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IMPROVING RECYCLING MARKETS

Nick Johnstone
Email: Nick.Johnstone@oecd.org

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FOREWORD

The OECD project on ‘Improving Recycling Markets’ has been concerned with the analysis of non-environmental market failures. As such, environmental externalities arising from waste generation (and recycling) are not the subject of inquiry, even if the environmental impacts associated with such externalities are the primary motivation for undertaking the study. Instead the project has focused on ‘classic’ industrial market failures and barriers such as information failures, technological externalities, market power, transaction costs, etc…. The distinction is important since in many cases environmental policies (explicitly designed to address environmental externalities) are introduced and assessed with the assumption that all other aspects of the market are functioning efficiently. As will be discussed, this may well not be the case.

No attempt is made to address whether levels of recycling are optimal or not for different materials in different OECD countries. This is beyond the scope of the project. In many cases they may be ‘too low’, and support for recycling markets may well lead to welfare improvements if the measures introduced are of reasonable efficiency. However, this is not a given and in some cases policy measures may be generating excessively high levels of recycling, imposing significant net welfare costs.

Whether recycling levels are ‘too high’ or ‘too low’, a case can be made for removal of non-environmental market failures (in addition to addressing environmental externalities). Even in the absence of environmental impacts associated with alternatives to recycling (i.e. incineration, landfilling, illegal disposal, waste prevention), the presence of non-environmental market failures imposes welfare costs and - if they can be addressed at reasonable cost through policy interventions - they should be addressed. If in doing so it is relatively less costly to meet given environmental objectives, this is all to the good.

This report brings together the five papers that have been prepared as contributions to the project on "Improving Recycling Markets". Chapter 1 provides an overview of market failures and barriers in markets for potentially recyclable materials. This is followed by three case studies: on used lubricating oils (Chapter 2), used plastic packaging (Chapter 3) and used rubber tyres (Chapter 4). The report concludes with a discussion of the public policy implications (Chapter 5).

EXECUTIVE SUMMARY
IMPROVING RECYLING MARKETS

EXECUTIVE SUMMARY

Markets for many recyclable materials are growing.

The growth of markets for many classes of potentially recyclable materials is due in part to policy incentives, but also more general commercial conditions. In many cases their development is supported directly by public authorities through measures such as collection schemes for recycled materials, deposit-refund systems, and public procurement schemes. Public authorities also provide indirect support for such markets through the internalisation of externalities at the waste management phase and upstream raw material extraction

However, market failures and barriers are constraining some markets.

The constraints for many potentially recyclable materials arise, in many cases, because such markets possess characteristics which undermine their efficiency. Factors such as information failures, technological externalities, and market power can affect the prices, quantity, and quality of materials traded. In addition, market barriers such as search and transaction costs can have an adverse effect on market development. Ultimately, such market failures and barriers can even undermine the market entirely.

Information failures related to secondary materials can be one important constraint on market development.

If sellers possess information about the characteristics of potentially recyclable materials which is not available to buyers (except at prohibitive cost), there are strong incentives for sellers to place low-quality waste on the market since they will not be penalised for doing so, at least in the short run or when reputation effects are not present. This can result in a downward spiral in the quality of recyclable materials placed on the market. For instance, where collection schemes are not well-monitored, this can create a problem of water contamination in used lubricating oils.

Initial perceptions (and misperceptions) concerning the quality of products made from recycled materials can be a problem.

Buyers may also be wary of entering the market because they do not have full information about the quality of the final product manufactured from recycled materials. In efficient markets such information is diffused effectively as market participants monitor the choices of other agents. However, for novel products - including those manufactured using recycled materials - there may be significant lags before this arises. Since initial buyers of such products are not rewarded for the information they generate for other market participants, take-up will initially be sub-optimal. This was thought to be a problem for recycled paper in former years, and continues to be a problem for retreaded tyres and other materials.

Technological externalities associated with product design can result in sub-optimal levels of recycling.

Similarly, if manufacturers are not rewarded in the market for designing products which are recyclable they are unlikely to do so even if it is in society's interest at large. Indeed, in many markets the increased complexity of product design and material use has driven up the cost of material recovery significantly. Clearly novel product design and material use brings benefits – otherwise, firms would not make such investments. However, in the absence of market signals which reflect the net benefits of recyclability, product design will be inefficient. Plastic packaging is an area in which such problems seem to be important.

Search and transaction costs can make it difficult for buyers and sellers to find each other and conclude a 'fair' transaction.

More generally, trade in recyclable materials can incur significant search and transaction costs. The markets are often diffuse, occasional, and in some cases include market participants with little market experience. Under such conditions it can be burdensome for buyers and sellers to find each other. Moreover, once they do so the effort expended to agree upon a 'fair' price may be considerable due to the heterogeneous and uncertain nature of the commodities being exchanged. While these costs may fall with time, they can be important barriers for a prolonged period. Amongst others, markets for some kinds of construction and demolition waste have high search and transaction costs.

The extent to which such failures and barriers exist varies widely according to the recyclable material in question.

While there is evidence for the existence of some market failures and barriers, there is also a need to look in detail at the functioning of individual markets. Indeed, many markets appear to be relatively efficient, while in other cases there may be one or more significant market failure or barrier constraining market development. 'Industrial' policies which address such market failures and barriers can be an important complement to more traditional environmental policies.

A range of effective policy measures have been developed that address specific problems.

Encouraging ever-higher recycling rates in an imperfect market may impose very high social welfare costs. In such cases it may be far less costly to address the imperfection within the market than to try and bring about increased recycling rates through increasingly ambitious recycling programmes. Relevant public policies for specific problems include:

- *Search costs*: To disseminate information to potential market participants (supply and demand), web exchanges to reduce costs of identification of market counterparts.
- *Transaction costs*: To develop standardised contracts, waste quality grading schemes for heterogeneous materials, establish dispute resolution mechanisms.
- *Information failure*: To introduce certification schemes, support for testing equipment, public procurement programmes, liability for product misrepresentation, establish dispute resolution mechanisms.
- *Consumption externalities*: To carry out demonstration projects, put in place public procurement programmes, disseminate information concerning product characteristics.
- *Technological externalities*: To implement extended producer responsibility, research and development on 'design-for-recycling', to develop product standards which incorporate impacts upon recyclability.
- *Market power*: To introduce and maintain general competition and anti-monopoly policy, market regulation of collection and processing which ensures competitive demand.

Such policies can effectively complement more traditional recycling policies.

The need for the use of policy mixes is emphasised. However, this mix relates not only to environmental policy, but also to market and industrial policy more generally. A thorough understanding of the markets and the means by which different policies interact with each and impact upon the market is key to the development of the right mix. This is particularly so if one recognises that a number of 'environmental' policies *(i.e*. extended producer responsibility') can impact significantly upon markets in which 'industrial' policies (i.e. market regulation) are also operational.

To purchase *Improving Recycling Markets*, and other OECD publications, visit the OECD Online Bookshop at http://www.oecd.org/bookshop or send an email to sales@oecd.org.

For more information contact : Nick Johnstone, Empirical Policy Analysis Unit, National Policies Division, Environment Directorate
Email – Nick.Johnstone@oecd.org

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TABLE OF CONTENTS

CHAPTER 1:

INTRODUCTION AND OVERVIEW OF MARKET FAILURES AND BARRIERS

by

Nick Johnstone and Soizick de Tilly

OECD Environment Directorate

1. Introduction

Markets for many classes of potentially recyclable materials are growing. This is due in part to policy incentives, but also more general commercial conditions. However, the markets for many potentially recyclable materials possess characteristics which are usually defined as evidence of market inefficiency. Factors such as information failures, technological externalities, and market power can affect the prices, quantity, and quality of materials traded. Ultimately, such market failures and barriers can even undermine the development of the market entirely.

As such, when designing environmental policies which have the objective of increasing recycling rates, it is important to have a good understanding of any potential inefficiencies which may exist in such markets, and how such inefficiencies may be overcome through public policy interventions. If such market conditions and policy options are not borne in mind the costs of meeting given environmental objectives may be significantly higher than is necessary.

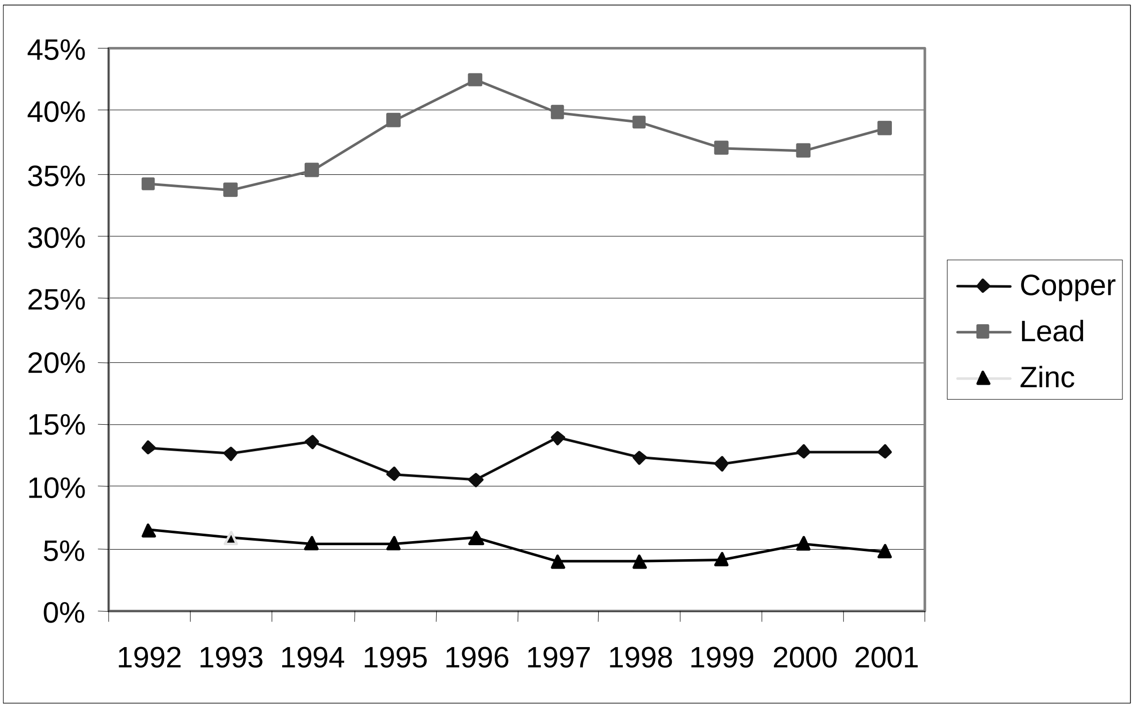
Before proceeding to a discussion of potential market failures in markets for secondary materials (Section III), this report briefly reviews the relative economic importance and the structure of the markets for recycling in selected OECD economies (Section II). In Section IV, we review the evidence on price volatility in secondary material markets since this is often (but by no means always) an indicator of inefficient markets. Section V provides a brief conclusion.

2. The Economic Importance and Structure of the Recycling Sector in OECD Countries

Recycling is an important economic sector in a number of OECD countries in terms of employment, turnover, and investments. However, since much recycling takes place within the firm (i.e. the reuse of ‘home’ scrap by the metal fabrication sectors) evaluating the economic importance of recycling can be problematic. In addition, in many sectors, post-consumption secondary material inputs may not pass through dedicated recovery facilities (i.e. use of used newsprint in production of pulp). The Bureau of Industrial Recycling has estimated that the 'recycling industry' employs more than 1.5 million people, with an annual turnover of $US 160 billion, and physical throughput greater than 500 million tonnes (www.bir.org).

Figure 1.1 gives the percentage of global production for selected non-ferrous metals which is based upon the use of secondary materials from 1992-2001. (UN DESA 2003) The figures are approximate since there are some breaks in the data for individual countries. However, in general it appears that with the possible exception of lead, the proportion is relatively stable.

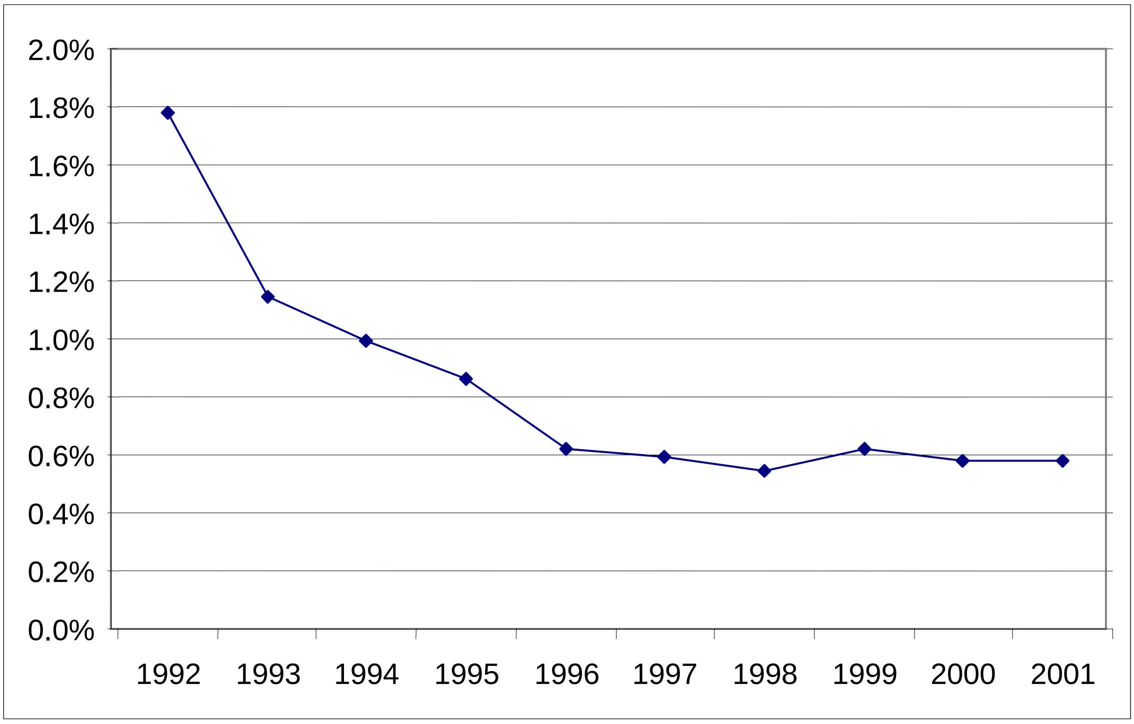
Figure 1.1: Approximate Percentage of Non-Ferrous Metal Production Based upon Use of Secondary Materials



Source : UN DESA 2003

Similar figures are not available for other commodities in which a mix of primary and secondary materials can be used – i.e. no distinction is made between primary and secondary production. For instance, no distinction is usually made in the case of pulp, glass, and ferrous metals, since a given product can be manufactured with a mixture of primary and secondary inputs. However, in the case of rubber, we can compare the proportions of synthetic and reclaimed rubber. Figure 1.2 shows a low overall percentage of reclaimed rubber production, with a marked decline in the early 1990's.

Figure 1.2: Approximate Percentage of Reclaimed Rubber Production in Total Rubber Production



Source: UN DESA 2003

Nonetheless, taking the United States, France and Japan as examples, this section will review the economic importance of the recycling sector in the economy.

United States

In the United States the waste collection[1] and material recovery[2] sectors involved approximately 9,000 firms in 1997 with the vast majority in waste collection. Total receipts for the sectors were $20 billion for waste collection and $1.3 billion for material recovery facilities (USDOC 2001). The establishments are generally very small, with waste collection firms being marginally larger than material recovery facilities. However, there are some large firms involved in waste collection and recovery, reflected in the fact that the four-firm concentration ratio is 45.0% for waste collection and 30.6% for material recovery.[3]

1. North American Industry Classification System (NAICS) Code 5621.

2. NAICS Code 56292.

4. Percentage of total national production accounted for by the four largest firms.

Table 1.1: The Structure of the U.S. Recycling Industry

	Waste Collection	Material Recycling
Number of Establishments	8234	765
Number of Employees	152744	10846
Employees per establishment	19	14
Total Receipts ($million)	20145	1299
of which from recyclable materials	427	1061
Concentration Ratios		
4-firm	45.0%	30.6%
8-firm	51.3%	37.2%

Source: USDOC COM Concentration Ratios in Manufacturing (2001)

It has been estimated that average throughput for material recycling facilities in 2000 was just under 62,000 tons per day (US EPA 2002). In the area of municipal solid waste generation, material recycling as a percentage of total municipal solid waste generation has risen from just 6.4% in 1960 to 30.1% in 2000.

France

In France there were approximately 3,600 establishments involved in material recovery and recycling in 2002, employing 28,800 people. This represented a decline from approximately 4,200 establishments in 1999, reflecting an on-going trend toward increased concentration in the sector. As in the United States the facilities are generally very small, with 81% of firms having less than 10 employees (FEDEREC 2003).

Total receipts in the sector in 2002 were almost EUR 7 billion, with annual investment of EUR 330 million. However, both of these figures were below figures for 2000 and 2001. Most of the receipts were in the area of non-ferrous metals (37%), ferrous metals (22%), and paper (12%). These areas also exhibited the fastest growth rates in recent years, while sales in some sectors (i.e. plastics and glass) actually fell.

In addition, to the relative increase in concentration in the sector, there has also been an increase in diversity within the sector with an increasing proportion of establishments involved in the recovery of more than one material. For instance 21% of establishments were involved in the recovery of at least three materials in 2002, while the figure in 1999 was only 10%.

Japan

In Japan the overall level of material recovery is 11% for municipal solid waste (MSW) and 41% for non-MSW. In the area of non-MSW, the highest levels are for metal scraps (75%), construction waste (70%), and slag (65%). Wastepaper has a recovery rate of 48%, while the figures for waste oil (27%), textiles (11%) and rubber (13%) are lower. For MSW the highest figures were steel cans (82.9%), aluminium cans (78.5%), glass bottles (78.6%) and paper (56.3%) in 1999. These figures have all been growing, with PET bottles showing by far the fastest growth rates over the last decade. Paper has shown the slowest rate of growth (OECD 2002).

In the area of recycling of metal and scrap, Table 1.2 gives figures for number of firms and total employment up to 2000. There was a moderate trend decrease in both variables in the late-1990s. Average number of employees by establishment was only in the region of 15 individuals, much smaller than in many other sectors. The sector has been given a large 'policy shock' in 2000 from the *Basic Law for Establishing a Recycling-Based Society* (http://www.env.go.jp/recycle/low-e.pdf) which was designed to help ease some of the pressures arising from decreasing landfill capacity. In particular, the obligatory recovery system for household appliances has resulted in a significant expansion of the sector, with the establishment of 38 additional recovery facilities.

Table 1.2: Employment and Establishments in the Recycling of Metal and Scrap in Japan

	Employment, total	Establishments
1997	10854	669
1998	10937	689
1999	9763	648
2000	10020	628

Source: OECD Structural Statistics for Industry and Services

Conclusion

The recycling sector is, therefore, not of negligible importance. It is, moreover, quite heterogeneous with recovery of some materials involving large enterprises with well-developed technologies, and others with smaller firms and less advanced technologies. In addition, the nature of the markets can differ widely, an issue to which we turn in the next section.

3. The Nature of Potential Market Imperfections in Secondary Material Markets

As noted above, in this section markets for secondary materials will be examined through the lens of the theory of market efficiency. Not all sources of market inefficiency which affect recycling rates are discussed. For instance, imperfect internalisation of externalities from raw material extraction will have a very significant influence on recycling rates, distorting markets in a manner which has negative consequences for material recovery. In addition, the imperfect internalisation of externalities from waste management such as leaching into groundwater, air pollution, etc... are not discussed. However, these issues - which are best described as policy failures - have been assessed in the literature (e.g. OECD 2005 and OECD 2004).

Instead the report focuses on 'classic' market failures in the recycling market itself. This includes issues such as information failures and market power. In addition, other market 'barriers' which are not strictly market failures (such as price volatility or risk aversion) are discussed. And finally, in an intermediate position are factors such as transaction costs which may or may not be market failures, depending on the context. In any event, in all cases such factors (if they are present) can affect the level of recycling, usually (but by no means always) negatively. The various imperfections to be explored are summarised in Table 1.3.)

Table 1.3: Potential Sources of Market Inefficiency

Causes of Market Inefficiency	Explanation
1. Transaction Costs in Secondary Material Markets	Arises from the diffuse and irregular nature of waste generation. May also arise from the heterogeneous nature of secondary materials.
2. Information Failures Related to Waste Quality	Arises from the difficulty of buyers to detect waste quality, and the relative ease with which sellers can conceal inferior quality waste.
3. Consumption Externalities and Risk Aversion	Perceived costs associated with the quality of final goods derived from secondary materials relative to those derived from virgin materials
4. Technological Externalities Related to Products	Complexity of recycling due to the technical characteristics of the recyclable material and products from which secondary materials are derived.
5. Market Power in Primary and Secondary Markets	Substitution between primary and recyclable materials may be restricted due to imperfect competition and strategic behaviour on the part of firms.

3.1. Transaction Costs and Search Costs in Secondary Material Markets

Transaction costs arise when there is ‘friction’ in the market – i.e. market transactions are not undertaken costlessly.[4] They can take many forms:

There may be costs associated with price discovery, whether due to lack of transparency or other factors;

There may be significant ‘search costs’ with buyers unable to identify potential sellers and sellers unable to identify potential buyers;

There may be extensive administrative costs associated with actually undertaking the transaction (i.e. due to permit requirements); and,

In still other cases, there may be significant negotiation and bargaining involved before the transaction is actually completed.[5]

However, to one extent or another all markets possess transaction costs. Do we have any reason to believe that transaction costs are likely to be particularly significant for recyclable materials? Unfortunately, identifying the presence of transaction costs is not as straightforward an exercise as one might suppose, since efforts to overcome them ‘cloud’ the picture, with transaction costs embedded in prices paid for other services (see below.) However, price variation be seen as indirect evidence of the relative importance of transaction costs (Williamson and Winter 1993.) In effect, in efficient markets, the ‘law of one price’ should hold, and if this is not found to be the case then this *may* be explained by transaction costs. For instance, the case study on plastics prepared by Ingham (see Chapter 3) shows single-period prices (DM/kg) for mixed German plastics, indicating significant variation. He finds much lower price variation for other plastic waste (such as HDPE) for which transaction costs are thought to be less important. In the market for low density polyethylene, Enviros (2003) also finds significant within-period price variations, with low and high prices differing by a factor of 10.

In the presence of transaction costs, a wedge is driven between the buyer’s price and the seller’s price, and gains from trade are lost. For Coase (1937) the alternative solution in the presence of transaction costs, was the administrative allocation of resources within the firm. However, in the presence of substitute goods for which transaction costs are lower, the solution may be altered purchasing decisions within the market. In the case of secondary materials which are substitutes for primary materials, this will mean that if transaction costs are more significant for secondary materials, they will enjoy much less market share than would otherwise be the case.

The implications can be seen in Figure 1.3. Sr and St are simplified representations of the supply curves for recycled and total materials,6 with the difference being the supply of virgin materials. With

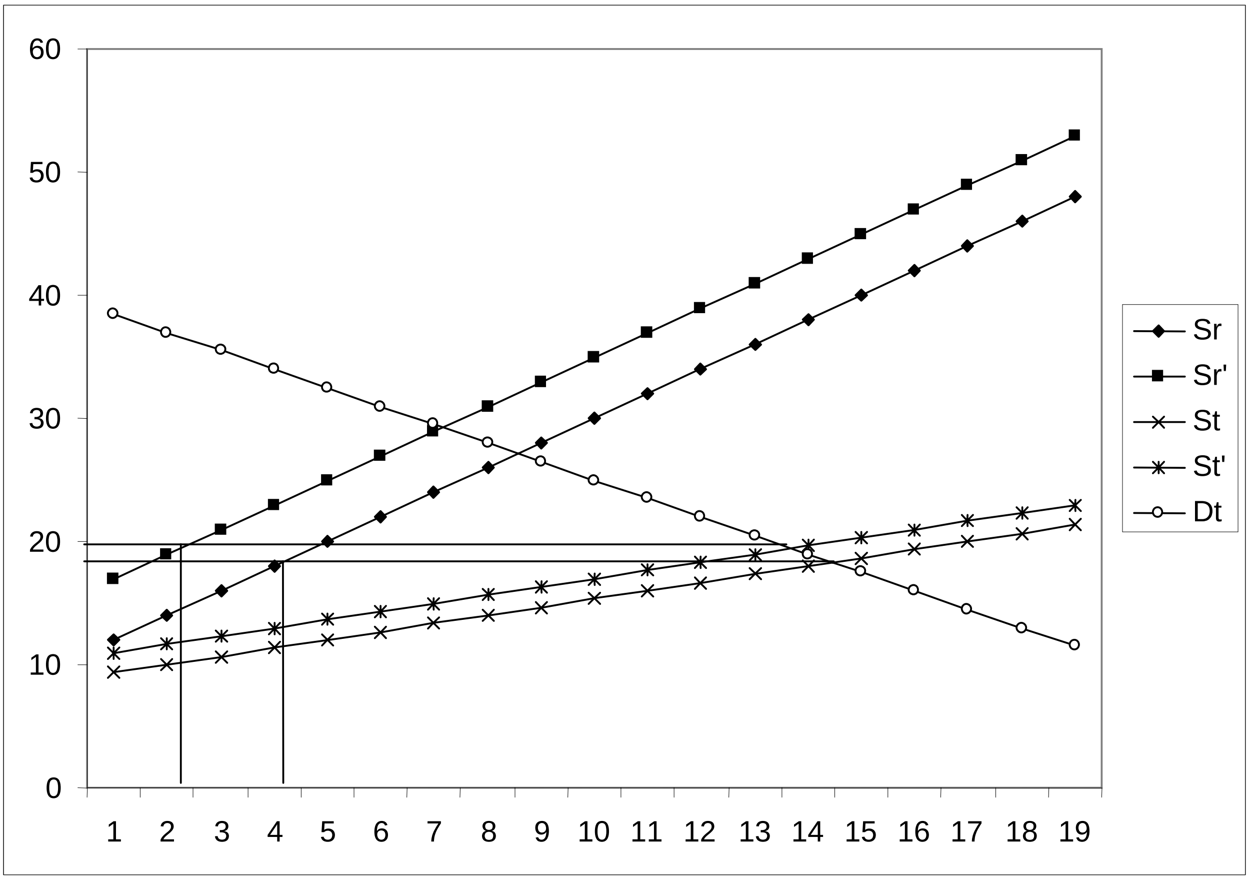
4. See Coase (1937) for the classic discussion. See also Williamson and Winter (1993).

5. To quote Coase (1937) - ‘If transactions are carried out through the market there are costs of discovering what the relevant prices are; there are costs of negotiating and completing separate contracts for each market transaction, and there are other costs besides.’

6. In reality supply curves for recycled materials are complex, not least because the main substitute for recycled materials are the very good which is used as the primary input in the supply of the recycled material. Since in most cases it is impossible to achieve 100% recycling rates, in a closed economy goods produced from primary materials need to be injected into the secondary material system. On the one hand, if the price of the product made from primary materials rises, this should increase the competitiveness of

transaction costs shifting the Sr curve up to Sr', the total supply curve shifts to St'. As noted above, potential gains from trade are lost. Moreover, activity within the market for secondary materials will be correspondingly less than would have been the case in the absence of transaction costs – there has been a shift from a level of recycling R to a level R'. At the same time, demand for virgin materials has increased. As such, if transaction costs are significant in markets for recyclable materials, the rate of recovery of recyclable materials will be lower.

Figure 1.3. The Effects of Transaction Costs on Recycling



The issue of price discovery costs has been raised again and again in the literature on recycling, with many (EC, 1998; and Sound Resources Management Group, 2000) arguing that there are many markets for which price information is not available. In such cases, considerable effort will have to be expended by buyers and sellers in determining an appropriate price. For instance, in a survey of recycled material sellers and purchasers in the United States, the prices provided in trade journals was not thought to be reliable, with the absence of information seen as a significant problem, particularly for buyers. (Sound Resources Management Group 2000, Tables 3 and 4).

The 'search costs' associated with identifying counterparts in the market may be particularly onerous for participants in secondary material markets. In comparison with a manufacturing firm producing specific commodities, production of recyclables is generally (but not always) much more 'ad hoc'. In particular, since 'production' is often spatially diffuse and temporally irregular, identifying counterparts may require considerable expenditure of effort. For instance, the generation of construction and demolition

that made from secondary materials. On the other hand, this will also reduce the supply of material inputs into production of the good, pushing up its price.

waste has these features due to the long-lived nature of the built environment. This is particularly true if the sector is dominated by small firms as is the case in many countries (see OECD, 2003). As such, if a firm wishes to identify a potential source of timber for construction, the costs of identifying a source for new materials are likely to be much less than for a source of used materials. The problem of high search costs might also exist in other areas, such as specialised metal scraps, low-volume plastic wastes, etc. While in all such cases search costs may reduce over time as markets mature, they may be significant constraints on the initial development of a market.

In addition, transaction costs associated with bargaining and negotiation can be considerable. For instance, negotiation costs can be high in markets for which products are heterogeneous, and for which the nature of the heterogeneity is not fully recognised by one or both of the parties or wherein there is potential for disagreement. This heterogeneity can relate to factors such as the degree of wear for some kinds of metal scrap, the precise mix of different plastics within a single shipment, the degree of contamination in waste oil, etc…. Even if both parties to the exchange were fully aware of the nature of the commodity being traded, this can lead to negotiation and bargaining costs since each transaction is, in effect, unique. As such, each transaction involves a degree of learning on both parties part, and perhaps some degree of strategic behaviour as buyers and sellers seek to gain advantage.

One possible internal market solution to such problems is the use of long-term contracts. This has been common in a number of secondary material sectors such as used paper. This can obviate negotiation costs to some extent. Alternatively, intermediaries are likely to arise in the face of such problems, serving as brokers between buyers and sellers who only enter into the market irregularly.[7] However, this does not obviate the problem of search costs entirely. Rather it is just a reflection of the fact that there are ‘economies of scale in search (and perhaps negotiation)’ and that these are significant enough to warrant the development of a specialised market participant.

While experience (on both sides) can reduce the costs associated with fixing a price, as noted above, the costs of doing so will always be greater than for homogeneous products for which the price can be taken as given by both the buyer and seller. Indeed, inadequate access to reliable price indicators for some types of recyclable materials is seen as a significant problem in a number of secondary material markets (EC 1998). However, these problems may diminish in time, even for some of the more ‘problematic’ waste streams. For instance, in the area of textiles the “Secondary Fiber Exchange” (www.sefex.com) provides data on prices by grade of secondary textiles. Many public authorities disseminate information on market prices in order to facilitate market transactions. (See, for example, California Integrated Waste Management Board (CIWMB, 1996).

With respect to product heterogeneity, the role of grading specifications is key (Ecotec 2001). The ISRI has played a key role in the development of specifications for wastepaper and metal scrap, and is starting to play a more prominent role in other areas. For instance, in the area of wastepaper there are 46 regular grades and 33 speciality grades. (See ISRI 2003 for a list of specifications of different waste grades.) While efforts have been made, there seems to be a clear need for improved grading in the areas of glass and plastics. (SRMG 2000, p. 52). While most of the schemes have been private market responses, the support of public agencies has been significant in some areas (i.e. crumb rubber). Since standards can serve as a reference point for use in negotiations and trading, this can reduce transaction costs. For instance, in the area of wastepaper a report prepared by the Waste and Resources Action Programme (WRAP, 2003) indicates that the British Standard (BS EN 643:2001) has been played a role in reducing transaction costs.

7. Although the presence of brokers can also be explained by other factors. See below.

One recent phenomenon has been the growth of web-exchanges in the area of secondary materials. One well-known example - the Chicago Board of Trade Recyclables Exchange (CBTRE) - received considerable support from the EPA (see SRMG 2000). In addition, Symonds et al. (1999) list a number of web-based exchanges that have been actively supported by different European governments. Their growth can be explained by the belief that in the presence of falling information costs, it is possible to overcome problems associated with search costs by centralising a market through the web. In addition, the problem of spatial diffusion and temporal infrequency which can result in 'thin' markets may be partially overcome.[8]

However, one of the few evaluations that has been undertaken of such an exchange (the aforementioned CBTRE) has found that they have not overcome problems within the market. It was only in well-developed markets (paper and ferrous metal) that transactions were widespread. In other markets, web-based exchanges may have reduced search costs, they have not obviated problems associated with the heterogeneity of the materials traded. In effect, in some (but by no means all) secondary material markets every shipment is distinct, possessing characteristics which are different from other shipments within the same 'market'.

What else might public authorities do to reduce 'friction' in the market? In the area of search costs, many governments have sought to 'match supply and demand'. For instance, the CWIMB (1996) has a number of programmes in place to help buyers and sellers identify each other. The Danish municipality of Kalundborg has pursued a very successful programme of this type (Ehrenfed and Gertler 1997). This includes the publication and dissemination of lists of plastic resin collectors and reprocessors to plastic manufacturers.

3.2. Information Failure and Uncertainty Related to Waste Quality

Full information is vital to the successful functioning of all markets. As we have seen above, inadequate information concerning prices can increase transaction costs. However, for recyclable materials in particular there may be another source of information failure related to the quality of information concerning the characteristics of secondary materials. In many cases this is either inadequate, unevenly distributed or altogether missing. Such information failures can have additional consequences, beyond the increase in transaction costs and search costs.

For instance, if information is unevenly distributed between demanders and suppliers - i.e. there is asymmetric information - adverse selection may occur. Akerlof (1970) provides the classic example, in his analysis of the used car market, describing the interplay between uncertainty and differentiated quality in the presence of asymmetric information. Assuming that suppliers are divided between those who offer low quality ("lemons") and high quality used cars respectively, adverse selection favours the sellers of "lemons", who will exploit their information advantage over the buyers. Sellers will, therefore have incentives to put low quality products on the market. Analogously sellers with high-quality products will not be able to attain a high price.

What implications does this have for secondary material markets? In this case 'quality' may mean the presence of contaminants in used waste oils, the structural strength of scrap, high water content in used lubricating oil, etc. All of these attributes will reduce the value of the material to the buyer. Most drastically, the presence of polychlorinated biphenyls (PCB's) in waste oils would transform something with a positive price due to its energy content or potential for material recovery, into something with a

8 . Examples include: http://www.cbot-recycle.com; http://www.ciwmb.ca.gov/calmax; http://www.efibre.com; http://www.recycle.net/recycle; http://www.scrapsite.net; and, http://www.sec-mat.com.

significant cost, since it would have to be treated as hazardous waste by the 'buyer'. Symonds et al (1999) believe this problem affects the construction and demolition waste sector, and Adant and Gaspart (2003) also cite the example of hazardous waste in steel scrap. More generally, the latter study provides empirical evidence on the discrepancy between assessed and actual quality of scrap.

However, if the buyer does not have access to the information which would be necessary to evaluate the 'quality' of the waste, the seller may have incentives to exploit this fact by putting low quality waste on the market. There will be a bias toward the sale of wastes which are more contaminated, structurally deficient, or which possess other negative attributes. Ultimately the market for secondary materials may be undermined altogether. As Papineshi (2003) points out with respect to the plastics sector, even a small number of unscrupulous traders can have significant negative consequences on the market as a whole.

In addition, the uncertainty associated with waste quality may mean that the waste is not directed toward its highest potential value-added use, even if it is 'clean'. In a review of the wastepaper sector in the United Kingdom, WRAP (2003) found that 25% of "mixed paper" was graded at too low a level, resulting in significant missed opportunities for revenue generation. And finally, the uncertainty associated with waste quality can have important consequences which are distinct from impacts associated with the quality per se. For instance, in the area of plastics (Enviros 2003) and glass (Enviros 2002) it has been found that material recovery facilities incur significant 'retooling' costs if the waste is of variable quality. Fitzsimons (2004) makes the same point in his review of lubricating oil. The likelihood of suppliers adopting a strategy of placing lower-quality materials dependent upon a number of factors, including:[9]

- The cost of concealing negative attributes of the secondary material;
- The cost of mitigating these negative attributes;
- The cost for buyers of detecting the negative attributes; and,
- The costs which arise ex post if the negative attributes are detected.

For instance, if it is relatively inexpensive for a buyer to test for the presence of contaminants in a given waste stream, the seller will have little incentive to try and conceal their presence. Similarly, if it is relatively inexpensive to remove the negative attributes then such measures are likely to be undertaken. And finally, it has been pointed out in other contexts that the 'lemons' problem is likely to be much more important for markets in which transactions are infrequent, and thus for which reputation effects are less important. Since upstream 'sellers' of some recyclable materials may only enter the market very infrequently, such reputation effects may be very weak.

As such, a market in which costs of concealment are low and costs of detection are high, and for which the implications of detection are insignificant is more likely to be subject to problems of adverse selection. This problem is likely to be particularly significant for wastes where materials from different sources are aggregated at the point of collection (i.e. used motor oil, some plastics, construction and demolition waste, etc….). Identification of specific sources of relatively low-quality waste is difficult. A study (RDC/PIRA 2003) conducted for the European Commission found that contamination was found to be a significant problem in glass, plastics and paper/board markets, and somewhat less of a problem for metals. There are many documented cases in which contaminated wastes have been sold on secondary markets for recovery. Clearly, in some cases the financial incentives are such as to encourage sellers to put 'lemons' (of whatever kind) onto the market.

9. See Kerton and Bodell (1995).

There are ways in which the market itself can seek to address such problems. Most obviously, significant tests of quality may be undertaken. In a review of the plastics sector Papineshi (2003) argues that on-site testing equipment is a pre-requisite for involvement in the plastics recycling sector. Unfortunately in many cases this is impracticable for technological or financial reasons. It is not financially viable for a used oil collector to invest in testing equipment for all of the potential impurities and contaminants that may be present (Fitzsimons 2004).

The only solution may be to rely solely upon sources of waste over which the firm has direct control, such as waste arising from the production process itself. In the case of glass cullet used for flat glass manufacturing this strategy has been adopted in many cases (Enviros 2002). As an alternative, firms involved in material recovery may integrate vertically back to the collection stage, thus taking responsibility for this stage of production and giving them greater control over the quality of inputs purchased. This strategy has been pursued in some cases in the plastics (Enviros 2003) and aluminium markets (Ecotec 2000). Even if vertical integration is not adopted, it may be possible to control quality by other means. In the construction and demolition waste sector, some buyers have responded to this problem by controlling the demolition process (Symonds et al. 1999).

The use of long-term supply contracts which reduce the sellers' incentives to misrepresent the materials being sold may also be adopted, as in plastics market (Enviros 2003) and elsewhere. This increases the cost of misrepresentation for sellers. Indeed, in the steel scrap study cited above (Adant and Gaspart 2003), it appears that the market was 'segmented' between ocassional sellers (who had fewer potential adverse reputation effects from selling 'lemons') and more regular and frequent sellers (for whom such effects were likely to be significant).

More broadly, within the market brokers who have specialised skills and economies of scale in identifying low-quality waste may enter the market. (See Wimmer and Chezum 2003 for examples from other markets). More fundamentally, they may take on the task of accepting risk for low-quality inputs, thus shifting the risk away from buyers. Since they are better able to ascertain the true quality of wastes they are able to reduce their risks relative to other market participants. These strategies are likely to be prevalent in markets when there are economies of scale and specialised skills associated with detection.[10]

Interestingly, the web-based exchanges discussed above – while reducing search costs - may exacerbate information asymmetries since much more is taken on faith. (See Huston and Spencer 2001 for a discussion related to coins.) This is particularly true for exchanges in which trade is bilateral, and the site merely serves as a point of contact. This can be overcome either through the provision of brokerage services in which guarantees are provided in the case of misrepresentation, or if a 'dispute settlement mechanism' is introduced. In the aforementioned Chicago exchange, the absence of the introduction of a dispute settlement mechanism was thought to be one of the main shortcomings (see SRMG 2000).

Some of the web-based exchanges allow buyers to post comments on the characteristics of wastes sold by particular sellers. This will, of course, increase the costs associated with detection, thus making adverse selection less likely. Even more radically, the ISRI has a transaction dispute settlement mechanism to deal with cases of misrepresentation (SRMG 2000, p.55). However, it has been used very rarely since the costs associated with pursuing a dispute are often out of proportion to the sums involved in the individual transactions.

However, there may also be other roles for public authorities in addressing information asymmetries. In effect, these relate to the need to penalise misrepresentation by sellers in secondary material markets,

10. In the words of Stigler: 'Some forms of economic organisation exist chiefly as a device for eliminating uncertainties in quality'.

and is thus a general theme addressed in industrial policy. The development of material testing protocols is one area in which governments have sought to overcome some of these problems (SRMG 2000, p. 54).[11]

More significantly, strict application of ex post liability regimes will increase the cost of detection for sellers and thus may be an effective deterrent. However, in most cases while the 'quality' of the waste may be inferior, it is unlikely to be sufficiently unambiguous to warrant launching suits of this kind. For this reason, public support for transaction dispute settlement mechanisms of the sort described may be preferable. In the case of plastics, Enviros (2003) cites the importance of agreements to renegotiate in the event of discrepancies between assessed and actual waste quality.

3.3. Consumption Externalities Related to Products Derived from Secondary Materials

The previous section examined information failures which may relate to sources of secondary materials at the production input stage. However, there may also be information failures and commercial disincentives to the use of secondary materials directly in final products. These failures relate to cases in which buyers have incomplete information about the suitability of a given waste for a particular use, i.e. the product in question contains characteristics related to its use that are not properly understood. Unlike the case described in Section III.2 above in which there are information failures due to nature of the material, this type of information failure stems from a lack of knowledge concerning its potential use.

In such cases, the buyer may not be making purchasing decisions in his or her own interest (Kreps 1999). The relative importance of these issues for different types of recyclable materials depends upon the ultimate use to which the material is put. Paper manufactured from recycled newsprint is not a perfect substitute for paper derived from virgin pulpwood for various high-end uses. However, for low-end uses, it may well be. Similarly, many types of metal manufactured from scrap are perfect technical substitutes for metals manufactured from virgin ore, but for specialised sub-segments of the market this is not the case.[12]

However, buyers may not be aware of the degree of substitutability between products derived from secondary materials and those derived from primary materials. One oft-cited example is the case of recycled paper, which initially was perceived to be of lower quality for numerous uses for which it was perfectly appropriate. Such information failures may constitute barriers to market penetration, resulting in sub-optimal levels of recycling. Thus, information barriers may negatively affect the adoption of a new technology based upon the use of secondary materials. Initial purchases of recycled products for use in particular applications may be lower than is optimal. Immaturity in use can be expected to create uncertainty about applications and suitability, which may delay or deter investment in machinery.

Such problems are likely to become less important in due course. Through learning, agents in more developed market are able to apply available information more efficiently. As such, hopefully these failures become less important as participants within the market become aware of the true attributes of products derived from different material sources. However, in extreme cases this can undermine long-run market potential for secondary materials, particularly if supply-side factors such as important economies of scale make initial market penetration costly.

11. Interestingly, Adant and Gaspart (2003) argue that lack of information concerning waste quality may reduce environmental impacts associated with the steel scrap sector since it encourages buyers to remove all potentially-recoverable waste from the 'environment' and not just high-quality waste. This depends upon whether the market remains viable even if waste of mixed quality is purchased.

12. Duchin and Lange (1998) provide some examples of degree of substitution possibilities in the area of plastics.

Inaccurate perceptions of the ‘true’ nature of a commodity can arise from imperfect internalisation of consumption externalities. In particular, even if buyers do not learn from the consequences of their own purchase decisions, there may be consumption externalities as information about the suitability of use is diffused throughout the economy. In effect, each potential buyer learns from the experience of other potential buyers. “The extent to which the adoption of the technology by some users is itself an important mode of information transfer to other parties, adoption creates a positive externality and is therefore likely to proceed at a socially sub-optimal rate”[13] (Jaffe et al. 2000). In effect initial buyers generate information which is of value to others, but because this value is not rewarded in the market early adoption may be lower than that which is socially efficient. This is one of the primary motivations for demonstration projects. By ‘demonstrating’ the consequences of a particular investment or purchase decision, the public authority reveals valuable information to other market participants.

Thus, the slow ‘take-up’ in the market of some products derived from secondary materials may be attributable in part to the imperfect internalisation of such consumption externalities. In a recent report (ECOTEC, 2000) it has been argued that these problems are significant for plastics, but also perhaps for paper and glass. Both Reid (2003) and Symonds et al (1999) feel that this has had important impacts on the construction and demolition waste sector.

Such problems can be exacerbated by the improper choice of terminology as well as standards based on inadequate quality criteria. At its most extreme, the very term “waste” has a negative connotation in that it causes cognitive association with risk of inferior quality. Consequently regulation, which requires recyclable materials to be referred to as “waste” may cause unnecessary market imperfections. These costs can take the form of increased administrative requirements, as well as negative attitudes among buyers, insurance and credit institutions (EC 1998).

While it is common to blame consumer ‘misperceptions’ for low levels of market penetration for products which are ‘approximate’ substitutes for goods derived from primary materials it is important to recognise that even a small degree of perceived uncertainty concerning the relative quality of some types of secondary materials can have significant consequences on a market. Thus, even if the probability of the product being inferior is relatively low, there may be very little demand.

This is best understood with reference to a particular form of risk aversion, known as ‘disappointment aversion’. (See Gul 1991 and Grant et al. 2000.) In such cases, buyers will ‘overweight’ low-probability risks of quality inferiority. There are many reasons that such behaviour might arise, and in many cases they are an appropriate reflection of underlying preferences. They are, therefore, best characterised as market barriers rather than market failures per se.

However, one which seems particularly pertinent is related to observed tendencies of economic agents to prefer the ‘status quo’. Known as reference effects, ‘it has been widely observed that a decision-maker will make significant economic trade-offs to remove the possibility of a net loss on a transaction’ (Bell 1985). For instance, even if the cost premium for the use of ‘virgin’ lubricating oil is non-negligible relative to the use of ‘re-refined’ lubricating oil, buyers may shun the latter if it is thought to increase the probability of engine failure even slightly. (See Chapter 3 for a discussion of the issues related to plastics.)

Similarly, in the case of retreaded versus new tires, if a household believes that that there is only a marginally increased probability of a retreaded tire causing a blow-out relative to a new tire, this might have disproportionate effects on demand. If potential losses (eg. a blow-out) and potential gains (eg. cost

13. Pharmaceutical products may serve as an example. Individual purchase decisions of pharmaceuticals are often derived from the extent that other consumers use the same drug. The overall consumption turns an indicator of product efficacy, safety and “acceptability” (Berndt et al., 2000, p. 1).

savings) are weighted differently even if the apparent cost is the same, then this can have very important implications for demand (see Annex 1 for an example).

Such effects are thought to be particularly important if the potential outcomes are 'catastrophic' and/or irreversible. (See Chichilnisky 2002 for a discussion related to climate change.) For instance, if a blow-out increases the probability of loss of life for family members, even a small perceived probability of inferiority may have very significant effects on the market. Other examples, might relate to increased potential legal liability for developers or manufacturers who use secondary materials in construction or production. Work in the area of 'experimental economics' has indicated that households do tend to 'overweight' low-probability risks, particularly if the outcomes are potentially catastrophic.[14] (See Allais 1984 for the initial evidence in this area.)[15]

It is, therefore, important for public authorities to recognise that preferences which appear not to be in the interests of the buyers themselves may in fact just be a reflection of normal levels of risk aversion evidenced elsewhere. If there are consumption externalities (and not just risk aversion), efforts should be made to transmit the relevant information consistently and repeatedly. However, perfect substitution is the exception and not the rule in secondary material markets, and thus efforts to make consumers indifferent between products derived from secondary and primary materials are likely to run up against the rock of human behaviour.

One possible solution is the use of performance standards which are generic across products, whether manufactured from primary or secondary materials. This can be justified in cases where the origin of the material used is extraneous information, which does not help consumers in making 'informed' decisions concerning the performance implications of the product. In the area of construction and demolition waste there has been a long-standing debate as to whether or not standards should be generic across products, irrespective of the origin (virgin or secondary) of the materials from which they are manufactured. While some countries (Austria, Denmark, Germany, the Netherlands) have favoured specific standards for products manufactured from construction and demolition waste, others (France, Italy, Portugal, and Finland) have favoured generic performance standards (Symonds et al 1999). The new European Construction Products Directive is generally consistent with the latter view (Collins and Nixon 2003).

Even if the standard appears to be generic and performance-related there may be subtle biases favouring the use of primary materials if the standards were originally designed with virgin material manufacturing in mind. Papineshi (2003) cites examples in the plastics area related to plastic bags, pallets and fencing, where products derived from secondary materials could not meet existing performance standards even if they were 'superior' in many senses.

Interestingly, the presence of negative perceptions associated with products derived from secondary materials indicates that in the case of goods targeted at final consumers, the use of 'eco-labels' which refer to recycled content may have perverse effects on the market, even for consumers which attach value to the preservation of the environment. If the 'negative' effects associated with perceived risk outweigh the 'positive' effects related to environmental preferences, demand may decrease. Thus, it is important to bear in mind the sometimes conflicting preferences of consumers.

14. 'Overweighting' in this case means that households attach more importance to such an outcome than would be implied by its 'certainty equivalent' – i.e. probability of the outcome occurring times the apparent cost of the outcome.

15. Paradoxically, low prices for products derived from secondary materials may exacerbate this problem, further undermining market potential by sending adverse 'quality' signals.

In order to overcome consumption externalities, demonstration projects can play an important role in fostering demand. In the area of construction and demolition waste Symonds et al (1999) cite policies in place in the United Kingdom, the Netherlands and Denmark. In addition, public procurement policies can play a potentially decisive role in diffusing information about the suitability of particular products manufactured using secondary materials. Unfortunately, there are examples in which public procurement has done precisely the opposite – reinforcing negative perceptions which are not borne out by technical studies. For instance, some schoolboards mandate the use of new tires on school buses. (See van Beukering 2003 and Gildea Research Center 1993.)

3.4. Technological Externalities Related to Recovery and Reuse of Secondary Materials

A technological externality exists when the production function of one agent enters another agent's production or utility function, without the latter being compensated (Kolstad 2000). Environmental externalities (when they affect productive processes such as polluted irrigation water) are, of course, specific examples of the more general case of technological externalities. However, they are by no means the only type of technological externality. In this section, the focus of the discussion is on "non-environmental" technological externalities which tend to reduce recycling rates.

In the area of waste, technological externalities would arise when one firm manufactures a product in such a way which increases the cost of recycling for the downstream processor, but for institutional reasons there is no means by which the potential waste recovery facility can provide the manufacturer with the incentives to change their product design (Porter 2002 and Calcott and Walls 2000). An illustrative list of areas in which technological externalities might exist would include the following:

The use of multi-layer plastics for food packaging which is incompatible with mechanical recycling (Wilson 2002).

The use of a wide variety of colours of glass bottles, resulting in costly separation for recovery (Ecotec 2002: Porter 2002).

The use of inking technologies in paper printing, which requires new technical solutions in paper recycling (Apotheker, 1993).

- Metal applications such as pigments in paints and as minor constituents in alloys, which make materials uneconomical to recycle (ICME, 1996).
- The use of a wide variety of resins (Sterner and Wahlber 1997) and blow-moulding vs. injection-moulding of plastics (Palmer and Walls 1999).
- The use of polymers in cable manufacturing which has reduced the potential for recovery of PVC (Enviros 2003).

All such product characteristics clearly provide benefits in the 'primary' market, or they would not be undertaken. For instance, the use of complex plastic mixes in food packaging which reduce recyclability is used to ensure minimum hygiene standards and reduction of food wastes. Thus, whether or not the examples of technological externalities cited above have adverse impacts on social welfare is dependent upon whether or not the benefits conferred by such designs are greater or less than the costs. Stated differently, would the firms have designed their products in such a way if they could capture the benefits of increased recyclability themselves? As such, in order to identify the existence of a technological externality it is not sufficient to identify attributes of products which restrict recyclability. Many characteristics of products restrict recycling, and even if there were no missing markets firms, would continue to design them in such a way.

The externalities will be internalised if subject to a market transaction between those who are deriving benefits and those who are bearing the costs (Kolstad, 2000). For instance, in cases where the manufacturer is linked with the recycler by commercial transactions – as for many types of industrial waste - the potential externality is internalised within the market. Similarly, the problem is likely to be obviated in cases where the waste stream exists within vertically-integrated industrial sectors. In this case internalisation is within the firm itself. In both cases the manufacturer will have incentives to change product design.

This is an area in which the distinction between the environmental externality and the non-environmental externality is particularly subtle. The sub-optimal level of recycling due to technological externalities has nothing to do with any loss of environmental benefits associated with recycling, but rather with the economic costs of not recycling. As such, the externality should be addressed, irrespective of environmental concerns. Of course, the level of recycling would be greater still if efforts to internalise associated environmental externalities favoured recycling over other strategies, but the two failures are quite distinct. Indeed, environmental internalisation may result in lower levels of recycling (if for instance incineration is a more efficient response from the environmental perspective), with technological internalisation pushing in the opposite direction.

There are good reasons to believe that technological externalities arising from the recyclability of product design are endemic in some waste streams due to the prevalence of ‘missing markets’. For instance, for municipal solid waste streams it is only rarely that manufacturers bear the cost of producing goods which can not be easily recycled. In practice, overcoming the joint presence of environmental externalities and ‘missing markets’ for product attributes is often achieved through the application of single instruments which seek to deal with both problems simultaneously.

Products which are subject to deposit-refund schemes or other forms of extended producer responsibility would be one such example. Indeed, the presence of technological externalities was often the (usually unstated) logic behind product take-back schemes. An alternative is to provide support for ‘design-for-environment’ such as through subsidies for research and development in product designs which are more recyclable. However, this does not address the fundamental problem of missing markets. Incentives have been changed, but in a rather crude manner, which requires large amounts of information for the public authority concerning alternative design options. Nonetheless, for markets which are relatively immature, ‘design for environment’ policies can play an important role in helping to overcome barriers to the market penetration of recycled materials.

Worse still would be support for recycling technologies which reduce the downstream impacts of technological externalities at the recycling stage, without simultaneously affecting the upstream incentives to internalise such externalities. Implicitly, this indirectly provides incentives for the design of products which possess technological externalities. The public authority may find itself ‘chasing its own tail’, providing increased levels of public support for recycling technologies to address technological externalities, generated by firms who face no incentive to design their products for recyclability. Thus, for example, if significant resources are devoted toward the development of sorting technologies to allow for the recycling of complex mixed wastes, product designers and manufactures will be discouraged for redesigning their products even if the net social costs would be less.

A final policy implication in this area relates to the formulation of product standard in light of potentially conflicting public policy objectives. For instance, in areas in which product standards are provided for other reasons related to the public good (i.e. hygiene standards for food packaging), it is important to balance the two objectives (environmental and health) appropriately. The examples cited above concerning the performance standards of certain products derived from construction and demolition waste are relevant in this regard (Papineshi 2003).

3.5. Market Power and Vertical Integration in Waste Recovery

One of the few areas in which there has been considerable work on the effects of market failures on recycling markets relates to market power. In effect, it is argued by some that dominance of the market by a limited number of firms in the production of primary materials has restricted recycling markets by giving primary material producers the power to influence markets in such a way as to undercut more competitive producers of secondary materials. In general it is found that economies of scale are few in collection, and of moderate significance in reprocessing (see Beede and Bloom 1995 for a review of some of the evidence), and as such market power is unlikely to be important.

Is there market power in those sectors where primary and secondary materials are at least potential substitutes? American data on the degree of market concentration in different manufacturing sectors indicate that for at least some sectors (particularly tire manufacturing, but also aluminium and newsprint mills), there is a high degree of concentration. (See Table 1.4.) Interestingly, in the case of tires, the tire retreading sector has four-firm concentration ratios[16] of 25.5 and 36.5 and a Hirschmann-Herfindahl Index[17] of 260.6 – i.e. it is much less concentrated than tire manufacturing from primary rubber. (USDOC, COM 2002).

Unfortunately, there is no recent database of comparable market concentration ratios across OECD countries. However, van Ark and Monnikhof (1996) provide figures for 1990, using the Schmalansee (1977) methodology.[18] Relative to median figures for manufacturing, it would appear that of the primary material sectors rubber, iron and steel and non-ferrous metals are the sectors for which degree of concentration are relatively important. (See Table 1.5.) However, it must be emphasised that the values are not such as to indicate particularly high degrees of concentration.

Table 1.4: Concentration Ratios in US Primary Material Sectors

	4 Largest	8 Largest	HH Index for 50 Largest[19]
Pulp, Paper and Paperboard Mills	28	45.7	356
Newsprint Mills	43.9	68.2	766
Plastics	3.9	6.9	14.1
Plastics Bottle	32.7	51.8	425
Rubber	36.8	46.4	444.8
Tire Manufacturing	72.4	90.8	1690.3
Primary Metal Manufacturing	13.8	22.3	97.4
Iron and Steel Mills	31.9	51.4	424.7
Alumina	48	64	816.3
NFM	24.3	34.7	242.6
Textile Product Mills	11.2	17.9	69.6

Source: USDOC COM – Concentration Ratios in Manufacturing (2001)

16. The percentage of national production represented by the four largest firms.

17. The Herfindahl-Hirschmann Index measures market concentration by adding the squares of the market share of all firms in the industry. In general, a HH Index in excess of 1,800 indicates a very high degree of market concentration. On this basis tire manufacturing is the only sub-sector for which the index is alarming.

18. Schmalensee (1977) suggested a variant of the HHI index. Instead of having a constant firm size within each size category, it can be assumed that firm size varies linearly within each class.

