROs must not be overlooked. However, it is possible that these compliance costs will not equate to the *real* costs of increased recycling – different situations within a country, within a region, cross-subsidisation, *etc*.

For *ex ante* studies, it is necessary to evaluate the potential costs involved. To achieve this, we must collect primary data on the costs of collection, sorting and reprocessing and the costs of alternative disposal options. These costs must be the net costs, taking account of the revenues received.

12 Based on personal experience of the author from helping UK companies achieve compliance.

Indirect costs and secondary effects of a proposed policy are difficult to identify and quantify. It is likely that these aspects can only be determined through close dialogue with industry and other stakeholders. It may only be possible to identify indirect and secondary effects. Quantification may be extremely difficult or highly uncertain.

4.3 Available internal cost data

Some countries have made detailed analyses of the financial costs of recycling (e.g. the US and Australia), but consistency and comparability of these data sets is limited. There are also often difficulties in applying this data for specific cost benefit analysis studies.

A recent review of some available internal cost for waste management activities identifies a wide range of costs. [13] Although much of this review concentrates on cost differentiations experienced in the UK and Europe, it serves to demonstrate the variables, which can influence the internal costs experienced. Where possible, internal costs for situations in the US have also been presented.

Key points arising from the review are:

- Internal cost estimates show considerable variation – this may be due to inclusion of different parameters / processes / approaches to the calculations;
- There is inconsistency in the methods applied and it is therefore difficult to make comparisons between different sources;
- Often it is frequently not transparent as to whether costs are gross or net of revenues from recycling /energy recovery;
- Often it is not transparent as to whether costs include capital investment;
- Often it is not transparent who the costs apply to, for instance, industry, local authorities, government; and
- Little is available on economies of scale.

Although these available data can be used to quantify the order of magnitude of costs for waste management processes, detailed CBA studies to support EPR policy decisions may require considerable primary data collection in order to be justifiable to stakeholders.

4.3.1 Internal cost data for collection and recycling activities (for household waste streams)

An Audit Commission survey investigated 21 authorities and notes a wide variation in costs between kerbside collection schemes. [14] It suggests the principal reasons for these variations are:

- Frequency and method of collection;
- Materials and tonnages collected;

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[13] Presented in “Beyond the Bin: The Economics of Waste Management Options”, ECOTEC Research and Consulting Ltd, 1999.

[14] Waste Matters: Good Practice in Waste Management, Audit Commission, 1997.

- Population density;
- Number of households served;
- Efficiency of the collection, transport and sorting system; and
- Materials income and recycling credits received.

The relative importance of these factors cannot be assessed in detail. The influence of less tangible factors such as scheme age and the information processes used to promote participation are not discussed, though experience in other countries suggest that these are influencing factor

The Audit Commission report analysed the effects of tonnages collected from each household on average costs per household. The analysis suggests that:

- Costs per household increase as the rate of materials capture increases; and
- Costs increase at a lower rate than materials collection (i.e. the costs of waste collection per tonne fall).

As the scheme matures and participation rates increase, marginal costs of additional collection may begin to approach zero. Costs may increase again with the inclusion of new materials (e.g. introduction of plastics collection). There is no consideration as to whether the costs would then begin to increase again as more and more material if recycling is pursued.

Table 2: Economies of scale in kerbside recycling[15]

Kg recycled per household	**21-25**	**26-50**	**51-75**	**76-100**	**101-146**
Average costs					
Gross cost per household (£)	4.74 (5.02)	7.01 (7.43)	7.56 (8.01)	12.55 (13.3)	14.59 (15.47)
Net costs (excluding recycling credits) per house hold (£)	4.66	6.2	6.2	8.37	10.63
Net costs (including recycling credits per house hold (£)	4.54	5.74	5.48	8.37	8.03
Net costs per tonne waste collected (excluding recycling credits)	200 (212)	160 (170)	100 (106)	90 (95)	90 (95)

Similar results are found in the US. Stevens 1994 reports cities with relatively high recycling rates that had costs of about one third that of schemes with lower recycling rates. [16] Ecologika 1998

15 Figures presented in parenthesis are 1999 values.

16 Recycling Collection Costs By the Numbers: A National Survey, Barbara Stevens, *Resource Recycling*, Sept. 1994, pp. 53-60.

show that costs of a blue box scheme fell over 3 years from £90 to £50 per tonne through reducing collection and sorting costs and increasing income through consortia selling. [17] Learning by doing and increased participation reduces costs over time, but income (revenue streams) is still vulnerable to swings in material prices.

Although these research reports suggest a correlation between costs and materials capture, the picture is actually more complex. The wide range of influencing factors identified, and the wide ranges reported in most literature, suggests that such conclusions may be premature. The term "kerbside collection" encompasses a wide range of activities and approaches, the costs of which will vary considerably. It is therefore difficult to draw conclusions about the costs of kerbside recycling schemes without a far wider reaching and dedicated study.

Decisions concerning the collection methods, materials coverage and the collection frequencies cannot be made separately from considerations of the vehicles and containers to be used in collection. In turn, this influences the labour requirements of the scheme. These decisions may also affect the quality of the collected material, and therefore the potential revenue levels that can be achieved.

Table 3: Summary of Recycling Costs[18,19]

Collection system	Collection cost (£/t)	Separation cost (£/t)	Sales income (£/t)	Recycling cost (£/t)
Bring scheme	16-36 (18-41)	0 (0)	0 (0)	16-36 (18-41)
Blue box	60-150 (69-173)	50 (58)	25 (29)	18-175 (98-201)
Green bin	25-40 (29-46)	50 (58)	20 (23)	55-70 (63-81)
Green bag	25-45 (29-52)	50 (58)	20 (23)	55-75 (63-86)

Table 4: Summary of collection costs[20]

System	1993 ECU	1999 £
Mixed collection and bring scheme	13	12
Co-collection (blue box)	62	56
Separate collection	160	146

17 Re-Inventing Waste: Towards a London Waste Strategy, Ecologika for LPAC and Environment Agency, 1998.

18 Landfill Costs and Prices: Correcting Possible Market Distortions, Coopers and Lybrand for UK Department of Environment, 1993.

19 Separation cost has been assumed to be the same irrespective of level of materials seperation prior to collection. Cost of collection for bring schemes is given net of sales revenue (i.e. revenue =0).

20 Assessing the Waste Hierarchy – A Social Cost Benefit Analysis of Municipal Waste Management in the European Union, Inger Brisson, 1997.

Table 5: Summary of collection costs [21]

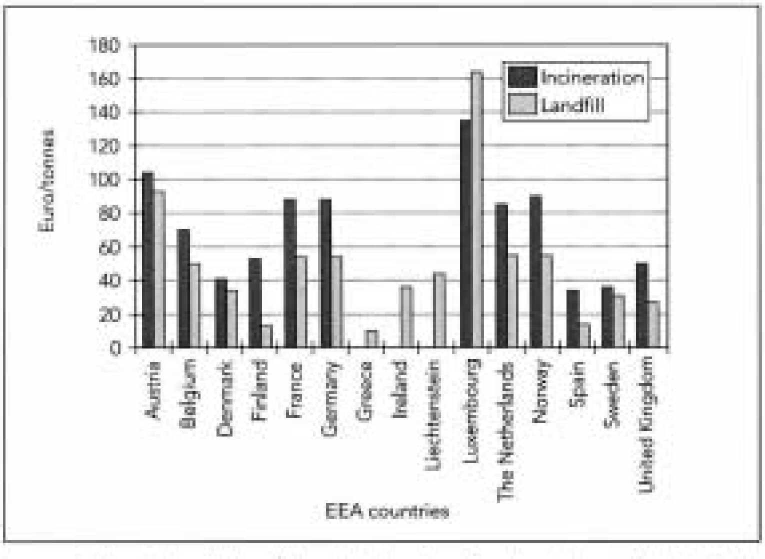
Scheme	Costs
Bring scheme	£40-49 (43-53) per tonne for low density bank systems (one site per 3500 households)
	£65-133 (71-145) per tonne for high density systems (one site per 500 households)
Separate collections [22]	£140-231 (153-252) per tonne
Integrated collection systems	£79-155 (86-169) per tonne

Reviewing this data, it is clear that it is not possible to formulate an "average" value for the collection and reprocessing of recyclables in the frame of EPR programmes for household waste streams. Too many factors influence the costs and benefits achieved. It is therefore necessary to consider carefully the internal costs of each proposed EPR scheme, and how this will impact on the marginal costs under a range of potential scenarios.

4.3.2 Disposal costs

Figure 4 illustrates the wide range of disposal costs experienced in Europe. Again, this illustrates the high variability of waste management costs in different countries and in different circumstances.

Figure 4: Average treatment prices for landfilling and incineration of non-hazardous waste [23]



Average treatment prices for landfilling and incineration of non-hazardous waste in selected EEA member countries (excl. waste tax and VAT). It should be noted that all prices are averages of observed prices and cover large variations between plants.

21 The Impact of Recycling Household Waste on Downstream Energy Recovery Systems, Atkinson, New, Papworth, Pearson, Poll and Scott, ETSU Report B/RI/00286/REP, 1996.

22 *i.e.* one collection round for recyclables and another collection round for residuals.

23 Environment in the European Union at the turn of the century, European Environment Agency.

Material specific allocation of these costs may improve the applicability of data for cost benefit studies. For landfill, allocations based on residual waste volume in the landfill, and contribution to landfill gas emissions and leachate may be appropriate. For incineration, consideration of contributions to emissions requiring pollution control and contributions to bottom ash production may be necessary. Calorific values may be used to indicate potential contributions to recovered energy.

4.3.3 Capital costs

Table 6 presents the capital costs for various waste management facilities in the UK. These data are especially useful for identifying the distribution of investment costs required over time in order to achieve an EPR programme.

Table 6: Capital costs of facilities for Household and commercial waste – Processing 200,000t per annum[24]

Facility	Costs (£M/200,00t)
Energy from waste plant	40 (43)
Anaerobic digestion plant	25 (27)
MRF and transfer station	10 (11)
Transfer station and civic amenity site	5 (5)
Landfill	4 (4)
Civic amenity site and MRF	2 (2)
Civic amenity site	1 (1)

5. Quantifying environmental benefits of EPR programmes

In a life cycle cost benefit analysis study, the environmental impacts are quantified using life cycle assessment techniques. These environmental impacts are then converted into monetary values using economic valuation, so that the environmental costs and benefits of the programme can be compared against the financial costs of implementation.

5.1 How are economic valuations for environmental impacts achieved?

Monetisation is a form of valuation (known in this case as economic valuation). Valuation seeks to convert the environmental impacts of a life cycle assessment study into a single comparable unit (in this case a monetary unit such as Euros or dollars).

Economic valuation of environmental impacts can be achieved through a variety of techniques:

- Contingent valuation is a state preference method which is used to ask individuals their willingness-to-pay (WTP) for the environmental good directly using a structured questionnaire;
- The Hedonic pricing method (HPM) considers that the price of a good is a function of its attributes, including environmental attributes. For example, the price of a house is a function of the characteristics of the house, its neighbourhood, and environmental variables such as ambient air quality or proximity to countryside. HPM regresses house prices to statistically isolate the price differential due to the relevant environmental characteristic; AND
- Averting behaviour method values environmental quality by looking at the expenditure people make for goods that can substitute or avoid for a decrease in environmental quality. For example, expenditure on bottled water can give an indication of people's WTP for preventing the adverse health effects from using bottles water.

[24] The Competitiveness of the UK Waste Management Industry, DTI, 1997.

5.2 Control costs versus damage costs

Some researchers argue that pollution prevention costs or clean up costs could be used to estimate the value of environmental characteristics. In order to understand this debate, consider emissions of greenhouse gases.

A damage cost estimate tells us what the loss of global wellbeing (utility, welfare) is from an emission of 1tC equivalents. Control costs are then compared to this damage. If it costs more to control the emission than the damage it does, then, in cost-benefit terms, one would not control that tonne. Thus, the control cost is in fact the internal cost against which we compare the external benefits.

Are there any circumstances in which control costs could be used?

"First, suppose that international negotiators are perfect economists and they estimate the globally optimal level of emissions reduction. Mathematically, this is where benefit minus costs are maximised, i.e. where marginal benefits of reduced damage equal marginal costs if control. Then, of course, it does not matter whether we use costs or benefits at the margin because they are equal. This may be important - see below. Second, one could interpret negotiated emission reductions as reflecting some kind of global consensus of what ought to be done. If so, the marginal cost of control at the point where the emission reduction goal just binds can be used. T he idea here is that the world would be saying that it judges that cost to be just worthwhile meeting.

The problem with both arguments is that they miss the whole point of a cost-benefit approach. If decision-makers always optimised we would have no need of cost benefit analysis (or any other decision-making advice!) because they would simply get things right and no amount of argument would show that they did not get it right. The arguments for using control costs therefore rest on a rather naïve model of political decision making. We do CBA ***precisely because*** *we want to illuminate politicians' decisions, to check whether they have made wise decisions, etc."* [25]

5.3 Benefit transfer

The costs of undertaking an economic valuation exercise are prohibitive. Therefore studies tend to use as a basis monetisation factors available in the literature. This involves the application of the principle of benefit transfer. Benefit transfer is the practice of using monetisation factors from previous studies which may focus on a different region or time period.

Three alternative approaches to benefit transfer can be applied:

- Direct transference of mean values – this is a very simple approach, but introduces uncertainties to the analysis. It is the most commonly used approach;
- Transference of adjusted unit values – this approach adjusts past estimates to correct for any original bias, or to take into account socio-economic characteristics of the particular project , the potential levels of damage reduction, regional and site characteristics and the availability of substitute goods; and
- Transference of a demand function – this approach takes demand functions from the previous studies and inputs new data relevant to the current project, then re-runs the analysis. This type of approach is preferable but difficult as the data needed to re-run the analysis is unlikely to be available.

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25 Taken from a personal communication with Professor David Pearce, January 2001.

5.4 Limitations to the use economic valuation

There are limits to the application of monetisation. In some cases there is currently no sound understanding of the pathway between the inventory data and effects on the protection area, or no reliable economic values are available. In such cases the environmental concern cannot be quantified within the monetary valuation but must still be considered in interpretation and in making a decision based on the study.

To overcome this, methodologies should consider two types of impact category:

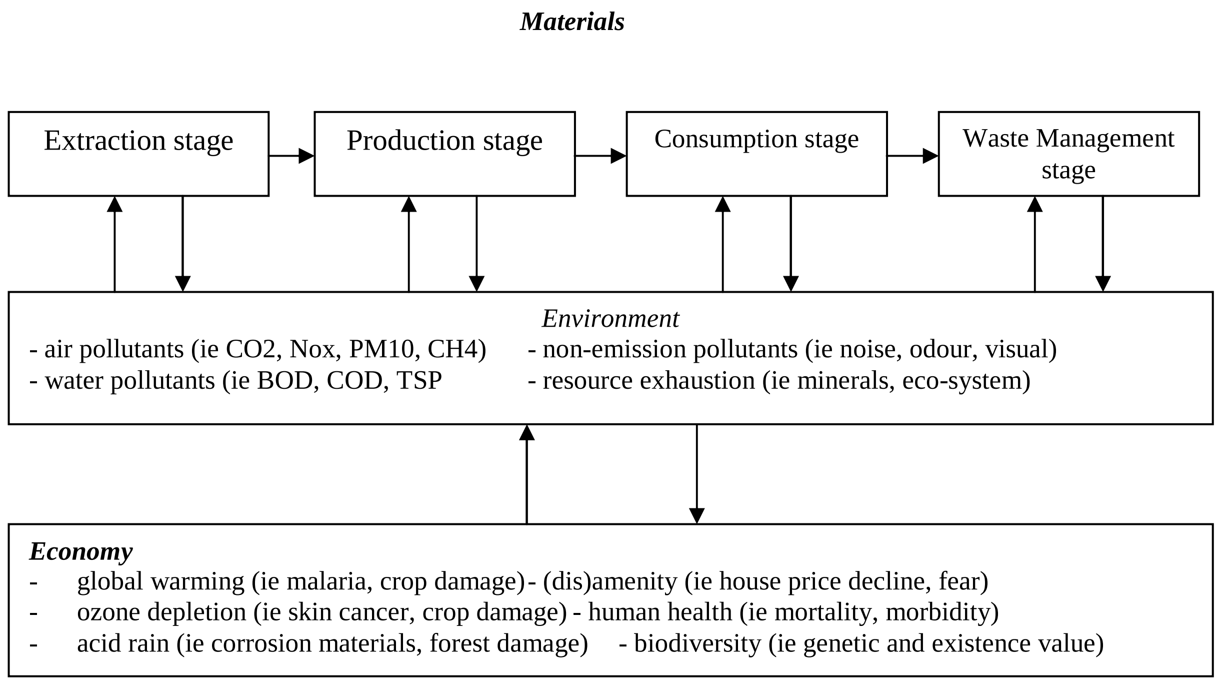
- Primary impact categories – those for which an economic valuation can be derived; and
- Secondary impact categories – those for which no economic valuation can be derived.

Where possible, these secondary impact categories should be quantified in traditional LCA impact assessment units. If this is not possible, the existence of the secondary impact category should at least be qualified for further discussion during the decision making process.

5.5 Which environmental impacts should be quantified and what economic valuations should be applied?

In Figure 5, the different dimensions of recycling-related external effects are identified. Recycling affects the whole life cycle of materials. As a result, an EPR programme may incur a wide variety of environmental costs and benefits.

Figure 5: Material, environmental and economic domain of recycling processes [26]



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26 From Valuation of waste-related externalities, unpublished paper prepared by Pieter van Beukering for RDC/Pira .International, April 2000.

In the extraction stage, recycling may reduce demands for raw materials, thereby avoiding ecosystem damage and exhaustion of scarce resources. Transportation impacts of raw materials may also be reduced.

In the production stage, substitution of primary resources by secondary materials may reduce energy demands, air emissions and waterborne emissions. For example, the production of recycled aluminium requires 97% less energy than primary aluminium production.[27] However, recycling can raise new environmental impacts, for example reprocessing impacts such as the decoupling of waste paper.

During the consumption stage, performance of the recycled product is critical. Secondary products may be less durable or cause operational problems. Conversely, some consumers may experience the purchase of recycled products as a positive environmental act, and thereby experience an improved feeling of well-being.

In the waste management phase, recycling prevents incineration and landfilling of solid wastes. Subsequently, emissions of pollutants to the atmosphere, surface waters and groundwater and dis-amenity impacts of landfill sites and incinerators are avoided, but the products of disposal (recovered landfill gas and recovered energy) are also displaced. Transport emissions associated with the collection of recovered materials and impacts associated with sorting must also be considered.

It is evident that the scope of the analysis must consider all aspects of the environmental costs and benefits across the entire life cycle. Table 7 identifies the main external effects that may be included in the evaluation of EPR programmes.

Table 7: Main externalities for consideration in the evaluation of EPR programmes [28]

Externality	Residual well-being effect
Global warming	Mortality and morbidity Damage to buildings, structures and materials Damage to forest resources and agricultural production
Air pollution	Chronic and acute morbidity Chronic and acute mortality Damage to buildings, structures and materials Damage to forest resources and agricultural production
Ground and surface water pollution	Safety and availability of drinking water Recreational value Biodiversity Damage to fisheries
Disamenity	Odour and Visual pollution Noise Congestion Willingness-to-recycle Convenience
Traffic accidents	Mortality Serious injury
Occupational health	Accidents Diseases
Macro-economic effects	Multiplier effects Employment

27 World Bank 1994. Market outlook for primary commodities. Energy, metals and minerals, International Trade Division, World Bank, Washington DC.

28 Adapted from Valuation of waste-related externalities, unpublished paper prepared by Pieter van Beukering for RDC/Pira International, April 2000.

In recent years, a number of research studies have sought to compile best available data for economic valuations, often for very specific reasons. It is not the intention of this paper to evaluate all available economic valuations and how these were achieved, but for each of the possible externalities a brief discussion is provided and some possible valuations are highlighted. It is intended that the valuations presented will provide a starting point for further discussion and methodology development.

5.5.1 Global warming

Various estimates of the marginal damage costs of greenhouse gas emissions have been reported, ranging between €6.3 and 228 per ton of carbon. The critical factors determining the variations are the assumed benchmark estimates for climate change, the time horizon considered and discount rate selected, and the vulnerability to climate change over time.

Table 8: Marginal costs of CO2 emissions (€^{1990}/tC)

Author	**1991-2000**	**2001-2010**	**2011-2020**	**2021-2030**
Nordhaus (1994)	12.4	18.6	27.4	-
Cline (1992, 1993)	6.0 – 128	7.9 – 159	10.1 – 192	12.2 – 228
Tol (1999)	11	13	15	19
Fankhauser (1999)	21.3	23.6	26.1	28.7
Maddison (1994)	6.3	8.7	11.9	15.7

A damage cost value frequently applied in the UK is $34 /tC. [29]

5.5.2 Air pollution

The valuation of human health impacts remains controversial. Values can be derived using a willingness-to-pay approach or based on the costs incurred in treating impacts (a cost of illness approach). An important factor in determining the monetary valuation achieved is the value of a statistical life (VSL) that is assumed. A frequently applied value is €3.1million (1997 prices). [30] Alternatively, mortality and morbidity can be valued on the basis of years of life lost (YOLL), an approach which is particularly recommended for risks arising from exposure to air pollution. Wide ranges of valuations have been reported, some are summarised in Table 9.

[29] Personal communication with Professor David Pearce, UCL.

[30] ExternE: Externalities of Energy, Vol 2: Methodology Part I (Impact Assessment) and II (Economic Valuation, Luxembourg, Office for the Official Publications of the European Communities, EC 1995.

Table 9: Economic valuations for human health impacts from air pollutants (€/kg of emission) [31]

Emission	Study 1 [32]	Study 2 [33]	Study 3 [34]
As	150	999	1,015,735
Cd	18.3	81.4	125,370
Cr VI	123	819	2,000,642
Ni	2.53	16.8	101,549
Dioxins	16,300,000	2,000,000	713,175,937

The characteristics of emissions and the local conditions of the receiving area play a crucial role in the monetary value of air pollutants. It may therefore be valuable to at least make a distinction between impacts from transport emissions and impacts from other processes, considering the pollutant range. Possible values for transport related emissions are summarised in Table 10.

Table 10: Local and regional impacts (€/tonne of emission)[35]

Emission	Local health impact	Regional health impact (more than 50km from source)
PM10	291,000	81,560
CO	2	1
SO2 (direct effects only)	2,180	260
Benzene (carcinogenic effects)	560	49
Butadiene (carcinogenic effects)	21,000	2,000
PAH (carcinogenic effects)	5,502,000	527,000
DME (carcinogenic effects)	1870	180
NOx (indirect impacts from nitrate aerosols)		3,154
SO2 (indirect impacts from sulphate aerosols)		3,961

31 From A Study on the Economic Valuation of Environmental Externalities from Landfill Disposal and Incineration of Waste, COWI for EC DG Environment, August 2000.

32 Study on Health Risks of Air Pollution from Incinerators, 1998.

33 Economic Evaluation of the Draft Incineration Directive, 1996.

34 Miljokostnader knyttet til ulike typer avfall, 1995.

35 Valuation of waste-related externalities, unpublished paper prepared by Pieter van Beukering for RDC/Pira International, April 2000.

Damage to structures refers to soiling of buildings and monuments caused by black smoke. The definition of black smoke is based on chemical properties of particles rather than on particle size, so the size composition of black smoke can vary considerably. However, roughly speaking black smoke consists of particles with a diameter of less than 15μm.

Damage to structures is measured in dust equivalents. The damage cost estimate applied by RDC/Pira was € 662 per tonne of dust equivalents. [36] This was based on the total UK emissions of black smoke and an assessment of the size of the UK market for cleaning buildings that is completely attributable to soiling from particle pollution.

Damage costs for forest damage and adverse effects on agricultural production have been estimated. Estimations of reductions in crop losses are carried out at an aggregated level by combining crop-specific dose response functions for O_3 damages with aggregate crop production figures for the European Union. This gives a damage cost estimate of €642 per tonne of VOC. In contrast, nitrogen may have a beneficial effect on drop yields, with some authors suggesting a value of €-697 per tonne of NOx. It is uncertain if these fertilisation effects are sustainable in the long-term. Uncertainty regarding damage estimates for forest losses is extremely high. Using various approaches, estimates in the range of €3.63 to €16.25 per tonne of SO2 have been achieved. [37]

5.5.3 Groundwater and surface water pollution

In relation to EPR programmes, groundwater pollution is most likely to be associated with leachate from landfills. Studies that focus on the external value of ground water pollution identify a wide range of willingness-to-pay for safe drinking water. Willingness-to-pay figures between €5 and €1306 per annum have been reported. [38] Bequest motives for future generations may play an important role in determining the values achieved.

Converting these values into damage costs is difficult, as knowledge of impact pathways is limited. Instead, many studies use control costs in the absence of available damage costs. Some possible valuations are summarised in Table 11.

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[36] Evaluation of the costs and benefits of reuse and recycling targets for materials in the frame of the packaging and packaging waste directive (94/62/EC), RDC/Pira International, draft report, May 2001, available from EC DG Environment.

[37] Reported in Valuation of waste-related externalities, unpublished paper prepared by Pieter van Beukering for RDC/Pira International, April 2000.

[38] Establishing priorities for groundwater quality: a contingent valuation study in Milan, J Press, Feem Newsletter 1. p. 7-10, 1995.

Table 11: Valuations of emissions to ground water and soil

Study Emission type	Study 1 (1993) per tonne waste landfilled [39]	Study 2 (1997) per tonne waste landfilled [40]	Study 3 (1995) per kg emission to water	Study 3 (1995) per kg emission to soil [41]
Leachate	0.77 (0-1.54)	0 – 1.09	-	-
Lead (Pb)	-	-	178	5
Cadmium (Cd)	-	-	622	1,514
Mercury (Hg)	-	-	1,022	37
Dioxins	-	-	62,824,889	n/a
Antimony (Sb)	-	-	121,366	121,366
Arsenic (As)	-	-	308	12
Barium (Ba)	-	-	31	37
Beryllium (Be)	-	-	44,928	44,928
Copper (Cu)	-	-	5	1
Chromium (Cr) [42]	-	-	17,479	320
Nickel (Ni)	-	-	12	4
Selenium (Se)	-	-	16,125	16,125
Zinc (Zn)	-	-	1	1

More valuation studies are available for surface water, but difficulties exist in transferring these for externalities relating to recycling processes:

- There is a lack of reliable data on dose response functions;
- The type of water pollution differs with the type of water contamination that is caused by recycling processes.

Empirical studies for pollution through atmospheric and soil based emissions range from €1.2 to €7kg NOx. Estimates based on cleanup costs range between €0.9 and €1.45 per tonne SO_2. [43]

5.5.4 *Disamenity*

Disamenity effects may make up a significant share of the externalities of waste related processes. Estimates of disamenity effects for visual and odour pollution have been reported using the

39 Externalities from Landfill and Incineration. Based on clean-up cost. Conversions performed with exchange rate in April 2000: 1£ = 1,71233 EURO.

40 Waste not, Want not: the Private and Social Costs of Waste-to-Energy Production. The emissions considered consist of As, Cd, Cr (type not indicated in source), Cu, Ni, Pb, and Hg. Conversions performed with exchange rate in April 2000: 1£ = 1,11495 EURO.

41 Miljøkostnader knyttet til ulike typer avfall. Conversions performed with exchange rate in April 2000: 1 NOK = 0.122792 EURO.

42 Split 50/1 between Cr III and Cr VI.

43 Reported in Valuation of waste-related externalities, unpublished paper prepared by Pieter van Beukering for RDC/Pira International, April 2000.

results of contingent valuation studies.[44] Using these estimates, disamenity values of €37 per tonne of waste and €10 per tonne of waste are calculated for landfilling and incineration respectively. To achieve this valuation, some very influential assumptions are made, such as the uniform nature of the disamenity and the neglect of income elasticity. The values should be treated with caution, and it is important to modify for variations in local circumstances such as house prices, density and processing capacity. Some practitioners also express concerns regarding potential for double counting of externalities – it is possible that the disamenity value expressed includes perceptions of environmental and health impacts that are also included in the valuations for other impact categories. No studies are available for the disamenity value of recycling facilities.

Valuing noise impacts associated with transport and disposal sites is difficult. Noise may be acute or nuisance. Important considerations are:

- The position of the noise source can very significantly change the size of the area it will affect;
- The number of people affected is dependent on population density;
- Background noise levels change throughout the day; and
- Noise is not linearly additive.

Studies have used hedonic pricing and contingent valuation in order to identify willingness-to-pay to halve noise exposure level (Table 12). Assuming a linear relationship between WTP and noise exposure an average WTP for reduction in noise exposure of €3.8 per dB(A). From this, Kageson (1993) determines the noise cost of road transport at €2-3 per 1000 kilometres and rail transport at €0.5-0.7 per 1000 kilometres.

Table 12: Summary of studies on the WTP to halve noise exposure level (in €) [45]

	Hedonic valuation	**Contingent valuation**
Pommerehne (1988)	51	46
Iten & Maggi (1988)	43	-
Willeke et al (1990)	-	81
Soguel (1994)	37	35-42

Greene et al (1997) summarises the results of studies estimating the marginal costs of congestion in the UK (Table 13).[46]

[44] Benefits Transfer for Disamenity from Waste Disposal, CSERGE Working Paper WM95-06, Inger Brisson and David Pearce, 1995.

[45] Reported in Valuation of waste-related externalities, unpublished paper prepared by Pieter van Beukering for RDC/Pira International, April 2000.

[46] The full costs and benefits of transportation: Contributions, to theory, methods and measurements, Greene, Jones and Delucchie (eds), Springer Verlag, 1997.

Table 13: Marginal costs of congestion, UK 1990[47]

	Marginal congestion costs (€/100HGV km)
Newberry (1995)	
Rural road	7.2
Urban central peak	10,742
Brossier (1996)	
Rural road	5.85
Motorway	2.95
National road	17.1

For recycling collection rounds, these estimates could be improved by incorporating details of the collection and transport processes, such as collection times, type of neighbourhood and type of waste collected.

From a purely private perspective, landfilling or incineration in a distant facility may be the optimal way to eliminate waste, but increased awareness of the issues motivates waste holders to participate at a cost to themselves. Willingness to participate in recycling schemes is crucial for the success of EPR programmes. For consumer waste streams, efforts have been made to value willingness-to-recycle as a positive externality of recycling, as consumers feel good about participating. A common method used to achieve a valuation is to quantify the time spent by the household on recycling-related activities and to value this time according to average income or at the value of leisure time. Cost-based approaches can also be used. The range of results achieved for willingness to pay for recycling using time and cost-based studies is €46-290 per household per year.[48] Huhtala identifies a marginal cost-based WTP of €20 per household per year.[49]

No studies are available that allow other convenience aspects of EPR programmes (e.g. increased weight of products through use of recyclate) to be quantified.

5.5.5 Traffic accidents

RDC/Pira (2000) value traffic accidents based on UK transport statistics. Little evidence was found in the statistics of a difference between HGV/commercial vehicles and passenger cars in terms of the accidents/deaths per km driven (Table 14) but road type is significant (Table 15).[50] Considering a VSL of €3.1million, a valuation for traffic accident fatalities of €16.9 per km travelled on an average road type is calculated.

47 Reported in Valuation of waste-related externalities, unpublished paper prepared by Pieter van Beukering for RDC/Pira International, April 2000.

48 Reported in Valuation of waste-related externalities, unpublished paper prepared by Pieter van Beukering for RDC/Pira International, April 2000.

49 Is environmental guilt a driving force: an economic study to recycling, ACTA Thesis 6, University of Lapland, A Huhtala, 1995.

50 Evaluation of the costs and benefits of reuse and recycling targets for materials in the frame of the packaging and packaging waste directive (94/62/EC), RDC/Pira International, draft report, May 2001, available from EC DG Environment

Table 14: Rate of serious & fatal accidents in UK, 1999

Vehicle type	Accidents (serious and fatal) / 100million vehicle km
Car	12
Light van	10
Goods vehicle	12

Table 15: Accidents by road type, UK 1999

	Fatalities	**Serious accidents**	**Total road traffic (billion vehicle km)**	**Deaths (per billion vehicle km)**	**Serious accidents (per billion vehicle km)**
Motorway	176	1218	83.6	2.1	14.6
Urban	1338	23011	200.2	6.7	114.9
Rural	1621	12176	183.3	8.8	66.4
All	3135	36405	467.1	6.7	77.9

5.5.6 Occupational health

Health impacts and accidents experienced by workers may be an important externality of recycling and waste related activities.

For recycling, no consistent dose-response functions can be found for health impacts, but estimates have been derived for waste related accidents. For example, in Denmark the average workforce stays at home for 1.7 days per annum as a result of accidents. In contrast, waste workers stay at home for 9.5 days per annum. Van Beukering et al (1998) estimates the overall marginal value of injuries related to waste collection at €2122 per worker per annum. [51]

5.5.7 Macro Economic effects

Standard economic theory says that it is not possible to create a job without displacing other employment. The argument is that for every job that is created, some other job is lost – the reason being that economics assumes full employment in the economy. Anyone not in employment is in a transitional stage between one job and another, rather than being "involuntarily unemployed", and has therefore internalised the costs of unemployment in their decision-making. Creating a new job for this person in recycling means that they are now not available for the job they would have taken if this job hadn't been created. There is therefore no social value in creating employment.

However, it may be that a proportion of the unemployed are not unemployed voluntarily (i.e. they are not in a transitional stage, and have not internalised the costs of unemployment in a decision).

[51] External Economic Benefits and Costs in Water and Solid Waste Investments: Methodology, Guidelines, and Case Studies, van Beukering, Drunen, Dorland, Jansen, Ozdemiroglu and Pearce, IVM Report No R98/11, 1998.

In such a case, the unemployment represents a social cost. If such involuntary unemployment represents a significant and long-term proportion of the total unemployment, then it may be argued that employment creation policies will have a positive social impact and employment should have an economic valuation.

Considering the European Union as an example, we must first decide how close EU Member States are to "full employment" (i.e. the situation where the unemployed are principally those in a transitional state). The official OECD unemployment figures by country from 1997 to May 2000 are presented in the table below.

Table 16: EU Unemployment rates

Country	**%**
Austria	3.2
Belgium	8.4
Denmark	4.7
Finland	9.5
France	9.8
Germany	8.4
Greece	No data
Ireland	4.7
Italy	10.7*
Luxembourg	2.2
Netherlands	3.0*
Portugal	4.5
Spain	14.3
Sweden	6.1
UK	5.7**

***	as at April 2000***
*****	as at March 2000***

Table 17: Percentage change in unemployment by Member State (excluding Greece), 1997–May 2000

Member State	% change
Austria	-27%
Belgium	-10%
Denmark	-16%
Finland	-25%
France	-20%
Germany	-15%
Ireland	-53%
Italy	-11%
Luxembourg	-19%
Netherlands	-42%
Portugal	-34%
Spain	-31%
Sweden	-38%
UK	-19%

From these figures, two conclusions can be drawn:

- Generally, unemployment rates in the European Union are low, half the Member States have an employment rate of less than 5%. The average unemployment rate in Europe is 6.8% (excluding Greece); and

- There is a significant downward trend in unemployment in all Member States.

It would not appear justifiable to apply an economic valuation for employment in all cases. The average unemployment rate is low (and getting lower), and from an economic standpoint the situation is approaching one of near full employment. However, the unemployment rate in some Member States is high, and probably includes a proportion of individuals who are involuntarily unemployed. In this situation, it may be appropriate to apply an economic valuation for employment. Employed people generally are healthier and happier than the unemployed.

However, measuring employment and deriving an economic valuation for employment is problematic. Whilst it may be possible to identify how many jobs are created by recycling activities, and how many jobs are displaced in disposal operations, it is not easy to establish the “net” employment created. This will be dependent on the number of jobs lost in virgin raw materials production and distribution compared to recyclate production and distribution. Even if this information can be ascertained or estimated, the difficulty remains of how to value employment, particularly whether different jobs should have different values. Can a sorters job be considered on a par with that of manager’s position at a sorting plant?

Despite these difficulties, some attempts have been made to value employment, most notably RDC/Pira 2001. [52] In the study, an employment valuation of approximately €2900 per job is applied, based on a consideration of the discounts in employment tax paid provided by the Belgian government for the creation of new jobs.

Further macro-economic effects caused by changes in waste management practices are not well researched, but may include a reduced dependency on imports, and thus improved balance of payments. This in turn may generate multiplier effects, causing national income to rise and therefore unemployment in other sectors to fall. Efforts are not generally made to quantify multiplier effects in CBA.

6. Examples of the costs and benefits of EPR programmes

It would be impossible in this paper to evaluate and draw conclusions on all EPR programmes. Therefore, this section of the paper aims to evaluate a selection of EPR programmes. In particular, instances where CBA studies have been used to evaluate EPR programmes are highlights. The implications of these studies for evaluating EPR programmes and policies are assessed.

6.1 General MSW

Within the OECD, waste volumes have been increasing at a rate similar to that of economic growth. In particular, private household consumption between 1980 and 1997 increased by 37.5% for OECD countries. During this time, municipal solid waste (MSW) production has increased by 40% in absolute terms (22% in per capita terms). To avoid increased demand for disposal facilities (landfill and incineration) for MSW, then policies to encourage improved resource efficiency must be pursued.

In both Europe and the US, the packaging waste management hierarchy has been put forward as a solution to increased MSW production. This hierarchy places source reduction at the top of the agenda, followed by reuse, recycling and final safe disposal in terms of priority.

Palmer and Walls (1999) evaluate the cost efficiency of different financial incentives for reducing MSW in the US. [53] They conclude that many options are available to achieve the twin goals of shared financial responsibility for waste management and more “design for the environment”. Manufacturer take-back coupled with collection and recycling targets is the best known of these options, but other instruments may provide a more cost-effective solution. In particular, an upstream combination tax/subsidy (UCTS) imposes fewer transaction costs than take-back approaches. Alternatively, a unit-based pricing policy could also be applied. The authors calculate that widespread application of this approach at a rate of $1.00 per 32 gallon trash bag could reduce MSW by approximately 13%.

However, the study concentrates on the cost-effectiveness of different options to achieve a defined goal. It does not attempt to ask the more fundamental questions asked by industry – is the goal justified, and what targets should be attained. To determine the answers to these questions, a cost benefit analysis approach is required. The results for two CBA studies for MSW disposal in the EU are presented below.

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52 Evaluation of the costs and benefits of reuse and recycling targets for materials in the frame of the packaging and packaging waste directive (94/62/EC), RDC/Pira International, draft report, May 2001, available from EC DG Environment.