19. The Hirschman-Herfindahl Index for the 50 largest firms.

Table 1.5: Concentration by Sector in Five OECD Countries (Schmalansee Measures)

ISIC Code	Manufacturing Sector	United States (1987)	Japan (1990)	Germany (1990)	France (1990)	UK (1990)
311	Food	3	2	21	12	19
313	Beverages	31	30	33	154	114
314	Tobacco	823	313	1622	0	2088
321	Textiles	8	3	27	25	17
322	Wearing apparel	3	1	24	17	23
323	Leather and products	43	11	119	144	74
324	Footwear	54	29	190	198	215
331	Wood products	3	3	25	23	17
332	Furniture and fixtures	8	6	30	25	20
341	**Paper products**	**11**	**27**	**70**	**75**	**33**
342	Printing and publishing	4	17	20	17	19
351	Industrial chemicals	19	38	121	165	75
352	Other chemicals	16	27	106	45	89
353	Petroleum refineries	107	288	960	1712	1422
354	Petroleum and coal products	15	104	0	4144	764
355	**Rubber products**	**42**	**54**	**242**	**396**	**195**
356	**Plastic products nec**	**4**	**6**	**22**	**28**	**16**
361	Pottery and china	138	47	260	289	163
362	Glass products	41	116	198	220	258
369	Non metallic products nec	6	4	23	58	47
371	**Iron and steel**	**40**	**42**	**106**	**193**	**195**
372	**Non-ferrous metals**	**22**	**43**	**152**	**260**	**92**
381	Metal products	3	3	10	8	10
382	Machinery nec	6	8	11	17	11
383	Electrical machinery	7	8	27	68	22
384	Transport equipment	21	26	73	92	66
385	Professional goods	16	34	64	77	54
Unweighted Means		**124**	**117**	**234**	**373**	**289**
Median		**16**	**27**	**67**	**76**	**70**

Source: van Ark and Monnikhof (1996)

A well-known example of purported anti-competitive behaviour in a sector for which recycling is potentially significant is the American aluminium industry in which an anti-trust case was brought against Alcoa in the 1960s. The case revolved around deciding whether or not the presence of a secondary market in aluminium scrap served to undermine Alcoa's dominant position in the market. However, the case spawned a rich theoretical and empirical literature on the links between primary and secondary material markets in concentrated markets, such as other metals and pulp and paper (Grant 1999, Martin 1982, Swan 1980 and Suslow 1986).

How can such market power manifest itself? In the area of pulp and paper it has been argued that vertical integration between the forestry sector (and thus pulpwood production) and pulp and paper production, served to restrict market penetration of used newsprint in pulping (See EPA 1993.) In the area of aluminium, it was argued by the Department of Justice lawyers that a price-making aluminium manufacturer had been able to exercise control over the secondary material market (see Grant 1999 for a discussion of the case.) In the area of used motor oil, it has been alleged that manufacturers of virgin lubricating oil use their influence over downstream supply chains to discourage 'switching' to re-refined oils (Fitzsimons 2003.) Other examples might include the UK glass industry (ENVIRO/RIS 1999) and the primary lead sector (Sigman 1995).

However, it must be recognised that there is little empirical evidence that the exercise of market power by primary material producers has suppressed markets for secondary materials. Moreover, there are also arguments to support the view that market power in primary material markets may serve to increase the use of secondary materials. If firms are able to 'segment' markets between primary and secondary materials, and price discriminate between the two markets, under plausible assumptions recycling rates will be higher than in the absence of market segmentation.

For instance, in the case of motor vehicle tires if firms are able to segment the market between buyers of re-treaded and new tires, they may be able to maximise profits by exploiting differences in price elasticities of demand in the segments. Is this likely to result in greater or less recycling? Empirical evidence has indicated that price elasticities of demand for secondary materials are generally lower than for primary materials. As such, market segmentation and price discrimination is likely to result in relatively greater rates of recycling than would be the case it its absence.

Moreover, it is not clear that secondary material markets are more imperfect than primary material markets. One example is the possibility of oligopoly and market power among UK re-processors of used newsprint (Ecotec, 2000). Market power may also be exercised within the secondary material market. Recycling operators at the collection and sorting level are commonly thought to be price takers in their transactions with buying traders. Moreover, they have little influence over their costs in terms of the price they pay for waste (EC 1998).

In the event that there is market power in virgin material markets, and that this has adverse implications for recycling, there are few tools at the disposal of public authorities beyond the usual application of competition policy. Moreover, it does appear that this is one area in which market failures do not seem to be particularly important, at least at the national level. However, in cases where markets are primarily local in nature (i.e. construction and demolition waste) there may be issues of market power which need to be addressed. This is particularly true if the buyers of recycled materials are granted concessions which are not subject to competition, whether in the market or for the market.

4. Testing for Market Efficiency

Testing directly for the 'efficiency' of markets is exceedingly difficult. In effect, the researcher must compare actual price levels (or trade volumes) with those that would exist in a counterfactual 'ideal' market. In principle this requires data on all the likely determinants of supply and demand for the commodity in question. In the area of recycling this is not possible. Moreover, in some cases the market itself may not exist as a consequence of failures (i.e. transaction costs are such as to remove all potential gains from trade), further complicating the analysis.

However, in an effort to cast some light on this question, efforts were undertaken to examine the determinants of recycling rates in OECD countries. This uses data which is submitted annually by OECD Member country governments to the OECD Environment Directorate as part of its work programme on environmental information and outlooks. The data covers 20 OECD countries, over the period 1985-2000, and includes information on glass and paper recycling. As a first step, a reduced form model of glass recycling rates in OECD countries was estimated in order to see if some important determinants had the expected effects. Indeed it was found that glass recycling rates are affected by variables such as GDP per capita, labour costs, degree of urbanisation, population density and policy shocks in the expected manner (see Annex 2 for details).

In a more informal manner, Chapter 4, van Beukering compares the ranking of different forms of recovery of used tyres (incineration, retreading, reuse, etc…) in a number of European countries against their ranking in terms of financial viability, and finds that the two diverge quite likely. However, he argues that this is likely to be largely explained by the different nature of policy interventions in the different countries rather than inefficiencies within the markets themselves.

In the absence of the possibility of comparing existing market performance with some counterfactual ‘ideal’, more rudimentary measures are required. Price volatility can be considered an indicator of market inefficiency. Indeed, it is widely argued that price volatility in secondary material markets is very high. (See, for example, Kinnaman and Fullerton 1999, Ackerman and Gallagher 2001, SRMG 2000, CIWMB 1996, Porter 2002, Blomberg and Hellmer 2000, and Ecotec 2000). However, price volatility in and of itself is not a reflection of inefficiency in the market: efficient markets can have very volatile prices (i.e. for seasonal products which are perishable); and, inefficient markets can have very stable prices (i.e. a monopolist market which is not subject to demand shocks).

It can also arise from the structural characteristics of the sector, such as the widespread observation that both supply and demand are inelastic for many secondary material markets, including glass, paper, aluminium, plastics, and lead. (See Kinnaman and Fullerton 1997, Palmer et al. 1996, Kinkley and Lahiri 1984, Sigman 1995, Blomberg and Hellmer 2000, Edgrenc and Moreland 1989, and Bingham et al. 1983). Supply inelasticity appears to be a particular problem due to capital-specificity. On the demand side, the use of the secondary market as a ‘residual’ can lead to significant shocks. Under such conditions, prices are likely to fluctuate wildly.

However, in many cases price volatility can arise from some of the market imperfections discussed above (i.e. transaction costs leading to thin markets and variability in quality). As such, price volatility can be (but need not be) a reflection of market imperfections. This is significant since price volatility can have adverse implications for the long-term development of recycling markets, by introducing and reflecting uncertainty in the market. For instance, Wilson (2002) believes this to have been the case for the plastic bottle recycling sector.

In the longer term, such uncertainty can discourage investments with long planning horizons, even if the average rate of return is comparable to other investments. If investments are irreversible but flexible in terms of timing, recent work indicates that a more cautious approach to investment is optimal if there is input or product price uncertainty (see Pindyck and Dixit 1994). In such cases, firms will be reluctant to invest in material recovery.

In fact, a number of economists have seen such price uncertainty in the market for secondary materials as an important ‘brake’ on investment. (See Kinnaman and Fullerton 1999 and Blomberg and Helmer 2000 for discussions.) It is, therefore, important to determine whether secondary material prices are actually particularly volatile? Table 1.6 compares the standard deviations of monthly price changes for different pairs of ‘substitute’ primary and secondary materials, using American data from the Bureau of Labour Statistics. In most, but not all cases, prices for the secondary material are more variable than prices for primary materials. The exceptions are reclaimed rubber relative to synthetic and (particular) natural rubber, and zinc scrap relative to lead and zinc ore.[20]

Is there anything that public authorities can do to address price volatility. Some governments have introduced ‘price-smoothing’ policies, as in California (i.e. CIWMB), but these are likely to be little more successful than they have for other commodities. Moreover, it is clear that some policies have actually exacerbated volatility - i.e. the ‘glut’ of supply of recyclables in the mid-1990s due to colleciton programmes. As such, perhaps there is little that can be done directly in this area - not least since price volatility may reflect a well-functioning market.

20. In some cases where substitution can occur at different points in the production chain more than one ‘primary’ material has been selected. Figures illustrating monthly price changes are presented in Annex 3.

Table 1.6: Standard Deviations of Monthly Price Changes for Selected Primary and Secondary Materials

Textiles	
Processed Yarns and Threads	0.36
Textile Waste	2.08
Rubber	
Natural Rubber	6.25
Synthetic Rubber	1.66
Reclaimed Rubber	1.40
Zinc	
Lead & Zinc Ore	4.59
Zinc Scrap	4.50
Aluminum	
Aluminum Ingot	3.68
Aluminum Scrap	6.29
Copper	
Copper Ore	1.64
Copper Scrap	5.57
Iron	
Iron Ore	1.64
Iron Scrap	5.73
Paper	
Pulpwood	1.80
Paper-Making Woodpulp	3.98
Wastepaper	9.24

5. Conclusions

Many OECD governments have introduced targeted policies to encourage recycling. Such policies, if designed and implemented efficiently can reduce the environmental externalities associated with waste generation to an optimal level. However, this assumes that all other aspects of the market are functioning efficiently. However, as we have seen, there may be market inefficiencies such as information failures, consumption and technological externalities, market power, and other factors at work, discouraging the uptake of recyclables in the market. Not all of these market inefficiencies affect all markets for recyclable materials. Indeed, in some cases none of them may be present. Markets for many types of metal scrap may be almost ideal and certainly less imperfect than markets for some primary materials.

However, in markets for some other types of secondary materials, significant market inefficiencies exist:

- If transaction costs are high there may be significant unrealised gains from trade as many potential buyers and sellers are unable to engage in profitable exchange;

- If information failures are important then waste quality may be sub-optimal, and this may ultimately undermine the market;

- If technological externalities are important, then products will be designed in a manner which is sub-optimal with respect to recyclability;

- If consumption externalities are important, then initial rates of market penetration for goods derived from secondary materials are likely to be too low; and,

- And finally, if market power is prevalent, processors of primary materials and manufacturers of commodities derived from primary materials may restrict the share of secondary materials through strategic behaviour.

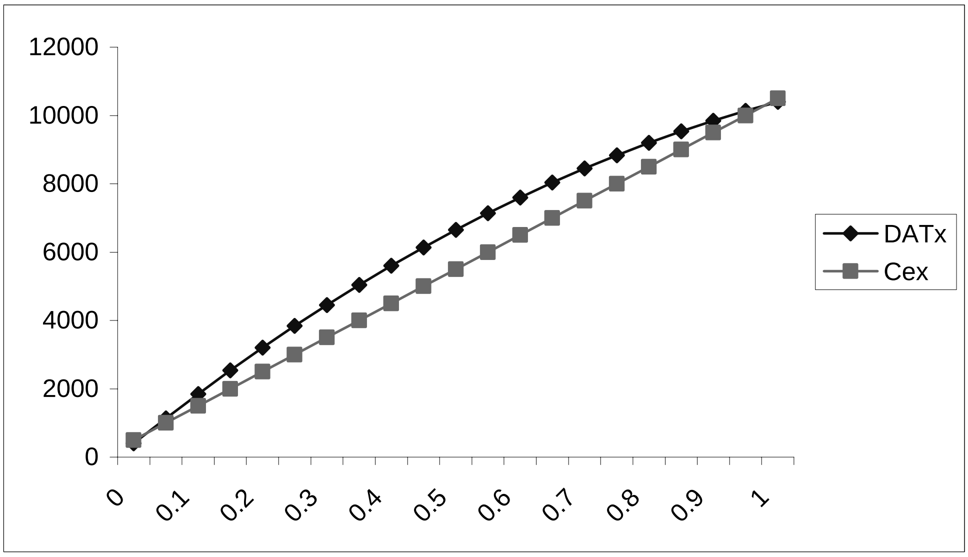
The effects of such market failures can lead to adverse economic impacts such as unrealised gains from trade as buyers and sellers do not find their counterparts, reduced investment levels due to general uncertainty and price volatility, and even the presence of low-quality secondary materials on the market. In many cases the market itself will find alternative ways to deal with many of these apparent inefficiencies in the market – i.e. grading specifications (information failures), vertical integration (technological externalities), waste brokers (search costs), waste certifiers, etc. However, this will not always be the case. In such circumstances some well-targeted public policy measures can usefully address particular market failures, and thus encourage optimal recycling rates. In the final chapter of this report a full discussion of the policy implications of the failures discussed, and some recommendations, will be presented.

ANNEX 1: LOSS AVERSION – AN EXAMPLE

Assume that the use of retreaded tires rather than new tires results in increased probability (p) of a crash due to a tire blow-out, and that in extreme circumstances this might result in estimated damages of $10,000 (i.e. due to a crash).[21] If the household 'weights' adverse outcome[22] relatively more than the 'benign' outcome y[23], it will behave as though the probability of a blow-out is much greater than it actually is. For instance, a probability of 2.65% under such conditions will result in a valuation by the household equal to 5% under 'certainty equivalent'assumptions.[24]

Assuming further that retreaded tires are less expensive than new tires ($400 rather than $500), if the household believes that the probability of a blow-out is 1%, under certainty equivalent (CE) calculations the household will be indifferent between the two types of tire (both are valued at $500). However, under 'disappointment aversion' (DAT) assumptions the household will 'cost' retreaded tires at almost $550, thus reducing demand. (The schedule of costs under certainty equivalents and DAT for all probabilities is given in Figure A1.1.)

Figure A1.1: Perceived Hypothetical Costs of Purchasing Retreaded vs. New Tires



21. These figures are purely hypothetical, and probability of a blow-out (and a crash arising from a blow-out) are of course much less than these figures. It is important to note however that what matters for the household is not actual probabilities but perceived probabilities.

22. A parameter (d) equal to 1.25 rather than unity.

23. A parameter (e) equal to 0.75.

24. The formula is px + (1-p)y + (e-d)p(1-p)(x-y), rather than the certainty equivalent px + (1-p)y.

ANNEX 2: THE DETERMINANTS OF GLASS RECYCLING RATES IN THE OECD

by Nick Johnstone, Julien Labonne and Pascale Scapecchi

The explanatory variables thought to be important determinants of glass recycling rates include socio-economic, demographic, policy and other variables. Gross domestic product per capita (GDPPC) is expressed in constant US dollars. It has been obtained from the World Bank's *World Development Indicators* (2002). It is commonly hypothesised that the effect on recycling rates is positive due to increased environmental consciousness with rising income levels. However, this hypothesis has not been consistently supported in previous studies, and there are good theoretical reasons to question the expected sign.

Labour costs (LABCOST) are average wage rates in manufacturing sectors (expressed in constant US dollars). This data was obtained from the American Bureau of Labor Statistics (http://www.bls.gov/fls). To the extent that wage rates reflect the opportunity cost of sorting and other labour-intensive aspects of recycling, it is expected that the sign on this coefficient is likely to be negative – as wage rates rise, recycling rates fall. The inclusion of LABCOST should also help us to gain more accurate estimates of the GDPPC variable.

A variable reflecting the percentage of the population in urban centres greater than 100,000 people (URB) has been included. This has been obtained from the World Bank's *World Development Indicators* (2002). The sign on this is expected to be positive, since there are thought to be significant economies of density in glass collection. POPD is a variable for population density and seeks to capture related effects. It is also from World Bank's *World Development Indicators* (2002).

A variable reflecting the production (in volume terms) of silica (SILICA) manufactured domestically in the respective countries has been included. It has been obtained from the UN Department of Economic and Social Affairs' *Industrial Commodity Statistics* (2002). The sign on this coefficient is expected to be negative, reflecting the fact that silica and glass cullet are substitutes in production.

A variable (ENERGY) has been included in order to reflect the price of energy (light diesel fuel oil), since energy savings are thought to be one of the principal benefits of the use of cullet rather than virgin silica. It has been obtained from the International Energy Agency's *International Energy Statistics*. The expected sign is positive, given the energy savings associated with the use of glass cullet relative to primary silica.

A dummy variable reflecting the EU's *Directive on the Recycling of Packaging Waste* is included (DEURPOL). It is hoped that this will capture the effects that the introduction of the Directive will have had on recycling rates in member countries.

And finally, there are thought to be a number of missing variables which have not been included in the model, due to data limitations. As such, a fixed effects model has been chosen in which dummy variables are included for each cross-sectional unit (the 18 countries). These reflect the time-invariant effect of individual countries on observed recycling rates.

In theory the full data set has 300 observations (15 years by 20 countries). However, in practice there are significant numbers of missing observations, including the dependent variable. As a consequence the models estimated use asymmetric pooled data sets of varying dimensions. Unfortunately, this means that direct comparisons between different regressions is not possible since the sample changes.

A large number of models were estimated in order to evaluate which variables were consistently statistically significant. The energy and silica variables were dropped since they did not prove to be significant in any of the models estimated. Moreover, due to the presence of a large number of missing observations for these two variables, their inclusion reduced the sample size markedly.

The results of the best-fitting model are presented in Table A.1 (coefficients for the cross-sectional dummies not included). An F-test indicated that the null hypothesis of equal intercept terms across cross-sectional units could not be accepted, indicating that the fixed effects model was appropriate. However, a Breush-Pagan test revealed considerable heteroscedasticity and thus the standard errors presented have been estimated using the White correction procedure.

As noted above, the inclusion of both labour costs and GDPPC has allowed us to tease out the demand side and supply side factors. As expected, the sign on GDPPC is positive. This indicates that demand for recycling (personal and social) rises with income levels. The income elasticity is marginally greater than unit elastic.

Unexpectedly, the sign on labour costs is also positive. Thus, the opportunity cost of time within the household (to the extent that it is reflected in this variable) does not appear to deter recycling. Given the close correlation between GDPPC and LABCOST (just over 0.7), the latter may be capturing such residual income effects. However, the estimates for both URB and POPD are very significant and of the expected sign (positive). Recovery and reuse of glass increase with the degree of urbanisation and the population density. Given the very significant density economies associated with collecting glass, such results are hardly surprising. However, the size of the coefficients are surprisingly large.

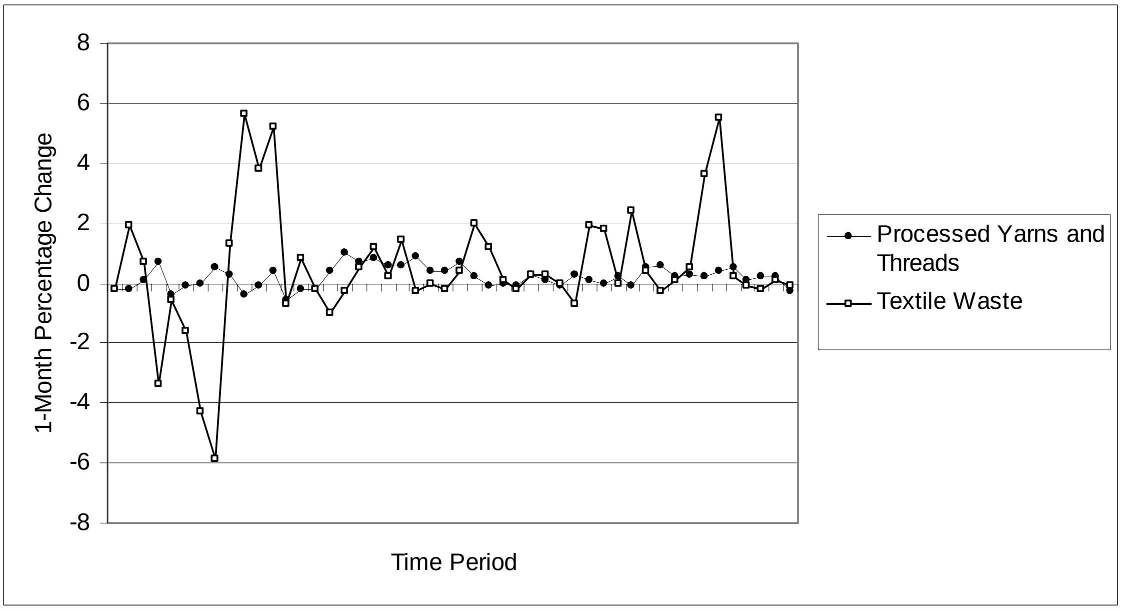
And finally the dummy variable for the EU Directive (DEURPOL) is significant and positive, indicating that the Directive did have an effect on recycling rates in member countries. However, it must be emphasised that such a result must be treated with considerable caution. The collection of household-level data across countries is required to assess such issues in greater detail and with greater reliability.

Table A2.1 – Results

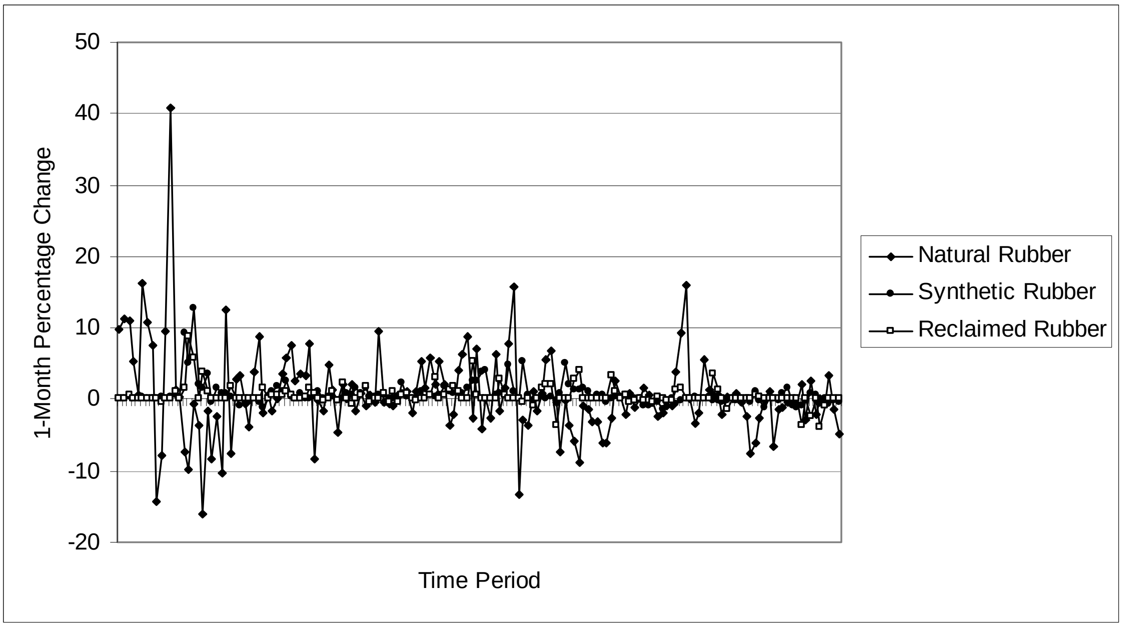
Variable	Estimate	Standard error	t value	Pr > $\|t\|$
Intercept	-22.485	2.8020	-8.02	< .0001
Lgdppcd	1.011	0.2005	5.05	< .0001
Llabcost	0.232	0.0633	3.67	0.0003
Lurb	1.049	0.3423	3.07	0.0025
Lpopd	3.420	0.8658	3.95	0.0001
Deurpol	0.109	0.0409	2.66	0.0084
F-value = 33.14, pr > F = < .0001				
R-square = 0.8563				
Number of observations = 211				

ANNEX 3: PRICE VOLATILITY FOR SELECTED PRIMARY & SECONDARY MATERIALS[25]

Textiles

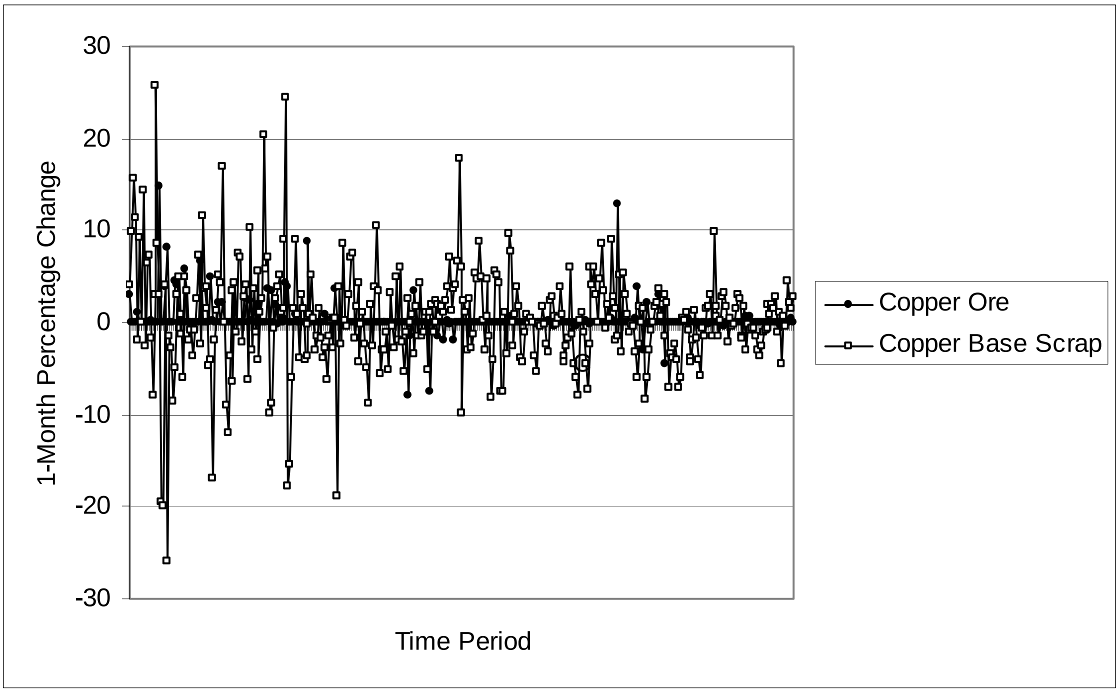


Rubber

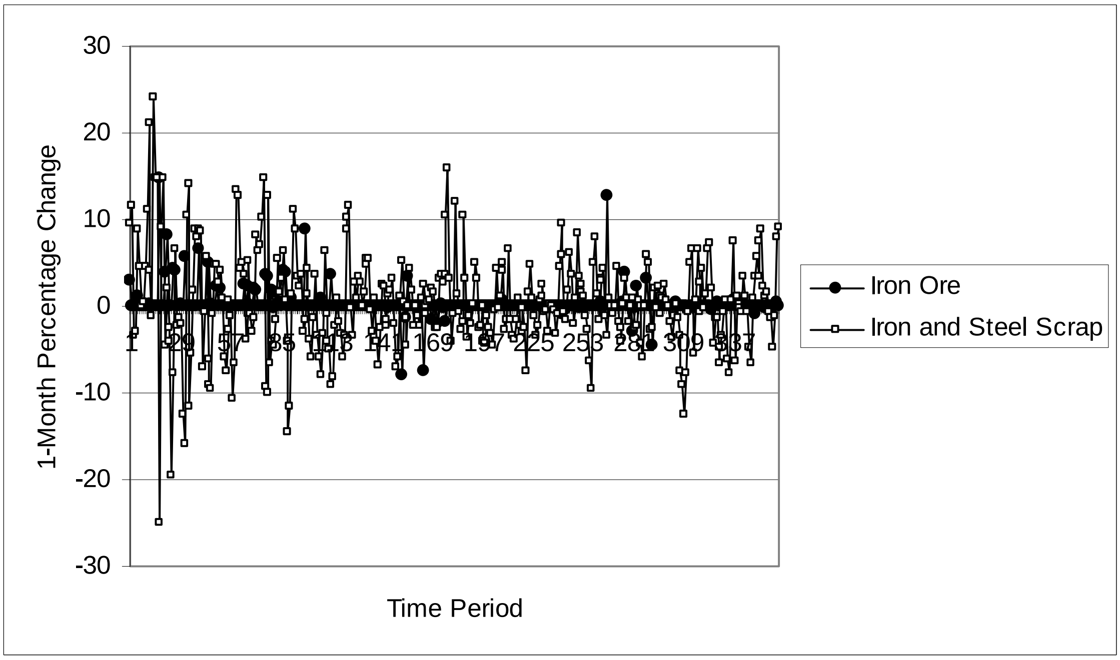


25. Standard deviations of monthly price changes. Source: US Bureau of Labour Statistics (www.bls.gov). Time periods are monthly and range from 1962 to 2001. However, due to missing data, in order to ensure consistency within commodity pairs, the time period varies.

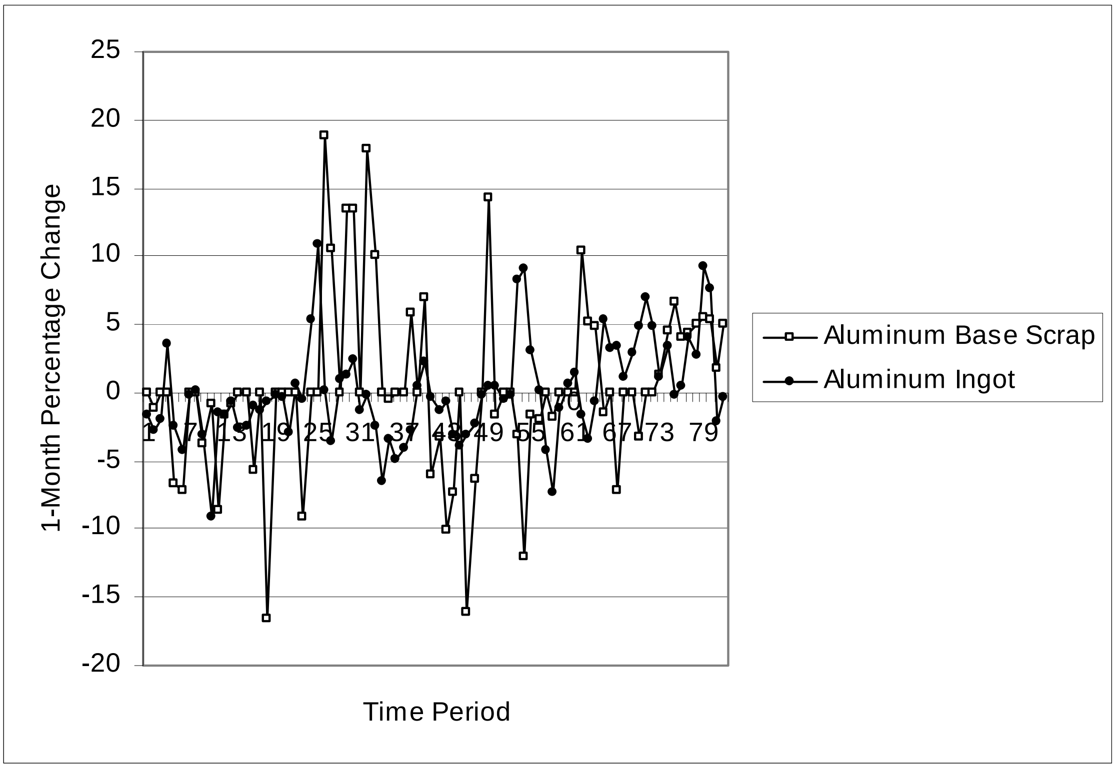
Copper


30
20
10
0
-10
-20
-30
1-Month Percentage Change
Time Period
Copper Ore
Copper Base Scrap

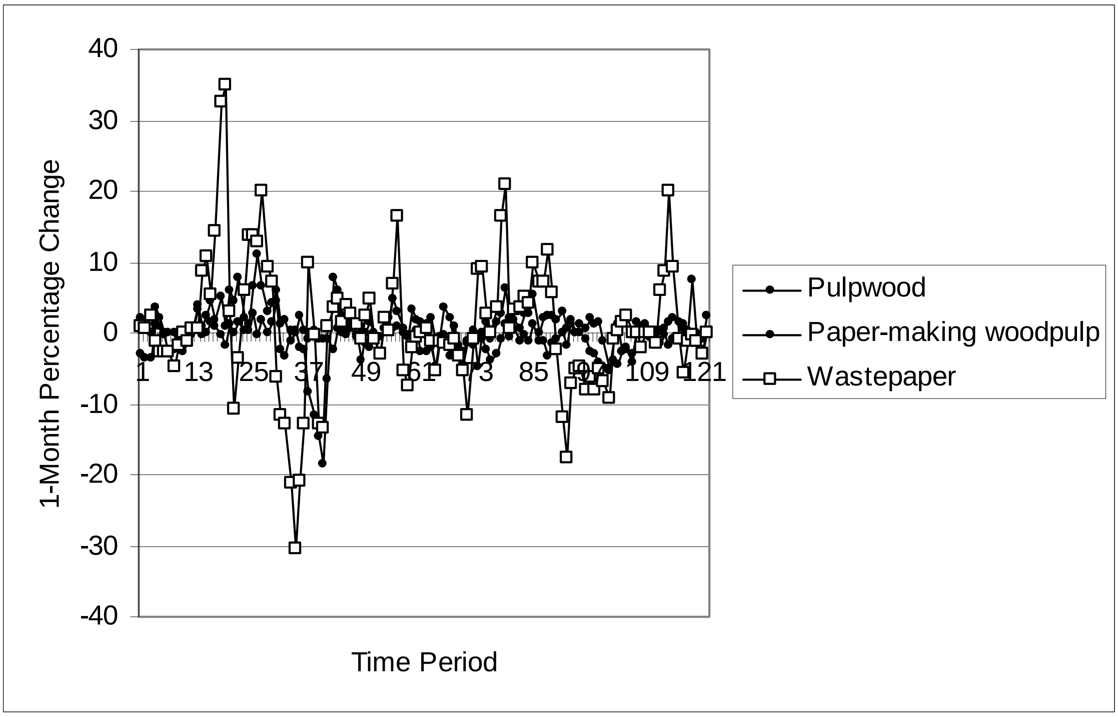
Iron


30
20
10
0
-10
-20
-30
1-Month Percentage Change
1
29
57
85
113
141
169
197
225
253
281
309
337
Time Period
Iron Ore
Iron and Steel Scrap

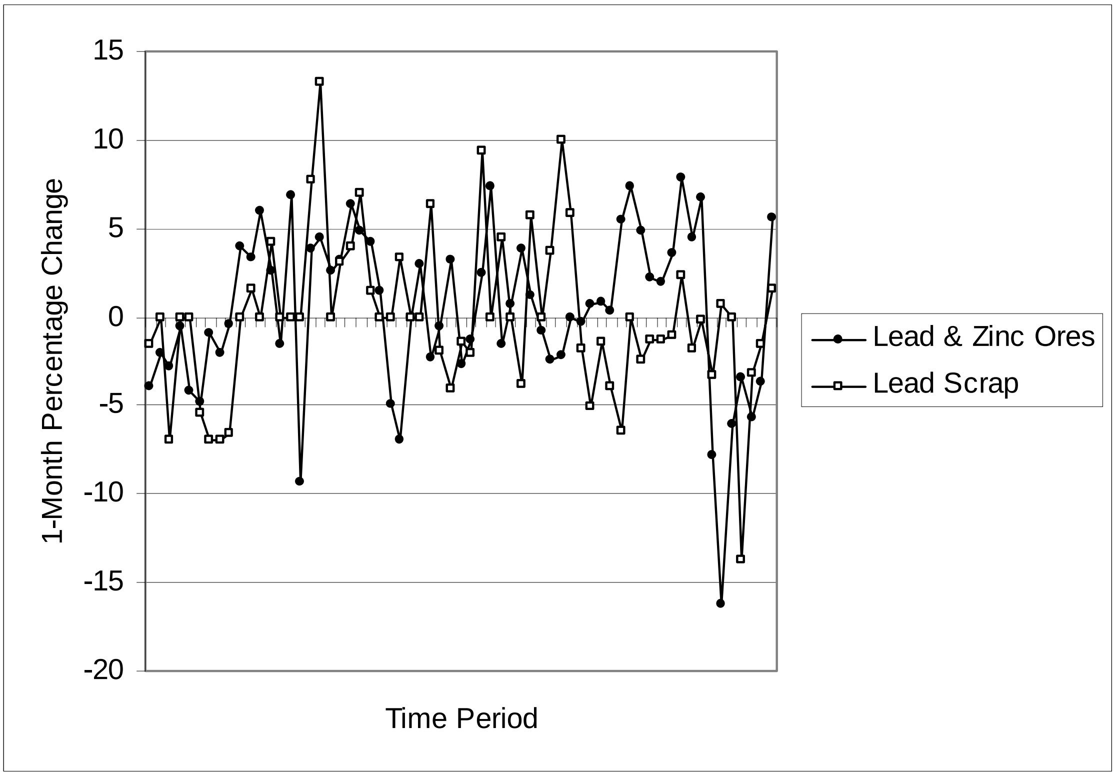
Aluminium



Paper



Zinc


15
10
5
0
-5
-10
-15
-20
1-Month Percentage Change
Time Period
Lead & Zinc Ores
Lead Scrap

REFERENCES

ACKERMAN, Frank AND K. GALLAGER (2001).. "Mixed Signals: Market Incentives, Recycling and the Price Spike of 1995" Tufts University, Global Development and Environment Institute, Working Paper 01-02.

AKERLOF, George, A (1970). "The market for "lemons"; quality uncertainty and the market mechanism", *Quarterly Journal of Economics*, 89(3).

ANDERSON, Simon P. et al. (1994). "Price discrimination via second-hand markets" in *European Economic Review*, Volume 38, Issue 1, January 1994, Pages 23-44.

APOTHEKER, Steve (1993). "It's black and white, and recycled all over", *Resource Recycling*, 12(7).

ARDANT, Ignace AND F. GASPART (2003). "Scrapping the Surface: Lemons, Informational Institutionals and Emerging Markets in a Recycling Branch' unpublished mimeo, Laboratoire d'économétrie de l'Ecole Polytechnique, Paris.

AYRES, R.U. (1997). "Metals recycling: economic and environmental implications", *Resources, Conservation and Recycling*, Vol. 21, pp. 145-173.

BARDOS, R. P. et al. (1991). 'Market Barriers to Materials Reclamation and Recycling' (Stevenage: AEA Technology).

BEEDE, D. N. AND D. BLOOME (1995). "Economics of the Generation and Management of Municipal Solid Waste" NBER Working Paper No. 5116, Cambridge Mass.

BELL, David E. (1985). 'Disappointment in Decision Making under Uncertainty' in *Operations Research*, Vol. 33, No. 1, pp. 1-27.

BERNDT, Ernst, R., ROBERT S. PINDYCK, PIERRE AZOULAY (2000)."Consumption externalities and diffusion in pharmaceutical markets; antiulcer drugs", National Bureau of Economic Research, NBER working paper 7772, Cambridge, Mass.

BEUKERING, Pieter van (2001). "Empirical Evidence on Recycling and Trade of Paper and Lead in Developed and Developing Countries", *World Development* 29(10), pp. 1717-1737.

BINGHAM, Talyer H. et al. (1983). 'Conditionally Predictive Supply Elasticity Estimates: Secondary Materials Obtained from Municipal Residuals' in *Journal of Environmental Economics and Management*, Vol. 10, pp. 166-179.

BLOMBERG, Jerry and Stefan Hellmer (2000). "Short-Run Demand and Supply Eleasticities in the Western European Market for Secondary Aluminium" in *Resources Policy,* Vol. 26, pp. 329-50.

CALCOTT, Paul and Margaret Walls (2000). "Policies to Encourage Recycling and "Design for Environment": What to Do When Market are Missing" (Washington, D.C.: RFF Discussion Paper 00-30).

CALIFORNIA INTEGRATED WASTE MANAGEMENT BOARD (1996). ‘Market Status Report: Waste Tires” (Sacramento: CIWMB).

CALIFORNIA INTEGRATED WASTE MANAGEMENT BOARD (1996). ‘Market Status Report: Postconsumer Plastics” (Sacramento: CIWMB).

CALIFORNIA INTEGRATED WASTE MANAGEMENT BOARD (1997). ‘Tire Recycling Program Evaluation” (Sacramento: CIWMB).

CHICHILNISKY, Graciela (2000). ‘An Axiomatic Approach to Choice Under Uncertainty with Catastrophic Risks’ in *Resource and Energy Economics*, Vol. 22, pp. 221-231.

COASE, R.H. (1937). “The nature of the firm”, reprinted in Williamson, Oliver E. and Scott E. Maston (eds.) (eds.) *The Economics of Transaction Costs* (Cornwall: MPG Books Ltd, 1999).

COASE, R.H. (1960). “The problem of social cost”, *Journal of Law and Economics*, VIII, October.

COLLINS, R. J. and P. J. Nixon (2003). “Implications of the Harmonisation of Construction Product Standards for the Use of Recycled and Secondary Aggregates” WRAP Research and Development Report: Aggregates, Oct. 2003.

CURLEE, Randall, T. (1986). *The Economic Feasibility of Recycling: A Case Study of Plastic Wastes* (New York: Praeger).

CURLEE, Randall, T. (1986). *The Economic Feasibility of Recycling: A Case Study of Plastic Wastes* (New York: Praeger).

DUCHIN, F., and G. Lange, 1998. “Prospects for the recycling of plastics in the United States,” *Structural Change and Economic Dynamics*, Vol. 9, No. 3, pp. 307-331.

EC (1998a). “The competitiveness of the recycling industries”, Report prepared for the European Commission, Directorate General for Industry, Brussels.

EC (1998b). “Waste management, Competitiveness of the recycling industries”, European Commission (abbreviated version of “EC 1998” above) http:/europa.eu.int/scadplus/leg/en/lvb/128065.htm

EC (2000). “Recycling Forum 1999-2000: Final Report”, European Commission, Directorate General for Enterprise, Brussels.

ECOTEC (2000). “Policy instruments to correct market failure in the demand for secondary materials”, Report prepared by Ecotec Research and Consulting Ltd., Birmingham.

EDGREN, John A. and Kemper W. Moreland (1989). “An econometric analysis of paper and wastepaper markets”, *Resources and Energy*, Vol. 11, pp. 299-319

EHRENFELD, John and Nicholas Gertler (1997). “Industrial Ecology in Practice: The Evolution of Interdependence at Kalundborg” in *Journal of Industrial Ecology*, Vol. 1, No. 1, pp. 67-79.

ENVIRONMENTAL PROTECTION AGENCY (1993). “Developing markets for recyclable materials; policy and program options”, Report prepared by Mt. Auburn Associates, Inc. and Nothreast-Midwest Institute, for U.S. Environmental Protection Agency, Washington D.C.

ENVIRONMENTAL RESOURCES MANAGEMENT LIMITED (2000). “Research study on international recycling experience”, Report prepared for DETR by ERML, London.

ENVIROS (2002). “Recycled Glass Market Study and Standards Review: 2003 Update” WRAP Research Report: Glass, July 2002.

ENVIROS (2003). “Survey of Applications, Markets and Growth Opportunities for Recycled Plastics in the United Kingdom” WRAP Research Report: Plastics, Aug. 2003.

FEDEREC (2003). ‘Le Marché de la Récupération du Recyclage et de la Valorisation en 2002’ (Paris: FEDEREC).

GOLDBERG, Victor, P. and John Erickson (1999). “Quantity and price adjustment in long-term contracts: a case study in petroleum coke”, in Carroll, Glenn R. and David J. Teece, (eds.) *The Economics of Transaction Costs* (Cornwall: MPG Books Ltd.).

GRANT, Darren (1999). ‘Recycling and Market Power: A More General Model and Re-Evaluation of the Evidence” in *International Journal of Industrial Organization*, Vol. 17, pp. 59-80.

GRANT, Simon, Atsushi Kajii and Ben Polak (2001). ‘Different Notions of Disappointment Aversion’ in *Economics Letters*, Vol. 70, pp. 203-208.

GRASSROOTS RECYCLING NETWORK (1999). “Welfare for waste”, Grassroots Recycling Network, Athens, Georgia.

GRAVELLE, Hugh, and Ray Rees (1992). *Microeconomics* (London: Longman Group.).

GUL, Faruk (1991). ‘A Theory of Disappointment Aversion’in *Econometrica*, Vol. 59, No. 3, pp. 667-686.

HUSTON, John and Roger W. Spencer (2002). "Quality, Uncertainty and the Internet: The Market for Cyber Lemons" in *American Economist*, Vol. 46, No. 1, Spring 2002, pp. 50-60.

INSTITUTE OF SCRAP RECYCLING INDUSTRIES (2003). ‘Scrap Specifications Circular’ (Washington, D.C.: ISRI).

INTERNATIONAL COUNCIL ON METALS AND THE ENVIRONMENT (1996). “Non-ferrous metals recycling; a complement to primary metals production”, ICME, Ottawa.

JAFFE, Adam, B., Richard G. Newell, Robert N. Stavins (2000). “Technological change and the environment”, National Bureau of Economic Research NBER working paper 7970, Cambridge, Mass.

JOVANOVIC, Boyan and Glenn MacDonald (1993). “Competitive diffusion”, National Bureau of Economic Research, NBER working paper 4463, Cambridge, Mass.

KERTON, Robert R. and Richard W. Bodell (1995). "Quality Choice and the Economics of Concealment: The Marketing of Lemons" in *Journal of Consumer Affairs*, Vol. 29, No. 1, Summer 1995, pp. 1-28

KINKLEY, Chu-Chu and Kajal Lahiri (1984). 'Testing the Rational Expectations Hypothesis in a Secondary Materials Market' *Journal of Environmental Economics and Management*, Vol. 11, pp. 282-291.

KINNAMAN, Thomas C. and Don Fullerton (1999). "The Economics of Residential Solid Waste Management" (Washington, D.C.: NBER Working Paper 7326).

KOLSTAD, Charles, D. (2000). *Environmental Economics* (Oxford: Oxford University Press)

KREPS, David, M. (1999). "Markets and hierarchies and (mathematical) economic theory", in Carroll, Glenn R. and David J. Teece (eds.). *The Economics of Transaction Costs* (Cornwall: MPG Books Ltd.).

MARTIN Robert E. (1982). "Monopoly Power and the Recyclying of Raw Materials" in *The Journal of Industrial Economics*, Volume 30, No. 4, pp. 404-419.

O'NEILL, William O. (1983). "Direct empirircal estimation of efficiency in secondary materials markets; the case of steel scrap", *Journal of Environmental Economics and Management*, Vol. 10, pp. 270-281.

OECD (2002). *Environmental Performance Review: Japan* (Paris: OECD).

OECD (2003). *Environmentally Sustainable Buildings* (Paris: OECD).

OECD (2005) *Environmentally Harmful Subsidies* (Paris: OECD)

WILLIAMSON, Oliver E. Scott Masten *The Economics of Transaction Costs* (Edward Elgar, 1999).

PALMER, Karen and Margaret Walls (1999). 'Extended Product Responsibility: An Economic Assessment of Alternative Policies' (Washington D.C.: RFF Discussion Paper 99-12).

PAPINESHI, J. (2003). "Standards and Specifications Affecting Plastics Recycling in the United Kingdom" WRAP Research Report, Jan. 2003.

PEARCE, David W. (1992). *MacMillan Dictionary of Modern Economics* (London: MacMillan Press Ltd).

PINDYCK, Robert, S. and Avinash, K. Dixit (1994). *Investment Under Uncertainty*, (Princeton, New Jersey: Princeton University Press).

PINDYCK, Robert, S. and Avinash, K. Dixit (1994). *Investment Under Uncertainty*, (Princeton, New Jersey: Princeton University Press).

PORTER, Richard (2002). *The Economics of Waste* (Washington D.C.: Resources for the Future).

RDC/PIRA (2003). "Evaluation of the Costs and Benefits for the Achievement of Reuse and Recycling Targets for the Different Packaging Maerials in the Frame of the Packaging and Packaging Waste Directive" Final Consolidated Report, March.

REID, Murray (2003). "A Strategy for Construction and Demolition Waste as Recycled Aggregates" WRAP Research and Development Report: Aggregates, Sept. 2003.

RESOURCE RECOVERY FORUM/ENVIROS (2000). *Recycling Achievement in Europe* (Skipton: Resource Recovery Forum).

RESOURCE RECOVERY FORUM/ENVIROS (2000). *Recycling Achievement in North America* (Skipton: Resource Recovery Forum).

SCHMALANSEE, Richard (1977). 'Using the H-Index of Concentration with Published Data' in *The Review of Economics and Statistics*, Vol. 59, No. 2, pp. 186-193.

SIELING, Mark S. (1990). "Productivity in scrap and waste materials processing", *Monthly Labor Review*, April.

SIGMAN, Hilary (1998). "Midnight Dumping: Public Policies and Illegal Disposal of Used Oil" in *RAND Journal of Economics*, Vol. 29, No. 1, pp. 157-178.

SOUND RESOURCES MANAGEMENT (1999A). "A tale of two realities", *The monthly Uneconomist* 1(1), Sound Resource Management, Seattle.

SOUND RESOURCES MANAGEMENT (1999B). "Forecasting recycled paper prices", *The monthly Uneconomist*, 1(2) Sound Resource Management, Seattle.

SOUND RESOURCES MANAGEMENT (2000). "The Chicago board of trade recyclables exchange; evaluation of trading activity and impacts on the recycling marketplace" Sound Resource Management Group, Inc., Seattle.

SUSLOW, V. Y. (1986). "Estimating Monopoly Behaviour with Competitive Recycling"in *Rand Journal of Economics*, Vol. 17, pp. 389-403.

SWAN, P. L. (1980). "Alcoa: The Influence of Recycling on Monopoly Power" in *Journal of Political Economy*, Vol. 88, pp. 76-99.

SYMONDS, Argus, COWI and PRC Bouwcentrum (1999). "Construction and Demolition Waste Management Practices and their Economic Impacts" Report prepared for European Commission, DG XI.

TILTON, John E. (1999). "The future of recycling", *Resources Policy*, Vol. 25, pp. 197-204

UN DEPARTMENT FOR ECONOMIC AND SOCIAL AFFAIRS (2001). *Industrial Commodity Statistics Yearbook* (New York: United Nations).

US DEPARTMENT OF COMMERCE, CENSUS OF MANUFACTURES (2001). *Administrative and Support and Waste Management and Remediation Services* (Washington, D.C.: 2001).

US DEPARTMENT OF COMMERCE, CENSUS OF MANUFACTURES (2001). *Concentration Ratios in Manufacturing* (Washington, D.C.: 2001).

US ENVIRONMENTAL PROTECTION AGENCY (2002). *Municipal Solid Waste in the United States* (Washington DC: EPA Office of Solid Waste).

VAN ARK, Bart and Erik Monnikhof (1996). 'Size Distribution of Output and Employment: A Data Set for Manufacturing Industries in Five OECD Countries" (OCDE/GD(96)18).

VARIAN, Hal R. (1992). *Microeconomic Analysis* (New York: W. W. Norton & Company, Inc.)

WILLIAMSON, Oliver E. (1999). "The vertical integration of production: market failure considerations" in Carroll, Glenn, R. & Teece, David. J. (eds.) *Firms, Markets and Hierarchies, the Transaction Cost Economics Perspective* (New York: Oxford University Press).

WILLIAMSON, Oliver E. and Sidney G. Winter (1993). *The Nature of the Firm: Origins, Evolution and Development* (Oxford: OUP).

WILSON, Scott (2002). "Plastic Bottle Recycling in the UK" WRAP Research Report: Plastics, March 2002.

WIMMER, Bradley S. and Brian Chezum (2003). 'An Empirical Examination of Quality Certification in a 'Lemons Market'' in *Economic Inquiry*, Vol. 41, No. 2, pp. 279-291.

WRAP (2003). "Assessment of Quality Arising from Existing Paper Collection Methods Against European Recovered Paper Grades" WRAP Research Report: Paper, Oct. 2003.

CHAPTER 2:

IMPROVING MARKETS FOR WASTE OILS

by

David Fitzsimons
Oakdene Hollins, United Kingdom

1. Introduction

This report reviews evidence of environmental damage caused by inappropriate management of used lubricating oil; provides an overview of the markets for used lubricating oil; reviews the market barriers and failures affecting some of the post-collection markets, and highlights the implications for public policy.[26]

Of the 37 million tonnes of lubricants sold globally in 2001, 40% to 50% was consumed or lost during use: the fate of the remaining 50% to 60% of the oil which is potentially recoverable is the subject of this report. Typically, 65% of the recoverable oil is engine oil, the remainder being from a wide variety of industrial sources.

Where it is not collected, there is a risk of pollution from waste oil that is dumped in sewers and rivers or tipped on soil. Waste oil can contaminate freshwater, with associated impacts on aquatic plants and fish, water treatment facilities, and even human health. Non-collection, and its attendant environmental risks, appears to be especially likely in sparsely populated regions. In addition, where it is not collected, it is also considered as a waste of resources since waste oils can be used as a fuel or re-refined to new base oils and save primary non-renewable resources.

After collection, used lubricating oil is traded as feedstock for a variety of uses:

- Fuel in small-scale space heaters
- Fuel in industrial-scale processes such as power stations and cement kilns
- Re-processing to be used as fuel
- Re-processing to produce various hydrocarbon products such as drilling fluids
- Re-refining or regeneration to produce new lubricants.

26. To assist in the preparation of this report we visited a number of existing used oil re-refining plants in Europe and used oil collection businesses. We are grateful in particular to Jean Vercheval of Groupement Européen de l'Industrie de la Régénération (GEIR), who provided us with useful information; Christian Hartmann of Baufeld Oel GmbH, Jacques Ledure of SITA Belgium, Renato Schieppati of Viscolube SpA and Paul Ramsden of OSS Ltd for the access given to their treatment and recycling plants. Whilst acknowledging this assistance all errors, omissions and opinions are those of the author.

The choice between these various options depends upon policy incentives and market conditions. Market demand for burning waste oil directly in small-scale space heaters and in industrial-scale processes is potentially very much greater than the supply. Consequently, depending upon local environmental regulations, the price paid for waste oil is close to its nearest substitute of coal or medium to heavy fuel oil, and can be lower if the environmental impacts of some post-consumption uses are unregulated (or existing regulations unenforced).

In the face of market pressure from using waste oil as fuel, several governments have introduced policies to promote re-refining of waste oil in preference to direct burning or production of fuels, in some cases through public expenditures. Such expenditures are usually justified on the basis of purported environmental benefits (conservation of a non-renewable resource and to avoid air quality issues arising from direct burning). For instance, a recent life-cycle analysis of waste oil regeneration vs. incineration of waste oils in cement kilns, finds in its main scenarios that for the majority of considered impact categories regeneration is less environmentally damaging.[27] However, as reviewed below, in general life cycle assessments are generally inconclusive over whether re-refining is preferable to recovery of energy uses from waste oils, except in some cases (such as burning in unregulated space heaters).

Moreover, designing policies that favour re-refining can be problematic as investors confront several market barriers and failures. The major barriers are: information failures relating to waste quality; risk aversion to using re-refined base oils; certain technological externalities and the market power of incumbents. In the past this has restricted the market, sometimes requiring significant public expenditures (implicit or explicit) to support the market for re-refining. While some have diminished in importance with time (e.g. risk aversion), the market remains far from ideal. In other cases the market for re-refining may be undermined by policy failures, such as inadequate enforcement of environmental regulations associated with incineration and disposal. Increased enforcement of regulations which are set at optimal levels of stringency will favour the least damaging waste management option.

A key policy question is therefore, the determination of the optimal point of intervention in the market. While the distinction is usually one of degree and not kind, some governments appear to focus on maximising collection rates (leaving the market to determine the ultimate recovery use), while others favour providing support for re-refining. This is environmentally significant. On the one hand, it is possible that the use of scarce public resources to encourage re-refining may have adverse implications for collection rates for a number of reasons reviewed below. On the other hand, some policies designed to maximise collection rates may make re-refining less economic. As such, the determination of the optimal point of policy intervention is key.

2. The Market for Lubricants

3,700 million tonnes of crude oil are refined worldwide, of which approximately 1% is used to manufacture lubricant products. Lubricants derived from mineral oils are by far the most common type of lubricant, and it is on these products that this paper is focused. The small segment of lubricant products derived from vegetable sources and esters are not considered in detail.

27. See Groupement Europeen de l’Industrie de la Regeneration (2005) « An Environmental Review of Waste Oils Regeneration » (www.geir.fedichem.be).

2.1. Characteristics of Lubricant Oils

A complex variety of compounds, such as detergents, dispersants, anti-oxidants, emulsifiers, corrosion inhibitors etc., are later added to base oils by specialist manufacturers to create an extensive range of lubricant products. These additives can contribute 5-10% of the weight of the lubricating oil.

Once base oils have been blended with additives, packaged and marketed the value-added-multiple for engine oil is in the range 5 to 60 times the average value of crude oil. The exact mix of product sold into each national market varies, but in broad terms 56% of lubricant sold is for automotive uses and 44% for a variety of industrial uses. The most complex lubricants are top-tier engine oils, in particular those offering extended life. Other lubricants are used in sensitive consumer products such as cosmetics and in a variety of medical applications. (Table 2.1 provides estimates of global market shares for different uses.)

Table 2.1. Global Market Shares for Different Lubricant Uses

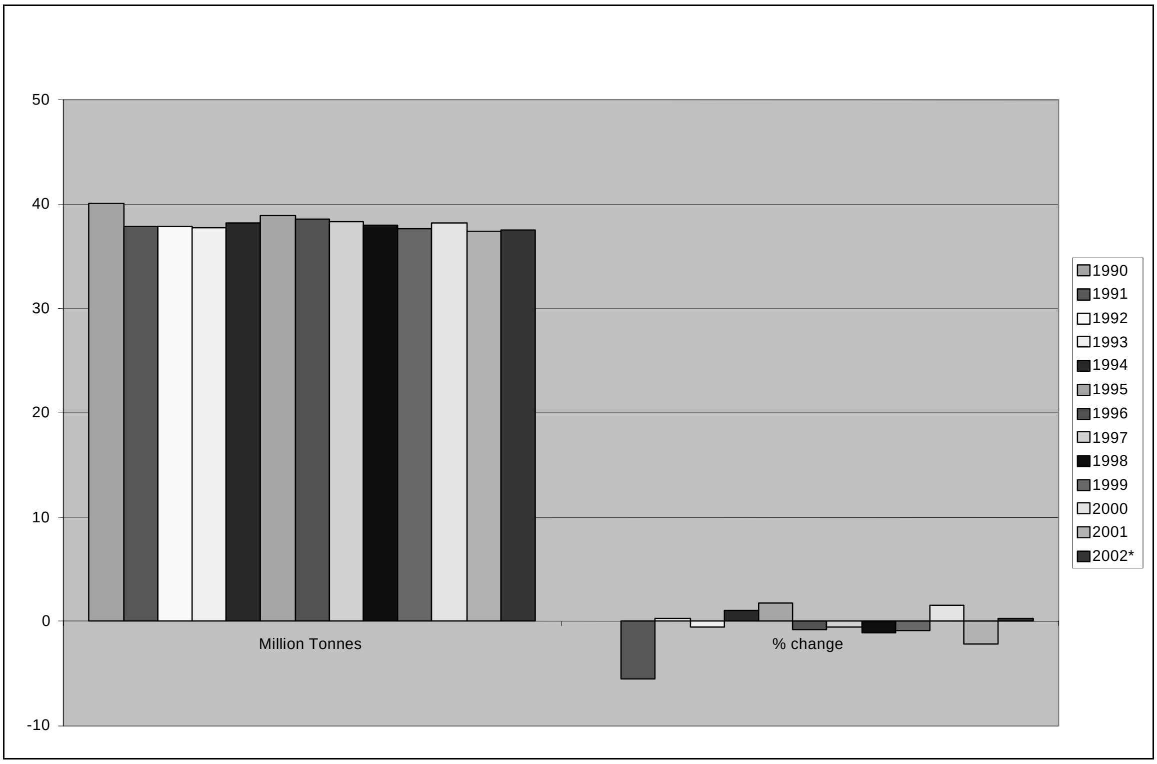
Type of Lubricant	Percentage of World Market
Automotive	55.1
Industrial Oils	24.5
Process Oils	11.6
Metalworking Fluids/Corrosion Preventatives	5.8
Greases	3.0

Source: FUCHS Petrolub AG Manheim - Paper to 7th ICIS LOR base oils conference 2003

2.2. Base Oil and Lubricant Product Trends

As can be seen in Figure 2.1 world demand for lubricants has been broadly static over the past 10 years, with a slight decreasing trend.

Figure 2.1. World Lubricant Demand (including marine oil)



Source: FUCHS Petrolub AG Manheim - Paper to 7th ICIS LOR base oils conference 2003

However, static global demand masks regional variations that show demand falling in North America and Western Europe and, to a lesser extent Latin America. Conversely, the Asia-Pacific region and Central and Eastern Europe are exhibiting growth rates in the region of 2% per annum (see Table 2.2).

Table 2.2. World Lubricant Demand (Excluding Marine Oils) by region

Region	2001 (000 T)	2002 (000 T)	%age change
N America	8,750	8,590	-1.8
Latin America	3,050	3,035	-0.5
Western Europe	4,930	4,885	-0.9
Central / Eastern Europe	4,610	4,690	1.8
Near/ Middle East	1,820	1,840	1.0
Africa	1,790	1,815	1.4
Asia-Pacific	10,610	10,820	2.0
WORLD	35,560	35,675	0.3

Source: FUCHS Petrolub AG Manheim - Paper to 7th ICIS LOR base oils conference 2003

This pattern in relative growth rates reflects a certain degree of "catch-up", with the fastest growing areas those in which demand is at relatively low absolute levels. Table 2.3 gives lubricant demand per capita for the same regions. Global average lubricant demand per capita of 5.9 kg masks significant regional variation, with North American demand over 10 times that of sub-Saharan Africa.

Table 2.3. Regional lubricant demand 2001 per capita

Region	Kg per Capita
N America	28.7
Australia/Oceania	18.1
Western Europe	12.6
Central / Eastern Europe	11.0
Near/ Middle East	10.7
Latin America	6.2
Asia	2.9
Africa	2.4
WORLD	5.9

Source: FUCHS Petrolub AG Manheim - Paper to 7th ICIS LOR base oils conference 2003

While demand is clearly shifting away from the large markets in North America and Western Europe, supply remains relatively concentrated in these regions. This is reflected in the significant capacity surpluses in these two regions. Asia-Pacific also shows a surplus (see Table 2.4).

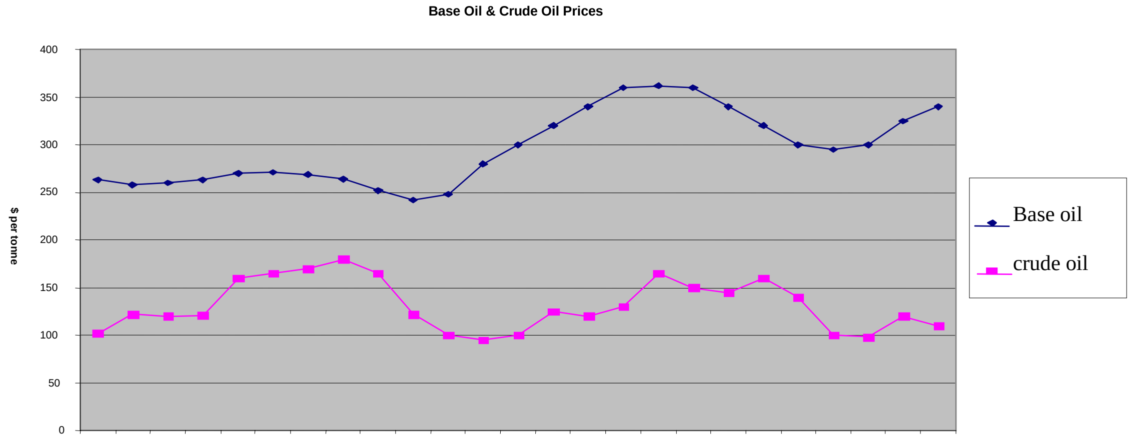
Table 2.4. Capacity Surplus/Deficit by Region

Region	Capacity Surplus/Deficit (-) 000's Tonnes
N America	2,410
Latin America	-535
Western Europe	2,405
Central/Eastern Europe	445
Near / Middle East	-230
Africa	-810
Asia Pacific	1,280
Marine	-1,585
WORLD (inc Marine Oils)	3,380

Source: FUCHS Petrolub AG Manheim - Paper to 7th ICIS LOR base oils conference 2003

ICIS LOR (Independent Commodity Information Services-London Oil Reports) prices since 1997 show the degree of correlation between crude oil prices and those of base oil. In broad terms, the refining process adds 2.6 to 3 times the average value of a barrel of crude oil whereas gasoline or diesel adds approximately 1.5 times. Given the high degree of value added in the production of base oil, it is not surprising that the correlation is not particularly strong (see Figure 2.2).

Figure 2.2. Base Oil & Crude Oil Prices (1997-2003)



But what is of interest is the variation of crude oil prices: because recycled lubricating oil can fully substitute for the product derived from virgin oil, the volatility of crude oil prices as well as high prices can benefit to markets for recycled lubricants by favouring the demand for such materials.

The market is relatively concentrated. Shell, Exxon Mobil and BP Amoco are market leaders in the world lubricant market with approximately 34% of the market. The remainder is shared between 13 companies such as LukOil, Nippon Oil, Conoco Phillips and PetroChina Sinopec.

2.3. Waste Oil and Used Oil

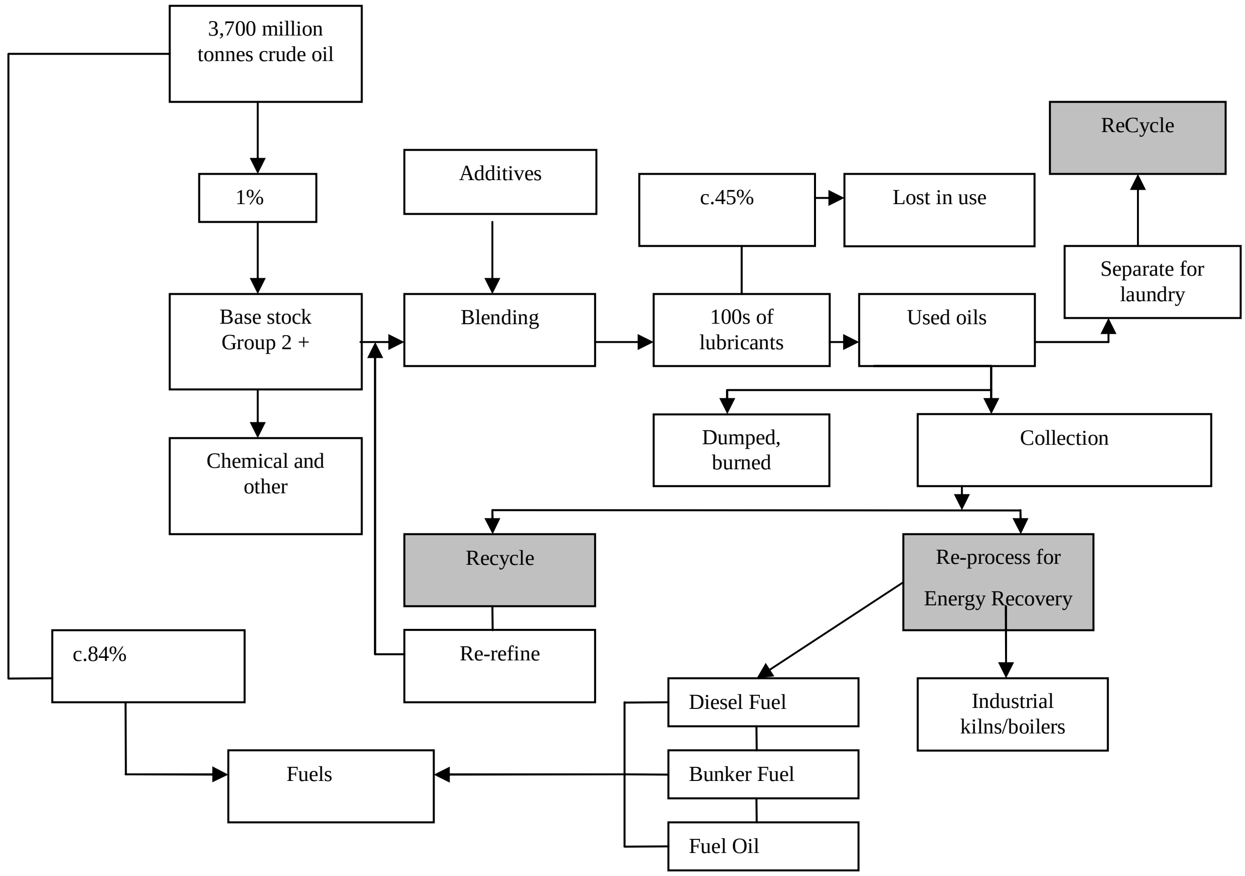
Approximately 40 to 50% of the lubricants sold are being lost through leaks or in the exhaust emissions during use: the fate of the remaining 60% to 50% of the oil is potentially recoverable (European Environment Agency, 2002).[28] Following a period of use that is typically less than 6 months, remaining lubricant is recovered along with various combustion, friction and heat-related contaminants. This oil - sometimes mixed with other contaminants such as water, solvents, antifreeze, brake fluid, paint and fuels – can enter the waste management market.

There is however a wide reported variation between similar economies in the amount of waste oil collected compared to the amount of lubricant product sold. In addition, potential recovery rates are likely to differ due to differences in use patterns. For instance, while recoverability is estimated to be as much as 90% for gear oils and 80% for aviation oils, the figures for brake fluids, process oils and greases are 0%

28. Although the GEIR (Groupement Européen de l’Industrie de la Régénération) estimates the percentage of recoverable oil in the EU at 46%, ranging from 32% (Italy) to 65% (Finland). www.fedichem.be/en/geir/figures.htm

(AATSE 2004). In addition, many rolling oils, quench oils, and other process oils are either incorporated into products or else they are lost during production.[29]

Figure 2.3 Summary Life Cycle of Lubricants



3. Environmental Impacts from Waste Oil Management

The environmental impacts associated with lubricating oil arise from extraction, processing, use and disposal. In this section some of the main environmental impacts are reviewed. Given the nature of the study, the focus is on impacts associated with collection and disposal (legal or illegal), and recovery of waste oil.

3.1. Post-collection environmental impacts

Before or during collection from multiple-point sources, waste oil can be contaminated from either in-use sources or materials introduced through mixing. As noted above, in-use additives include a variety of compounds added during manufacture to meet performance specifications. In addition, waste oil is also likely to contain metals from engine wear; unburned fuel; PAH (polyaromatic hydrocarbons) from

29. In addition, some waste oils are generated which have not been 'used'. This includes oil wastes such as fuel storage tank bottoms. In some cases these may be 'laundered' for limited types of re-use without undergoing re-refining.

polymerisation and incomplete combustion of fuel; particulates and water. It may also contain - from mixing with other wastes - solvents from parts-cleaning in garage workshops, paints, brake fluids and non mineral-oil based lubricants such as biodegradable vegetable oils. In most countries, waste oil is considered “hazardous waste”.

The nature and extent of in-use contamination in waste oil is changing. In the 1970's, lead and chlorine were present in significant concentrations but are now declining due to the phasing out of lead additives in vehicle fuels and the tighter restrictions on the use of chlorine as an additive in many industrial lubricants. On the other hand, the presence of esters and vegetable oils is expected to increase, but remains negligible (Boyd 2002).

The removal of these contaminants from waste oil through a process of re-refining is complex. Once removed, the contaminants, in the form of a thick black and malodorous sludge, the asphalt bottoms, are usually incinerated in a cement kiln or similar energy-intensive facilities or used to create roofing materials, as in Canada and the United States. This contaminated sludge represents 15% to 20% of the weight of waste oil delivered to re-refining plants.

If waste oil is burned in small-scale space heaters, the environmental impacts can be similar to those calculated for oil-fired power stations, i.e. emissions of atmospheric pollutants derived largely from metal contamination in the waste oil in addition to typical pollutants from fuel oil incineration (CO_2, CO, NO_x, and SO_2). Use of waste oil for energy recovery in unregulated markets can have much more significant environmental impacts. For instance, in Australia there are concerns that the use of recovered oil in the hydroponics and flower industries may produce toxic emissions, including polychlorodioxins (AATSE 2004). It has been estimated that this use may represent as much as 25 ML per annum, although there is considerable uncertainty.

A comprehensive and critical review of several LCA (life cycle assessment) studies was published by the EU in 2002. The study, prepared by TN Sofres, summarised the findings of four major LCA studies as follows:

- "For almost all environmental impacts considered, incineration in cement kilns (where waste oil replaces fossil fuels) is more favourable than incineration in an asphalt kiln (where waste oil replaces gas oil);
- A modern regeneration[30] plant may, according to the impact considered, be more favourable than or equivalent to incineration in an asphalt kiln; and,
- Compared to incineration in a cement kiln (where waste oil replaces fossil fuels), waste oil regeneration has environmental advantages and drawbacks depending on the impact considered." [31]

Twelve environmental impact criteria were used to evaluate eight comparative studies carried out in Norway, France and Germany since 1995. The study highlights that several environmental impacts were excluded from some or all of the LCA studies. Specifically: noise, odour, biodiversity, land use, toxic emissions and the displacement of non-fossil fuels by waste oils.[32] Significantly, the LCA studies do not provide firm evidence to justify support for either re-refining or direct burning in cement kilns (see Table 2.5). This view is largely supported by a recent review undertaken for the Australian Department of Environment and Heritage (AATSE 2004).

30. Regeneration can be read as being identical to re-refining in this paper.

31. One of 10 summary points made in the study.

32. The study goes on to point out that there is a lack of data to evaluate some of these excluded factors.

Table 2.5. Results from Comparative Studies of Incineration and Various Re-Refining Technologies[33]

	UBA (2000)	ADEME (2000)	ADEME (2000)	ADEME (2000)	N - EPA (1995)	ADEME (2000)	ADEME (2000)	ADEME (2000)
Total energy	R=I	R=I	R=I	R=I		R=I	R=I	R=I
Fossil fuels	R	I	I	I	R	R=I	R=I	R=I
Water inputs	R=I	R	R	R		R	R	R
Climate change	I	I	I	I	R	I	I	R=I
Acidifying potential	R	I	I	R=I	R	R	I	R
VOC	R	I	I	I	R	I	I	I
Dust to air	R	I	I	I		I	I	R=I
Human toxicity	R	I		R		R		R
Ecotoxicity (aquatic)		I		R=I				R
Ecotoxicity (terrestrial)		I		R=I		I		R
Eutrophication (water)	R=I	R=I	R=I	R=I	R=I	R=I	R=I	R=I
Solid waste					R=I	R=I	R=I	R=I

Source: TN Sofres Critical Review of Existing Studies and LCA on the Regeneration and Incineration of WO (December 2001) 20 AW 83-5 Page 115. The different ADEME study results refer to differing assumptions concerning the parameters (e.g. substitute fuel for burning).

The results from environmental impact studies are especially sensitive to assumptions made concerning: the type of re-refining technology; the type of fuel that is being displaced by the burning of waste oil; and, the scale and type of incineration plant. Studies undertaken by ADEME during 2000 concluded that the latest re-refining technologies produce fewer environmental impacts than older technologies using clay[34]. In general, the studies indicate that the comparative environmental benefits are finely balanced between incineration in cement kilns and re-refining. However, it should be noted that the waste oil re-refining process to lubricants emits less atmospheric pollutants and consumes less energy (50% to 85% less) than the refining of virgin oil into lubricants.

In a study undertaken in California, Boughton and Horvath (2004) compare the impacts from re-refining used lubricating oil back into base oil for reuse, distillation into marine diesel oil and burning as untreated fuel oil for space heating. They find that the impacts with respect to air and water pollution emissions and generation of solid waste are approximately equal, but that emissions of heavy metals are much more severe for burning. However, they assume that burning is undertaken for space heating, with zero emission controls.

In summary, the present LCA studies are not conclusive about the best management option for waste oils in terms of environmental impacts: re-refining or incineration in industrial furnaces provided with the appropriate pollution control device. On the other hand, it can surely be claimed that production of lubricants from the re-refining process is more environmentally friendly than direct burning in individual small space-heaters or in unregulated industrial furnaces using old technologies. It may also be more

33. "R=I" represents equivalent impact between regeneration and incineration
"I" represents incineration having a lesser impact
"R" represents regeneration having a lesser impact
N-EPA is the Norwegian Environmental Protection Agency
UBA : Umwelt Bundes Amt (Germany)
ADEME : Agence de l'Environnement et de la Maîtrise de l'Énergie (France)

34. The UOP (HYLUBE) process was compared with older distillation processes that require clay to finish the base oil output.

environmentally-friendly than production of lubricants from virgin oil, but the aforementioned LCA studies do not make such an assessment.

3.2. Illegal Disposal

The most significant environmental impacts associated with lubricating oils almost certainly arise from illegal disposal. The extent of deliberate dumping in the US was highlighted in a survey by the Bureau of Transportation Statistics in 2002. It concluded that 5% of people who change their own motor oil dump the recovered oil in the drainage system, pour it on the soil, or place it in the municipal waste bin. The Bureau estimates that 43 million US residents change their own motor oil (BTS 2002). A recent survey commissioned by the Australian Government found that one in three households have motorists who change their own vehicle oil, and that nearly half of those people inappropriately dispose, store or reuse the oil: uncollected waste oil has been estimated to be in the region of 40-65 million litres (AATSE 2004). Environmental Choice New Zealand estimated that in 1998 of the 30 million litres of lubricating oil requiring disposal in New Zealand, as much as 23 million litres may not have been disposed in an environmentally-sound manner (ECNZ 1998). Similarly, as much as 40 million litres of waste oil in Western Canada is unaccounted for by the collection data and estimated in-use losses (CLMC 2002).

Thus, significant additional quantities of used lubricant are deliberately spilled on the ground or in surface waters. The extent of these illegal activities can be estimated by comparing the amount of waste oil collected with the amount that is potentially recoverable. In most cases this will be between 50% and 60% of the amount of lubricant sold. Studies of collection effectiveness show a wide variation in performance (see Table 2.6).

Table 2.6. Waste Oil Collected Compared To Lubricant Sales (1997)

Country	Lubricant sold (1997)	Waste oil collected	% Collected
Denmark	85 000	36 337	43
Finland	97 000	47 000	48
France	900 000	242 000	27
Germany	1 159 000	485 000	42
Italy	713 000	172,000	24
Spain	451 000	110 000	24
UK	872 378	422 000	48

Source: Collection data from Report to Council and European Parliament "Implementation of Community Waste Legislation 1995 - 1997"

There are three main reasons for doubting the accuracy of the collection data.

- At the point of collection, waste oil may contain significant quantities of water and other contaminants. In the case of bilge oil and emulsions, 20% or more of the weight of waste oil may be water. Oily waters are a common type of industrial waste.

- Where waste oil is heated as rudimentary treatment process, waste oil is used as a fuel source and thereby may not be included in the data.

- Waste oil collectors also collect confiscated fuels and these may be included in waste oil data.

Calculating collection rates on the basis of potentially collectable oil (differentiated by country), the GEIR comes up with the following figures for 2002 (see Table 2.7). For instance, on this basis the figure for the United Kingdom is 88%. Since the calculation of 'collectable' is necessarily fraught with difficulties there is no straightforward manner to resolve this discrepancy. Calculating collection rates directly from consumption yields a rate of 42%, not out of line with the figure given above (48%).

Table 2.7. Tonnes of Waste Oil Consumed, Collectable and Collected (2002)

	Consumption	‘Collectable’ Oil	Oil Collected	% Total Oil Collected	% 'Collectable' Oil Collected
Austria	109000	53622	33500	31%	62%
Belgium	173100	63105	60000	35%	95%
Denmark	71718	46909	35000	49%	75%
Finland	88809	49596	39677	45%	80%
France	841356	422197	242500	29%	57%
Germany	1032361	463304	460000	45%	99%
Greece	87800	40161	22000	25%	55%
Ireland	39800	17794	15303	38%	86%
Italy	617594	196737	189595	31%	96%
Luxembourg	10170	4652	4564	45%	98%
Netherlands	152694	66468	60000	39%	90%
Portugal	102000	52842	39620	39%	75%
Spain	510980	255236	160000	31%	63%
Sweden	142814	77232	61786	43%	80%
U.K.	840834	401474	352500	42%	88%
EU	4820130	2211329	1776044	37%	80%

www.geir-regeneration.org

Policy measures can have a significant influence on illegal disposal rates. Using American data, Sigman (1998) estimated the extent to which oil dumping incidents increased in response to changes in the costs of collection and disposal caused by tighter regulations. For instance, it was found that the imposition of a ban on landfilling increased incidents of illegal dumping by 28%. Although the correlation between dumping incidents and costs will vary between countries, the environmental impact of any increase could be significant due largely to the persistence of mineral oil in the environment.

The environmental impacts associated with illegal disposal are likely to be similar to those arising from the discharge of other oil products. These include:

- Harm to wildlife through depletion of the oxygen supply for fish and other aquatic life;
- Aesthetic and recreational impacts through contamination of surface waters;
- Loss of soil productivity and contamination of groundwater; and,
- Human health impacts associated with the contaminants commonly found in waste oils.

For all these reasons more and more countries, are trying to maximise the collection rate of waste oils by making the collection a legal obligation or by providing financial incentives. Different policy instruments have been put in place throughout the OECD to finance the collection costs. They will be analysed at a later point in the report.

3.3. Conclusion

An important conclusion to draw concerning the types of environmental risks described in this section is that those arising before collection, such as illegal dumping, are considerably greater when compared to the net increase in the risk of environmental damages from preferring any one recycling route over another. Consequently, if the policy objective is to minimise the risk of environmental damage, maximising the collection of the recoverable proportion of lubricants should be a priority. In the case of waste oil, the threat presented to water resources by just a few litres demands attention.

The oil industry group Concawe concluded their 1995 study by recommending a policy focus on collection efficiency.[35] While they may have a vested interest in reaching this conclusion, there is little question that collection is key to reducing environmental impacts. However, at the post-collection stage, there is considerable debate on the best environmental waste management option: material recycling (i.e. re-refining to lubricants) versus energy recovery. Direct burning in unregulated small space-heaters or small industrial boilers and furnaces without any pollution control devices has been clearly identified as being the worst environmental option, to be rejected or even forbidden.

Member countries are more and more concerned about environmental problems and remediation costs related to waste oils (including used lubricating oils) and are trying to solve them by developing various recycling policies and regulations. However, in many cases those policies are facing difficulties. The markets which are needed to support the different options are not operating as efficiently as required. In some cases this may be due to the presence of non-environmental market failures and barriers.

4. Market Failures and Barriers

Clearly the most important determinant of market participants to adopt one waste management option over another is financial cost. Some routes are more costly than others due to the importance of different factor inputs required. Public policy measures can play an important role in determining these factor input costs, whether implicitly or explicitly. For instance, a landfill tax will clearly have an impact on waste generation costs if the effects of the tax are transmitted back to the generator through waste fees. Analogously, technology standards for landfill sites will affect the costs of waste management, and thus costs for generators.

Costs can also be significantly affected by the efficiency of the conditions in which the market operates, and thus any failures or barriers which may exist in such markets. Since the market for the collection and processing of waste oil for sale as an industrial fuel appears to work efficiently,[36] we are concerned here only with barriers confronted by those seeking to re-refine waste oil as to base oil for non-energy uses, such as lubricants. Investors in this market confront several market failures and barriers: information failures relating to waste quality; risk aversion to using re-refined base oils, technological externalities associated with lubricating oil specifications, and the market power of virgin product incumbents.

4.1. Information failures relating to recovered oil quality

Re-refining plants are more sensitive to contaminants in the waste oil pool than all other competing uses. While tankers tend to have separate compartments to allow for segregation of waste oils of different

35. CONCAWE report 5/96 concluded "It is therefore more important in the first instance to ensure that this oil is collected and safely disposed of than to specify any particular disposal option."

36. This is not to say that there may not be important policy failures (i.e. non-internalisation of externalities) or tax distortions, a point to which we return below.

qualities, collectors (buyers) of waste oil can never be certain that the oil is free of these and other liquid contaminants to which re-refining is sensitive. Unless a comprehensive mobile testing programme is in place, the cost of concealment is negligible and the cost of detection substantial since most waste oil collected is immediately mixed with other oil in a road tanker, and the risk of those responsible for contamination being identified is small[37]. Even disposers (sellers) may not be certain of the quality since waste oil stores are sometimes less than secure (Allen Consulting Group 2004).

The impact of this asymmetry is revealed through the costs of testing undertaken in Italy before waste oil is accepted into the re-refining process. Tests are carried out before waste oil is placed in the 26 regional storage tanks and again on acceptance at the re-refining plant. These costs are estimated by the Italian waste oil consortium to be €45 - €60 per tonne of collected waste oil.[38] Conversely, where waste oils are taken for direct burning in a cement kiln, the testing protocols are less costly since the acceptance criteria at cement kilns are less sensitive than those required by re-refiners.

As such, in the absence of effective control of waste oil quality at the collection stage a disposer of waste oil has an incentive to place contaminated oil on the market. For instance, excessive water content will lower the value of the waste oil and if sellers believe that this will not be detected, this can result in significant market inefficiency. This problem is particularly important if the waste oil is contaminated with excessive levels of metals, PCB's or halogens. The diffuse nature of waste oil generation – and thus the mixing of a variety of sources in collection strategies in order to reduce overall collection costs - makes detection difficult. Moreover, "reputation effects" may be negligible. As such, the cost of discovery falls most especially on re-refiners who are forced to direct a greater proportion of their oil feedstock to incineration than would otherwise have been necessary.

4.2. Risk aversion to using re-refined base oils

As noted above, buyers of base oils typically blend base oils with additive packages to achieve a consistent product that delivers specified performance attributes. These buyers are highly sensitive to the risk of using materials that may cause their own products to fail in use or for their own production process to be stopped as additive packages fall out of suspension. Moreover, lubricants offer performance-critical attributes for a value that is typically insignificant when compared to the equipment motor vehicle or industrial equipment) it is protecting.

Consequently, a significant degree of "disappointment aversion" (see Gul 1991 and Grant et al 2000) best describes the conservative approach of buyers in the market for base oils. Downstream, buyers of lubricants are reluctant to risk damaging their vehicle engine or industrial equipment by purchasing re-refined oils, whereas the quality of re-refined lubricants is equivalent and even said better than primary lubricants. This undercuts the market. This risk-aversion can persist for a considerable length of time, even if a strong technical case can be made for the equivalent quality attributes of re-refined oil, relative to lubricants manufactured from base oils derived from virgin crude.

Thus, re-refiners, who are almost always independent of oil refiners and lubricant manufacturers, sell their base oil and with it the negative association of waste. Risk aversion amongst buyers causes re-refined base oil to be directed into lower value markets. Typically, re-refined base oil attracts a price 20% to 25% below that which might have been expected for a product with similar viscosity and related features supplied by crude oil refiners. For instance, in Australia the base oil prices for a re-refined product are 10-12 cents per litre lower than for a virgin base oil of comparable specifications (see AATSE 2004).

37. Although at least some re-refiners confirm that they receive very few 'bad batches' (less than 1 in 500), and can identify the culprits with relative ease (Mr. J. Vercheval, GEIR, personal communication).

38. This represents less than 1% of the retail purchase price of standard lubricating oil for car engines.

Moreover, this is reinforced by industry standards. The American Petroleum Institute restricts the use of “alternate” base oils (including re-refined) to 10% of lubricating oil production input. Such intense conservatism in the market for lubricating oils for passenger cars is less a feature of markets for products such as gear oil, transformer oil, hydraulic oil and some engine oils. For some of these products, the additive packages are simple to blend. In some cases, re-refiners have invested in blending plants to enter these markets[39].

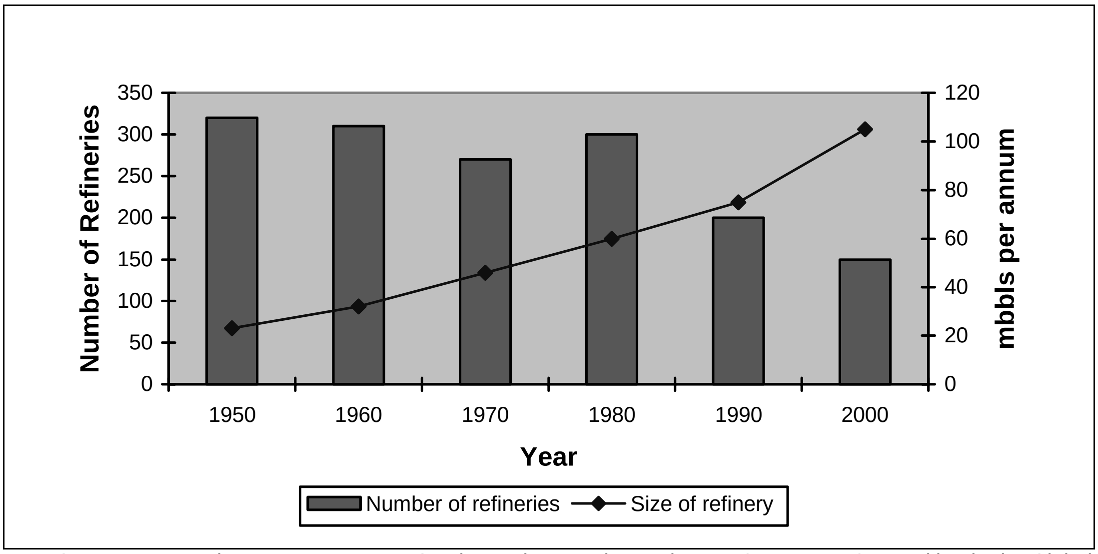
This risk-aversion on the demand side in some markets is reinforced by the strategies pursued by many suppliers. While lubricant demand in most economies is static or declining in volume terms, it is increasing in value terms. Market incumbents seek to signal the quality of their products to their customers by adding brand names such as "Mobil 1" and meeting international and OEM (Original Equipment Manufacturers) product standards. Although it is technically possible to manufacture the branded and highly-specified lubricant products from a base oil manufactured from re-refined base oils, buyers avoid doing so for fear that competitors will signal this to lubricant users (see AATSE 2004).

In an attempt to distance re-refined base oils from the association with poor quality, the Viscolube plant in Italy and the Puralube plant in Germany recently began shipping high quality oils manufactured from waste oil. The quality of the base oil will be superior to that produced in the majority of European refineries. It remains to be seen whether this strategy of exceeding expectations of technical standards amongst buyers is effective.

4.3. Switching costs and barriers to entry

Incumbents seek to protect their market position and since volume growth in the lubricant market is small, it is customers of the incumbents who must switch to the entrant. The minimum efficient scale of 60,000 tonnes for a re-refining plant presents a major challenge in such a market. Once production starts, the existing market will need to accommodate the new source of supply. The existing vendors are typically large integrated oil companies who can respond to the new competition from re-refiners. Moreover, the scale of such refineries is growing (see Figure 2.4).

Figure 2.4. Rationalisation of US Refineries



Source: Suzan Jagger, Director, Downstream Services, The Petroleum Finance Company. "Competition in the Global Base Oil Business". The Seventh ICIS-LOR World Base Oils Conference. 20th & 21st February 2003, London.

39. Dollbergen in Germany did this during the 1980's and early 1990's.

Faced by competition from virgin oil refiners, re-refiners are susceptible to a short-term change in pricing policy from such large-scale incumbents. A reduction in the price of base oil for 12 months may be sufficient to undermine the profitability of any new re-refining plant. This is, of course, a risk that investors will consider and value in the cost of capital. (See Allen Consulting Group 2004 report for discussion of uncertainty and risk aversion.)

Moreover, before the re-refinery is able to sell re-refined base oil to the engine oils market, a sector that forms approximately 50% of lubricant demand, the oil must meet international quality specifications (i.e. Society of Automotive Engineers or American Petroleum Institute specifications). Testing costs are typically $500,000 and since the scale of re-refining plants is less than 60,000 tonnes output annually, these costs can form a significant barrier to investment, and thus market entry.

The re-refiner must meet the testing costs without certainty of success in selling the output to a mature and risk-averse market. The product tests cannot be carried out until the re-refining plant has been made operational. Furthermore, as the tests require at least several months to complete, the base oil output will have to be sold to alternative markets. This can increase the opportunity cost of testing.

Frequent changes to engine oil standards make it more difficult for re-refiners to enter this market. However, API (American Petroleum Institute) base oil inter-changeability guidelines permit up to 10% of base oil to be substituted without requiring a re-test. This offers re-refiners an opportunity to sell to independently-owned blenders. In practice, this market is constrained by the risk that competitors will inform customers that waste oil sources are being used in certain branded lubricant products.

4.4. Technological externalities

As noted above, a wide variety of additives are added to base oils to produce lubricating oils. All of these additives provide benefits in terms of performance but can create difficulties for re-refiners. Technological externalities can reduce economic efficiency if the benefits (improved performance) are not as great as the costs (reduced recoverability). This can arise if incentives are not provided to ensure that the product designer does not take the downstream costs into account.

Some of the most problematic additives for re-refiners are chlorinated hydrocarbons and dithiocarbamates (containing lead) used as extreme pressure agents, polysulphides and several sulphur containing compounds that can cause emissions. Additives such as styrene butadiene (SBR) and styrene isoprene (SIP) used in engine oils to provide shear stability appear in the waste from re-refining as asphalt or as a substitute for heavy fuel oils used in cement kilns.

If the costs associated with re-refining are not transmitted back up the product chain, it is possible that sub-optimal specifications for lubricating oil will be developed. As noted above, a technological externality results in sub-optimal design if the performance advantages associated with the additive are of smaller positive value than the negative values associated with reduced potential for re-refining. Indeed, it has been argued that minimum quality standards can be redrafted with a view toward increasing the potential for re-refining (AATSE 2004).

Technological externalities caused by the use of certain lubricant additives and new lubricant products may be best addressed through restrictions on their use. For instance, a restriction on the use of chlorinated additives has been especially significant. However, continued use in certain, mostly high-pressure applications, continues to present a diminishing contamination risk for reprocessors.

5. Policy Responses

The policy responses in the countries reviewed can be broadly divided into two categories; those targeted primarily at collection efficiency and those designed to address the multiple market barriers to re-refining waste oil. While all countries have sought to improve collection rates by establishing some form of free collection system and regulating the management of the collected waste oil, there appears to be significant differences in the effectiveness of these systems. Similarly, the support provided to re-refining relative to reuse of waste oils as fuel differs markedly.

5.1. United Kingdom

A Government statement from 1985 describes the UK position on post collection uses.

"The Government favours the regeneration of waste oil as lubricant wherever practicable but sees no reason, environmental or otherwise, to discriminate against the use of base oil as supplementary fuel...It is highly questionable whether regeneration is always the most rational way of re-using waste oil, and the decision as to whether to regenerate as lubricant or to use as fuel is best left to the operation of market forces"

William Waldegrave
Parliamentary Under Secretary of State, Department of Environment 2nd April 1985

Although the policy is supposed to be indifferent between post-collection uses, the use of waste oil as a substitute for fuel oil is supported by derogation in excise duty on the waste-derived fuel product. At 2002/03 prices this is worth the equivalent of £27.40 per tonne, or approximately 15% of the delivered product price. This has the effect of guaranteeing a positive value for most waste oils at the point of collection and favouring the demand for recovered fuel oil to the extent that the United Kingdom is importing about 100,000 tonnes (in 2001) of waste oils from Europe and elsewhere.

The costs of managing pollution from waste oil, collecting oil at public facilities without charge and for the ‘Oil Care’ campaign are met from general taxation and sponsorship from retailers and oil companies. Lubricant carries a sales tax of 17.5% recoverable by almost all commercial and industrial users but not by those buying oil for personal use.

Table 2.8. Basic Data on UK Lubricating Oil

Measure	
Lubricant Sold (a)	803 667 tonnes
Waste oil Collected (b)	380 000 tonnes
Collection Rate (b/a * 100)	47%
Recycled as a Base Oil	0%
Other Recycling/Reuse	0%
Direct Burning /Energy Recovery	100%

Source: Defra (2001)

There is competition to collect and treat waste oil. Attempts to operate a re-refining plant to produce new lubricants and a merchant laundry operation in this market ended in closure. The operations were not viable, in part because they confronted the market barriers detailed in the previous section. Policy makers have not actively intervened to try and support the market for re-refining to lubricants.

5.2. Italy

In contrast to the UK, long-term enthusiasm for the re-refining of waste oil has led policy makers to support the manufacture of base oil from waste oil sources. From July 2003, new regulations were introduced to reform a system of subsidies paid to re-refiners and collectors of waste oils

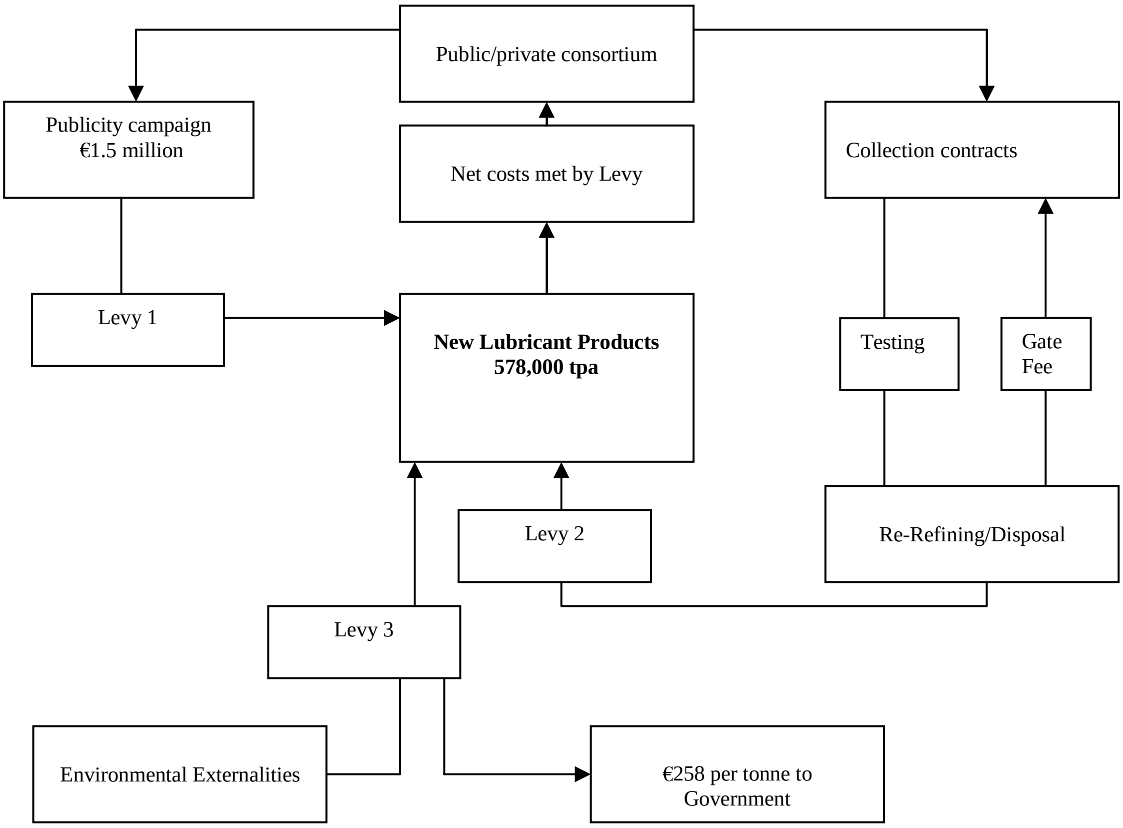
A statutory consortium consisting of representatives from the manufacturers of lubricants, waste oil collectors and re-refiners as well as public regulators manages the contractual arrangements for waste oil collection and processing in Italy. It also manages the equivalent of the UK Oil Care campaign, albeit with an annual publicity and promotion budget of €1.5 million, significantly greater than the equivalent UK scheme.

All lubricants attract a three-part variable levy of €325 per tonne. Of this variable levy, €67 is to pay a subsidy to operators of certain re-refining and fuel manufacturing plants. The average cost of subsidy per tonne of waste oil collected in 2002 was €166.50 or approximately €31 million. Base oil output will receive a subsidy more than twice that of fuel products, although the final ratio has not yet been agreed.

The consortium requires that the 85 contracted collectors deliver a minimum of 90% of the collected waste oil to re-refiners and that not more than 10% is delivered to cement kilns for energy recovery. The contracts offered by the consortium are arranged over three regions, north, central and south. The waste oil can be stored in 26 registered storage tanks before delivery. Cement kilns pay a market based price that is currently €30 per tonne for each tonne delivered and re-refiners €50 per tonne. (The structure of the policy is set out in Figure 2.5.)

The collection of waste oil is free of charge, regardless of location and amount. Testing of the waste oil prior to collection and again at the 26 registered storage tanks is required and ensures that the least acceptable waste oils are diverted to less sensitive cement kilns and other fuel uses. Testing costs are met from the variable levy on lubricant products. The implied price signal in this system is that a free collection service is available for qualifying waste oils. Those oils that fail to meet the quality criteria fall outside the system and should be collected by independent (non-consortium) businesses at a cost. Consequently, the collection performance of the Italian system appears to be inferior to that in other countries, notably the UK and Germany (see Table 2.9).

Figure 2.5. The Structure of Lubricating Oil Policy in Italy



Source: Used Oil Management Consortium (2002).

Table 2.9. Waste oil Collection in Italy

Year	Waste Oil Collected 000 tonnes	Lubricant Sold 000 tonnes	Collection Rate
1997	172	625	28%
1998	177	636	28%
1999	183	634	29%
2000	183	648	28%
2001	192	605	33%
2002	190	578	32%

Source: Consortium Data 2002

Given the support provided to re-refining it is not surprising that there is a relatively active industry with six plants, with capacities ranging from 9,000 to 100,000 tonnes per annum. However, it is significant that output is considerably below capacity for many of the plants (see Table 2.10.)

Table 2.10. Base Oil Manufactured From Waste oil in Tonnes in 2002

Operator	Location	Nameplate Capacity	Base Oil Output 2002
Viscolube	Ceccano	84,000	38,000
Viscolube	Pieve Fissiranga	100,000	48,000
SIRO	Milan	9,000	5,200
RAM Oil	Naples	35,000	7,900
DISTOMS	Porto Torres	18,000	4,300
ECENER	Ravenanusa	15,000	Nil
TOTAL		261,000	103,400

TFD = Thin Film Distillation. Source: Industry Interviews 2003

The outcomes of the policy are summarised in Table 2.11. Re-refining rates are high relative to most other countries, but the collection rates are relatively low.

Table 2.11. Summary of Programme Outcomes in Italy

Measure		Comment
Lubricant Sold (a)	578,000 tonnes	
Waste oil Collected (b)	190,000 tonnes	Consortium data only - other waste oils are collected outside this system
Collection Rate (a/b *100)	33%	Excludes the non-consortium collection for direct burning
Recycled as a Base Oil	54%	Output of base oil compared to (b)
Other Recycling/Reuse	36%	Various fuels and water
Direct Burning /Energy Recovery	10%	To cement kilns

Indirectly, the policy may have supported the market for re-refined oils on the demand side. When AGIP was a state-owned oil company, it was possible for the Government to require the company to integrate with Viscolube to overcome the multiple barriers to marketing base oil manufactured from waste oil. Although AGIP is only one of several customers for re-refined base oil, the availability of this marketing route allows re-refined base oil to lose its separate identity. AGIP and its many related companies are able to add their brand names to guarantee buyers that the base oil and lubricants meet the quality standards expected by buyers.

In addition, the inclusion of lubricant manufacturers on the consortium encourages a flow of information between manufacturers and re-refiners. This allows the waste management implications from the introduction of new additives or products to be balanced with the performance advantages expected from the new product. International trade in lubricant products and the exclusion of companies such as Cargill who manufacture vegetable-based biodegradable oils undermines the value of such an information exchange in this forum.

However, some features of the programme – i.e. limiting deliveries of waste oil to cement kilns to 10% and excluding many sources of waste oil - may be inhibiting collection rates since this would have the effect of reducing the potential scope for a use of waste oil which is financially viable. Since non-consortium oil is collected at a cost, this may have perverse environmental consequences.

80% of the levy on lubricants is paid to Government for environmental enforcement and clean up costs. Moreover, whether or not it is equitable to impose an additional €325 per tonne price on all lubricant users when recovery rates for different types of lubricant vary considerably is questionable. It is further complicated by the differential rates of excise duty on oil products. Two-stroke engine oil for example may qualify as a lubricant but it is effectively used as a fuel.

5.3. Australia

"The Commonwealth will fund the development of a comprehensive product stewardship arrangement and provide transitional assistance to ensure the environmentally sustainable management and re-refining of waste oil and its reuse. It will support economic recycling options and the development of stewardship arrangements. Any diesel extenders or other products manufactured from recycled waste oil will be required to meet the relevant Commonwealth environmental standards"

Prime Minister's Statement 31st May 1999

Australian policy (the Product Stewardship Arrangements for Waste Oil) might be categorised as being somewhere between that of Italy and the UK. On the one hand it is focused on improving collection efficiency by using funds generated from an advanced disposal fee (a product stewardship levy) on lubricants to increase the number of free collection points. However, the funds generated are returned in part through a benefit scheme, for which benefit rates for recovered oil used for re-refined lubricating oil are 10 times that received for heavy grade burning oil, and 7 times that for diesel fuel (AATSE 2004). (See Table 2.12.) Such a difference can hardly be explained by differences in the externalities mitigated (AATSE 2004). Significantly, in the case of high-quality re-refined base oil the sales price is only approximately 10% greater than the subsidy provided.

Table 2.12. Subsidies for Different Oil Products in Australia

	Category	Subsidy (cents/litre)
1	Re-refined base oil achieving the highest quality standard	50
2	Re-refined base oil for chain bar oil etc	10
3	Diesel Fuels	7
4	Diesel Extenders	5
5	High Grade industrial burning oils	5
6	Low Grade industrial burning oils	3
7	Industrial process oils that are laundered	0

Source: Environment Australia PSA Benefit Rates 2002/03

In practice, the Australian system has excluded certain lubricants from the levy payment on the grounds that the lubricants are unavailable for recycling and pose little risk to the environment. In addition, an effective exemption through the provision of a benefit payment for multi-use oils that cannot be recycled and pose little risk to the environment, has recently been provided.

The system does encourage the development of markets for waste oil through subsidies (funded by the levy on new oil). Significant subsidies ($34.5 million) are provided to a range of schemes. The majority is expected to go toward support for collection schemes. By the end of 2003 approximately $6 million had gone toward collection infrastructure (AATSE 2004). However, the single largest grant ($1.3 million), excluding those made to State governments, was recently disbursed for the capital costs of a re-refining plant (AATSE 2004). Other grants have gone toward "waste oil market development" projects. An Oil Stewardship Advisory Council including representatives from oil producers, waste management companies, trade associations, NGOs and public authorities advise on the management of the fund.

Prior to the introduction of the programme, waste oil was collected free of charge by commercial businesses except in rural areas where a modest fee of between 8 and 12 cents per litre was charged. Public provision of collection facilities was poor. Since the programme has been introduced, collection rates have risen from 165 ML per year prior to the introduction of the programme to 220 ML in 2002-2003. Estimated collection rates are now 81% of potentially recoverable oil (AATSE 2004).

The effects on post-collection practices are less apparent. Prior to 2000, approximately 75% of collected waste oil was used as a burner fuel in a variety of industrial furnaces including cement and lime kilns, 20% was used to manufacture a high sulphur diesel fuel and a small fraction was re-refined. Re-refining remains very low (3 ML in 2002-2003) and 85% of the oil recycled is still for energy recovery. However, it is expected to grow with the new subsidised plant at Wagga Wagga, which has a projected capacity of 15 ML (AATSE 2004).

It should be noted that the Australian policy for waste oil management recently changed direction to improve the collection rate instead of financially support re-refiners. The wisdom of this change in direction has been supported by recent reports (AATSE 2004 and Allen Consulting Group 2004). Indeed, recent figures “suggest that up to 100 million litres of waste oil go missing each year”, said Dr Kemp, the Federal Minister for the Environment and Heritage in May 2004. An extensive infrastructure of 300 new collection sites (in addition to the 400 existing ones) will be developed by the end of 2004 through a fund of 10 million Australian $ provided by the Australian government. The development of more free collection facilities, especially in rural areas, should increase the proportion of waste oil that is collected. However, the mixed oil quality may limit the amount of waste oil that can be directed to re-refining.

5.4. Alberta, Canada

A comprehensive description of the Alberta EPR system was prepared for the OECD in December 2002 (CMCL 2002). The system imposes an ADF (the Environmental Handling Charge) on most lubricant and oil filter products at point of sale and distributes payments (Return Incentives) to collectors to provide an incentive to return waste oil, packaging and oil filters to almost 600 specified facilities. The policy is broadly revenue-neutral, and the programme, which has been developed and is directed by major oil refiners and filters manufacturers, does not receive any financial public support.

The focus is on increasing collection in order to ensure that waste oil is not disposed of inappropriately. There are no quantitative data available on how much of the waste oil collected is burned for energy recovery versus re-refined. While there is a recovery market, the waste oil goes mainly toward asphalt paving or burned for energy recovery. Moreover, since the return incentive is based strictly on volumes (discounted by water content) and the oil collected is not tested for contaminants, the potential for re-refining from the oil collected is limited.

The programme has been very successful with respect to collection and recovery for the two uses mentioned above. Based on an assumption that 65% of lubricant sold is recoverable, the scheme has achieved a recovery rate of 71%, equivalent to 46%% of lubricant sold. This is similar to the 47% achieved in the UK and 48% in Germany. The system is setting higher targets for future years.

The focus on collection reduces the greatest risk of environmental damage; that of deliberate or accidental spillage into water resources before collection. The system is much less concerned with distinctions between re-refining waste oil and recycling it as a fuel substitute or asphalt additive. The Alberta re-refiners complain about the fact that the Western Canadian EPR programme has not been designed to avoid contamination of collected waste oils, which severely restricts the re-refining potential.

Indeed, the funding of the collection of used lubricants without testing procedures has created an incentive for waste oils generators to blend used lubricants with “no value” hazardous recyclables such as parts washer solvents, glycol or unburned fuels. Such waste oils are thus more suitable for incineration rather than re-refining. The high demand for fuel by industry for energy purposes combined with imperfect enforcement of emissions regulations result in making the re-refining process less competitive. And finally, there is also some concern on the part of re-refiners that refiners have undue influence over the programme.

5.5. European Union

The Commission of the European Communities issued a Directive on Waste Oils in 1975[40], whose purpose is to protect the environment against the harmful effects of illegal dumping and of treatment operations by ensuring safe collection, storage, recovery and disposal of waste oils. It requires Member States to give priority to the regeneration of waste oils in preference to other disposal methods, regeneration being considered as the most energy efficient option. Otherwise, waste oils have to be burned under environmentally acceptable conditions.

However, the report on the implementation of the Directive on waste oils indicates that, in 1997, more than half of the EU countries do not regenerate any waste oils - or only a very small quantity - and that re-refining is decreasing relative to incineration in many of the countries where there is a market (France, Germany, Italy and the United Kingdom). Presently most of the EU countries have not complied with the Directive and the Commission has taken legal action against Austria, Belgium, Denmark, Finland, France, Germany, Ireland, the Netherlands, Greece, Sweden and the United Kingdom for non transposition of the EU Directive into their legislation.

One should say that the Directive itself states that, "where technical, economic, and organisational constraints so allow, Member States shall take the measures necessary to give priority to the processing of waste oils by regeneration" (Article 3). Thus, a great number of countries have argued that, given a number of constraints, they were not in a position to implement the Directive. In summary, most of the countries targeted give the following reasons:

- There is a high demand for waste oils used as a fuel compared to a small demand for regenerated lubricants;
- There is the typical obstacle of "risk aversion": consumers think that regenerated lubricants are of lower quality;
- There is a base oil overcapacity (France and Germany are closing plants)
- New lubricants are of lower cost.
- Re-generation is not economically viable.

Also it should be noted that countries which do not re-refine waste oils (Austria, Belgium, Denmark, Ireland, and the Netherlands according to the EU report) are small countries, where the quantities of collected waste oils are not sufficient to economically justify investing in a domestic re-refining industry. This means that re-refining is likely to be increasingly concentrated in a small number of countries, importing used lubricating oil. In summary, the countries targeted argue that the demand for regenerated lubricants is relatively small in comparison to their high production costs, and does not make the domestic regeneration of waste oils an attractive commercial option.

40. Council Directive 75/439/EEC as amended by Council Directives of 22 December 1986 (87/101/EEC) and 23 December 1991 (91/692/EEC)

Moreover, the European Commission has given contradictory signals to potential re-refiners: it has issued, on one hand, the Directive on Waste Oils which encourages industrial investment in re-refining capacity, and, on the other hand, a Directive on Waste Incineration[41], which allows lower emission limit values and less strict requirements for co-incineration, i.e. industrial boilers or cement kilns burning hazardous wastes, than for municipal waste incineration. In addition, there is a duty derogation on waste oil used as a fuel until 2006 for 11 Member States. This creates incentives to burn waste oils rather than to recycle it, which is not at all in line with the Waste Oils Directive and may explain why so many Member States have not been able to favour waste oil regeneration.

5.6. Comparative Assessment

Given the paucity of data and the differences across the programmes it is difficult to provide a systematic comparative assessment of their relative efficiency. However, there does seem to be a distinction between governments which have introduced policies designed to favour the re-refining of waste oil and those which have sought to maximise collection rates. The former have necessarily sought to overcome the barrier of unsuitable quality waste oil undermining the viability of the re-refining process.

On the other hand, where policies favour a high collection rate of waste oils by making the waste oil collection mandatory for example, as in France (80 % of waste oils are collected) and grant subsidises according to the volumes collected, the quality of waste oils is usually low because all kinds of waste oils of different qualities are mixed and the water may be illegally used to inflate volumes. This makes the re-refining to lubricants more difficult and costly and most of waste oils must be directed to incineration processes.

In general, policies which seek to maximise collection, make it free of charge or provide positive incentives to ensure that generators bring back their waste oil to the collection point. In addition, it is common to exclude certain types or sources of oil from the collection system to reduce potential for contamination and increase the potential for re-refining. This exclusion is applied as a proxy method to segregate poorer quality oils in the absence of accurate information about the precise quality attributes at a more disaggregated level. This proxy in the form of an exclusion rule such as "no marine slops" or "no small-volume amounts" is designed to reduce the risk of whole batches being made unsuitable for re-refining.

Table 2.13. Summary of Policies and Outcomes in Selected Countries

	Lubricant sold (tonnes)	% Collected	% Re-refined as base oil[1]	% Reused /Recycled
Australia	450 000 000	38	0	38
Alberta	141 000	51	0	51
Catalonia	81 000	33	17	16
Germany	1 100 000	48	9	39
Italy	578 000	33	18	15
UK	803 667	47	0	47

As a percentage of lubricant sales
1. "+ " Represents the additional subsidy from the derogation of excise duties when substitute fuels are made from waste oil. This is estimated at €34 per tonne in European countries. The Australian rate is A$70 per tonne

Sigman (1988) highlights one relationship between price signals and dumping and it is likely although uncertain, that once excluded from the free collection system, some of this waste oil may be dumped. Nevertheless, policies are needed that encourage segregation of good and bad waste oil without causing

41. Council Directive on Waste Incineration of 4 December 2000 (2000/76/EEC)

this externality to arise. The development of a low cost, rapid result testing kit for use at the point of collection would be helpful in this regard but only if used to enable waste oils unsuitable. for re-refining to be collected separately. This would reduce the information asymmetries in this market.

The Catalonian policy of offering free collection for a minimum of 400 litres of pre-tested waste oil reflects the high cost of testing a batch of waste oil relative to its market value. The policy of volume segregation could successfully improve the quality of oil delivered to a re-refinery, but at the risk of excluding a significant proportion of waste oil from multiple small point sources.

The Australian policy of offering comparatively large output subsidies to all recycling uses was designed to compensate re-refiners and others for the additional treatment, maintenance and production costs from receiving mixed poor quality waste oil. Policies were not directed at improving segregation and no particular waste management option for waste oils (re-refining versus energy recovery) was favoured by the government. However, it is interesting to note the change of policy from the Australian government who quite recently decided to improve the collection rate of waste oil instead of financially support re-refiners.