53 Extended Product Responsibility: An Economic Assessment of Alternative Policies, Karen Palmer and Margaret Walls, Discussion Paper 99-12, Resources for the Future, January 1999.

6.1.1 Cost-benefit analysis of Different Municipal Solid Waste Management Systems: Objectives and Instruments for the Year 2000, Coopers and Lybrand/CSERGE, March 1996

This *ex ante* analysis, which aimed to provide input into the elaboration of a comprehensive waste management strategy, remains one of the most complete assessments of general MSW management options. Having generated detailed internal cost data for waste management activities in twelve European countries, the study projected the likely costs and benefits under three different scenarios:

- An "as is" or business as usual baseline scenario;
- A green scenario, which assumed a progressive greening of consumer tastes and preferences, reflected by increased recycling and increased demand for recycled materials; and
- A technology scenario, which builds on the green scenario but includes favourable technological changes which reduce unit costs.

From an environmental perspective, the study concludes that recycling offers significant environmental benefits, but these vary considerably between Member States due to differences in transport costs, energy savings, and the mix of recycled products. Municipal composting results in net environmental costs (mostly associated with transport), and therefore home composting may be a more attractive option. Incineration with energy recovery leads to significant environmental benefits if it offsets higher polluting marginal power sources. Landfill has a relatively small net environmental cost with or without landfill gas recovery.

From a total social cost perspective, the study suggests that the net total social costs of recycling are significantly less than those of landfill or incineration. They also suggest that total social costs of landfill are less than those of incineration. The small economic costs associated with composting schemes are more than outweighed by the net environmental costs. Although no attempt has been made to evaluate source reduction, the researchers conclude that for *a priori* reasons this option will provide greater potential cost effective environmental improvements.

The report draws the following important policy implications:

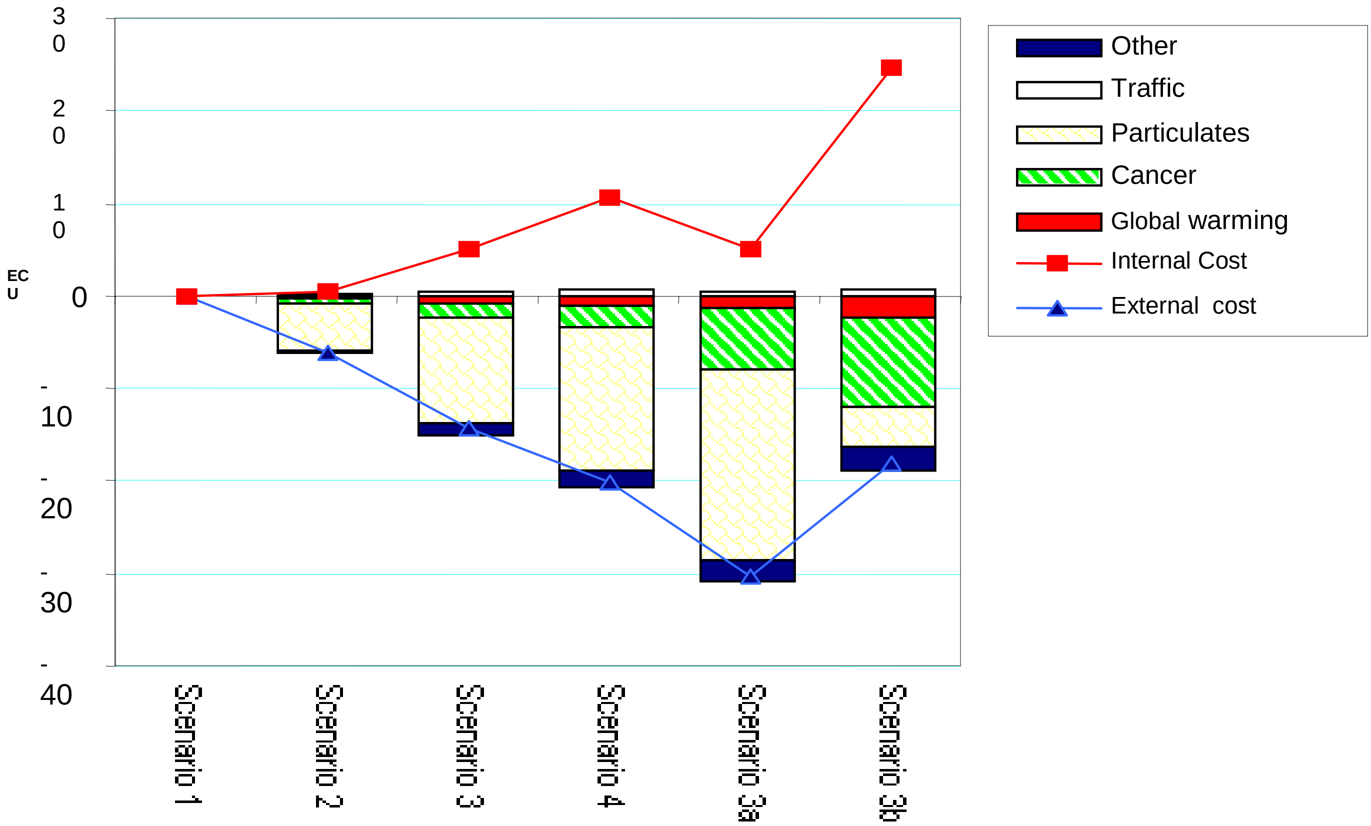
- Policy should be geared towards source reduction and recycling;
- Economic instruments are a potentially useful tool but are insufficient alone. There is a need for continuing regulation;
- Most of the costs and benefits identified arise from transport and energy processes – the ideal policy would integrate transport and energy policy, but such issues go beyond the scope of the research;
- The existing arrangements for funding collection and disposal of MSW in many countries inhibits the development of an economically desirable pattern of waste management, since in many cases the costs to users of services do not depend on their level of use of these services; and
- The differences in environmental costs and benefits between Member States suggest that a uniform policy across the EU would not be appropriate. Indeed, certain aspects of waste management are best dealt with at a local level.

However, the authors recognise that many environmental impacts have not been valued, including important impacts such as disamenity, leachate emissions from landfill, and some air emissions from incineration. Advances in cost-benefit analysis methodologies would now allow these impacts to be considered.

6.1.2 Life Cycle Inventory Development for Waste Management: Development of a Waste Management Scenario, Project Record P1/392/9, by Pira International for UK Environment Agency, 2000

This study, conducted as part of a wider CBA methodology development project, investigated the costs and benefits of a scheme to introduce separate collection and recycling of a range of materials from the MSW stream for a typical UK small town or semi-rural situation.[54] The study results are presented in Figure 6, the scenarios considered are summarised in Table 18. In defining the scenarios, no account was taken of available markets for recyclate. The scenarios aimed to investigate the achievement of a proposed 25% recycling target for MSW in the UK.

Figure 6: Costs and benefits alternative MSW management options



[54] Development of a combined methodology to evaluate recycling processes based on life cycle assessment (LCA) and economic valuation analysis (EVA), Pira International, IVM Amsterdam, Fraunhofer Institut, 1996-1999.

Table 18: Scenarios studied

Scenario	**Description**
Scenario 1	Base case – All material to landfill except a small quantity to of glass delivered to bring banks (recycling rate achieved = 3%
Scenario 2	As base case, but additional bring facilities at civic amenity cites and supermarkets (Recycling rate = 6%)
Scenario 3	As scenario 2, but including a green box scheme to collect aluminium and steel cans, plastic bottles and paper fortnightly (Recycling rate = 12%)
Scenario 4	As scenario 3, but including composting of green waste delivered to civic amenity cites, plus addition of glass, other plastic waste, and textiles to green box scheme (Recycling rate achieved – 25%)
Scenario 3a	As scenario 3, but considering replacement of the landfill site's spark ignition gas utilisation engine with higher specification turbine based engine
Scenario 3b	As scenario 3, but involves building of an incinerator which takes all material from residual collection and all combustibles from CA sites

The following conclusions can be drawn:

- Within the systems studied, increased recycling appears to give substantial environmental gains, although there may be some evidence of diminishing rate of returns on capital employed;

- The improved gas utilisation engine gives a clear environmental improvement at an apparently negligible cost. This suggests that an environmental policy which focuses solely on recycling activities may overlook opportunities for more cost effective environmental improvements;

- The potential benefits of central composting schemes are unclear; and

- Incineration appears environmentally preferable over landfill, but the high internal costs identified may reduce the desirability of this option.

The secondary impact categories are also presented and evaluated. Generally, these follow similar trends to those of the primary impact categories.

6.1.3 Implications for evaluating EPR programmes

Although there are limitations to the methodologies and data applied in the studies they both appear to lend support for the implementation of EPR programmes for household waste streams. The additional costs of EPR programmes that divert materials from disposal to recycling can be justified in terms of the reduced environmental impact of waste management and the avoided environmental emissions from virgin production.

However, it is apparent that both the costs and benefits of recycling programmes will be dependent on the specific products or materials addressed. Whilst the two studies presented here provide some basic guidance, EPR programmes for specific materials or products will require a more focused analysis of the environmental and economic costs and benefits.

Both studies highlight the importance of the framing of the question to be addressed. A narrow focus on recycling activities without due consideration of the wider issues could lead to missed opportunity costs.

6.2 Packaging waste

Packaging and packaging waste has been singled out for EPR programmes for several years. In Europe, the implementation of the EU directive on packaging and packaging waste (94/62/EC) has been central to the EPR debate. In Canada, packaging related EPR policies have developed over many years (for example, the development of the Beverage Container Stewardship Programme 1997 in British Columbia is the culmination of many years of activity dating back to the 1960's. Bottle bills have been adopted in more than a dozen states in the US. Minimum recycled content laws have been implemented for glass and/or plastic containers in California, Oregon, and Wisconsin. The Laws for the Promotion and Utilisation of Recycled Plastic Containers implements EPR thinking for plastic packaging in Japan. The application of the National Packaging Covenant in Australia is ongoing.

The mature nature of EPR policies for packaging means that many studies are available for this sector. Most studies focus on cost-efficiency aspects, but in response to the implementation of the controversial EU Directive on packaging and packaging waste Directive (94/62/EC) more recent studies in Europe have begun to adopt a cost benefit approach. These packaging studies have been key to the further development of CBA methodologies.

Two packaging related CBA studies (one for Europe and one for Australia) are discussed below, in order to demonstrate the broad findings and to illustrate some of the issues involved in conducting CBA studies.

6.2.1 Evaluation of costs and benefits achieving reuse and recycling targets for the different packaging materials in the frame of the packaging and packaging waste directive 94/62/EC, by RDC/Pira International, for EC DG Environment, Draft final report May 2001

This is the most high profile of the studies performed for the packaging sector in Europe, as it has informed the Commission review of the Directive (94/62/EC).

The consultants recognised that to perform a full cost benefit analysis for all packaging recovery and recycling options in all EU Member States would be an immense task, and its added value might be limited. The scope of the study was limited so as to produce a representative but accessible analysis, using only readily available data sources. Choices had to be made which lead to simplifications that do not always represent the full details of real conditions. Despite these limitations, the study was able to draw a number of important conclusions.

The study concluded that (with some notable exceptions) the selective collection of both household and industrial packaging for recycling is better for the society than its treatment together with unsorted waste. For packaging waste collected from households, separate kerbside collection is frequently the preferable options due to the higher collection and recycling rates that can be achieved. Notable exceptions are:

- Glass – which should be collected from bottle banks ;
- Metals – which should not be collected selectively in areas where the MSW is incinerated with metals recovery, and may also be collected selectively by a bring system in areas with a low population density;

- LBC, composites and mixed plastics – which should not be collected selectively, but should be treated by landfill or incineration as part of the mixed MSW stream; and

- Plastic bottles - which should generally be collected by separate kerbside collection, but in low population density areas where MSW is incinerated with efficient energy recovery the material should be collected by a bring scheme.

For packaging waste collected from commercial and industrial premises, generally it is preferable to collect industrial packaging separately for recycling, except in instances where only a small amount of packaging waste is produced.

Considering the results of the analysis the report recommends material specific recycling targets for each Member State. The recycling rates recommended are dependent upon:

- Population density distribution;
- Available MSW disposal option (landfill or incineration); and
- Packaging mix available on the market.

In defining these recycling targets, the research considers the maximum collection and recycling rates achievable where an efficient and mature scheme is applied. No consideration is given to available end-use markets for the recyclate.

Table 19: Member State recycling targets recommended in RDC/Pira

	Global Target Industrial waste		Global Target Household waste		Global target (Industrial + Household waste)	
	Min	Max	Min	Max	Min	Max
Austria	56%	74%	42%	60%	49%	67%
Belgium	54%	70%	42%	65%	48%	67%
Denmark	54%	70%	53%	66%	53%	68%
Finland	57%	73%	35%	48%	48%	63%
France	53%	72%	45%	68%	50%	70%
Germany	56%	72%	45%	71%	51%	72%
Greece	53%	70%	39%	52%	46%	61%
Ireland	50%	67%	27%	38%	40%	54%
Italy	54%	71%	44%	65%	49%	68%
Luxembourg	54%	70%	46%	66%	50%	68%
The Netherlands	55%	71%	44%	64%	51%	68%
Portugal	57%	75%	46%	64%	47%	65%
Spain	50%	66%	47%	65%	49%	65%
Sweden	59%	76%	44%	54%	52%	66%
United Kingdom	56%	72%	39%	64%	49%	69%
EU	54%	71%	45%	65%	50%	68%

Thus, there is no uniform recycling rate that can be applied across Europe or across materials. Local factors and material specific factors dictate a wide range for the recycling targets. The conclusions would seem to provide support for differentiated material specific targets for packaging waste, and suggests the following range of targets (see Table 20).

Table 20: Material recycling targets proposed by RDC/Pira

	Minimum recycling rate	Maximum recycling rate
Plastic	28%	38%
Steel	60%	75%
Aluminium	25%	31%
Wood	47%	65%
Paper & board	60%	74%
Glass	53%	87%
Composites	0%	0%

The study also investigates the costs and benefits of reuse of beverage packaging. The study concludes that, from a total social cost perspective, there is no case for universally encouraging the use of returnable beverage packaging. The environmental and economic costs and benefits are dependent on a range of local and regional factors, including number of reuses and distance to market.

Despite drawing these conclusions, the report makes considerable efforts to highlight the limitations of CBA techniques and draws attention to sensitivity analysis. Subsequent to the issuance of the final report, a stakeholder consultation process has been established by the European Commission. This process has revealed that some of the underlying assumptions and environmental and economic data applied could be improved, thereby improving the quality of the study.

6.2.2 Independent Assessment of Kerbside Recycling in Australia, Enviros RIS and Nolan ITU, June 2000 for the National Packaging Covenant Council

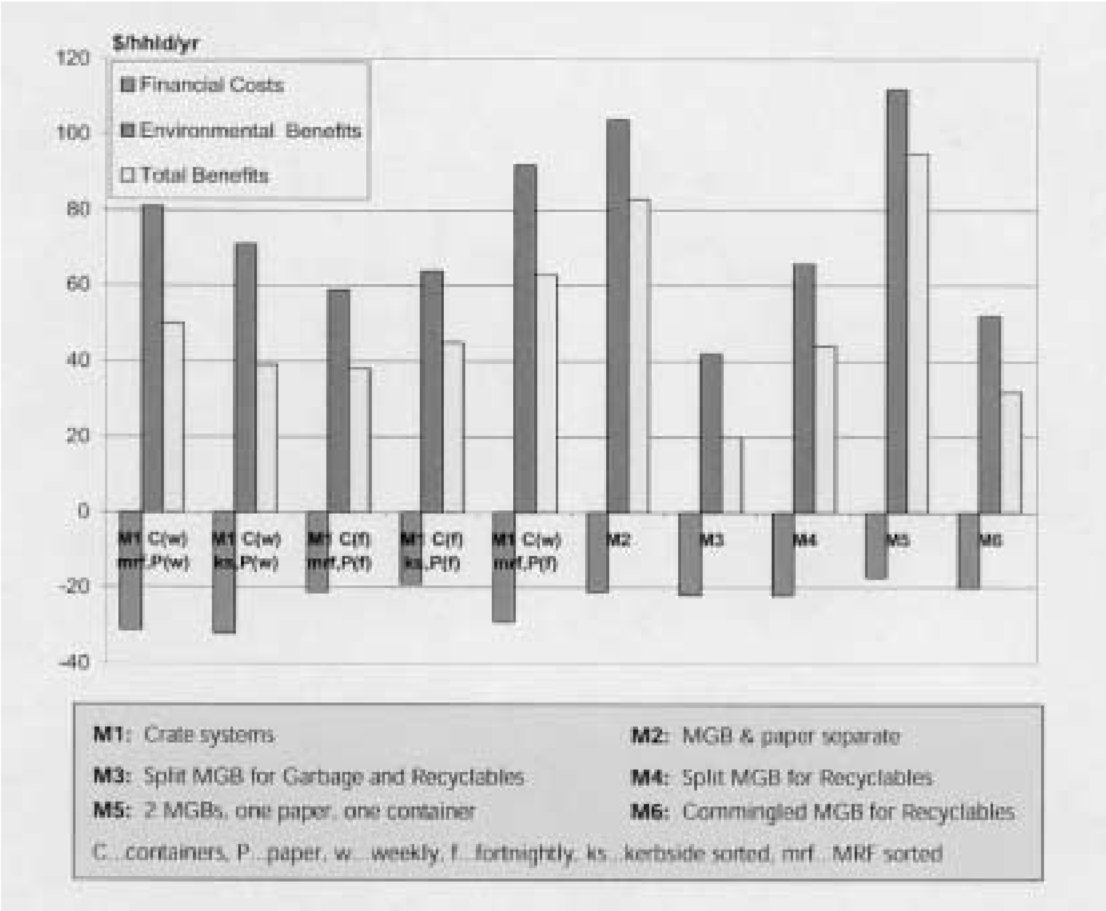
The Covenant represents a landmark agreement to foster efficient and environmentally sustainable systems for managing used packaging materials in Australia. In response to the development of the Covenant, and ex ante cost benefit analysis was performed. The objectives of the study were:

- To assess viability of kerbside collection and recycling systems;
- To provide a transparent framework for decision-making on a sound economic, environmental and social basis; and
- To provide a framework through which industry can assist local government to improve the long-term sustainability of recycling programmes.

The study uses a CBA approach, with economic evaluation of environmental impacts achieved using published Australian Government monetisation factors. The environmental costs are then compared against the financial costs of recycling. The financial assessment used assumes that the net cost of recycling is the cost of collection, sorting and delivery of the material minus the material revenue and the avoided costs of MSW collection and disposal.

The results for each of the schemes studied are shown in Figure 7.

Figure 7: Cost and benefits of Metropolitan Kerbside Recycling Systems – National Average



The study confirms the common perception that current kerbside collection schemes in Australia provide a net benefit to Australian communities. This finding is consistent with the results of other packaging CBA studies. The net environmental benefit of recycling over landfill in Australia is put at Aus$68, with a range between Aus$41 – 119 depending on the system and location chosen. The study identifies that recycling yields and contamination are important factors in determining the environmental performance of a system. The study also demonstrates that the benefits of recycling in urban areas are greater than for recycling in rural areas. Sensitivity analysis shows that the results achieved and conclusions drawn are not dependent upon the economic valuations used.

6.2.3 Implications for evaluating EPR programmes

The conclusions drawn by the RDC/Pira study for Europe seem to confirm the general inference of the waste management hierarchy that material recycling is preferable to disposal by landfill or incineration. This lends support to EPR programmes that encourage recycling of packaging waste.

However, the study also consolidates the view that the hierarchy should be applied with caution – in specific cases the ranking of waste management options may even be reversed. The conclusion that there is no universal case for reusable beverage packaging would seem to be counter to the general inference of the waste management hierarchy, that reuse is generally a preferred option (as a means of source reduction of waste).

In particular, the study highlights that the potential costs and benefits of any EPR programme are dependent on a number of factors, including:

- The materials targeted;
- The waste streams targeted; and
- Local circumstances, including the population density and alternative MSW disposal options.

Inclusion of a stakeholder consultation process as an integral part of the analysis can improve the quality and validity of a study.

The Enviros/Nolan study seems to confirm that recycling of household packaging waste shows a net benefit from a combined environmental and economic viewpoint. The conclusion for Australia that recycling in urban areas delivers greater environmental benefit than recycling in rural areas is seems intuitive, but contrasts to the findings of the RDC/Pira study for Europe. Closer analysis of the two studies demonstrates that the underlying assumptions made about the systems and the data sources used are fundamental in determining the results achieved and may influence the conclusions drawn. The author believes that this demonstrates the necessity to carefully determine the study goal and scope of the study.

6.3 Batteries

In the US, the Mercury-Containing and Rechargeable Battery Management Act (the Battery Act) 1996 does not establish specific take-back requirements or collection / recycling rates to be achieved. Instead the law establishes ground rules for promoting the improved waste management of batteries covered by the act. The Rechargeable Battery Recycling Corporation (RBRC) started a nationwide take-back scheme in 1994, which was expanded in 2001 to encompass the entire US industry sector.

Despite the significant financial (and environmental) commitment that this programme represents there appears to have been no attempt to evaluate the environmental and economic costs and benefits of the policy using a CBA approach. To understand the potential environmental and economic trade-offs afforded by the activities it is again necessary to turn to a study performed in Europe, as presented below.

6.3.1 Analysis of the Environmental Impact and Financial Costs of a Possible New European Directive on Batteries, ERM for the UK Department of Trade and Industry, November 2000

In response to discussions on a Batteries Directive that would introduce collection and recycling targets for spent batteries, the UK DTI commissioned a study to evaluate the environmental impacts and financial costs of alternative scenarios. In the analysis, a cost effectiveness approach has been adopted.

The study does not present the results using economic valuation to evaluate the scale of the environmental impacts. Instead, the changes to specific environmental emissions and flows of specific heavy metals to landfill are plotted against the financial costs of compliance for alternative scenarios.

The study concludes that the results are relatively complex, and reveal that there are trade-offs both between the estimated financial costs and the reduction of materials in batteries remaining in waste and between these levels of materials in waste and other environmental impacts such as global warming, resource depletion, acidification, ozone depletion and eutrophication.

From an environmental perspective:

- For all battery types considered, constituent metals are progressively diverted from landfill as recycling rates rise;

- All other environmental impacts increase as collection and recycling rates rise, due to collection and transport requirements;

- For consumer batteries, at a given collection rate, all environmental impacts are reduced as recycling rates are increased. It is more environmentally effective to recycle as great a proportion of collected batteries as possible;

- There is little evidence to suggest that there are significant environmental impacts associated with the materials in batteries which are currently sent to disposal in the UK. However, the level of materials disposed of is heavily dependent on the recycling rate achieved for industrial and automotive batteries – should lower rates for these occur, lead and cadmium to landfill would rise; and

- The impacts of batteries reported to waste are likely to be strongly mitigated by other environmental legislation, especially the Landfill Directive, the End-of-Life Vehicles Directive, the WEEE Directive and the Incineration Directive.

From a financial cost perspective:

- There are unlikely to be significant additional costs associated with the collection and recycling of automotive and industrial batteries. Costs are dominated by collection and transport, with the recycling of lead acid batteries close to break even at a lead price of €400 per tonne;

- Additional costs of collection and transport of consumer batteries are significant. Bring schemes are more favourable than kerbside, but if the collection of batteries is incremental to other kerbside collection initiatives then the costs are comparable to bring. Even in the most favourable circumstances, the collection and recycling costs exceed €1100 per tonne;

- Financial costs for consumer batteries are strongly influenced by whether the electric arc furnace route is available for recycling; and

- Implications of the proposed WEEE Directive may be significant, but cannot easily be quantified. Accurate estimates of the quantity of WEEE likely to be collected and recycled in the UK are not available, nevertheless it is fair to conclude that this will result in a small decrease in the estimated costs of consumer batteries. The costs of battery recycling would be unchanged but some of these costs would fall on the producers of EEE.

6.3.2 Implications for EPR programmes

The complexity of the results presented makes it difficult to draw precise conclusions. The complexity of these results could have been reduced by using economic valuation to achieve a single environmental impact cost, but this may have reduced the transparency of the results and suggested a simplicity that does not exist.

The approach applied highlights the application of cost effectiveness analysis as opposed to cost benefit analysis. In cost effectiveness analysis, the most economically efficient way of achieving a specified target is identified. Applying cost-effectiveness analysis to determine how to achieve a pre-set target makes the assumption that the target is rational and must be achieved. This contrasts with cost benefit analysis studies, which asks more fundamental questions of whether targets are justified and at what level targets should be set. This suggests that the goals and aspirations of EPR programmes should be set on the basis of CBA studies. However, where regional EPR programmes are implemented, cost effectiveness analysis approaches may be used to identify the best way to achieve these objectives at a national or local level.

7. Discussion and conclusions

7.1 Conclusions on the application of EPR programmes

The simplified waste management models presented in this paper demonstrate that, from a traditional economic viewpoint, conditions for achieving an optimal balance waste management mix do not exist without market intervention. The scale of the disparity is greatest for products with a low end-of-life value and/or disproportionately high collection and reprocessing costs.

If external environmental effects are also considered, the disparity between the achieved recycling rate and optimal recycling rate may become even more pronounced.

This would suggest that EPR programmes, which encourage a reduction in disposal and an increase in recycling, are potentially justifiable. To some extent this statement is further supported by the results of some of the case studies presented in Section 5, but these studies show that in some cases the additional economic costs of EPR programmes are not justified in terms of the environmental benefits afforded.

In order to determine whether an EPR programme is appropriate for a specific product, product group or waste stream, it is necessary to evaluate the economic costs of achieving the programme against the environmental benefits delivered by the programme. This approach can also be applied to determine what the goals and aspirations of an EPR programme should be

In particular, EPR programmes appear to have a significant role to play in situations where the end of life value of products is low but the scale of environmental impact of disposal is medium to high. The scale of the environmental impact may be due to specific hazards associated with the waste stream or due to the size of the waste stream considered.

7.2 Conclusions on the evaluation of EPR programmes

The matrices and models presented in this paper provide ground rules for making initial assessments as to which products, product groups or waste streams *might* be addressed by EPR. However, considering the scale of the potential costs involved in implementing, enforcing and realising EPR programmes it is prudent that a more thorough analysis of the environmental and economic implications is performed.

Subsequently, this paper advocates a life cycle cost benefit analysis (CBA) approach for evaluating EPR programmes.

In CBA, the environmental impacts of a policy are converted into monetary units by using economic valuation techniques. These external costs and benefits of the policy can then be compared against the internal costs of realising the policy.

However, CBA is a developing technique. As with all tools and techniques, there are limitations to the methodology. The limitations of CBA should be recognised and appreciated before the methodology is applied:

- Ethical issues;

- Many critics of CBA question the underlying ethics of monetary valuation of environmental impacts. However, without CBA, some other form of value judgements must be made, and these may have no rational basis;

- Methodological limitations of LCA;

- The environmental analysis that is performed prior to the economic valuation of the environmental impacts is based on life cycle assessment methodologies. Although life cycle assessment methodologies have been subject to international standardisation, there are methodological limitations;

- The nature of choices and assumptions made in LCA (e.g. system boundary setting, selection of data sources and impact categories) may be subjective;

- Models used for inventory analysis or to assess environmental impacts are limited by their assumptions, and may not be available for all potential impacts;

- Results for LCA studies focused on global or regional issues may not be appropriate for specific local applications – specific local conditions may not be adequately represented by general regional or global conditions;

- The accuracy of LCA studies may be limited by accessibility or availability of relevant data, or by data quality and data gaps;

- A lack of spatial and temporal considerations in inventory data that are subsequently used for impact assessment may introduce uncertainty to the results;

- Achieving economic valuation of environmental impacts;

- Not every externality can be valued in monetary terms at present. Several external effects are difficult to measure or the impacts are too site specific to be transferred from specific studies to a general cost benefit analysis methodology. These impact categories must still be considered as part of the evaluation, even if only a qualitative judgement is made; and

- Methodological difficulties arising from attempts to perform monetisation.

Even where monetisation can be performed, the reliability of the values derived may be questioned. A variety of techniques can be applied to derive economic valuations. In many cases, application of different techniques results in conflicting valuations being achieved, suggesting inherent bias in valuation methodologies.

- Difficulties of identifying and isolating internal costs

Estimates for internal costs of waste management related activities show wide ranges. Complex financial flows make evaluation of internal costs a difficult task. In some cases, obvious surrogate data is available (for example producer responsibility organisation fees), but sometimes this data may be misleading. Increasingly, national environmental policies have attempted to internalise some aspects of external costs. For example, emission permits and landfill taxes effectively internalise some elements of pollution. However, the charges for these permits and taxes have rarely been based on a detailed evaluation of the external costs of the avoided environmental impact. In many cases, the scale of the charges is politically motivated. It is therefore impossible to accurately determine the level of internalisation. This may lead to some double counting in the methodology.

- Quantifying indirect and secondary effects

The difficulties of quantifying indirect costs and secondary effects means that many studies focus only on the direct costs. This limited approach could have a significant influence on the true "social cost". In some cases, wider effects can only be taken into account by inclusion of broad assumptions, which may be limiting.

- Consideration of opportunity costs

Typically CBA is used to evaluate alternative approaches for implementing a selected policy measure, rather than looking at alternative policy measures. Therefore, the opportunity costs of pursuing a particular policy measure over an alternative policy measure may be overlooked. It is possible that greater gains could be achieved with the same investment in a fundamentally different policy.

However, these uncertainties and difficulties do not render CBA valueless. CBA provides decision-makers and stakeholders an insight into the trade-offs within and between economic costs and environmental benefits that are inevitably made when selecting and implementing policies. In this way, CBA becomes a useful tool for informing the development of EPR policies and programmes.

By asking the right questions it opens up the discussion and identifies key issues. Methodological and data limitations may prevent CBA from providing definitive answers to all the questions raised, but this does not render the exercise valueless. In this way, CBA makes decision-making more transparent, but it is only an aid to the decision-making process, not a substitute for it. The uncertainties mean that CBA does not provide a decision rule. The decision-maker must still judge how to weigh up environmental effects, economic costs and distributional impacts of the different policy options and select the preferred option.

It is therefore recommended that CBA type analyses be used to evaluate the environmental and economic performance of EPR programmes. *Ex ante* analysis should be performed to inform decisions on the development of new EPR programmes, but as it is difficult to predict how individuals/organisations or markets will react (the potential implications of EPR programmes are far reaching) *ex post* studies should be performed as EPR programmes progress. The implication is that EPR programmes should be fluid and iterative. In particular, the strict application of regional policies may be misleading. The wide range of internal and external costs and benefits experienced at a national or local level suggest that CBA studies may inform the development of regional objectives, but that some discretion on how to achieve these targets must be left to national or local decision-makers. At best, ranges of targets can be established at a regional level based on CBA, then a cost efficiency approach may be more appropriate for evaluating national implementation.

7.3 Recommendations for improving future work

There are a number of research areas that need to be addressed and considerations that need to be made in order to improve the quality of future CBA studies:

- Methodological framework

At present, there are many practitioners performing CBA studies. In order to improve the accessibility of CBA studies to policy makers and decision-makers, a common methodological framework for conducting CBA studies may be helpful. Due to the diversity of applications for CBA studies and considering the continuing developments in the field, this should not be a prescriptive methodology, but a set of ground rules for structuring studies. The framework proposed in Section 3 of this paper may provide a basis for further development.

As part of this consideration, closer attention to the presentation of results is required. The use of absolute numbers (often to several decimal places) is highly misleading, and suggests a degree of confidence that simply does not exist. Greater emphasis on the presentation of data ranges or use of alternative economic valuations is required, in order to ensure that the sensitivity and uncertainty of studies is fully appreciated.

- Further economic valuation research

Clearly, further work is needed in the area of available economic valuations. Further verification of existing values is required, and development of new values for impact categories not currently valued is required. This will involve efforts to determine dose response functions, etc. Continued research into the issues presented by benefit transfer is required.

- Further research into internal costs of waste management activities

Available data demonstrates that internal costs of waste management activities, especially collecting and sorting activities, are highly variable. Further research into the factors that determine internal waste management costs is required in order to inform ex ante studies in the future.

- Consideration of opportunity costs

Careful attention needs to be given to the framing of the questions asked by CBA studies. A narrow focus on recycling activities may lead to opportunity costs being overlooked.

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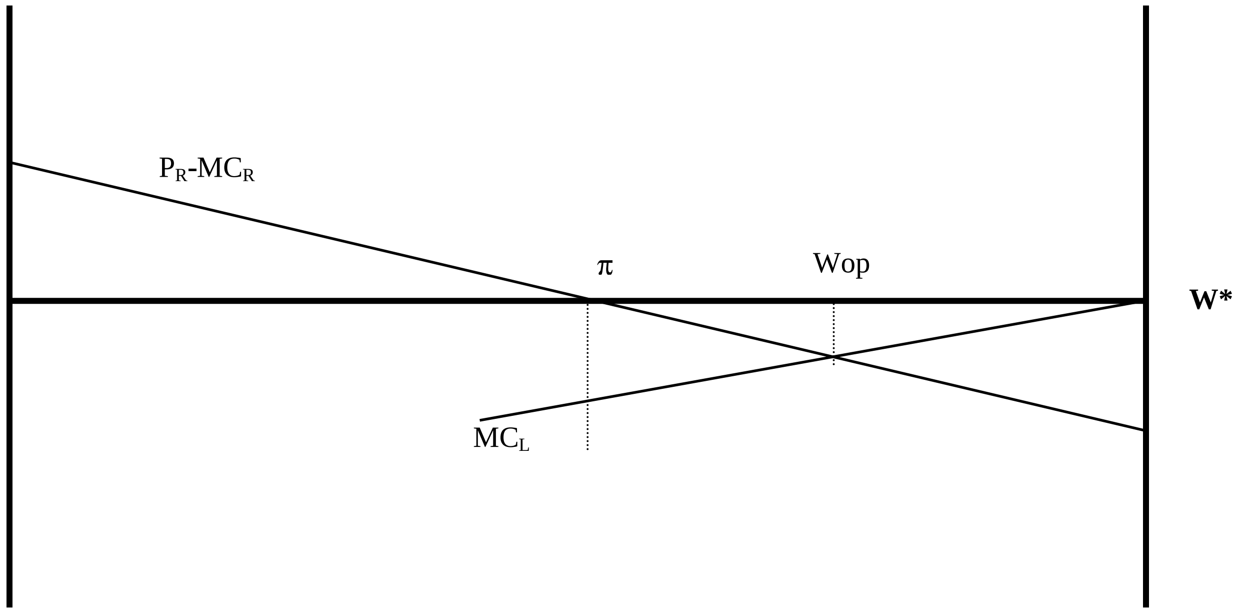
ANNEX

A BASIC WASTE MANAGEMENT SCENARIO

Figure A.1 presents a simplified economic model of a basic waste management scenario.[55] In the diagram, W* is the quantity of waste to be managed. This can either be landfilled, L, or recycled, R. It is assumed that recycling generates revenues for recyclers P_R. The marginal cost of recycling is MC_R. Thus, the net marginal profit from recycling is P_R-MC_R. MC_L is the marginal cost of landfill.

The recycling profit function is read from left to right. The landfill MC_L is read from right to left.

Figure A.1: Optimal Mix of Waste Management Options



This model can be used to investigate the recycling rates that will be achieved under various scenarios. Four possible scenarios can be considered.

55 Sourced from *"What is a Rational Waste Management Policy?"*, Professor David Pearce, included as Section I of "Development and Application of Policies for Promoting Sustainable Waste Management", draft report, International Forum of the Collaboration Projects, for the Economic and Social Research Institute, Cabinet Office, Government of Japan, August 2001.

Scenario I – recycling is operated by the private sector and landfill is operated by the municipality

In this instance the recycling level that will normally be achieved will be W_{Π} – where marginal net revenues are zero, or where total revenues (profits) are maximised.

However, the Point Wopt gives the optimal combination of landfill and recycling. This is the point where the marginal revenues from recycling equal the marginal costs of landfill. As long as the costs of landfill and recycling are borne by separate operators then there is no incentive for the private recycling company to go beyond W_{Π}. The remaining waste (W^*-W_{Π}) is landfilled.

The resulting net gains of W_{Π} are area A-area (B+C+D), whereas the resulting net gains of W* are area A-area (C+D). Area B is thus a dead-weight cost.

Scenario 2 – recycling is operated by the private sector, and landfill is operated by the private sector

Assuming that separate companies operate the recycling operation and the landfill activity, then the landfill operators charge a price to cover costs – again, the recycling rate achieved will be W_{Π}. If a single private operator operates the recycling and landfill operations, then conditions exist which may encourage W_{opt}.

Scenario 3 – recycling is operated by the municipality, landfill is operated by the private sector

In this instance, Pearce identifies that it may be possible that regulations are introduced to secure that the optimum is achieved. The publicly owned recycling operation may strive for W_{opt} even though it may provide negative net revenues, with the private sector taking the residual amount and charging for it. This would be a "planning solution" in that a deliberate command and control policy can be used to ensure that optimal recycling takes place.

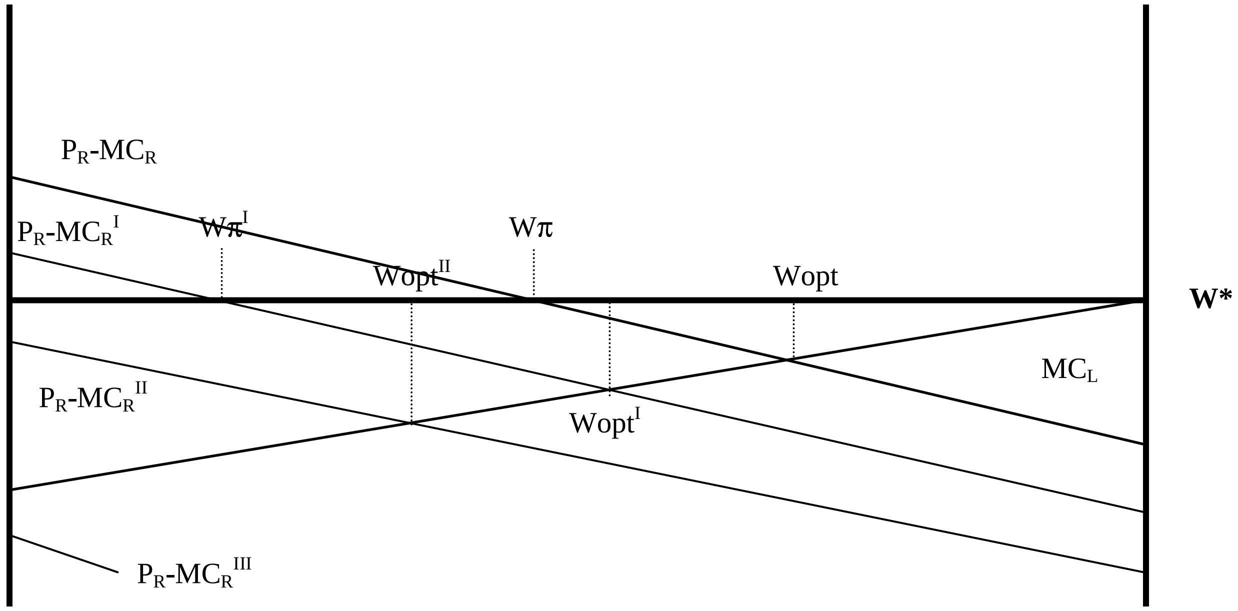
Scenario 4 – both recycling and landfill is publicly operated

In this case, a full planning solution can come about, ensuring that the optimal is achieved.