Moreover, even after waste oil feedstock has been delivered to a re-refinery further intervention has been required from Governments in Italy and Catalonia to address risk aversion in the base oil market. In Italy, the "arranged marriage" of AGIP and Viscolube was effective in giving re-refined base stock a guaranteed market outlet. In Catalonia, public vehicles were used as a market for engine oil manufactured at the single re-refining facility (C.A.T.O.R) In the United States, the Postal Service and the National Park Service use re-refined oil in their vehicles. Public purchasing policies can usefully guarantee outlets for re-refined waste oils and promote their use.

The development of high-quality lubricating oil by Viscolube and Puralube is likely to address risk aversion amongst consumers more effectively. This base oil product will display superior attributes to that manufactured in almost all European oil refineries. However, the €20 million capital costs of the hydro treating plant have been underwritten by the consortium distributing the ADF (advanced disposal fee) in Italy. There is, therefore, an important cross-subsidy which may be difficult to justify on purely environmental grounds.

Switching costs make base oil buyers less likely to use re-refined base oils. This barrier can be addressed for market entrants by meeting the costs of engine tests to enable re-refiners to demonstrate that base oils meet the minimum quality standards required by the market. In Spain, the costs of engine tests for the development of a new branded motor oil product have also been met by the state. Seeking to develop a new motor oil brand manufactured from re-refined base oil is ambitious but as a first step, making grant awards toward the cost of establishing a quality signal for re-refined base oil may be an initial means of seeking to address this barrier.

6. Conclusions

Getting the shares between re-refining, different forms of energy recovery, and other waste management options is clearly a delicate balancing act. Moreover, it is a moving target – with demand for lubricating oil stagnant and changing technologies of production, re-refining, and even detection of impurities. Against this background, some countries have placed the emphasis on maximising regeneration, while others have targeted collection without favouring one post-consumption use over another (except to restrict illegal and unregulated energy recovery). As a general rule it is, of course, best to ensure that environmental damages are regulated effectively and efficiently, and then let the market determine the optimal shares. However, this is easier said than done in the area of lubricating oils and a few specific policy recommendations emerge from the case study.

6.1. Targeting environmental policy objectives

It is clear that use of waste oil in unregulated small burners or dumping of waste oil into the natural environment results in significant environmental damages. As such, it is important to maximise collection rates, ensuring, to the greatest extent possible, that waste oils do not end up being discharged into the natural environment, whether through illegal dumping or incineration.

A secondary issue is the ultimate use to which the waste oils are put after they have been collected. LCA studies are not conclusive about the best environmental option between re-refining to lubricants and reprocessing waste oil to recovered fuel oil in industrial furnaces provided with the appropriate pollution control device. Depending upon assumptions made, one or the other may be preferable.

As long as environmental regulations for both of these processes are appropriate, there is no environmental reason to favour one over the other. If an environmental case can be made for the adoption of one waste management option over the other (eg. re-refining over energy recovery), this should be directly reflected in the environmental regulations to which they are subject, and not indirectly through interventions in the market (eg. subsidies, tax preferences, etc…). Thus, it is vitally important to ensure that air pollution regulations for 'legal' energy recovery of waste oils are enforced, a point which is taken up below.

6.2. Providing information to market participants

For markets to operate efficiently, market participants must have access to reliable information. There is, therefore, a strong case to be made for an active government role in informing small and medium enterprises generating waste oils and "do-it-your-self" households about the potential environmental damages to the environment of illegal dumping and unregulated burning, and the fines they will incur if they adopt such a strategy. This should be complemented by information campaigns concerning the location of collection points, and other practical advice to ensure safe management and avoid mixing of waste oils of different quality or contaminated with hazardous substances.

In addition, it may be possible to support the demand for re-refined lubricants by 1) informing customers on the quality of re-refined lubricants; 2) developing identical standards for virgin and re-refined lubricants; and 3) using the public purchasing tool as a demonstration tool. However, it must be recognised that government measures designed to change 'preferences' in markets where is significant risk aversion are likely to be more successful if they focus on the quality characteristics of the product themselves, rather than any associated environmental benefits.

6.3. Discouraging illegal dumping and burning

It is not a straightforward task for the regulatory authorities to discourage illegal dumping and/or burning. Waste oils are generated diffusely and it can be relatively easy to dump them clandestinely. Moreover, in many countries there is a vibrant market for burning in a variety of unregulated uses, such as space heaters and greenhouses. The administrative costs associated with monitoring and enforcement to prevent such uses of waste oils can be costly.

In the face of high administrative costs, it is preferable to complement the 'stick' of regulatory incentives (i.e. fines) with a 'carrot' (i.e. benefits payments or refunds). The resources required to fully prevent illegal uses of waste oil through the application of penalties are beyond the capacity of most regulatory agencies. It is for this reason that many jurisdictions try to pull the waste oil back into the system through positive incentives rather than trying to prevent illegal options through negative incentives.

Administratively, the simplest means of providing these positive incentives is through return incentives, often financed out of upstream charges on lubricant oil sales. However, these return incentives are usually granted on the basis of volumes returned in an undifferentiated manner. Unfortunately this can provide perverse incentives with respect to the quality of oils collected (see below). Differentiating return incentives according to waste oil quality would, however, be dependent upon accurate and timely assessment procedures.

6.4. Maximising and optimising collection without restricting re-refining

Unfortunately, efforts to maximise collection rates (e.g. through return incentives) can have a negative effect on the potential to re-refine waste oils. Since it can be costly to run parallel collection schemes due to significant density economies (particularly in remote regions), there is a tendency for heterogeneous oils to be mixed at the point of collection. In some cases oils with high water content may be collected. In other cases, there is a danger that the waste oil may contain hazardous substances. All of these restrict the potential for downstream re-refining. Benefits payments or refunds could be proportional to the quality of the collected waste oil.

This possibility is exacerbated by the fact that it can be easy to conceal (and difficult to detect) waste oils of low quality at the collection stage. Indeed, this can provide a ‘race to the bottom’ with many suppliers dumping low quality waste oils into the collection system. Since reputation effects within the market can not always prevent this arising, there is a strong case to be made for government intervention. Support for the development of low-cost testing equipment is one route.

Even in the absence of comprehensive testing it is possible to combine periodic and random testing with stringent penalties for the dumping of low-quality waste oil. Even if the probability of detection is low, if the penalty is sufficient this may dissuade the entry of contaminated waste oil into the collection system. Such enforcement schemes have been used successfully in other markets where the costs of monitoring are high.

6.5. Removing market and policy distortions

As noted above, allowing direct incineration of waste oil without any pollution control device or imperfect enforcement of air emissions regulations result in an unfair competition between re-refining and energy recovery. In effect, there is a market distortion if externalities are left internalised in one waste management option but not in the other.

However there are other potential distortions in the market(s) which may favour one or the other waste management option. For instance, in the United Kingdom it has been argued that derogation on excise duties for waste-derived fuels oils used for energy recovery provides a significant cost advantage, undermining the potential development of the re-refining industry which is competing for the use of the same material input. It has been estimated that the duty would add 50% to the current cost of recovered fuel oil.[42] This derogation is to be rescinded in 2006, removing a significant market distortion.

Needless to say, unless removal of the derogation (and other market and policy distortions of this kind) is combined with effective enforcement, there is a danger that it will reduce collection rates by driving the waste oil into illegal dumping or burning, rather than into the re-refining industry. The need for effective co-ordination is key.

42. Roger Creswell “UK Waste Oil Sector” presentation to the NIHWF, Nov. 2003.

6.6. Ensuring policy consistency

If it was possible for market participants to switch easily between different waste management options, changes in relative costs and policy incentives could be absorbed relatively easily by the market. However, as has been noted the start-up costs for re-refining can be very significant, not only because of the physical capital but also due to other factors associated with market entry such as testing costs.

As such it is extremely important that policymakers provide clear incentives to investors. For instance, the development of inconsistent regulations has sent contradictory signals to investors. On the one hand, the priority given to regeneration in the of the European Waste Oils Directive appeared to provide the basis for the development of a market. On the other hand, other incentives such as the Incineration Directive and the duty derogation on waste used as a fuel, have favoured waste oil incineration to the detriment of re-refining. In addition, the Waste Oils Directive has been imperfectly enforced due in part to the apparent inability of some countries to support such an industry without significant public financial support.

More generally, there is a clear need to ensure that policy measures which are designed to increase the supply side (i.e. collection schemes) are co-ordinated with measures to increase the demand side (i.e. procurement policy). In the event that inconsistent signals are given, there is likely to be significant disequilibria in the market and thus great uncertainty. In the long run this will discourage investment.

REFERENCES

AEA Technology Environment (1998). "Power Generation and The Environment - a UK Perspective", Ref AEAT 3776.

AKERLOF (1970) "The Market for Lemons: Quality Uncertainty and the Market Mechanism"*Quarterly Journal of Economics*, Vol. 89, No. 3.

AL-AHMAD et al (1991). "Techno economic study of re-refining waste lubricating oils in the Arabian Gulf countries" *Resources, Conservation and Recycling* Vol. 6 Issue 1, pp 71-78.

ALI, MOHAMMAD *et al.* (1996). "Techno-economic evaluation of waste lube oil re-refining", in International Journal of Production Economics Vol. 42 Issue 3 pp 263-273 (April 1996).

ALLEN CONSULTING GROUP (2004) 'Independent Review of the Product Stewardship (Oil) Act 2000' Final Report Prepared for the Australian Minister for the Environment and Heritage, Canberra.

AMERICAN PETROLEUM INSTITUTE (API) (1996). "National Used Oil Collection Study", May 1996 plus addendum in August 1997.

AUSTRALIAN ACADEMY OF TECHNOLOGICAL SCIENCES AND ENGINEERING (2004) "Independent Review of the Transitional Assistance Element of the Product Stewardship for Oil Program" Report Prepared for the Australian Minister of the Environment and Heritage, Canberra.

BDA GROUP (2003). "A Tradable Certificate System for Used Oils" Department of the Environment and Heritage, Australia.

BOUGHTON, BOB AND ARPAD HORVATH (2004) 'Environmental Assessment of Used Oil Management Methods' in *Environmental Science and Technology*, Vol. 38, No. 2.

BOYD, S. (2002). "Green Lubricants. Environmental benefits and impacts of lubrication", *Green Chemistry 2002* Volume 4 pp 293-307.

COMMISSION NATIONALE DES AIDES USAGEES (2002) "Rapport d'activité: 2002" (Paris: ADEME).

COMMISSION OF THE EUROPEAN COMMUNITIES (2000) Implementation of Community Waste Legislation for the Period 1995-1997 – Directive 75/439/EEC on Waste Oils (COM(1999)752final).

CONCAWE (1996). *Collection and Disposal of Used Lubricating Oil*, Report No 5/96 Brussels.

CORPORATE LINK MANAGEMENT CONSULTANTS (2002). "Economic and Environmental Performance of Alberta's Used Oil Program". A discussion paper presented to OECD workshop on EPR in Tokyo, Japan, December 2002.

CSERGE (1992). *The Social cost of Fuel Cycles.* Report to the Department of Trade and Industry ISBN 0 11 414288 2.

DAGMAR SCHMIDT ETKIN (1999) “Oil Spill Intelligence Report” presented to the International Oil Spill Conference, 1999

DAGMAR SCHMIDT ETKIN (1999). *Estimating Cleanup Costs for Oil Spills,* report to the 1999 International Oil Spill Conference.

D'ARCY Brian *et al* (1998). "Initiatives to tackle diffuse pollution in the UK", *Water Science and Technology,* Vol. 38 Issue 10 pp 131-138.

DEFRA (2001). “Waste Lubricating and Other Oils”, U.K. market study.

ENVIRONMENT AUSTRALIA (2003) “PSA Benefit Rates 2002/03” (Canberra: Environment Australia)

EUROPEAN COMMISION DG XII Science Research and Development, *Externalities of Fuel Cycles ExternE Project,* EU Bruxelles.

EUROPEAN ENVIRONMENT AGENCY (2002) “Review of Selected Waste Streams: Sewage Sludge, Construction and Demolition Waste, Waste oils, Waste from Coal-Fired Power Plants and biodegradable Municipal waste”. European Topic Center on Waste- EEA- Copenhagen.

FUCHS PETROLUB AG MANNHEIM (2003) “The World Lubricants and Base Oils Market”, Paper presented to 7th ICIS LOR base oils conference 2003

GRANT, S. *et al* (2000). "Different notions of disappointment aversion", *Economic Letters*, Vol 70 pp 203-208.

GUL, Faruk (1991). "A theory of disappointment aversion", *Econometrica*, Vol 59, No 3, pp 667-686.

JAGGER, S. (2003) "Competition in the Global Base Oil Business". Paper Presented to the Seventh ICIS-LOR World Base Oils Conference. 20th & 21st February 2003, London.

OGDEN, Joan *et al* (2004). "Societal costs of cars with alternative fuels/engines", *Energy Policy,* Volume 32, Issue 1 pp 7-27 (Jan 2004).

PORTER, R. (2002) *The Economics of Waste* (Washington, D.C.: RFF)

SIGMAN, H. (1998). "Midnight Dumping: Public Policies and Illegal Disposal of Used Oil”, *RAND Journal of Economics*, Vol.29 No1, pp 157-178.

SOFRES, TN. (2001) “Sustainable Resources Consumption and Waste Critical Review of Existing Studies and Life Cycle Analysis on the Regeneration and Incineration of Waste Oils", Report to the European Commission DG Environment, December 2001.

UNITED KINGDOM ENVIRONMENT AGENCY (2003) (2003 in text?) *Fuel and Pollution Incidents by Type, 2002,* Available at http://www.environment-agency.gov.uk.

UNITED STATES BUREAU OF TRANSPORTATION STATISTICS (2002). *Omnibus Household Survey on Disposal of used motor oil*. Available at http://www.bts.gov/omnibus.

USED OIL MANAGEMENT CONSORTIUM (2002). Conversation with Renato Schieppati. Milan.

CHAPTER 3:

IMPROVING MARKETS FOR WASTE PLASTICS

by

Alan Ingham
Principal Research Fellow
Economics Division
University of Southhampton

1. Introduction

Compared to other potentially recyclable materials, levels of recycling of plastics are seen as being relatively low in OECD countries, particularly for post-consumer plastics. Moreover, there is wide variation in recycling rates across different countries. At the same time many countries are introducing measures to encourage increased rates of recycling.

This report is concerned with commercial markets for recyclable plastics. The intention is to review market conditions in order to identify any barriers or failures which may result in sub-optimal levels of material recovery. These might include information failures, market power, search costs, and other factors which inhibit the development of markets. In addition, policy frameworks in three OECD countries (United States, Sweden and Germany) are reviewed in order to assess whether they are effective means of bringing about increased rates of recycling. Particular attention is paid to the role such policies play in relation to the market barriers and failures identified.

In addition to the introduction, the report has six sections. The second section provides a broad overview of the role of plastics in OECD economies. The third section reviews the recycling processes and industry. The fourth section reviews some barriers and failures which can affect markets for plastics recycling, while the fifth section looks in some detail at the implications of price volatility in the market. The sixth section reviews the policy framework in three OECD countries. There is also a concluding section.

2. The Importance of Plastics in OECD Economies

Plastics cover a large range of different materials and uses. Since the introduction of celluloid in the 1870s, plastic has been used in an increasing number of products. The commercial development of plastics began in the 1930s, when polystyrene, acrylic polymers and PVC started to be mass-produced from petroleum. During the Second World War, the production of polyethylene, polystyrene, polyester, nylon and silicones grew, and polyethylene terephthalate was discovered in 1941. Since the 1950s the manufacture of plastics has grown into a major industry. Sales of plastics increased, primarily due to greater demand for thermoplastic plastics such as polyvinyl chloride, polystyrene and polyethylene. For example, consumption of polystyrene doubled, due in part to the use of polybutadiene as an impact modifier to improve its strength.

Development of reinforced fibreglass led to a 20-fold increase in consumption of unsaturated polyester between 1954 and 1960. New production methods resulted in increased use of polyethylene, including its use in containers. The use of polyester, lycra and nylon in clothing grew. In the 1960-70s there was increasing use of plastics in engineering and electronic applications and medicine. This was then followed by a slump in plastic production due to rising oil prices and the world energy crisis of 1973-1975, although new manufacturing techniques resulted in an increased use of PET. More recently in the 1980-90s there has been accelerated growth in the manufacture of plastics due to widespread applications in the electronics and communications industry and increased use of plastics in cars, due in part to the development of high temperature polymers. In addition, plastics are increasingly used in a variety of packaging applications.

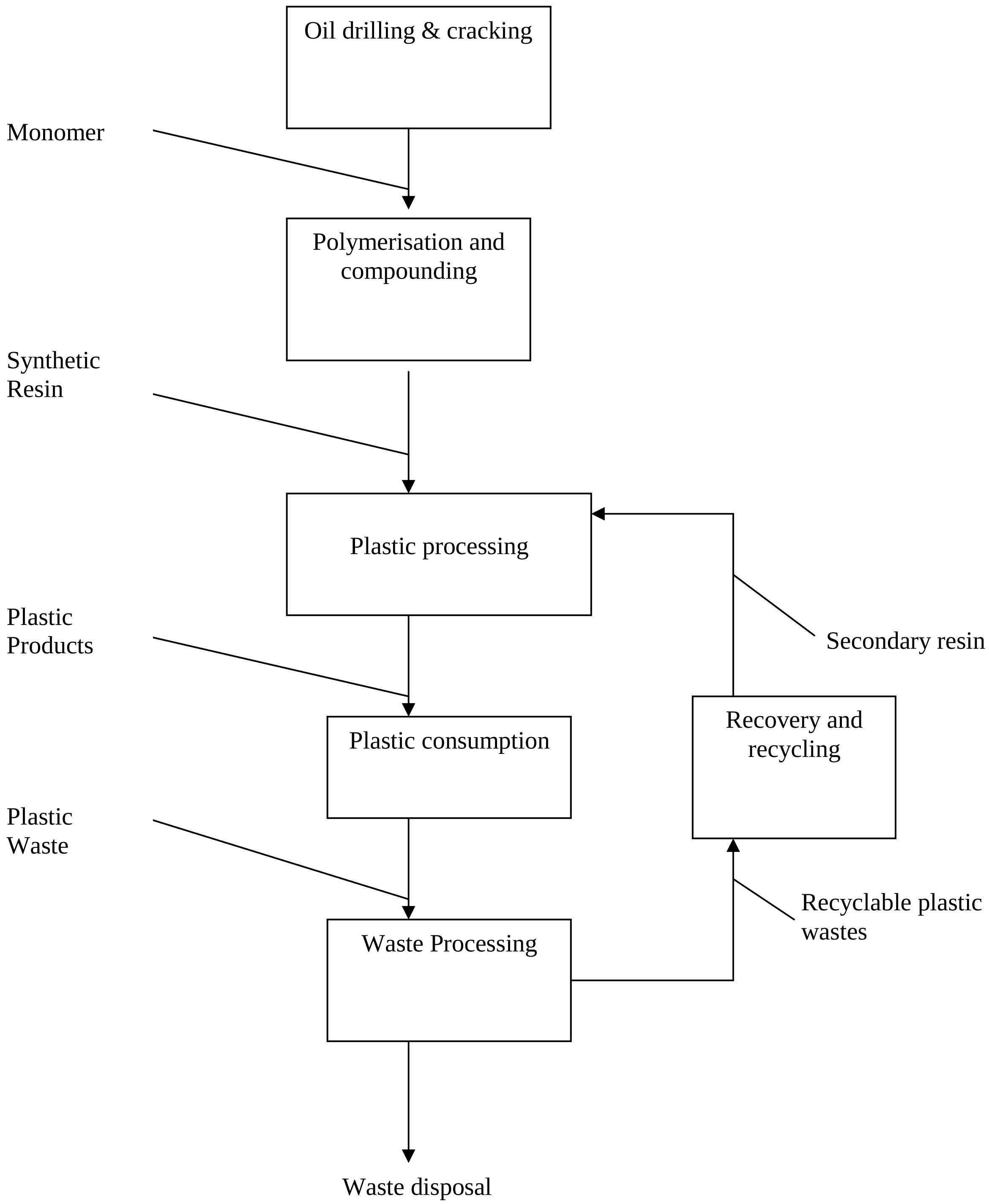
2.1. The Process of Making Plastics

Plastics can be divided into two major categories: thermoplastics and thermosets. Thermoplastics can be remelted and reformed many times into different shapes. For this reason, they are the most commonly recycled plastics. Thermosets can only be formed once. After that, they may be ground and used as filler for future plastic products. Thermoplastic resins are the primary focus of this report because they are more readily recycled. The major categories of thermoplastic resins are high-density polyethylene (HDPE), low density polyethylene (LDPE), linear-low density polyethylene (LLDPE), polyethylene terephthalate (PET), polystyrene (PS), polypropylene (PP), and polyvinyl chloride (PVC). Of the different types of products, the plastics recovered from the waste stream for recycling are mostly those used in packaging.

Virgin plastic resins are produced in large-capacity facilities from the monomers that are the building blocks to plastic polymers. Oil and natural gas are the main raw materials for producing the monomers. Other natural resources such as wood and coal can be used, but this is not common. Four per cent of annual world oil production is used as a feedstock for making plastics, in addition to the four percent used as energy to process plastics (Wastewatch, 2001). Extracted oil is made up of a mixture of hydrocarbons, which can be separated into fractions, such as gasoline, naphtha and bitumen. Some of these can then be processed, by cracking or reforming, into compounds that provide the starting material for the petrochemical industry to produce materials such as plastics and synthetic rubber.

The monomers undergo polymerisation to form the polymer. Additives can be added to the resin pellets to achieve desirable properties. Only minor changes in chemical structure can lead to significant changes in the material characteristics. The fact that additives can change the characteristics of the polymer considerably has made plastics a very versatile and widely used material. For example, PVC is thermally unstable, so stabilisers and lubricants are added in a compounding process. This produces rigid PVC. Flexible PVC is made with the addition of plasticisers. A wide variety of additives are used to help in further processing. Unfortunately, as discussed below, these additives can make recycling difficult and lead to a substantially increased cost of doing so.

Figure 3.1: Flow Diagram of Plastic Cycle[43]



43 . Figure based upon van Beukering (2001).

2.2. Uses of Major Plastics

Given their versatility, it is not surprising that plastics have found a large number of commercial applications. Products based upon the production of plastics include the following (APME 2000b):

- High-density polyethylene (HDPE): Containers, toys, housewares, industrial wrapping and film, gas pipes
- Low-density polyethylene (LDPE): Film, bags, toys, coatings, containers, pipes, cable insulation
- Polyethylene terephthalate (PET): Bottles, film, food packaging, synthetic insulation
- Polypropylene (PP): Film, battery cases, microwave containers, crates, car parts, electrical components
- Polystyrene (PS): Electrical appliances, thermal insulation, tape cassettes, cups, plates
- Poly Vinyl Chloride (PVC): Window frames, pipes, flooring, wallpaper, bottles, cling film, toys, guttering, cable insulation, credit cards, medical products.

Table 3.1 gives an indication of the proportional use of plastic resins in different commercial product streams in 2000 in the United States.

Table 3.1. Use of Plastic Resins in the U.S.

Industrial Machinery	1%
Adhesives/Inks/Coatings	2%
Furniture and Furnishings	4%
Electrical/Electronic	4%
Transportation	5%
Exports	10%
Others	12%
Consumer & Institutional	14%
Building and Construction	22%
Packaging	26%

Source: APC (2001)

The role of plastics in packaging is particularly important. Three types of plastics play a particularly important role in packaging:

1. PET is clear, tough, heat-resistant and provides a barrier to gas and moisture. It is used mainly in drink bottles but also in fibre applications, using recycled materials.
2. HDPE is stiff, resistant to chemicals and moisture but permeable to gas. It is easy to process and mould and is used in bottles, tubs and bags.
3. LDPE is used primarily in films for wrapping and covering.

Table 3.2. gives data on the demand for PET in Western Europe for different uses in 1999 (actual), as well as a forecast for 2004. Food contact containers make up almost two-thirds of the total demand, with fibre the second-most important use.

Table 3.2. Western European Demand for PET for 1999 (Actual) and 2004 (Forecast) (ktonnes)

Market Sector	Total Market size incl. virgin		% penetration by Recycled PET		Tonnage used	
	1999	2004	1999	2004	1999	2004
Fibre	448	605	20	33	88	199
Food Contact Containers	1224	1955	<1	16	5	314
Non-food containers	208	247	<1	14	1	35
A-sheet	89	134	6	35	16	47
Strapping	27	40	22	75	6	30
Injection Moulding	100	122	2	12	2	15
Polyols	30	52	3	33	1	17
Total	2126	3155	6	21	118	657

Source: WRAP (2002)

Plastics offer significant resource savings in packaging as they reduce the amount of material needed to provide the same function as other materials, but are also much lighter, resulting in considerable transport cost savings. This advantage comes at a cost. Plastics are relatively bulky so that whilst plastics have contributed to a reduction in the weight of waste, this is not matched in terms of volume. This low weight/volume ratio causes problems in the economics of recycling where volume becomes a critical factor in transport costs.

2.3. The Environmental Impacts of Plastics and Plastics Recycling

The rapid growth of plastics production is reflected in the growth of plastic waste as a proportion of total Municipal Solid Waste (MSW). This can be seen in Table 3.3, which gives figures for the United States.

Table 3.3. Proportion of plastic in MSW

Year	% by Weight
1960	0.5
1970	2.6
1980	5.0
1988	8.0
1990	9.8
1992	10.6
1994	11.2
1995	11.5
1996	12.3

Source: Subramian (2000).

The difference in the plastic component of European waste in the early 1990s is shown below in Table 3.4, with figures ranging from 6% (UK) to 11.5% (Ireland).

Table 3.4: Proportion of Plastics in MSW (%) by Weight

Year	1975	1980	1985	1990	1993	1995	2000
Country							
Belgium	5	6		4		6	6
Denmark	4	7	4		7	0.3	0.5
Germany	8	6	5	0.6		3	
France	5	6	9	10	10	11	11
Greece		7	7	11	9	10	9
Ireland	4	11	14			9	10
Italy	5	7	7				
Luxembourg	5		6		8	8	
Netherlands	6	7	7	8	9	5	5
Portugal		3	3	9	12	12	
Spain		6	7	7	11	11	12
Sweden		10		7			
UK	4	7		10			
USA	4	5	7	8	9	9	11

Source: OECD Environmental Data Companion various years, UNEP Environmental Data Report various years, Eurostat (1995) .

Moreover, the range of plastic types, together with their additives makes separation into sufficiently homogeneous types for recycling purposes a serious problem. On the other hand, plastics are difficult to compact, have a high volume /weight ratio, and are resistant to degradation. All of these make landfill, as an alternative to recycling, an unattractive, even if sometimes unavoidable, option.

Incineration can also pose problems, and has been seen as unattractive because of public opposition. Some of this arises from opposition to incineration of MSW in general, some from a concern with the pollution that might arise from the burning of additives. For instance, an important attribute of PVC is that the additives have significant environmental implications for air quality when incinerated. However, in a LCA study, Bruvoll (1998) found that the costs of recycling exceeded those for incineration even when the social cost of dioxins (which has high marginal costs) is included.

A study by Scharai-Rad and Welling for the FAO (2002) compared the use of wood to substitute materials in various building components. For instance, PVC can be used as a substitute material for window frames and for floor coverings. It is not immediately obvious whether wood would dominate PVC in environmental terms for window frames, for whilst PVC has various environmental costs associated with its production and disposal as waste, wooden frames will require regular applications of preservatives and paints which also have environmental costs. However, on balance, once all of the lifetime impacts are taken into account, it turns out that wood dominates PVC almost completely.

Using recycled materials in production generally requires about three times less energy than for virgin production because the recycled products are already partially converted into the final product. The US EPA estimates that the energy savings from recycling rather than landfilling PET bottles are equivalent to 24 mil Btu/ton (EPA 1999). Over 95 percent of the total energy required to produce one kilogramme of plastics is due to oil extraction and refining. Avoiding these steps by recycling can thus result in significant energy savings.

There are other adverse environmental impacts associated with other kinds of plastics. For instance, PS can be a problem because of the nature of its waste. EPS that comes as moulded packaging is very bulky. It is light so often blows away in the waste disposal process. As it easily breaks into small pieces, it can cause environmental damage through the blockage of waterways, and damage to marine wildlife.

For the majority of mixed plastics, and plastics of lesser quality, a steady supply of plastic waste is a requirement for market development and industrial investments. Therefore, the separate collection of plastic waste and dismantling of complex products has been seen as a prerequisite to ensure environmental benefits. High quality mechanical recycling offers the largest environmental benefits. The potential for this is steadily increasing due to improvements in dismantling and separation technologies.

3. The Recycling Process

Plastics can be recovered for either material or calorific values. This report focuses on recovery for material use, but it is important to note that energy recovery is more important in most countries. Table 3.5 gives data for European Union countries for 1999. The degree of variation is marked, with some countries recovering very little plastic (eg. Ireland) and others a very high proportion (eg. Denmark). As the percentage of unrecovered waste increases across countries, the proportion going to energy recovery tends to rise, as in Denmark, Netherlands and Sweden. The only country for which recovered waste represents more than 20% of total consumption and for which material recycling is more important than energy recovery is Germany.

Table 3.5. Plastic Recycling and Energy Recovery (%) in Europe in 1999

	Recycling	Energy Recovery	Unrecovered Waste
Denmark	5.6	66.6	27.8
Netherlands	10.0	50.8	39.2
Sweden	10.1	50.2	39.7
Germany	29.9	25.4	44.7
France	7.9	33.1	59.0
Austria	18.8	21.0	60.2
Belgium	12.8	25.8	61.4
Greece	1.7	17.9	80.4
Italy	8.5	7.2	84.3
Finland	10.3	5.2	84.5
Spain	7.5	5.6	86.9
UK	6.2	5.9	87.9
Portugal	3.0	8.7	88.3
Ireland	4.1	0.0	95.9
EU AVG	11.3	19.3	69.5

Source: APME (2001)

Plastic waste arises at different stages of the production/consumption cycle and can be recycled in different ways. The basic distinctions are three-fold:

- Post Industrial vs. Post Consumer;
- Closed Loop Recycling vs. Open Loop Recycling; and,
- Mechanical Recycling vs. Feedstock or Chemical Recycling.

In addition to these is the distinction between material recovery and energy recovery through incineration, or waste-to-energy. These distinctions are discussed below.

3.1. Source of Waste: Post-Industrial vs. Post-Consumer Recycling

On the basis of data from the Netherlands, it is found that the quantity of plastic waste being reabsorbed into the production system almost matches the quantity of post-industrial waste (Joosten, Hekkert and Worrell (2000). This is not surprising as the recycling rate for post-industrial material has been high for a long time, whilst the post consumer recycling rate has been low. In-house recycling of scrap plastics process waste is carried out extensively.

It makes financial sense to recycle post-industrial plastic waste as it is contaminant-free, consists of a single, identified polymer type and there is a demand for the recycled product. Thus, it is standard practice amongst resin producers, and most large fabricators to gather, recycle, and rework as much scrap as is possible in their processes. Scrap plastic is produced at every stage of manufacture. Resin producers, fabricators and converters all generate wastes from off-grade products, spillage and equipment cleaning. Fabricators and converters also produce waste from trimming, moulding and forming operations. Where certain smaller operators may not find it economic to employ in-plant reclamation, reprocessors collect, grind and blend these wastes for recycling as secondary resin. In such cases, plastic reprocessors know that it corresponds to a currently used material, is clean and can be reabsorbed without loss of quality. Both internal and external recycling from post-industrial scrap is therefore developed to a high degree.

This is in marked contrast to post-consumer recycling which, despite many efforts and policy initiatives, remains at a low level. There are a number of barriers to increasing the levels of post-consumer plastic recycling which are related to uncertainty concerning waste quality and complexity. These barriers are inter-linked and include the fact that plastic waste for recycling needs to be free of contamination and relatively homogeneous. Household plastic waste is often mixed with other household waste such as paper, metals and food waste. To ensure that the plastic is suitable for recycling this type of waste needs to be sorted and cleaned prior to processing.

Thus, for post-industrial material comprising clean scraps from the production process, estimates for the USA in the early 1990s put recycling of production scrap in the range between 75% and 90%, whilst post consumer recycling was around 1% to 3%. (Derry 1989). However, because of the already high levels of post-industrial scrap recycling, further recycling would need to come from MSW and other post-consumer waste

3.2. Nature of Recovery: Closed Loop, Open Loop, and Energy Recovery

Closed loop recycling refers to the situation where the recycled item is re-made, e.g. using PET bottles to make new PET bottles. This reduces the amount of raw material going into this product. Open Loop recycling refers to the case where different items are made, e.g. using PET from collected bottles to make synthetic fibre or plastic lumber. In this case, the amount of raw material going into PET bottle production is not reduced, and the savings arising here from the recycling of PET are at the expense of a renewable raw material – timber or natural fibres. Most current recycling is of the open loop form. Curlee (1986) identified three separate classes of recycling activity for plastic.

Primary. This corresponds to closed loop recycling. It is where waste is processed into a material with characteristics similar to the original product. For most materials with long established and successful recycling markets – eg. gold or aluminium - this is what happens. For post-industrial plastic scrap primary recycling is the main activity. For post-consumer plastics this has proved to be difficult because of contamination, and the difficulties of separation, associated with the complex and mixed nature of the waste.

Secondary. This includes reprocessing waste into materials that have characteristics less demanding than the original material. An example would be the recycling of plastic waste into plastic lumber used as fence posts. Performance may still be important though, and the uncertain characteristics may limit applications. An example of this is the use of plastic lumber in marine applications where its resistance to rot might make it an attractive material. However, laboratory studies have shown that its engineering properties are inferior to those of wood, it is prone to damage from UV radiation, and suffers from deformation under load. Plastic lumber has variable elasticity and this has been judged to limit its areas of application. (Breslin et al. 1998).

Tertiary. Tertiary recycling produces basic chemicals and fuels by taking the plastics back to the basic monomers and polymers. Pyrolysis and hydrolysis are examples of this. This has been an area of much research. It has proved to be feasible although expensive. Although it is the area where the petrochemical companies would have comparative advantage, economic considerations lead them to continue to rely on oil as a basic input, rather than recycled plastics.

If material recycling cannot be undertaken efficiently, waste plastic can be used as a fuel. It has a high calorific content, and where MSW is incinerated, fuel such as oil might need to be added if plastic content were not present. Where it is burnt by itself it has an average calorific content of 12,000 BTU per lb, which is similar to that of anthracite.

3.3. Mechanical vs. Feedstock Recycling

The distinction between mechanical recycling and feedstock (or chemical) recycling is that mechanical requires physical separation, whereas feedstock recycling converts mixed PE/PP/PS/PVC into basic hydrocarbon building blocks. Thus, mechanical recycling is appropriate for the primary and secondary recycling routes discussed above, through the grinding of plastic products into pellets or flakes. Feedstock recycling technologies, whilst they can handle contaminated and mixed material, are relatively expensive.

Mechanical recycling is primarily a method of treating single polymer waste. The waste is converted into pellets or granules for extrusion into new products. This allows the addition of virgin polymer materials, pigments and stabilisers. Feedstock recycling is a relatively new recycling technology, which can be used to treat mixed polymer waste. The method used depends on the nature of the waste. For example, thermolysis is used to convert mixed plastic wastes back into the component monomers. Feedstock recycling plants require major capital investment but can handle relatively high tonnages of plastics.

3.4. The Plastics Recycling Industry

The recycling of plastics involves various separate activities. These are: collection; sorting; cleaning and granulation; and re-processing. Typically, each of these different activities are undertaken by different firms or public bodies. They are very different activities from those of firms involved in virgin plastics production, with much less technological sophistication. This is not to say that there are not difficult problems associated with separation.

Factors affecting the profitability of recycling include the price paid to the collector or intermediate processor, processing costs, and selling price. The price paid to the collector is dependent on the collection method used and the distance from generation to the intermediate processor or recycler. Processing costs are determined by the quality of the material and the throughput of the facility. The price paid by the plastic product manufacturer for the processed resin is generally lower than that of competing resins. Vertical integration and economies of scale realised in virgin resin production are not generally available to recycled plastics processors and compounders, making the margin that they work in very narrow.

How the recycling industry is organised depends significantly upon government rules and regulations and so varies from an integrated system such as that in Germany to a decentralised scheme such as is found in the United States, although there is a considerable difference between different states. Many of these firms are relatively small. Re-processing firms are typically SMEs in the range 5,000 - 20,000 tonnes per annum. One explanation for this is the diversity of polymers and products, especially in comparison to steel and aluminium. Another explanation is the scale of investment required to set up in business. For the

United Kingdom, direct capital requirements for mechanical recycling fall in the range £10,000 to £ 1.5 m. In the UK in 1996 there were over 100 re-processor companies.

Reclaimers tend to be the smallest of the enterprises involved. However they are at the heart of the recycling process. The problem that follows from this is the ability of these small firms to withstand the financial shocks that arise from price volatility. While the volatility may be not that much greater for recycled plastic prices than for virgin, the size of the enterprises involved in virgin production means that they are better able to smooth out profits and losses. Indeed, it is almost always the case that the cost of collecting, sorting and transporting plastic bottles to reprocessors usually exceeds revenue generated by the sale of recovered bottles. This is usually supported by some form of subsidy or other financial transfer such as the payments made by DSD in Germany (see below).

4. Barriers to Recycled Material Use

The following section looks at what are generally seen as being barriers to increased recycling rates and use of recycled material. Not all of these may actually be market failures in that, for example, they correspond to prevailing consumer preferences or costs of supply. As such, they are not all amenable to policy intervention. Nonetheless, by focusing on barriers it is possible to obtain an improved understanding of why plastic recycling is not greater than it is at present.

4.1. Technological Externalities

Technological externalities relate to the use of plastics with characteristics that make recycling more costly or difficult than would otherwise be the case. Examples include the use of composite plastics, special additives or mixing of colours. However, such characteristics can not be described as 'market failures', unless there are "missing markets" in the production-consumption chain, which discourage manufacturers from using optimal product designs. For instance, if manufacturers face no incentives to use more easily recyclable plastics, they will design their products accordingly, even if the social costs of doing so outweigh the social benefits.

For instance, a range of modifications to standard PET bottles (e.g. barrier layers in bottles, barrier coatings etc.) have been developed to allow products such as beer to be packaged in PET bottles. These modifications bring important commercial benefits (or would not be undertaken). However, they are to a greater or lesser extent, incompatible with the existing processes for mechanical recycling. Recycling may be afforded a low priority during packaging design/specification, designers and buyers may well be ignorant of the impact of packaging design on recycling activities, and may face few incentives to take them into account.

Restrictions on the use of plastics for food packaging are an area where these potential trade-offs are most acute. Such restrictions can constrain the market in two ways: it can prohibit certain applications outright; and, it can require the 'mixing' of plastics in a way which severely restricts the potential for recovery. For instance, EU legislation lays down strict criteria regarding the qualities of plastics that are allowed to come into contact with food.[44] This is to prevent any potential food contamination, and clearly provides significant social benefits. Because of the stringency of these directives, plastics recyclers are reluctant to use their product for this type of packaging. This means that the potential market for recycled

44. Council Directive 89/109/EEC on *the Approximation of the Laws of the Member States Relating to Materials and Articles Intended to Come in Contact with Foodstuffs* and Commission Directive 90/128/EEC, as amended, *Relating to Plastics Materials and Articles Intended to Come in Contact with Foodstuff)*

plastics in the packaging industry is limited since much of the plastics used in food packaging comes into contact with food.

In the USA the Food and Drugs Administration (FDA) will allow recycled plastics to come into contact with food under certain conditions and at least three different processes have been approved for this purpose. RECOUP, the UK Recycling of Used Plastic Containers group, claim that it is possible to use post consumer recycled (PCR) plastics in contact with food provided that one of three recovery methods is used: feedstock recycling, multi-layer extrusion processing (where the contaminated layer from recycled plastics goes between two virgin layers of plastic), and super-clean recycling processes. Coca Cola have pioneered the use of chemically recycled PET for post consumer use in other food applications. In 2001, the Coca-Cola Company made a requirement of incorporating 10% of post consumer RPET in their bottles[45].

4.2. Collection, Transaction and Search Costs

Commercial and, in particular, domestic plastic waste arisings are found in diverse locations in low volumes. The cost associated with collection from the point at which the waste arises and transportation to a recycling facility can often be prohibitive. Table 3.6 below gives illustrative figures for the UK and is based on information published by RECOUP in November 2000. Market prices for collected material are linked to the value of virgin material, which are cyclic in nature. The bottle sales revenue figure of £10/tonne is based on RECOUP's estimate of a typical point in the HDPE price cycle over the period 1996-1999 without any element of price support.

Table 3.6. Estimates of Plastic Bottle Collection Scheme Costs

Scheme Activity	Debit (£/tonne)	Credit (£/tonne)
Plastic Bottle Collection	140	
Scheme Promotion	14	
Manual Sorting by Polymer type (labour)	100	
Sorting site and capital depreciation	50	
Baling	25	
Delivery to reprocessor	20	
Scheme Administration and other overheads (@11%)	41	
Collection scheme gross cost	**390**	
Avoided disposal cost		25
Baled bottle sales revenue		10
Collector gross income		**35**
Collector net income (loss)	**(355)**	

Source: RECOUP

Collection and transport costs are an important problem because post consumer material has low density even when compacted. The cost (£/tonne of material recovered) incurred by collection schemes is generally higher for plastic bottles than other post consumer recyclables such as paper and glass. Two main factors contribute to this higher cost: the relatively low bulk density of plastics and the low proportion of plastic bottle material (by weight) in the domestic waste stream.

The bulk density of un-flattened post consumer plastic bottles is typically 20-30 kg/m^3, while the bulk density of mixed post consumer recyclables (paper, card, plastic bottles, cans and textiles) is 50-100 kg/m^3. The density of baled plastic bottle varies with polymer and baling press used, bale densities of 230-370 kg/m^3 are typical. Kerbside collection vehicle capacity may be limited by volume of material recovered rather than weight. Plastic bottle collection banks recover less weight of material/bank than banks of

45. Recycling Today (3/6/2003 http://www.recyclingtoday.com/News/news.asp?Id=3719)

equivalent volume recovering materials with higher bulk density. Low bulk density is also a factor during transport of baled plastic bottles.

Plastic bottle collection schemes focus on the collection of two or three types of plastic bottle; HDPE, PET and less frequently PVC, which collectively account for more than 90% of the plastic bottles in the domestic waste stream. A range of polymers including polypropylene (PP), polycarbonate (PC) and PEN polymers make up the balance. Reprocessors require material matching established specifications with low levels of contaminants such as non-specified polymers, paper etc. Recently there has been a trend to avoid costs associated with sorting by selling recovered material unsorted to waste management companies for subsequent sorting on a regional basis or export.

Unreliability of supply can also affect price information. For example, WRAP, a government company set up to support recycled material markets in the UK, has currently undertaken an investigation into price data relevant to the UK recycling sector (WRAP, 2003). They have found that “behaviour of markets for recyclable materials has suggested that a lack of price transparency in the sector is a key barrier to market participation and maturity. Without quality price information, economically sub-optimal deals will be made and participants will be forced to spend valuable time attempting to find reliable price data. It is suggested that markets for recyclable and recycled materials tend to be particularly opaque for a number of reasons, including the lack of readily accessible, reliable, price information.”

Transaction costs of this type can lead to market inefficiency. However, identifying whether or not markets are efficient is not straightforward. One standard measure to use is to ascertain whether or not the ‘law of one price’ holds. If a market is efficient, the price for individual commodities within specific time periods should exhibit little variation. However, as the data for Germany in Figure 3.s 3 and 4 below shows, prices for plastics are indeed variable within individual time periods, especially for mixed plastics. (EUWID, various years).

Figure 3.2. The Variability of German Recycled Plastic Prices (DM/kg)

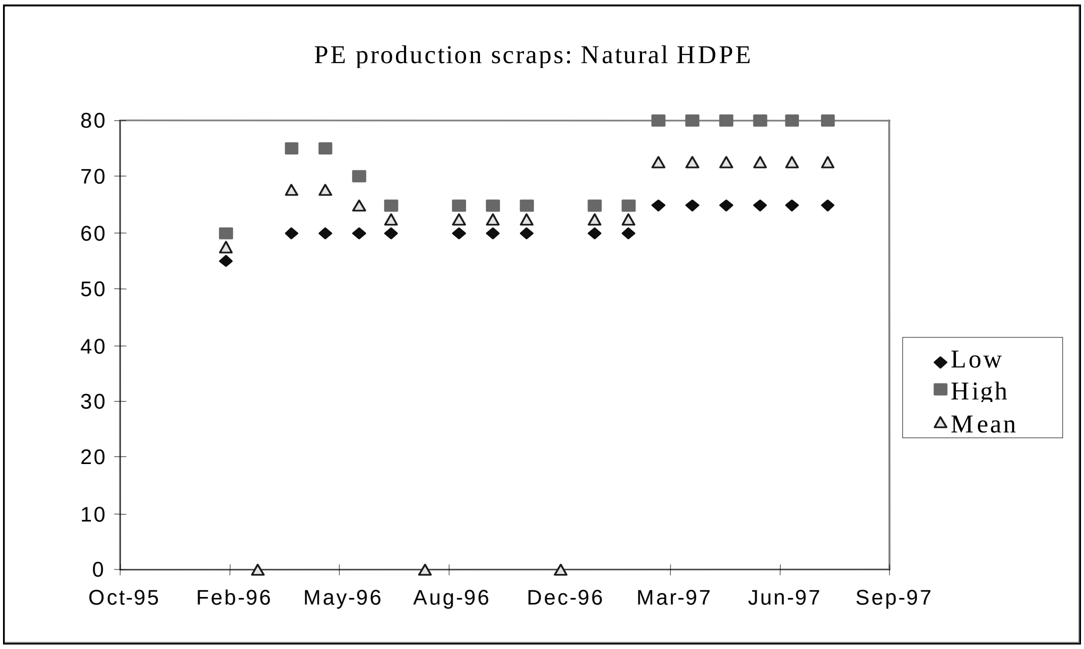
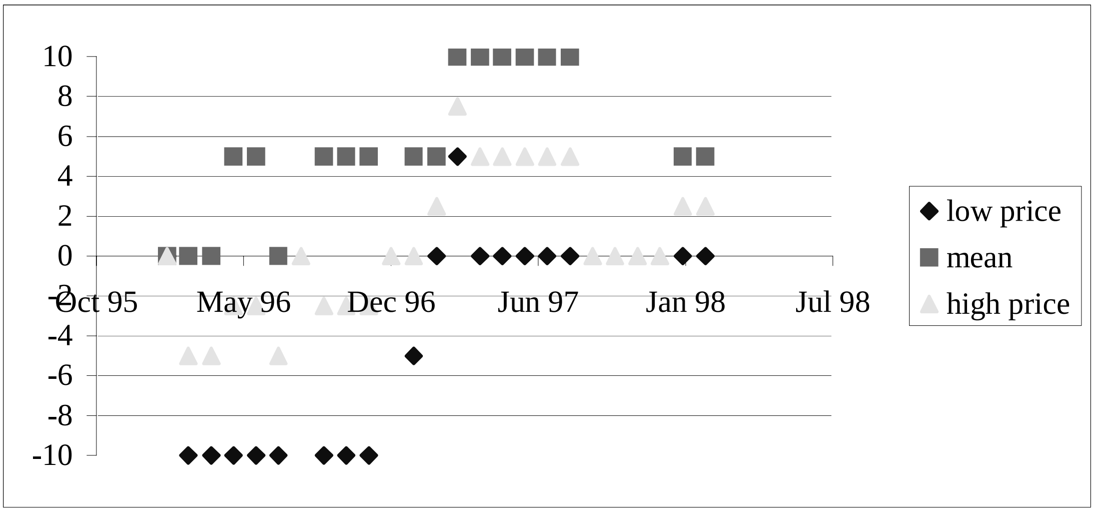


Figure 3.3: The Variability of German Recycled Plastic Prices (DM/kg)



4.3. Waste Quality and Demand-Side Information 'Failures'

It is often asserted that demand for secondary materials is less than optimal because consumers have an incorrect and irrational perception of the quality of products containing recycled material. This is thought to be especially important where the recycled product enters as a final good. Where recycled material enters as an intermediate good consumers are unlikely to be aware of its presence. An observation often made is that a stigma attaches to recycled materials and this reduces the take-up of recycled material, with consequent effects on prices. For example, the section quoted in Box 1 below comes from the Report of the California Integrated Waste Management Board on *Markets for Recycled* Plastics (CIWMB, 1996).

Box 3.1. Perception of Recycled-Content Products as Inferior

> Another market barrier to increased plastic recycling is the lack of knowledge on the part of the public and businesses about plastics recycling in general. People may perceive products manufactured with recycled content as inferior because they believe that recycled content products are not as durable or as reliable as products made from virgin materials. On the manufacturing side, some product designers prefer to use virgin resin with well-known physical properties, and are hesitant to use PCR because of supply and quality concerns. However, many reprocessors have proven that they are able to meet supply and quality specifications.

In general, difficulties in the reprocessing technology, because of deterioration in material structure arising out of the presence of contaminating additives, leads to recycled products being perceived as of inferior quality. The seriousness of this is often not easily ascertained by purchasers. The reason for this is that the extent of deterioration may not be easily verifiable, and if it were it may not be possible to convince final purchasers of the extent of this deterioration if in fact any actually occurs.

However the attitude of consumers to new products, and especially products that generate a perception of being potentially uncertain in application may be a reflection of their attitude to risk. Thus, consumers are faced with a product of uncertain quality, in contrast say to homogeneous commodities or commodities where quality is authenticated (i.e. by hallmarking in the case of gold). The presence of this uncertainty can lower the valuation that consumers place on the recycled product to an extent that may appear unwarranted. This depends on how consumers respond to uncertainty.

In order to assess the potential implications of this we could consider the choice between a product made out of virgin materials and one made out of recycled as being a choice subject to risk.[46] In the case of recycled plastic products, the 'risks' to which consumers may be averse include the following types of examples:

- Unknown extent or nature of additives in plastic compounds;
- Uncertain potential for structural failure due to weakening through repeated recycling; and,
- Uncertainty of mixtures of different plastic types in collected plastic waste.

All of these will affect the consumer's perception of the quality of the plastic waste or recycled plastic product which they are purchasing as either a final or intermediate good. The allowance that should be made for this risk is measured by the risk premium. The risk premium depends upon the magnitude of the risk and by individual dislike for risk. Empirical evidence indicates that individuals have very high levels of aversion to risk, which is not easily explained by apparent probabilities.[47]

A solution that has recently been proposed for this puzzle is that of a form for individual's preferences over uncertain outcomes known as Disappointment Aversion Theory (DAT) (Gul, 1996). According to this theory, which has gained widespread acceptance in analyses of a wide variety of markets, people often attach greater value to losses than gains (Camerer and Ho 1994). In such circumstances, only a small probability of an adverse outcome (i.e. contaminated plastic waste or a low-quality recycled plastic product) occurring would lead to the risk being assessed almost as though only the bad outcome matters. This provides a strong case for those using a predictable product being reluctant to change to a substitute with uncertain characteristics.

4.4. Market Power and Price Discrimination

Whereas the scale of activity in the plastics reclamation and reprocessing industry is small (see figures in Table 3.6 above for USA), the firms involved in the virgin resin industry are large and are usually off-shoots of petrochemical concerns. Table 3.8 below gives capacities for Europe in 1994, and corresponding capacities for the same firms in 1997 where available. The comparison is very limited as many of the firms listed for 1994 do not appear for 1997 and vice versa.

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46. Whilst information on this type of choice is limited, there is much information on choices involving risky financial assets such as shares or equity. These give higher returns than safe assets such as Treasury Bills. This extra return is payment for the risk involved. If a consumer is faced with a choice between a product made out of virgin and recycled material, the choice of which to use will involve price and an assessment of the possible performance. This should also include an allowance for any risk.

47. Attempts have been made to measure the risk premium that attaches to financial assets, but these have been unable to explain adequately the magnitude of the differences in returns found between risky and safe assets. This problem is known as the "equity premium puzzle" (Mehra and Prescott, 1985).

From Table 3.7, it can be seen that for 1994, and each of the plastic types there are 5 or 6 firms, which dominate production. The same is true for 1997. The structure of the industry appears to be that of a small group of large producers with two fringes. One fringe group consists of smaller virgin resin producers, and another group of even smaller reprocessors. If virgin resin producers are ‘price-makers’, they may be in a position to disadvantage other possible entrants, including plastics recyclers. For instance, if there is a significant competitive threat from recyclers, virgin plastics producers can ‘dump’ their products for a period of time, until the threat has been removed.

Market power can also manifest itself in other ways. For instance, Ehrig and Curry (1992) suggest that at times post-industrial recycling may be lower than it might have been for competition reasons within the plastics industry. They suggest that as the plastics industry grew, unwanted post industrial scrap was sold by virgin producers on to independent reprocessors usually selling into different markets. Once these reprocessors began to compete with the virgin industry, the scrap was landfilled or incinerated in order to reduce competition.

Table 3.7. Capacities of Main European Resin Producers

HDPE	,000 tons/yr.		PVC	,000 tons/yr.		PS	,000 tons/yr.		PET	,000 tons/yr.
Firm	Cap. 94	Cap. 97	Firm	Cap 94	Cap. 97	Firm	Cap 94	Cap 97	Firm	Cap 94
Borealis	600	510	Aiscondel	145		BASF	545	1138	Inca	100
BP	370	-	BASF	250		BP	115		ICI	100
Dow C.	130		Buna	150		Buna	30		Eastman	125
DSM	230	520	Cires	100		Dow C.	445	1620	Akzo	90
Elf A.	190	-	EKO	100		Elf A.	450	530	Hoechst	85
Enichem	250		Elf A.	710	735	Enichem	200	225	Shell	90
Hoechst	475		EVC	970	980	Huls	220	230	Radici	20
Huls	120		Hispavic	130		Huntsman	80	842	Zipperling	5
Petrochim	430		Huls	380		Lin Pac	25		Brilen	10
PCD	200		Ind. Gen.	100		Montefina	105		La Seda	20
Repsol	215		LVM	390		Neste	12		Elana	5
Row	215		Norsk H.	380		Shell	80		Sasa	20
Solvay	330	1160	Rovin	200					Venpetil	10
Vestolen	160		Shell	205						
Others	115		Solvay	850	1420					
			Vinnolit	610	720					
Total W. Europe	4030			5670			2307			680

Source : European Commission 1997, Modern Plastics, Jan. 98.

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4.5. Signalling and Market Segmentation

Other market characteristics may also inhibit the take-up of recycled plastics. For instance, 'market signalling' may play a role.[48] Similar processes occur with other goods and services where product quality or similar attributes may be unobservable. Thus high quality cars are sold from opulent showrooms, and inferences will be made about cars sold from more basic premises. This could be used as a means of price discrimination, where a manufacturer will separate customers into those with a high willingness to pay for a good from those with a low willingness to pay.

In this case, the market cannot be relied on to provide the right range of products. This is an important market failure for which government policy may be necessary since the presence of competition in both price and quality can lead to some unexpected and perverse results. In the area of recycling, this may result in sub-optimal levels of material recovery.

In setting prices, the producer will take account of the preferences of the different consumers, and will want to 'price discriminate'[49] in order to maximise profits. For instance, if environmentally-aware consumers are prepared to pay a premium, it will be in the producers' interests to charge a higher price for reprocessed than virgin-based plastics, even if they are qualitatively identical or even inferior. Moreover, as the proportion of 'environmentally-aware' consumers increases, the quality of the recycled plastics may well fall since this will allow the producers to continue to segment the market in the face of rising demand for reprocessed plastics.

The precise impact of these different effects will depend on the parameters of demand, and even the direction of the effect on recycling levels can not be known with certainty. The conclusion to be drawn here is that just as demand and price are endogenous variables determined within the market, so is the price and quality difference between virgin and recycled products. This quality difference is limited by technical considerations.

5. Market Conditions and Price Volatility

Price volatility is the reason most often cited as the cause of inefficiency in post-consumer recycled plastics markets. Such price volatility can be a consequence of the market barriers and failures discussed in Section 4 above, since they can lead to uncertain and thin markets. In this section, we review the evidence for price volatility in secondary plastics markets, seek to find empirical evidence for the reasons for the price volatility which does exist, and undertake some simulations which examine the economic implications of price volatility in the market.

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48. The theory of market signalling was introduced by Spence (1974) in the context of hiring and other screening processes. He wished to explain why, for example, certain educational qualifications might be required for a particular job when it seemed that the education and training acquired had no particular relevance to the job concerned. His explanation was that the inherent acquisition of the qualification involved a cost which was correlated with the skill and ability of the person hired that could not be directly observed. High ability individuals can acquire qualifications more easily and cheaply than low ability individuals can. Thus qualifications can signal ability and be used as a hiring requirement.

49. This is not strictly price discrimination as that is charging different consumers different prices for the same product. Here for some consumers, the 'environmentally aware' virgin-based and recycled-alternative are different products.

5.1. Evidence of Price Volatility

Prices for secondary plastics are collected by various bodies in different countries. In this section we will review some of the data from Germany, the United States and the United Kingdom. Generally speaking waste plastic prices in all three countries exhibit considerable volatility. Moreover, such variability tends to be greater for 'lower quality' plastics, such as mixed PE plastics.

Table 3.8 gives the standard deviations of prices for different recycled plastic products in three countries (UK, USA and Germany) and compares them with the standard deviations of virgin plastic prices in the United States. In general, recycled plastic prices appear to be more volatile than virgin plastic prices, in some cases (PET) considerably so.