We can now use Pearce's model to demonstrate what happens as the end of life value of the material being disposed of changes.

Figure A.2 investigates the optimal recycling rate considering a situation where the recycled material has a low (or even negative) marginal net profit curve. This situation will arise where the recycled material has a low value and/or where the costs of reprocessing are disproportionately high.

Figure A.2: Reduced end of life value



The results are of course very predictable. At P_R-MC_R`, both the achieved and the optimal recycling rate are reduced ($W_{\Pi}$`and W_{opt}` respectively). At P_R-MC_R``, the marginal costs of recycling exceed the revenues received. The recycling activity is no longer economically viable and no recycling will occur without market intervention, even though there is still an optimal recycling rate ($W_{opt}$``). At P_R-MC_R```, there is no longer an optimum recycling rate. From a traditional economics viewpoint, recycling should not be pursued under these conditions.

So far the models only present the internal economic costs of waste management options considered. To facilitate a full cost benefit analysis the environmental costs and benefits (the externalities) must also be considered. This can be achieved through economic valuation techniques, which convert environmental impacts into a monetary unit (for further discussion see Section 3 of this paper). The effect of introducing externalities to the model is shown in Figure A.3.

Figure A.3: The effect of introducing externalities to the model

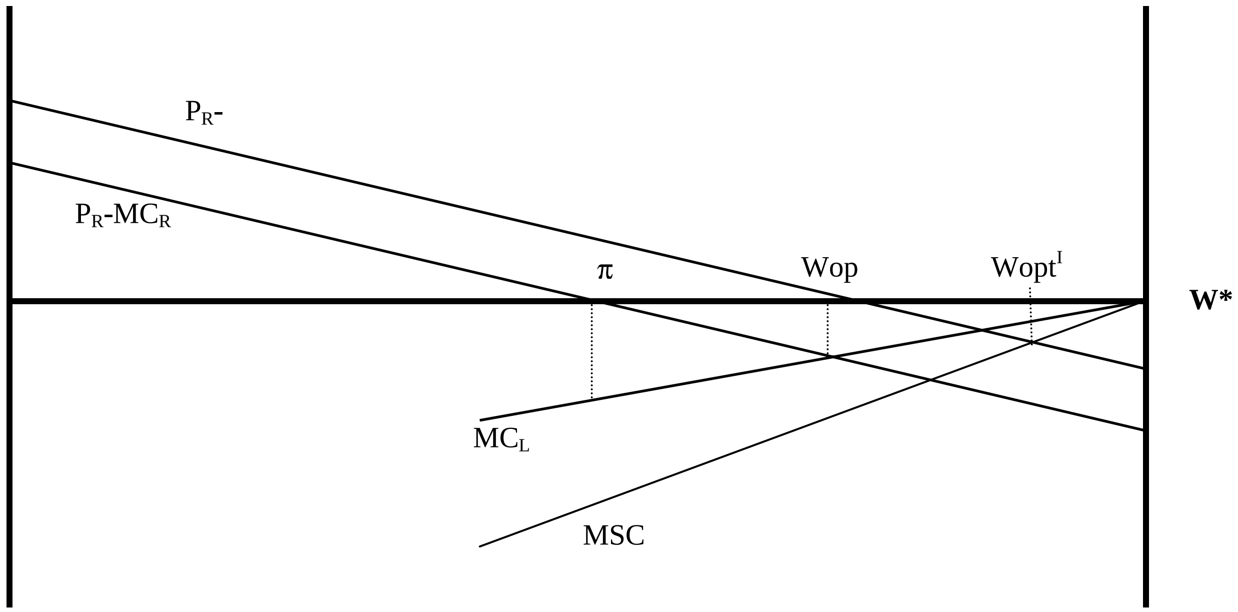


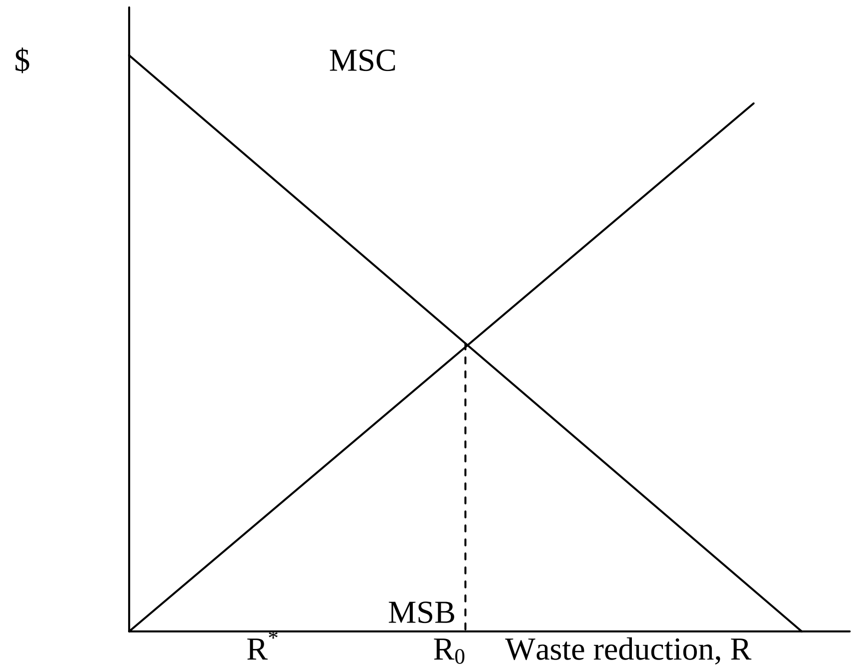
Figure A.3 considers that landfill has a negative externality (*i.e.* a net environmental cost). The combined marginal internal and external costs of landfill are the marginal social cost of landfill (MSC_L). Figure A.3 assumes that recycling has a positive environmental externality (ie a net environmental benefit – the environmental benefits of offset virgin production outweigh the environmental costs of collection, sorting and reprocessing). In this case, the influence of considering the social costs (*i.e.* combined internal costs and external costs) is to further increase the optimal recycling rate (W_{opt}`).

DISCUSSANT COMMENTARY

Comments on the Use of Cost-Benefit Analysis and EPR

by
Margaret Walls
Resources for the Future
Washington, DC

A cost-benefit analysis (CBA) is useful for setting an environmental goal or target. For example, to determine whether a given reduction in solid waste disposal is worthwhile, policy-makers might want to estimate the full social costs of achieving such a reduction and weigh those costs against their best estimates of the full social benefits. Ideally, the level of waste disposal would be set at the point where the difference between social benefits and social costs is greatest, or equivalently, where the marginal social benefits of an additional reduction in waste just equals the marginal social costs. This social efficiency result is illustrated in the following diagram.



There are several important points to be made about the CBA exercise and the diagram above. First, it is important that the marginal social cost curve indicate the *lowest* possible costs for achieving the reductions in waste. Thus, when moving to the right along the horizontal axis – i.e., when reducing waste more and recycling more – the marginal costs incurred are the lowest possible costs for achieving those reductions. So, for example, if at some point along the waste reduction axis, a further reduction is most efficiently achieved by product redesign, that is incorporated in the marginal social cost curve. Likewise, if material substitution is called for or any of a host of other changes, all of those things are reflected in the estimated costs. One can think of a set of MSC curves in the diagram above; if no product redesign takes place, for example, the MSC curve would lie above and to the left of the curve shown.

Second, all of the relevant benefits of reducing waste must be included in the marginal social benefit curve, including upstream environmental benefits that result from using fewer virgin materials and more secondary materials in production. However, it is important that these upstream benefits be net of any environmental costs that might be incurred from material substitution or other changes that are made upstream. In other words, emissions of some kinds of pollutants are likely to be lower in the production stage, but emissions of other kinds might be higher. All environmental effects need to be accounted for. Moreover, it is important that what is counted as a benefit indeed be a benefit – e.g., a reduction in pollution, not simply a reduction in the use of a particular material.

Third, as the first two points make clear, doing a good and comprehensive CBA is tremendously difficult. Estimating the full life-cycle benefits is exceedingly difficult; a line must be drawn somewhere, but where should the line be drawn? Estimating costs is often no less difficult, especially when uncertain product design changes have a large influence on those costs. Because of these difficulties, it is sometimes more useful to carry out a more limited CBA, perhaps one that ignores upstream environmental effects, for example, with some ancillary information about those effects provided but not included in the benefit estimates per se. And regardless of what approach is taken, it is essential that all the information about how costs and benefits are calculated be included in the written output so that readers can make their own judgments. Sensitivity analysis of key parameters should be done whenever possible.

Finally, the main point I would like to emphasize about CBA is that it provides information for setting an environmental goal or target, but not about what policies should be used to achieve that goal. Several CBAs are briefly discussed by Sturges in his paper, and they tend to show that the benefits are greater than costs for a sizeable reduction in waste. From these results, however, we cannot go the next step and conclude that an EPR-type policy option is called for.

EPR can encompass a number of different policy instruments but it always includes a major role for the producer in managing end-of-life product waste, either through take-back and recycling of those products, paying for recycling, and/or ensuring that recycling targets are met. One of the strengths of EPR is that it is thought to encourage DfE. As explained above, in terms of the diagram, if DfE is an important part of any waste reduction strategy, then obtaining waste reductions without it would lead to higher costs. On the other hand, EPR policies often impose extra costs in the form of administration and transaction costs. For example, if a PRO is formed, there are costs associated with setting it up and operating it. There may be costs from labeling products. If producers must ensure recycling targets are met, there are extra monitoring and information costs incurred. To fully assess whether a specific EPR policy option is a good public policy choice, it is necessary to include all of these effects and it is necessary to compare EPR to other waste reduction strategies.

DISCUSSANT COMMENTARY

Economic Efficiency of Extended Producer Responsibility

by
Thomas Eichner
Department of Economics
University of Sieben

1. Introduction

The following paper responds to the paper offered by Michael Sturges (2001). It examines the issue of allocating a resource that is first extracted, used for producing consumer goods, then recycled and finally landfilled. An Extended Producer Responsibility policy consisting of a take-back obligation, an output tax and recycled content and recyclability standards is analysed and recommended, since this policy option establishes efficiency on markets. Finally, the paper describes a procedure to evaluate Extended Producer Responsibility policies based on computable general equilibrium models.

2. Efficiency of Extended Producer Responsibility

When analysing the life-cycle of products it is necessary to begin with the first stage of the life cycle which is the extraction of raw materials. These raw materials are then used to produce consumer goods. In the post-consumption phase consumer goods are turned into residuals that are recycled. The recycling process generates two outputs: secondary materials and recycling waste. Secondary materials, like virgin materials, are used as inputs in the production of the consumer goods, whereas recycling waste has to be landfilled. Next to the quantity of the consumer goods the producers determine the product design which is a product attribute affecting the recycling and waste treatment process.

Recycling materials from post-consumer goods aims to kill two birds with one stone: (1) it diminishes the detrimental flow of waste to be landfilled and (2) it economizes the use of scarce natural resources (virgin materials). Generally speaking, economists are interested in the question of whether real economies provide an *efficient* allocation. An allocation is considered efficient if it is impossible to make some individuals better off without making some other individuals worse off, and if no resources are wasted.

There are at least two sources of inefficiency present in real economies: an *environmental externality* and a *recycling externality*.[56] From a methodological point of view Heller and Starrett (1976) make an applicable observation to this end: 'One can think of externalities as nearly synonymous with non-existence of markets.' The classical environmental externality is a so-called consumption externality, which is caused by the absence of suitable price signals for waste. As a consequence the amount of waste generated by consumers is too large.

The recycling externality is a production externality. This externality arises when producers design their products without taking into account the recyclers' needs or wants. Missing markets for

[56] The recycling externality is sometimes denoted as product design externality or missing markets of product design.

recyclability, or in other words, the missing trade of product design between producers and recyclers are responsible for the existence of the recycling externality. Both externalities render the economy inefficient, and in order to restore efficiency, policy makers have to intervene to regulate the markets.

In the last decade different policy schemes have been proposed to help restoring market efficiency. One policy concept is the so-called Extended Producer Responsibility (EPR). EPR aims at making the producer responsible for the environmental impact of his produced good at its post-consumption phase. EPR can be thought of as an umbrella concept encompassing specific policy measures. The two types of EPR instruments include:

- take-back requirements; and
- economic taxes, subsidies, standards *etc*.

The theoretical literature on recycling and product design has made strong progress in the last five years. This is due, in particular, to the publication of an article by Fullerton and Wu (1998) in the Journal of Environmental Economics and Management. In reviewing literature on policies for green design, two key questions are raised: (1) how should the government intervene if there exist no markets for product design or recyclability and (2) how should the environmental externality caused by waste be internalised?

Let us first turn to question (1). Regulatory possibilities depend on the kind of product design policy-makers have in mind. For instance, one can distinguish between product design that is non-related to material flows (*e.g.* disassembly properties, see Fullerton and Wu 1998 and Choe and Fraser 1999, 2001), and material-related product designs (*e.g.* material content or weight) as discussed by Eichner and Pethig (2000, 2001). In Calcott and Walls (1999, 2000) both types of product design are found. With respect to regulation some authors such as Fullerton and Wu (1998) propose a subsidy/tax on product design whereas Calcott and Walls (1999) think that it's not reasonable to expect that the government can make an appropriate observation of the level of product design needed to establish efficiency on markets.

In case of the environmental externality, compare question (2), a deposit-refund system which is an output tax combined with a subsidy on recycled material has been recommended as a policy choice by some authors (*e.g.* by Calcott and Walls 1999, 2000).

It should be noted that the above mentioned models differ with respect to recycling activities. Fullerton and Wu (1998) assume recycling takes place at households whereas Eichner and Pethig (2000, 2001) focus on a recycling industry, *i.e.* recycling activities are organized by firms. Clearly, whether households or recycling firms must be regulated or not depends on the structure of the model.

In this paper, EPR is understood as an intervention targeted on producers. Governments should not levy policy instruments on other actors in the product chain. Based on this understanding, this paper proposes a very simple EPR scheme consisting of the following features:

- take-back obligation;
- output tax; and
- recycled content and recyclability standards.

Based on a general equilibrium model (compare the annex) which models the whole life-cycle encompassing extraction, production, consumption, recycling and landfilling we analyse whether the above mentioned EPR scheme is efficient. After some analysis, it was found that this question can be answered affirmatively. Additionally, the analysis conducted indicates that recycling and landfilling activities should be organised privately and governments should make sure that competition takes place on markets.

The proposed policy suggested in this paper directly applies to durable consumption goods, especially to high volume and brand products, such as automobiles, washing machines or electronic sets. But even in the case of consumption residuals, such as packaging waste, our EPR policy is implementable, if mandatory take-back is organised by a private waste management organisation.

A remark is in order concerning transaction costs. Whether producers should be regulated, and which policy instruments should be introduced, depends on the transaction costs of implementing and monitoring the policy schemes. The transaction costs of our proposed EPR policy when applied to durable goods are relatively low. However, they are expected to be relatively high when applied to packaging waste. We conjecture that the transaction costs are at least lower than the benefit the EPR-regulation provides through protecting the environment.

3. Evaluation of Extended Producer Responsibility

This section addresses the evaluation of policy reforms. Sturges (2001) outlines methods to evaluate EPR programmes. He advocates cost-benefit analyses to evaluate the environmental benefits - avoided emissions or reduced waste in monetary units - and the implementation costs of EPR programmes. Obviously, if benefits are greater than the costs, EPR programmes are recommended. Sturges (2001) gives a very interesting introduction into the practical problems of evaluating benefits and costs. However, this paper proposes another way to evaluate EPR policies.

As previously mentioned, the theoretical literature on product design and recycling has made strong progress and the fundamental relations are well understood. Although the model found in the annex offers interesting insights, it does not allow for making predictions for concrete values of the policy instruments. However, the next step in this research is to provide sound empirical investigations. We are not in the position to offer empirical results but instead in this section we provide suggestions on how to do empirical simulations. Since economists have recognised the advantages of general equilibrium models in studying the material and waste flows from production, to recycling and landfilling, applied general equilibrium (*AGE*, for short) or computable general equilibrium (*CGE*, for short) models seem to be predestined to carry out evaluations of allocative and welfare impacts of policy reforms based on empirical data. AGE or CGE models became very popular in many fields of economics, *e.g.* in environmental economics (estimation of carbon taxes, see Böhringer and Rutherford 1997) or in international trade (estimation of trade policies, see Lopez-de-Silanes, Markusen and Rutherford 1996).

In the following we describe the steps which are usually parts of AGE analyses. Starting point are the industries of an economy described by input-output tables.[57] The following is then to determine:

1. production sectors: extraction, production (agriculture goods, industry goods, services ...), recycling, waste treatment;
2. primary factors: capital, labour, energy, materials;

[57] In Germany these input-output tables are available from the 'Statistisches Bundesamt'.

3. production functions: use of separable nested CES functions, estimation of the substitution elasticity;
4. consumer sector: arguments of the (representative) consumer's welfare function, which could be modelled as nested separable CES function, are consumer goods, labour supply and savings; estimation of the elasticity of substitution; and
5. public sector: consumes services and produces public goods; these expenditures are financed with tax revenues.

As is customary in AGE analyses, models are based on economic transactions in a particular benchmark year. The data of this year determine the parameters of production and consumption functions. In a final step tax reforms, *e.g.* introducing or increasing output taxes or increasing standards, could be studied and both the allocative and welfare impacts could be employed.

Of course, we offered only some brief ideas on the conceptual framework of an AGE analysis. There are experts in this field which have longstanding experiences with AGE analyses. My proposal is that AGE experts, who know the tricks of empirically modelling, together with theorists, whom have worked on the economic efficiency of extended producer responsibility, or more generally on the economics of recycling and product design, should carry out empirical investigations in order to benefit from the synergetic effects.

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ANNEX

A Simple Model and the Efficient Allocation

In this subsection a simple model is presented which is based on building blocks of the works of Fullerton and Wu (1998), Calcott and Walls (1999) and Eichner and Pethig (2001). The amount *x* of a consumer good *X* is produced using labour ℓ_x and two types of material, virgin material *v* and recycled material *r*. In its production decisions the firm also chooses the product's recyclability α. Thus the production function is given by

$$(x,\alpha) = \tilde{X}(\ell_x, v, r). \qquad (1')$$

Moving α on the other side of the equation we obtain the production function

$$x = X(\ell_x, v, r, \alpha) \qquad (1)$$

which is assumed to have the properties $X_\ell > 0$, $X_v > 0$, $X_r > 0$ and $X_\alpha < 0$. Clearly, to make the consumer good more recyclable the firm needs to use up some inputs. After consumption good *X* is turned into consumption residuals

$$x = z. \qquad (2)$$

The residuals are taken back by the producers and then supplied to the recycling sector which uses residuals *z*, labour ℓ_r and recyclability α as inputs to generate recycled material according to the recycling function

$$r = R(\ell_r, z, \alpha) \qquad (3)$$

with $R_\ell > 0$ and $R_z > 0$. In addition, an increase of the input recyclability will increase the amount of recycled material, $R_\alpha > 0$. Next to secondary material *r* the recycling process generates the output recycling waste *w* which add up in terms of weight

$$w + r = z. \qquad (4)$$

The recycling waste must be disposed of from the landfilling firm at labour cost

$$\ell_w = C(w) \qquad (5)$$

with $C_w > 0$. The extraction of virgin material is described by the extraction function

$$v = V(\ell_v) \qquad (6)$$

with $V_\ell > 0$. The household's preferences are represented by the utility function

$$u = U(w, \ell, x). \qquad (7)$$

The interpretation of $U_w < 0$ is that landfilled waste causes environmental degradation which, in turn, adversely affects the consumer's utility. ℓ is the endogenous labour supply which is demanded to produce the quantity *x*, (ℓ_x), to extract primary material *v*, (ℓ_v), to recycle material *r*, (ℓ_r), and to landfill waste *w*, (ℓ_w). Thus our model is closed by the labour supply constraint

$$\ell = \ell_x + \ell_r + \ell_v + C(w). \tag{8}$$

Now we focus on the Pareto optimal allocation which follows from solving the Lagrangean

$$L = U(w, \ell, x) + \lambda_x \left[X(\ell_x, v, r, \alpha) - x \right] + \lambda_v \left[V(\ell_v) - v \right] + \lambda_r \left[R(\ell_r, z, \alpha) - r \right]$$
$$+ \lambda_z [x - z] + \lambda_w [w - z + r] + \lambda_\ell \left[\ell - \ell_x - \ell_r - \ell_v - C(w) \right]. \tag{9}$$

The first-order conditions listed in the first column of table I can be rearranged to yield:

Proposition 1. *(Properties of the efficient allocation): The efficient allocation of the economy (1) – (8) is characterized by*

$$-\frac{U_x}{U_\ell} + \frac{R_z}{R_\ell} = \frac{1}{X_\ell} + C_w + \frac{U_w}{U_\ell}, \tag{10}$$

$$\frac{X_r}{X_\ell} + C_w + \frac{U_w}{U_\ell} = \frac{1}{R_\ell}, \tag{11}$$

$$\frac{X_v}{X_\ell} = \frac{1}{V_\ell}, \tag{12}$$

$$\frac{R_\alpha}{R_\ell} = -\frac{X_\alpha}{X_\ell}. \tag{13}$$

Each of the equations (10) – (13) is of the type: marginal benefits (terms on the left-hand sides) are equal to marginal costs (terms on the right-hand sides). Equation (10) represents the allocation rule of good *X*. Efficiency is attained if the sum of consumer's marginal willingness-to-pay for good *X* in terms of labour, $-U_x / U_\ell$, and the recycling benefit of increasing *x*, $-R_z / R_\ell$, equals the sum of marginal production costs, $1/X_\ell$, landfilling costs, C_w, and marginal environmental damage, U_w / U_ℓ. Equation (11) determines the efficient allocation of recycled material. The left-hand side captures the benefit of increasing *r* in the production sector, X_r / X_ℓ, the saved marginal landfilling costs,[58] C_w, and the reduced marginal environmental damage, U_w / U_ℓ. Obviously, increasing recycled material requires additional labor costs in the recycling process which we find on the right-hand side of (11), $1/R_\ell$. Equation (12) governs the efficient allocation of virgin material and equation (13) the efficient allocation of recyclability. In case of virgin material, the marginal productivity of *v* in the production process, X_v / X_ℓ, is required to be equal to labor costs in virgin material extraction, $1/V_\ell$. In case of recyclability, increasing recyclability leads to a (productivity) benefit in the recycling process, R_z / R_ℓ, and simultaneously causes production costs, $-X_\alpha / X_\ell$.

[58] Observe that increasing recycled material decreases the amount of waste, see also (4).

Table A.I: Efficiency and markets (notation: $\mu_x = \lambda_x / \lambda_\ell$ *etc.*; $\mu_\ell \equiv 1$)

	Row	Pareto efficiency	Markets
column		1	2
consumption	1	$-\frac{U_x}{U_\ell} = \mu_x - \mu_z > 0$	$-\frac{U_x}{U_\ell} = p_x$
production	2	$\mu_x X_\ell = \mu_\ell > 0$	$\left(p_x - t_x - p_z - \gamma_\beta \frac{r}{x^2}\right) X_\ell = p_\ell$
	3	$\mu_x X_v = \mu_v > 0$	$\left(p_x - t_x - p_z - \gamma_\beta \frac{r}{x^2}\right) X_v = p_v$
	4	$\mu_x X_r = \mu_r - \mu_w > 0$	$\left(p_x - t_x - p_z - \gamma_\beta \frac{r}{x^2}\right) X_r = p_r - s_r - \frac{\gamma_\beta}{x}$
	5	$\mu_x X_\alpha = \mu_r R_\alpha > 0$	$\left(p_x - t_x - p_z - \gamma_\beta \frac{r}{x^2}\right) X_\alpha = s_\alpha + \gamma_\alpha$
material	6	$\mu_v V_\ell = \mu_\ell > 0$	$p_v V_\ell = p_\ell$
recycling	7	$\mu_r R_\ell = \mu_\ell > 0$	$(p_r + p_w) R_\ell = p_\ell$
	8	$\mu_r R_z = \mu_w + \mu_z$	$(p_r + p_w) R_z = p_w - p_z$
waste	9	$\mu_\ell C_w + \frac{U_w}{U_\ell} = \mu_w$	$p_\ell C_w = p_w$

Competitive Markets and Regulation

In this subsection, we turn to the allocation of resources via markets. In the economy, there are competitive markets for labour with price p_ℓ, the consumption good with price p_x, virgin material with price p_v, recycled material with price p_r and residuals with price p_z. As a consequence the profit maximisation problems of the household, the virgin material extraction firm, the recycler and the landfilling firm are given by

$$\mathcal{L}^H = U(w, \ell, x) + \gamma_c \left[p_\ell \ell + \phi - p_x x \right], \qquad (14)$$

$$\mathcal{L}^V = p_v v - p_\ell \ell_v + \gamma_v \left[V(\ell_v) - v \right], \qquad (15)$$

$$\mathcal{L}^R = p_r r - p_\ell \ell_r + p_z z - p_w w + \gamma_r \left[R(\ell_r, z, \alpha) - r \right] + \gamma_w \left[w - z + r \right], \qquad (16)$$

$$\mathcal{L}^W = p_w w - p_\ell C(w) \qquad (17)$$

where ϕ is a lumpsum transfer of profits and of the net tax-subsidy revenue to the representative household.[59] The EPR policy instrument is restricted to producers. That is the reason why we consider only those policy instruments which are exclusively levied on producers. We denote t_x an output tax,

59 It is interesting to observe that the household does not pay for waste disposal. Thus there are no incentives for moonlight dumping.

s_r a subsidy on recycled material, s_α a subsidy on recyclability, β a recycled content standard[60] and $\bar{\alpha}$ a recyclability standard. While t_x, s_r, s_α and $\bar{\alpha}$ are self-explanatory, the recycled content standard β requires that a certain fraction of the output consists of recycled material, or formally

$$\frac{r}{x} \geq \beta. \tag{18}$$

Then the producer's optimization problem reads

$$\mathcal{L}^P = (p_x - t_x)x - p_z z - p_v v - (p_r - s_r)r - p_l l_x + s_\alpha \alpha + \gamma_x \left[X(l_x, v, r, \alpha) - x\right] + \gamma_z [x - z]$$

$$+\gamma_\beta \left(\frac{r}{x} - \beta\right) + \gamma_\alpha (\alpha - \bar{\alpha}). \tag{19}$$

The specification of (14), (16) and (19) reveals a particular sequence of trading residuals. The producer takes back the residuals from the consumer and sells them at price p_z to the recycler. First-order conditions of the Langrangeans (14) – (17) and (19) are listed in the second column of table I. In the economy, there are two distortions which leave the economy inefficient if there is no regulation. These distortions are:

- Environmental externality. The environmental damage caused by waste and suffered by the consumers is ignored in the producer's decisions on the amount of good *X* which in turn determines the amount of waste;
- Recyclability externality. The recyclability of residuals affects the productivity of recycling, but is chosen by the producer irrespective of the recycler's needs or wants.

In the following we investigate how the inefficiency gap caused by these distortions can be bridged. Let us begin with the environmental externality and for the sake of simplicity we will assume for a moment that the recyclability externality is absent, *i.e.* $X_\alpha = R_\alpha = 0$. Then proposition 2 offers two alternatives to internalise the environmental externality.[61]

Proposition 2. *Suppose there is no recyclability externality* $(X_\alpha = R_\alpha = 0)$. *Then either*

$$t_x = s_r = \frac{U_w}{U_l} \tag{20}$$

or

$$t_x = \frac{U_w}{U_l} \cdot \frac{w^*}{x^*} \quad \text{and} \quad \beta = \frac{r^*}{x^*} \tag{21}$$

implements the efficient allocation.

Proof: Insert $p_l = \mu_l = 1$, $p_x = \mu_x - \mu_z$, $p_z = -\mu_z - \frac{U_w}{U_l}$, $p_v = \mu_v$, $p_r = \mu_r - \mu_w + \frac{U_w}{U_l}$, $p_w = \mu_w - \frac{U_w}{U_l}$ and either $t_x = s_r = \frac{U_w}{U_l}$ or $t_x = \frac{U_w}{U_l} \cdot \frac{w}{x}$ and $\gamma_\beta = \frac{U_w}{U_l} \cdot x$ into the second column of table I which summarises the first-order conditions characterising markets and regulation. Thus the second column is turned into the first column which characterises the efficient allocation. □

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60 A recycled content standard is also discussed in Palmer and Walls (1997).

61 Optimal values of variables are indicated with a *.

Equation (20) describes a deposit-refund system. The deposit (output tax) must equal the refund (subsidy on recycled material) and both must set equal to the environmental damage caused by waste. The tax t_x induces the producer to reduce the produced amount of the consumer good and the subsidy s_r stimulates the producer to increase the use of recycled material. Both effects, *i.e.* the reduction of *x* and the increase of *r*, together drive back the amount of waste to its efficient level. If politicians should not like subsidies we offer an interesting alternative in equation (21), a recycling content standard in combination with an output tax. The recycled content standard fixes the ratio of *r* and *x* to the efficient ratio. Since there is still a degree of freedom in choosing *x* and *r*, an additional output tax is necessary to restore efficiency.

Now we pay attention to the recyclability externality and assume that the environmental externality is absent. Our results are summarized in

Proposition 3. *Suppose there is no environmental externality* $(U_w = 0)$. *Then either*

$$s_\alpha = \frac{R_\alpha}{R_1} \tag{22}$$

or

$$\bar{\alpha} = \alpha^* \tag{23}$$

implements the efficient allocation.

Proof: Insert $p_1 = \mu_1 = 1$, $p_x = \mu_x - \mu_z$, $p_z = -\mu_z$, $p_v = \mu_v$, $p_r = \mu_r - \mu_w$, $p_w = \mu_w$ and either $s_\alpha = \frac{R_\alpha}{R_1}$ or $\gamma_\alpha = \frac{R_\alpha}{R_1}$. Then the market conditions in column 2 of table I turn into the efficiency conditions listed in column 1 if table I. □

It is worth mentioning that the recycler does not obtain any price signals for recyclability, compare Lagrangean (16). The producer determines the level of recyclability of good *X* and the recycler takes the level of recyclability as exogenously given, he exhibits a Nash-like behavior of optimally responding to the 'prevailing' level of α. In the production process increasing recyclability is coupled with costs. Since the producer does not receive any price signal for recyclability, too, he will choose the level of recyclability as low as technically possible. Thus the task of the government is to induce the producer to determine a higher level of α. As shown in equations (22) and (23) this can be done by subsidising recyclability or by the command and control policy instrument $\bar{\alpha}$. Both policy options are feasible and which option is chosen depends on the preferences of politicians.

SESSION 3: EPR PROGRAMME IMPLEMENTATION: INSTITUTIONAL AND STRUCTURAL FACTORS

EPR PROGRAMME IMPLEMENTATION: INSTITUTIONAL AND STRUCTURAL FACTORS

by
Naoko Tojo and **Thomas Lindhqvist**
International Institute for Industrial Environmental Economics
Lund University, Sweden
and
Gary A. Davis
Center for Clean Products and Clean Technologies,
University of Tennessee, USA

EXECUTIVE SUMMARY

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 is written for the OECD Seminar on Extended Producer Responsibility: Programme Implementation and Assessment held in December 2001.

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

The evaluated results 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. The difference in the calculation method of collection/reuse/recycling rates, in the range of end-of-life management that is covered by the paid fee, as well as the question on the reliability of the data, posed limitation when comparing the results.

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 uncovered (*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 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 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 without consumers demanding it (*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 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 source. Due to the uncertainty as to when a product comes to its end-of-life, it is non-trivial to set collection targets for durable products (*e.g.* EEE, cars).

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).

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 increase in customer numbers.

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.

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.

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.

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.

I. 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. 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.[62] 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

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

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 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 section 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 1 summarises the EPR programmes for packaging in selected countries.

2.1.1 Collection, reuse and recycling rates

2.1.1.1 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 organised 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).

2.1.1.2 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.

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

Country	Germany	Austria	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)

(Table 1 continued)

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.

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

In Sweden, refillable PET bottles achieved the reuse rate of 91%, and refillable glass bottles, 98% in 1999, with the target between 1997-2000 being 90% and 95% respectively (ENDS, 2000h).[63]

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).

2.1.1.3 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). 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)

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

(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.[64]

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.

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).[65] 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, 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

2.2.1.1 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).[66] 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

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64 Exchange rate: SEK 1 = USD 0.095443 (Forex, 2001).

65 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.

66 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).

for the organisation and financing of the collection.[67] 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 mandated producer responsibility for these batteries (Fishbein, 1997; RBRC, 2000).[68] 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).

2.2.1.2 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

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

68 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).

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 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.[69] 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%.[70] The European Battery

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

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

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 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 3.5, 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 indicates an 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.[71] 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.[72] 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

[71] For example, In Europe, approximately 2 million tons 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).

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

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

2.3.1.1 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 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,[73] 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.

2.3.1.2 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

73 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).

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 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 practised 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 utilised 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).[74]

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 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).

[74] 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).

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 ITC (information technology and communication) 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.[75] Improper management of the discarded products also leads to the abandonment of useful resources in the landfill.

[75] 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.

In Europe, the European Union has been discussing the development of the Directive on Waste Electrical and Electronic Equipment at length.[76] 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

2.4.1.1 Collection

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 ITC 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).

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[76] 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.

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.

2.4.1.2 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.

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: 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 weight	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)

2.4.3 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).

2.4.4 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.

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).

2.4.5 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 stream, 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.

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.

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 ITC 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.

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.

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 previously discussed, 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 changes 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, s, 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 as discussed in section 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. 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, 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

PR 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).

Convenience, financial incentives, and available 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 seem 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 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 producer financial 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.

An end-of-life fee (or 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.

LIST OF ABBREVIATIONS

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

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DISCUSSANT COMMENTARY

HOW FAR DOES PRODUCER RESPONSIBILITY EXTEND?

by
Pieter van Beukering
Institute for Environmental Studies
Vrije Universiteit
Amsterdam

1. Summary

This paper aims to supplement the paper by Tojo, Lindhqvist and Davis (2001) with several dimensions that have so far been underrepresented in discussions on EPR. The central question is how far producer responsibility extends in terms of the waste hierarchy, the product lifecycle, and the global market and the harmonisation. Moreover, the issue of harmonisation of EPR programmes across countries is addressed. Several conclusions are drawn from the paper. First, setting recycling targets as part of an EPR programme should be closely coordinated within the context of integrated waste management to avoid conflicts within the waste hierarchy. Second, the emphasis on the waste stage within EPR is not necessarily justified. Other stages in the life cycle, such as the consumption stage, also require attention within EPR programmes. Third, EPR programmes should not make the mistake to reinvent the wheel as far as international markets for recyclables is concerned. Rather than developing isolated trade channels, existing markets should be used to links international recycling programmes. Fourth, the international differences in EPR concepts are a natural phenomenon. Harmonisation of EPR programmes is therefore not immediately required. This does not imply that information on EPR should not be made available in a more uniform and consistent manner. Information currently available on EPR programmes is heavily fragmented, inconsistent and unreliable. There appears to be a great need for an international effort to systematically collect information on EPR using uniform definitions of performance rates and other terminology.

2. Introduction

Extended producer responsibility (EPR), a policy option requiring producers to be financially or physically responsible for their useful life, is rapidly growing in popularity especially among European countries. EPR requires that producers take back spent products or manage them through reuse, recycling or remanufacturing, or delegate this responsibility to a third party, a so-called producer responsibility organisation (PRO), which is paid by the producer for spent-product management. The underlying idea of EPR is that placing responsibility for waste management with producers creates a strong incentive for them to redesign products with an aim toward less material use and improved recyclability (Hanisch, 2000).

This paper discusses a study by Tojo, Lindhqvist and Davis (2001) that provides a comprehensive overview of EPR programmes for packaging, batteries, end-of-life-vehicles and electrical and electronic equipment (EEE) and evaluates these programmes on their performance. Tojo, Lindhqvist and Davis pay special attention to the role of institutional and structural factors in the success of EPR programmes and the existence of institutional and structural barriers that constraint the success of EPR programmes.

Besides commenting on the content of the study by Tojo, Lindhqvist and Davis, this paper aims to supplement the paper with dimensions that have received little attention in the debate about EPR. Special attention is raised on the question how far the producer responsibility extends? Several dimensions have been formulated to initiate a discussion on the type of EPR desired. These include:

- *Recycling versus general environmental performance*: recycling targets predominantly drives most EPR programmes. This dimension questions this rather specific interpretation of extended producer responsibility.
- *Local versus global*: How does EPR perform in a globalising world? What are the implications of take-back programmes on existing trade flows of post-consumer products? This dimension addresses the need for EPR programmes to operate locally or globally.
- *Harmonised versus individual EPR*: Substantial differences in EPR concepts exist across countries. This dimension considers the advantages and disadvantages of this situation.

3. Review

Tojo, Lindhqvist and Davis (2001) are the first to systematically evaluate EPR programmes worldwide on their content and performance. The authors provide a comprehensive overview of EPR programmes for packaging, batteries, end-of-life-vehicles and electrical and electronic equipment (EEE). They pay special attention to the role of institutional and structural factors in the success of EPR programmes and the existence of institutional and structural barriers. The evaluation focuses at the performance, impact on innovation, cost of implementation, soft effects, and approaches to overcome barriers.