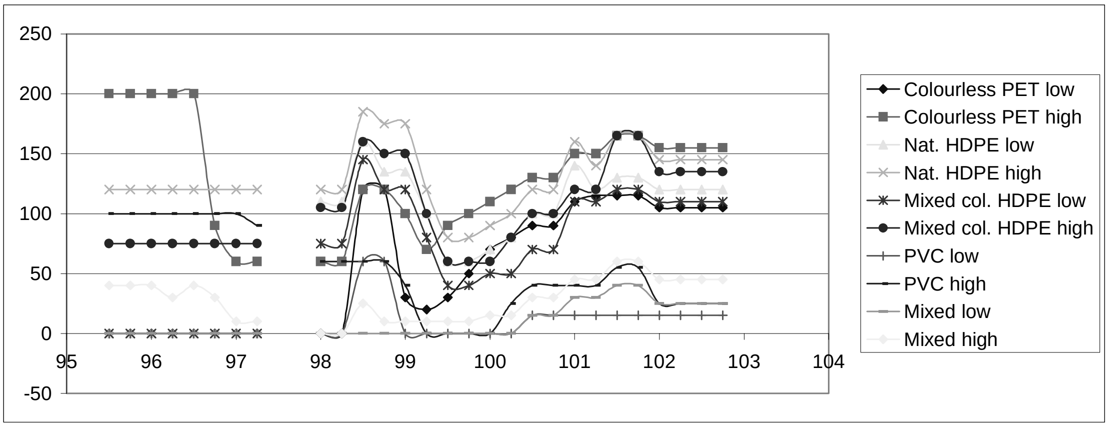
Table 3.8. Standard Deviation of Price divided by Mean Price

	USA	USA	UK	Germany
	Virgin	Recycled	Recycled	Recycled
HDPE Natural	0.15	0.19	0.26	0.14
HDPE Coloured	0.15	0.19	0.35	0.84
PET Natural	0.18	0.31	0.37	0.80
PET Coloured	0.18	0.29		
Polypropylene	0.16	0.24		0.12
Polystyrene	0.08	0.09		0.10
Mixed			0.92	2.85

Source: Calculated from Data for USA, - Plastics News, Recycling Times, UK - Materials Recycling Weekly, Germany - EUWID

Figure 3.4 shows time-series for recycled plastics prices in the UK. In general, volatility appears to have risen in the period 1996 - 2000, with a period of stability beforehand and, to a lesser extent, afterward.

Figure 3.4: Recycled Plastics Prices in the UK 1995 - 2003, £/tonne



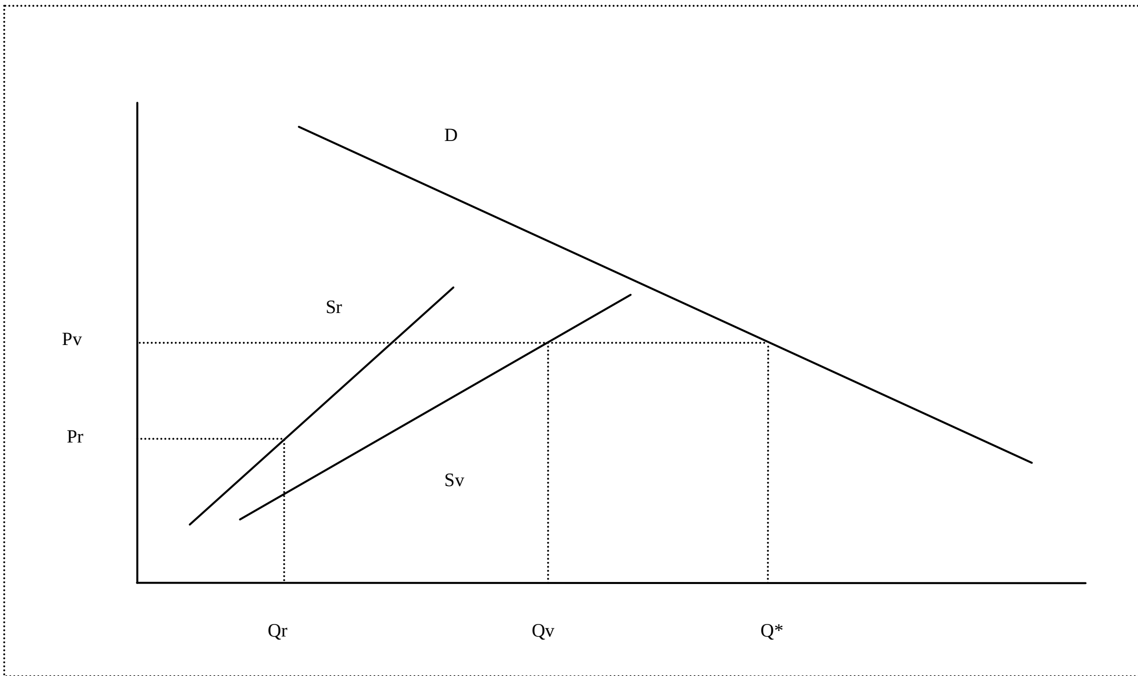
Source: Materials Recycling Weekly, various issues.

5.2. The Reasons for Price Volatility

The financial viability of reprocessing firms arises out of the relationship between waste plastics and virgin resins. Where these are close substitute goods then the two markets need to be considered in parallel as demand in one market will depend on price in the other. Thus, forces determining the level of demand in one market will affect related markets.

For many uses, reprocessed materials are imperfect substitutes for plastics made from virgin resins. In other cases, they are – at least in functional terms – almost perfect substitutes. However, as noted above, issues of quality perception, will lead to the reprocessed material selling at a lower price. The market for all material will determine an overall price level, and quality considerations will then determine the premium that is attached to virgin plastics. Equilibrium in this market is illustrated in Figure 3.5 below where total demand (derived as a function of the virgin price) equals $q_v + q_r$.

Figure 3.5: Interactions between the Markets for Virgin and Secondary Plastics



What is important for the reprocessed market is the residual demand, that is demand left unsatisfied after virgin supply at the equilibrium price. Virgin supply will be limited in the short-run by capacity in the industry, and when there are shortages in capacity then recycled material will take up all of the slack up to the new equilibrium quantity, which will result from a higher market price. This is typically the story for exports of recycled plastic material from the West Coast of the USA to Asia, China in particular.

> "2001 saw the U.S. markets for PET bottle bales dominated for the first three quarters by North American buyers and then by Chinese buyers during the fourth quarter. A strong economy allowed North American buyers to push prices to levels that forced Chinese buyers out of the market for a short period of time in May. Conversely, the Chinese took advantage of the dramatic U.S. economic downturn in the fourth quarter to purchase large quantities of bales at the lowest prices in years. It must be noted that during this period, competing Chinese buyers often drove prices higher while North American buyers were absent from the market." (NAPCOR, 2001 Report on Post Consumer PET Container Recycling Activity Final Report)

Similarly when there is excess capacity in the virgin resin industry, recycled material will only compete to the extent that it can be supplied in matching quality at the same or lower cost. As recycled material has collection, sorting, processing and transport costs this has led to the use of recycled material being low except in certain situations. This is made more severe by the general decline in virgin resin prices. The explanation that is given for these decreasing prices by trade journals is one of excess capacity. Together with the structure of the industry and the apparent nature of competition within it, virgin resin prices are pushed down – which is desirable for competition in the virgin sector, but disastrous for the recycling sector.

One particular episode of great volatility in prices has been investigated by Ackerman and Gallagher (2002). They seek to explain why prices for most recycled materials (including HDPE and PET) were suddenly extremely high in 1995 and then fell back to traditionally low levels. For example Cascadia Consulting group (1998) reported for PET and HDPE in NW USA that "a spike in prices occurred in 1994-1995 and, since then, prices have settled to levels generally higher than those in 1993". Ackerman and Gallagher's (2002) main concern is the incorrect signal that this sent to the recycling industry as recyclers expanded capacity on the assumption that high prices were a long run phenomenon. Six explanations for a possible price spike were considered:

1. Inventory fluctuations
2. Macroeconomic Trends
3. Capacity Adjustment lags
4. Government Intervention
5. Export Demand
6. Speculation

On the basis of their econometric analysis they find that there is seen to be a small influence of 4 and 5 on prices, but the main explanation for the spike in prices of recyclables was speculative behaviour (6). However, it must be emphasised that the empirical evidence is not unambiguous and that the study only relates to one particular episode.

5.3. The Consequences of Price Volatility

Price fluctuations in the overall market will have important consequences for profitability and the long term survival of firms involved in recycling. Fluctuations in the quantities of different types of materials collected will interact with this, and firms will need to be able to cope with both types of uncertainty. Such fluctuations can arise from collection where the amount of particular materials included in the waste stream will change on an unexpected basis. This type of problem will be particularly serious in the collection and reprocessing of plastics where there is considerable variability in both the type of material collected and in its price.

Various possible sources of uncertainty exist for plastics reprocessing. In order to examine these issues, a series of simulations was undertaken, in which different sources of uncertainty are modelled as independent random variables. Four aspect of randomness are considered:

- The balance between HDPE and PET material;
- The balance between natural and coloured HDPE;
- The balance between natural and coloured PET; and,
- The price of collected unsorted waste.

This is then used to calculate the revenue for a reprocessing firm, for which the composition of waste input and sorting/sales is based on the business plan for an example of an American recycler in 1993 used by the Association of Small Business Development Centres. This example provides figures for costs as well as throughput. Cost data is generally impossible to obtain, but is crucial to understanding long term viability. The data is unlikely to be particularly accurate for present day Europe but provides a base illustration for problems involved. These costs are taken to be independent of the randomness in the sorting process and for the waste price fluctuations. The random term is used to multiply the basic figure, have a mean of 1 and a standard error of 0.05. Thus the variables fluctuate between +10% and -10% of the base figure.

For the price series two sources are used. The first is 19 months of price data provided by the German Market research firm EUWID, for various grades of plastics in Germany in the period 1996-1998. In the figures below, the months are indexed by the numbers 1 to 19. The top grade of sorted material is taken as the price that can be obtained for the material in each month. The price for the mixed grade is taken as the input price for material into the sorting firm. The profitability is therefore that of the re-sorting activity. For purposes of comparison, data from the USA is used, and this may be more appropriate as the cost data for the sorting facility is from the USA.

There are three main objectives for undertaking the simulations. The first is to see what the level and pattern of profitability for sorting is likely to be. The second is to see how this profitability may vary over time. The third is to see how much of variation in profitability is due to basic price fluctuations and how much to the accuracy of sorting and the composition of waste. The answers to the first two questions are provided in Figure 3.6 and 3.7.

Figure 3.6: Simulation of Profitability of Reprocessing

(Collected material price based on EUWID data for Germany)

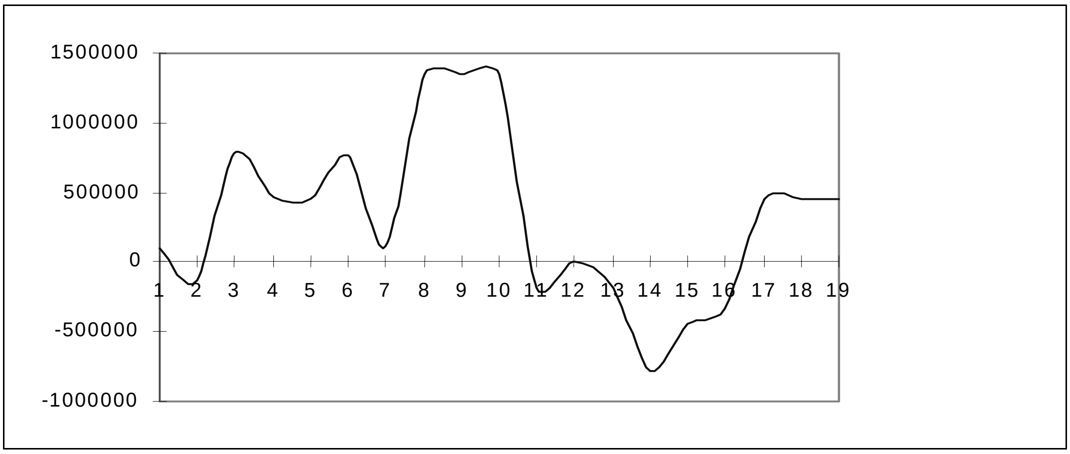
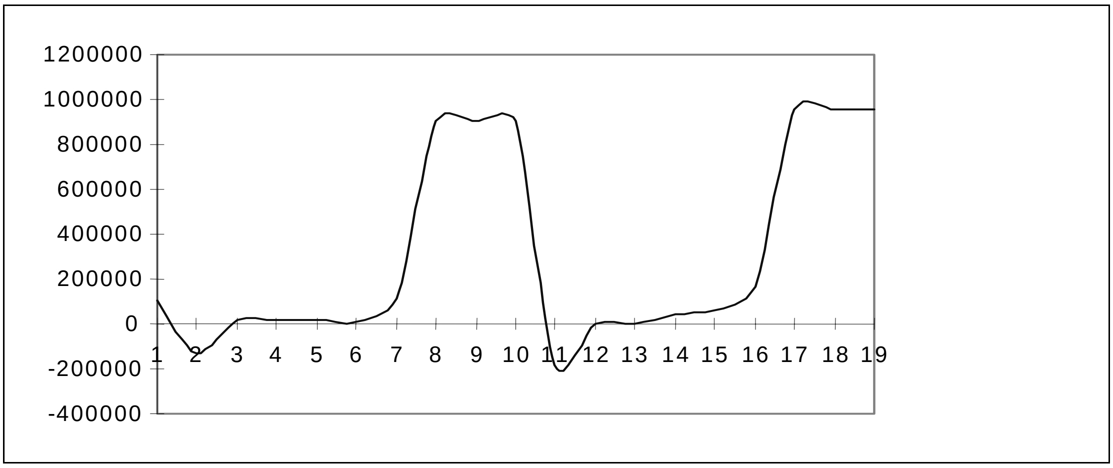


Figure 3.7: Simulation of Profitabilty of Reprocessing

(Collected material price based on data for USA)



Both simulations indicate periodic profitability. The main difference between profitability based on US prices and those based on German prices is that the former has a minimum profit of approximately zero, whilst the latter has more months of profitability and more of loss making. This may reflect the market structure and legislative differences between the two countries.

6. Policy Case Studies

In this section the markets and policy frameworks in three OECD countries (United States, Germany and Sweden) are examined in order to cast further light on how public policy may have affected the markets for recyclable plastics.

6.1. United States

The most important sources of plastic waste in MSW in the United States are durable goods, followed by non-durable goods, plastic bags and containers. LDPE is the most important plastic type in US waste, followed by HDPE, PP, PS and PET in that order (see Table 3.9.) In the late 1980's there was almost no plastic recycling in the USA from MSW. Since then, the infrastructure for post-consumer plastics recycling has grown significantly to a mature industry with more than 2,000 companies. The plastics industry sees the uncertain supply of plastic bottles from municipal recycling programs as the industry's most significant barrier to higher recovery levels. Post-consumer plastics are in demand as cost-effective raw materials for making a range of products.

Table 3.9. Plastics in US waste 1996 (1000 tons)

By Type of Good	
Durable Goods	6260
Non-durable goods	5350
Bags	3220
Soft drink, milk containers	1350
Other containers	1280
By Type of Plastic	
PET	1700
HDPE	4120
LDPE	5010
PP	2580
PS	1990
Other	3130

Source: Subramian 2000.

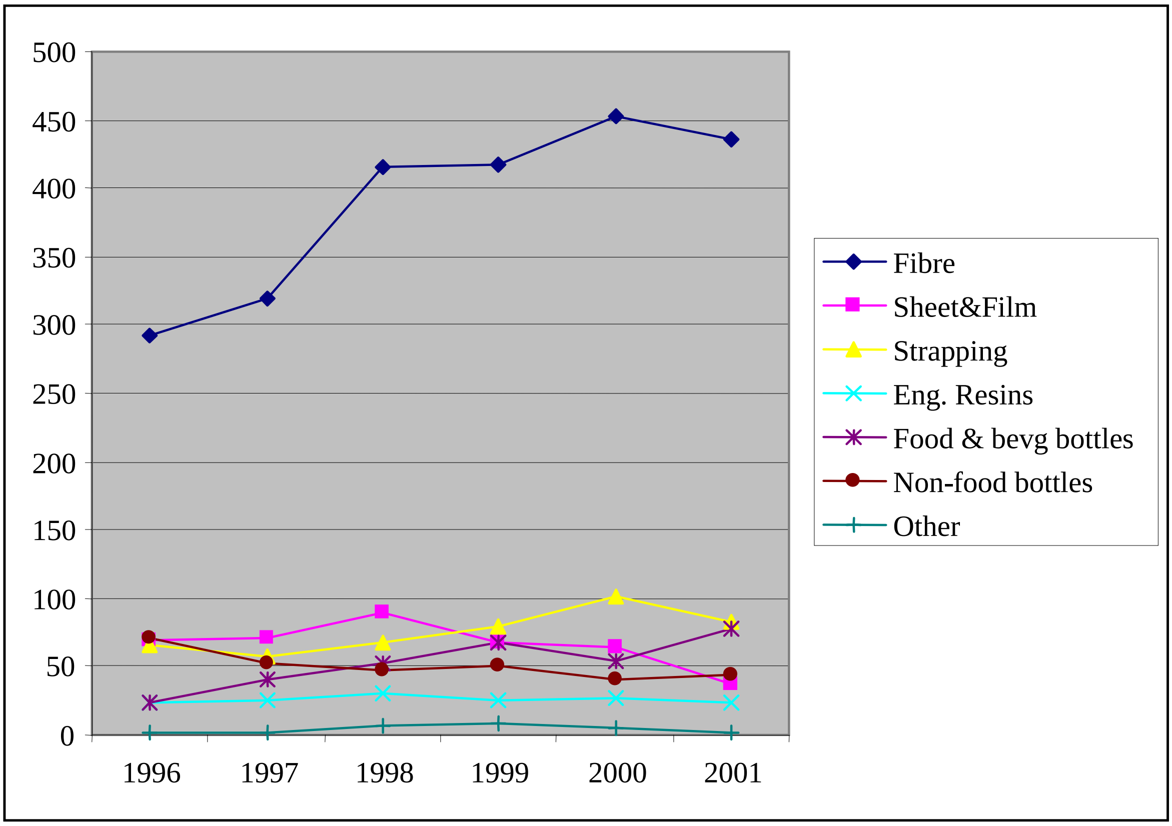
In total, 53 companies reclaimed plastic bottles during 2000. Of these, 16 domestic reclaimers reported recycling PET bottles, and 37 companies reported recycling HDPE bottles. The domestic PET and HDPE bottle-reclaiming industries are each dominated by a relatively small number of companies; however, there are nearly three times as many HDPE bottle reclaimers as PET bottle reclaimers.

Five companies, that together recycle 77 percent (478 million pounds) of the recovered post-consumer PET bottle resin in the country, dominate the PET recycling industry. The remaining 11 companies recycle 139 million pounds, with five of the eleven companies recycling 87 percent of these pounds. Similarly, there are five companies that recycle 71 percent (454 million lbs.) of the post-consumer bottle HDPE resin. Beyond those dominant industry players, the HDPE recycling industry is somewhat more diverse. Thirty-two recyclers process the remaining 29 percent of the HDPE pounds recycled. PET and HDPE bottle reclamation capacity continues to exceed supply. Based on operations on a continuous basis (24 hour per day and 365 days per year), domestic reclamation capacity exceeds utilization by 38 percent for PET and by 46 percent for HDPE.

R. W. Beck (Beck, 2001) data on the recycling of non-bottle post-consumer plastics shows that in some cases, the amount recycled is quite significant. The 2000 recycling figures for non-container plastics only reflect data provided by survey respondents and do not extrapolate to a nationwide total; however, the figures do provide insight into the size and complexity of the post-consumer plastics recycling market. A total of 167 companies reported the recycling or export of non-bottle post-consumer plastics that totalled 1,175 million lbs.

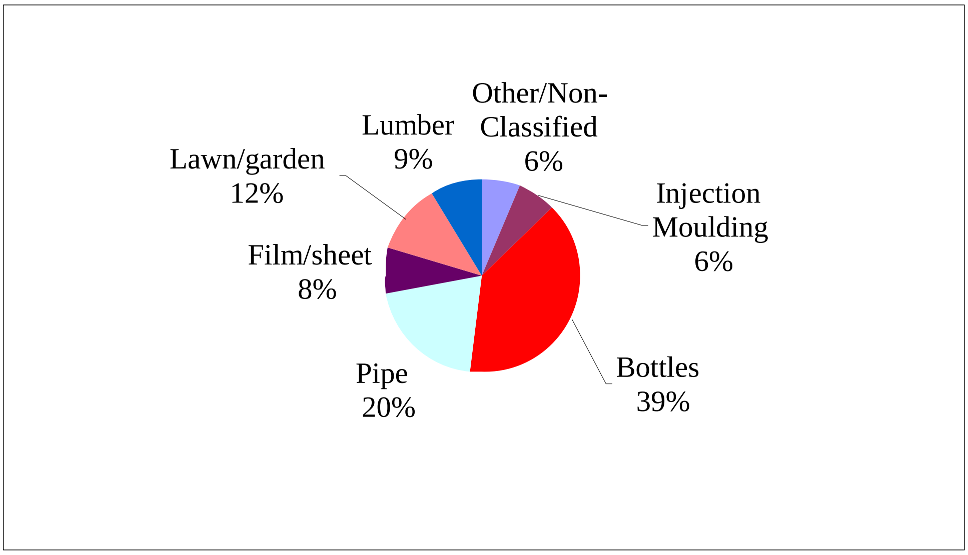
As shown in Figure 3.8 below over half of all domestic recycled PET bottle end use in the United Sates is for fibre. Strapping is the next most important use, with only 13% of the total. Very little is re-used as containers.

Figure 3.8: Demand for Recycled PET Bottles in the USA (million lbs)



Conversely, as shown in diagram below, over one-third of all HDPE plastic bottles recycled in the United States were remanufactured into HDPE bottles. Other major HDPE bottle end-markets include pipe and plastic lumber; film and sheet; lawn and garden (e.g., products such as edging and flower pots); and injection moulding (e.g., applications such as buckets, crates, and automobile parts).

Figure 3.9: End Use of Domestic Recycled HDPE Bottles in the United States



However, only 0.7 of the 19.3 million tons of plastic waste in MSW were recycled in the United States in 1993. This gives a rate of only 3.5% (GAO 1995). However, PET has much higher levels of recycling, albeit with present collection rates being much lower than their peak in 1995. (See Table 3.10.) HDPE bottles have comparable collection levels – 745.4 million lbs in 2000.

Table 3.10. US PET recycling rates (m. lbs)

	Bottles collected	Bottles on shelves	Gross Recycling rate %
1995	775	1950	39.7
1996	697	2198	31.7
1997	691	2551	27.1
1998	745	3006	24.8
1999	771	3250	23.7
2000	769	3445	22.3
2001	834	3768	22.1

It was argued by Curlee and Das (1991) that efforts towards plastic bottle recycling should be directed particularly at PET and HDPE, as these were seen as the most appropriate. Their suggestion was the adoption of a flexible approach to recycling management. They did not, in 1991, anticipate significant barriers to the expansion of PET and HDPE as collection increased, particularly for post-industrial sources. However, in the case of post-consumption plastic waste they did stress the importance of ensuring that contamination is kept to a minimum.

Whilst recycling policy in the USA is organised at a state rather than federal level, many states have looked towards the EPA to provide leadership and guidance in waste planning and recycling, not least because inter-state issues are important. State officials often want national standards to require products to contain a proportion of recycled material. State efforts are necessarily limited because markets typically extend across state boundaries.

State- and Municipal-Level Experience

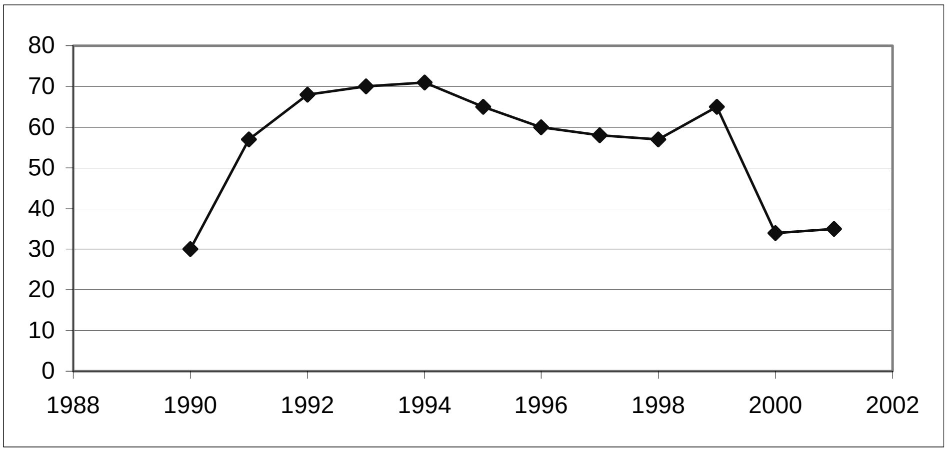
Some states and municipalities have had active recycling programmes related to plastics for a number of years. For instance, in the case of New Jersey a programme was instituted to encourage recycling of paper, metal, glass, plastic containers and food waste. However, as noted above plastics can be costly to collect and recycle, and as such most municipalities concentrated on the other waste streams in order to meet the overall objectives of the program. The relatively more positive experience in two other states will be discussed.

<u>California</u>

The California Integrated Waste Management Board (CIWMB) has recently investigated how its recycling policies have applied to the recycling of plastics, with the premise that current recycling policies have failed (NewPoint Group 2003). As such, it makes a useful candidate for a case study.

Plastics recycling has increased substantially in recent years. For instance, 26.4 m plastic containers were recycled in 1988, but this increased to 1600 m in 2001. However, this has not kept pace with the volume of plastics so that recycling rates for PET have declined from a peak of 71% in 1994 to 36% in 2001. The range of plastics has increased also so that some plastics which are not being recycled much can have considerable environmental problems associated with inappropriate disposal (e.g. PS). Currently recycling consists mainly of PET and HDPE, although these comprise 13.8% of plastic waste by weight. (Figure 3.10 gives time-series of PET recycled in California in recent years.)

Figure 3.10: PET Recycling Rate in California



Source: California Dept. of Conservation (2002)

California has a substantial recycling infrastructure. There are 241 recycled plastics processors and 8 reclaimers (the US as a whole has 60). Current recycling policy started with the Integrated Waste Management Act of 1989. This followed the EPA hierarchy of re-use first then recycling and finally disposal. Targets were set at 25% reduction by 1995 from the 1990 base and 50% by 2000. For plastics, this converted into a market capacity goal of 0.4 m ton per year, whereas the overall actual figure for 2000 was of the order of 0.05 m tons (NewPoint Group, 2003).

CIWMB (1996) sets out the market development programme that was expected to be necessary to meet the 2000 target. The first market development plan was set out in 1993, the 1996 market development plan was based on the changes that the initial plan had caused. Three areas were seen as requiring attention:

1. Collection and processing of materials
2. Manufacturer use of recycled feedstock
3. Product marketing and consumer use

Plastics were seen in 1996 as requiring special attention as the CIWMB had two specific minimum recycled content programmes, mandated by state law. One was for garbage bags, the other for rigid plastic packaging containers. Market development for plastics was also seen as being necessary as local authorities had invested in plastic collection programmes and the state wished to ensure that demand would be present to absorb this material. The 1996 plan was to ensure compliance with the minimum content legislation, to promote discussions between plastic durable goods manufacturers and plastics reprocessors, and to provide loans for the establishment of reprocessors and recycled content manufacturers. The CIWMB also provided a quarterly market report on recycled plastics monitoring developments in both virgin and recycled resins. The 1995 market report envisaged that high costs of collection, sorting, and cleaning would act as barriers to post-consumer plastic recycling (CIWMB, 1996).

Together, these costs have led to recycled resins having prices close to or above that of virgin resin as determined by capacity considerations. Whereas a 20% advantage of recycled resin over virgin is thought to be necessary to stimulate sufficient demand by plastics fabricators. Prices for recycled resins closely follow those for virgin resins and so are variable. Long term contracts which smoothed out fluctuations would help reprocessors who may be more dependent on price stability than are virgin resin producers.

Contamination of post-consumer plastics has been seen to be a particular problem in California. This comprises two separate types of problem:

> *Use of recycled plastics.* The US FDA has guidelines for the use of recycled plastic in food containers and packaging. The guidelines relate to chemical contamination in mechanically recovered plastics. The responsibility for ensuring compliance with FDA guidelines falls on packaging manufacturers. They retain liability for safety of packaging even if approval has been given for the recycling process used.
>
> *Incompatible resin types.* This can arise from uncertainty about what is recyclable. Where contamination from incompatibility arises it can reinforce the perception of recycled content products as being inferior.

There are a number of laws which provide incentives for recycling for plastics. These include:

> *B 939 Recycling and Landfill Legislation (1989)* established the hierarchy of waste provision, goals for waste diversion (25% reduction by 1995 from the 1990 base and 50% by 2000) with requirements that all local authorities meet these. It is a weight-based law, and so favours heavier materials such as paper to plastics.
>
> *SB 235 Rigid Plastic Packaging Container Legislation (1991)* which is directed at plastics and was designed to "spur markets for plastic materials collected for recycling by requiring manufacturers to utilize increasing amounts of post consumer recycled material in their rigid plastic packaging containers and to achieve high recycling rates for these plastic packaging containers." In 1996 food and cosmetic packaging was made exempt
>
> *SB 951 Plastics Trash Bag Law (1993)* which initially required that all trash bags of 0.75 mm. and greater thickness have 10% recycled plastic post consumer material, eventually increasing to 30% content. This law was replaced by SB 698 in 1998 which eliminated the 30% requirement, and introduced two compliance options, of either 10% recycled content, or at least 30% recycled content of all the manufacturers plastic products intended for sale in California.

AB 2020 Beverage Container Recycling and Litter Reduction Act (1986) which is a redemption programme for carbonated mineral and soda water beverage containers. Payments are made by beverage distributors on each container sold in the state. A payment is made to consumers who return empty containers to certified recycling facilities. In 2000 this was extended by SB 332 to cover a wider class of non-carbonated drinks, and to cover all resin types, rather than just PET and HDPE which had previously been included. In 2002, it was extended further by SB 1906, to cover an even wider range of drinks containers. The scheme now covers all food liquids except milk, wine and spirits.

SB 235, which specifies details on recycled content of rigid containers, has been difficult and expensive to monitor, and appears to have been largely ignored by small rigid container producers in the past. This may have been because of a lack of awareness of the requirement. Larger producers tended to be in compliance throughout, so that the law was not affecting those intended. CIWMB found that between 1996 and 1999 there was only 10% compliance with the act. Throughout the late 1990's there seems to have been little impact on the amount of recycling for rigid containers, and the recycling rate declined from 24.6% in 1996 to 17.9% in 1999. Only in 2000 was there an upturn in the amount and recycling rate. The law also creates incentives to switch to other forms of packaging, or to change the size in order to avoid it.

SB 951, the trash bag minimum requirement, was intended to reduce the amount of PE, from which trash bags are made, that was going to landfill. It also could be seen as part of a closed loop recycling system as the PE typically comes from waste plastic bags. It has been made obsolete by two separate means of avoidance. Firstly there are alternative uses to which recycled PE can be put. Plastic lumber is one and this has had high demand in recent years. Plastic lumber has no quality or other requirements. It is also able to take more PE from the waste stream. However, it corresponds to an open loop rather than closed loop solution. Although, as the plastic will be locked up for a much longer time as plastic lumber than as short-life trash bags this may not be a particularly serious problem. Consequently, trash bag manufacturers found it difficult to access enough waste PE to meet the 10% minimum requirement. The second problem was that the law applies only to one-quarter of the trash bags produced for use in California. Two-thirds of the trash bags produced in California are sold to out-of state users.

AB 939 has several problems in providing an adequate solution to recycling problems. The first is one common to recycling policies in several countries in that it "wills the end without willing the means to ensure that end". The second problem for plastics recycling is that it sets overall targets by weight. As light products this is biased against plastics. The third problem is that there is no incentive for alternatives to landfill for plastics that cannot - or are expensive to - be recycled, such as waste-to-energy. Overall, the aim of AB 939 in leading to 50% diversion has not been met, in part because waste generation has grown much faster than recycling activity, in part because it does nothing to solve the problem of high collection and sorting costs.

AB 2020 is crucial to supporting recycling activities in California, even though it is only concerned with 3% of total waste. Although recycling has not kept pace with the growth in plastic beverage containers, PET containers recycled through it increased threefold between 1998 and 2001. So whilst the recycling of beverage container goals in AB 2020 are only met for aluminium, this should be seen perhaps a reflection of the inefficiency of the level of the target set rather than the programme itself. For PET, recyclers receive $1140 per ton, of this $200 comes from the value of the plastic scrap collected, with $940 form AB 2020 payments. For HDPE, recyclers get $445 per ton in total with the scrap value being $185.

AB 2020 is limited in application though. For example, HDPE containers used for milk are not included in it. It also does not currently provide any incentive for the use of plastics that have existing recycled markets. Thus there is the possibility of proliferation of plastic types, which will be returned by consumers to recycling facilities, but end up being disposed as waste.

King County, Washington State

King County is the area surrounding Seattle. Plastics form only a small part of waste disposed there, and, as with California, recycling is confined mainly to PET and HDPE bottles. In 1998 the County commissioned Cascadia Consulting Group to investigate recycling markets and the possibility of public sector aid in their development. Plastic film was seen as a medium priority material for recycling as it makes up a significant proportion of commercial waste. Whilst there is some demand for the material, this is at some distance and has stringent specifications. The other area of plastics they considered was to extend the collection of PET and HDPE from beverage containers to rigid plastic containers. However, there was limited scope for this. Only one processor was willing to accept mixed rigid containers, and then charged a fee for accepting them. Cascadia focused on four types of plastic, PET bottles, HDPE bottles, rigid containers, and films.

23% of plastic containers were being recycled. This compares to a national figure of 24.5% As in other areas, virgin plastic supply is growing quite rapidly, and whilst recycling is also growing, this is at a lower rate. Overall recycling rates are therefore at best static, and usually decreasing. Supply of PET bottles was growing fastest. However, various new developments appeared imminent such as the use of composite materials such as PET with an impermeable core, PEN (Polyethylene naphthalate) and HDPE bottles with PVC sleeves (acting as a labelling device). The separation of these materials, to avoid the recycled material being regarded as contaminated, will be important if use of recycled material is to be maintained.

Baled HDPE was shipped to reclaimers in British Columbia, California or Texas. These then produce recycled resin, which is sold to processors. Most PET was exported, mainly to China. King County is well placed for this through the port of Seattle. 25% of the PET is shipped to the south east of the USA. Plastic film collected is shipped to the south west USA for plastic lumber and similar applications. Whilst there is very little local demand for post consumer recycled material, several local plastics processors used post-industrial scrap. Some products in King County advertise to consumers their recycled plastic content. For this reason they are often prepared to pay a higher price for the recycled material.

Recycled material from King County enters into the following consumer products: HDPE as 25% of the content in detergent bottles; plastic lumber; PET as fibre; HDPE and PVC in plastic pipes; plastic bags in California. In all of these there is strong competition in terms of both price (e.g. in pipes where the competing virgin resin has a low price), and in quality.

Markets for recycled plastics from King County are international in scope, and the price depends on that of the competing virgin resin. This was seen by Cascadia (1998) as being driven by oil prices, economic activity in Asia, and virgin production capacity and production in the USA. In 1998, depressed economic activity in Asia, and the growth of virgin production capacity in China, leading to the initiation of exports into the USA, was seen as depressing prices for recycled plastics. Barriers to market development were seen by Cascadia as being:

Lack of local critical mass in plastic processing and manufacturing - recycled plastics are typically shipped elsewhere - and this adds to costs;

- Increases in virgin capacity in Alberta, Canada;
- Preference for virgin plastics by manufacturers due to better performance characteristics;
- Increases in supply of imported resins and consequent lower prices; and
- Lack of infrastructure for sorting plastics other than PET and HDPE and applications for these materials.

Of these 2, 3, 4 are concerned with supply in the competing virgin industry and so are not barriers that can be changed by policy without affecting competition overall. 1 is concerned with the local geography of the area, and 5 is mainly concerned with the nature of the material. Nevertheless it was seen that there was opportunity for public sector action for market development. These were:

Monitor developments in new packaging materials so that recycling programmes were not undermined;

- Expand supply of PET and HDPE bottles and film, by improving recovery rates from consumers;
- Improve efficiency of recycling so as to both lower recycled material price and increase its quality;
- Expand procurement policies for recycled plastic products, mainly by the private sector; and

- Expand collection of and markets for plastics currently not being recycled

Of these 4 and 5 would necessitate that recycling of waste plastics into new products should take place, and that incentives be provided to ensure that consumer demand meets this supply.

6.2. Germany

Like other countries, plastics use in Germany has grown rapidly over the last two decades, although very recently there has been a marked levelling-off of production. However, Germany remains the third largest producer and consumer of plastics in the world. Production of plastics grew at a rate of 2.2.% per annum from 1976 to 1995 and consumption of plastic products grew at a rate of 3.8% per annum. (OECD, *STAN Industrial Database*). In addition, Germany is the second largest consumer of plastic packaging in Western Europe. In recent years, this market has shown considerable dynamism, with annual rates of growth averaging 7% a year. The market for plastic packaging is the most fragmented in the entire plastics industry. Germany has some 380 companies operating in this field. The highest concentration of suppliers is in the PET bottle sector, with Schmalbach-Lubeca the largest manufacturer in Europe. From 1996 to 2000, German production increased by an average of 6 % per year.

In the 1980's, plastic waste was about 5.5% of German MSW by weight. This comprised 60-65% PE, 15 –20% PS, 10-15% PVC and 5 – 10% others. There was very little PET then because of the ban on non-returnable PET bottles in Germany. Total figures for post-consumer plastic waste grew from 2,830,000 tons/year in 1990 to 3,650,000 tons/year in 1996 (Patel 1996).

Waste and Recycling Policy

Subsequent to the first "German Waste Law" of 1977, the Government sought to encourage the industries involved in packaging to agree to a standstill agreement to retain the existing reuse packaging systems. Those agreements were enacted in an informal way in 1978. In 1979-80, the Government started to run a market test where reuse packaging and different kinds of one-way packaging were tested in a competitive way in relation to their economic and environmental aspects.

In April 1989, specific agreements with the beverage industry were established as a final option before legal instruments would be brought into force. The Government analysed the developments which followed in the beverage industry, and 6 months later started to outline the first proposal for an ordinance to protect reuse packaging systems for beverages. This proposal came under pressure from a lobby of packaging producers and fillers. The outcome was the German packaging ordinance of 1991 (VVO). This ordinance covers all kinds of packaging and laid down recycling targets for primary packaging. A quota of 72% to cover all primary beverage packaging was fixed. If this quota was not upheld by the industry, a compulsory deposit on one-way packaging would be put into force.

At present, Germany has a compulsory deposit with a high recycling target (Golding, 1999). 74% of mineral water bottles in Germany are re-used. Germany has a substantial pool system covering, beer, mineral water, soft drinks, juices and milk products. The 1986 Waste Avoidance Act had a provision for banning certain items of packaging and this was used in 1987 to force the withdrawal of a one-way PET bottle for soft drinks.

The Duales or Green Dot System

The *Duales System Deutschland* (DSD) is meant to reduce disposal of packaging material either by recycling or by reduced use.[50] It is based on the concept of Producer Responsibility. Packaging was divided into different types each with its own regulation. The basic idea was that packaging should be taken back by producers. Initially, the German government planned to collect fifty percent of all packaging materials by January 1994. The second, and more aggressive step required collection by July 1995 of eighty percent of all packaging.

In response to the VVO, German industry took advantage of one section which permitted companies to call on third parties to fulfill their obligations. The result was the creation of a dual system, in which a group of over six hundred companies formed the *Duales System Deutschland* (DSD) and authorised it to work with local governments to collect recyclable packaging materials. The German government gave DSD exclusive rights to handle this business and set certain conditions on its operation. These conditions included requirements that the programme offer national coverage, locate collection bins close to consumers, establish routine collection schedules, integrate the collection plan with state and local systems and meet the qualitative goals for collection and recycling set forth in the packaging ordinance.

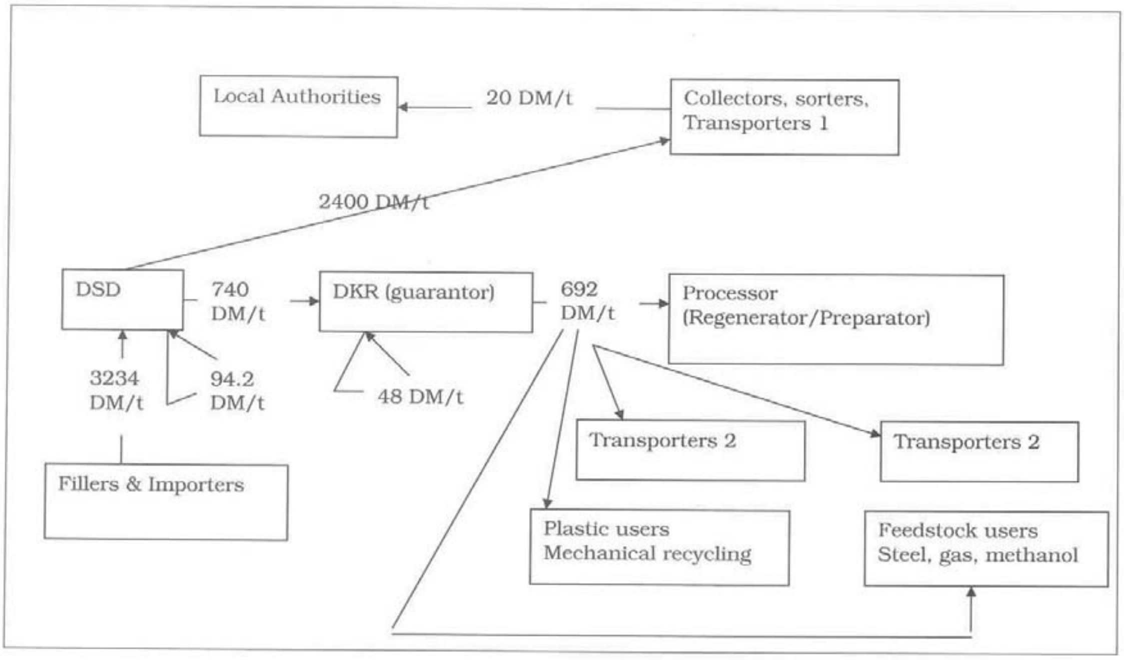
Companies who wish to participate in the DSD program, thereby complying with the German statutory take-back requirement without the necessity of creating their own system, must apply for permission to use a "Green Dot" symbol on their packaging materials. A product bearing the Green Dot is guaranteed to be composed of recyclable packaging. For a fee, DSD licenses the use of this symbol to companies whose materials DSD is willing to accept. Consumers and retailers may dispose of sales packaging bearing the green dot in DSD collection bins. The practical result is that retailers who do not want to have to send materials back to their suppliers, even if the supplier pays for any expenses, insist on using the symbol. VVO requires collection facilities at or near stores and return to the manufacturer.

To join DSD there is a fee per package based on packaging volume. As DSD also organises recycling there is a cost for that, but for most materials it was envisaged that reprocessing revenues would be sufficient to pay for costs incurred. However, this was not thought to be likely for plastics and so a separate organisation VGK was set up to recycle plastics. Packers and fillers make arrangements with VGK for reprocessing. These fees are much higher than for Green Dot. A charge of 0.02 DM was made by DSD for containers of 0.2l. to 3l., whereas the VGK fee is 0.5 DM. This makes the overall Green Dot payment for plastics much higher than for other materials. The figure below shows the financial flows arising out of the Green Dot.

Up until 1998, the reuse quota remained above 72%. Thus PET forms only a small part of waste, which does not have a direct recycling/re-use channel. The highest reuse rates occur with mineral waters (88%), carbonated soft drinks (77%) and beer (78%). 36% of non-carbonated soft drinks like juices and iced tea etc. and 28% of the wine are sold in reuse bottles. Reuse packaging systems were faced with a straight decline beginning in the mid 70’s when one-way packaging entered the market. The soft drink market is much more heterogeneous, with carbonated soft drinks being bottled by mineral water bottlers, breweries and specialised soft drink producers like Coca Cola etc. The decline of reuse systems was even faster and fell to a market share of 72% in the early 1990’s. After legal action was taken against one-way plastic bottles in 1989, the introduction of reuse PET bottles brought a new increase to the reuse market share.

50. A basic outline of the “Green Dot” system is given by Klepper and Michaelis (1994), Reynolds (1995) and OECD (1998).

Figure 3.11: Financial Flows for Plastics Recycling in Germany)



Source: Taylor, Nelson, Sofres Consulting (2000)

Assessments of the DSD System

Many concerns have been expressed about the DSD system, including the extent to which DSD limits competition. There are two ways, at least, in which this might happen. One is the monopoly position of the DSD system in waste management and recycling in Germany (OECD, 2002). The other is as a trade barrier, for example the concerns expressed by the FAO in relation to the import of fruit and vegeTable 3.s into Germany especially from outside of the EC (Carter, 1997). A second concern is about economic efficiency, and whether the goals being set are the correct ones. These are covered more fully in the next section. The main concern is that the high recycling rate comes at a high cost and that this is maintained only by high fees for use of the DSD system. Feedstock recycling is expensive and only justified by the need to process large quantities of material in ways which count as material recovery rather than energy recovery. It can be argued that there needs to be a wider interpretation as to what counts as reprocessing/recycling and whether energy recovery should count towards this. Data in Table 3.14 shows that incineration with energy recovery is a substantially more economic option for plastic waste in Germany.[51]

<u>Economic Efficiency</u>

As Green Dot fees are designed to cover the collection and reprocessing costs, then the price of disposal should be reflected in product prices. Further, as different fees are charged for plastics then this

51. See para. 19 of Assessment And Recommendations, OECD Economic Survey of Germany, 2002, http://www.oecd.org/dataoecd/50/45/2484170.pdf

will direct packaging material towards packaging materials with lower disposal costs. However, various assessments in the past ten years of the economic efficiency of the programme estimate that the cost of collecting, sorting and reprocessing of plastic packaging by DSD are substantially more than the cost of incineration. Further these costs are substantially higher than the environmental benefits obtained night justify[52].

Table 3.11. Recycling Costs of DSD and Incineration Costs

Costs of Collection, Sorting and Reprocessing	Incineration Costs	Year	Source
2500 to 6600 DM per ton	150 to 330 DM per ton	~1990	Klepper and Michaelis (1994)
Up to 6,000 DM per ton	300 to 500 DM per ton	1995	OECD (1998)
€1654 per tonne	€177 per tonne	1998	Taylor, Nelson, Sofres (2000)

Where recycling is economically viable, as it is with aluminium and steel materials because of the high value of raw materials and the relatively low costs of processing, and where low processing costs and efficient markets make recycling feasible, government mandates may push industries to reduce packaging waste without enormous economic disruptions. However, when recycling faces higher technical and economic barriers, as it does with plastics, government mandates may create enormous disruptions.

The costs of DSD rose as a consequence of including plastic packaging in the scheme. Thus, despite the fact that the DSD system does take a large amount of plastic material out of the waste stream, it is costly. Figures from Taylor Nelson Sofres (2000) show that whilst DSD achieves a much higher recycling rate than, for example France, the costs are much higher.

Table 3.12. The Costs of Recycling Plastics in France and Germany

		Germany	France
Financing need			
	€/capita /yr	12.2	1
	€/tonne sorted	1654	1294
	€/tonne on market	1141	64
Recycling rate (%)		69	5
Quantity Recycled		8 kg/cap./yr.	0.8 kg/cap./yr.
HH Packaging Cons.		12kg/cap./yr.	15 kg/cap./yr.

Source: Taylor Nelson Sofres (2000)

In addition, there may be other distortionary impacts on the economy. Important considerations here are entry barriers for new firms. Indeed, there are three separate aspects of competition that should be considered: competition aspects of waste collection and recycling; competition between firms selling packaged products that are affected by DSD; and, finally competition between domestic and foreign firms, i.e. the extent to which the DSD creates a trade barrier. The first and the third seem to be most problematic.

Competition aspects of waste collection and recycling

Arguments can be made that there are likely to be restraints of competition within DSD. If companies fulfil the take-back obligations on their own, then no competition issues arise. However, DSD was

52. See, OECD Economic Survey of Germany, 2002, p. 163 and Staudt and Scholl (1999)

introduced to help realise some element of cost reduction – i.e. through economies of scale. As such, most firms use the system, and this is likely to have implications for the degree of competition. By its very nature, it introduces an element of co-operative behaviour between certain firms. Moreover, it may be controlled by small number of firms. It can represent a monopoly for recycling, with no incentive to minimise collection costs or organise in an efficient manner (OECD, 1996).

Competition in the packaging industry

Recycling companies can set up within the system. There is scope for cartel behaviour in purchases from packaging companies or of sales to waste material traders. Whilst the Federal Cartel Office scrutinises statutes of companies for cartel behaviour, the general intention of the retail trade to only buy and sell Green Dot products is a possible violation of the German cartel ban. In addition, local government and waste disposal companies have set up joint ventures with long term exclusive dealing aspects. There is a danger that this establishes a monopoly in waste disposal. DSD has attempted to extend its operations into other sectors, used sales packaging and transport packaging in order to gain economies of scale. This was prohibited by the Federal Cartel office, as it would have resulted in the elimination of small and medium sized firms outside of the DSD system.

Competition between domestic and foreign firms

DSD does not discriminate against foreign suppliers in so far as they can obtain a Green Dot on their packaging. However, in practice the quota for refillables could be barrier as imported packages are more likely to be one-way. The deposit/refund will only be appropriate for domestic firms. So a refillable quota could limit imports. A tax on one-way containers would lead to increased supply of domestic returnables and tend to leave the amount imported unchanged, so there would be no problem as long as quota is being set at the right level. Indeed, it could be that it is easier to set a tax by determining external costs, than to determine a quota on amount of refillables. An example of this is contained in the discussion for Sweden.

Another question is how the quota is divided up between importers and domestic suppliers. This could be solved by an auction of non-refillable quota. This would allow importers to buy up quota where transport costs justify this. But a question remains whether refillables are more environmentally friendly than non-returnable when substantial transport costs are involved. A more general problem is the consequences of uncoordinated policy. There may be merits to the German policy on its own. But there can also be adverse consequences of uncoordinated policies. These come through distortions of trade and trade barriers. To some extent, the recycling system will serve to give domestic industry an advantage. Prices of recycled material will be lower than they otherwise would. The main advantage for domestic firms is the way that foreign firms are disadvantaged by having to pay the non-refillable packaging charge rather than that corresponding to returnable packaging.

Environmental Effectiveness

The German system has been remarkably successful at collecting waste materials. In general, the rate of mechanical recycling of plastic waste in Europe is fairly low at 11%, but in Germany it is 29.9% (APME, 2001 and EPA Ireland, 2002). For high quality mechanical recycling, it is necessary to ensure that waste plastics are not contaminated. This is ensured by segregation at source, as well as good collection and recovery schemes. The high levels of recovery and mechanical recycling in Germany is only achieved by efficient and widespread bring sites and kerbside collection schemes.

The German government believed that demand for recycled materials would develop because of the operation of markets, led by consumer demand, for packaging and products made with recycled materials. Since consumers have no incentive, beyond their own personal commitment, to demand less packaging and/or recycled content in packaging, the system may not discourage wasteful practices. Consumers, who could be charged based on how much they throw away, instead can avoid costs by using specified collection bins. DSD has produced an overabundance of plastic material, leaving Germany with the challenge of selling this plastic in a soft world market.

DSD fees are 2.5 times the price of virgin polymers and it would be expected that this has an effect on plastics use. It seems that the effect has been to reduce the growth in plastics packaging compared to other countries. Since the establishment of the DSD system, the consumption of packaging materials has declined by 12% overall, and for plastics by 4%. Indeed, the DSD has made the German recycled plastic market the largest in Europe.

However, it remains the case that only 20–25% of post consumer plastic waste overall is recycled. Post consumer waste is 3 to 3.5 times post-industrial, which has little scope for increases. It seems that plastic packaging has grown at a lower rate in Germany than in other countries. However, there are no direct incentives for consumers to recycle as there are no waste disposal fees. Patel (1998) reports that, in 1997, plastics packaging handled amounted to 800 kt or about 60% of the total plastics packaging and 11% of total plastics consumption. DSD handles 20 –25% of total plastics waste and 70% of total plastics recycling.

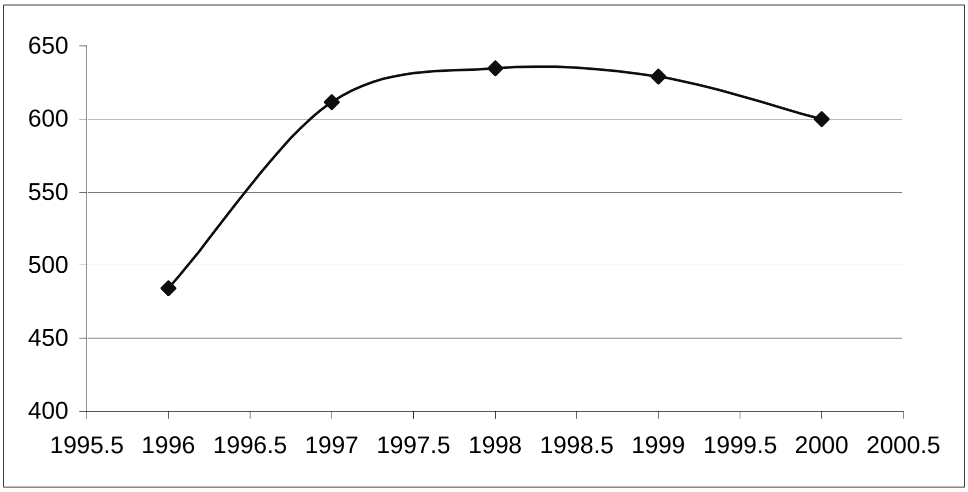
Table 3.13. Plastics Packaging Recycling under DSD

Total Recycling	Sales packaging recycled by DSD	Non sales packaging recycled by DSD	Total packaging recycling %	Sales pack recycling. %	Non sales packaging recycling %	Recycling rate by DSD %
675	580	96	45	69	14	71

Source: Taylor Nelson Sofres 2000.

Indeed, there seems to have been some levelling-off in recycling quantities in the late 1990s, although the figures remain high (see Figure 3.12).

Figure 3.12: Plastic Package Recycling of DSD (kt)



Source: Taylor Nelson Sofres 2000.

The severity of the problems associated with large volume waste collection varies with the type of packaging material. The biggest problem facing DSD is plastic packaging materials. Most metal, glass and paper packaging materials are readily recycled, although not necessarily profitably. Also, although DSD found end-user markets for recycled materials in Germany, it had to look elsewhere in Europe to make its recycling program profitable. This poses some problems for foreign companies that sell recycled paper and glass materials. These companies allege that German recyclers sell waste materials at extremely low prices, driving their competitors out of business. Germany's objective to get rid of excess plastic materials has been very problematic. Soon after DSD was established, attempts were made to export excess plastic materials to other countries in Europe, including France, that incinerate plastics for energy recovery

Conclusions

One weakness of the German system is that its goals are not based on a cost-benefit analysis and, as such the level is not chosen on the basis of an economic rationale. (The dominant hierarchy of waste management gives recycling a priority, and so to some extent is pursued for its own sake.) Whilst Germany achieves high recovery rates, the DSD system shows that this is expensive, and substantially exceeds the costs, including environmental costs, of alternative waste disposal methods. The targets set by the German government imposed have perhaps been too ambitious. High goals may be reasonable for materials like aluminium and steel where recycling is technically and economically practical, but such goals may not be practical for materials like plastics. Moreover, no incentives at the level of the consumer are provided, other than those associated with the Green Dot labeling on products.

6.3. Sweden

Reidy (1992) reports that the average composition of waste in Sweden in the years up to 1990 is 35-45% paper, 8-10% plastics, 20-40% other assorted recyclable materials, and 25-35% food and garden waste. A high content of paper and increasing content of plastics with reduced content of metals leads to high calorific value of 10-12 MJ/kg. Packaging waste accounts for about half of all municipal solid waste in Sweden. Around 35 billion packaged products are sold each year. Attempts have been made to divide domestic waste into fractions, and to segregate products at source. Three different systems are operated:

- Households separate waste into four fractions prior to collection
- Source recovery from the ordinary collection of refuse
- Recovery of resources from centrally located facilities

All municipalities in Sweden are served by a drop-off system for packaging collection. (One drop-off point / 1100 inhabitants). 5-10% of households are served by door-to-door collection. Collection is undertaken mainly by private companies[53].

The packaging collection scheme affects almost the whole country, despite parts of Sweden being very sparsely populated with long distances from large centres of population. For waste from companies there is a system with 32 delivery points, where they are paid for their sorted plastic packaging according to quality and quantity. Sorting into different type of polymers is done by four privately owned sorting plants. Rigid packaging from households and companies are sorted. Sorting is done according to demands from the recycling industry.

53. EPRO(2001); DKR (Deutsche Gesellschaft für Kunststoff-Recycling) Recycling plastics in Sweden, Markets / International / Country portrait, http://www.dkr.de/en/maerkte/212.htm

Table 3.14. Collection (tons)

Year	LDPE Transport film	Total Collection of Plastic Waste
1997	13689	31672
1998	13219	43158
1999	15437	48074
2000	16071	45641
2001	15163	41823

Source: EPRO(2001)

Derry (1989) reported that for Sweden the plastic content of waste is between 6-10%, and around 24kg per person. PVC content was 0.4% and waste also contains PET from various forms of PET bottle. In 1999 just over 50% of plastic waste was used for energy recovery, with just over 10% being recycled and the balance unrecovered (APME 2000a). Attempts were made to recover paper and plastic as material resource recovery, as well as in the form of Refuse Derived Fuel (RDF). As the quality, yields and markets for recycled plastic material were all inadequate at the separation plants, attention had switched to RDF.

Public awareness of environmental issues is generally high in Sweden. There is strong public support for measures to tackle waste issues (demonstrated by a good response to the user delivery system for collecting glass which has been in operation since the 1980s) and industry acceptance of the principle of producer responsibility for waste. Although the Swedish Parliament first recommended legislation on producer responsibility in 1975, it was only introduced into Swedish law with the Ecocycle Bill in 1993. The objective of the Ecocycle Bill is to improve the management of materials so as to reduce the high consumption of material resources and mitigate associated environmental impacts, through the re-use, recycling and recovery of energy from materials. It makes manufacturers responsible for their products after the end of their useful life.

The Ecocycle Bill sets recycling targets for packaging. Agren reports that a number of actors interviewed, for her *Environmental Assessment*, agreed that the targets were political in nature. The Centre and Green Parties were keen to establish stringent legal requirements for producer responsibility for packaging, with ambitious recovery and recycling targets (Agren, 2000). The Swedish Environmental Protection Agency conducted a number of studies including: life-cycle assessments, to assess the environmental effects, cost effects and technical options for these targets. These suggested a softening of the targets for 1997. This was based on an assessment of the technical feasibility of reaching the targets for plastics and led to the target being reduced to 30%.