Most of the literature on EPR programmes so far has been limited either to non-empirical analysis or to a national evaluation of only one type of post-consumer product. Generating an evaluation of such a wide range of post-consumer products worldwide, as conducted by Tojo, Lindhqvist and Davis, is far from easy. Information is heavily dispersed and often incompatible. The authors rightly complain about non-uniformity of the definitions and the unreliability of the data. There appears to be a great need for an international effort to systematically collect information on EPR using uniform definitions of performance rates and other terminology.

Despite this handicap the authors were able to draw a number of important conclusions with regard to the general characteristics of EPR programmes across nations. In general, they are optimistic about the actual and potential achievements of EPR programmes. EPR programmes stimulate recycling significantly for several reasons. EPR programmes make resources available that governments would not be able to generate and therefore can achieve substantially higher recycling rates. Moreover, EPR bridges the gap between recyclers and designers, thereby increasing the recyclability of post-consumer products.

The data supporting the link between EPR and recycling performance, however, is too fragmented to support conclusions about the substantive positive relation between the two. As the authors themselves confirm, recycling performance is not only due to EPR programmes but also to many other initiatives. These include the increase in environmental awareness of consumers. This is a crucial aspect to the success of recycling programmes. Still, rather than attributing it to EPR programmes, environmental awareness is much the result of general education and dissemination of information.

Another aspect is the existing recycling markets that are developing independently from EPR programmes. Therefore, the authors are overstating the achievements of EPR. It is a challenge to the research community to unravel the complex mechanisms underlying EPR. The work Tojo, Lindhqvist and Davis provides a solid foundation for further studies.

4. Recycling versus general environmental performance

Most EPR programmes have adopted the objective of maximum recycling. However, higher recycling does not necessarily imply better environmental performance. According to the waste hierarchy[77], which is adopted by the EU as a guiding principle, recycling indeed is one of the preferred waste management options. The waste hierarchy, however, has always been subject to fierce criticism. For example, many believe that the options presented in the hierarchy should not be ranked in a particular order but considered as a 'menu' of alternatives. "It is not a question of good and bad waste management options. Rather, each option is equally appropriate under the right set of conditions addressing the right set of waste stream components" (Schall, 1995). Therefore, rather than supporting maximum levels of recycling, regardless of the conditions, I rather support a more general approach focused at minimising environmental impact.

In the same line of reasoning, EPR needs to be deeply embedded in the overall integrated waste management polices to avoid internal conflicts in the waste hierarchy. For example, in the Netherlands producers were required to meet high recycling targets for plastics while simultaneously the incineration capacity of municipal waste was expanded significantly. This is one of the reasons why the recycling targets in the Netherlands have not been met. The government is responsible for these conflicting strategies. Recycling programmes should be designed in a comprehensive manner, using policy instruments that match economic incentives.

Most attention in EPR goes towards the post-consumer stage of products. One reason for the emphasis on the waste stage is that waste is generally the most visible environmental impact in the life cycle. Therefore the post-consumption stage receives most attention among policy makers and producers. Waste functions as a figurehead of environmental policies. Especially for durable goods, however, improvements in other stages of the life cycle such as the consumption stage, may be much more significant.

A typical example is the life cycle of tyres. A recent study evaluated tyres from a perspective of industrial metabolism, to investigate potential novel and practical ways to reduce the environmental impact of tyres (Van Beukering and Janssen 2001). This may be achieved by focusing on technological issues such as choosing materials, designing products, and recovering materials or by looking at institutional and social barriers and incentives such as opening waste markets or changing consumer behaviour. A model is presented for the lifecycle of truck tires in Western Europe that is dynamic in nature and values both environmental and economic consequences. Various scenarios are simulated including longer tire life time, better maintenance of tire pressure, increased use of less expensive Asian tires, and increased use of fuel efficiency-enhancing tires ("eco" or "green" tyres). Figure 1 presents the main result of the study.

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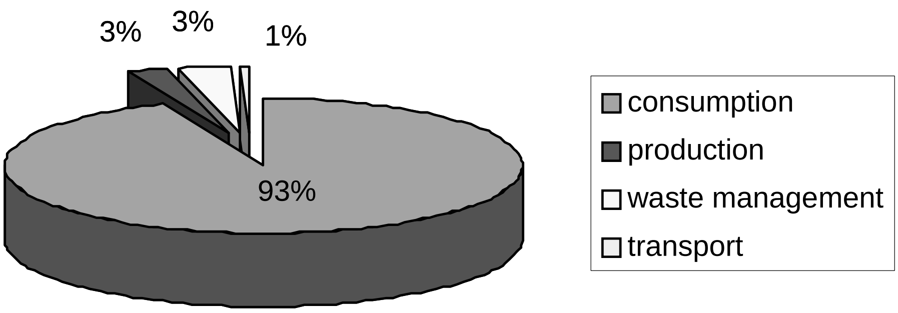
[77] The hierarchy, is based on environmental principles, and implies that waste, depending on its characteristics, should be handled by different methods: a certain amount should be prevented or reused, another share of the waste stream needs to be recycled, composted or used as a source of energy, and the remaining may be landfilled.

Figure 1: Environmental impact in the lifecycle of tires

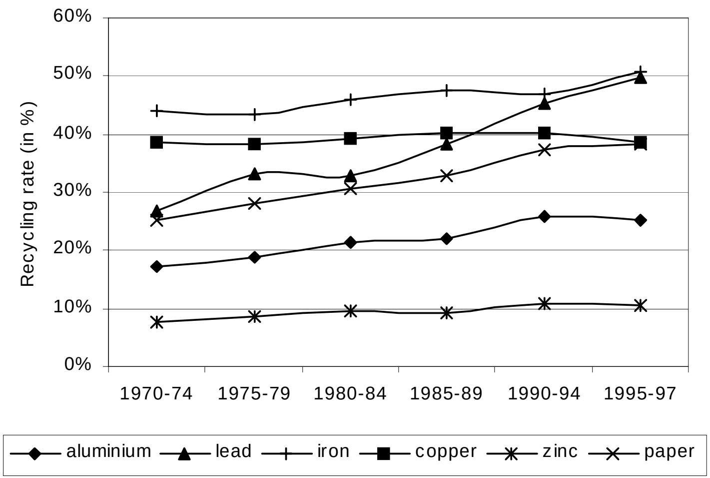


The production process of tyres generally is seen as an extremely polluting activity. Also the waste stage of tyres is perceived as damaging due to the frequent images of the mountain-high piles of discarded tyres. As Figure 1 indicates, however, 94 percent of the overall environmental impact during the life of a tire occurs before disposal during the use of the tyre, due to the impact of tires on automotive fuel efficiency. Better maintenance of tire pressure and use of eco-tires produce greater environmental and economics benefits than more durable and or less expensive (Asian) tires. These results imply that the emphasis in environmental policies related to tires should shift from the production and the waste stage to the consumption stage. It also suggests that the focus on materials throughput and associated improvements through factor 4 or factor 10 advances in reduction in mass are less important than the quality of the tires and their management.

5. Local versus global

As shown in Figure 2, many countries have experienced large increases in recycling in recent decades. The rationale behind this development varies between the developed and the developing world. In the North the increase in recycling is assumed to have mainly resulted from higher disposal costs, increased public and private concern about the health and environmental impacts of waste disposal, and a general perception that recycling can result in resource conservation. Recycling in the South is mainly driven by more economic motives (Van Beukering 2001).

Figure 2: Global recycling rates for the period 1970-1997[78]

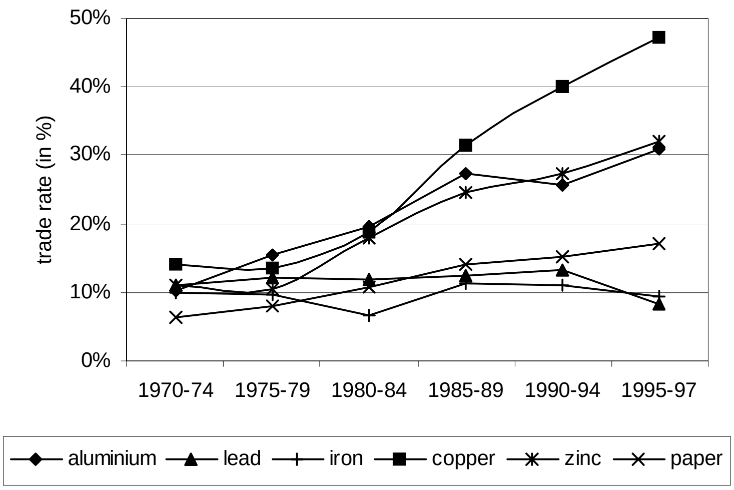


78 Global quantity of recycled materials utilised as a share of total global quantity of final production, five-year averages.

Besides domestic causes, international trade has played an important role in the expansion of the global recycling sector. In the last decades, international trade of recyclable materials has increased significantly. The combined trade volume of secondary aluminium, lead, zinc, copper and paper increased from 2.5 million tonnes in 1970 to 21.5 million tonne in 1997. Iron and steel dominate the international recycling market: the iron and steel scrap trade increased from 20 to 37 million tonnes.

To compare and evaluate the trade of secondary commodities on a global scale, the indicator 'trade rate' is used. The trade rate is defined as the total amount exported (imported) internationally as a share of the total amount globally produced (consumed). Figure 3 shows the five-year average trade rate of secondary materials. The first typical feature is the high growth rates of four of the six materials considered. For example, the trade rate of secondary copper increased from 15 percent in 1970 to 48 percent in 1997. The trade rate of waste paper and aluminium also increased significantly: namely, from 11 to 31 percent. These trends indicate how for most secondary materials the local market has changed into a global market. The development of trade in iron scrap is similar to the dynamics in recycling. The trade rate remains constant at approximately 10 percent. Trade in lead scrap also lags behind.

Figure 3: Global trade rate of secondary materials for the period 1970-1997



The explanation of the cause of these developments lies in the distinction between recovery and utilisation of recyclables. Developed countries are particularly talented in recovery. High-income countries have plenty of recyclable waste, enabling them to realise scale effects in the recovery process. Also, Western governments and industries are very active in providing incentives and the necessary infrastructure to facilitate recovery operations. Minimisation of landfilling and incineration has a high priority in the West, while consumers are generally alert to the need to minimise waste and therefore cooperate by separating a huge amount of recyclables at a low cost.

Recovery in developing countries is subject to far greater limitations. Despite the presence of waste pickers who comb the streets for valuable waste materials, recovered volumes are still insufficient - both in terms of quantity and quality - to support a mature recycling industry. That said, developing countries have a great talent for processing recyclable materials. Cheap labour ensures accurate sorting of materials, thereby improving the quality of the recycled products. Moreover, recycling industry technologies are generally less sophisticated than those in most primary industries, such that detailed knowledge and large amounts of cash for investment - both of which are particularly scarce in the Third World - are less important for recycling operations. It should also be remembered that cheap products made from secondary materials are more popular in low-income countries.

This trend in international trade raises various questions with regard to EPR. To what extend is EPR equipped to deal with a globalising recycling market? Should industries try to prevent export of recyclables to take place or should they encourage this trend? How far does the responsibility of producers reach? Does it stop with the recovery of their disposed products or are they also kept responsible for the processing of these recovered products? How do the take-back obligations of producers relate to these international flows of recyclables? Is there a need for a completely isolated trade system for EPR related material flows or should EPR markets build on existing trade channels? These are all questions that require further exploration before reliable answers can be given.

6. Harmonised versus individual EPR programmes

Table 1 shows the variety of the configuration of the waste management hierarchy throughout Europe for solid waste in general. The countries are ranked on the basis of their share of landfilled municipal solid waste.[79] Denmark, the Netherlands and Switzerland reveal the lowest landfilling shares. Italy and the UK perform poorly according to the waste management hierarchy. On the one hand, these differences may be the result of the implementation of different EPR and waste management policy instruments. On the other hand, the variation in the Europe-wide hierarchy may be the natural consequence of country specific conditions such as culture (*i.e.* environmental awareness and thus the participation rate of recycling programmes), geography (*i.e.* the extend of urban and rural communities determines transport costs), market forms (*i.e.* a monopolistic collector may charge higher prices than a more market-exposed collector).

Similar to the variations in recycling performance, differences in EPR concepts exist across countries. Is this a problem? Countries are different in many respects and therefore naturally apply different targets and policies. For multinationals, importers and exporters, however, it is a great handicap to deal with the different legislation and customs procedures.

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[79] Note that the base year differs across countries.

Table 1: Waste management hierarchy in Europe

	Year	Landfill	Incineration	Composting	Recycling
Denmark	1996	0.11	0.58	0.02	0.29
Netherlands	1998	0.12	0.42	0.07	0.39
Switzerland	1996	0.13	0.45	0.11	0.31
Sweden	1997	0.30	0.36	0.08	0.26
Austria	1996	0.35	0.17	0.14	0.34
France	1993	0.49	0.39	0.06	0.06
Germany	1993	0.54	0.18	0.05	0.23
Norway	1995	0.62	0.15	0.01	0.22
Spain	1997	0.74	0.06	0.17	0.03
Italy	1997	0.80	0.07	0.10	0.03
UK	1996	0.85	0.06	0.01	0.08
Average Europe	n.r.	0.60	0.19	0.07	0.14

Source: Resource Recovery Forum (2000)

For the most part, it is better to have country-specific policy strategies, while for some part EU-wide policies are more appropriate. This is confirmed by a study by Cagnot, Monier and Le Doré (2000) which shows that the differences in the size of the environmental costs and benefits between Member States for specific waste management systems suggests that a uniform EU policy would not be appropriate since such a policy would not reflect the fundamental differences in costs which exist. On the other hand, some of the environmental impacts associated with waste reflect either regional or transboundary impacts. These impacts are most appropriately addressed at an EU level, if not at a broader level.

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SESSION 4: INTEGRATED PRODUCT POLICY (IPP) AND EXTENDED PRODUCER RESPONSIBILITY (EPR)

INTEGRATED PRODUCT POLICY AND EXTENDED PRODUCER RESPONSIBILITY

by
Claudia Fénérol
OECD Environment Directorate

1. Introduction

At the November 2000 meeting of the OECD Environment Policy Committee (EPOC), delegates asked the Secretariat to examine the role of extended producer responsibility and its relationship to integrated product policy. In light of this request, session 4, Integrated Product Policy and EPR was added to the agenda of this seminar. The purpose of this paper is to: i) provide a general description of integrated product policy and EPR; and ii) to raise issues for discussion during session 4 of the EPR seminar.

A focus on products

Environmental policy has traditionally focussed on manufacturing facilities and the control of their emissions. Yet, what has been found over the past decade is that products also create significant impact on the environment over their life cycle. The increase in pollution and environmental impact from products is primarily due to the change in composition and complexity of products, coupled with their increased consumption. In response to the increase in environmental impacts from products, new product-related environmental policies have emerged. Sometimes these policies are cohesive and complementary, and sometimes the policies might act in contradiction to one another.

EPR is a policy used to address products at their post-consumption stage, sending strong signals upstream to the producer to take environmental considerations into the design of their products. Many OECD countries have implemented EPR policies to address impacts from specific waste streams, products and product groups.

Over the past few years several OECD countries began work to examine how environmentally related product policy could be better integrated, increasing its overall effectiveness. In light of this, integrated product policy has emerged as a new strategy for managing the environmental effects of a product over its life cycle. Its aim is to integrate policies from the resource extraction phase until the post-consumption phase, strengthening and refocusing product-related environmental policies. The European Commission has undertaken a project on IPP and recently completed a Green Paper on the subject. Currently, the Commission is developing a White Paper on IPP.

2. Background

What is EPR?

EPR is a policy in which the producer's financial and/or physical responsibility for a product is extended to the post-consumer stage of the product's life cycle. It specifically focuses on reducing the environmental impacts of a product at the post-consumer phase. There are two key features to an EPR policy: i) the responsibility for a product at its post consumption phase is shifted upstream in the production-consumption chain, to the producer, and ii) it provides incentives to producers to incorporate environmental considerations into the design of their products.

What is Integrated Product Policy?

IPP seeks to reduce the lifecycle impacts of products from resource extraction, production, distribution, use, and waste management. It focuses on decision points that influence the reduction of environmental impacts from the product, particularly on those policies, instruments and measures that would increase the environmental compatibility of a product, more informed consumer choices and the internalisation of environmental externalities into the price of the product.

What is different about IPP?

IPP is considered to be a life cycle *or holistic* approach to product policy, seeking to address environmental problems *up and down the product lifecycle* and *across different types of environmental impacts*. At the same time, however, the point of policy intervention should usually be designed to be as close as possible to different types of environmental impacts generated at different points. Through IPP, this is accomplished by *searching for synergies or links between current policies* to reduce environmental impacts over a product's life cycle.

3. Issues for Discussion

The IPP - EPR Relationship

IPP focuses on reducing the environmental impact of a product over the life cycle, targeting different actors at different points. With EPR, the policy intervention is directed at the product's post-consumer phase and sends clear signals upstream to influence product design to be more environmentally compatible. The producer has primary responsibility under EPR, however, all actors in the product chain need to participate to effectuate the policy. In contrast, each policy instrument under IPP would generally address a different actor and a different part of the product chain. Both EPR and IPP seek to vertically integrate signals across the product chain.

The basic objectives of IPP, as it is applied to the product chain, can be considered as:

- reduction of environmental impacts from the extraction of raw materials;
- improving environmental compatibility of products;
- reduction in emissions and impacts during product consumption; and
- reduction of impacts at the post-consumption phase.

All EPR instruments would help implement, either directly or indirectly, the basic objectives of IPP.

- ➢ *How can EPR policies and programmes help to meet the objectives of IPP and its implementation?*
- ➢ *What role do governments foresee for using EPR to implement IPP (e.g. internalisation of environmental externalities into the price of the product, more environmentally compatible products, reduction of waste volume, etc.)?*
- ➢ *Are any Member governments designing, or applying, EPR instruments to help implement IPP? If yes, how is EPR being used to this end (e.g. which EPR instrument, policy goals and programme objectives, which product, product group or waste stream is being targeted, etc.)?*

EPR and the mix of instruments for IPP

Member countries traditionally apply a number of instruments to address a particular environmental impact. Within the context of IPP, what other instruments can be combined with EPR to effectively implement IPP. Other measures attributed to IPP that also support the goals of EPR policy and programme objectives, include, *inter alia*, green government purchasing, ecolabelling, unit based pricing, landfill bans and taxes; removal of subsidies on virgin materials, material bans and restrictions, product bans and restrictions, virgin material taxes, waste charges, marketable permits and recycling credits.

The following table gives an overview of a particular EPR instrument and its application.

Table 2: Application of EPR Instruments

	Product or waste stream	**Stage in product chain**	**Direct response to intervention**
Deposit/refund	Specific products (*e.g.* beverage containers)	Disposal	Re-use and design
Take-back	Product and waste streams (and sectors)	Disposal with strong signals to resource extraction and design stages	Re-use, recycling some source reduction and design
Materials tax	Product (specific inputs)	Resource extraction and design stages	Reduced inputs of targeted materials and design
Advance disposal fee	Product	Disposal	Recycling and some reuse and recovery
Combined upstream tax/subsidy	Product specific	Design and disposal	Reduced material input and recycling
Recycled content	Product (*e.g.* paper and plastics, *etc*.)	Design	Design, reduced raw material input

- ➢ *What other instruments can governments combine with EPR to effectively implement IPP?*
- ➢ *Are governments combining EPR with other policy instruments to implement IPP? Which instruments and how are they co-ordinated with other environmental policies?*

Incentives and Information in the Supply Chain

Both IPP and EPR are useful in cases where incentives to improve the environmental performance of products are not being transmitted up and down the product lifecycle (design, production, use, and disposal). For instance, "markets" for some environmentally-relevant attribute of the products may be missing. By introducing appropriate policies (such as deposit-refund, take-back, material taxes, *etc*.) such signals can be transmitted when markets are unable to do so on their own. In some cases, the provision of information concerning environmental impacts up and down the production-consumption-waste chain can support more binding policies. Actors along the chain need systematic information at a level that they can easily understand to help effectuate IPP and EPR.