There is concern over the potential harmful impacts associated with landfills and reducing the use of landfills is one of the goals of national environmental policy. There is also some concern amongst sections of the public over the environmental impact of the incineration of waste. There are currently 21 waste incineration plants in Sweden, all with energy recovery programmes. In 1994, 1.2 tonnes of MSW were incinerated.

The Packaging Ordinance

A specific Packaging Ordinance came into force in October 1994. The Ordinance requires producers, including manufacturers, importers, producers of packaged goods and retailers, to collect, re-use and recycle packaging waste. From 1 January 1997, 90%-95% of glass bottles must be refilled. The Ordinance also sets quotas for recycling other materials. This ranges from 30% for paper to 70% for glass. “Recycling” legally means resource recovery or incineration with energy recovery. The Ordinance sets targets for the year 2001 of between 50 and 65% of materials to be recovered and between 25 and 45% to be recycled. A minimum of 15% by weight of each material type must be recycled. The bill also establishes the following waste treatment hierarchy:

- re-use,
- material recovery,
- energy recovery, and then
- landfill.

However, it is stated that the method chosen should be that which makes the best use of resources. The Ecocycle Law provides the government with a mandate to demand more producer responsibility where this can lead to resource use which is environmentally beneficial and technically and economically viable. The aim of producer responsibility, laid down in the Ecocycle Bill, is to reduce the amount of packaging waste by reducing the use of packaging and promoting re-use of the waste. Producer responsibility is seen as providing incentives for:

- cleaner production;
- the development of products with a better environmental performance;
- recycling and re-use of materials and improved use of waste products; and
- minimising the use of landfills.

Producer responsibility is one means of internalising the waste externality, and should lead to the waste disposal and recycling costs being included in packager costs, and reflected in consumer prices. Where targets for recycling are added to producer responsibity, then a command and control rather a market solution will arise. Legislation also requires consumers and other waste generators to sort packaging from other waste at source and leave it for collection by producer. Separation should be between rigid and flexible packaging. This is done by contractors who organise waste collection stations in residential areas. These have five collection bins one for separate materials. Major commercial users recycle through recovery centres.

The legislation set the recycling rates for plastic at 30% excluding PET bottles, in 1997, and 70% recovery with 30% recycling in 2001. PET bottles are covered by a deposit return system. Packaging waste amounts to around 200,000 tonnes. It includes containers such as bottles and crates, but also shrink and stretch film. 90% of packaging plastic waste is PE or PP. The intention is that packaging similar to plastic bottles will be recycled into a timber substitute. This allows for mixtures of colour and of type of material. Plastic film is intended to be melted to form granulate which can be re-used in production of virgin plastic. There remains the problem though of contaminated and mixed plastic material, and the schemes intention is that this will be incinerated for energy recovery. Unlike Germany, producers are not obliged to take back packaging. Instead, they are obliged to organise and finance collection schemes with minimum goals.

The REPA scheme was established by representatives of the sectors affected by the Packaging Ordinance. The firms were in favour of the ecocycle approach and producer responsibility but wanted the flexibility to determine the means by which the targets set should be met. The producers were concerned about the costs of waste collection and treatment, which would be pushed up by separate collection, re-use and re-cycling. It was important for them to have control over these costs, rather than have a charge imposed by municipalities or central government. In particular, the representatives were keen on establishing control over the whole waste stream, including collection, so as to control costs.

In the absence of a system for spreading responsibility for managing packaging waste, the burden of complying with the Ordinance would probably have been borne by the retailers. However, all companies who produce, use or sell packaging must provide a system for the collection and recycling or re-use of that packaging and report to the EPA. If they are not registered with the materials companies via REPA, they must have established an alternative system. This provides a strong incentive to register under the REPA scheme.

8,200 companies have registered with the 'REPA' scheme. For a fee, determined by the amount and type of packaging used by the producer, each firm can register with the appropriate materials companies, who undertake to meet the company's legal responsibilities as established under the Ordinance. Fees for recycling plastics are based on weight SEK 1.5 per kg to REPA which co-ordinates the system collecting data on how the system works. This provides an incentive to use thinner and lighter plastic packaging, which may result in its own waste problem. There is also a one-off affiliation fee of SEK 400, and fees for the different types of packaging based on the amount used. For companies with a turnover below SEK 3 million, a standard annual charge can be paid instead. This amounts to SEK 500 per year for a turnover below SEK 0.5 million, and SEK 2,000 for a turnover between SEK 0.5 and 3 million.

Table 3.15. REPA Fees by Material Type

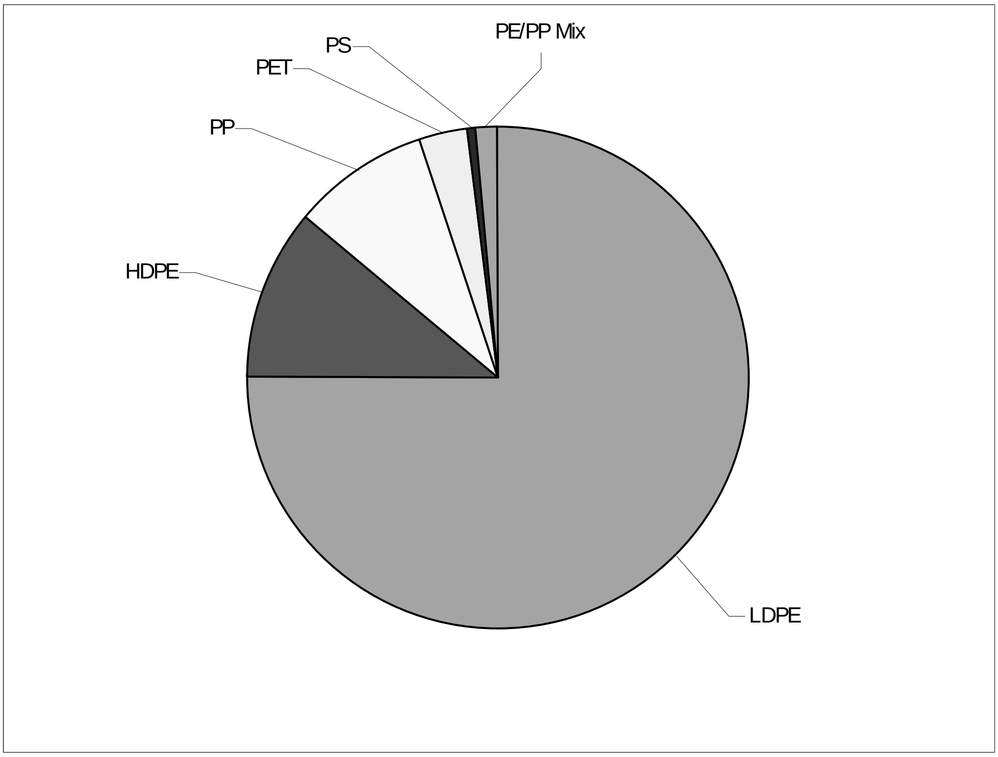
Material	Fee
Metal	100 öre/kg
Plastics	150 öre/kg
Paper/ board	40 öre/kg
Corrugated board	12 öre/kg

Source: Agren (2000)

Companies have the option of registering with and paying the fee to REPA, or setting up their own collection, recovery, recycling and reporting system to fulfil their legal obligations under the Ordinance. Charges are levied by this body on packaging, and are paid by packers or importers of packaged products. Thus the system is based on the idea of internalising the social costs of waste. The materials companies provide collection bins for the centralised collection of packaging waste. There has been some debate over the issues of bin numbers and siting. Although the material companies need to meet the targets set out in the Ordinance, the provision of more bins increases their investment and collection costs. Municipalities are still responsible for the door-to-door collection of municipal solid waste; the less packaging waste collected, the higher their costs per ton/volume of waste. The companies originally intended to provide one collection point per 10,000 inhabitants. Some municipalities have negotiated the provision of one per 1,000 inhabitants.

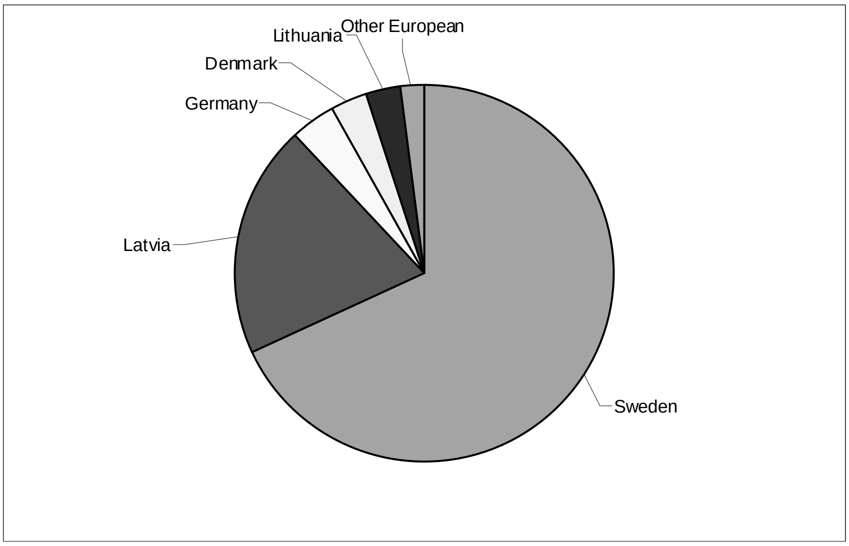
The packaging industry has set up five organisations to organise collection, sorting, and recycling, both from households and commercial packaging, allowing for competition between the materials on the basis of cost. The organisation responsible for plastics is Plastkretsen AB. Plastkretsen ("Plastics Circle") organises the recycling of plastics and its re-use, apart from PET bottles which are included in refundable deposit schemes. Plastkretsen is jointly owned by Association of Swedish Plastic Producers and Converters (60%), Fillers and Packers (15%), Trade and Retail Syndicates (15%), Petrol Stations (10%). It recycles all plastic packaging (household & commercial) which are collected by Plastkretsen.

Figure 3.13. Sales of Recycled Material Collected by Plastktretsen (2000)



Source: EPRO (2001)

Figure 3.14: Destination of Recycled Material from Plastkretsen (2000)



Source: EPRO (2001)

From households there is separate collection of:

- Rigid packaging for mechanical recycling (bottles and cups);

- Flexible packaging for energy recovery (films, bags, vacuum packs);
- From industry and trade there is separate collection of;
- Transport film (LDPE) for mechanical recycling;
- Rigid packaging (drums, big bottles and barrels) for mechanical recycling; and,
- Other plastic packaging for energy recovery.

The REPA scheme should allow for target-sharing between companies for given materials, by spreading the costs of collection and treatment over many firms, and allowing savings through economies of scale. In addition, the fact that there are separate companies responsible for the different materials, allows for competition between different types of materials, preventing cross-subsidisation between materials activities and allowing packaging users to opt for the least costly material.

The view of the stakeholders is that the REPA scheme and materials companies allow the Ordinance on producer responsibility for packaging to be implemented more efficiently than it would be under many other options (such as the DSD system in Germany). It is likely, given the cost-savings, that the EA has been more cost-effective than the existing municipal collection scheme. On the other hand, the system also imposes external costs to the public who bring waste to the collection points. The following table the progress made in meeting the targets. The data shows that the progress for plastics is poor compared to the target.

Table 3.16: Compliance with the Packaging Ordinance

Level of recycling and re-use (%)			Target
1995	1999	2001	January 1997 (%)
5	10.1	15	30

Source: Agren(2000) ; APME (2000a); Swedish EPA[54]

Bottle Recycling

PET bottles are covered by a 1991 Act on Recycling of Certain Beverage Containers. This requires fillers to show that they are taking part in recycling or re-filling scheme. If they fail to do this they pay a tax. This levies a charge on returnable containers of 8 ore, and a charge on one way bottles based on volume. For returnable bottles there is a different charge for each material once differing volumes are taken into account so that returnable glass corresponds to a charge per litre per trip of 0.27 ore assuming 30 trips, whilst for PET the charge is 0.8 ore per litre assuming 10 trips for a 1 litre bottle.

AB Svenska Returpack was set up to organise deposit scheme on aluminium cans. Following on from the success of this, a pilot scheme for PET bottles was set up on island of Gottland. 0.5 Skr deposit was imposed on all PET bottles 0.5-3 l and recovery rate of 60-70% obtained. Four different kinds of reverse vending machine were used and collected material was sent to Wellman in Ireland for reprocessing. The Wellman plant there converts PET into fibre, but Wellman do have clearance from the US FDA for a post consumer recycled PET for food applications. On the basis of this trial it was suggested that PET bottle recovery should be extended to the whole of Sweden with deposit of 1.5 Skr in order to increase recovery rate to 80-90%.

54. http://www.internat.naturvardsverket.se/index.php3?main=/documents/issues/prodresp/packag.htm

Soft drinks (mainly carbonated) have a market share of 60%. They are predominantly filled in re-use PET bottles of 1.5 litres. One-way bottles dominate the small volume soft drink market. The very high share of PET bottles for soft drinks and water is different from the situation in non-Scandinavian countries. Market shares in percent of bottled (mainly carbonated) of beverages (beer, soft drinks, water) sold and produced by members of the Swedish Brewers Association for 1997 is given in the table below. They have 99% of the market.

Table 3.17: Market Share (based on filled quantity) for Re-Used and One-Way PET Bottles

Packaging	Beer	Soft Drinks	Bottled Water	Total
Re use PET	0.2	32.8	25.2	18.6
One way PET	0	12.1	7.9	6.7

Source: Golding (1999)

Time series figures for soft drinks are not available but glass reuse systems declined until 1991 to a market share of around 40%. After the introduction of re-use PET bottles in the beginning of the 90's, the share went up again to around 60%. The deposit for the bottle is 4 SEK and has a return rate of 98% compared to a target for PET return of 90%. The volume of PET return has been steadily increasing since its introduction in 1991 (Golding, 2000). For imported bottles, a code label of the re-use PET-bottle shows that the bottle is connected to the Swedish system. Breweries and importers of beverages in PET-bottles need to show that bottles are connected to a system for reusing or recycling. That means the producer/importer must make an agreement with the official recovery company, Returpack PET AB.

A study of the Packaging Charge was undertaken by Millock (1994). This uses the model of Pearce and Turner (1992) to calculate a charge corresponding to a Pigovian tax for beverage containers such as plastic PET bottles. Ideally, to derive an optimal charge this would be calculated as the marginal damages from final disposal, which would be set equal to the marginal costs of preventing waste. The social cost of packaging is set equal to the marginal costs of: packaging production (a private cost), waste collection, waste disposal, litter and landfill user cost. After allowing for recycling which in this case is solely the return of packaging for re-use, and then dividing by either the volume, or weight, of packaging type a tax per unit volume or weight could be obtained. This can be compared to the actual packaging charge implemented, and the actual return rate with the target return rate to assess how close the implemented system might be to one based on an optimal charge[55].

Because of the large difference between how much the total amount of waste is comprised of packaging by weight or by volume, two sets of charges are derived. One is a weight based charge, the other a volume related charge. Further, in order to reflect uncertainty in measuring environmental externalities and the possibility that higher environmental standards may be imposed a 50% enhanced volume related disposal cost is also used. Various data problems arise. It is not possible to obtain data on marginal costs so data on average costs is used instead. Data for waste disposal was derived from the charges levied on households for waste collection. Data on litter showed large variability and so these costs were excluded and landfill costs are also left out.

Table 3.18 gives the calculations of Millock for one-way and returnable PET bottles. The first three rows are the tax in ore per litre (or kr/100l) calculated on the basis of 1990 data. The final row is the charge actually levied in 1990. The results show that the actual charge was substantially more than the Pigovian calculation would suggest. The low level of the Pigovian tax in relation to the overall price of the product

55. The taxes calculated by Millock follow the methodology of those advocated by the OECD for use by the DSD in the German Green Dot System (see OECD, 2003 p. 165).

would provide little incentive for changes in consumer behaviour. However, the difference between a one way and returnable charge could be sufficient to alter the behaviour of producers in favour of returnable PET bottles. The high level of the charge would be justified if the environmental costs of litter and landfill, which are excluded from these calculations, were sufficiently high.

Table 3.18: Assessment of the Packaging Charge on PET bottles (kr/100 litre)

	Collection cost	Disposal Cost	Returnable 1.5 l. PET bottle	One Way 1.5 l. PET bottle
Pigovian Tax	Weight related	Weight related	0.72	3.92
	Weight related	Volume related	0.92	5.05
	Weight related	Volume related 50% higher costs	1.1	6
Packaging Charge			5	36

Source: Millock (1994)

If the charge is high in relation to the optimal level, too much will be done by way of return and waste reduction. Comparing the amount of return that the charge generates to the level that the charge was intended to achieve will provide further evidence on this. Golding (1999) reports that, in 1997, the return rate actually achieved for PET was 98%, whereas the target rate was set at 90%. So on the basis of both the packaging charge being higher than the Pigovian Tax calculated by Millock, and the target level for PET return being exceeded, it might be concluded that at a 98% return rate perhaps too much in the way of PET bottle return is taking place.

7. Conclusions and Policy Recommendations

The light-weight and wide range of resin qualities make plastics an effective material for many applications. The lighter weight of plastics packaging has benefits in transportation as well as reduced material in the waste stream. On the other hand, these same characteristics can make plastics difficult to recycle. Plastics packaging resin sales are increasing about four times faster than plastic packaging recycling. Since 1995, U.S. plastics packaging resin sales have increased at an average annual rate of 5.9 percent, while plastics bottles recycled have increased at an annual average rate of 3.4 percent plastics as a percentage of containers and packaging.

Newpoint (2002) report that in the USA, approximately 25 percent of plastics resin sales are for packaging. The share of plastics in the total packaging generated has increased from less than one percent in 1960, to 14.7 percent in 1999 – the percentage has increased steadily each year. The share of plastics in the amount of packaging waste discard has also increased – from less than one percent in 1960, to 21.1 percent in 1999. Packaging has made up about one third of the MSW generated over the last 40 years.

Prices for recycled PET and HDPE rise and fall with virgin resin prices. There are three primary factors influencing virgin resin prices:

(1) the price of natural gas and petroleum;

(2) available production capacity relative to demand; and,

(3) general economic conditions.

Plastics markets for both virgin and recycled materials are part of a global market, and subject to much greater volatility than markets for many other recycled materials. Plastics recycling and markets are heavily impacted by the export of plastics to the Pacific Rim – however the demand for plastics in Asia is increasingly being met by virgin PET production capacity in China. In particular, California is very dependent on the plastics export market, 70-80 percent of the PET is exported; 40 to 50 percent of the HDPE, and all the other resins are being exported. But the export market is very volatile. Such volatility may have restricted the development of recycling infrastructure.

Nonetheless, post-industrial plastic recycling markets appear to be functioning relatively well, and recycling rates are high. Where there is very clean waste (and quality is easily verifiable), such as production scraps, this is already re-used at a high rate. There are established markets in post industrial scrap. Suggestions have been made that competition between virgin producers and producers using recycled resins might have limited the amount to which this scrap is traded. However, overall, post industrial waste plastic markets appear to exist and to work.

For post-consumer waste, there are some working markets particularly for PET, which result in high levels of recycling. There are strong markets for recycled plastics (particularly PET) in fibre, including carpets, clothing, and strapping, as well as markets in other applications such as furniture, buckets, bins, drums, lumber, cassette cases, and drainage pipes (particularly for HDPE). However, it is often asserted that there ought to be greater use by consumers of recycled material, and that demand is constrained because consumers have an incorrect and irrational perception of the quality of products containing recycled material. This is thought to be especially important where the recycled product enters as a final good. Where recycled material enters as an intermediate good, consumers are unlikely to be aware of its presence. This can explain part of the differences in success of recycling different materials. The attitude of consumers to new products - and especially products in which there is a perception of potential uncertainty with respect to suitability for certain applications - may just be a reflection of a consumer’s preferences concerning risk. Consequently, there may be not much that be achieved by policy motivation to address this.

Contamination is clearly a serious problem for material obtained from post-consumer waste. Contaminants such as caps, labels, adhesives, and dirt are not volatised during remolding, as they are for other materials, thus requiring manual and mechanical upgrading.[56] In addition, plastics sorting costs could increase further in the future with the increase in single- serving containers and alternative colors and barriers The use of barriers made of non-PET materials in PET containers provides enormous growth potential for the PET market, however there are concerns about the impact of barriers on recycling PET. Additionally, recycled materials have limited uses in packaged food and pharmaceutical products, which are significant markets for plastic packaging. The characteristics of the end product made from plastic resins depends crucially on the presence of additives. Even a small amount of additive often makes a big difference. While their application makes plastics an attractive material, it introduces considerable problems for recycling, for example in the case of PVC. This problem is even more severe when dealing with mixed plastics in waste.

Thus, plastic packaging materials are difficult and expensive to recycle. Because different resins cannot be mixed, the materials must be sorted and processed separately. This labour-intensive process increases costs considerably. Even when prices for recycled materials are favourable, many companies cannot use recycled plastic because it is of lower quality and consistency than virgin material. Because of the lightness of plastic waste, transport costs are important in determining whether recycling will take place, and whether or not it is beneficial overall. For example, LCA analyses of plastics recycling,

56 . This is because the melt temperature for plastics is low – 210F, relative to 1500 for aluminum, 2800F for glass, and 3000F for steel.

undertaken by Craighill and Powell (1996) find that net returns are negative: –£7 per tonne for PET, –£3 for HDPE, –£4 for PVC. These negative results arise from transport costs outweighing emissions savings arising at the manufacturing stage. Together, these costs make recycled plastics as costly as virgin resin. There is, therefore, little economic incentive to use potentially contaminated PCR if virgin is available for a similar or even lower price. Virgin plastics prices fluctuate, and tend to be low relative to the cost of recycled plastics.

Hence the "one size fits all" policy, which is often advocated, is unlikely to be satisfactory. This approach has led to a universal hierarchy of waste management approaches, which puts material reduction first and then recycling/reuse. Incineration for waste to energy comes below. Incineration for waste to energy is generally opposed by the public. It is, however, not clear that pollution and environmental problems are as bad as they are perceived. One aspect of the forcing of recycling as a solution through a hierarchy of waste management solutions has been that recycling has been encouraged, whether it is economically viable or not. Perhaps the most notable example is Germany, where recycling has taken place through expensive feedstock recycling. Whilst Germany achieves high recovery rates, the DSD system shows that this is an expensive approach.

Conclusions from life cycle analyses depend on whether mixed waste or individual plastics are being considered. Bruvoll finds recycling a more expensive option than incineration or landfill. European Commission (1997) find that whilst for 7% of plastic waste, mechanical waste recycling is economically profitable, an extra 23% to 33% is technically possible but not economically justifiable. In this case, incineration with energy recovery is seen as an appropriate strategy. Whilst incineration may not lead to a net profit, it may well be the least costly waste management option. Whether such waste should be generated in the first place will depend on the overall net benefits of the material and the feasibility of direct solutions such as a product/packaging tax to capture the environmental externalities. Waste-to-energy should be considered as a potential option, though it may not be best solution in all locations. If regulated properly, all of landfill, incineration and recycling can be suitable solutions. Material recycling turns waste into raw materials. Incinerators can be utilized to recover energy created by burning wastes.

Thus material recycling through market development should be seen as just one part of the overall solution for waste management, and the most important policy initiatives are clearly those which internalise environmental externalities. If recycling is environmentally beneficial this will provide a spur to the sector. However, given the nature of the market for used plastic packaging there are other measures which can be considered. For instance, the presence of technological externalities which decrease recyclability to a sub-optimal should be addressed. To take one example mentioned in Chapter 1, getting the balance right between the hygiene objectives and environmental objectives requires good co-ordination in the area of product standards since the two may have contradictory effects. Deposit-refund measures may also be applied.

Support for improved product design (e.g. Design for Environment) may also play a role, but this places a large information burden on public authorities, not least because consumer reactions to changed design need to be assessed. Upstream support for the development of sorting recycling technologies which allow for the recovery of mixed and composite plastic waste may also be valuable, but it should be noted that this will not encourage optimal product design. Indeed, such measures provide indirect financial support for design which incorporates technological externalities.

Information failures and transaction costs should also be addressed. For instance, it has been noted that plastic wastes are one area where the benefits from improved grading schemes are significant (SRMG 2000). This gives buyers and sellers a clearer idea of what is being exchanged – a valuable service for heterogeneous commodities such as plastic wastes. However, this needs to be reinforced by the provision of incentives to discourage unscrupulous trading. Papineshi (2003) has pointed out that even a few 'lemons' in plastic waste supply can undermine the market. This responsibility extends beyond the environmental policy sphere, and into more general issues of market regulation.

Within the environmental policy sphere it does appear that a case can be made for helping to overcome some misperceptions about the quality of products derived from plastic waste (ECOTEC 2000). Amongst other examples, the case of plastic lumber is often cited. Demonstration projects (eg. through public procurement) can help to overcome some of these barriers. However, in the longer run such barriers are likely to diminish through the normal functioning of the market.

REFERENCES

ACKERMAN F. Gallagher K., (2002). "Mixed signals: Market incentives, recycling, and the price spike of 1995", *Resources, Conservation and Recycling*.

ÅGREN H. (2000). Sweden: Agreement on Producer Responsibility for Packaging, AFR, Naturvårdsverket, Stockholm.

APC (2001). American Plastics Council, The Resin Review 2000 edn.

APME (2000a). "Plastics – a material choice for the 21st century: insight into plastics consumption and recovery in Western Europe 1997". APME, Brussels. www.apme.org. Association of Plastics Manufacturers in Europe.

APME (2000b). "An analysis of plastics consumption and recovery in western Europe 1998". Brussels, 14pp. www.apme.org. Association of Plastics Manufacturers in Europe.

APME (2000c) "Plastics – at work for a sustainable future". Brussels. http://www.apme.org/media/public_documents/20010727_151803/sustainable_future.pdf

APME (2001). "An analysis of plastics consumption and recovery in western Europe 1999". Brussels, www.apme.org. Association of Plastics Manufacturers in Europe.

BECK, R.W. (2001). U.S. Recycling Economic Information Study Prepared for The National Recycling Coalition. http://www.epa.gov/epaoswer/non-hw/recycle/jtr/econ/rei-rw/pdf/n_report.pdf

BPF (2003)**.** Sustainability Through Plastics, British Plastics Federation http://www.bpf.co.uk/bpfissues/Sustainability.cfm

BRESLIN V., SENTURK U., BERNDT C. (1998). "Long term engineering properties of recycled plastic lumber used in pier construction", *Resources, Conservation and Recycling*.

BRUVOLL A. (1998). *The Costs of Alternative Policies for Paper and Plastic Waste*, Statistics Norway.

CASCADIA CONSULTING GROUP (1998). *Assessment of Markets for King County Recyclable Materials*, King County Commission for Marketing Recyclable Materials.

CALIFORNIA DEPT. OF CONSERVATION (2002). Calendar Year 2001 Biennial Report of Beverage Container Sales, Returns, Redemption and Recycling . Rates. Sacramento, California May 23, 2002 California Dept. of Conservation

CAMERER and HO (1994)**.** Nonlinear Weighting of Probabilities and Violations of the Betweenness Axiom, Journal of Risk and Uncertainty 8, 167-196.

CARTER, S. (1997). *Global agricultural marketing management,* W5973E FAO Rome http://www.fao.org/docrep/W5973E/w5973e00.htm#Contents

CIWMB (1996). *Meeting the 50% Challenge: Recycling Market development Strategies Through the Year 2000*, California Integrated Waste Management Board.

CIWMB (1996)**.** *Market Status Report: Postconsumer Plastics, CIWMB Publication Number: 421-96-066, http://www.ciwmb.ca.gov/Markets/StatusRpts/Plastic.htm*

COUNCIL FOR SOLID WASTE SOLUTIONS (1989). *The Environmental Impacts of MSW Incineration*.

CRAIGHILL, A.L. and POWELL, J.C. (1996) Life cycle assessment and economic evaluation of recycling: a case study. Resources, Conservation and Recycling, 17(2), 75-96.

CURLEE R. (1986). *The economic feasibility of recycling, A case study of Plastic wastes*, Praeger.

CURLEE R. and Das S. (1991). "Identifying and assessing targets of opportunity for plastics recycling", *Resources, Conservation and Recycling*.

DERRY, R. (1989). *Plastics Recycling in North America,* Warren Spring Laboratory.

EHRIG R. (1992). *Plastics Recycling,* Hanser.

EPA (1999). *Recycling Works; State and Local Solutions to Waste Management Problems*, OSWER, United States Environmental Protection Agency. EPA530-K-99-003 http://www.epa.gov/epaoswer/non-hw/recycle/recycle.pdf

EPA (Ireland) (2002). AStrategy for Developing Recycling Markets in Ireland, Environmental Protection Agency of Ireland, http://www.epa.ie

EPRO (2001). Report on Plastkretsen: Recycling plastics in Sweden, 19/10/01, http://www.epro-recycling.org

EUROPEAN COMMISSION (1997). *Elements for a Cost-Effective Plastic Waste Management in the European Union*.

EUROSTAT (1995). Europe's Environment: Staistical Compendium for the Dobris Assessment, Office for Official Publications of the European Community

EUWID (various issues), Euwid Recycling Markets, http://www.euwid-recycling.com/

GAO (1995). *Solid Waste: State and Federal Efforts to Manage Non-hazardous Waste*, United States General Accounting Office.

GOLDING A. (1999). Reuse of Primary Packaging. Brussels: European Commission, 1999. Country-by-Country report,

GUL F. (1991). A Theory of Disappointment Aversion, Econometrica, vol 59(3), pp. 667-686

JOOSTEN L., HEKKERT M., WORRELL E. (2000), Assessment of the plastic flows in The Netherlands using STREAMS, Resources, Conservation and Recycling.

KLEPPER G. AND MICHAELIS P. (1994). “Economic Incentives for packaging waste management : The Dual System in Germany”, *Developments in Environmental Economics*.

MEHRA, RAJNISH AND EDWARD PRESCOTT (1985). The Equity Premium Puzzle, Journal of Monetary Economics 15, 145-161

NAPCOR (2001). National Association for PET Container Resources, 2001 Report On Post Consumer Pet Container Recycling Activity, ww.napcor.com New Point Group

MILLOCK K. (1994). “Packaging waste management in Sweden: a product charge”, *Resources, Conservation and Recycling*.

NEWPOINT GROUP (2002). “Issues and Solutions” Plastics White Paper: Optimizing Plastics Use, Recycling, and Disposal in California , Workshop Presentation, Scaramento California, June 24-25, 2002 available at http://www.ciwmb.ca.gov/Plastic/Events/WhitePpr2002/Presentation.pdf

NEWPOINT GROUP (2003). Plastics White Paper: Optimizing Plastics Use, Recycling, and Disposal in California , report for California Integrated Waste Management Board .http://www.ciwmb.ca.gov/Plastic/WhitePaper02/#Background

OECD (1996). Competition Policy and Environment, OCDE/GD(96)22.

OECD (1998). *Extended Producer Responsibility, Phase 2, Case study on the German Packaging Ordinance,* Paris.

OECD(2002) Annual report on Competition Policy Developments in Germany, 2001-2002, www.oecd.org/dataoecd/34/6/2489057.pdf

OECD(2003). Economic Survey of Germany, 2002.

PATEL M., VON THIENEN N., WORRELL E., (2000). Recycling of Plastics in Germany, Resources, Conservation, Recycling, vol. 6(1-2), pp. 65-90

PEARCE D.W. and R.K TURNER. (1992) The Economics of Packaging Waste Management: Conceptual Overview, Working paper WM 92-03 CSERGE

PORTER R. and ROBERTS T. (1985). “Energy Savings by Wastes Recycling”, *Elsevier Applied Science for European Communities*.

RECOUP, Survey 2000 Report, http://www.recoup.org

REIDY R. (1992). Solid Waste Recycling, Elsevier Advanced Technical

REYNOLDS S. (1995). The German Recycling Experiment And Its Lessons For United States Policy, *Villanova Environmental Law Journal* , Vol. 6 No. 1

SCHARAI-RAD M., and WELLING J. (2002). “Environmental and Energy balances of wood products and substitutes”, FAO.

SCHAUMBURG G., DOYLE K. (1994). “Wasting Resources to Reduce Waste”, *Policy Analysis 202,* Cato Institute.

SOUND RESOURCE MANAGEMENT GROUP, “The Chicago Board of Trade Recyclables Exchange: Evaluation of Trading Activity & Impacts on the Recycling Marketplace”, National recycling Coalition.

SPENCE A.M. (1974). *Market Signalling; Informational transfer in Hiring and Related Screening Processes*, Harvard UP.

STAUDT and SCHOLL (1999). The German Packaging Ordinance: the questionable effects of a fragmentary solid waste management approach, Jouirnal of Material Cycles and Waste Management, vol. 1

SUBRAMIAN P. (2000). “Plastics Recycling and Waste Management in the US”, *Resources, Conservation and Recycling.*

SWEDISH ENVIRONMENTAL PROTECTION AGENCY) (1996). Rapport 4748 *Packaging in the Ecocycle*. English Summary.

TAYLOR, NELSON, SOFRES CONSULTING (2000). *Cost Efficiency of Packaging Recovery Systems: The Case of France, Germany, The Netherlands, and the United Kingdom*.

VAN BEUKERING (2001). Plastics Recycling in China: an international life cycle approach, Paper W99/05, Institute for Environmental Studies, IVM, Amsterdam.

WASTEWATCH(2001). Wasteline Information on Plastics, http://www.wasteonline.org.uk/resources/InformationSheets/Plastics.htm

WRAP (2002a) (The Waste and Resources Action Programme), Plastic Bottle Recycling in the UK, www.wrap.org.uk.

WRAP (2003) (The Waste and Resources Action Programme), Review of Market Price Information in the UK Recycling Sector, Research Report..

CHAPTER 4:

IMPROVING MARKETS FOR USED RUBBER TYRES

*b*y

Pieter van Beukering
Vrije Universiteit Institute for Environmental Studies, Amsterdam.

1. Introduction

This chapter provides a review and analysis of the markets and policies for recyclable rubber. Particular attention is paid to used tyres because tyres account for approximately 50% of the total rubber market by weight and are responsible for a much larger share of the environmental problems caused by rubber materials.[57] In addition to littering landscapes, the improper disposal of used tyres often results in health and environmental hazards, though used tyres are not considered as a hazardous waste:

- they provide convenient habitats for rodents;
- they hold water and become excellent breeding grounds for mosquitoes that carry diseases, mainly in warm-climate countries;
- they can contaminate soils, surface and ground water when illegally burnt through run-offs of oils and soot; and
- they present a fire hazard when improperly stored. Such fires emit noxious, air polluting smoke.

Because of all these potential environmental hazards, most OECD countries now regulate landfilling of used tyres, often banning them altogether, and strongly encourage their reuse, recycling and recovery. The objective of the case study is to reassess recycling policy in light of an improved understanding of the functioning of the secondary rubber markets. These issues will be discussed in light of experience in three OECD member countries: the Netherlands, the United Kingdom and the United States.

The report is structured as follows. Section 2 describes the lifecycle of tyres. Section 3 focuses particularly on the economic aspects of the tyre lifecycle and its market barriers and failures. Section 4

57. In 1997, the share of tyres in the overall consumption by weight was as follows: USA 64%; Germany 47%; Japan 59%; UK 47%; France 55% (Derived from IRSG 1998, p.39).

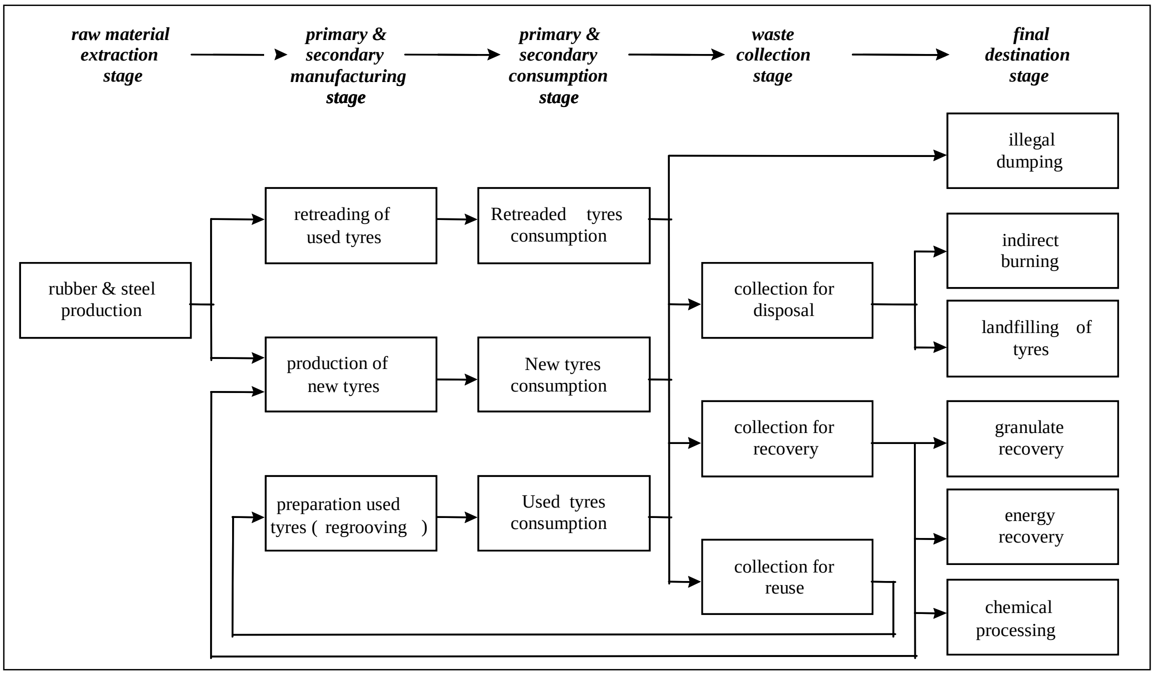
discusses the influence of policies on secondary rubber markets and provides policy recommendations concerning the most effective ways to solve market failures or redefine recycling policies. Throughout the paper specific examples of the secondary rubber markets in the Netherlands, the United Kingdom and the United States are provided in the boxes. The appendix provides several elaborations on specific aspects of the market for secondary rubber and its failures.

2. The lifecycle of tyres

The increased number of vehicles has led to a tremendous growth in the volume of used tyres. Over a billion tyres reach their end of life in the world each year (Brown and Watson, 2002) of which about 200 000 000 arise in Europe and 290.000.000 in the United States (RMA 2003). From 1998 to 2008 this is expected to change by ± 2 % every year. Vast quantities of tyres are stocked piled in designated landfills or illegally dumped, since a small quantity is presently reused and recycled (see www.rma.org/scrap_tires/). Besides posing a pressure on the environment and the existing waste management sector, this enormous waste flow also creates opportunities for new recycling markets to evolve.

The tyre lifecycle traditionally comprises four main stages. These include production, consumption, collection of used tyres and waste management. A simplified version of the tyre cycle for one region is depicted in Figure 4.1.

Figure 4.1. The lifecycle of a tyre



2.1. Production

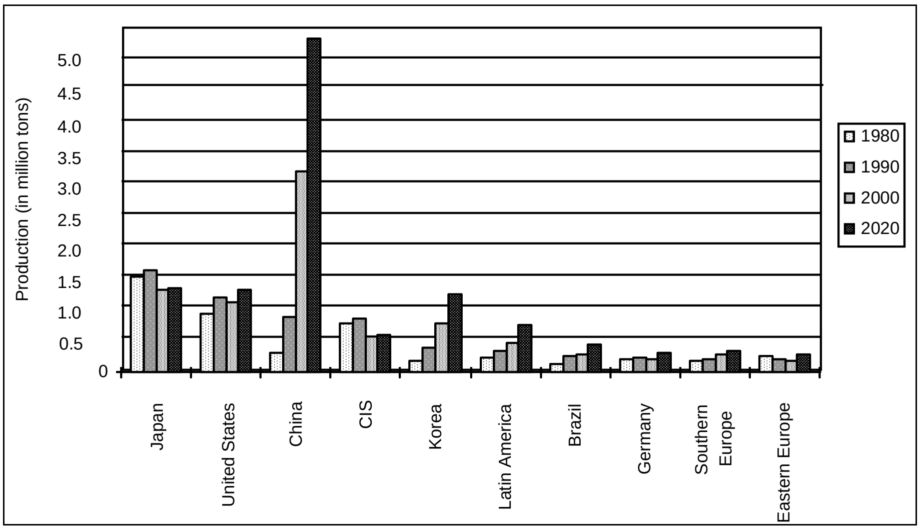
The basic components used to produce a tyre are synthetic and natural rubber, textile, steel and chemical additives. The proportions in which these components are used depend heavily on the specific characteristics of the tyre. This is clearly demonstrated if we look at the ratio between natural and synthetic rubber in different tyres: generally, truck tyres have a larger natural rubber content than passenger car tyres.

Natural rubber comes from the sap collected from *Hevea brasiliansis* (rubber tree) plantations. Synthetic elastomers are obtained from petroleum and coal requiring several stages to produce. The most important chemical additive is zinc oxide, which is used as an activator. Carbon black is added to further improve the rubber properties, prevent oxidation and provide greater abrasion resistance. All the input materials are finally mixed and vulcanised using sulphur. Vulcanisation is a thermochemical process that gives the tyres their performance characteristics in the *consumption* phase of the product lifecycle, but which can make further processing in post consumption stages more difficult.

An alternative component for rubber is "reclaim" which is the material recovered from used rubber products. Because its physical properties (i.e. elasticity, flexion, and chemical resistance) are not as good as 'new' rubber, the proportion of reclaim in tyre applications is limited to 10% (Guelorget et al. 1993).

It is expected that the greatest increase in tyre production by year 2020 will occur in the Asian newly industrialised economies. Already today, China has the largest global market share. Supported by its close proximity to the major natural rubber suppliers and cheaper labour supply, it is expected that China's market share in the world market will increase even further (see Figure 4.2).

Figure 4.2. World Tyre Production in 1980-2020;



Source: Smith, 1995

2.2. Consumption

An important product characteristic of the demand for tyres is the type of application: i.e. passenger car and truck tyres. This distinction is important as the two types of tyres have rather different material properties, thereby influencing their retreadability and recyclability. Approximately 70% of total weight of car tyres and 65% of truck tyres is rubber compound that is a combination of natural and synthetic rubber hydrocarbons. In general, truck tyres have larger natural rubber content compared to passenger car tyres. Steel, textile and carbon black are the main reinforcement materials of tyres. The amount of steel wire used in tyres is larger for truck tyres compared to passenger car tyres. The tyre weight varies from 7 kg for a passenger car tyre to about 55 kg in average for a heavy truck tyre. In terms of numbers of tyres, passenger car uses by far outweigh the commercial applications. Approximately 85% of the used tyres in the US and the UK are passenger car tyres. In weight, however, truck tyres (i.e. 60%) outweigh passenger car tyres (i.e. 40%) in both the US and in the UK (Rosendorfová et al. 1998).

Driving behaviour and tyre maintenance are the main factors in the consumption stage. Improvements in tyre manufacturing over the past 40 years have more than doubled the mileage of tyres, yet this technical limit is rarely met. Quick acceleration, excessive speed, abrupt braking, improper inflation of tyres and not taking into account the state of road surface are all forms of driver behaviour that cause the original tread to dwindle at a high rate. Currently, steel-belted radial passenger tyres last about 65,000 kilometres. If these tyres are properly inflated, rotated, and otherwise cared for, a lifetime of 95,000 to 128,000 kilometres lifetime may be achieved.

A tyre loses up to 10% of its weight until it is disposed. Most of the dissipated material comes from the tread, which is made of rubber only. If the casing is in a good state once the tread is finished, tyres can generally be retreaded (U.K. Environment Agency 1998).

2.3. Collection and Waste Management

The final stage in the lifecycle describes the ultimate destination of “used” tyres. Used tyres are generated after replacement or when shredding a vehicle. Various actors are involved. Generally, tyres are collected in tyre service centres. In some countries, consumers pay a limited fee to the service centre for proper disposal of the used tyre. For example, in Canada, consumers pay approximately € 2 per tyre for disposal. In turn, the service centre passes part of that fee (roughly 50%) on to the broker, who separates out the reusable and retreadable tyres. However, in most countries illegal dumping is a common way to get rid of a cumbersome waste at no financial cost.

There are four different options to put used tyres back into the economic circuit, which governments try to promote:

- reuse;

- retreading and regrooving;

- recycling through material recycling; and

- recovery through incineration with energy recovery.

In Europe in 1999 these options concerned on average nearly 40% of total discarded tyres, compared to 15% in 1994; landfilling is still the preferred route (52%), with the remainder being exported.

The various recovery routes for used tyres are evolving differently: for example, the market for tyre-derived fuel in the United States was 24 million used tyres in 1990 and grew significantly until 1996 to reach 152 millions. Since then, the trend is decreasing: 125 millions in 2000. In the same time period, tyre recycling for civil engineering applications has grown from 0.5 millions tyres in 1990 to 30 millions in 2000. This can be explained by the development of applications for recycled tyres and the publication of better information on environmental impacts related to respective uses of used tyres.

In Europe, the same trend is observed: from 1994 to 1999, post-consumer tyres consigned to landfills decreased significantly while other outlets, such as energy recovery and material recycling, increased proportionately (see Figure 4.3 and 4.4):

Figure 4.3: Fate of Used Tyres in Europe (1994)

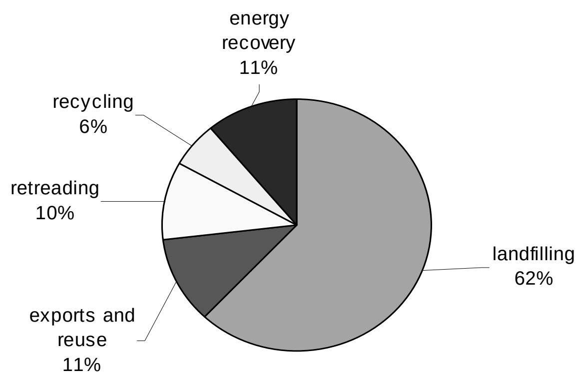
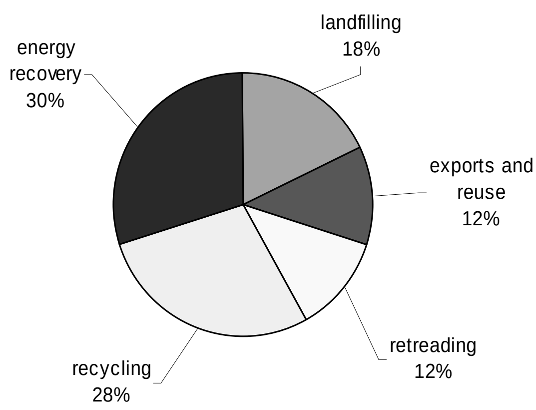


Figure 4.4: Fate of Used Tyres in Europe (2003)



Source: BLIC 2004 (European Association of the Rubber Industry)

2.3.1. Material reuse

Many civil engineering projects reuse discarded tyres to create artificial reefs, boat bumpers, crash barriers for racetracks, slope stabilisers, irrigation channels and erosion layers on dams. This is not "product reuse" since the used tyre is not used for its original purpose, but it is still material "reuse" because the tyre has not been subject to any reprocessing.

"Product reuse" mainly concerns part-worn tyres, which are discarded when a minimum tread depth is reached, and are then exported to be directly reused in developing countries or countries having less stringent safety standards. From an environmental perspective, direct reuse prolongs the life span of a tyre. However, it transfers the "waste burden" to countries that may not have the adequate infrastructure to manage such wastes.

2.3.2. Retreading and Regrooving

The tyre roughly consists of the casing or carcass that forms the skeleton of the tyre and the tread that is mainly made of rubber and therefore in most cases can be renewed. Retreading consists of stripping the old tread from a worn tyre and reclothing the old casing with a tread made from new materials.[58] Retreading brings environmental benefits, as it extends the tyre life span. It saves 80 % of raw material and energy necessary for production of a new tyre and reduces the quantity of waste to be discarded. Although the price of retreaded tyres is between 30-50 % lower than the price of a new tyre, they deliver the same mileage as new tyres (Ferrer 1997).

The potential market for reuse and retreading is quite important since 40-50% of discarded car tyres and 60-80% of truck tyres are suitable for reuse and retreading. However, though standards and regulations tend to ensure the security of retreaded tyres and most retreaders claim that there is no quality difference between new tyres and retreaded tyres (U.K. Environment Agency, 1998), there are still unjustified and mistaken suspicions on the part of consumers associated with retreaded tyres.

Partly worn tyres may also be regrooved: a new pattern is grooved into the tread base that remains after the pattern has been worn away by use. This technique is carried out primarily on truck tyres because they are designed with significant tread thickness. If the process is carried out correctly, about 30% extra mileage will be obtained for only 2.5% of the cost of a new tyre (World Tire Industry 1997). Retreaders oppose direct reuse and regrooving because it makes further retreading more difficult, more expensive and in most cases even infeasible.

2.3.3. Recycling (mechanical and chemical processing)

a) Mechanical processing:

Used tyres are shredded and ground to be used as a base material in asphalt paving and road construction instead of sand, gravel and stone. It increases the pavement life by four to five times, thus reducing the amount of resurfacing materials required and also reducing noise. Rubber granulates can also be used to produce new tyres, as well as many different products such as adhesives, wire and pipe insulation, brake linings, conveyor belts, carpet padding, hose pipes, sporting goods wheels of roller blades and suitcases, etc. They are also commonly used in the building of tennis courts, sport fields and children’s play areas.

In view of its environmental performance, grinding is an energy-intensive process and has relatively high dust emissions. The economic and environmental advantage of grinding is that it generates recyclable rubber and useful by-products such as steel and textile, which can also be recycled. The most common application of granulate is in rubberised asphalt. Although this seems to be a promising outlet for recycled rubber because rubberised asphalt lasts longer than conventional asphalt, its relatively high cost prevents its widespread application.

b) Chemical processing

Chemical processing of size-reduced tyres, such as pyrolysis, produces monomers. Although the end product is inferior to virgin rubber, it can still be used as a component in high value commercial

58. Today retreading is more common for truck than for passenger car tyres. The retread rate of truck and passenger tyres is approximately 80 and 20 percent, respectively (U.K. Environment Agency, 1998). Car tyres can be retreaded only once, truck and bus tyres 3 to 4 times and an aircraft tyre 8 times. In practise, truck tyres are retreaded fewer times (average 1.5 times in the Netherlands, 2.5 times in Eastern Europe).

applications requiring high performing rubber such as tyres, bicycle tyres, automotive moulded parts, soles and heels. Chemically recovered rubber is approximately half the price of virgin rubber. Using recovered rubber can be economically feasible for the tyre industry, especially when production waste is recycled and reused within the factory where it is generated. This might result in additional savings from the elimination of disposal fees and transportation costs.

2.3.4. Use as a fuel

Because of their high calorific value, used tyres are used as a supplement fuel in pulp and paper mills, industrial boilers, cement kilns and power plants. A tonne of tyres is equivalent to a tonne of good quality coal or to 0.7 tonne of fuel oil.

Energy recovery through incineration is presently the leading recovery for scrap tyres in most OECD countries. Depending on the technology used, tyres can represent up to 25% of the total fuel of cement kilns. A major advantage of using worn out tyres in cement kilns is that it does not generate solid waste because the ash residues from the tyre combustion are bound to the final product. Furthermore, sulphur emissions are not of a major concern as the sulphur is transformed and bound into gypsum, which is added to the final product (Jones 1997). In Europe, the USA, Japan and Korea, cement kilns are among the most common end users of tyres for their energy content. In some countries, such as Austria, France, Germany and Sweden, up to 65% of the total quantity of used tyres is incinerated in cement kilns. In an alternative to cement kilns, totally dedicated tyres-to-energy power plants have been built in Europe and the US.

2.3.5. Landfilling

Traditionally, landfilling has been the most common method for disposal, mainly due to the low price of landfilling. Two forms of disposal can be identified: disposal in a landfill and disposal in a monofill. A scrap tyre monofill is a landfill that only stores tyres. When disposed of in landfills, scrap tyres occupy a large space and remain intact for decades posing increased environmental and public health risks related to possible leakage and the danger of uncontrolled burning. Furthermore, when whole tyres are buried in a landfill they trap air and have tendency to migrate to the top of a closed landfill breaking the landfill cap and causing costly damages to the landfill cover that increases the instability of sites. Also, used tyres easily trap rainwater and therefore create a favourable environment for insects, which increases the risk for mosquito-borne diseases (URI, 2001). In 1998, the European Commission accepted a directive that bans the disposal in landfills of whole tyres by 2003 and shredded tyres by 2006 (EC, 1997).

Monofills are more desirable than landfills as they facilitate material and energy recovery in the future. After the European ban on landfills is operational, monofills form a temporary solution in those European countries where capacities for processing used tyres are limited. The potential advantage of such monofills is that they can be reconsidered as future used tyre collection sites and distribution centres. However, examples around the world show that monofill sites frequently become abandoned without processing the stored tyres. Monofills also are a serious source of fire outbreaks, especially when they are unsecured. Yet, in assessing tyre-related risks on the U.S.-Mexico border, Blackman and Palma (2002) show that despite the large numbers of disposed tyres, fire outbreaks and propagation of mosquito-borne diseases are minimal in this region. Therefore, they warn that generalisation of tyre-related risks is misleading, and that instead case-by-case analysis is needed to determine which risks are important.

2.4. International trade

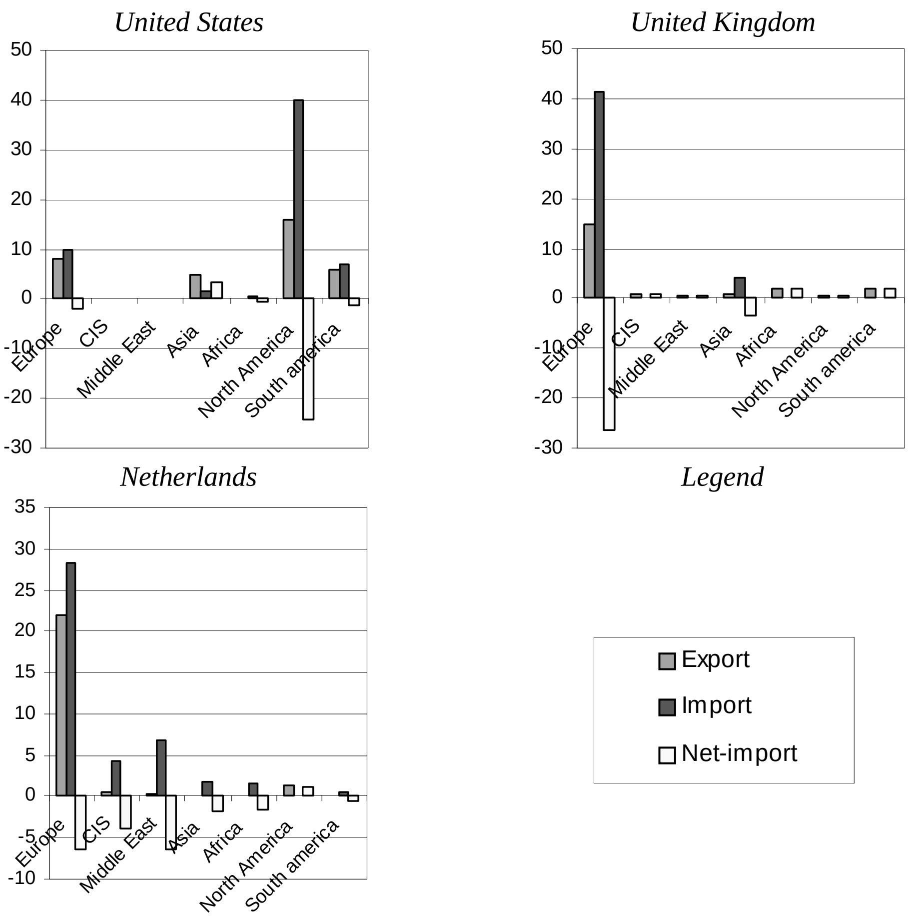
International markets are important throughout the lifecycle of tyres. Natural rubber can be produced only in tropical areas and high-quality tyres are manufactured in just a limited number of industrialised countries. The waste management stage is also increasingly subject to international trade. Yet, knowledge on the commodity and regional patterns of international trade in used tyres and tyre-derived rubber waste is scant (UNCTAD, 1996). However, based on the limited data available, it can be concluded that trade in used tyres has become more important in the 1990s. Moreover, typical patterns in the trade of used tyres can be recognised. For example, Rosendorfová at al. (1998) show that the share of imports of used tyres by Eastern European countries in Europe expanded from 19% in 1992 to 47% in 1997. They conclude that trade of used tyres with neighbouring countries still dominates. Transport costs of the bulky tyres still seem to be a crucial factor vis-à-vis trading distance. This is confirmed by the trade situation for the UK, the US and the Netherlands, presented in Box 4.1.

Several arguments are put forward to explain the increasing flow of tyres from high- to low-income countries (Van Beukering and Janssen, 2001). First, due to the low wage level and the relatively simple process of retreading and recycling, low-income countries may have a comparative advantage in the labour-intensive retreading and recycling of tyres. Second, many tyres are imported for reuse purposes. Safety standards regarding minimum tread depth and the enforcement of these standards are less strict in lower income countries. Third, international differences in disposal fees promote the trade of used tyres that are not recyclable. Because disposal fees are much lower in low-income countries, it is a lucrative business to collect tyres in the North, collect the disposal fee paid for by consumers in the North, and export these tyres to low-income countries. For example, the disposal fee in the Netherlands is twice the fee in the Czech Republic (Rosendorfová et al., 1998).

Box 4.1. Trade patterns for used tyres

The majority of the traded used tyres moves between neighbouring countries. The bulkiness of tyres make it an expensive commodity to transport. This is confirmed if we consider the trade flows in 2001 for the US, the UK and the Netherlands (see Figure below). The Figure also shows that all three countries are net-importers of tyres.

Trade volume of used tyres in 2001 in terms of sources and destinations for the United States, United Kingdom and the Netherlands (in 1000 tons)



Source: UNSD Comtrade Database 2002

3. The market for secondary rubber

The configuration of the market for secondary rubber in each country depends on local economic and institutional conditions. Therefore, differences in recycling activities and waste management of used tyres occur naturally between countries. Moreover, market failures and distortions may be present in certain countries that reduce the efficiency with which markets for recyclable material operate, possibly leading to sub-optimal levels of recovery. This section describes the current configuration of the market for used tyres in various countries and presents evidence for the proposition that this configuration is not optimal from an economic point of view.

3.1. Current configuration of the used tyre market

The international comparison depicted in Table 4.1 shows how the configuration of options to process used tyres varies widely between countries. On the one hand, Northern European countries have all achieved their objective to ban or significantly reduce tyres from the landfill. The Eastern European countries and some of the Southern European countries, on the other hand, still landfill the majority of their used tyres. Another typical feature in Table 4.1 is that countries such as Denmark and Finland are fully dedicated to material recycling while Austria, Germany and Sweden favour energy recovery of tyres. Also note the situation of Portugal which recovers 100% of its used tyres and has the highest rate of retreading (33%).

In general, the options that have become relatively less important include reuse and landfilling. This can be explained by a number of factors. Given the announced landfill ban in Europe for 2003 it is not surprising that landfilling has declined from 35% to 32% between 2002 and 2003. In addition, the retreading industry was significantly affected by imports of less expensive tyres from Russia and Asia, with a loss in market share from 23% in 1996 to 9% in 2003. Recycling has grown from 11% in 1996 to 27% in 2003. This is due in part to improvements in technologies. However, the aforementioned implementation of a European landfill ban also provided a strong incentive for the material recycling industry. In addition, the net-export of used tyres increased during this period, reaching 8% in 2003, from a base of less than 2% seven years earlier.

Table 4.*1*. Processing options of used tyres in Europe for 2003 or the most recent available estimate (in %)

	Tyre Arisings (000 tonnes)	*Reuse %*	*Export %*	*Retreading %*	*Material recycling %*	*Energy recovery %*	*Landfill & other disposal %*	*Used Tyre recovery rate %*
Austria	55	0	0	0	45	55	0	**100**
Belgium	75	0	0	4	34	45	17	**83**
Czech	57	0	0	0	0	0	100	**0**
Denmark	44	0	0	9	75	16	0	**100**
Finland	36	0	0	4	80	16	0	**100**
France	390	12	0	14	33	19	22	**78**
Germany	582	4	6	11	19	53	7	**93**
Greece	50	4	2	4	14	4	72	**28**
Hungary	45	0	10	0	4	23	63	**37**
Ireland	16	0	0	25	0	0	75	**25**
Italy	355	8	13	14	20	37	8	**92**
NL (pc tyres only)	35	0	85	0	15	0	0	**100**
Poland	124	0	0	13	7	37	43	**57**
Portugal	61	0	0	33	50	17	0	**100**
Slovak Rep.	18	0	0	0	0	0	100	**0**
Spain	275	1	4	14	9	12	60	**40**
Sweden	77	1	15	4	35	45	0	**100**
UK	444	11	3	12	39	17	18	**82**
EU average 2003	**2739**	**2.3**	**7.7**	**8.9**	**26.6**	**22.0**	**32.5**	**67.5**
EU average in 2002	***2617***	***6***	***6***	***13***	***19***	***21***	***35***	***65***
US (estimates)	3,600	11	5	9	29	34	12	

Source: European estimates from BLIC 2004;

US estimates for the year 2000 from Sunthonpagasit and Duffey (2003b), U.S. EPA (2003); and RMA (2002).

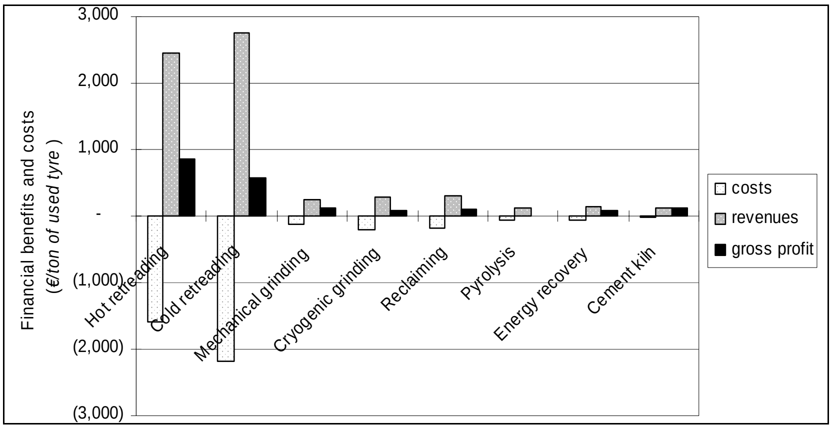
3.2. Optimal configuration of the used tyre market

In order to assess the efficiency of the market, it is first important to assess whether the current configuration of processing options for used tyres is optimal from an economic point of view. Deviations from the optimal configuration can indicate the presence of market failures and distortions. Thus far, no studies have sought to identify the optimal configuration of the complete range of waste management options for used tyres and only a limited number have made a comparison of the financial viability of specific processing options. These studies can provide some insight in the desired ranking of options.

Rosendorfová et al. (1998) study the options for post-consumed tyres and tyre related rubber waste in Europe in order to create a ranking on the basis of financial feasibility. Considering the average performance of the industry in Western Europe, they compared the financial viability of retreading, grinding, reclaiming, pyrolysis and incineration in cement kilns. The net costs have been calculated by subtracting the total financial costs from revenues. No account is taken of environmental externalities, except those that are presently incorporated in the costs of the alternative waste management options. The results of this analysis are presented in Figure 4 5.

Retreading is clearly the most desirable option from a financial point of view. A significant gap exists between the benefits of retreading and the other options. The financial performance of mechanical grinding, incineration in cement kilns and materials recycling do not vary widely. Although the revenues and costs differ among the processing options, their total gross profit varies only between €115-118 per tonne of used tyres. Assuming the end products of pyrolysis are to be utilised as fuels, leads to the conclusion that pyrolysis is not a financially viable process, unless the production costs decline substantially over the coming years. Given the modest progress of pyrolysis applications for comparable waste materials such as plastics, this is unlikely to happen (US EPA, 1991).

Figure 4 5. Financial costs of various recycling and waste management options in Western Europe



Other works have been conducted that provide a better understanding of the optimal configuration of the used tyres market. Ferrer (1997) demonstrates the economic rationale of maximisation of retreading before alternative processing options are considered. Ferrer shows that, when retreading is no longer technically feasible, heat generation is the only true recovery alternative because it is financially viable, the technology is readily available, and the demand is stable. The other options have technological, economical or demand limitations that make them less attractive. It should be mentioned that techniques have progressed since Ferrer's study and that his conclusion are unlikely to be valid any longer for a number of alternatives, such as asphalt application.

Sunthonpagasit and Duffey (2003b) examine the engineering economics of crumb rubber facilities in the US. Following a literature review and interviews with producers, a financial model of a nominal processing operation was created to aid the analysis of different market, crumb size, and production scenarios. The profitability of a crumb facility appears to be particularly sensitive to crumb rubber prices, operating costs, and raw material availability. In particular, the lack of standards leading to large variations in quality of rubber crumb is a handicap for the market of material recycling.

Using information on scrap tyre composition and the market conditions in which they are used, Amari et al. (1999) examine the technologies used in the US to recover their value either for energy or as rubber. They conclude that retreading offers the best strategy for value recovery, requiring least new material and energy to achieve the highest value-added use in the economy. Energy recovery is considered as the second best option after retreading but further expansion of TDF is hampered by local regulations and lack of adequate infrastructure for collection and transport.

The conclusion from this literature review is that the current configuration of processing options of used tyres is far from optimal. Generalising across countries, retreading seems to be practised below the level which is economically optimal. The findings for TDF are more diverse, but predominantly indicate the need for expansion of energy recovery. The economic prospects for material recycling are the most uncertain, which leads to the conclusion that no immediate economic justification exists for its expansion. Without exception, landfilling and stocking is considered to be both environmentally as well as economically unjustifiable. However, the actual 'market shares' for these different options are very different than this ranking would imply. A review of potential market failures and barriers may shed light on why this is the case.

4. Market failures and barriers

The conclusion concerning the sub-optimality of the relative shares of different processing alternatives is an indication of the possible presence of market failures and barriers. This section provides an overview of the potential market failures in the used tyre sector. These market failures will be described subsequently for the main categories of recycling and waste management options: retreading, material recycling, energy recovery and landfilling/illegal dumping.

4.1. Retreading

When examining market failures for retreads, it is important to make a distinction between truck tyres and passenger car tyres. Although the technique of retreading tyres has been established for decades it has never broken through in the passenger car tyre market as a serious substitute for new tyres. Conversely, retreads are fully accepted for truck tyres, where about 50% of used tyres are retreaded. The limited share of retreads in the passenger car market is somewhat surprising but can be partly explained by the lack of supports in place from which trucks benefit. Truck tyres are generally inventoried, tracked when used commercially, properly inflated, rotated and checked on a schedule. According to the above mentioned engineering studies, retreading is clearly indicated as the most efficient method of recovering values from used tyres. Retreads are 30% to 50% cheaper than new tyres and appear to have functional attributes which are not dissimilar from “new” tyres. Therefore, the main question to be addressed in this section is why the market for retreaded tyres is not more important than it is. Several possible explanations are put forward.

4.1.1. Technological Externalities

Technological externalities arise when goods are designed in an economically sub-optimal manner. In the case of tyres this would arise if incentives are not in place to ensure that the optimal degree of recyclability is present. Thus, if costs associated with the landfilling (or illegal disposal) of tyre waste are not transmitted up the product lifecycle to the point of purchase, consumers will have no incentive to demand (and manufacturers to supply) tyres with the socially optimal design.

An issue affecting the potential for retreading is the introduction of energy-efficient tyres (often labelled as eco-tyres, or smart tyres). These can save up to 6% of a vehicle’s fuel. The retreadability of these tyres, however, is said to be limited. It is possible that the reduction in externalities associated with increased fuel efficiency is not as great as the increased externalities associated with reduced recoverability. However, there is a ‘market’ for fuel efficiency. Conversely, if the costs of disposal are not transmitted back to the consumer there is not a market for recoverability. As such, design will not be socially optimal.

Recent changes in trade patterns have exacerbated such problems. In the last decade, the import of new tyres from China and other Asian newly industrialised countries to Europe has increased significantly. These tyres are much cheaper than the high-quality tyre produced in Europe and the US. Due to the lower quality of the casing, these cheaper tyres are less suitable for retreading and have a shorter life span. The concern expressed by Rosendorfova et al. (1998) regarding the negative impact of this trend on the overall performance of the retreading sector has proved to be valid. The continuing decline in the passenger car retread market has forced the closure of a number of retread companies in the UK. Despite the improved quality of retreads, sales have dropped by over 50% in the past five years. Traditional key selling points for retreads were that they were significantly cheaper. However, the growth in the market for budget tyres, where prices may be marginally higher than retreads, has significantly affected retread business (DTI, 2001).

4.1.2. Information Failures and Risk Aversion

It is hardly surprising that retreaders claim that there is no quality difference between new tyres and retreads (U.K. Environment Agency, 1998). They have a vested interest in encouraging their use and as such their claims must be considered in that light. However, a number of reputed government agencies confirm the fact that retreaded tyres are perfect substitutes, at least in terms of functional attributes, for new tyres. US EPA (2000) dispels the myth of retreads being less safe than new tyres. “Statistics compiled by the U.S. Department of Transportation show that nearly all tyres involved in any tyre-related accidents were under-inflated or bald. Properly maintained tyres, both new and retreaded, do not cause accidents.” US EPA (2000) also repudiates the notion that retreads have a higher failure rate than new tyres. “Rubber on the road comes from both new tyres and retreaded tyres, primarily from truck tyres that are overloaded, under-inflated, or otherwise abused.” Studies of tyre debris in the UK have shown that it arises from a mixture of around 46% new tyres and 54% retreads (TRL, 1992). This is representative of the use pattern of truck tyres. (These findings are further discussed in Box 4.2.)

No evidence has been found in the literature or elsewhere of reports that claim a structural lower performance of retreads compared to new tyres. This leads us to reject the hypothesis that the engineering studies are wrong about their conclusion of complete substitutability of retreaded and new tyres. However, despite the lack of scientific evidence for the existence of quality differences between new and retreads, the public image of retreaded tyres remains poor among passenger car owners. Thus, there may be an information failure within the market. This failure may, however, be less prevalent amongst road freight operators due to the strong competitive pressures to operate efficiently, and thus have full information about relative tire quality.

There may also be other factors at work, undermining the market even if there is full information concerning their relative quality. This can partly be explained by the fact that consumers are generally risk averse. The ‘risk aversion’ literature points out that even a slight inferiority in quality (or perception of slight inferiority in quality) can have significant repercussions on demand.

Indeed, many studies in other areas have shown that people are disproportionately averse to low-probability high-impact risks that may lead to morbidity and mortality. This is likely to be particularly true for car tyres that are crucial for the safety of a car. The incident of a blow-out at full speed on the highway is every car driver’s nightmare. Moreover, although retreads are up to 50% cheaper than new tyres, the price difference is still insignificant compared to the value of the car. Therefore, it seems crucial for the retread market to eliminate the perception of car users that retreads are inferior in quality to new tyres.

In addition, there is also evidence that consumers are often reluctant to change consumption patterns in the face of perceived uncertainty concerning quality. Encouraging consumers to ‘switch’ from a product (new tyres) whose quality attributes are thought to be known through previous experience to a substitute (retreaded tyres) which is perceived to be of potentially lower quality may require relatively greater price incentives than is implied by the standard economic assumptions.

Indeed, both of these factors can undermine the market for retreaded tyres to a greater extent than might be imagined. In effect, the standard assumptions of certainty equivalence (i.e. in which the value of the good is discounted by the probable degree of inferiority relative to a substitute) may not hold. Consumers need even stronger incentives than is usually the case.

Box 4.2. The truth behind the gators.

In the United States several organisations, such as the Maintenance Council of the American Trucking Association and the American Retreader's Association and government agencies from Virginia and Arizona have researched the performance of retreaded or recapped tyres compared to new ones. This research was initiated because of public concerns about the quality of retreaded tyres. In the eye of the public many of the failed tyres routinely found lying on and alongside roadways – the so-called road gators – are believed to originate from retreaded tyres often associated with heavy-duty trucks and other large road equipment. Unlike for passenger car and light truck retreaded tyres there are no federal standards for retreaded heavy-duty truck tyres, nor are there any plans to establish them in the near future (Federal Motor Vehicle Standard Number 117 sets norms for retreaded passenger car tyres comparable to those for new tyres).

From this research it was established that poor maintenance of tyres was the foremost cause of failure (especially under-inflation). Even though retreaded tyres were over-represented in the recovered tyre parts, this was mainly due to their use on failure-sensitive vehicles like tractor pulled trailers (this was the conclusion in the Virginia study; the Arizona study did find other reasons for the over-representation of retreaded tyres).

Sources: Virginia Department of State Police, 2000; Carey, 1999; AAMVA, 2000.

4.1.3. Consumption Externalities and Signalling

Closely related to the issue of information failures, are the issues of 'consumption externalities' and 'signalling' effects, both of which may contribute to the likelihood of passenger car owners favouring new tyres over re-treaded tyres. 'Consumption externalities' refer to the effect of consumers take their 'cue' from other consumers (i.e. consumption patterns from one consumer provides information to other consumers). This is often the motivation behind demonstration projects and public procurement policies. Indeed, public procurement favouring the use of retreaded tyres could have the effect of reducing other consumers' degree of risk aversion.

However, quite frequently policies have the opposite effect, favouring new tyres over retreads. For instance, although the US EPA (2000) strongly promotes the market for retreads by encouraging government agencies to purchase retreads for their fleet, legislation has also been implemented in the US that is likely to strengthen the public in its mistrust towards retreads. This legislation prohibits the use of retreaded (and regrooved) tyres on the front wheels of buses. Although this legislation is relatively unknown to the general public, and therefore have only a small influence, it does show the government has mixed feelings about retreads that are not justified by its own research or that of others.

In addition, the consumer's perception of the value of their car (and not just the tyres) can be affected by the choice of retreaded over new tyres. This is known as 'signalling,' and occurs in situations where the perceived value of a car is negatively affected by the use of retreads. Because the use of retreads sends a negative signal to the potential buyer of a car, the cost savings of using retreaded tyres are more than nullified by the reduction in the value of the car itself. This may partially explain why car manufacturers (and even second-hand car dealers) sell cars with tyres manufactured from 'new rubber'.

4.1.4. Market Power and Market Segmentation

There may also be factors on the supply side which are restricting the use of retreaded tyres. Tyre manufacturing industry can be characterised as an oligopolistic market with a limited number of large brands. In 1999, the major production of tyres in the global market was held by six companies: Michelin (France), Bridgestone (Japan), Goodyear (USA), Continental (Germany), Sumitomo (Japan) and Pirelli (Italy). Based on the global market sales, the top three controlled 57% of the world market (See Table 4.2).

Table 4.2. Production data of major tyre producers in 1999

Rank	Company	Output/tonnes (in US$ million)	Market share (%)
1	Bridgestone	13,750	20
2	Michelin	13,200	19
3	Goodyear-Dunlop	12,725	18
4	Continental	4,955	7
5	Sumitomo	2,783	4
6	Pirelli	2,600	4
7	Yokohama	2,514	3
8	Cooper	1,803	2
9	Toyo	1,323	2
10	Kumho	1,235	2

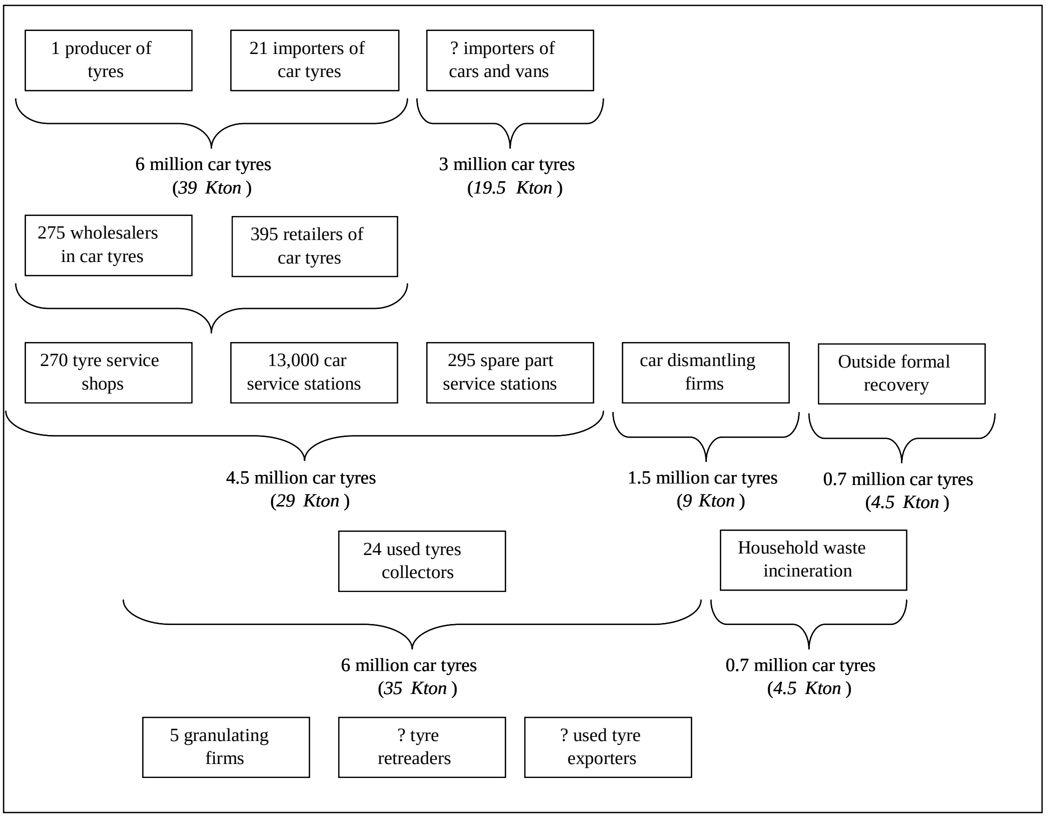
Source: BLIC, 2002.

As opposed to the tyre manufacturing industry, the majority of tyre retreading and recycling companies in Europe are small and medium enterprises. Recycling operators at the collection and sorting level are commonly thought to be price takers in their transactions with buying traders. Moreover, they have little influence over their costs in terms of the price they pay for waste. Their low market power is further influenced by competition with other ways of waste elimination as well as the fragmented characteristics of collection and sorting stages.

The literature provides little evidence of primary tyre manufacturers exercising market power (i.e. through dumping and other means) to constrain the share of retreaded tyres. However, retreaders in the UK were said to face difficulties in reaching their customers due to the fact that the main retailers have strong ties with the primary manufacturers of tyres, and therefore only promote new tyres (Aardvark Associates, 2001). Random checks at tyre retailers in the Netherlands confirm this experience but this test cannot be considered representative for Europe as such.

Another important segment in the market structure of the tyre supply chain is the collection and processing network of used tyres. Figure 4.6 shows the actors and the level of concentration in the Dutch tyre supply chain. As mention earlier, the level of concentration at the stage of the producers and importers is particularly high. At the retail and car service centres, the dispersion of the market is large. Approximately 13,000 car service stations, 279 tyre replacement shops and 300 spare part centres are involved in the replacement of old for new tyres. The large number of entrepreneurs makes it necessary to implement collective initiatives to improve efficiency in the post-consumption stages rather than pushing for individual initiatives.

Figure 4.6. Supply chain of passenger car tyres in the Netherlands in 2002



Source: VROM, 2002.

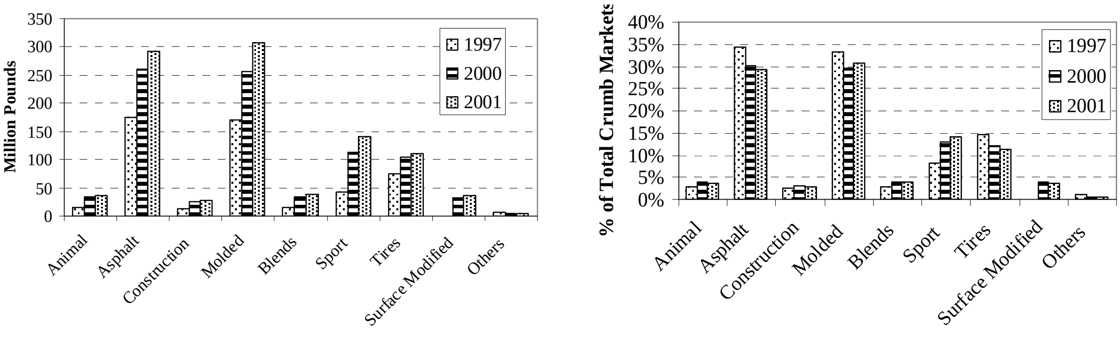
In addition, however, it is possible that tyre manufacturers are able to exercise discrimination through their links with wholesalers and retailers. A number of manufacturers of tyres made from primary rubber are themselves involved in the manufacturing of retreaded tyres. Therefore, they have incentives to optimise the share of each. In the presence of market power, they may be able to maximise their profits by segmenting the market between new and retreaded tyres, and exercise price discrimination in particular markets.

4.2. Rubber recycling

4.2.1. Characteristics of the rubber recycling market

Shredded tyres, also called crumb rubber, have various applications in the rubber recycling sector. Figure 4.7 shows that in the US in 2001, of the total 0.45 million ton (996 million pounds) of crumb rubber produced, asphalt and moulded products were almost equal in market share, and combined had approximately 60% of the total market. The remaining 40% was used for the manufacturing of new products. Figure 4.7 shows that although consumption by all crumb rubber markets increased from 1997 to 2001, the crumb rubber market share of animal bedding, construction, plastic blends, and sport surfacing increased while the other crumb rubber markets shares decreased.

Figure 4 7. Crumb Rubber Markets: North America



Source: Sunthonpagasit and Duffey, 2003a.

The aforementioned engineering studies provide limited support for further relative expansion of rubber recycling operations. Also, most governments are cautious in unambiguously promoting rubber recycling. For instance, the UK Government's Department of Trade and Industry reject the notion that recycling is underdeveloped due to the lack of capacity (DTI, 2003). DTI shows that there is capacity in place now capable of handling many more tyres than are being processed at present.

The main reason for the limited share of rubber recycling is the low profit margin in the crumb rubber sector. This is mainly caused by technical constraints and the low quality of the end product. Whether this situation will change depends on price developments of primary natural and synthetic rubber and on the options available to reduce the cost of recycling or expand the market for products made of secondary rubber further. In addition recycling costs are heavily penalised by high collection and transport costs which represent 40-60% of the total recycling costs, according to a French study on the costs of used tyres management (ADEME). Another reason for the limited market of rubber recycling is the still too restricted number of applications for secondary rubber.

World rubber prices are expected to fluctuate around the current level or drift lower in the short and mid-term. Low prices for primary rubber discourage the use of secondary rubber in various products. Further price improvements are unlikely because of the great potential for increased supply from more intensive tapping and from the increased production capacities in new producing countries (FAO, 2002). Given these exogenous developments in the primary market for rubber, the main issue with material recycling is whether or not the market for products of secondary rubber (i.e. rubber crumb) is 'immature' and thus whether or not it should be supported to get over learning curves, etc.

Whether strong market distortions are present remains unclear. High price volatility is often recognised as a symptom of the presence of market distortions. There are some claims in the United States of high month-to-month price instability and shifting demand, especially for lower quality crumb rubber, but this only seems to apply to a limited number of markets within the rubber-recycling sector (Sunthonpagasit and Duffey, 2003a). In the following paragraphs, several other potential market failures will be discussed.

4.2.2. Information failure

Because of the large variety in crumb rubber size and quality, every rubber recycler in effect is said to produces a different product (Owen, 1998). Lack of uniform specifications in the crumb rubber market implies that suppliers have to submit their products for approval and testing in each new locality where they are proposed. The resulting high marketing costs seriously hamper the maturation of the crumb rubber market. The development of crumb rubber specifications could help to improve this situation. The

American Society for Testing and Materials (ASTM) and the European Committee for Standardisation (CEN) have already developed standards for crumb rubber, so the problems relating to uncertainty about quality and other matters should be substantially reduced in the near future. Also for TDF, the ASTM has developed standard relating to the wire content, size etc..

Another information failure is not directly related to used tyres, but to granulated rubber processing equipment. In the US, the reported maintenance costs of this equipment are reportedly 200-300% higher than the costs claimed by equipment manufacturers. One reason for this is the shorter than projected service lives of perishable items, such as shredder knives. Maintenance costs and the timing of maintenance seem to be important, since one processor claimed his shredder machine worked twice as fast with new knifes. Such information failures make the planning of market entry and production volumes difficult. Sunthonpagasit and Duffey (2003b) consider this to be one of the reasons many companies fail within several years.

4.2.3. Technological Externalities

Tyres are designed – for safety and durability reasons – to make them withstand the wear and tear of their everyday use as well as possible. Tyre strength originates from the added steel and fibre but mostly from the vulcanisation of its rubber. The steel and fibre increase recycling costs because it blunts the knives or wears down the hammers that cut or ground the tyres into smaller pieces. After cutting and grounding the steel and fibre have to be separated from the rubber, which also adds to the costs. Of course, we – as drivers – want tyres to be strong, we want them to be safe and we want to get a good mileage. These are the basic requirements a tyre has to live up to. Still, as noted above with reference to retreading, one could ask if the current balance between strength and recoverability is the right one if the incentives for one product attribute (i.e. recoverability) are missing (Amari et al., 1999).

A possible way to increase the likelihood of a correct or optimal balance would be to make the producers of tyres responsible for the cost of disposing or recycling them when they are worn down. This could be achieved through an Extended Producer Responsibility (EPR) approach, which will be discussed in the next section. Appropriately designed, such measures can transmit incentives from the post-consumption stage back up to purchasing decisions and thus manufacturing and design decisions.

Recycled rubber is not a perfect substitute for the virgin material due to vulcanisation, which is part of the production process. In this process rubber, carbon and sulphur are mixed together using heat and pressure to form a strong and durable material. This chemical process has proved hard to reverse and has been likened to "unbaking a cake and then reusing the eggs". Recycling of useful, uncontaminated materials is even more complex for multi-material laminated products such as modern tyres (Brown and Watson, 2000). There are currently many recycling techniques but these either produce an end-product which is inferior to virgin rubber, such as the size reduction techniques mentioned above, and can only be used for its original purpose in small percentages, or are at present economically unfeasible due to high energy costs (e.g. pyrolysis and gasification). New techniques, such as chemical and biological devulcanisation, mechanochemical recycling, and de-wiring technologies need to be developed further to produce a satisfactory product (Brown and Watson, 2000).

4.2.4. Policy failure

In the US, each state has its own laws and regulations with regard to waste management and recycling of tyres. Regulatory practices to stimulate recycling include landfill bans and disposal (tipping) fees. Ironically, however, many in the industry have perceived government grants and subsidies to some rubber recyclers as counter-productive. By granting some facilities access to exemptions from tipping fees and subsidies for capital or operating costs, they have put downward pressure on secondary rubber prices, scrap tyre availability, and tipping fees. This affects the survival of existing un-subsidised plants. Numerous subsidised rubber recycling facilities in the US closed down because they could not sustain profitability or they failed to find markets for processed material. For example, tyre-recycling programs in Oregon and Wisconsin provided end-users a subsidy of US$20 per ton. The used tyres were collected and processed when the subsidy was in place, after which most of the new end users stopped processing.

Efforts to create a level playing field in the tyre-recycling sector have also been made at the Federal level in the US. Congress passed the Intermodal Surface Transportation Efficiency Act (ISTEA) in 1991, requiring the states to use scrap tyres for surfacing of federally funded highways. If this Act had been implemented, proponents claim it would have increased the use up to 70 million used tyres in 1997. However, in 1995 ISTEA was repealed, presumably as a consequence of lobbying efforts of the asphalt industry. As a result, the use of crumb-rubberised asphalt (CRA) now depends on the willingness of the state to initiate and sustain programmes. After the repeal of the ISTEA most processors who came into this business in anticipation of this potential market left due to downward pressure on secondary rubber prices (Sunthonpagasit and Duffey, 2003a).

4.3. Energy recovery

4.3.1. Characteristics of the energy recovery market

Financial-engineering studies are generally positive about the prospects of expanding the capacity of energy recovery. Still, as was shown in Figure 4 4 the relative importance of energy recovery has declined in the past five years. This trend can partly be explained by the increasing popularity and promotion of material recycling. Another important reason for the decline in energy recovery in a number of countries is the introduction of strict environmental laws. In Austria, for example, energy recovery declined from 70% in 1993 to 40% in 2000, partly due to special emission regulations for TDF in 1993 (MoA, 1993). The Austrian cement industry had been using the thermal input of tyres since 1979 successfully (Reiter and Stroh, 1995). According to the Regulation, the emission limits for particulates are 50 mg/m^3, for SO_2 200 mg/m^3 and for NO_2 500 mg/m^3. The incineration of other waste is not subject to the same regulation but individually by other competent authorities. As a result, it has become more attractive to use alternative waste materials (e.g. plastics or animal flours) as fuel input for the cement industry.

Similar to material recycling, there is sufficient capacity available in the incineration and cement industry to use a higher proportion of used tyres than is presently the case. The British Cement Association estimates that in 1997 the UK cement industry could potentially recover up to 190,000 tons of used tyres (U.K. Environment Agency, 1998). This is much more than the 22,000 tons processed by the cement industry in that year. In some regions, local environmental regulations and the lack of adequate infrastructure for collection and transportation may hamper further market development for energy recovery (Amari et al., 1999). Moreover, with the landfill ban in place in Europe and several States in the US, it seems likely for energy recovery to increase as well, more especially as energy prices are expected to increase in the near future. However, several potential market failures and barriers may be causing tyre-derived fuel (TDF) to remain underdeveloped.

4.3.2. Information and technological failures

As in the granulated rubber market, different users of scrap tyres in the energy utilisation segment require different characteristics of waste. Some cement kilns can process whole tyres, while others use shredded ones. Cement kilns typically burn at sufficient temperatures to oxidise the wire and benefit from both the energy release from oxidation and the resultant iron oxide that becomes a critical component in cement chemistry. At lower temperatures, however, the energy contribution from the wire is non-existent and will account for a lower energy value than that of either a wire-free or relatively wire-free TDF.

In these facilities, steel leads to more significant ash disposal problems, so wire-free TDF will be preferred. If the steel is not removed, the TDF must be cleanly cut with minimal exposed wire protrusion from the chips to facilitate mechanical handling. The only tyre incinerator in the UK at Wolverhampton closed down mainly because the available technologies did not allow separation of good quality steel. Investments to upgrade the technology were considered to be economically infeasible (Green Consumer Guide, 2001).

Other important specifications for TDF are the fuel's combustion characteristics, the handling and feeding logistics and environmental concerns. The recent development of American Society for Testing and Materials (ASTM) standards for TDF must be recognized as another step toward making tyre-derived materials a more accepted fuel input. The main advantage in this effort is that end users and potential end users now have an industry-accepted standard against which to compare all tyre chips. The other benefit to the industry is the development of a single sampling and testing protocol (RMA, 2002).

4.3.3. Search and transport costs and market power

Several observations can be made concerning search and transport costs. Given the fact that spatially dispersed waste generators and a fragmented recycling industry can be an indication for high transport costs, one can assume that this problem is less severe in highly populated countries such as the Netherlands. Nevertheless, the bulkiness of tyres implies relatively high transport costs. The transport of shredded tyres is about 30% to 60% cheaper, simply because fewer trips are necessary (Jang et al. 1997). High costs limit trade in used tyres in Europe and require recyclers to be near the raw material sources (Rosendorfová et al. 1998). In various studies it was estimated that transporting whole tyres for material recycling and energy recovery is not economically feasible over distances beyond approximately 150 to 250 kilometres (Sunthonpagasit and Duffey 2003b).

As a result, the area in which used tyre processors can retrieve their used tyres is relatively small. This limits the number of buyers and sellers thereby hindering the retrieval of a reliable flow of tyres, which is especially important for energy generators. This would also make an often-heralded solution to the problem of thin markets for secondary materials, the internet, less suitable for used tyres, even though there are many sites that do offer the possibility to sell or buy tyres. Moreover, it also increases the potential for market power to be exercised in the market.

4.4. Incineration and landfilling

Landfilling of tyres is categorically rejected as inefficient by most financial-engineering studies. These options are commonly considered a waste of valuable materials and causing environmental problems. Still, despite the implementation of the landfill ban in Europe by 2006, these options are still widely used in some countries.

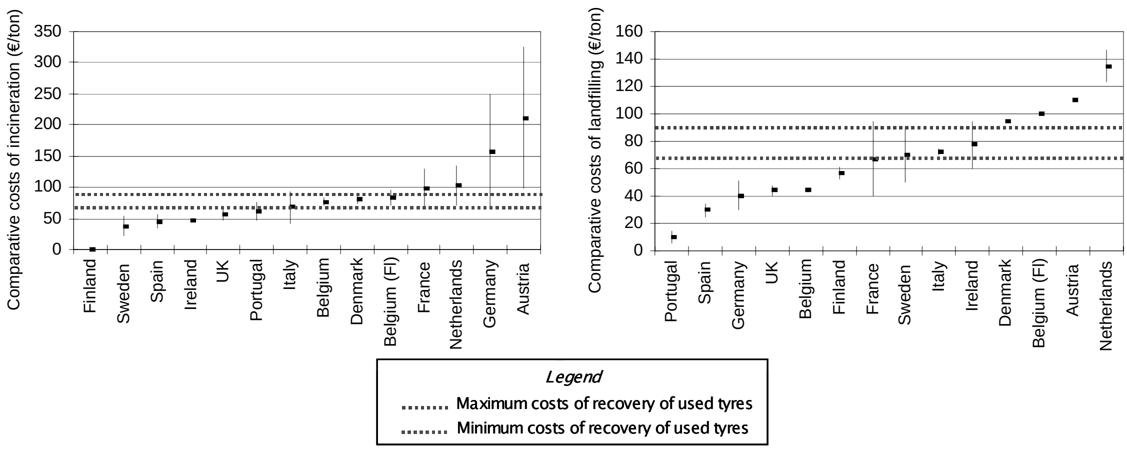
4.4.1. Policy failure

The cost for used tyre collectors of landfilling and incineration versus alternative management options is often the main driver of the configuration of the waste stage of used tyres. In the Netherlands, the disposal fee is the outcome of the commercial deals between the collectors and the processors. Because the disposal fee represents an important source of income for the tyre processing companies, it is difficult to obtain exact and reliable information on disposal fees for individual processing options. Some of the Dutch collectors claim that the disposal fee is the same for material and energy recovery options, which represents €83 per tonne of passenger car tyres and €68 per ton of truck tyres (Rosendorfova et al., 1998).

Based on several interviews in the sector and taking into consideration a relatively high share of energy recovery options, it can be assumed that the disposal fee paid to cement kilns is lower than the disposal fee paid to grinding companies. The disposal fee for shredded tyres (TDF), in turn, is usually lower than the disposal charge for whole tyres. Figure 8 shows incineration and disposal costs compared to recovery costs across Europe. It is clear that in a number of countries it is cheaper to landfill or incinerate tyres, rather than to recover them.

International differences in rules and regulations can be a cause for the transboundary movement of used tyres. Typical examples of such differences are the existence of regionally variation of tipping and hauling fees. Besides state and local environmental regulations, regional conditions and local labour rates influence these costs. In 2001 US tipping fees for passenger car and truck tyres ranged from $34-300 per ton. For used tyre processors these tipping fees are essentially a "negative cost" for raw materials and impact the producer's ability to keep crumb-selling price competitive with other scrap processors and thereupon virgin counterparts. Therefore the high variance of these fees seriously hamper the conditions needed for a level playing field. As shown in Figure 4.8, the cost of different processing options varies widely within Europe.

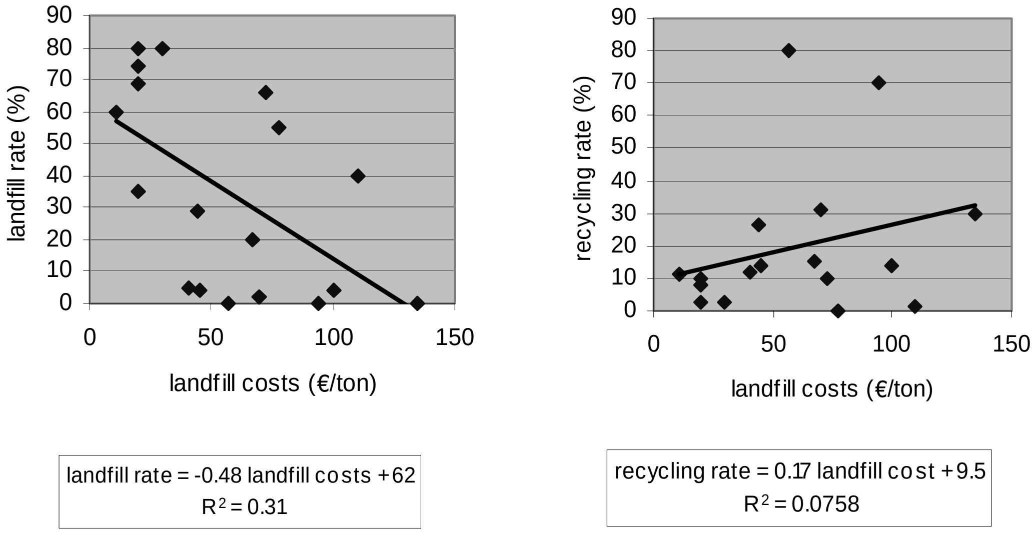
Figure 4.8. Comparison between incineration and landfill costs in Europe with (€/tonne)



Source: based on Hogg (2001).

By comparing these costs with the levels and costs of recovered and landfilled used tyres reported in Table 4.1. Respectively, it is possible to see some degree of correlation between the two variables. Figure 4.9 shows that landfill costs are a reasonable indicator for the landfilling level of used tyres. However, despite the correct sign, this is not true for the recycling rate of used tyres. In other words, high landfill costs are a pre-requisite to divert used tyres from the landfill, but are not a guarantee that they will be recycled.

Figure 4.9. Correlation between landfill costs and the levels of landfilling and recycling of used tyres in Europe in 2000.



Source: Data on costs from Hogg (2001) and volumes from ETRA (2003).

4.5. Illegal dumping

A growing problem in the waste management stage is the increasing incidents of illegal dumping of used tyres. Generally these tyres have already been sorted: only the tyres that are no longer retreadable or reusable are fly-tipped. Related to this problem is the abandonment of monofill sites without processing the stored tyres. Such examples have been recorded in Canada, USA, the Netherlands and the UK (U.K. Environment Agency 1998, US EPA 1998). Unfortunately, there is limited information available on the extent of the problem, although it could be about five per cent of fly-tipped waste by incident (Greater Manchester Waste Regulation Authority, 1995). UK's Environment Agency responded to over 1,300 incidents involving tyres in 2002. The actual total is likely to be substantially higher since many incidents are reported directly to local authorities and do not feature in Environment Agency statistics (UK Parliament, 2003).

The introduction of landfill taxes and collection fees are closely linked to this trend. After the introduction of a landfill tax in the UK in 1996, the Agency's North East Region found a 25% increase in fly-tipping the year after (U.K. Environment Agency, 1998). Generally, collectors accumulate used tyres from car repair shops, receive the collection fee, sort out the useful tyres and dump the remainder. In response to this trend, many governments have increased monitoring efforts of fly-tipping and strengthened the enforcement of penalties for this illegal practice. The UK's Department of Trade and Industry is considering the introduction of statutory reporting requirements through both the new tyre supply and used tyre disposal chains with the aim of improving the information base, promoting responsible tyre recovery and disposal practices while at the same time making it increasingly difficult for those fly-tipping tyres (UK Parliament, 2003).

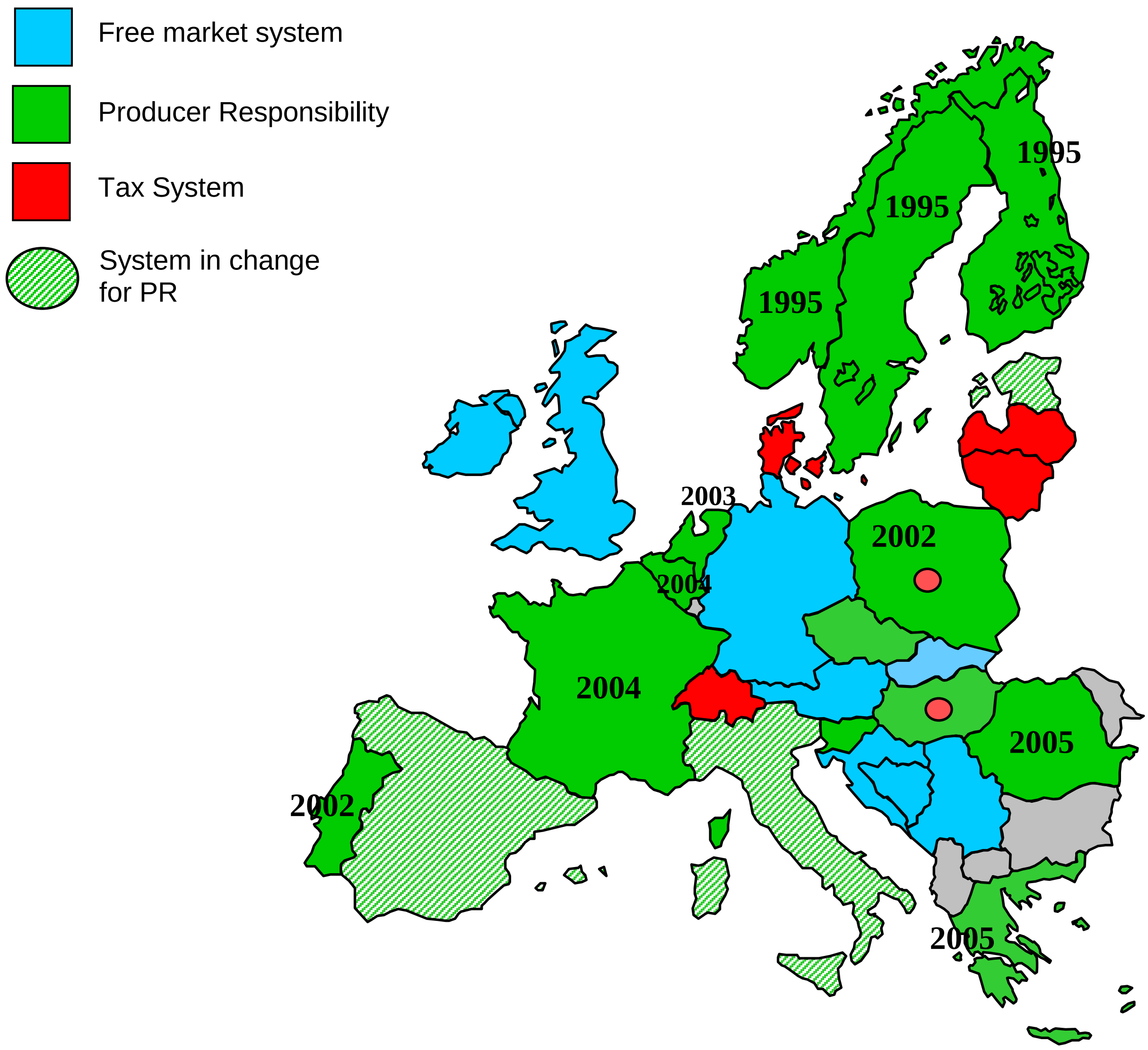
5. Policy influences in the market for secondary rubber

Public policies can have both a positive and a negative impact on the economic and environmental performance of recycling and waste management of used tyre sector. On the one hand, government strategies can be developed to correct the above market failures. On the other hand, if applied inappropriately, policies are sometimes counterproductive in making the waste stage of tyres more sustainable. This section aims to evaluate the existing tyre-related policies and suggest potential improvements in policy strategies.

5.1. Existing tyre-related policies

The different policy regimes in the tyre waste management sector are shown in Figure 4.10. The policy instruments that deal with waste and therefore encourage recycling, either directly or indirectly, can be divided into three broad categories: direct regulation, economic and communication policy instruments (Opschoor and Turner, 1994). In the following overview these instruments are briefly explained, supplemented by examples from the used tyre sector.

Figure 4.10. Policy regimes in the used-tyre sector of Europe in 2004.



Source: BLIC, 2004.

5.1.1. Direct regulation instruments

Direct regulation instruments include the compulsory legislation that prevents environmentally undesirable practices. The most commonly used direct regulative instruments are series of regulations, permits, technical standards, bans and restrictions, which provide the regulatory framework for environmental policy. Additionally, setting targets defined in legislation is becoming an important instrument (Rosendorfová et al. 1998).

A crucial regulative instrument for the used tyre sector is the *ban on landfilling* of whole and shredded tyres. Landfill bans stimulate the recycling of tyres by making dumping impossible (except via illegal disposal), and therefore other ways of dealing with the tyres have to be found. Currently several European countries, such as Belgium, Denmark, France and the Netherlands have already introduced a landfill ban on (whole or shredded) tyres and the EU Directive 1999/31/EC on the landfill of waste obliges all EU countries to ban the disposal of whole tyres to landfill from July 2003 and shredded tyres from July 2006. This will probably result in a significant increase of used tyres being put on the market after 2008.[59] In the US most municipalities also no longer permit the inclusion of tyres in regular landfills (Owen, 1998). The undesirable effects of tyres in landfills instigate landfill bans.

Specific targets for recycling sometimes supplement these bans. The new Dutch legislation on passenger car tyres, which is scheduled to be implemented in July 2003 demands that material recycling will be employed for 20% of the collected used tyres. Because current processing capacity or other outlets for material recycling in the Netherlands are insufficient to achieve the target, this will stimulate the various options for material recycling. The Dutch government envisages that producers will add a fee on new tyres to raise the money needed for this investment. The new law does not state a target for product reuse. One of the underlying reasons is the fact that tyres suitable for reuse – either directly or through retreading – have an economic value and are therefore not expected to end up in the take-back systems that will be used for scrap tyres. Another reason is the lack of influence producers are thought to have on the many factors that affect the extent to which this reuse will be adopted.

Technical standards also fall into the category of direct regulative policy instruments. In an environmental context standards usually state a maximum allowable content of certain substances in products or emissions, but in the case of used tyres – and secondary materials in general – they serve a different purpose, namely to enhance uniformity and certify a minimum quality. In the previous section the new standards for retread tyres have already been discussed. First and foremost these are meant to increase the safety of these tyres, but because safety perceptions by the public have given retread tyres a bad name, they may also increase demand for this very desirable option of dealing with used tyres.

Thus, the EU has attempted to tackle the problem of consumer's distrust of the safety of retreads by introducing a more uniform (safety/quality) standard. There were various national standards for retread tyres before, like the British BSAU 144 requirements, but in June 2001 the EU has acceded to the UNECE (United Nations Economic Commission for Europe) Regulations 108 and 109 for retread passenger car and truck tyres respectively. These Regulations require more or less the same quality controls as the widely applied UNECE Regulations 30 and 54 for new car and truck tyres. It is not yet mandatory for member states to adopt these Regulations as national standards, but several countries, such as Belgium, Denmark, Portugal, Spain, the Netherlands and the UK have done or will do so in the next two years (2003-2005) (VCA, 1999). This might help to free retreads from their bad image.

Initiatives are also being taken to develop standards for granulated rubber and recycled rubber products. European co-operation processes have started to produce standards for recycled tyre materials.

59. Calculations show an increase of 3,500,000 tonnes per year.

The European Tyre Recycling Association (ETRA) has prepared a basis document (CWA) for a standard, in close co-operation with a large number of organisations and companies. The European standardization body (CEN) will formulate the final standard (CEN, 2000). Ultimately, standards for granulated rubber are aimed at creating a more transparent market for this rubber product by increasing its uniformity.

Box 4.3. The British approach: will voluntary prove viable?

The Extended Producer Responsibility (EPR) approach, whereby producers and retailers accept responsibility for the waste stage of their products, is gaining popularity in the tyre sector. There is a discussion about the way this responsibility should be implemented: through imposed regulations from governments or voluntary initiatives from the industry itself. In Britain the tyre sector has clearly expressed their preference for the voluntary approach. The Government is willing to follow this route as long as the legal requirements are met. For the most part, these requirements stem from the national "Duty of Care" requirement, which is part of the Environmental Protection Act of 1990 and the EU Landfill Directive. Under the Duty of Care requirement anyone dealing with waste tyres is required to make sure tyres are not handled illegally and are only transferred to an authorised person together with a waste transfer note. The EU Directive banned landfilling of whole tyres from July 2003 and tyre chips three years later. Because in Britain almost 30% of waste tyres are currently being landfilled, the industry will have its work cut out for it. In order to comply with both the national legislation and to fulfil the obligations arising from the EU Directive the industry has set up several initiatives, the largest being the Responsible Recycler Scheme (RRS). This scheme was created in 1999 by the Tyre Industry Council. Its goals – besides preventing regulation – are to make sure used tyres are properly reused or recycled, to reduce illegal dumping, to create consumer awareness and to encourage best practice in the industry. Independent auditors monitor participants in the scheme annually and award a certificate, which can than be used as a marketing tool to reassure customers that tyres are being disposed of responsibly. A similar initiative, launched by Waste Tyre Solutions, the British market leader in the disposal and recycling of waste tyres, also aims to provide evidence that waste tyres have been reused and recycled in an environmentally friendly way. Both these schemes work with the Waste Transfer Note system, which records the movements of waste materials. If the combined efforts of the British tyre sector prove unsuccessful the government is, among other options, considering a statutory Producer Responsibility arrangement.

Sources: DTI, 2002; UTWG, 2001; TIC, 2001.

5.1.2. Economic instruments

Economic instruments are defined as instruments that affect the market conditions under which people and firms make their decisions, without directly reducing the decision space available to them. Instruments are labelled "economic" when they provide financial incentives for the balancing of costs and benefits of alternative actions open to economic agents (OECD, 1998). Thus, economic instruments can also be used to internalise the external costs of various activities. The most common examples of economic instruments are taxes, subsidies and deposit-refund schemes.

Besides landfill bans, governments also raise *taxes or levy charges* on the deposition of tyres going to landfill, as is currently the case in the UK and in some parts of the US (Sunthonpagasit and Duffey, 2003a). Like a ban, this also stimulates the use of other, more desirable, ways of dealing with tyres by making them comparatively cheaper. Examples of a tax used to internalise the costs of handling used tyres are the Danish and Hungarian taxes levied on the producers of new tyres. These taxes are passed on to consumers through higher prices. The revenues of the tax are used to pay for the organisation and recycling of the tyres. A similar system is currently in place in the Netherlands, and has lead to an increase of the price of a new tyre of approximately €2. In France, the costs of collection and recycling of used tyres, assessed at about €2.65 per car tyre and between €13 and €32 per heavy vehicle tyre, is passed on to the consumer since 2004. In Norway, the fees paid in 2003 for new (and retreaded) tyres are €1.25 for cars and €8 for trucks (Recycling International).

Secondary material industries can also be stimulated through the use of direct *subsidies*. This policy instrument has in general not proven to be very efficient. A telling example, as reported in Section 4.2.4, is the American use of grants and subsidies to some granulated rubber producers, which created unfair competition for the existing un-subsidized plants. Another problem related to subsidies is the failure to find markets for the processed materials, resulting in stockpiling of shredded tyres (Sunthonpagasit and Duffey, 2003a). Some of these problems could be solved by better instrument design.

5.1.3. Communication instruments

Communication instruments are aimed at bringing about changes in behaviour, in a more environmentally friendly (in this case reuse or recycling enhancing) direction. The 'communication' role of standards has already been discussed. In addition, information and education campaigns, product policies, eco-labels or voluntary sectoral agreements (covenants) and public procurement (by which governments set an example by purchasing for example recycled products and thereby may pave the way for advancement in products and technologies) are all examples of policy tools with a large potential in the waste management.

An example from Italy illustrates the use of government procurement to support retread tyres. The Financial Act of 2002 demands that 20% of replacement purchases of public motor vehicle fleets are retreads (AIRP, 2002). In addition to the direct effect, this stimulates the retread sector, already important in Italy, and helps convince the public of their safety. The US government is also active in promoting retreads through public procurement programmes. US EPA (2002) recommends that procuring agencies establish preference programs consisting of two components: (a) procurement of tyre retreading services for the agencies' used tyre casings; (b) procurement of tyres through competition between vendors of new tyres and vendors of retread tyres. In the event that identical bids are received in response to a solicitation, all other factors being equal, procuring agencies should provide a preference to the vendor offering to supply the greatest number of retread tyres.

Government procurement can also be found in the use of granulated rubber in asphalt. In 1991, the US Congress passed the Intermodal Surface Transportation Efficiency Act (ISTEA), requiring the states to use scrap tyres for surfacing of federally-funded highways. If it had been implemented, proponents claim the act would have increased the use of scrap tyres by up to 70 million in 1997. However, in 1995 ISTEA was repealed, presumably as a consequence of lobbying efforts of the asphalt industry. ISTEA had initially shown promise of developing granulated rubber markets for asphalt but after repeal most processors who came into this business in anticipation of this market left due to downward pressure on granulated rubber prices (Sunthonpagasit and Duffey, 2003a).

5.2. Comparative assessment of instruments

None of the above-mentioned instruments is capable of addressing all of the identified market distortions and failures. Each policy type has its specific pros and cons. Although economic and communication instruments seem to be most in line with the generally recognised principles of promoting economic efficiency, it is unlikely that these instruments eliminate the need for regulation entirely. Regulatory instruments provide a greater degree of certainty of outcome than other types of instrument. The establishment of a ban on landfilling used tyres, for instance, is considered to have been the strongest incentive for tyre recycling activities in the last decade, although potential increases in illegal disposal must be addressed. Therefore, a pragmatic approach should be followed in designing policy schemes to reduce failures in the secondary rubber market.

Sets of measures should be tailored to suit the local circumstances, the problems addressed and product types for which they are applied. Ogilvie and Poll (1999) address the relationship between the instrument choice and the problem addressed. They suggest the implementation of a variety of policy instruments for different purposes. Technical problems should ideally be addressed through research and subsidies, while the imbalance between the supply and the demand for recyclable materials requires awareness campaigns and waste charges to be introduced. Alternatively, waste exchanges could bridge this gap. What instruments are best suited to address problems such as illegal disposal and consumer perception will be described in the following section.

6. Policy recommendations

If recycling has significant environmental and economic benefits, then the expansion of the capacity of the recycling industry and the improvement of the secondary rubber market has to be promoted further. Most countries discussed in this study acknowledge this. The question remains how this aim should be accomplished. Several policy recommendations have been derived from the above analyses that are aimed specifically at reducing the market failures and barriers for rubber recycling markets. By combining these principles, a more effective market for secondary rubber can be achieved.

6.1. Set clear targets while maintaining flexibility

In Europe, the implementation of a European wide ban on landfilling has had an enormous impact. Although such uniform ban may not have necessarily led to the most efficient outcome in economic terms, it certainly created large momentum in the tyre recycling business. Countries and organisations are forced to come up with creative solutions to meet with the European legislation. Earlier attempts on a purely voluntary basis, such as those attempted in the Netherlands, failed miserably. (An argument in favour of voluntary agreements is that the industry is faced with considerable flexibility in meeting their obligations.)

The earlier experiences with voluntary schemes show that such systems only function well if economic gains can be achieved, indicating the possibility that such measures only confirm existing trends in the market. As shown in the US, voluntary programmes without these gains will only be set up to avoid legislation. Another drawback of voluntary schemes is that they are particularly difficult to implement if there are many players, as seen in the Dutch experience. The advantage of mandatory policies is the creation of a clear set of rules for all players in the market. Indeed, the European industry favours mandatory systems to minimise the problem of free-riders and reduce uncertainty surrounding future legislation. Within mandatory schemes producers should have the flexibility to operate in a way they consider optimal.

The use of economic instruments should also be more actively considered by governments. For instance, deposit-refund systems for tyres could provide incentives for recovery and reduce the potential for illegal disposal (see below). Other options include ‘advance disposal fees’ or ‘subsidy-tax systems’. The latter are similar to deposit-refund systems, but provide greater flexibility in implementation. If tyres with different downstream environmental consequences can be targeted through such measure applied upstream, this may prove effective in internalising environmental externalities.