- *What policy measures are likely to be most effective in ensuring that incentives up and down the product lifecycle?*
- *How would asymmetric information affect the implementation of EPR as a tool for implementing IPP?*
- *What are governments doing (or intending to do) to overcome these challenges?*

Transaction/administration costs

Transaction and administration costs are often raised as a significant barrier to the implementation of policies for both governments and for the private sector. How would these costs affect the implementation of EPR instruments that might be used for the implementation of IPP?

- *What type of administration/transaction costs would be higher (or different) with an EPR policy designed to implement IPP objectives (e.g. resources -- monetary outlays, time, etc.)?*
- *Are there examples of policy implementation measures in an OECD country in which the administration and transaction costs were reduced, or for which specific measures were implemented to limit such costs?*
- *What type of components can be included in the design of an EPR instrument for implementing IPP that could reduce transaction/administration costs?*
- *Do transaction/administrative costs decrease over time as the EPR programme is implemented? If yes, in what way would these costs decrease?*

Product or sectoral affects IPP and EPR

EPR is sometimes applied to a product group such as with electronics and electrical appliances, or a sector such as the automobile sector, in lieu of a particular waste stream (*e.g.* packaging).

- *Would EPR be a more effective tool for IPP if it were applied to particular sectors?*
- *Are there EPR instruments that are more effectively applied to a particular product group or industrial sector in implementing IPP?*

INTEGRATED PRODUCT POLICY (IPP) AND EXTENDED PRODUCER RESPONSIBILITY (EPR)

by
Jacques Desarnauts
ATOFINA
France

It has been proposed that an Integrated Product Policy (IPP) should "encourage innovation of products and services and aim at continuous improvement of products and services with regard to their environmental performances within a life-cycle context" and that it should be the role of Public Authorities "to set the framework for product management by other actors and stakeholders"[(1)].

Under the Responsible Care initiative, the Chemical Industry is committed to Product Stewardship which seeks to reduce the health, safety and environmental (HS&E) risks posed by its products while also meeting customer and public demands for new products which are safer and have lower environmental impact.

The design of Sustainable Products should be based on an analysis of market needs combined with the use of tools such as Life Cycle and Risk Assessments, along with a dialogue between all involved stakeholders on the governance of risk of hazardous activities.

The common objective of both the Public Authorities with the IPP concept and of the Chemical Industry with the Product Stewardship practice is "Sustainable production and consumption" to design and market products in an environmentally-sound and sustainable manner.

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

a) **EPR only considers the waste impact of the post-consumer stage of a product's life cycle, while IPP lead principle is to cover all environmental effects on a life cycle perspective.**

b) **EPR points very clearly at a single actor in the Supply Chain -the Producer-while IPP must involve all relevant stakeholders along the product chain.**

In this context, one can question whether EPR can help promoting IPP and under what conditions.

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(1) *IÖW (Institute for Ecological Economy Research) - Heidelberg, Germany*

Two cases will be analysed:

1. The EU End of Life Vehicle Directive, with very detailed prescriptive measures which could lead to non optimal solutions from an environmental point of view. This case shows clearly that under EPR, Authorities should set key environmental objectives and leave to Producers the responsibility for product design and management of waste options.

2. A supplier of a CMR bromine alkylating agent, taking back brominated effluents of its customer in the pharmaceutical industry to regenerate bromine, demonstrates that Sustainability can often be better achieved by Voluntary Initiatives of the actors on the product chain under a Product Stewardship policy than by regulations from Public Authorities (e.g. EPR).

Sustainability can be served by different policy instruments.

- In Command and Control Regulations, Authorities tell to the stakeholders "This is the objective; this is how to do it; do it";

- EPR should be implemented in the spirit of Negotiated agreements with Authorities telling "This is the objective; you have to get there; propose how to do it". Otherwise, there will be a conflict with IPP; and

- Voluntary Instruments work when there is an economic incentive; Economic instruments can help to set up the proper conditions.

Role of Authorities vs. Producers

POLICY INSTRUMENTS:	**Command & Control Regulations**	**EPR and Negotiated Agreements**	**Voluntary Instrument**
WHAT is the objective?	***Authorities***	Authorities	Producer
HOW to achieve it?	Authorities	**Producer**	Producer

To sum it up, Industry message to Authorities is: "Don't make detailed prescriptive regulations when you can simply specify the goal and allow business to decide how to achieve it."*

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* *"A Guide to Regulatory Appraisal" - UK Cabinet Office Deregulation Unit 1998 under Prime Minister John Major.*

PRINCIPLES OF IPP AND EPR: THE PERSPECTIVE OF A GLOBAL CONSUMER GOODS COMPANY

by
Dr. Marina Franke
Procter & Gamble Services GmbH

Introduction

As a leading global consumer goods manufacturer, P&G welcomes efforts to harmonize and integrate public policies associated with ensuring the protection of human health, the environment, and the efficient use of resources. If properly designed and implemented, we believe Integrated Product Policy (IPP) has the potential to contribute to achieving this goal. It is especially important that IPP, and its individual tools such as Extended Producer Responsibility (EPR), take into account not just environmental effectiveness, but also economic efficiency and social equity, in order to be consistent with the three pillars of Sustainable Development.

Description and Basic Characteristics of IPP

As it is generally defined, IPP is intended to bring together a variety of existing and proposed tools aimed at stimulating improvements in consumer and commercial products. Therefore, it is useful to describe some of the key elements and principles that should influence the development of IPP in general, before discussing EPR specifically. Importantly, IPP should focus on harmonizing existing policies, tools and regulatory instruments so that they work together more effectively. IPP should thus establish a process that fulfils:

- <u>Environmental scope and effectiveness:</u> Promoting approaches through which all stages of the product life cycle are included in the evaluation of a product's environmental profile, including product manufacturing (from raw materials to final product), the use phase and recovery and/or disposal at the end of the product's life.

- <u>Economic efficiency:</u> It is important that the product's costs are well-managed in all stages of the product life cycle, and kept to a minimum. This is important not just to the individual consumer, but also to society in general, since efficient conservation of the public's 0money enables a broader range of societal needs to be met. This might involve efforts to include the external (environmental and social) costs in the product's costs, though there are many difficulties with such schemes.

- <u>Consumer choice:</u> It is important to keep in mind that there are different interests and priorities among consumers, including price, performance, environment, safety, and other factors, and that the consumer makes decisions based largely on personal judgments about his/her priorities.

- <u>Political and social acceptance:</u> Improvements to any public policy need to reflect both public expectations, and what can be supported from a scientific and technical standpoint. Transparency of processes applied and criteria used to make decisions about specific products or policy instruments is indispensable, so that IPP is viewed as a partnership approach. An organizational framework of a political character is necessary to allow interested parties to be kept informed, and to have opportunities for appropriate participation. Several important participants that are sometimes overlooked are small and medium enterprises (SMEs) and importers.

As noted above, IPP should consider environmental safety and burdens over the full life cycle of a product in order to not to shift environmental burdens from one stage to another, and be sure that the key contributions are being addressed. A variety of tools are available to accomplish this:

- LCA (Life Cycle Assessment) can be a useful tool to help identify and prioritize the life-cycle attributes. The series of International Standards Organization (ISO) standards 14040 to 14043 can be used here on a voluntary basis. But LCA is only one of such tools.

- Risk based decisions need to be included, which are key in product assessments to ensure overall safety. Therefore Risk Assessment should be used with other tools of Environmental Management to come to an overall scientifically sound product evaluation.

IPP should stimulate consumer information and education. Consumers need to have easily accessible, relevant and credible information about the products they want to buy:

- Consumer decision patterns should be studied extensively, so that the information provided is able to meet genuine consumer interests. The number of labels on a product should be reduced to those that provide real value (according to the consumer), to avoid consumer confusion, and existing labels need to be harmonized.

- Greater emphasis should be placed on information-based approaches, such as Type II and Type III labels, which may provide consumers with more direct and concrete information and also increase Business-to-Business communication and information. Regardless of approach, it is important that such information sharing tools follow the guidelines established by ISO 14020-14024.

IPP might include the tool of EcoDesign to support decisions and provide information to product designers. An EcoDesign tool has to include the following requirements:

- EcoDesign should not become an obstacle to innovation that provides products of value to the consumer. Innovation is essential for a sustainable society. A promising approach is provided by the activity of ISO elaborating the integration of environmental aspects into product development (*e.g.,* Technical Report ISO 14062 Design for Environment).

- EcoDesign needs to fully recognize the importance of consumer choice which includes a very personal balancing of individual expectations on product safety, performance, quality, reliability, affordability, environmental protection and other factors. Communication and cooperation with consumers is therefore essential.

- IPP has to be a partnership approach, in which roles and responsibilities are defined from the beginning of the process, taking into consideration the consequences of decisions:

- Communication is therefore a very important point in order to make IPP function. We support the arrangements of multi-stakeholder debates (*e.g.* as initiated by the European Commission).

- Expert workshops on specific topics will be helpful and provide relevant information about environmental, economic, technical and social aspects.

- IPP must facilitate coherency among different policy_instruments in a way that supports Sustainable Development. We therefore support the hierarchy of “Smart public policy making” as outlined by the WBCSD (World Business Council for Sustainable Development):

- Voluntary instruments - should be preferred as they offer the most flexibility and opportunity for innovation
- Negotiated agreements – can be effective if focused on the ends, rather than defining the means to achieve them
- Economic instruments – can offer incentives for improvement if carefully designed to avoid perverse consequences
- Command and control regulations – might be necessary to outlaw unsafe, unacceptable behaviour.

As noted above for LCA and environmental labelling, during recent years several voluntary tools and instruments have been developed and practiced to help organizations. These instruments are usually in the form of international standards, both overall organization-oriented standards such as for management systems, auditing and evaluation of environmental performance as well as product-oriented standards such as for Life Cycle Assessment, environmental labels and declarations. These standards are introduced and practised and will most likely form a solid base for the type of responsibilities the business sector are willing to undertake with regards to voluntary environmental activities in the future.

Relationship of IPP and EPR

The concept of IPP is based on the consideration of the impacts related to products throughout their life cycle, from natural resources, through their use phase until their recovery and/or disposal at the end of the product's life. EPR (Extended Producer Responsibility) has mainly developed through practical experiences of recovery and waste management systems, i.e. has a focus at the end of a product's life. Therefore EPR can be seen as one tool under the umbrella of IPP, along with the others described above.

According to the three pillars of Sustainable Development, the social dimension of IPP requires a need for open and constructive dialogues, cooperation and also appropriately shared responsibilities between all sectors of society – legislators, authorities, business, retailers and consumers. This should also be a guiding principle for EPR under IPP. For the success of IPP, a positive response from business and promotion of activities on the company level is essential to lead to overall environmental, economic and social improvements.

Under the umbrella of IPP, EPR should meet each of the key elements and principles outlined above. But in particular, it is strongly recommended to add a detailed economic assessment of the impact of existing EPR schemes at a practical level and a comparison between EPR and other models. Here the economic and environmental effectiveness need to be analysed and assessed. The experience of all stakeholders with EPR systems, including consumers/citizens, waste management industries, authorities and business is essential for further development of EPR under IPP. Current EPR policies should broaden their focus on the policy objective – improved environmental and economic and social performance of products through their life-cycle (including waste management) – and offer a range of alternative tools that could be used to meet that objective. It is very important to recognise that improvements in environmental performance cannot be made at the expense of product safety, quality or consumers choice / customer satisfaction.

DISCUSSANT COMMENTARY

COMMENTS ON INTEGRATED PRODUCT POLICY (IPP) AND EPR

by
Bette Fishbein
INFORM, Inc.

1. Parallels with discussion at President's Council on Sustainable Development

IPP has not been a topic of discussion in the US but the debate on this in Europe is very reminiscent of the one that took place at the President's Council on Sustainable Development (PCSD). This Council was appointed by President Clinton in 1993. It was a very high level multi-stakeholder task force which ultimately recommended that the US government adopt a policy of EPR. However, it redefined what that means.

The following four points were at the heart of the debate at the PCSD and the re-definition of EPR and are again being raised in the discussion of IPP in Europe.

1. emphasis on life-cycle environmental impacts of products rather than a focus on waste;
2. emphasis on shared responsibility;
3. focus on providing information;
4. recommending that programs should be voluntary not mandatory.

US industry supported the broadening of EPR to include all life-cycle impacts rather than focusing on waste. In Europe, the same argument is being made and some are going further—arguing that social and economic factors must be included, in addition to environmental ones, in developing IPP. This broadening is useful as an umbrella concept. However, specific policies must be implemented if progress is to be made in developing sustainable products. Clearly, there are trade-offs between different parts of the life cycle and different goals, yet these need not cause paralysis in developing policies. Different tools can be used to address different phases of the life cycle. For example, fuel efficiency standards can address the use phase of vehicles but this does not preclude EPR from addressing the end-of-life phase. Of course, safety must be paramount in considering other goals.

INFORM's research on EPR programs indicates that they all have some form of shared responsibility. Even when producers pay the fees for end-of-life management, consumers must bring products back and the fees are often passed on to consumers in higher prices. The key in implementing "shared responsibility" is to assure that producers have enough financial and/or physical responsibility for their products at end-of-life to provide a strong incentive to design products that are less wasteful and more recyclable.

EPR is criticized for only addressing the waste phase. In fact, this is the point of intervention for EPR but it has impacts across the life cycle. Products that are less wasteful and more recyclable reduce the amount of virgin materials that need to be extracted, processed and manufactured thereby significantly reducing environmental impacts well beyond those caused by disposal.

Providing information about environmental impacts of products to consumers is useful in encouraging them to purchase green products. However, it is important that this be a supplement to EPR not a substitute for it.

INFORM research has documented many voluntary EPR programs. Our findings on this issue follow.

2. Voluntary vs. Mandatory EPR Programs

The distinction between voluntary and mandatory is not really either/or for in reality there is a continuum. It ranges from truly voluntary programs that companies implement because it is in their financial interest to do so, to those in the mid-range that are implemented to avoid product bans or take-back mandates, to those that are actually required by legislation.

Xerox and Kodak have take-back programs that are truly voluntary. Xerox implemented "asset recovery" to generate profits. It redesigned its products to enhance the recovery at end of life, implemented extensive re-manufacturing and recycling systems and created a program that is highly profitable. Kodak implemented take-back to save a new product---its single-use camera. The throwaway camera met with strong opposition from environmentalists and consumers because of the waste it would generate. Kodak claims this is now one of the most highly recycled products in the world. The company is now the recipient of environmental awards for this camera rather than public reprobation and the take-back is profitable. Programs like these that are profitable for the companies that implement them, work well on a voluntary basis.

Battery take-back in the US is a good example of a program mid-way on the continuum. This is known as a "voluntary" program. It was implemented because eight states passed legislation requiring take-back of Nickel-Cadmium (Ni-Cds) batteries and many other states had proposed such legislation. There was also the threat of bans on Ni-Cds. Industry decided that rather than being subject to different laws in each state and facing bans, it preferred to launch a national take-back program. It formed the Rechargeable Battery Recycling Corporation (RBRC)---a PRO that sets and collects the fees and administers the take-back and recycling program.

This program entails a net cost to producers, not a profit, and illustrates some key weaknesses of voluntary programs with net costs.

1. the programs often have no firm targets for amounts collected or amounts recycled;
2. no reporting is required;
3. there are benefits for "free riders"; and
4. the programs do not reward the companies that "do the right thing"—in fact, such companies have a competitive disadvantage.

RBRC originally set a goal of recycling 70% of Ni-Cds by 2001. It later put off the 70% goal to 2004. In 1998 RBRC published estimates of the amounts it expected to recover each year, as shown in the table below.

Ni-Cd Battery Recycling in the United States and Canada

Calendar Year*	Total Recyclable Pounds Entering Waste Stream	RBRC Market Penetration	RBRC Program Pounds Entering Waste Stream	RBRC Program Pounds Recycled	RBRC Program Recycling Rate
1993	14,221,000	-	14,221,000	284,000	2%
1994	15,760,000	-	15,760,000	630,000	4%
1995	17,921,000	-	17,921,000	2,703,000	15%
1996	20,542,000	-	20,542,000	3,078,000	15%
1997	22,454,000	75%	16,840,500	3,782,000	22%
1998	23,231,000	80%	18,584,800	4,646,200	25%
1999	26,330,000	81%	21,327,300	6,398,190	30%
2000	27,917,000	82%	22,891,940	8,012,179	35%
2001	28,242,000	83%	23,440,860	9,376,344	40%
2002	28,199,000	84%	23,687,160	11,843,580	50%
2003	28,032,000	85%	23,827,200	14,296,320	60%
2004	28,035,000	86%	24,110,100	16,877,070	70%
2005	28,027,000	87%	24,383,490	19,506,792	80%

* Numbers for 1998 to 2005 are projected; numbers for 1997 are under review by RBRC.
Source: Rechargeable Battery Recycling Corp., "Charge Up to Recycle," Fall 1998.

RBRC has not reported on its recycling rates since 1998. It has provided some information indicating that it recycled 3.5 million pounds in 2000, far less than its own earlier estimate of 8 million pounds. Since this is a voluntary program there are no consequences due to the shortfall. As is generally true in voluntary programs, "free riders" are a problem---amounting to about 10-20% of the "producers."

Another weakness of voluntary programs is that they do not usually specify what should be done with the products that are taken back. In many cases they end up in landfills, incinerators or are exported. RBRC has not been criticised in this respect as it sends all of the spent batteries it collects to a recycler, INMETCO in Pennsylvania.

In the US electronics industry, companies like IBM and Hewlett Packard are implementing voluntary take-back programs for computer equipment. These programs charge consumers for sending back equipment and require them to arrange for the packing and shipping of the used equipment. There is no public reporting which gives rise to concerns that little equipment is being recovered.

A ray of hope for "voluntary" initiatives is the recently negotiated agreement for the take-back of carpet, described in the paper authored by Wilt and Hickle. The agreement includes negotiated targets and requires reporting. This is close to mandated programs on the continuum. Evaluation of the accomplishments of this program will provide very useful information on the effectiveness of this approach in the US.

3. Conclusions

A broad umbrella concept like IPP can be useful in coordinating policies aimed at development of sustainable products. IPP can include tools to complement existing legislation on chemicals and waste such as using the tax code and procurement guidelines to provide incentives for making such products and creating markets for them. INFORM has done extensive work on procurement policies and strongly supports this as a tool for sustainable development. Some criticisms of suggested design guidelines in IPP are well taken. The challenge is to drive the design changes through incentives that can stimulate rather than stifle innovation.

We can all support the stated goal in the *Green Paper on Integrated Product Policy*—"Products of the future shall use less resources, have lower impacts and risks to the environment and prevent waste generation already at the conception stage." One danger posed by IPP is that it rekindles the old debate on EPR and can be used to redefine EPR, to dilute it, or to circumvent it. It is important that "extended producer responsibility" be included as a key policy in any comprehensive IPP that is adopted.