6.2. Well-targeted support for research and development

At present, the rubber recycling sector suffers from a number of technological impediments. Governments could support research and development (R&D) to overcome these problem areas in the recycling industry. First, the currently limited market of secondary material applications could be expanded. For example, products could be designed in such a way that they could contain a higher content of recycled materials or could be used in different production processes: this is the case for example in the French iron and steel industry where the rubber of used tyres is used as a carbon reducing agent in electric arc furnaces. Approximately 15% of used tyres (about 60 000 Tonnes) will thus be recovered in the near future by this industry (l'Usine Nouvelle).

Secondly, improved product design can increase the recyclability of tyre-related products. In addition, it may be feasible to provide support for the development of improved recovery technologies, overcoming some of the obstacles which increase costs at present. However, as mentioned earlier, government support in the form of subsidies in technological development have not proved to be very effective. Therefore, governments should be cautious in providing these, particularly in the longer term.

6.3. Actively fight illegal dumping

In the case of used tyres, one of the main market distortions to be addressed includes illegal disposal. Incentives for illegal disposal will arise from imposition of a ban on landfilling or from the application of charges. Illegal disposal is generally dealt with through the establishment of deposit refund schemes because these provide a strong incentive for the consumers to return their used products in a proper state. In the case of used tyres, however, illegal disposal does not occur at the consumer level, but at the level of collectors. Therefore, illegal dumping of tyres may be better addressed through the certification of tyre collectors and through increased monitoring of illegal dumping events. The UK Environment Agency is running a Tyre Watch campaign, which both makes businesses aware of their responsibilities and pushes for stiffer sentences for illegal dumping. The Agency actively seeks prosecution. The Environmental Protection Act makes provision for fines of up to £20,000 and possible imprisonment for each illegal deposit of waste (U.K. Environment Agency, 2003).

6.4. Promote retreads to the public

Retreading should be seen as the first step in management of post consumed tyres. It saves 80% of raw materials and energy necessary for production of tyres and reduces the quantity of waste to be disposed. Retreading, therefore, should be promoted by prolonging the lifecycle by producing retreadable tyres and improving their image, thus creating markets for retreaded tyres. Possible measures to achieve that are to implement quality standards for tyre manufacturers to produce retreadable casings, promote public procurement policy (use of retreads on government vehicles) and, most importantly, provide information to the public on the quality and existence of retreaded tyres. Retailers play an important role in this respect. If retailers promoted retreads more actively, the retreading industry would probably not suffer such heavy losses. Why they have not done so thus far is unclear – but some possible explanations related to market power and segmentation have been forwarded in this report.

Governments are in a strong position to boost the demand for recycled materials. Government procurement can benefit the recycling sector of secondary rubber tremendously by supporting the adoption of recycled products or setting minimum content rules, for example for rubber crumb in road surfaces. Effects of public procurement are not negligible as shown by the example of California: indeed the elimination of the federal requirement that states use of rubberized asphalt in federally-funded highway projects is reported as having slowed the development of this market (Market Status Report: Waste Tyres, State of California, October 1996). Moreover, by prescribing government-owned vehicles to use retreaded tyres, government procurement schemes can have a direct (i.e. expanding the market for retreads) and an indirect (i.e. signalling function for citizens) impact on the market for retreaded tyres. As mentioned before, several stimulation measures have been taken in the US to promote the use of retreaded tyres. In 1998 President Clinton signed Executive Order 13101, which mandated the use of retreaded tyres on all government vehicles. Combined with strong endorsements from for example the EPA, most federal government fleet vehicles are now using retreaded tyres. The US Postal Service has, for instance, successfully utilised retreads on all their vehicles for several years and the US Army Tank-Automotive & Armaments Command won a recycling award in 1999 from the Clinton Administration for its program in retreading tactical tyres. In this program retreaders are tested and certified annually and lists of certified retreaders are made available to other government institutions.

6.5. Encourage extended producer responsibility

Extended producer responsibility (EPR) is an approach increasingly being used in Member countries to address the environmental impacts from used-tyres at their post-consumption phase. For instance, the tyre manufacturing industry in Europe has formally accepted its responsibility in early 2002. Plans for setting up the EPR scheme are currently being developed by BLIC, the European Association of Tyre Manufacturers.

In each country under an EPR scheme, the tyre manufacturers collectively create a joint company in charge of the collection and the recovery of end-of-life tyres. The costs of this scheme are covered by a fee paid by the consumer when buying a new tyre; the consumer then returns the end-of-life tyre free of charge to the dealer. By giving an incentive to the consumer and using exclusively approved companies and network for the collection and recovery, this scheme has proven its efficiency in terms of recovery, cost and environmental protection.

The primary tyre manufacturers are a powerful player in the tyre supply chain and therefore are in a good position to stimulate the use of retreaded tyres. EPR can also form an incentive for the Asian manufacturers that produce the cheap non-retreadable tyres, to increase the retreadability of their product.

6.6. Implement harmonised standards for retreads, crumb rubber and TDF

Safety should of course not be compromised just to reduce the problem of waste tyres, but studies have shown that retreads can be just as safe as new tyres and the retread industry even claims they are of a higher quality than the budget tyres from Asia and Russia. The main problem then is to convince the public of this fact. One way to do this is the introduction of clearly communicated safety standards for retreaded tyres that should be very similar to standards for new tyres. The newly adopted UNECE standards by the EU, referred to earlier, are a good example but because it will not be mandatory for countries to adopt these standards, a lot of the benefits of having one uniform standard might be lost. It is therefore required that as many (future) Member States as possible adopt the standards. It should not however be expected to be a panacea for retreaded tyres. In the US where standards for retreaded passenger car tyres do exist, their use still falls short of potential. Other stimulants like public procurement are necessary.

Even though the TDF and crumb rubber industries are certainly not in their infancies anymore and have proven their market viability, it cannot be said that they have fully matured yet. TDF has not taken the place among other fuels it deserves on the basis of its characteristics. Though this can partly be explained by the fact that there are reliability issues in the supply of TDF, compared to conventional fuels, there are also institutional factors that limit its growth. Another example is the extensive testing phase British cement kilns have to undergo before they are authorised to use TDF. With clear air emission standards for tyre using facilities it ought to be much easier to check if the requirements are met and then issue the permits. Of course there should be proper air emissions for TDF, but this should not needlessly hinder its adoption. Where air emission standards should clearly be a government issue, other standards for TDF relating to the wire content, size etc. can be developed by the industry itself.

Indeed, this has already happened in the US, where the American Society for Testing and Materials (ASTM) has developed such standards. Where such standards do not yet exist governments could stimulate their development or adoption of existing foreign standards. Governments might also have a role to play in harmonizing different existing standards. Both the ASTM and the European Committee for Standardisation (CEN) have already developed standards for crumb rubber, so the problems relating to uncertainty about quality and other matters should be substantially reduced in the near future. Again, in countries where these standards do not yet exist governments could promote their adoption.

REFERENCES

AAMVA (2000). *The Commercial Driver's Handbook*. American Association of Motor Vehicle Administrators, Washington DC.

AARDVARK ASSOCIATES (2001). *Post Consumer Tyres: Shifting Perceptions to Create Opportunities*, Working Paper prepared for the Retread Manufacturers Association, Dorset.

ADEME (2001) France. « Étude du coût des filières de traitement des pneus usagés » Rapport Final.

AIRP (2002). *Drop in energy consumption and environmental pollution: One million of oil barrels were saved in 2001 thanks to retreads*, Press Release 3 December 2002, Italian Association Tyres Retreaders, Bologna.

ALIAPUR (Filière française de valorisation des pneus usagés). Website address : http://www.aliapur.fr

AMARI, T., N.J. THEMELIS, and I.K. WERNICK (1999). "Resource recovery from used rubber tires". *Resource Policy* 25; 179-188.

BLACKMAN, A. and A. PALMA (2002). "Scrap Tires in Ciudad Juárez and El Paso: Ranking the Risks". Resources for the Future (RFF), Washington DC. Discussion paper 02-46.

BLIC (2003). "BLIC Facts and Figures". September 2003. Bureau of International Recycling. Brussels.

BLIC (2004) "End of Life Tyres in Europe An overall picture of ELTs management", presentation at International Automobile Recycling Conference, Geneva March 11, 2004

BROWN, D.A. and W.F. WATSON (2000).. *Novel Concepts in environmentally friendly rubber recycling*, paper presented at the International Rubber Forum 2000 held in on 9 November 2000, Antwerp.

BURGER, C.P.J. and H.P. SMIT (2001).. "Economic growth and the future of natural rubber", presented at Yamoussoukro, Cote d'Ivore, 5-9 November.

CAREY, Jason (1999). *Survey of Tire Debris on Metropolitan Phoenix Highways*, Arizona Department of Transportation, report no. ATRC-99-11, Phoenix.

CEN (2000). *CEN Workshop Business Plan N2", CEN Workshop on the preparation of standards for post-consumer tyre materials and applications*, European Committee for Standardisation 26 May 2000, Brussels.

DTI (2001). *Tyre Recycling*, Environment Directorate, Department of Trade and Industry, London.

DTI (2003). *Discussion Paper on a possible Producer Responsibility Model for Used Tyres*. Sustainable Development Directorate. Department of Trade and Industry (DTI), London. http://www.dti.gov.uk/sustainability/pub.htm

EC (1997). "Proposal for a Council Directive on the landfill of waste". COM(97)105, European Commission EC, Brussels.

ENVIRONMENT AGENCY (1998). "Tyres in the environment". Government of the UK, London. http://www.environment-agency.gov.uk/commondata/105385/ea_tyres_report.pdf

ENVIRONMENT AGENCY (2003). "Tyres Watch Programme". Government of the UK, London. http://www.environment-agency.gov.uk/business/444251/444707/288582/?lang=_e&region=

ETRA (2003). "Introduction to tyre recycling 2003". European Tyre Recycling Association. Brussels. http://www.etra.eu.com/

FAO (2002). "Commodity notes: Rubber". Food and Agriculture Organisation of the United Nations. Economic and Social Department. Rome. http://www.fao.org/

FERRER, G. (1997). "The Economics of Tire Remanufacturing". *Resources, Conservation and Recycling*, No.19, pp. 221-255.

GUELORGET, Y., V. JULLIEN and P.M. WEAVER (1993) "A lifecycle analysis of automobile tires in France". INSEAD Working Paper 93/67/EPS. Fontainebleau.

GREATER MANCHESTER WASTE REGULATION AUTHORITY (1995). "Fly-tipping in Greater Manchester". Greater Manchester Waste Regulation Authority, Manchester.

GREEN CONSUMER GUIDE (2001). "Tyre group push recycle scheme". Newsletter #62. Greenmedia Publishing. 24 August 2001. http://www.greenconsumerguide.com/news62.html

HOGG, D. (2001). *Costs for Municipal Waste Management in the EU*. Final Report to Directorate General Environment, European Commission. Eunomia Research & Consulting.

IRSG (1998). International Rubber Study Group, *Rubber Statistical Bulletin*. Vol. 52, No.4, Eng-land.

JANG, Ji-Won, Taek-Soo YOO, Jae-Hyun OH and Iwao IWASAKI (1997) "Discarded Tire Recycling Practises in the United States, Japan and Korea", *Resources, Conservation and Recycling* 22, pp.1-14.

JONES, K.P. (1997). "Rubber and the Environment". International Rubber Research and Development Board. Paper presented at the International Rubber Forum, 12-13 June 1997, Liverpool. International Rubber Study Group. London.

MASSACHUSETTS OPERATIONAL SERVICES DIVISION (2000). "Retread Tires", *Recycled Product Fact Sheet # 9*, available on the internet at: www.state.ma.us/osd/enviro/products/tires.htm.

MoA (1993). "Regulation of the Federal Ministry of Agriculture for Air Emission Limits in Cement Kilns", (BGBI No. 63/1993), Vienna.

OECD (1998). *Evaluating Economic Instruments for Environmental Policy*, OECD, Paris.

OGILVIE, S. and J. POLL (1999). *Developing Markets for Recycled Materials*, AEA Technology Environment, AEAT-5538 Issue 1, Oxfordshire.

Opschoor, J.B. and R.K. Turner eds. (1994). *Economic Incentives and Environmental Policies Principles and Practice*, Kluwer Academic Publishers, Dordrecht

OWEN, K.C. (1998). "Scrap Tires: A Pricing Strategy for a Recycling Industry", *Corporate Environmental Strategy*, 5(2): 42-50.

PYANOWSKI, D. (2002). "European End of Life Tyres (ELT)", Presentation for the Rubber Manufacturers Association (RAC/RMA) Conference held on 23-25 October 2002, Montreal.

RMA (2002 and 2003). *U.S. Scrap tire markets 2001*. Pub #: MAR-2001. Washington DC. Available at http://www.rma.org/

RECYCLING INTERNATIONAL (June 2003, N°5) "Collection Fee Systems for Scrap Tyres".

REITER, B. and R. STROH (1995). *Use of Waste in the Cement Industry*. Federal Environmental Agency-Austria.

ROSENDORFOVÁ, M., I. VÝBOCHOVÁ and P.J.H. van BEUKERING (1998). *Waste Management and Recycling of Tyres in Europe*, R98/13, Institute for Environmental Studies, Amsterdam.

SMITH, F. G. et al (1995). "Testing and Evaluating Commercial Applications of New Surface - Technology Utilizing Waste Tires". *Resources, Conservation and Recycling*, No.15, pp. 133-144.

SUNTHONPAGASIT, Nongnard (2003a). "Scrap Tires to Crumb Rubber: Market and production Issues for Processing Facilities". Doctoral dissertation. The George Washington University, Engineering Management & Systems Engineering Department (EMSE), Environmental & Energy Management Program (E&EM). January 16, 2003. Washington DC.2003a

SUNTHONPAGASIT, N. and M.R. DUFFEY (2003b). "Scrap Tires to Crumb Rubber: Feasibility Analysis for Processing Facilities", *Resource, Conservation and Recycling*, (In press).

TIC (2001). "Responsible Recycler Scheme", in *TIC Report*, Tyre Industry Council, July 2001.

TRL (1992). *The incidence of HGV tyre debris on the M4 motorway*. WP/VS/222. Transport Research Laboratory (TRL), Crowthorne, Berkshire.

UNCTAD (1996). "A Statistical Review of International Trade in Tyre and Tyre-Related Rubber Waste for the Period 1990-1994". United Nations Conference on Trade and Development, Papers on Commodity Resources, Trade and Environment, No.1 Geneva.

U.K. PARLIAMENT (2003). Question on tyres by Norman Baker for the Secretary of State for Environment, Food and Rural Affairs [99271]. London.

UNSD (2002). COMTRADE United Nations, Geneva.

URI (2001). *Mosquitoes, disease and scrap tires*. University of Rhode Island. Department of Environmental Management, Office of Mosquito Abatement Coordination. http://www.uri.edu/research/eee/tires.html

US EPA (1991). “Markets for scrap tires”. In: *Scrap Tires Technology and Markets*, published by Noyes Data Corporation. Park Ridge.

US EPA (1998).”Illegal Dumping Prevention Guidebook”. United States Environmental Protection Agency. Washington DC, EPA-905-B-97-001

US EPA (1999). “State Scrap Tire Programs. A quick Reference Guide: 1999 Update”. United States Environmental Protection Agency. Washington DC, EPA-530-B-99-002.

US EPA (2000). *2000 Buy-Recycled Seri1es: Vehicular Products*. United States Environmental Protection Agency. Washington DC. www.epa.gov/cpg/pdf/vehi-00.pdf

US EPA (2002). Comprehensive Procurement Guidelines: Retread Tires. US Environmental Protection Agency (EPA), Washington DC. http://www.epa.gov/cpg/index.htm

L’USINE NOUVELLE, France, N°2867, 17 April 2003.

UTWG (2000). *Fourth Annual Report*. Used Tyre Working Group, London.

UTWG (2001). *Fifth Annual Report*. Used Tyre Working Group, London.

VAN BEUKERING, P.J.H. van and M.A. Janssen (2001). A dynamic integrated analysis of truck tires in Western Europe, *Journal of Industrial Ecology* 4(2): 93-115.

VCA (1999). "VCA Guide to the Type Approval of Retread Tyres", VCA 036, Vehicle Certification Agency, Bristol.

VIRGINIA DEPARTMENT OF STATE POLICE (2000). "Need for Standards for Recapped Tires", Commonwealth of Virginia, Richmond.

VROM (2002). Decree on waste management of used-tyres/ Ontwerp-besluit beheer autobanden. Dutch Ministry of Environment (VROM). Staatcourant 10 October 2002. No. 195. p.11.

WORLD TYRE INDUSTRY (1997). Tyre Recycling and Disposal. World Tyre Industry, The Economist Intelligence Unlimited, pp. 22-38.

CHAPTER 5:

PUBLIC POLICY AND RECYCLING MARKETS

by

Nick Johnstone and Soizick de Tilly
OECD Environment Direcorate

1. Introduction

It has often been argued that levels of recyc7ling for a number of potentially recyclable materials are sub-optimal, both from an economic and from an environmental perspective. For this reason many governments have sought to encourage recycling through measures such as subsidised municipal collection schemes, mandated sorting of wastes, recycled content standards, government procurement policies, etc. Such measures directly seek to affect the level of recycling either by decreasing the cost of supply or by increasing the level of demand (OECD 2004).

The fundamental public objective for such measures is not to encourage recycling for its own sake. While recycling may feature prominently in standard classifications of the 'waste hierarchy' – behind waste prevention, but before energy recovery, incineration and landfilling – such public policy preferences are implicitly motivated by a belief in the need to reduce environmental impacts to the greatest extent possible. In many, but by no means all, cases general adherence to a standard waste hierarchy is approximately consistent with such an objective. In some cases, incineration may be preferable to landfilling, which may in turn be preferable to recycling (see Rasmussen et al. 2005). Such cases highlight the potential for strict adherence to a waste hierarchy to result in perverse environmental impacts.

Moreover, it can never be strictly optimal in an economic sense to adhere strictly to the waste hierarchy. The overall social costs of different waste strategies depend not only on their environmental implications, but also on the financial costs of their attainment. For some waste types in some locations, the financial costs for a given strategy which is environmentally preferable may be such as to result in greater overall social costs relative to another strategy 'lower' in the hierarchy. Indeed, extreme adherence to the waste hierarchy would imply that waste prevention would be pursued indefinitely at whatever the financial cost. With rising marginal financial costs for any given strategy[60], the social cost implications can be significant.

Taken broadly, therefore, an optimal waste hierarchy is more concerned with the relative proportion of different waste strategies, and this will differ by waste type and location. Recycling is, therefore, a means (a waste strategy) to an end (the mitigation of environmental bads). Specifically, increased recycling is thought to be closely associated with reductions in the generation of many environmental public bads (i.e pollution emissions from incinerators or leachate from landfills) related to solid waste generation, and as such policy measures which increase recycling are thought to contribute to the mitigation of such impacts.[61]

However, markets for recyclable materials are affected by a large array of other policy measures – some of which may increase recycling rates, and others which may have the opposite effect. Thus, prior to (or simultaneous with) introducing targeted recycling policies, governments can undertake a wide variety of policy reforms in order to ensure that the rate of recycling of wastes is optimal. These different types of policy reforms can be classified into two broad categories:

The introduction of policy measures to directly internalise waste-related environmental externalities. Any measure which seeks to internalise the external environmental impacts of waste and waste management will affect recycling rates. Thus, unit-based waste fees and landfill taxes can provide incentives to recycle materials. Similarly, technological restrictions on alternative measures of waste disposal and incineration will affect recycling rates (Porter 2002).

The removal of policy failures in substitute primary material markets. The development of recycling markets is, in many cases, dependent on the substitution from virgin to recycled material inputs. However, subsidies favouring virgin materials usage are reported for several resources (e.g. fossil fuels, minerals, pulpwood, in OECD 2000) which compete with recyclable materials. In addition, inappropriate product and material standards may discourage the use of recyclable materials in place of primary materials. All of these can prejudice the development of markets for recyclable materials.

The key point is that in neither of these cases is the fundamental public policy objective directly related to rates of recycling per se: in the first case the objective is to reduce environmental damages; and; in the second case it is to remove policy-induced distortions in the market. Changes in recycling rates are the incidental outcome of the achievement of other legitimate public policy objectives.

Unfortunately, it may not always be sufficient to introduce policies which change the relative costs of different waste management options and the relative prices of primary and recyclable materials. Indeed, it has been argued in this report and elsewhere that markets for some recyclable materials are inefficient (see ECOTEC 2000, EPA 1993 and OECD 2004). This is environmentally significant since factors which reduce the efficiency with which markets for recyclable material operate can result in sub-optimal levels of recovery and recycling. As such, policy measures which seek to remove market failures and barriers will tend to result in more efficient markets and, perhaps, increased rates of recycling.

60. For instance, 'preventing' the last 5% of a given waste stream for recycling is likely to be much greater than 'preventing' the first 5%. Similarly, recovering the final 5% of a given material is likely to be much greater than recovering the first 5%. While such relationships are likely to be non-linear, there is little question, that marginal costs eventually rise, and probably quite dramatically.

61. While it is sometimes argued that recycling can result in the protection of upstream environmental goods associated with primary resource extraction, these are much more efficiently and effectively addressed through resource management policies in the relevant sectors (i.e. forestry, mining, oil exploration, etc…).

The general objective this concluding chapter is, therefore, to place recycling policies in a broader public policy context in a very general way, briefly reviewing the potential impacts of:

- Policies targeted directly targeted to increase recycling rates (Section 2);
- Other policies targeted at waste-related externalities (Section 3); and,
- Policies which support the use of substitute 'virgin' materials (Section 4).

The more specific objective is to examine what measures should be undertaken to address some of the specific market failures and barriers which appear to affect markets for recyclable materials (Section 5), drawing upon the previous chapters. The chapter concludes (Section 6) with some general policy conclusions.

2. Policies Targeted Directly at Recycling

Recycling is not an economic activity which should be supported by government policy measures in and for itself. However, environmental externalities associated with solid waste generation and management can be considered public bads. Moreover, solid waste is an area in which it is notoriously difficult to apply "first-best" environmental policies to reduce these bads (Eichner and Pethig 2001, Calcott and Walls 2000). This is due in part to the "mixed" nature of much of the solid waste stream. A wide variety of substances with very different potential environmental impacts are aggregated. This is commonly the case for municipal solid waste. For industrial waste, waste separation may be less costly since it is often (but not always) more homogeneous, but there is still considerable heterogeneity in many cases. Distinguishing between all types of waste in terms of their different externalities would impose exorbitant administrative (private and public) costs.

It is for this reason that much of environmental policy in the area of solid waste is designed to disaggregate heterogeneous waste streams, and "pull out" those elements which have particularly high values, whether positive (recyclables) or negative (hazardous). It is, of course, with the former case which we are primarily interested in this study. Since recyclable waste is "embedded" in the waste stream (and within individual products in the waste stream) public authorities seek to provide the right short-run (separation) and long-run (product design) incentives to separate out those elements which can be recovered cost effectively. However, it is also important to note that many recyclable materials are themselves potential sources of significant environmental damages (i.e. used lead-acid batteries or used lubricating oil), and there are, therefore, both negative and positive incentives to recover such waste (OECD 2004).

As noted above, targeted recycling policies can be distinguished between those that seek to increase supply for recyclables and those that seek to increase demand. Table 5.1 provides a partial list of measures, including the precise nature of the incentive provided and some of the potential strengths and weaknesses of the different measures.

A number of empirical studies have been undertaken in order to determine which of these instruments is likely to be most environmentally effective and economically efficient. (For a survey related to municipal solid waste see Kinnaman and Fullerton 1999.) The choice will, of course, depend upon the specific material to be recycled. There are, however, some general principles which emerge from the literature:

Table 5.1. An Overview of Targeted Recycling Policies

Instrument	Incentive	Comments
Supply-Side		
Ban on Landfill or Incineration of Recyclables	By preventing disposal, encourages increased supply of recyclables.	Depending upon conditions in recycling markets, may divert some waste into illegal disposal. It may also reduce recovery-for-energy for some types of waste.
Deposit-Refund Schemes for Goods with Recyclable Content	By increasing the opportunity cost of disposal (legal or illegal), encourages recovery of recyclables.	Deals effectively with wastes for which illegal disposal is an option and provides strong upstream incentives for redesign. May have high administration costs for some types of waste.
Public Collection Schemes for Recyclables	By reducing the opportunity cost of collection, increases supply of recyclables.	Effectiveness depends upon whether sorting at source is required and upon programme design (sorting, frequency, location, etc…).
Subsidies for Material Recovery Facilities	By increasing the capacity and decreasing the cost of recycling, increases supply and reduces costs of recyclable materials.	Imposes significant information requirements for public authorities. May have negative effects on waste prevention since it reduces overall material input costs.
Product Take-Back Targeted on Goods with Recyclable Content	By increasing the opportunity cost of designing products which can not be recycled, increases the supply of recyclable material.	May impose high administrative costs. Unlikely to be economically efficient unless targets are disaggregated by product and material.
Mandated Sorting of Recyclables	By facilitating downstream waste separation, decreases the financial cost of recycling.[62]	Significant benefits for downstream processing. May reduce aggregate collection rates due to increased opportunity costs (time and effort) for waste generators.
Demand-Side		
Recycled Content Standards	By requiring the use of recyclable materials in products, increases demand.	Likely to be environmentally effective, but economically inefficient unless standards are highly differentiated by material, sector and location.
Tradable Credits for Recycled Content	By encouraging the use of recyclable materials in products, increases demand.	Similar to recycled content standards, but introduces flexibility to allow for cost equalisation, and thus economic efficiency.
Public Procurement Preferences for Goods Made from Recycled Materials	By encouraging procurement officers to purchase goods made from recyclable materials, increases demand.	Only targets a part of the market - may result in unintended negative effects on private markets. May encourage product innovation.
Eco-labels Related to Recycled Material Content	By encouraging consumers to purchase products using recyclable materials increases demand.	Allows consumers to express preferences for environment through purchasing decisions. Effectiveness depends upon consumer preferences.
Support for Research and Development for Use of Recyclable Materials	By increasing potential uses for recyclable materials may increase demand.	Likely to lead to product innovation, but may have unintended negative consequences on waste prevention.

To the extent possible, efforts to encourage recycling should not come at the expense of reducing waste generation at source (increasing waste prevention). For instance, subsidies for material recovery facilities, recycling collection schemes and recycled product development will encourage substitution of inputs toward recyclables, but will also (if not supported by other measures) increase overall demand for

62. Although non-financial costs for the household (i.e. sorting time, storage space, etc…) are likely to rise.

material inputs.[63] While the negative scale effects are unlikely to be as important as the positive substitution effects, they should be borne in mind – particularly if the environmental benefits of recycling relative to other waste management options are marginal.

Instruments with a single point of incidence (advance disposal fees, subsidies for collection or processing, etc…) are unlikely to be able to provide incentives throughout the entire design-production-consumption-disposal chain due to missing markets for some product characteristics at some stages. In such cases incentives for recycling (and designing for recycling) are not transmitted all the way along the product cycle (e.g. used electrical appliances). Two-part instruments (such as tax-subsidy and deposit-refund schemes), or the joint application of two separate instruments are preferable.

If there is potential illegal disposal of recyclable wastes, it is important to particularly important to provide incentives for their recovery. For those materials for which the costs of concealment are low (e.g. used lead-acid batteries) positive incentives (e.g. deposit-refund) are likely to be more effective since the costs of enforcing negative incentives (e.g. fines) may be particularly high.

If policies are introduced on both the supply-side (i.e. collection schemes) and the demand-side (i.e. public procurement) or at different points within the supply and demand chain, it is important to co-ordinate their introduction closely in order to ensure that price volatility is not exacerbated. Arguably some of the wide swings in used newsprint prices in the early years of the market arose out of such a lack of co-ordination.

However, the key point to bear in mind is that while support for recycling through public policy measures can be justified as a second-best policy, the ultimate objective should never be to encourage recycling for its own sake. If there are better ways to target the public bad, then these should be applied. Unfortunately, this is not always possible, an issue to which we turn in the next section.

3. Environmental Policies Designed to Reduce Waste-Related Environmental Damages

As noted above, the ultimate objective of waste-related environmental policies is to reduce environmental impacts to their economically efficient level. (See OECD 2004 for a full discussion.) If the use of recyclable materials has fewer waste-related environmental impacts than primary materials, the internalisation of waste-related externalities should increase recycling rates, even if the ultimate policy objective is to decrease environmental damages rather than to increase recycling rates per se. Other types of externalities, such as those associated with upstream primary resource extraction and exploitation, may also be reduced. However, such impacts are best addressed at their point of generation (i.e. forestry, mining, etc…) and not downstream at the point of commodity production or waste generation.

If the incentives are appropriate (and there are no other market or policy failures), the level of recycling should be such as to equalise the marginal damages from primary and secondary resource use. However, this is, of course, often exceedingly difficult to implement in practise and it is for this reason that the policies outlined in Section 2 above have been introduced.[64] Nonetheless, despite their inability to target waste-related externalities directly, more general solid waste policies will have impacts on recycling

63. This is perhaps the clearest example of the cost of targeting environmental policy on a proxy for environmental damages, rather than on the damages directly. In such cases there is always potential for "slips between the cup and the lip", particularly in the long run. In this example, the output effect (reduced material costs) will undermine the substitution effect (improved material choice). This can never happen if the policy objective is targeted directly.

64 As noted, such targeted recycling policies are indirect means of reducing damages by supporting an apparent means of mitigation.

rates by changing the relative attractiveness of different waste management options. In particular, by increasing the cost of other waste management options (such as landfill or incineration) they will increase incentives for the recycling of recyclable materials. However, the extent to which they do so will depend upon the precise nature of the measure introduced, and the broader policy context in which it is placed. Table 5.2 provides some examples of waste-related environmental policies which are likely to affect the level of recycling significantly, even if this is not their primary objective. (For empirical evidence and literature reviews see Johnstone and Labonne 2004, Kinnaman and Fullerton 1997, and Beede and Bloom 1995.)

Table 5. 2. The Effects of Selected Solid Waste Policies on Recycling

Instrument	Incentive	Comments
Unit-Based Waste Fee	By increasing the cost of disposal, encourages recovery of recyclables.	Empirical studies indicate that the effects are likely to be much greater if such measures are accompanied by collection schemes.
Landfill Tax	By increasing the cost of landfill disposal, may encourage the recovery of recyclables.	Environmental effectiveness depends upon the extent to which the cost of tax is ultimately passed through to waste handlers and generators in marginal terms.
Landfill Standards (Performance or Technological)	By increasing cost of landfilling waste, may encourage the recovery of recyclables.	Environmental effectiveness depends upon the extent to which the cost of the standard is ultimately passed through to waste handlers and generators in marginal terms.
Incinerator Standards (Performance or Technological)	By increasing the cost of incinerating waste, may encourage the recovery of recyclables.	Environmental effectiveness depends upon the extent to which the cost of the standard is ultimately passed through to waste handlers and generators in marginal terms. Effects on recycling are different for wastes which can be recovered for material or energy.
Landfill or Incinerator Bans	By preventing the landfilling or incineration of recyclables, will increase supply.	This can increase recycling rates for those recyclables which have been banned for disposal due to concerns about environmental impacts (i.e. used motor oil, lead-acid batteries, etc….)[65] However, it may divert waste into illegal disposal.

Indirectly, almost all solid waste policies will affect rates of recycling. In some cases these effects may be even more important than directed recycling policies. For instance, the combined application of strict technological and performance-based standards for landfill and incineration, with unit-based waste fees which are set at a level which recovers all costs (financial and environmental) will certainly lead to increased rates of recycling. Indeed, some empirical studies have found that the price elasticity of recycling rates in the face of unit-based waste fees can be as high as 0.4 (see Kinnaman and Fullerton 1999).

However, unit-based pricing of solid waste is not applied systematically across all OECD countries. Moreover, such measures will not result in optimal levels of recycling by material type, since waste fees and taxes can not be set at reasonable administrative cost at a level which reflects management costs for individual constituent parts of the waste stream. Even when used in conjunction with technology-based or

65 Note that some municipalities have introduced bans on recyclables which do not have particularly acute environmental impacts but which can be recycled – i.e. composting of organic waste. Such measures fit under the heading of targeted recycling policies discussed above.

performance-based standards they are crude means of targeting externalities. It is for this reason that they need to be accompanied by the types of policies outlined in Section 2 above.

4. Removing Policy Failures Which Discourage Recycling

Another public policy area which directly impacts upon recycling rates concerns the existence of policy failures which affect the degree of substitution between primary and recyclable materials.[66] There are two areas which seem to be particularly important:

- The provision of subsidies for substitute primary resource extraction and exploitation; and,
- The use of material standards which discourage optimal material choices.

It has often been argued that various types of subsidies for the extraction and processing of primary materials have inhibited the development of markets for some substitute recyclable materials. This can include various forms of direct financial subsidies (support for exploration and development), as well as tax preferences (percentage depreciation allowances) and in-kind public support (road construction in isolated regions), amongst others. Non-internalisation of associated public environmental goods (i.e. forest habitat, natural landscape, etc….) are also significant types of subsidies associated with primary resource exploitation. Such subsidies may result in significant negative environmental impacts generally (see, for example, Porter 2002 and OECD 2003 and OECD 2004b), and their impacts on recycling are just one manifestation of this.

Such programmes have the potential effect of reducing the relative price of a number of primary materials (minerals, pulpwood, and oil) which are in direct competition with recycled inputs (e.g. ferrous and non-ferrous metal scrap, used newsprint, plastic bottles, etc). This can result in reduced reuse and recycling of such materials, and thus increased solid waste generation rates. The extent to which such subsidies have implications on recycling and waste generation rates will depend upon the degree of substitutability between primary and recyclable materials.[67] This differs not only by material, but also by end use for individual materials (see Enviros Services/RIS 1999).

Another area of policy failure appears to be the use of product standards which restrict the use of recyclable materials as inputs in the manufactures of different types of goods. For instance, in many cases restrictions are placed on the use of recovered plastic in packaging or structural materials in construction waste. In some cases these restrictions are appropriate - i.e. if there are legitimate and verifiable concerns about hygiene effects associated with the use of recovered plastics in food packaging. However, as was discussed in the case study on plastics, in some instances the balance of costs and benefits between different public policy objectives (public health and environmental protection) may not be reflected appropriately in existing policy frameworks (see Ingham 2004).

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66 There is a third type of policy failure which will not be discussed – ‘soft budget constraints’ amongst municipalities and local authorities responsible for both waste management and recycling schemes. Unless such authorities have financial incentives to optimise their waste management strategies (in strict financial terms), they may not discourage landfilling and incineration sufficiently. However, this may also work in the opposite direction, with excessive support (in financial terms) for recycling schemes (see below).

67. It should also be noted that subsidies for recycling can also be significant. As noted above, publicly-supported programmes have been introduced in a number of OECD member countries in order to reduce the cost of both collection and processing of recycled materials. The relative importance of such subsidies in comparison with upstream subsidies for primary materials has not been examined.

In such circumstances the appropriate policy intervention is the use of standards which are targeted at the performance of the product and not its material composition (ECOTEC 2000). In this way, unfair discrimination against the use of recyclable materials will be removed. If products made with recyclable materials are functionally identical to those made with primary materials they will meet the performance standards set. Even if they are not functionally identical, the use of performance standards will ensure that any product differentiation which does arise will not be unfairly discriminatory, but will instead reflect these differences.[68]

Irrespective of the degree of substitutability between primary and recyclable materials, unless policy failures such as primary resource subsidies or discriminatory product standards are removed, levels of recycling will be sub-optimal. This will be the case even if environmental policies are in place to internalise waste-related externalities and market failures associated with recyclable material markets have been addressed through complementary policies. It is to these latter set of issues – the primary focus of the project – to which we now turn.

5. Addressing Market Inefficiencies in Recyclable Material Markets

If markets operate in conditions of maximum efficiency and regulatory standards and other policy measures are introduced for different waste management options such as to reflect their external environmental effects, recycling will be at its optimal level. However, in addition to the potential for non-internalisation of environmental externalities, other types of market failure may constitute an impediment to the realisation of commercial opportunities in recycling markets. This project has sought to identify the relative importance of such conditions, with a particular focus on used lubricating oil, rubber tires, and plastics.

Market failure is present when there are unexploited gains from trade. In effect, profitable opportunities are not being exploited. The fundamental causes of market failure are related to imperfect information, market power, technological and consumption externalities, and other factors. As the case studies on used lubricating oil, plastics and rubber tires (Fitzsimons 2004, Ingham 2004, and van Beukering and Hess 2004) have documented, many of these factors are prevalent in markets for at least some recyclable materials. The prevalence of barriers and failures in recyclable material markets can be attributed to their very specific nature of supply and demand. Based upon the case study reports and the overview report, Table 5.3 provides an overview of some of the market failures and barriers which affect particular markets.

68 Although the introductory chapter gives examples in the area of plastic timber and plastic bags where even performance standards may discriminate against the use of recyclable materials.

Table 5.3. Types of Market Failure and Potentially Affected Waste Streams

Class of Market Imperfection	Potentially Affected Waste Streams	Cause of Market Imperfection
Market power	Construction & demolition waste	Localised monopsonistic markets which may constrain market and reduce prices
	Used glass	Localised monopsonistic markets which may constrain market and reduce prices
	Metal scraps	Vertical integration and entry barriers which may restrict entry of recyclables
	Rubber tires	Market segmentation and price discrimination between retreaded and new tires
Information failure	Used lubricating oil	Presence of water or contaminants which reduces value of oil
	Construction & demolition waste	Uncertain quality of materials which restricts potential uses
	Used plastics	Presence of contaminants which increase costs of recovery
	Scrap metal	Presence of contaminants or materials which can result in high unanticipated disposal costs or damage reprocessing equipment
	Textiles	Presence of large (and easy to conceal) amounts of low-quality textiles in batches.
	Used paper and board	Mixed paper streams which result in high sorting costs, or lower value uses
Technological externalities[69]	Plastic packaging	Use of composite plastics which increase reprocessing costs and restrict potential uses
	Used lubricating oil	Presence of additives and contaminants which undermine potential for re-refining
	Scrap metal alloys	Use of composites which restrict or prevent potential for recovery
	Electrical appliances	Difficulty of sorting/separation due to mixed nature of waste, although technologies are fast-improving.
	Rubber tires	Newer tire types often less suitable for retreading
Search & transaction costs[70]	Construction & demolition waste	Heterogeneous nature of material and diffuse generation of materials
	Used lubricating oil	Heterogeneous nature of material and diffuse generation of materials
Consumption externalities[71]	Rubber tires	Interdependence of preferences between consumers and excess risk aversion
	Used lubricating oil	Interdependence of preferences between consumers and excess risk aversion
	Used newsprint	Interdependence of preferences between consumers

69. As noted in the introductory chapter, all of these characteristics provide benefits (or would not be introduced), however they also increase the costs of recycling and since there may be no means by which these costs are transmitted back to the product design stage – depending upon policy frameworks – it is possible that the benefits will be less than the costs for certain product designs.

70. While transaction and search costs are better characterised as a market barrier than a failure, they can be significant sources of unrealised market opportunities and have therefore been included in the discussion.

71. ‘Excess’ indicates aversion which is greater than that which would be implied by standard ‘certainty equivalence’ theory – i.e. reflecting the actual probability and cost of quality inferiority.

The existence of such failures and inefficiencies can result in missed opportunities of mutual benefit to potential buyers and sellers. For instance, if there is some uncertainty about the quality of re-refined lubricating oils or used paper they may be sold for lower-value uses than is necessary. Similarly, if transaction or search costs are high for construction and demolition wastes, trades of mutual benefit may not be consummated. In addition, if technological externalities are important, benefits from product characteristics such as multiple plastic layers may be less than the net costs of potential recovery opportunities which are lost as a consequence.

In many cases the market itself will find alternative ways to deal with many of these apparent inefficiencies in the market – i.e. standards organisations which provide 'grades' for scrap (transaction costs), vertical integration between material recovery and manufacturing stages (technological externalities), waste brokers which serve as market intermediaries (search costs), waste certifiers who identify waste quality (information failures), etc. However, this will not always be the case, and in such cases public policy interventions will be required.

However, the nature of the intervention may be different than that which is usually found in the environmental policy-maker's 'toolbox'. In the presence of such market imperfections, "environmental" policies (i.e. policies which have the explicit and sole objective of internalising environmental externalities) will not suffice to result in the optimal level of recycling. Thus, in addition to introducing policies which internalise environmental externalities, a strong case can be made for the use of market policies which improve efficiency within the market itself. Examples which relate to each of the broad categories of market inefficiency are discussed in turn. Responses to a questionnaire to Member countries of the OECD revealed that many governments are introducing policies in these areas.

5.1. Market Power

Market power can restrict the market for recyclable materials. In most cases problems of market power should be dealt with through standard competition and anti-trust law. For instance, if it is found that downstream manufacturers are exercising market power in a manner which discriminates in favour of the use of primary materials (and thus prejudices the penetration of recyclable materials in the market), this should be addressed through standard competition law. However, it appears that clear examples of such discrimination are the exception and not the rule. The rich literature spawned by the aluminium example mentioned in the introductory chapter reached ambiguous conclusions.

However, the review of the literature and the case studies did identify some areas in which discrimination *may* exist, but it frequently takes on very subtle forms, associated with strategies such as market segmentation and price discrimination. The cases of re-refined lubricating oil and retreaded tires are, perhaps, examples in which a degree of market segmentation has arisen. However, these are very difficult to regulate against, and the case for doing so is much less clear.

There are, perhaps, some examples which are more easily addressed. For low-value, bulky waste (such as paper, glass, and C&DW), there may be clear examples of monopsonistic behaviour by reprocessing facilities. This can result in insufficient returns for collectors, resulting in inefficiently low collection rates. Indeed, in some cases local authorities may be playing a direct role in encouraging monopsonistic or monopolistic behaviour in such markets. Since collection and/or reprocessing are often 'natural' monopolies, concession contracts or service contracts are issued. If the procedures by which these contracts are awarded - and more importantly, the means by which they are regulated – are not appropriate there is significant potential for the creation of monopoly/monopsony rents.

5.2. Information Failures

Information failures do appear to be a widespread problem in a number of markets for recyclable materials. In so far as they relate to the use of material inputs in the production of goods, there do appear to be markets in which the presence of contaminants can not be easily detected by potential buyers (plastics, construction and demolition waste). In other cases there may be undetected presence of materials which can affect downstream reprocessing equipment. In this case there is an asymmetry of information, and the market will be undermined with adverse economic and environmental implications.

In such cases the public authorities should seek to provide incentives for market participants to reveal the true quality of the wastes being placed on the market. This increases transparency in the market and reduces the potential for adverse selection (low quality 'lemons' crowding out high quality waste) and moral hazard (encouraging efforts to sort and 'clean' waste prior to sale). Appropriate incentives should encourage 'collectors' and others to undertake efforts to provide waste of high quality will help to increase efficiency in the market.

Thus, research and development on 'testing' equipment may prove beneficial in some cases, reducing the potential for concealing the presence of 'bads' by sellers and thus making material quality more easily detectable. The growing use of infra-red technologies in plastics sorting technologies may help. In other cases, support for dispute resolution mechanisms and development of appropriate liability standards may be useful, giving buyers recourse to compensation in cases of misrepresentation by sellers. However, in all cases it is important that the reputation effects of providing low-quality waste are transmitted back to the seller or else the 'reputation' effects of providing low-quality (or even hazardous) waste are limited.

5.3. Consumption Externalities

The degree of 'risk aversion' associated with goods manufactured from some recyclable materials (retreaded tires, used lubricating oil, etc…) appears to be out of proportion to the actual inferiority of the products relative to substitute goods manufactured from primary materials. In cases in which consumers' preferences are affected significantly by the preferences of other consumers (i.e. consumption externalities), this can undermine the market for recyclables significantly. The cases of retreaded tires and re-refined oils pointed this out clearly, but it also appears to affect other markets such as recycled newsprint, plastic lumber, etc… Since initial buyers of new products are not rewarded in the market for the information about product quality that they provide to other market participants, there can be barriers to the development of such markets.

There are three other useful roles which public authorities can play in such cases:

- Serve as a 'trusted' source of demand (i.e. in public procurement), demonstrating to other consumers that the two products are functionally identical or at least close substitutes. New Zealand, Austria, the United Kingdom, the Netherlands, France, Turkey, and Korea report having introduced such measures in a variety of waste areas;

- Ensuring that product standards are based upon performance criteria, and not material (recyclable vs. virgin) content. Efforts to ensure that such biases have been removed are reported for Austria, Turkey, United Kingdom, the Netherlands, and Korea; and,

- Provide information on the quality of products manufactured from secondary materials. For instance, both France (www.marque-nf.com) and the Slovak Republic (www.druhasanca.sk) have active programmes in this area. In some cases certificates of quality for goods produced from recyclable materials are provided. The United Kingdom's *Quality Protocol* in the area of aggregates is one such example.

In many cases government policies may be unintentionally having the opposite effect, undermining confidence in the market.[72] In addition, public authorities should be aware of the potential impact of consumption externalities and risk aversion on the market, and avoid using eco-labels which specify recycled material content for affected goods. Even if consumers have 'preferences' for the use of recycled material for environmental reasons, in some markets this may be partially undercut by their risk aversion for perceived quality reasons. Without support information programmes, in such cases the positive consequences of the 'eco-label' on demand may be undone by the negative consequences.

5.4. Technological Externalities

Due to specific product and material characteristics, there are many wastes in which technological externalities are widespread (eg. composite and multi-layer plastics, metal composites, post-consumer electronic appliances, etc….). All of these product characteristics provide positive benefits, or they would not be undertaken. And these benefits are reflected in the value of the goods in the market place. However, they also generate costs (i.e. increased sorting or reprocessing costs), and if there is no means whereby the 'costs' are transmitted back up the product chain to product designers and manufacturers, these costs will not be reflected in the marketplace.

Box 5.1 Technological Externalities

A technological externality exists when the production function of one agent enters another agent's production or utility function, without the latter being compensated (Kolstad 2000). Environmental externalities (when they affect productive processes such as polluted irrigation water) are, of course, specific examples of the more general case of technological externalities. However, they are by no means the only type of technological externality. In this section, the focus of the discussion is on "non-environmental" technological externalities which tend to reduce recycling rates. In the area of waste, technological externalities would arise when one firm manufactures a product in such a way which increases the cost of recycling for the downstream processor, but for institutional reasons there is no means by which the potential waste recovery facility can provide the manufacturer with the incentives to change their product design (Porter 2002 and Calcott and Walls 2000).

As noted in the synthesis report, markets often address such problems.[73] But, what can public authorities do to contribute to their resolution? Possible avenues would include:

- In cases wherein product standards are promulgated to ensure some public objective (i.e. hygiene standards for plastic packaging), ensure that environmental public objectives are also taken into account if there are missing markets associated with such impacts. This is particularly important if there are trade-offs in product design between the two objectives.

72. The Austrian government undertook a study in 1994 which examined whether product standards have discouraged the use of recyclable materials. .

73. Similar rules apply in the Austrian *Ordinance on Take Back of Batteries and their Limitation of Hazardous Substances*.

- Introduce policies which 'bracket' the missing market -- for instance, a deposit-refund or subsidy-tax scheme which is levied according to the degree of recyclability of the product, would encourage 'design for recycling'. In theory a take-back scheme would have the same effect, but may allow for less differentiation according to product characteristics and has potentially high administrative costs.

- Introduce measures which reduce the costs of such externalities – i.e. by providing information to demanders of recyclable waste on means of recovering reusable materials.[74] For instance, in Austria, the *Ordinance of End-of-Life Vehicles* facilitates material and component recovery through the application of material and coding standards and dismantling information.

In addition, many countries (Japan, France, UK, Korea, Austria) report having provided support for research and development into product design which facilitates recycling. It may also be possible to provide support research and development for technologies used in sorting and reprocessing facilities. However, while this may increase the degree of recyclability of affected goods by overcoming the technological externality, it must be recognised that this does not remove incentives for product design which results in technological externalities in the first place, and may even provide incentives for their exacerbation.

5.5. Search and Transaction Costs

Search and transaction costs are common in all markets, but may be particularly problematic for markets for recyclable materials due the particular nature of their generation and their heterogeneous nature. This can result in significant costs to identify market counterparts, to agree upon a price, and to conclude a transaction. While such costs may reduce as markets mature, if they are important enough at the outset, they may prove to be an insuperable barrier and the market itself never becomes viable.

Public authorities can play a role in all of these stages. For instance, it can serve as a 'midwife' to the market, helping buyers identify sellers, and vice versa. A number of public authorities have taken on this role. For instance, in the Netherlands, public authorities have sought to reduce friction in the market for recoverable residues of energy production. In Korea, a monthly "Market Survey on Recyclable Materials" provides information to 1,000 subscribers on prices and trends in markets for waste paper, synthetic resins, glass bottles, metal cans, and used tires.

Many governments have sought to play this role through support for the use of on-line web-sites.[75] However, it is not clear that public authorities have a comparative advantage in fulfilling such a role, and many sites designed to fulfil this role have closed. Perhaps more fruitfully, in cases in which such intermediary services do not arise spontaneously in the market efforts should be undertaken to determine why this is the case.

More evidently, public authorities can play a role in encouraging or promulgating the development of grading schemes for scrap and other wastes. For instance, Austria's *Compost Ordinance* specifies three grades, designed to facilitate the identification of potential uses. These have the effect of reducing the 'space' for negotiation, and thus potentially reducing transaction costs. In addition, if dispute resolution

74. An interesting private sector initiative in this case is the "Internnational Dismantling Information System" (www.idis2.com) which is a consortium of 24 automotive manufacturers, providing information of value to vehicle dismantlers.

75. Examples include exchanges for construction and demolition waste in Austria (http://www.recycling.or.at), and aggregates in the UK (http://www.AggRegain.org.uk).

mechanisms are put in place, it may also reduce transaction costs by reducing the need to identify potential sources of disagreement at the stage of contract preparation. Many trade associations have recognised this, sometimes with the support of public authorities. Similarly, the dissemination of 'standardised' contracts may also help to reduce transaction costs. The Netherlands has recognised the value of this with respect to secondary construction materials, as has the United Kingdom in the area of wastepaper.

5.6. Summary

Table 5.4 summarises some of the possible public policies which can be applied to address particular market failures and inefficiencies. The list is by no means exhaustive, but it does indicate the potential scope for public intervention. Whether or not such interventions increase efficiency within the particular market will very much depend upon local conditions.

Table 5.4. Types of Market Failure and Associated Public Policies

Market Failure	Relevant Public Policies to Address the Failure
Search costs	Information dissemination to potential market participants (supply and demand), web exchanges to reduce costs of identification of market counterparts
Transaction costs	Development of standardised contracts, waste quality grading schemes which 'commodify' heterogeneous materials, establishment of dispute resolution mechanisms
Information Failure	Certification schemes, support for testing equipment, public procurement programmes, liability for product misrepresentation, establishment of dispute resolution mechanisms
Consumption externalities	Demonstration projects, public procurement programmes, information dissemination concerning product characteristics
Technological externalities	Extended producer responsibility, research and development on 'design-for-recycling', product standards which incorporate impacts upon recyclability
Market power	General competition and anti-monopoly policy, market regulation of collection and processing which ensures competitive demand

However, the key point is that the policy needs to address the relevant market failure. If not the policy may be at best irrelevant. Introducing waste quality grading schemes will serve little benefit if the market is mature and buyers and sellers do not face uncertainty with respect to waste quality. Web-based waste exchanges will not be of value if search costs are low. Worse still, the policy may have perverse impacts. For instance, an eco-label which trumpets the recycled content of a particular product will have negative consequences (even if environmental preferences are significant) if consumers are risk averse to potential quality discrepancies. Similarly, public support for the development of sorting technologies may reduce incentives for 'design-for-recycling' by reducing the costs associated with less environmentally-benign product design.

However, if well-directed at the failure or barrier which exists in the given market, then such policies can be an important complement to more traditional 'environmental' types of policy measure. More generally, in the absence of such complementary 'market' policies, 'environmental' policies which directly seek to increase recycling rates are likely to be much less efficient and effective. In effect, such policies will be "running uphill", fighting against imperfections in the market which discourage recycling. It is likely to be more cost-effective to address the failure directly than seek to overcome it by introducing increasingly stringent environmental policies.

6. Conclusions

Many OECD governments have introduced targeted policies to encourage recycling. However, increased rates of recycling is not a legitimate public policy objective for its own sake. If there is full internalisation of waste-related environmental externalities, and there are no market failures in recyclable material markets nor policy failures which discourage the substitution of primary materials for recyclable materials, no case can be made for the introduction of targeted public policies to encourage recycling.

In practice such conditions rarely hold. Recyclable material markets can be inefficient, support is sometimes granted to substitute primary resource exploitation,[76] and environmental externalities are not fully internalised in solid waste policies. This leads to sub-optimal rates of recycling. The best solution is, of course, to address such policy and market failures at source, and let the markets (primary and secondary) take their course. This should lead to more nearly optimal levels of recycling.

However, this is not always possible. It may be impossible to target waste-related environmental externalities directly at reasonable administrative cost, there may be political obstacles to the removal of public support for primary resource exploitation, and market failures may not be subject to effective policy intervention. In such cases targeted recycling policies can and should be introduced, although they need to be designed with care, taking into account not only the precise characteristics of the market, both in terms of the waste stream and local conditions.

For instance, if products made from recyclable materials are identical with those made from primary materials, but there is a negative perception of the latter, introducing eco-labels to increase demand will have a perverse effect. Similarly, if buyers of recyclable materials enjoy monopsony power in local markets, policies to encourage increased collection rates are less likely to be effective since market outlets may not be sufficient. Failures, objectives and policies need to be closely matched.

As is so often the case, the need for the use of policy mixes is emphasised. However, it must be noted that this mix relates not only to environmental policy, but also to market and industrial policy more generally. A thorough understanding of the markets and the means by which different policies interact with each and impact upon the market is key to the development of the right mix. This is particularly so if one recognises that a number of 'environmental' policies (i.e. extended producer responsibility') can impact significantly upon markets in which 'industrial' policies (i.e. market regulation) are also operational.[77]

76 However, as noted above, financial support is also often provided to recycling schemes. The net effect depends very much upon location-specific and material-specific conditions.

77 See Adant and Godard (2004).

REFERENCES

Adant, Ignace and Olivier Godard (2004) ‘Economie de recyclage: Les conditions d’une régulation efficace sous incertitude et opacité’ Rapport d’activité, ADEME – Service Economie, Paris.

Akerlof, George, A (1970) “The market for “lemons”; quality uncertainty and the market mechanism”, *Quarterly Journal of Economics*, 89(3).

Beede, David N. and David E. Bloom (1995) “Economics of the Generation and Management of Municipal Solid Waste” (Cambridge: NBER Working Paper No. 5116).

Berndt, Ernst, R., Robert S. Pindyck, Pierre Azoulay (2000)“Consumption externalities and diffusion in pharmaceutical markets; antiulcer drugs”, National Bureau of Economic Research, NBER working paper 7772, Cambridge, Mass.

Beukering, Pieter van (2001) "Empirical Evidence on Recycling and Trade of Paper and Lead in Developed and Developing Countries", *World Development* 29(10), pp. 1717-1737.

Calcott, Paul and Margaret Walls (2000) "Policies to Encourage Recycling and "Design for Environment": What to Do When Market are Missing" (Washington, D.C.: RFF Discussion Paper 00-30).

Curlee, Randall, T. (1986) *The Economic Feasibility of Recycling: A Case Study of Plastic Wastes* (New York: Praeger).

Ecotec (2000) “Policy instruments to correct market failure in the demand for recyclable materials”, Report prepared by Ecotec Research and Consulting Ltd., Birmingham.

Eichner, Thomas and Rüdiger Pethig (2001) "Product Design and Efficient Management of Recycling and Waste Treatment" in *Journal of Environmental Economics and Management*, Vol 41, pp. 109-134.

Environmental Protection Agency (1993) “Developing markets for recyclable materials; policy and program options”, Report prepared by Mt. Auburn Associates, Inc. and Nothreast-Midwest Institute, for U.S. Environmental Protection Agency, Washington D.C.

Enviros Services/RIS (1999) "Developing Markets for Recyclable Materials" (Cambridge, UK: Enviros).

Grassroots Recycling Network (1999) “Welfare for waste”, Grassroots Recycling Network, Athens, Georgia.

Johnstone, Nick and Julien Labonne (2004) “Generation of Household Solid Waste in OECD Countries: An Empirical Analysis Using Macroeconomic Data” in *Land Economics*, Vol. 80, No. 4.

Kinnaman, Thomas C. and Don Fullerton (1999) "The Economics of Residential Solid Waste Management" (Cambridge: NBER Working Paper 7326).

Kinnaman, Thomas C. and Don Fullerton (1997) ‘Garbage and Recyling in Communities with Cubside Recyclling and Unit-Based Pricing’ (Cambridge: NBER Working Paper 6021).

O’Neill, William O. (1983) “Direct empirical estimation of efficiency in recyclable materials markets; the case of steel scrap”, *Journal of Environmental Economics and Management*, Vol. 10, pp. 270-281

OECD (2000) *Improving the Environment through Reducing Subsidies* (Paris: OECD).

OECD (2001) "Improving Recycling Markets" (ENV/EPOC/WGWPR(2001)1/REV1).

OECD (2003) *Environmentally Harmful Subsidies: Policy Issues and Challenges* (Paris: OECD)

OECD (2004) *Addressing the Economics of Waste* (Paris: OECD).

OECD (2004b) “Draft Synthesis Report on Environmentally Harmful Subsidies” (SG/SD(2004)3/Rev1)

Porter, Richard (2002) *The Economics of Waste* (Washington D.C.: Resources for the Future).

Rasmussen, Clemen et al. (2005) “Rethinking the Waste Hierarchy” (Copenhagen: Environmental Assessment Institute)

Sigman, Hilary (1998) "Midnight Dumping: Public Policies and Illegal Disposal of Used Oil" in *RAND Journal of Economics*, Vol. 29, No. 1, pp. 157-178.

Sound Resource Management Group (2000) "The Chicago Board of Trade Recyclables Exchange: Evaluation of Trading Activity and Impacts on the Recycling Marketplace" (Seattle: SRMG)

US EPA (1993). “Developing Markets for Recyclable Materials. Policy and Programs Options”. Prepared by Mt. Auburn Associates, Inc. and Northeast-Midwest Institute for US Environmental Protection Agency.